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RESEARCH

A quantitative comparison of tools for monitoring the abundance of crown-of-thorns starfish and coral cover

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Abstract Effective monitoring of crown-of-thorns starfish (COTS) populations and coral cover is critical for informing management of COTS outbreaks throughout their tropical Indo-Pacific range, including Australia's Great Barrier Reef. However, existing monitoring tools such as manta tow and cull diver surveys have well-known limitations including accuracy and resource demands. This study evaluates a range of monitoring tools for assessing both COTS and coral populations. Tools that measure both COTS and coral include manta tow surveys, scooter-assisted large area diver-based (SALAD) surveys and surveys undertaken using the ReefScan towed camera platform. For COTS-only monitoring, we examine cull diver surveys and environmental DNA (eDNA) sampling. Data from side-by-side deployments on seven to ten reefs (depending on the tool) with varying COTS densities and coral cover were collected and analysed to calibrate estimates between tools. The calibration models developed will aid in integrating data from diverse sources and facilitate the translation of estimates based on one monitoring tool to another. Critically, the models provide a means of estimating COTS density (COTS per hectare) for monitoring tools that do not directly collect this measure, allowing more meaningful decision-making around ecological thresholds and enabling threshold-based management regardless of which tool collected the data. The study emphasises that each tool has specific applications for addressing knowledge gaps and informing management decisions, advocating for an integrated, multi-tool monitoring strategy that leverages the strengths of each tool to improve the overall effectiveness and efficiency of COTS and coral monitoring.

Keywords Crown-of-thorns starfish (COTS) · Coral monitoring · Monitoring tool calibration · Tropical Indo-Pacific

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Introduction

The crown-of-thorns starfish (COTS; *Acanthaster* spp.) is a large-bodied starfish that predominantly feeds on reef-building corals (Foo et al. 2024). COTS are endemic to tropical reef ecosystems across the Indian and Pacific Oceans, including Australia's Great Barrier Reef (GBR)), and constitute a species complex of at least five species (Uthicke et al. 2024a). While COTS are naturally present in these ecosystems, their populations can sometimes surge, leading to outbreaks (Babcock et al. 2016). During outbreaks, the increased size and abundance of COTS can cause significant damage to coral reefs due to the cumulative feeding pressure that can cause extensive coral loss (Pratchett et al. 2017). Despite emerging pressures, such as increasing prevalence and severity of bleaching events (e.g. Bozec et al. (2022)), outbreaks of COTS remain one of the leading causes of coral loss and reef degradation on the GBR (Emslie et al. 2024), and undermine resilience to other escalating disturbances and threats (Pratchett et al. 2014). For example, De'Ath et al. (2012) estimated that 42% of the observed coral loss from 1985 to 2012 on the GBR was due to COTS outbreaks.

Given the loss of coral caused by COTS, direct population control is an obvious and direct solution (e.g. Yamaguchi (1986)). The largest existing control programme, the GBR COTS Control Program (managed by the GBR Marine Park Authority), uses an Integrated Pest Management (IPM) framework to deliver a tactical response to COTS outbreaks and to mitigate damage to the reef system (Fletcher and Westcott 2016; Westcott et al. 2016; Rogers et al. 2023; Matthews et al. 2024). The COTS IPM uses a combination of data, modelling and real-time observations to prioritise reefs based on their ecological and economic value. The Program, formally established in 2012, has led to substantial coral benefits at reef and regional scales (Matthews et al. 2024). While the COTS IPM Program is a single example, it is clear that the success of any control program depends on the availability of quality data on which management decisions are based. In general, monitoring data on COTS and coral is essential for: 1) strategic decision-making, such as determining which regions or reefs should be prioritised for on-water control operations, 2) making tactical decisions about which parts of reefs require the most COTS control and 3) assessing and reporting the outcomes of the control program (Westcott et al. 2021; Matthews et al. 2024).

A key challenge in COTS management is monitoring populations relative to ecological thresholds that determine when control interventions are warranted. Fletcher et al. (2020) operationalised the ecological thresholds from Plagányi et al. (2020) by establishing discrete CPUE targets of 0.04 COTS culled per minute where coral cover is less than 40% and 0.08 COTS culled per minute where coral cover is 40% or greater. However, these thresholds are defined in terms of cull dive CPUE, which is resource intensive and spatially limited. For other monitoring tools to inform threshold-based management decisions, their observations must be translatable into equivalent density or CPUE units. This translation capability is currently lacking, limiting the operational flexibility of the control program.

There are several established and emerging tools for measuring the density of COTS on a reef (e.g. Miller et al. 2009; Uthicke et al. 2018, 2022; Chandler et al. 2023), and some of these tools are also able to simultaneously measure coral. Each tool has strengths and weaknesses, and these will make some tools more/less appropriate for different applications. The GBR COTS Control Program currently relies heavily on data collected by manta tow for broad-scale COTS surveillance and coral cover estimates (Miller et al. 2009), and COTS Control Program divers for finer-scale measurement of COTS densities that are estimated during cull dives (Fletcher and Westcott 2016). The manta tow technique was developed in 1969 for surveying reefs in Micronesia that had been impacted by COTS (Chesher 1969) and is now the predominant COTS and coral monitoring tool used on the GBR. The method is generally used to provide a broad-scale assessment of large areas of reef, including detecting inter-annual changes in the abundance and distribution of COTS and coral (De'Ath et al. 2012; Vanhatalo et al. 2017; Emslie et al. 2020). Coral cover and COTS densities have also been estimated via manta tow at a consistent set of GBR reefs across large scales since 1985, by the AIMS Long-Term Monitoring Program (LTMP). Thus, LTMP data already provide a long time series that is an important historical data source in the GBR (Emslie et al. 2020).

Manta tow surveys can be conducted over large areas quickly and using minimal equipment, although the method suffers from low detectability of COTS, due to their often cryptic habit (Fernandes 1990; MacNeil et al. 2016). Accordingly, sampling of COTS is often complemented by recording conspicuous feeding scars that indicate the recent and local occurrence of COTS (e.g. Kayal et al. (2012)), though it is not possible to verify the cause of apparent feeding scars during manta tow surveys. This greatly constrains the capacity to effectively survey COTS and guide culling interventions during periods of coral bleaching, white coral disease or *Drupella* infestations which may leave similar scars (white coral skeleton). The measurement variation of COTS counts and coral cover from manta tow surveys can also be influenced by the skill and experience of the observers and may lead to inconsistent observations and potential bias in the data (Moran and De'ath 1992; Miller et al. 2009). Fernandes et al. (1990) and Fernandes (1990) attempted to quantify the degree of undercounting of COTS during manta tows along coral reefs. An average of 23% of starfish counted during SCUBA swim surveys were observed during manta tows over the same transects. The factors that were shown to have a significant influence on the degree of undercounting were the proportion of starfish that are cryptic, the site complexity, underwater visibility, the individual observer, whether or not the boat driver was steering into the sun, and the state of the tide (Fernandes et al. 1990). In short, the appeal of manta tows is in the ease of deployment, its ability to detect COTS at medium to high densities, ability to estimate COTS and coral on a common transect and the extent of historical data. On the other hand, a detraction of manta tows is that they are known to produce biased and noisy measurements and are less effective at low COTS densities.

