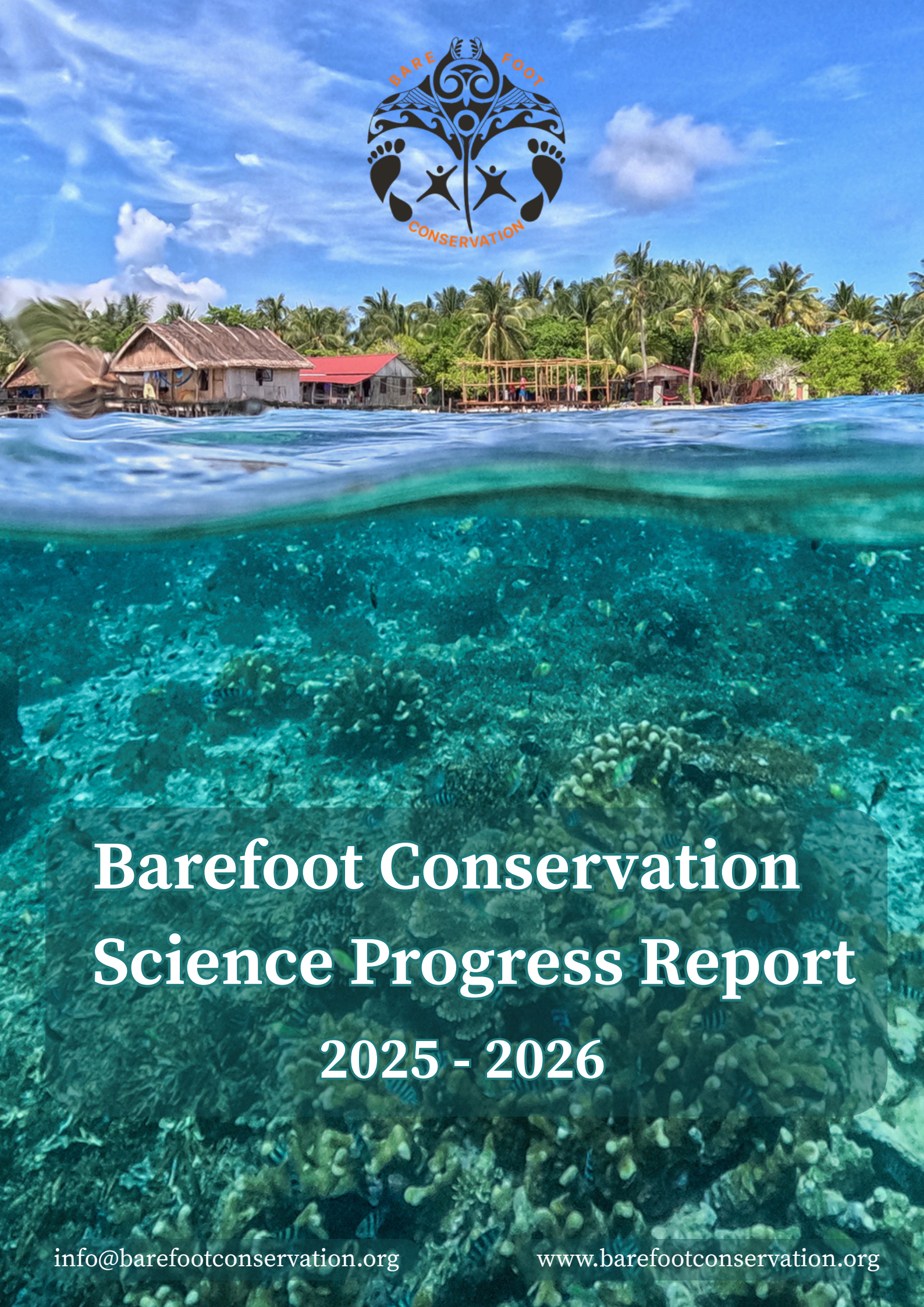


Barefoot Conservation Science Progress Report

2025 - 2026

info@barefootconservation.org

www.barefootconservation.org

Science Progress Report
2025 - 2026

For any questions regarding the data, findings or projects mentioned in this report, please contact our Head of Science via science@barefootconservation.org

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Introduction

Barefoot Conservation is an Indonesian registered NGO (Yayasan Konservasi Jejak Kaki Indonesia: AHU-0004531.AH.01.04. Tahun 2018) working to conserve the unrivalled marine life of Raja Ampat, through monitoring, research and science training of the local community. Barefoot has been running since 2016 and has been collecting data on reef health, manta populations, marine debris and crown of thorns starfish for several years, providing invaluable long-term datasets of the ecosystem over time. Our scientific output is constantly evolving and expanding and we now have 11 science projects running in parallel. This report provides an update on the progress of 7 of our major science projects in 2025. Our Science team at Barefoot currently consists of Septya Putri (Jenderal Soedirman University), Ilman Fari (Diponegoro University), Fahri Widiatmoko (IPB University), Natalie Petrovski (University of Queensland) and Head of Science Josie Chandler (James Cook University), who is working remotely from Australia. Other scientists who have been involved with the science at Barefoot during 2025 but are no longer working with us are: Mathilda Bates (James Cook University) and Nikolai Madland Shorter (University of Cambridge). Our Head of Operations (Iris Uijttewaal), Dive Manager (Matt Perrodou), and Divemasters (Hawe and Rama) are also heavily involved in the science projects. We also thank Erika Gress (James Cook University) for her involvement with Barefoot Science and dedication to the reefs of Raja Ampat during 2025.

