Coral Reefs
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RESEARCH

Coral bleaching and mass mortality at Lizard Island revealed by drone imagery

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Abstract Quantitatively assessing mortality post coral bleaching at scale is inherently difficult, yet can be achieved with georeferenced imagery from aerial drones. Here, we assess the coral bleaching mortality rate from the 2024 global bleaching event at the iconic Lizard Island, Australia. Using drone-derived orthomosaics of the northern and southern sides of the island collected during and after the bleaching event, we measured the area of bleached coral and the area of live coral remaining after bleaching. Across twenty 10×10 m quadrats, mean bleaching mortality was $92.2 \pm 6.8\%$, with bleaching affecting $96.92 \pm 2.03\%$ SD of living coral cover (mean, SD) of quadrat areas. This is one of the highest rates of bleaching mortality ever recorded, despite corals at Lizard Island being exposed to lower levels of cumulative heat stress than others in many parts of the Great Barrier Reef during this bleaching event.

Keywords Coral bleaching · Remote sensing · Climate change · Mortality · UAV · Drone

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Introduction

The frequency and severity of mass coral bleaching is increasing due to marine heat waves instigated by anthropogenic climate change (Hughes et al. 2018). However, direct connections between bleaching intensity, coral mortality, and resulting habitat loss are less clear (McClanahan 2004). High bleaching levels (> 90%) may result in less than 50% coral mortality (Riegl et al. 2012; Sakai et al. 2019), and some coral species may be more susceptible to mortality following bleaching events than others (Matsuda et al. 2020). These variations and potential adaptations are not well understood or quantified at scale.

While adaptation to marine heat waves through natural selection may allow some corals to survive (Coles and Brown 2003; Császár et al. 2010), it is uncertain whether adaptation of thermal tolerance can keep pace with the rising frequency of bleaching events under climate change (Hughes et al. 2018; Lachs et al. 2023; Brown et al. 2023b). Tradeoffs to high thermal tolerance have been noted in corals from extreme contrasting environments of reefs versus mangroves (Scucchia et al. 2023), but are not necessarily ubiquitous for outer reef corals (Lachs et al. 2023). Moreover, bleaching events severe enough to affect most corals and result in high (> 90%) mortality can result in substantial loss of coral reef habitat (Hoegh-Guldberg 2014). Such losses of habitat can greatly reduce the biodiversity and function of associated fishes and other reef-associated organisms (Pratchett et al. 2018). Therefore, having a clear understanding of the relationship between bleaching incidence and coral mortality over broad areas is crucial for determining whether coral reefs will persist into the future.

Post-bleaching coral mortality assessments are uncommon due to timing challenges with short-term bleaching events, and the difficulty of quantitatively assessing coral

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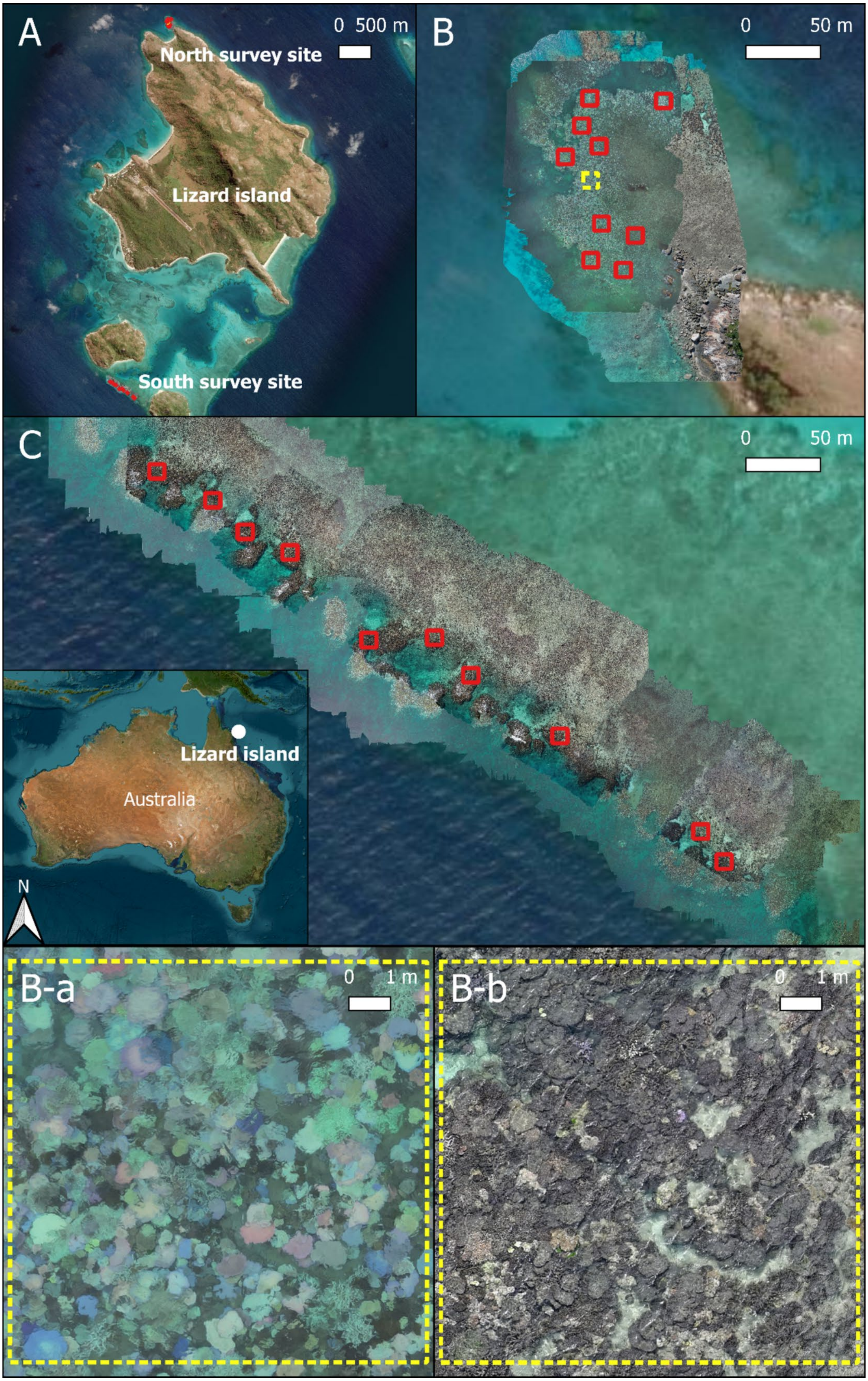
A
0 500 m
North survey site
Lizard island
South survey site
B
0 50 m
C
0 50 m
Lizard island
Australia
N
B-a
0 1 m
B-b
0 1 m