The cull dive data is a natural by-product of the culling process itself and because of this, the amount of historical data is already large (e.g. divers spent 16,657 h searching for COTS in the 2023/4 financial year). Divers record the number of COTS Control Program dive teams located and culled over a period, resulting in a measure of catch per unit effort (CPUE). As a measurement tool, cull diving is laborious as it requires multiple divers (GBR IPM protocol is for six to eight divers for any one dive). The COTS Control Program follow this protocol, as a team with less divers may not be able to search the entire cull site, likely focussing attention on the habitat with maximum available coral prey and/or highest COTS densities which would invariably increase the CPUE. The amount of dive time is restricted by dive plan protocols and so there are a limited number of surveys that can be taken within each voyage. However, estimation of COTS counts via dive surveys is perceived to be relatively accurate (MacNeil et al. 2016): there are many divers searching for COTS, including under complex parts of reef, within a relatively small area. Fernandes et al. (1990) demonstrated that under experimental conditions, SCUBA divers are likely to underestimate available COTS by 11%, though more recent research suggests this underestimation may be substantially higher, with at least 20% of starfish evading detection even during highly intensive surveys in small sample areas (Pratchett and Caballes 2025). This is likely to be even higher for cull divers covering larger search areas under varying conditions but is accurate compared to other methods like manta tow. Importantly, CPUE data must be approximated to density estimates before they can be related to the ecological status of reefs or incorporated into ecological models. For example, Plagányi et al. (2020) estimated that the ecological threshold in COTS density is reached when control actions achieve a CPUE of 0.04 COTS per minute dive bottom time in locations where coral cover is lower than 40% and 0.08 where coral cover is greater than or equal to 40%.

The limitations in the established tools for monitoring COTS are well known, prompting the recent development of new tools such as environmental DNA (eDNA) (Uthicke et al. 2018, 2022), Scooter assisted large area diver surveys (Chandler et al. 2023) and ReefScan technology (Australian Institute of Marine Science 2024). Each tool has been developed for a different application and no one tool is expected to meet all the diverse information and monitoring needs of the GBR COTS Control Program. For example, ReefScan is designed to simultaneously estimate COTS and coral, while SALAD surveys undertake complementary coral transects, and the current application of eDNA is for COTS only. Some immediate questions around the implementation of these methods are: "Which measurement tools are suitable for each COTS Control Program process (strategic, tactical or reporting)?" and "How do the measurements relate to each other so the resulting data can be combined into single analyses?" To answer these questions, it is important to understand how the tools compare in terms of deployment characteristics, including detectability, effective search effort sampled (and units) and what the collected metrics represent (e.g. visible COTS only, COTS and scars, or in the case of eDNA, all COTS including cryptic and juvenile animals). The emerging tools are likely to improve the effectiveness of the monitoring program by facilitating monitoring of lower COTS densities and reefs that may not be suitable for in-water monitoring, increasing data volume, and ultimately decreasing the uncertainty surrounding the process of decision-making. New tools also have the potential to change how and when COTS surveys are undertaken, potentially leading to increased efficiency. For example, rapid assessment tools such as eDNA can be used to check on areas not surveyed by other methods and existing methods can be adapted using emerging technologies for use by community, NGO and Traditional Owner groups. This allows for new survey designs and improved logistics giving a wider range of spatial and temporal data to managers.

As monitoring programs evolve, a key challenge lies in balancing the adoption of innovative technologies while maintaining the integrity and continuity of long-term datasets (Murray and Krebs, 2024). While this challenge will become increasingly prevalent as new technologies are introduced to long-term monitoring programs, it remains largely unaddressed in the current literature. Existing studies on measurement comparisons typically assume both measurements are in the same units (Choudhary and Nagaraja 2017), which is often not the case when comparing novel technologies with traditional methods. Before integrating new tools into established monitoring programs, it is crucial to understand their statistical properties and relationship to existing methodologies, despite these unique challenges (Olsen et al. 1999). This understanding guides strategic decisions about measurement methods and timing, addressing several critical questions: 1) How do estimates from different methods compare, including the magnitude and direction of variations, and potential systematic differences, particularly when the measurements are in different units? 2) Do certain tools demonstrate superior performance in specific environmental contexts? 3) How do emerging methodologies perform relative to established techniques when neither can be assumed to provide the "true" value?

To answer these questions, this work focusses on statistical calibration (Osborne 1991), which attempts to quantify the sources of variation that can lead to differences among measurements produced by different tools.

Calibration allows measurements from one monitoring tool to be expressed in the units of another and so enables the decision-making process to proceed with multiple sources of data. The most efficient and effective method to perform this comparison is to deploy novel and established monitoring tools "side by side". In this paper, we describe an experiment where we deployed the COTS and/or coral monitoring tools previously described, side by side. The experiment visited a total of 10 reefs with different environmental characteristics. Statistical models were developed to calibrate the COTS and coral cover estimates produced by each tool with each other relevant tool.

Materials and methods

Monitoring tools

The study compared five monitoring tools for measuring COTS and/or coral cover on the GBR: two methods that measure both COTS and coral simultaneously (manta tows (Miller et al. 2018) and ReefScan camera systems (Australian Institute of Marine Science 2024; Bainbridge and Coleman 2024)), two methods that measure COTS only (cull dives (Fletcher and Westcott 2016) and eDNA (Uthicke et al. 2018, 2022)) and SALAD surveys that measure COTS over a 1 km transect but also undertake a standard 50 m linear coral transect at the beginning and end of the SALAD survey (Pratchett et al. 2009; Chandler et al. 2023). Two differing camera systems were tested, a transom mounted fixed-depth camera system (ReefScan Transom) and a towed variable-depth platform (ReefScan Deep), each using the same camera design. Each method offers unique advantages and operational challenges (Table 1).