In 2024, we initiated several new projects, most notably establishing a coral spawning project and expanding our coral bleaching monitoring methodologies. Building on this momentum, 2025 has seen the launch of additional projects and the development of new collaborations that we intend to strengthen further in 2026. These include hosting University of Papua (UNIPA) interns for six-month placements to complete their university research projects, an initiative we plan to continue for many years into the future.



We have also continued our long-term Reef Check monitoring, meeting quarterly monitoring targets and expanding our dataset through the addition of bonus sites. In parallel, we have contributed to ongoing data collection across a wide range of projects, including marine debris, manta rays, cyanobacteria, crown-of-thorns starfish, and others. With an emerging crown-of-thorns starfish (CoTS) outbreak, we have renewed focus on this research area and led an intensive culling programme, alongside revitalising associated research efforts. All of these projects are outlined in detail within this report, including their background, progress to date, and objectives for 2026.

One of the most significant initiatives this year has been our work to improve water quality in Raja Ampat. In collaboration with local and regional stakeholders, we have implemented a pilot wastewater treatment system, currently being trialled at two sites. If successful, this system has the potential to be rolled out more broadly across Raja Ampat in the coming year. We continue to work closely with other stakeholders to collect data and support informed reef management decisions in the region. More details of this project can be found within the Community Outreach section of the report.
All of the projects currently running at Barefoot Conservation collect observational data only. Correct permissions were sought from both Chief of Arborek Village, Bapa Juan, and Head of BLUD, Pak Syafri, before commencing any of the projects mentioned in this report.

AUTHORS
Josie Chandler, *Head of Science*
Report coordination and preparation

Natalie Petrovski, *Project Scientist*
Coral ecology, marine debris, community engagement, report preparation and design

Septya Putri, *Project Scientist*
Manta ray research, reef restoration and community engagement

Ilman Fari, *Science Officer*
Reef Check, Crown-of-Thorns starfish and community engagement

Fahri Widiatmoko, *Science Officer*
Cyanobacteria and community engagement



Where is the data going?

Reef Check
Data from our Reef Check project is shared with Reef Check Indonesia and Reef Check Worldwide. These data contribute to national and global reef health monitoring initiatives using standardised monitoring methodologies, providing insight into changes in reef health across multiple locations worldwide. We contribute valuable long-term monitoring data on coral reef condition in Raja Ampat.

Coral Bleaching Project
Data collected through our coral bleaching project are shared with the University of Queensland, CoralWatch (ALA's Biocollect CoralWatch Data Portal) and NOAA Coral Reef Watch. These data contribute to projects that improve our understanding of global coral bleaching patterns and taxa-specific bleaching responses.

Photogrammetry & Temperature Monitoring Project
Our photogrammetry and temperature monitoring project is a collaborative research project with researchers from NOAA Coral Reef Watch and Atlantic Oceanographic and Meteorological Laboratory (AOML). Data collected through this project is shared with these research teams and contribute to the development and validation of coral bleaching prediction models, while also supporting broader research into coral bleaching.

Coral Spawning Project
Data collected through our coral spawning project are contributed to CORALog (COsmaRindo datA Log), the data platform of COSMARINDO (Coral Spawning Monitoring and Awareness for Restoration Network Indonesia). These data contribute to a growing open-access database that improves understanding of coral spawning patterns across Indonesia and supports coral reef conservation, management and restoration initiatives.

Cyanobacteria Project
Data collected through our cyanobacteria project has been presented to BLUD UPTD (the Raja Ampat Marine Park Authority and Management Unit) in a formal report and shared with local stakeholders. The findings contribute to ongoing discussions regarding wasterwater mangement in Raja Ampat and support efforts to identify effective solutions to this growing environmental challenge.

Marine Debris Project
Data collected through our marine debris project is shared with BLUD UPTD and other environmental organisations across Indonesia to support local management actions, inform decision-making and encourage collaborative solutions to marine waste challenges in Raja Ampat and the wider region.

Manta Project
Data collected through our long-term manta ray monitoring project contributes to our growing photo-identification database and is also shared with researchers from the Elasmobranch Institute Indonesia. These data complement ongoing manta ray research in Raja Ampat and support regional conservation and management efforts.

Crown-of-Thorns Starfish Project
Data collected through our Crown-of-Thorns Starfish (COTS) project is shared with BLUD UPTD and uploaded to the Raja Ampat COTS Portal managed by SeaPeople. These data support regional monitoring efforts and provide an early warning of increasing COTS abundance and potential outbreaks in Raja Ampat.

BLUD UPTD Pengelolaan Kawasan Konservasi di Perairan Kepulauan Raja Ampat





Coral Ecology Project Report

Introduction

The fourth global coral bleaching event, from January 2023 to September 2025, is the most widespread coral bleaching event recorded to date, impacting an estimated 84% of the world's coral reef area across at least 83 countries (NOAA, 2025; Image 1). This event has been driven by a combination of prolonged anomalies in ocean temperatures attributed by broader climate patterns, including El Niño conditions, and long-term anthropogenic warming (Santer et al. 2007). The global annual mean sea surface temperature (SST) in 2025 ranked among the three warmest years on record, with 2023 - 2025 representing the warmest three-year period across multiple global temperature records (WMO, 2025). Despite 2025 beginning and ending under cooling La Niña conditions, the ongoing accumulation of heat-trapping greenhouse gases in the atmosphere contributed to these elevated temperatures. As bleaching events become more frequent and severe due to increases in SSTs, and recovery intervals between thermal stress events continue to shorten, the risk of long-term reef degradation increases (Baker et al. 2008). These impacts extend beyond ecological consequences, affecting coastal economies, food security, and the livelihoods of communities that depend on healthy reef systems worldwide.