◂**Fig. 1** Images of orthomosaics of the two surveyed reefs (**B**, **C**) at Lizard Island (**A**), Queensland, Australia, produced from drone imagery overlaid onto satellite imagery. Locations of the 10 × 10 m digital quadrats used for bleaching mortality assessments overlaid with red (solid outline) squares. Example during and after bleaching imagery from one quadrat in dashed yellow (**B-a**, **B-b**). Note the shift from bleached and fluorescing corals to predominantly dead corals overgrown with dark algae

mortality. Most surveys investigate impacts via permanent transects and monitoring mortality of individual coral colonies (Depczynski et al. 2013; Banha et al. 2020). The underwater work required generally means such studies survey a relatively small area compared to the reef extent (e.g. 360 m^2 in Depczynski et al. 2013 and 451 m^2 in Tebbett et al. 2022), or a small number of colonies relative to the whole reef population (e.g. 462 colonies in Byrne et al. 2025). Such assessments of individual coral colonies over small areas may not extrapolate well to represent the reef impacted by bleaching, making it difficult to perform comparative assessments of bleaching mortality at the reef level. Drone (unoccupied aerial vehicle, UAV) imagery has the advantage of producing verifiable data of bleaching and mortality over substantially larger areas and geographic regions than in-water surveys and can be used as baseline data for future events.

With the fourth global coral bleaching event officially announced for 2024 by the National Oceanic and Atmospheric Administration in April (NOAA 2024), the importance of accurately monitoring bleaching and mortality is paramount. Widespread bleaching in 2024 was confirmed on Australia's Great Barrier Reef (GBR) through this process; however, rates of bleaching mortality linked to this bleaching event are yet to be determined. To fill this gap, we used drones to assess two reefs at the iconic Lizard Island on the GBR that experienced bleaching in March 2024. We asked: (1) How extensive was the March bleaching event on these reefs; (2) What proportion of the coral survived to June 2024; and (3) Was bleaching mortality homogenous across the island's reefs?

Methods

Data collection

Drone surveys were conducted over two semi-exposed reefs at Lizard Island in March and June 2024: one in the north of the island (North Point Reef 145.45472°, – 14.6462°) and one to the south (Palfrey Island Reef 145.4473°, – 14.6990°). These surveys were conducted opportunistically as part of other research projects, and thus formal in-water validation was not possible within the available field time. However, haphazard snorkels were conducted in the same areas as part of other work, which validated high bleaching and subsequent high bleaching mortality over the area of interest (Figs. 1, 2).

Flight missions parameters were selected using best practice for mapping coral reef environments, targeting low tides in the mid-mornings or mid-afternoons that produce the best results regardless of wind speed, wind direction, and sun orientation (Joyce et al. 2018). For the March dataset, images were collected on 19 March 2024 using a DJI Mini 3 Pro with a 12 MP, 1/1.3 inch CMOS camera pointed at nadir (directly down). Flight missions in March were conducted at low tide (0.87 m) after 1300 local time. For the June dataset, images were collected on the 4th and 5th of June 2024 by an Autel Evo II drone with a 24 MP, 1 inch CMOS mechanical shutter camera and a DJI Phantom 4 Pro with a 20 MP, 1 inch CMOS mechanical shutter camera at nadir. Flight missions in June were done during peak low tides (0.18 m and 0.17 m low tides for the two days) in the afternoons between 1300 and 1600 local time. Due to variable weather conditions and desired resolution, the flight mission conducted by Autel Evo II at North Point Reef flew at 30 m altitude with an 85% overlap and 80% sidelap, whereas the southern flight mission at Palfrey Island flew at 20 m altitude with an 85% overlap and 75% sidelap.