Primary side-by-side experiment

To obtain directly comparable data for calibration between all monitoring tools, a side-by-side experiment was performed in the central Great Barrier Reef (GBR) in March 2023. The structure of the experiment was straightforward: take measurements with each of the monitoring tools for a specified area of a reef (coinciding with an IPM-defined cull site (~ 10 ha)) using the protocols that each monitoring tool would usually employ. This method of side-by-side deployment minimises sources of spatial and temporal variability by taking measurements at the same sites within a short timeframe but includes all sources of variation that would be present if any of the monitoring tools were deployed separately.

Sampling was conducted at a total of 16 sites across seven reefs (Table 2), with the table showing the number of times each method was deployed. Multiple measurements were typically taken within each site (e.g. multiple manta tow transects, multiple eDNA samples), and this within-site variation is incorporated into our hierarchical model structure described below. Sites were selected across the region in a preferential way to cover a range of COTS densities (as much as the available information would permit) to ensure that the monitoring tools are calibrated across as wide a range of values as possible. This is beneficial for this calibration experiment as it provides the most information for the relationships between the different tools' observations. The plan was to deploy every monitoring tool at every site (complete block design), and this was achieved at most sites. Departures were due to diver protocols and gear repairs. The standard order of operation at each site consisted of manta tow, eDNA, ReefScan Transom, SALAD, ReefScan Deep and then COTS culling. This order was chosen for logistical and scientific considerations (e.g. cull diving should occur after eDNA collection). We acknowledge that this fixed ordering is not random (a best practice), but full randomisation was not achievable due to resource constraints, and this fixed order not only satisfied the constraints but also increased in-field efficiency and safety. Deployment was almost always completed in similar weather conditions (light, wind, tide state).

We designed this study to produce calibrations applicable to routinely collected monitoring data under operational conditions. While highly standardised, replicated experimental designs (e.g. Fernandes et al. 1990; Moran and De'ath 1992) maximise statistical power to detect differences between methods, they do so by minimising sources of variation that are unavoidable in operational deployment, such is the case for the COTS Control Program. Our approach deliberately incorporates realistic operational variation including natural differences in spatial coverage between methods (Table 1), standard deployment protocols for each tool, observer variation within operational teams, temporal variation within site visits, environmental variation (visibility, currents, tide state), and the reality that exact spatial overlap is not achievable between methods with fundamentally different deployment modes (e.g. eDNA water samples vs cull dives). This design prioritises generalisability to real-world operations over precision under controlled conditions. The resulting prediction intervals reflect the uncertainty managers will experience when using these calibrations to make operational decisions.

Complementary data

To provide complementary data for a subset of the monitoring tools, a second sampling expedition was carried out in January 2024 in the southern GBR, departing from Gladstone (Table 2). The measurement tools utilised were manta tow and ReefScan Transom, with both being deployed

Table 1 Summary of the COTS and coral monitoring methods, including operating procedures, metrics, scale features and limitations. * Denotes the metric was used in the calibration analysis

Method	Description	Metrics captured	Scale	Key features	Primary limitations
Manta tow	Towing a snorkel diver at ~ 2 knots for a 2-min transect (~ 200 m), visually assessing COTS and coral cover	COTS count, COTS scars category*, coral cover category* (e.g. 0%, 1–5%, 6–10%, 11–20% etc.)	~ 200 × 10 m, (0.2 ha, multiple transects per cull site)	Quick, broad-scale surveys, minimal equipment needed, standardised methodology	Limited to good visibility conditions, observer fatigue and skill can affect accuracy, restricted to surface observations, underestimates cryptic COTS
Cull dive surveys	6—8 divers search cull sites, cull COTS and record data on numbers, size and search time to calculate a CPUE	COTS count*, COTS size, CPUE*	~ 200 × 500 m (10 ha)	Actively removes COTS, relatively high detection accuracy, provides size data	Labour and resource intensive, requires specialised training, limited dive hours
SALAD surveys	Diver pair uses scooters, each covering ~ 1 km transects within a 5-m-wide belt to record COTS, feeding scars and GPS location. Scars are only recorded if they cannot be attributed to a nearby COTS, i.e. they are a proxy for cryptic COTS	COTS count*, COTS scars*, COTS size, GPS location	~ 1 km x 10 m (1 ha)	Can detect low COTS densities, GPS tracking, records both COTS and feeding signs, sufficient time to search for COTS and analyse scars to determine if multiple scars belong to a single COTS	Dependent on diver experience, specialised equipment and maintenance required, may miss small or cryptic individuals
Coral composition transects (before/after SALAD)	50 m linear coral transects pre-/post-SALAD with coral composition recorded at 100 points along the transect	Coral composition, coral cover*	~ 50 × 5 m (0.025 ha)	Very well-established method	COTS are also recorded but not analysed due to known patchiness and/or inaccuracies at such a small spatial scale, transects are very short
ReefScan surveys	Machine learning-driven underwater camera systems (towed (ReefScan Deep) or transom mounted (ReefScan Transom)), analysing benthic images to determine coral cover	Coral cover*	Continuous (~ 8 m width at 5 m from bottom)	Can be towed continuously or subset to transects, automated analysis reduces human bias, permanent visual record, consistent methodology	Limited to visible surface features, accuracy dependent on ML model training, ML model training being performed for COTS and COTS scars but not yet deemed reliable at time of experiment
eDNA surveys	Environmental water sampling analysed via digital droplet PCR to detect COTS DNA fragments	Proportion of samples containing COTS eDNA*, average copy number	Unknown	Can detect low COTS densities, non-invasive sampling, can be undertaken in areas of low visibility/high dive risk, can detect cryptic and juvenile COTS	Laboratory processing delay, influenced by water movement, no direct coral cover measurement

Table 2 Number of cull sites each method was deployed at for each reef. The manta tow and ReefScan typically recorded multiple transects per site when towing the perimeter of the reef

Reef	Departing from	Cull dive	Manta tow	SALAD	eDNA	Reef-Scan Deep	ReefScan Transom
Banfield North	Townsville	2	2	2	2	2	2
Darley	Townsville	4	4	3	4	4	4
Davies	Townsville	2	2	2	2	1	2
Faith	Townsville	1	1	1	1	1	1
Lynchs	Townsville	1	4	4	4	4	4
Prawn	Townsville	2	2	2	2	2	2
Shrimp	Townsville	1	1	1	1	1	1
U/N Reef (21–057)	Gladstone	0	18	0	0	0	18
U/N Reef (21–465)	Gladstone	0	10	0	0	0	10
U/N Reef (21–551)	Gladstone	0	16	0	0	0	16

simultaneously. Unlike the Townsville trip, this approach maximises the common ground measured by both monitoring tools. We present coral cover data from the manta tow and ReefScan Transom to provide an alternative view of the calibration relationship for coral. Like the initial fieldwork, the monitoring transects were matched to the closest cull site for analysis purposes.