Raja Ampat lies at the heart of the Coral Triangle and is widely recognised as one of the most biodiverse coral reef systems globally (Turak et al. 2003). Reefs within the Dampier Strait have historically been considered a thermal refugium (Dixon et al. 2022), however, they have not been immune to the impacts of recent marine heatwaves (De Clippele et al. 2023). Following a mild bleaching event in 2023, elevated sea surface temperatures associated with the ongoing global bleaching episode led to more severe bleaching across parts of the Raja Ampat reef system during late 2024 and early 2025. This represents one of the most significant bleaching events documented in the region to date and highlights the need to better understand both local vulnerability and the capacity for reef recovery.

In response, systematic and long-term monitoring is essential to quantify bleaching impacts, identify drivers of vulnerability and resilience, and inform effective management responses. Our coral ecology project has remained one of our highest research priorities, designed to capture bleaching dynamics across multiple spatial and temporal scales, integrating colony-level and reef-scale observations.

Throughout the bleaching event, NOAA Coral Reef Watch Bleaching Alert Levels consistently remained high for the Dampier Strait (NOAA Coral Reef Watch, 2019). In response, we prioritised field-based data collection to document bleaching severity, spatial extent and species-level responses. The core bleaching monitoring sub-project was maintained to track coral condition and bleaching progression through time. This was complemented by continued and expanded in-situ temperature monitoring to better characterise fine-scale thermal variability, including short-term fluctuations and potential upwelling events, to assess their relationship with observed bleaching patterns. The photogrammetry sub-project has also progressed to provide spatial context for reef-scale impacts, while the coral spawning sub-project was initiated to improve understanding of reproductive timing and its implications for post-bleaching recovery.

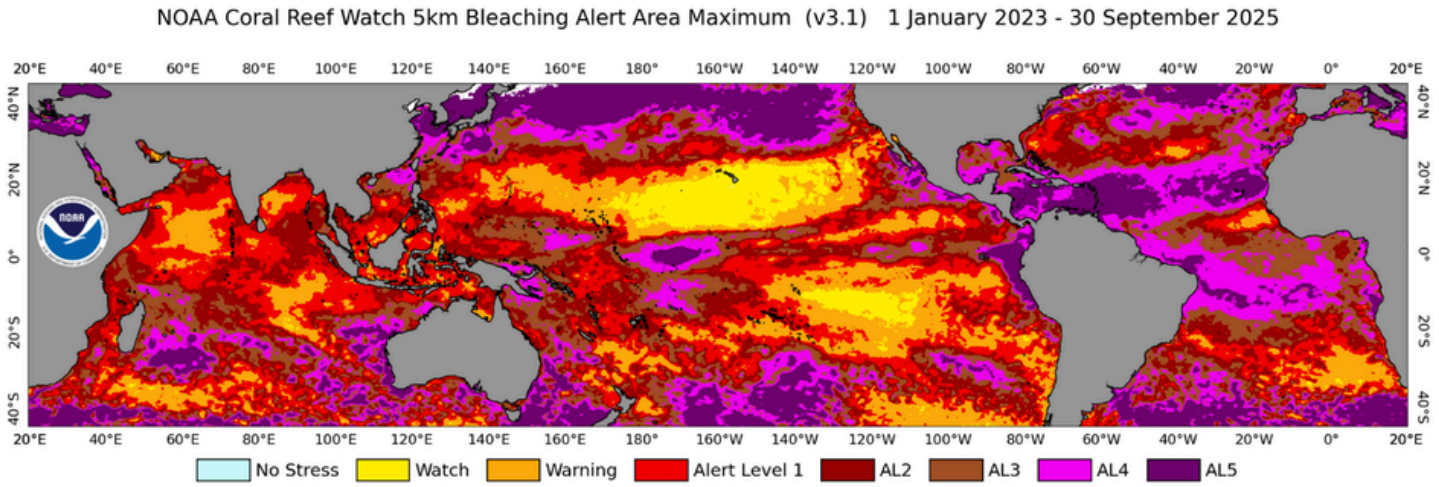
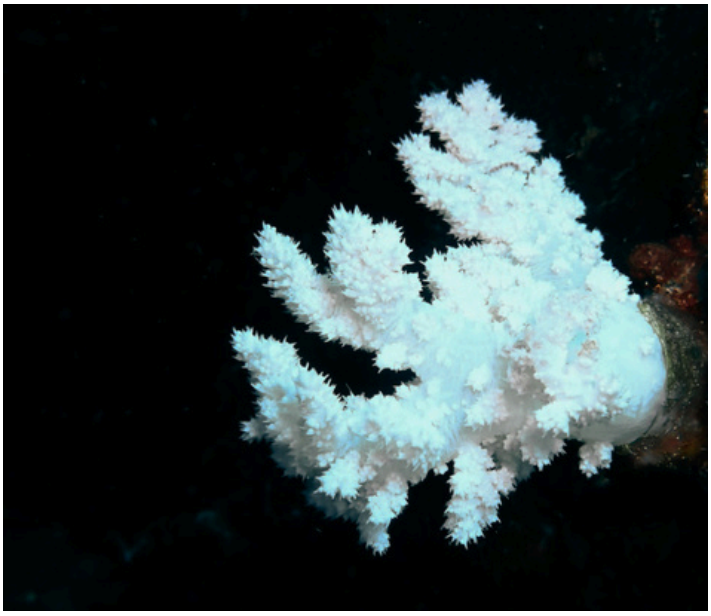