Imagery was used to produce orthomosaics using the GeoNadir portal, which are publicly accessible (https://tinyurl.com/lizard-island-bleaching). Resulting orthomosaics overlapped across a ~ 18,000 m^2 area with a ground sampling distance (GSD, effectively image resolution) of 0.72 cm per pixel in North Point Reef, and an area of ~ 80,000 m^2 with a GSD of ~ 0.7 cm per pixel in Palfrey Island Reef (Fig. 1B, C). All subsequent geospatial analyses were also conducted using the GeoNadir portal.

Analysis

To ensure our approach was comparable to traditional habitat surveys while applying it to large area drone orthomosaics, we subsampled each reef using 10 × 10 m georeferenced, fixed digital quadrats placed haphazardly across the drone survey areas over time (Fig. 1). This approach aligns with other bleaching monitoring studies relying on fixed monitoring plots (Eakin et al. 2010; Depczynski et al. 2013; Sakai et al. 2019) albeit ours was at a much broader scale. The quadrats covered areas with consistent shallow depths (< 5 m) with high to low initial coral cover, representing the different reef crest and reef edge habitats present.

Using the polygon tool and freehand tracing in the GeoNadir web portal, areas of bleached coral within the March (bleaching) quadrats were outlined and calculated. Any coral that was fluorescing, white, or in the early stages of death was included within these areas, and these were visibly distinct from healthy corals in the imagery (Fig. 3). Dead corals