Statistical analysis

To demonstrate the overall coherence of the observations from the different monitoring tools, we calculated Spearman's rank correlations among all pairs of site-level COTS metrics and among all pairs of coral cover metrics. These correlations, which measure the strength of the relationship irrespective of linearity, scale and distribution, provided an initial check that the tools responded monotonically to comparable ecological gradients.

The goal of the statistical analysis was to form a quantitative calibration for each pair of measurement tools. This was achieved by forming multiple generalised linear mixed models (GLMMs; McCulloch (2003)) that explain each measurement tool's observations from each of the other measurement tools' cull site-level observations. For example, models were developed to predict cull divers' data from each of the manta tow COTS measurements, SALAD data and eDNA data, and vice versa. The exact form of each of the regression models depends on the nature of the outcome variable (e.g. whether a count or a binary variable) and the effort metric (e.g. dive time or area searched).

As an example, consider explaining the cull data from the SALAD data. The cull data consists of: 1) the number of COTS culled at each site and 2) the total time taken at each site. Apart from an intercept, the GLMM contains a fixed effect for the log of the SALAD CPUE (count divided by time), a random effect for the reef and an offset of log cull dive time. A log link was used, and the sampling variation was assumed to follow a negative binomial distribution. The relationship between the cull data and the SALAD data is log–log, which allows for both asymptotic and exponential growth. The reef random effect allows for reef-to-reef variability and accounts for possible within-reef correlation.

We chose GLMMs for several important reasons. First, our monitoring tools produce diverse outcome types (counts, proportions, ordinal categories that are translated to proportions) that are not well accommodated by standard correlation approaches. Second, the hierarchical structure (sites nested within reefs) requires random effects to account for reef-to-reef variability and within-reef correlation. Third, the monitoring tools all operate at different spatial scales (0.025 to 10 ha; Table 1) and use different effort units (time, area, transect length), which GLMMs can accommodate through appropriate link functions and offsets. GLMMs provide the predictions with uncertainty that managers require: given an observation from Method A, what would Method B likely observe, with what prediction limits? Finally, as this is a calibration study, not a causal analysis, the regression relationships are symmetric and bidirectional.

We performed all pairwise calibrations between methods (available in the RosettaCOTS package); the choice to present results organised by certain methods (cull dives for COTS, manta tow for coral) reflects current operational practice and presentation clarity rather than assumptions about which method is most accurate. Inference for the example model (explaining the cull data from the SALAD) was conducted using Bayesian methods. This required specification of priors for each of the parameters: vague Gaussian priors for the fixed effects, a vague log-normal prior for the variance of the random effect and a vague gamma prior for the dispersion of the negative binomial. The posterior distribution was approximated using samples from a Gibbs sampler in the JAGS software (Plummer et al. 2016). The posterior samples are made available through the R package

RosettaCOTS (Foster 2024), along with functions that predict all sampling methods from all others.

The models for data from the other sampling tools follow a similar pattern. Differences among models arise from different types of outcomes (negative binomial for counts, binomial for success/failure data and beta for proportion data), different link functions to follow the outcome type and different structure in the sampling strategy requiring extra random effects for sites (e.g. eDNA has samples and laboratory replicates).

To characterise the extent of within-reef and within-site variability available in our dataset, we extracted variance components (reef-level variance and site-level variance where relevant) from each GLMM. These components quantify how much unexplained variation is attributable to the different spatial scales of sampling after accounting for the fixed effect linking two methods. Importantly, the degree to which within-site variability can be estimated differs among monitoring tools because several methods do not include within-site replication in the operational program (e.g. cull dives produce a single site-level metric). We report the corresponding variance components for each calibration model, noting cases in which the site variance component is non-estimable due to the structure of the underlying sampling design.

Results

All monitoring tools showed positive rank correlations, although the strength of the relationships varied substantially (Fig. 1-S2). Cull rates showed moderate correlations with SALAD (time and distance) and eDNA, while manta tow feeding scar prevalence was only weakly correlated with these measures. SALAD time and distance metrics were very strongly correlated with each other and strongly associated with manta tow observations. These patterns suggest that while the tools generally respond to similar patterns in COTS presence, the strength of agreement among methods varies considerably. Coral cover metrics showed moderate to strong agreement between most pairs of tools, with the strongest relationship observed between the two ReefScan platforms, while coral transects and manta tow estimates showed a weaker but still positive association.

The SALAD COTS rate is presented as both density of COTS and scars (per hectare), in addition to the frequency of encounter (per minute). While the per minute metric allows direct comparison with the culling rate (also directly measured per minute), the dive area searched (ha) can be estimated and density of COTS is the most meaningful measurement, both from an ecological and management perspective. Predicting the SALAD observed COTS or scars per minute from the COTS culled per minute (not including scars), reveals a mean higher encounter rate on SALAD at the full range of CPUEs observed by the cull divers (0–0.2 COTS per minute, Table 2).

The proportion of manta tow transects with scars (used because no COTS were observed on manta tow) predicted by the cull rate reveals a highly uncertain relationship with the prediction interval spanning the full range between 0 and 1 (Fig. 1). The proportion of positive eDNA samples can be predicted via the cull data with less uncertainty than the manta tow, with a strongly increasing trend at low densities starting to flatten out around 0.1 COTS per minute (equivalent to approximately 31 COTS and scars per ha on SALAD surveys, Table 3).

These predictions enable managers to translate observations from any tool into equivalent COTS densities or rates. For example, at the lower ecological threshold for COTS control (0.04 COTS per minute cull CPUE; Plagányi et al. 2020), SALAD surveys would be expected to observe approximately 20.6 COTS and scars per hectare, while eDNA would show approximately 54% of samples positive. The width of prediction intervals varies substantially among tool pairs, with manta tow showing the widest uncertainty (Table 3, Fig. 1C), indicating that different tool combinations have different reliability for cross-method translation. The high variability reflects the inherent noise in measuring rare and cryptic organisms, as well as the hyper-stable relationship between CPUE and density (Plagányi et al. 2020).

The relationship between SALAD coral cover transects (measured using point transect methods) and manta tow coral cover is highly uncertain. This is likely because the SALAD coral cover uses short 50 m transects that do not spatially overlap with the manta transects, which is recorded in ordinal categories (the mid-points are used in analysis), also contributing to the high degree of uncertainty.