Figure 1. *NOAA's Coral Reef Watch Alert Area for the period January 1, 2023 - September 30, 2025. This map displays the maximum accumulated heat stress experienced by coral reefs during the fourth global mass bleaching event.*

Although Raja Ampat has remained largely unaffected by large-scale bleaching until now, both the magnitude of the impacts and the success of recovery remain uncertain. The region's exceptionally high coral cover and species diversity provide a strong foundation for recovery. However, outcomes will depend on the frequency of future thermal stress events and the influence of local stressors (such as pollution, wastewater and unmanaged tourism). Ongoing analyses aim to quantify the extent of bleaching impacts, identify taxa that are most vulnerable or resilient to thermal stress, and determine recovery trajectories over time. Understanding how global climate stress interacts with local environmental pressures will be important for evaluating long-term resilience of reefs in Raja Ampat under continued climate change.



Bleaching Monitoring

INTRODUCTION

The coral bleaching monitoring project has been ongoing for two years, commencing in November 2023 in response to the first documented bleaching event in Raja Ampat. Since its establishment, the project has generated a consistent long-term temporal dataset. Bleaching data have been collected across multiple coral genera and morphologies along defined depth gradients. By repeatedly surveying the same permanently tagged colonies, this approach allows for accurate documentation of bleaching onset, duration, severity, and subsequent recovery at the colony level. This monitoring program provides important insights into how local coral communities respond to thermal stress and contributes to ongoing efforts to understand reef resilience in Raja Ampat.

METHODS

Bleaching surveys were conducted at the JPP dive site, a seamount located south of Arborek Island (0°34'02.6" S, 130°31'18.6" E). This site has been monitored consistently throughout the project and was selected due to its accessibility, limited diver traffic, and suitability for long-term monitoring. Surveys were conducted weekly, with each depth surveyed fortnightly under normal conditions and increased to weekly during periods of elevated bleaching alert. Monitoring was undertaken along permanent 20m transects established at the three depths: 6m (seamount summit), 8m (upper slope), and 16m (lower slope). Typically, only one depth was completed per dive.

Each transect was laid between two permanent stakes to ensure that the same section of reef was monitored during each survey. Prior to survey commencement, metadata including time, weather conditions, and temperature were recorded. Bleaching surveys comprised two components: (1) photo-quadrat surveys and (2) tagged colony assessments.

For the photo-quadrat component, a diver positioned a quadrat rig at the 0m transect mark, ensuring the transect line ran through the centre of the frame. A second diver captured photo-quadrats at 1m intervals along the transect using an Olympus TG-7 camera, shooting in RAW format with manual white balance adjusted for each depth. A CoralWatch Health Chart was affixed to the top-left corner of the rig and included in every image as a standardised bleaching severity scale. The chart consists of four colour hues across a six-point brightness/saturation scale corresponding to symbiont density within coral tissue.

The second survey component focused on 20 tagged coral colonies located adjacent to each transect (60 colonies total across three depths). Colonies were marked using numbered cattle tags, with tag numbers cross-referenced against morphology records to ensure accurate identification. During the 2025 survey period, some colonies experienced breakage or mortality, these were replaced with new colonies and assigned new identification numbers. Each tagged colony was photographed with the CoralWatch Health Chart positioned alongside the coral.

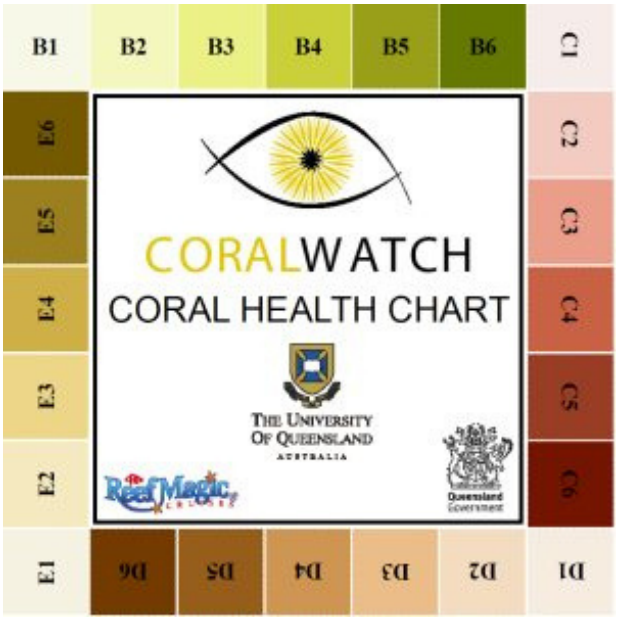


Figure 2. *CoralWatch Coral Health Chart used to assess coral condition and bleaching severity*

Colonies were assigned CoralWatch scores by recording the lightest and darkest colour observed on the colony (Fig. 2). Colonies with a colour score of 1 were classified as bleached, while scores ≥ 2 were classified as not bleached. This threshold was used to minimise misclassification of naturally pale colonies as bleached and to provide a conservative estimate of bleaching prevalence. In addition, a bleaching severity score was assigned to provide a more detailed assessment of colony condition and allow bleaching intensity and bleaching-related mortality to be assessed independently of CoralWatch scores. Severity scores ranged from 0 to 7 following a scale adapted from the Australian Institute of Marine Science (AIMS): 0 = healthy, 1 = pale (<5% bleaching), 2 = partially bleached (5 – 50%), 3 = partially bleached (51 – 90%), 4 = severe bleaching (>90%), 5 = minor mortality (5 – 50%), 6 = major mortality (51 – 90%), 7 = full mortality. Mortality scores represent recently dead tissue associated with bleaching. Colonies could therefore receive mortality scores across multiple surveys if recently dead tissue remained visible or if tissue loss expanded. Once dead areas established algal overgrowth and were no longer considered recent mortality, only the remaining live tissue was assessed. Any additional observations, including signs of physical damage or other potential causes of mortality were also recorded.