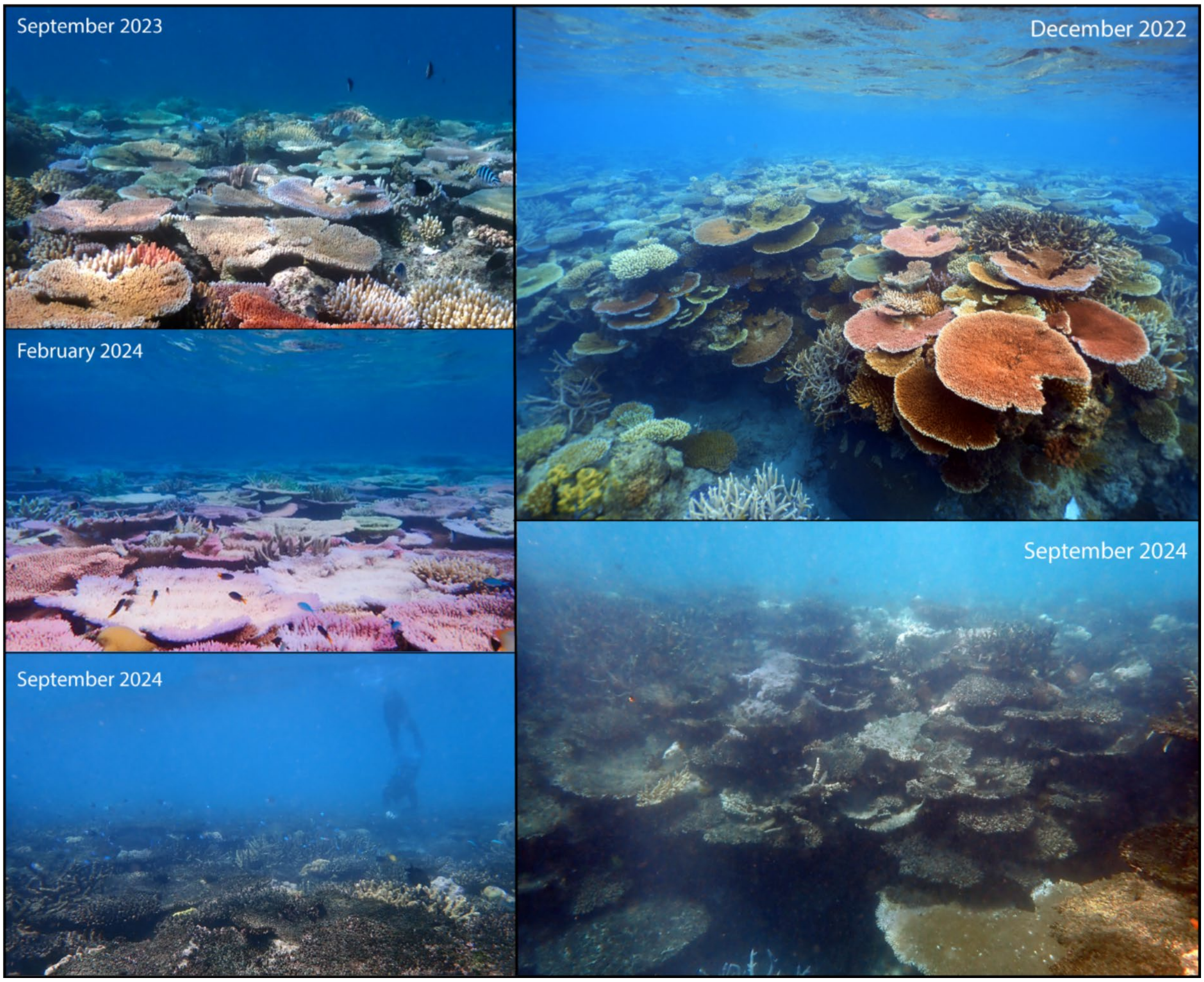

Fig. 2 Underwater images captured in the northern Lizard reef at various time points before, during and after bleaching. Note the very dark, almost black colouration of recently dead corals as a result of algae growth, which gave us confidence we could separate live from dead corals from drone imagery

in the June (post-bleaching) dataset were also distinct, with a dark grey shade. In the post-bleaching data, all living corals were outlined and a spatial polygon layer was created, and the living area after the bleaching event was calculated. Coral bleaching mortality was also visually qualitatively validated on snorkel during the June trip. Soft corals were not separated from hard corals, but the reefs in question are typically dominated by hard corals with interspersed algal tufts (Tebbett et al. 2022). If corals could not be identified as living or dead post-bleaching they were conservatively recorded as alive.

Snorkelling in the area (Fig. 2) gave us a high degree of confidence we could separate bleached from dead or live coral, especially given the much darker hue of corals covered in turfing algae (Fig. 3). However, as qualification of 'bleached' and 'living' corals from aerial imagery is subject to professional perspective and we were not able to include formal in-water validation during fieldwork, we first assessed inter-observer differences in area and bleaching mortality measurements with a subset of three quadrats (total area = 300 m^2) on North Point Reef that were measured independently by four observers. Resulting residual standard deviation measurements were used to assess precision of the approach. While not able to validate 'true' accuracy of the bleaching mortality assessments, this would provide confidence that observer bias was unlikely to impact results and be repeatable should other researchers attempt similar measurements with the dataset. This approach is commonly used in fisheries science, for example, to validate visual measurements of fish age (Campana 2001).

To comprehensively assess bleaching and mortality on the two reefs on opposing sides of the Island, an additional 17 quadrats were then assessed by one of the observers, resulting in a total of 11 quadrats at North Point Reef and 10 quadrats at Palfrey Island Reef (total quadrat area = 2000