The relationship between manta-derived estimates and ReefScan Deep measurements is less uncertain. At lower coral densities (up to around 15%, Table 4), the estimates from both monitoring tools are similar, but at higher densities, ReefScan Deep estimates tend to be higher than those from manta tows. Results for ReefScan Transom are presented separately for the two trips (primary and secondary, Fig. 2). The primary trip (off Townsville) shows that coral cover estimates from ReefScan are almost double those from manta tows, while the secondary trip resulted in much lower estimates. The estimated relationships are affected by the range of coral cover estimates encountered which were much broader on the second trip (0–80%) than the first (0–25%). High inter-diver variability (Miller et al. 2009) or possible differences in deployment protocols in the secondary experiment (where the manta tow and ReefScan were co-deployed to ensure they ran over the same survey area, see Methods) could also contribute to the discrepancies.

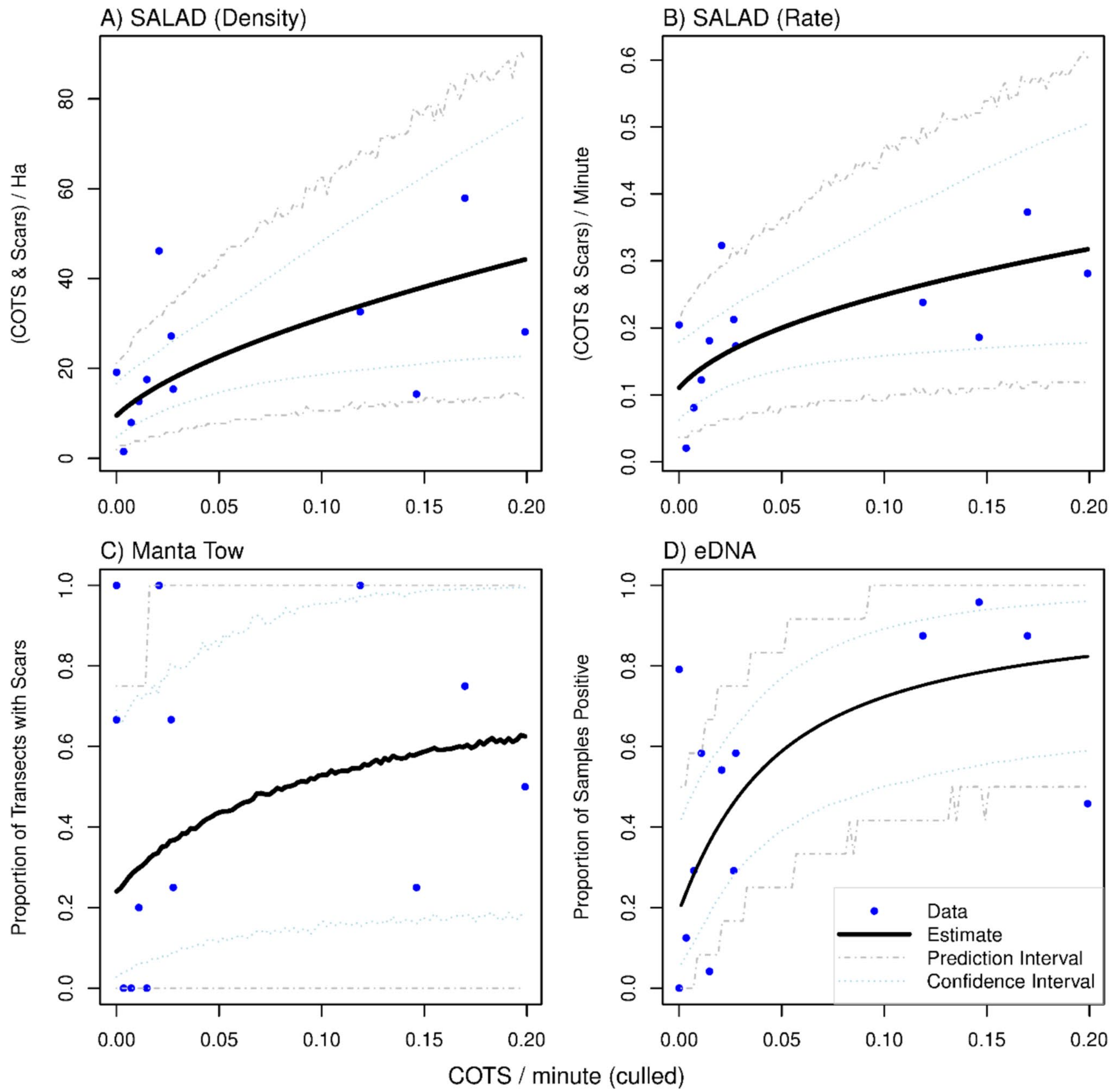

Fig. 1 COTS densities/rates (y-axis) predicted by cull diver surveys (x-axis) for A) SALAD surveys (COTS and scars per hectare), B) SALAD surveys (COTS and scars per minute), C) manta tow density (proportion of tows with scars) and D) eDNA (proportion of samples positive). The black solid line is the estimated relationship, the grey dashed line is the prediction interval, and the blue dashed line is the confidence interval

The coral cover relationships show tighter prediction intervals than COTS metrics for some tool pairs (Fig. 2), though substantial uncertainty remains. The two separate deployments of ReefScan Transom (primary and secondary experiments) show different calibration relationships with manta tow (Fig. 2C-D), highlighting the influence of deployment conditions and the importance of continued data collection to refine calibrations.

Estimated variance components from the GLMM analyses revealed differences among monitoring tools in the scale and structure of random variation (Tables S1–S2). In the coral models, reef-level variance was generally modest, which confirms relatively similar coral cover estimates across reefs. Among COTS models, reef-level variance was large, reflecting the aggregated spatial distribution of COTS across reefs. For the calibration analyses where a site-level

Table 3 Predicted COTS densities/rates and associated prediction intervals for each COTS measurement tool, based on cull rates of 0.00–0.20 COTS per minute culled

Cull rate (COTS per min)	SALAD density (COTS & scars per ha)	SALAD rate (COTS & scars per min)	Manta tow (proportion with scars)	eDNA (proportion samples positive)
0.00	9.50 (2.71, 22.30)	0.11 (0.02, 0.27)	0.24 (0.00, 1.00)	0.19 (0.00, 0.50)
0.02	15.97 (5.42, 34.60)	0.16 (0.05, 0.34)	0.34 (0.00, 1.00)	0.41 (0.08, 0.75)
0.04	20.62 (6.61, 42.44)	0.19 (0.06, 0.38)	0.40 (0.00, 1.00)	0.54 (0.17,0.92)
0.06	24.52 (8.35, 53.10)	0.21 (0.06, 0.44)	0.46 (0.00, 1.00)	0.63 (0.25, 0.92)
0.08	27.98 (9.35, 62.98)	0.23 (0.07, 0.47)	0.50 (0.00, 1.00)	0.68 (0.33, 1.00)
0.10	31.13 (9.59, 71.05)	0.25 (0.07, 0.53)	0.52 (0.00, 1.00)	0.72 (0.33, 1.00)
0.12	34.06 (10.41, 76.68)	0.26 (0.09, 0.56)	0.55 (0.00, 1.00)	0.75 (0.33, 1.00)
0.14	36.82 (11.57, 80.21)	0.28 (0.09, 0.62)	0.58 (0.00, 1.00)	0.78 (0.42, 1.00)
0.16	39.43 (11.68, 91.75)	0.29 (0.09, 0.66)	0.60 (0.00, 1.00)	0.80 (0.42, 1.00)
0.18	41.93 (11.72, 98.63)	0.31(0.09, 0.69)	0.61 (0.00, 1.00)	0.81 (0.42, 1.00)
0.20	44.33 (12.67, 111.94)	0.32 (0.09, 0.69)	0.62 (0.00, 1.00)	0.82 (0.42, 1.00)