Following each survey, CoralWatch scores were uploaded to the Atlas of Living Australia (ALA) Biocollect CoralWatch Data Portal, while bleaching severity scores were submitted to NOAA Coral Reef Watch via the coral bleaching data and observations reporting system.

RESULTS

CoralWatch scores

A total of 73 unique colonies were monitored during the 2025 survey period, reflecting replacement of colonies that were lost or died throughout the year. Bleaching (CoralWatch score = 1) was observed across all depths (6m, 8m and 16m) and varied throughout the year (Fig. 3). Bleaching was most common during the first half of the year, with relatively high proportions of colonies affected between January and June. Bleaching increased again toward the end of the year, with December recording one of the highest monthly values. The highest levels were recorded in May and December (both 15.3%), with similar elevated bleaching in April (14.7%) and January (14.6%). In contrast, bleaching declined during the middle of the year, reaching its lowest levels between July and October (approximately 7 – 8% of colonies).

Depth influenced the strength of these patterns. Colonies at 8m consistently showed the highest levels of bleaching and the greatest temporal variability, with several surveys exceeding 20% of colonies affected. In comparison, colonies at 6m showed lower and more stable bleaching levels, while colonies at 16m followed a similar temporal pattern but generally remained below the levels observed at mid-depth. These patterns indicate that bleaching occurred across the full depth gradient but was most pronounced at mid-depth.

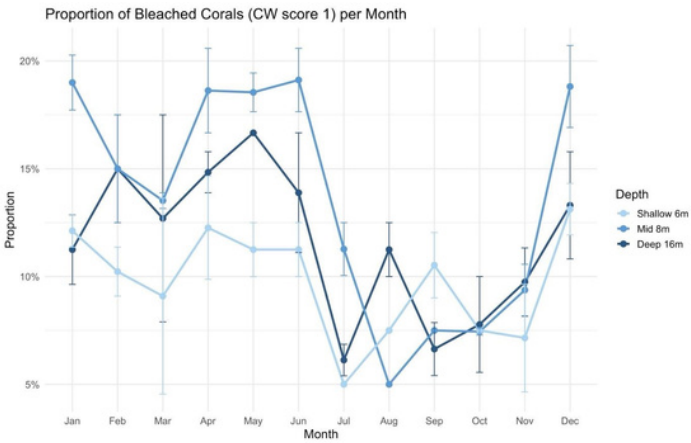


Figure 3. *Monthly proportion of bleached coral colonies at three reef depths (6m, 8m, 16m) at the JPP monitoring site during 2025, based on CoralWatch colour scores (score = 1). Points represent the mean bleaching proportion per month and depth calculated from survey-level observations, with lines connecting monthly means. Error bars indicate variability among surveys within each month.*

Growth form and depth

Bleaching susceptibility varied strongly among coral growth forms and across depths (AIC = 728.7; Fig. 4). Branching corals were the most affected, particularly at 8m where bleaching was significantly higher than at other depths ($z = 3.76$, $p < 0.001$). Massive and plate corals showed lower bleaching overall, especially at 16m, but experienced increased bleaching at shallower depths (6m and 8m; $p < 0.01$). Encrusting corals showed consistently low levels of bleaching, with reduced susceptibility at both 6m and 8m compared to deeper colonies ($p < 0.001$). Overall, these results indicate that bleaching patterns were primarily driven by coral morphology, with depth playing a secondary but important role. Soft corals were excluded from this analysis due to low sample sizes and limited bleaching observations.

Bleaching and mortality severity scores

Across all observations, most coral colonies remained healthy throughout the year (Fig. 5). Healthy colonies (score 0) accounted for the majority of observations (54 - 69% across depths), while a smaller proportion showed paling (score 1; approximately 8 – 11%). Moderate to severe bleaching (scores 2 – 4) was observed in 14% - 24% of cases and bleaching related mortality (scores 5 – 7) accounted for 10 – 13% of observations. Across the 73 tagged colonies, 44 colonies (60.3%) exhibited bleaching at least once, while 34 colonies (46.6%) experienced some level of bleaching-related mortality. Complete colony mortality (score 7) was recorded in 6 colonies (8.2%).

Clear differences were observed across depths. Colonies at mid-depth (8m) experienced the highest levels of bleaching (23.7% of observations), followed by 16m (20.4%), while shallow colonies at 6m showed the lowest bleaching levels (13.9%) and the highest proportion of healthy observations (~68.5%). In contrast, bleaching related mortality occurred at similar levels across all depths (approximately 10 – 13% of observations). As mortality scores represent recently dead tissue, repeated observations may include the same colony across multiple surveys while mortality remained recent.