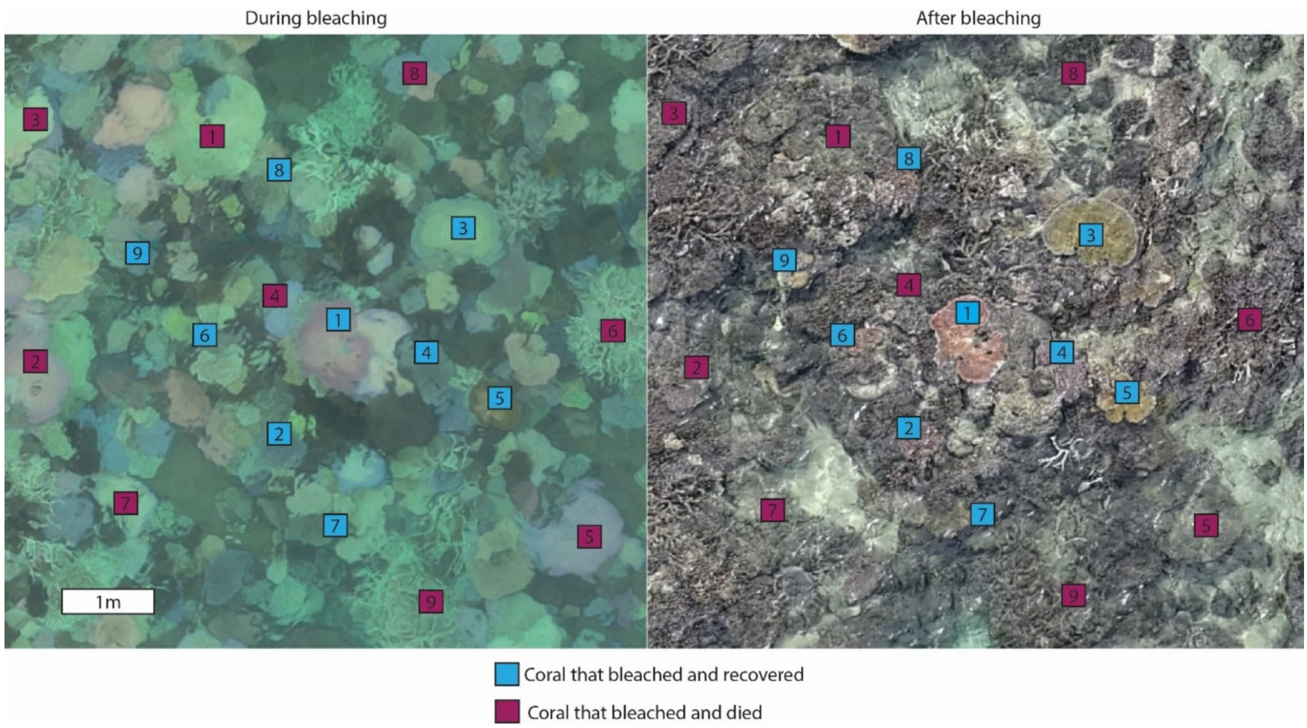

Fig. 3 Example fate of coral colonies during bleaching and after. Numbers correspond to the same coral colony across both instances. Note the typical black/brown appearance of dead corals covered in algae compared to green/pink hues of surviving corals that was readily separated in drone imagery. Also, note the already dead corals during the bleaching event in dark brown hues that are clearer in the post-bleaching image

m^2). All image analyses were conducted using tools available in the GeoNadir web portal.

To more accurately quantify the survival and mortality rate of bleached corals, the resulting spatial polygon layers of bleached corals and post-bleaching surviving corals were intersected. The objective of this was to exclude area of post-bleaching corals that were not in the initial bleached layer (and thus were living corals within the quadrat that had not bleached). We then quantified bleaching mortality using the initial 'bleached coral' layer and the new 'surviving coral minus corals that were not in the initial bleaching layer' (Fig. 4). This also allowed us to add the living but non-bleaching coral layer to the initial bleaching layer to calculate total initial coral cover, and subsequently the proportion of total coral cover that bleached. To assess whether the northern and southern reefs had similar bleaching mortality rates, a *t*-test was run with location as a factor and bleaching mortality rate as a response variable in R V. 4.2.2.

Using this method, individual colonies were not tracked, rather bleached polygon areas were summed relative to total quadrat area. This approach was preferable to individual tracking of colonies due to the many thousands of colonies that were captured within the 20 quadrats and the time it would have taken to process them in this fashion. In addition, preliminary tests found that in some quadrats, colony boundaries were difficult to define in both the bleached and post-bleaching datasets, which increased the likelihood of discrepancies occurring that would produce outputs that were less accurate than areal calculations. Machine learning segmentation approaches could address these issues and should be tested in future studies but were not within the scope of this project.

Timing of bleaching surveys is an important factor to correctly identify bleaching severity and mortality (Claar and Baum 2019). Publicly available data from the NOAA DHW portal suggested thermal stress at Lizard Island in 2024 was the third highest on record, with both 2016 and 2017 achieving higher DHW (Fig. 5). Sea surface temperature data and NOAA bleaching alert status suggests the initial drone surveys were conducted just after the peak of thermal stress, while the follow-up surveys were conducted approximately 2 months after thermal stress was lower than alert level 1. The reproducible code for extracting the data used for this figure is available here: https://github.com/marine-ecologist/nGBR-DHW.

Results

Measurement precision between observer measurements of bleaching mortality was very high across all three quadrats