Table 4 Predicted coral cover (%) for each measurement tool, based on estimated manta tow coral cover ranging from 5 to 25%

Manta tow	Coral transect	ReefScan Deep	ReefScan Transom (primary)
5.00	10.97 (5.86,18.00)	7.79 (0.47, 23.39)	11.99 (2.41, 27.69)
10.00	13.47 (7.62, 21.38)	11.93 (1.98, 29.80)	17.56 (5.11, 34.51)
15.00	16.45 (9.50, 25.50)	18.04 (4.71, 39.46)	25.12 (9.72, 44.73)
20.00	19.94 (12.00,29.75)	26.53 (9.33, 48.29)	34.64 (16.13, 56.05)
25.00	23.98 (14.50,35.50)	37.25 (15.48, 61.22)	45.53 (23.29, 67.75)

variance component was estimated (those predicting a measure with within-site replication: manta and eDNA), the estimate of this component was large, indicating substantial variation. This site-level variance component is not estimable for all the remaining methods (no within-site replication), but an over-dispersion parameter was estimated. These over-dispersion parameters also showed that there was substantial variation, with some methods being extreme (e.g. all calibration models predicting manta tows). These variance patterns indicate that uncertainty in calibration relationships arises from a combination of reef-level, site-level and observation-level variability, and that the GLMMs appropriately accommodate the sampling structure of each tool.

Discussion

This study develops calibration models that enable translation between COTS and coral estimates from different monitoring tools, facilitating the integration of data from diverse sources. The models provide managers with both point predictions and quantified uncertainty for converting estimates between tools. All pairwise calibrations are available through the RosettaCOTS R package, allowing bidirectional translation between any pair of methods, depending on operational needs.

Monitoring tools differ in their sampling structure and the degree of replication they provide, which directly influences the precision of calibration models. Methods such as eDNA and manta tows generate multiple observations within a site, allowing estimation of within-site variability, whereas ReefScan and cull data are analysed at the cull site level. These structural differences help explain variation in calibration precision among methods and the relative widths of prediction intervals. While all monitoring tools showed positive relationships, the strength and uncertainty of the relationships varied, reflecting spatial heterogeneity, observer variation, environmental conditions and differences in spatial coverage. As tools such as ReefScan, eDNA, SALAD and manta tow are more routinely deployed together, increasing availability of co-located data will enable continued refinement of calibration models and progressive reduction of uncertainty.

We acknowledge that there is no "gold standard" monitoring tool, each tool has its own drawbacks and benefits. However, for estimating densities and size structure of sub-adult and adult COTS, we suggest that SALAD surveys may provide estimates that are closest to the (unobserved) "true" densities due to their large search area