Bleaching also showed clear temporal patterns. The highest levels of bleaching were recorded early in the year (January – March), particularly at mid-depth (8m) where up to 47% of observed colonies were affected in January. Deep colonies (16m) also experienced elevated bleaching during this period (~31%). From April to June, bleaching declined across all depths, while mortality increased, indicating delayed tissue loss following earlier bleaching. Mortality peaked during this period, particularly at 16m where it reached 30.6% of observations in June.

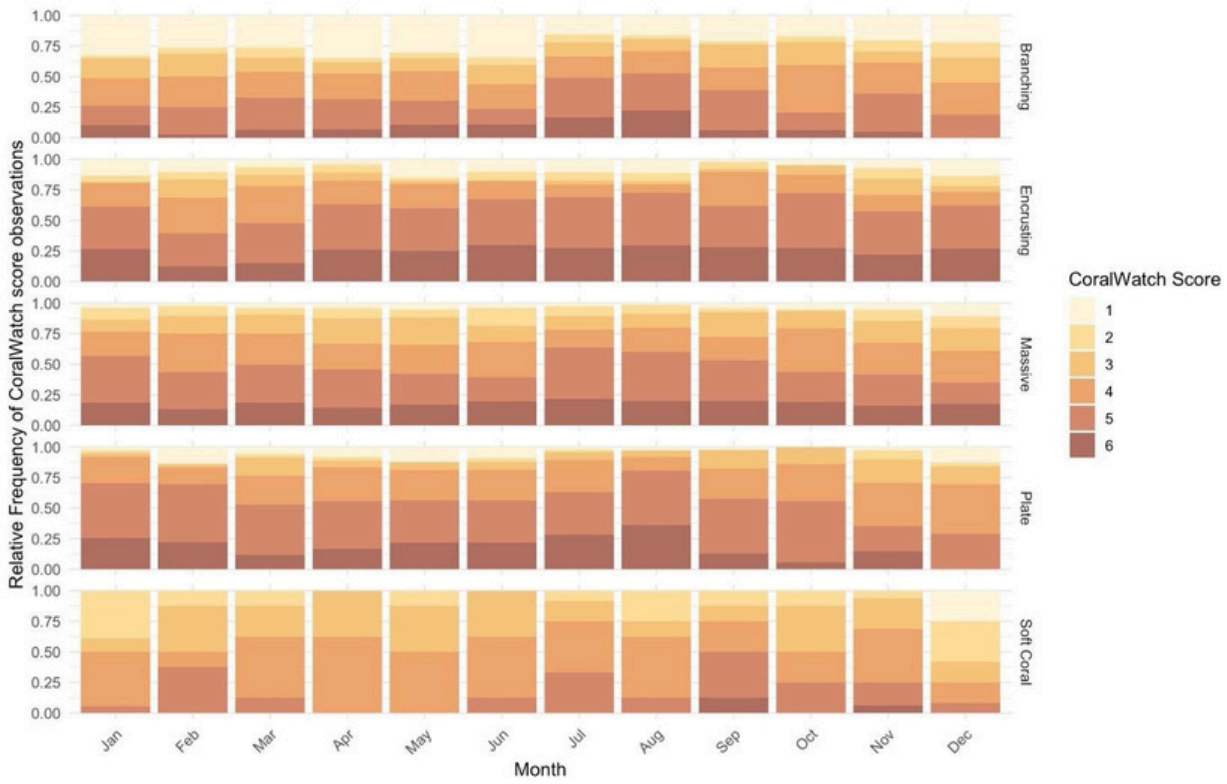


Figure 4. *Temporal variation in CoralWatch colour scores recorded from tagged coral colonies at the JPP monitoring site during 2025. The relative frequency of colour scores is shown by month and separated by coral growth form. Lower colour scores indicate paler colonies and increased bleaching, while higher scores indicate healthier pigmentation.*

During the second half of the year (July – December), the proportion of healthy colonies increased across all depths, although low to moderate bleaching persisted. An increased in paling towards the end of the year (November – December), particularly at 8m and 16m, suggests that some colonies were beginning to recover following earlier bleaching.

Overall, while bleaching and mortality were observed across the tagged population, most colonies remained healthy. The greatest impacts were concentrated at mid-depth colonies (8m), while shallow colonies maintained the highest proportion of healthy observations.

Growth form and depth

Bleaching severity varied among coral growth forms and across depths (Fig. 6). Branching corals were the most affected, showing the highest bleaching severity, particularly at 8m where severity was significantly greater than at 16m ($p = 0.027$). In contrast, massive and plate coral experienced significantly lower bleaching severity than branching corals ($p < 0.007$), while encrusting corals showed consistently lower bleaching severity, with reduced bleaching at both 8 m and 6m relative to 16m ($p < 0.001$).

Patterns of bleaching-related mortality closely mirrored these trends. Mortality was highest in branching corals at 8m ($z = 4.32$, $p < 0.001$), while massive, plate and encrusting corals experienced lower mortality at this depth ($p < 0.007$). No clear differences in mortality among growth forms were observed at 6m ($p > 0.05$).

Overall, these results indicate that the most severe bleaching and mortality were concentrated in branching corals at mid-depth, while other growth forms were comparatively more resilient across the depth gradient.

DISCUSSION

Severe bleaching was observed on local reefs in late 2024, with bleaching levels increasing from August and reaching peak intensity in December 2024 and January 2025. Surveys conducted throughout 2025 therefore captured the later stages of this bleaching event and the subsequent recovery period. At the beginning of the year, bleaching remained evident across all depths, indicating that many colonies were still affected by the thermal stress experienced during late 2024.