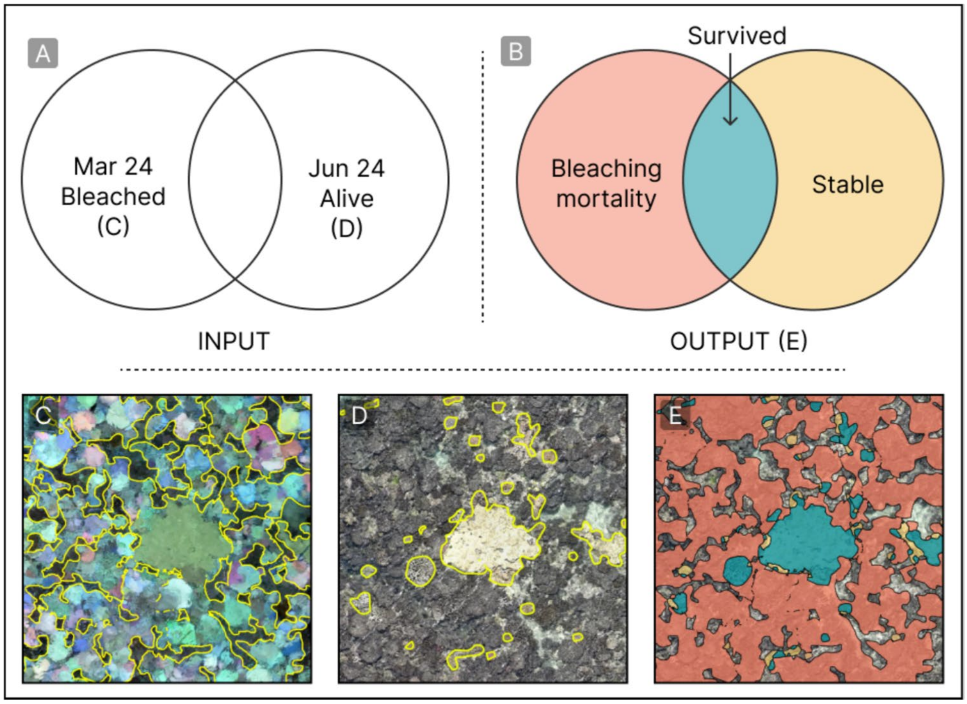

Fig. 4 Conceptual diagram of spatial layers created and used to calculate bleaching mortality. Bleaching layer and post-bleaching living coral layers were used (**A**). These two spatial layers were intersected to exclude living corals that were not initially bleached (**B**). Example layers of bleached (**C**), living post-bleaching (**D**), and bleaching mortality and survival layers (**E**) overlaid onto drone orthomosaics in one of the surveyed quadrats

examined by the four observers (1.4, 1.8, and 0.7% relative standard deviation), suggesting high repeatability of bleaching mortality assessments. The coefficient of variations between observers for bleaching areal measurements was slightly higher (9.3, 11.5, and 9.3% relative standard deviation) but not substantially different, suggesting alignment in assessment between observers and confidence in our approach.

Using the intersect of the layer of surviving corals that had not bleached in June, and the layer of bleached corals in March, we calculated that bleaching affected at least $96.92 \pm 2.03\%$ SD of living coral cover across the two reefs (Fig. 6B), and a mean $59.2 \pm 20.5\%$ SD of the area across all quadrats (Fig. 6A). In most cases, lower bleaching area reflected lower total available coral cover rather than a reduced incidence of bleaching, with bleaching often affecting all coral present in the quadrat. The largest area covered by living coral post-bleaching was 23.9% of the quadrat, with mean cover of surviving corals of $7.5 \pm 6.1\%$ SD of quadrat area.

No significant difference in bleaching mortality rates between the southern and northern reefs of Lizard Island were observed ($df = 1, 19$; $F = 1.82$, $p = 0.19$). Bleaching mortality rate, the proportion of initially bleached coral that subsequently died, was consistently high, ranging up to 99.9%. Mean bleaching mortality rate was $92.22 \pm 6.28\%$ SD across all quadrats (Fig. 6D). We note that some corals in the initial bleaching dataset had already died and these were not included as 'bleached' in analyses, and other corals were still bleaching in the post-bleaching dataset; thus, despite already very high values, total bleaching mortality during the 2024 event is likely to be higher than our results suggest.

Discussion

Coral bleaching mortality at the two reefs assessed at Lizard Island during the 2024 bleaching event is the highest ever documented at the site. To our knowledge, only one other study from the west coast of Australia measured similar overall bleaching mortality rates of approximately 90%, though this was recorded over a much smaller 360 m^2 area (Depczynski et al. 2013). Understanding the drivers of our alarming coral bleaching mortality rates will be important to better predict these events and their ecological impacts.

In recent years, Lizard Island has experienced multiple large-scale disturbances, including bleaching, cyclones, pollution, and crown-of-thorns outbreaks, whose effects have likely accumulated over time and diminished the corals' capacity to recover (Hughes et al. 2019). Work conducted across two decades from 1980s already suggested that disturbance events produced cyclical changes in coral communities and reductions in hard coral cover (Wakeford et al. 2008). Severe bleaching and cyclones occurred in both 2016 and 2017 at Lizard Island, which led to a coral cover decline from ~ 30 to < 10% over 2 years (Madin et al. 2018; Brown et al. 2023a) or an approximate mortality of 66%. Coral cover at Lizard Island at the start of 2024 was much higher than in early 2016, with some quadrats in this study exhibiting > 90% coral cover during the bleaching event.