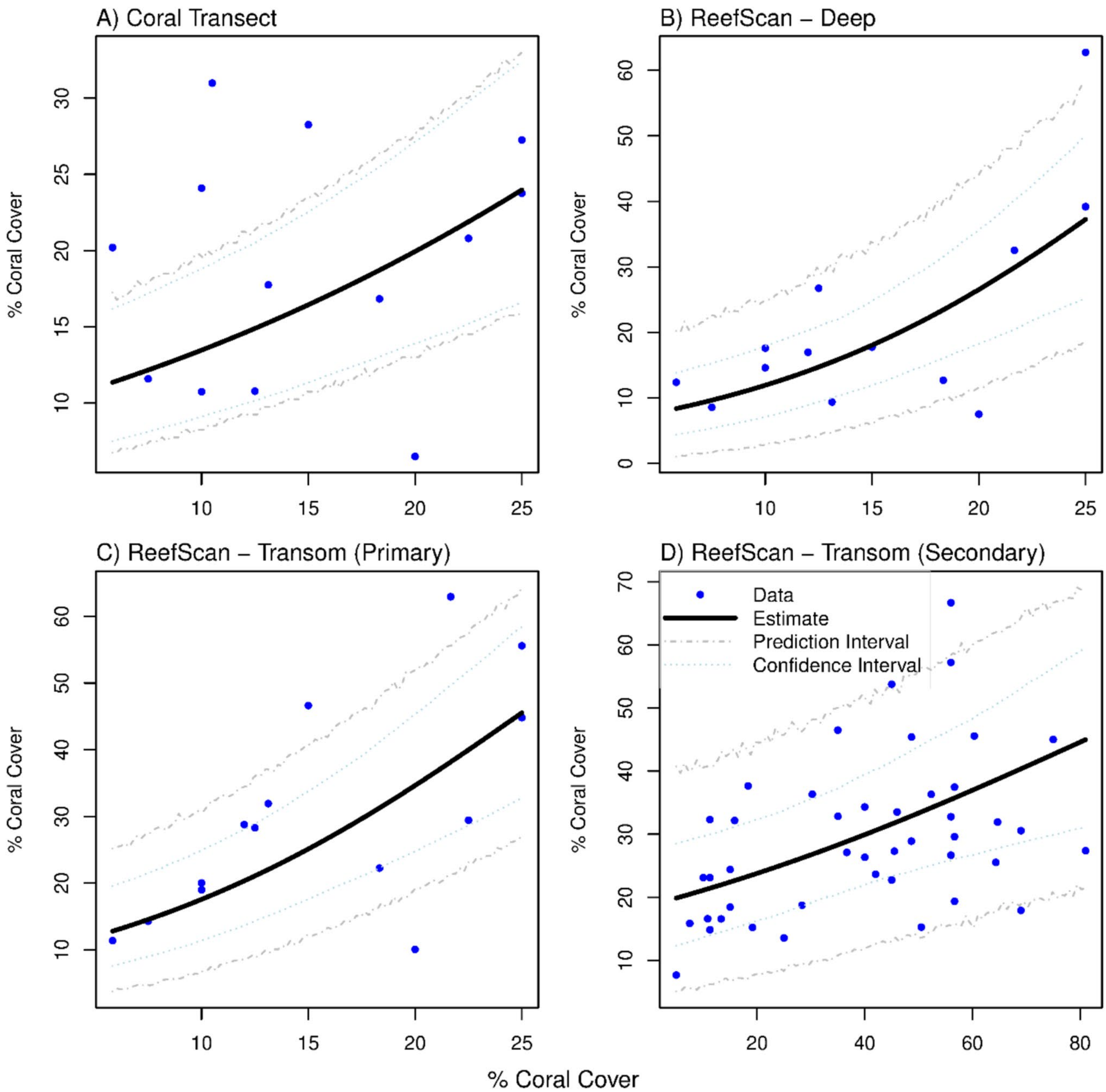

Fig. 2 COTS coral predictions (y-axis) from manta tow (x-axis). A) Coral transect adjacent to SALAD survey, B) ReefScan Deep, c) ReefScan Transom (from the primary experiment) and d) ReefScan Transom (from the secondary experiment). The black solid line is the estimated relationship, the grey dashed line is the prediction interval, and the blue dashed line is the confidence interval

(1 ha per transect), systematic coverage and divers' ability to thoroughly search complex three-dimensional reef structure. Expressing observations as densities (COTS per hectare) provides a common currency for integrating data across methods and is particularly valuable for linking observations to ecological thresholds. We note that among the methods considered here, only eDNA can detect juvenile COTS, though additional fine-scale approaches may be required for comprehensive juvenile monitoring.

Comparison of COTS monitoring tools

The manta tow surveys provide broad-scale estimates of COTS densities and coral cover over large areas. However, as previous studies have noted, the accuracy of manta tow surveys can be impacted by factors such as observer experience, visibility, reef complexity, proportion of cryptic COTS and environmental conditions (Fernandes et al. 1990). Our results align with these past findings but go further by quantifying the relationship with other methods. In general, manta tows tend to underestimate COTS abundance compared to other monitoring tools such as cull diver surveys. At the most basic level, more than 50 COTS were culled at four separate cull sites, yet no COTS (only scars) were observed on manta tow, suggesting that manta tow might underestimate the true density of COTS far more than the 23% suggested by (Fernandes et al. 1990), at least at lower densities. For this reason, the GBR COTS Control IPM uses the presence of COTS or COTS scars on manta tow as a trigger to start culling (Fletcher et al. 2020). The focus in this study was calibrating the monitoring tools based on the way they are/would be deployed in the COTS Control Program where, by necessity, the area searched by manta tow and cull divers is different, but both are used as measures of relative abundance for decision-making. This is a strength of the present study, as it quantifies the relationship between monitoring tools and provides a method to convert to a common density unit (COTS/ha) that can be directly implemented in a large-scale operation.

SALAD surveys showed much higher detection rates for COTS compared to manta tows, likely due to ability to vary dive speed and to stop and better search the 3D coral structure, including looking for COTS in the presence of scars (Chandler et al. 2023). SALAD surveys detected (on average) more COTS than cull diver surveys at lower COTS densities; however, as the detection rate increases, the discrepancy in density estimates between the two monitoring tools decreases. The estimate of ten COTS per hectare on SALAD being equivalent to zero CPUE on cull divers is unrealistic and suggests that more data should be collected to improve the model estimates, particularly at the lower densities. Importantly, the team of culling divers (6–8 divers) have much more consistent survey effort across a broader reef swathe than the SALAD surveys, which leads to greater detection at increasing densities. At relatively low COTS densities (up to 15 COTS per hectare), SALAD surveys will provide equivalent coverage and therefore much greater efficiency than entire culling teams. In addition to COTS densities, SALAD surveys provide more detailed data on COTS size distribution, detectability and recent feeding history (Chandler et al. 2023), crucial for understanding population dynamics and reproductive potential (Pratchett et al. 2021).

eDNA has emerged as an effective tool for detecting and quantifying the presence of COTS, especially at low densities where traditional methods may struggle (Uthicke et al. 2018, 2022). Uthicke et al. (2024b) demonstrate that eDNA sampling can successfully identify inter-annual changes in COTS populations at densities below outbreak levels, a task previously thought to be achievable only through SALAD surveys. The metric chosen here (proportion of positive samples) showed some degree of saturation (most values reaching 100%) at higher COTS densities. It was previously demonstrated that for higher densities an alternate metric (average copy number) may be more appropriate (Uthicke et al. 2022). Similarly, Uthicke et al. (2024b) showed a higher proportion (> 50%) of variance explained between SALAD and eDNA when reef wide estimates are considered (as opposed to location estimates as considered here). The ability of eDNA to provide reliable data across a range of conditions and its scalability makes it a complementary tool alongside existing monitoring tools, offering a cost-effective, safe and efficient alternative for monitoring COTS populations across the Great Barrier Reef.

While ongoing research aims to develop algorithms for automated detection of crown-of-thorns starfish (COTS) and associated feeding scars using ReefScan technology, the methodology was not deemed sufficiently reliable to provide accurate quantitative measures during the field trips conducted as part of this project. Successful implementation of this automated detection technology would significantly enhance data collection capacity due to increased efficiency associated with continuous towing, automated data recording and reduced dive-related constraints. Additional benefits of such data acquisition include precise GPS coordinates of COTS occurrences, facilitating both targeted culling efforts for the IPM process and future studies on COTS aggregation mechanisms.

Tool performance for ecological threshold detection

A primary motivation for this study is enabling more meaningful decision-making around ecological thresholds for COTS management. Plagányi et al. (2020) established and Rogers et al. (2024) validated that the ecological threshold for COTS control is reached at a CPUE of 0.04 COTS culled per minute where coral cover is less than 40% and 0.08 COTS culled per minute where coral cover is 40% or greater. Our calibration models now allow these operationally defined thresholds to be translated into equivalent values for other monitoring tools, facilitating threshold-based management decisions regardless of which tool collected the data.

At the lower ecological threshold (0.04 COTS/min), our models predict a mean value of 20 COTS and scars per hectare on SALAD surveys (95% PI: 6.6–42.4 ha^{-1}), 54% of eDNA samples positive (95% PI: 17–92%) and 40% of manta tow transects with scars (95% PI: 0–100%). At the higher threshold (0.08 COTS/min), our models predict a mean value of 28 COTS and scars per hectare on SALAD surveys (95% PI: 9.4–63.0 ha^{-1}), 68% of eDNA samples positive (95% PI: 33–100%) and 50% of manta tow transects with scars (95% PI: 0–100%). The wide prediction intervals for certain tool pairs (particularly manta tow, Fig. 1C, Table 3) indicate that additional data collection should be prioritised to improve calibration precision. As emerging tools are routinely deployed alongside established methods, incorporating these data into updated calibration models will progressively reduce uncertainty and improve threshold-based management decisions.