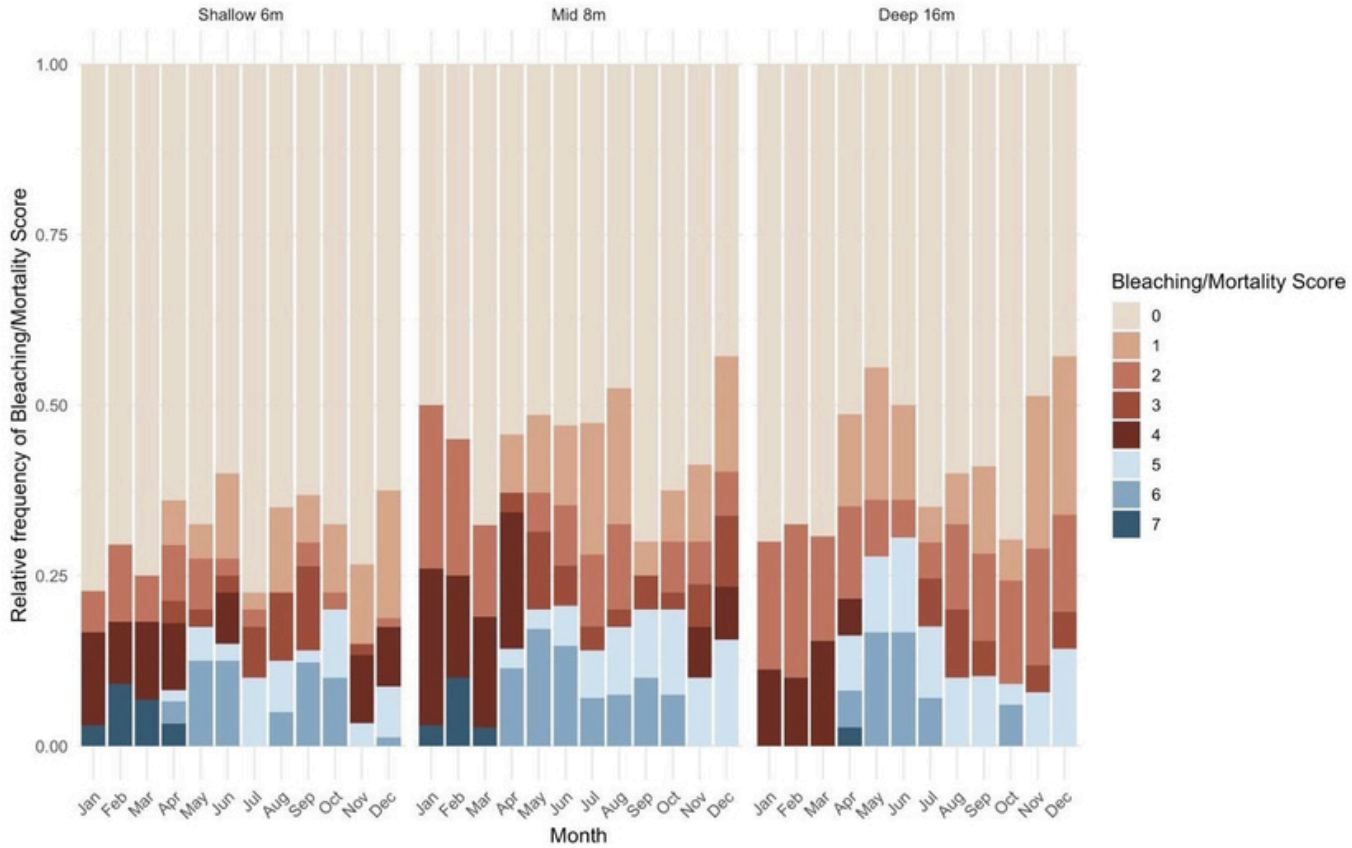


***Figure 5**. Temporal variation in bleaching and mortality severity scores recorded from the tagged coral colonies at the JPP monitoring site during 2025. The relative frequency of severity categories is shown by month and separated by depth. Bleaching scores (0 – 4) are shown in red tones, while mortality scores (5 – 7) are shown in blue, representing increasing bleaching severity and bleaching-related tissue loss, from healthy colonies (0) to complete colony mortality (7).*