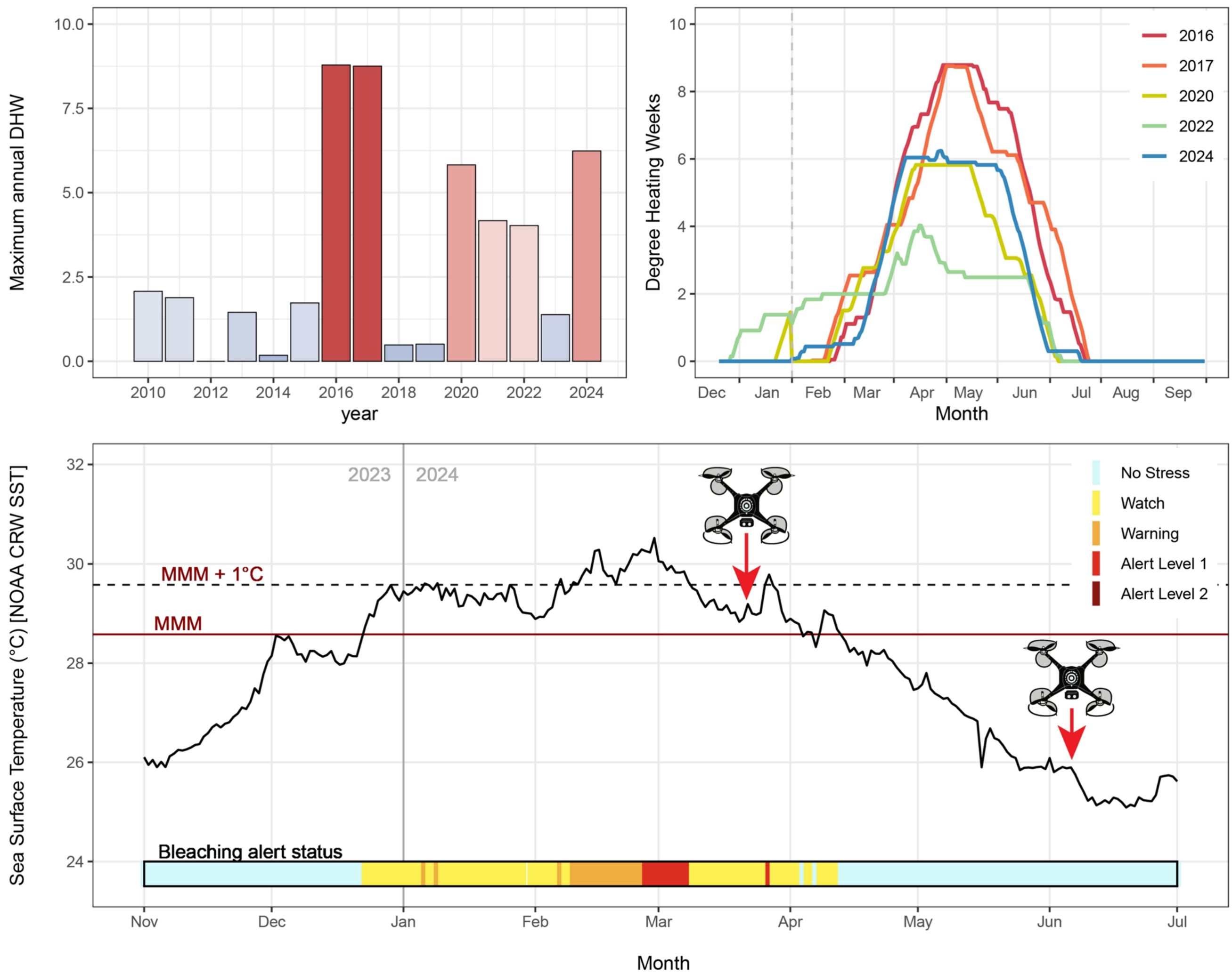

Fig. 5 Bleaching alert level inferred from sea surface temperature (SST), and degree-heating weeks (DHW) for Lizard Island, QLD, Australia in 2024 (bottom), and in previous years (top). Note the drones and arrows indicating when imagery was collected relative to sea surface temperatures. The maximum monthly mean (MMM) is typical warm season temperature of the baseline period used by NOAA (maximum of monthly means climatology), with MMM + 1 set as the level above which heat stress accumulates

We note that most corals we observed in the survey areas were tabular *Acropora* spp. that rapidly grew across the area after previous bleaching (Anderson 2025), and it is possible the greater coverage of this morphotype relative to previous years could explain higher mortality. Tabular *Acropora* spp. are less susceptible to bleaching than other acraporid morphotypes (e.g. digitate or arborescent, see (Hoogenboom et al. 2017)), but *Acrapora* spp. are more susceptible than other genera like *Montipora* spp. and *Porites* spp. (Pratchett et al. 2018). Thus, the dominant coral species in these sites could explain the high mortality.

In the broader context of the 2024 global bleaching event, the mortality rates we have measured are concerning given that cumulative heat stress at Lizard Island was not as high as other parts of the Great Barrier Reef, and was lower than during previous bleaching events occurring on this reef. NOAA Coral Reef Watch recorded a maximum 6 °C-Heating Weeks (DHW) at Lizard Island, where it ranged up to 14 °C-weeks in the southern parts of the Great Barrier Reef. In our surveys, nearly 90% of coral cover was bleached or dying, so it is feasible to expect similar or worse post-bleaching mass mortality in areas with similar habitat characteristics and coral assemblages that were exposed to more thermal stress. Early reports from southern parts of the Great Barrier Reef seem to confirm this, with bleaching mortality rates ranging from 23 to 92% depending on the coral genera (Byrne et al. 2025), with the fast-growing species driving most coral bleaching recovery also the most likely to perish.