Our model predicts that even at zero cull rates (0.00 COTS/min), SALAD surveys would observe approximately 9.5 COTS and scars per hectare (95% PI: 2.71–22.30), while eDNA would show 19% of samples positive (95% PI: 0.00–0.50). These non-zero predictions at zero CPUE may seem counterintuitive but reflect an important ecological and sampling reality: at very low COTS densities, time-limited culling surveys may fail to encounter any COTS even though small populations persist on the reef. This phenomenon is well documented in fisheries science as a hyper-stable relationship between catch rates and abundance (Hoyle et al. 2024). Plagányi et al. (2020) demonstrated this hyper-stable relationship for COTS showing considerable scatter in observed CPUE–density data points at low densities, with some zero-CPUE observations occurring at densities of 50–100 COTS/ha. The practical implication for management is that observations near zero do not necessarily indicate complete absence of COTS but instead indicate very low densities where detection becomes probabilistic.

Coral monitoring

Effective coral monitoring is essential for assessing the impact and efficacy of the COTS Control Program. Using manta tow surveys for coral estimation is a cost-effective method that allows for rapid assessment of large reef areas. However, the accuracy of these estimates can be significantly influenced by several factors, including observer variability, environmental conditions and logistical constraints. Estimates are subject to observer bias, as they are based on visual assessments, which can vary widely between individuals due to differences in perception and experience (Moran and De'ath 1992). While precision can be improved through training and standardisation of methods (Miller and Müller 1999), these processes can be time-consuming and may reduce the time available for other aspects of fieldwork. This presents a challenge for control programs that have limited time for monitoring and rely on a relatively transient workforce, as maintaining data quality requires consistent observer training and calibration (Miller 1997).

The ReefScan technology shows promise in addressing these challenges. There appears to be a positive relationship between Reefscan estimates and manta tow estimates. Its ability to conduct continuous, automated surveys could significantly increase the spatial coverage of coral monitoring. Additionally, the machine learning algorithms employed by ReefScan have the potential to provide more consistent and objective coral cover estimates, potentially reducing measurement uncertainty. Lastly, the images obtained are a resource that can be re-analysed when new image analysis approaches emerge, or to provide data for emerging science and management applications.

Importance of survey design in monitoring programs

While our study focusses largely on comparing different monitoring tools, it is crucial to emphasise that tools are only one component of an effective monitoring program. A comprehensive and reliable monitoring program depends heavily on a robust survey design (Gitzen and Millspaugh 2012; Reynolds 2012; Foster et al. 2020b). The statistical design of a monitoring program determines where, when and how often measurements are taken. It influences the ability to detect changes over time, differentiate between natural variability and human-induced changes, and make inferences about larger areas based on sampled sites (Foster et al. 2020a). By combining an optimal set of monitoring tools with a robust statistical design, the COTS Control Program can significantly enhance its ability to detect and respond to COTS outbreaks, as well as accurately assess its impact on coral preservation.

Limitations and future directions

Several limitations of this study should be acknowledged. First, our sample size (60 sites across 7–10 reefs depending on tool) provides initial calibration relationships but results in wide prediction intervals for some tool pairs, particularly at low COTS densities where early detection is most critical. Second, our sampling was conducted over a relatively short timeframe (March 2023 primary trip, January 2024 secondary trip) and may not capture seasonal or inter-annual variation in tool performance. As the COTS Control Program continues to deploy multiple tools in parallel during routine operations, these data should be systematically compiled to progressively reduce calibration uncertainty. The Bayesian framework we have employed facilitates these updates and the RosettaCOTS package is designed to be updated as improved calibrations become available, ensuring the management community benefits from ongoing data collection.

Conclusions

This study provides a mechanism for converting between the COTS and coral estimates obtained by different monitoring tools, including density estimates for COTS that are essential for management decisions. While this study provides valuable side-by-side experimental data from multiple tools, the current calibration models based on the experiment data alone have very wide prediction intervals.

As emerging tools are introduced into the COTS Control Program, it will be important to continue to collect data to reduce this uncertainty and ensure robust calibration with established tools such as manta tow surveys. This could be achieved by collecting monitoring data simultaneously using both established and emerging tools. For example, routine manta tow surveys could be conducted in parallel with ReefScan surveys. A key limitation lies in the need for substantially more SALAD data to improve the precision of estimates linking culling CPUE to SALAD-derived density metrics. Reduced uncertainty around the estimates would significantly enhance our ability to make informed management decisions, particularly in targeting areas for effective COTS control.

This study demonstrates the potential benefits of integrating multiple monitoring tools in the COTS Control Program. By leveraging the strengths of traditional and emerging monitoring tools and implementing a statistically rigorous survey design, we can significantly enhance our understanding of COTS populations and coral dynamics. An integrated approach such as this will improve the effectiveness of control efforts and our ability to quantify and report on programme outcomes across the GBR, particularly through the translation of estimates obtained via one tool to another. New monitoring tools create opportunities for innovative survey designs that can engage additional stakeholders, enabling the work to scale and extend to previously unmonitored areas, thereby enhancing the spatial coverage and timeliness of management data. The implementation of such a strategy, coupled with ongoing refinement of technologies, methods and design, represents a significant advancement in our capacity for effective COTS management and coral reef conservation.

Our calibration models enable threshold-based management regardless of which monitoring tool collected the data. By translating observations into the CPUE units in which ecological thresholds are currently defined (Plagányi et al. 2020), managers can make consistent decisions across datasets from diverse sources. However, the wide prediction intervals for certain tool combinations indicate that some methods (e.g. manta tow) are less reliable for threshold detection than others (e.g. SALAD, eDNA). As routine operational data accumulates from parallel deployment of multiple tools, these data should be incorporated into updated calibration models to progressively reduce uncertainty, particularly at the ecologically critical low-to-moderate density range where early intervention is most effective.

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Author contributions EL and SF designed the field experiment. SF, DW, MP, JD, SB, PD and MB participated in field data collection. SU, MP, JD, SB, PD, BK and MAA provided curation and interpretation of individual monitoring tool data. SF and EL conducted statistical analysis. EL and SF wrote the initial manuscript draft. All authors revised and contributed to the manuscript.

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Data Availability Data is provided https://doi.org/10.25919/b5jn-vg48 and as part of the RosettaCOTS R package (R package).

Declarations

Conflict of interest The authors declare no competing interests.

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