Calibration and validation of novel approaches is a key aspect of their integration (Naethe et al. 2023), and unfortunately formal in-water validation was not possible in this

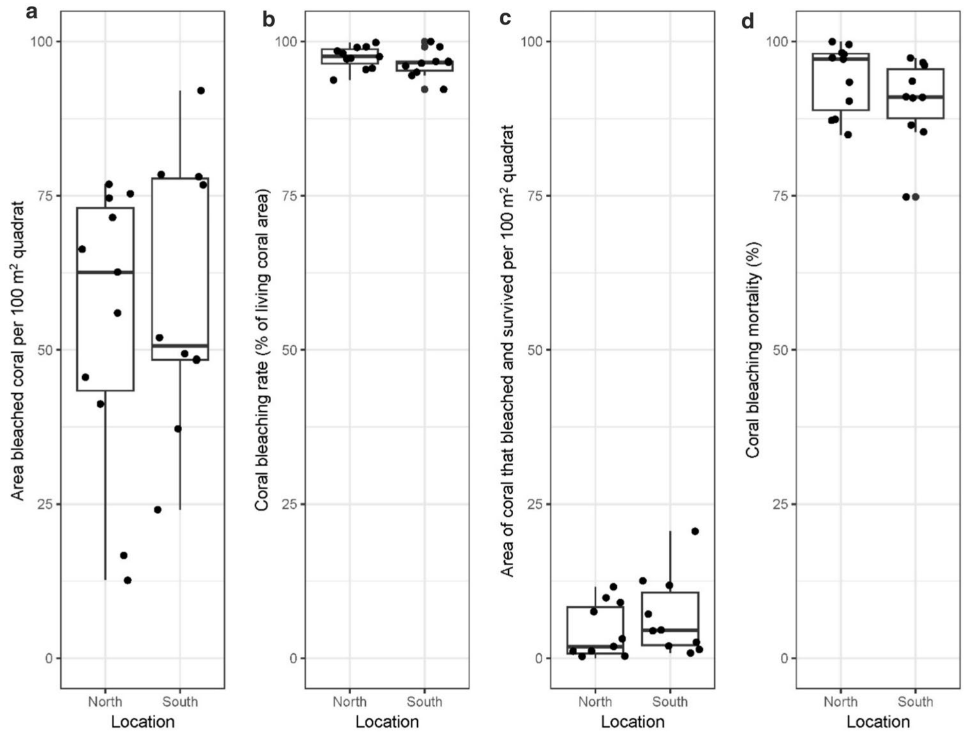

Fig. 6 Median, 5th, 25th, 75th, and 95th percentiles overlayed with raw data of area bleached (**A**), per cent coral area bleached (**B**), surviving (**C**), and bleaching mortality (**D**) of corals in the 20, 10 × 10 m survey quadrats across North Point Reef (north) and Palfrey Island Reef (south) at Lizard Island, Great Barrier Reef

instance due to the opportunistic nature of the surveys. Haphazard snorkel surveys did validate bleaching and subsequent mortality, but more consistent validation at locations distributed throughout the study that could be georeferenced with drone imagery would have been preferable. The independent and highly congruent classification of live/dead coral does provide a measure of consistency in the observations that suggests other coral scientists would agree with our mortality assessment, with relative standard deviation values much lower than what other fields view as 'good precision' (~1% for bleaching mortality here, while 5% RSD is generally acceptable inter-observer variation (Campana 2001)). Objective classification using image parameters, independent of observer visual qualification, would add another layer of validity to these assessments. However, in-field RGB colour calibration of these images would have been necessary (Menesatti et al. 2012), and in our case made more complex since different drone platforms and cameras were used. Harnessing machine classification approaches such as segmentation tools would allow more objective and rapid classification of live/dead coral (e.g. (Bennett et al. 2020; Giles et al. 2023)). Such an approach would also allow automated tracking of the fates of individual colonies at the reefal scale (Stone et al. 2024) rather than the 'small' quadrats and areal cover approaches used herein, to test factors such as colony size that may impact bleaching susceptibility (Brandt 2009). Future studies should integrate these machine learning approaches to further validate data outputs and provide greater insights.

Our results are concerning for the corals, considering predictions of increasing frequency and intensity of extreme events impacting coral reefs in the near future (Emslie et al. 2023), along with several studies showing a growing inability of corals to successfully recover (e.g. Tebbett et al. 2022; Brown et al. 2023b), and little apparent natural selection for heat-tolerant corals despite predictions (Lachs et al. 2023). As of writing this article, NOAA (Liu et al. 2018) is forecasting Coral Bleaching Alert Level 2 for the March and April 2025 periods at Lizard Island, meaning the few survivors or new coral recruits in the measured areas could be exposed to successive annual bleaching events. Our data emphasize the importance of ongoing and targeted research to understand why coral bleaching resilience is not increasing despite repeated bleaching events, and the need to rapidly reduce fossil fuel use and emissions. The use of drones for these purposes, as in this project, combined with advanced computer vision approaches should form a key part of responding to these events.

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Author contributions VR, KJ, GR, JEW conceptualised the study, GR, JL, GC captured the drone mapping data, VR, KJ, JL, GR, JEW analysed data, VR wrote the initial draft of the manuscript, VR, KJ, JL, GR, GC, JEW edited, reviewed and finalised the manuscript for submission.

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Data availability All drone orthomosaics are available on the Geonadir FAIR portal at https://tinyurl.com/lizard-island-bleaching. Code for DHW analysis https://github.com/marine-ecologist/nGBR-DHW.

Declarations

Conflict of interest The authors declare that they have no conflict of interest.

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