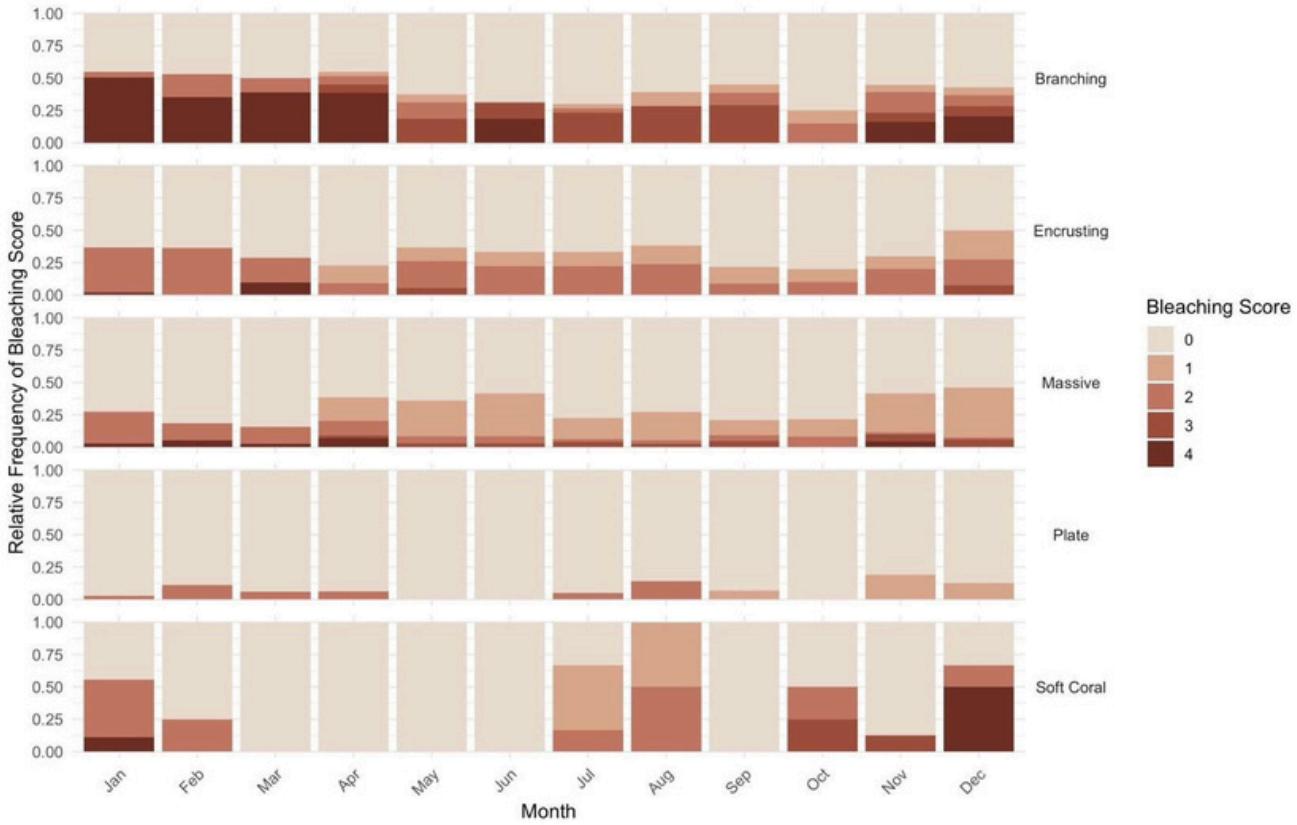


Figure 6. *Temporal variation in bleaching severity scores recorded from tagged coral colonies at the JPP monitoring site during 2025. The relative frequency of bleaching categories (0-4) is shown by month and separated by coral growth form. Increasing scores indicate greater bleaching severity, from healthy colonies (0) to severe bleaching (score 4 = >90% of the colony affected).*

Bleaching patterns observed using the CoralWatch scores were broadly consistent with those identified using the AIMS bleaching and mortality severity scores. CoralWatch scores captured changes in bleaching prevalence over time, indicating when colonies were visibly pale or bleached, while the bleaching severity scores provided greater detail on bleaching intensity and associated mortality at the colony level. Together, these complementary metrics show that while bleaching was relatively common during the early part of the year, severe bleaching and colony mortality were limited to a smaller subset of colonies. Using both scoring approaches therefore provided a more comprehensive understanding of bleaching dynamics and recovery following the late-2024 bleaching event.

Bleaching prevalence was highest during the early months of 2025, particularly between January and March, and gradually declined throughout the year. This pattern is consistent with bleaching dynamics observed on many reefs, where corals remain visibly bleached for several months after peak thermal stress has passed. As the year progressed, the proportion of healthy colonies increased across depths, suggesting that many corals were able to recover their symbiotic algae following the initial bleaching event.

Patterns of recently dead tissue indicated that some colonies experienced delayed mortality following bleaching, with mortality observations highest during the middle of the year. In this monitoring program, mortality scores represent recently dead tissue, meaning older dead areas are no longer recorded. The decline in mortality observations later in the year therefore likely reflects both reduced bleaching stress and the absence of new or expanded mortality, rather than a complete recovery of previously affected corals.

Bleaching susceptibility varied strongly among coral growth forms. Branching corals were consistently the most affected, exhibiting higher bleaching prevalence, greater bleaching severity and higher mortality compared to other morphologies. In contrast, massive, plate and encrusting corals generally experienced lower bleaching severity and mortality rates. These patterns are consistent with observation from reefs worldwide, where fast-growing branching corals such as *Acropora* are often among the most sensitive to thermal stress, while slower-growing massive corals tend to be more resistant (Burn et al. 2023; Yadav et al. 2023).

Depth-related differences were also observed in bleaching prevalence and severity. Colonies at mid-depth (8m) experienced the highest levels of bleaching and mortality, while shallow colonies (6m) generally maintained the highest proportion of healthy observations throughout the monitoring period. This differs from the commonly observed trend where shallow corals experience greater bleaching due to higher light exposure and heat stress, while deeper colonies may be partially sheltered by reduced light levels and cooler water (Baird et al. 2018). The patterns observed at JPP may instead reflect local environmental conditions associated with the seamount, including differences in water movement, current exposure and light intensity across the reef slope, which can influence how heat stress is experienced by corals.

Although the JPP monitoring site experienced notable bleaching, observations across the wider Dampier Strait indicate that some reefs were more severely affected, particularly on the eastern side of the strait where bleaching occurred across multiple coral morphologies and at greater depths. Local differences in ocean currents, heat stress exposure and reef structure likely contributed to the spatial variability. Overall, the results from 2025 indicate that while the late 2024 bleaching event affected a portion of the coral community at JPP, many colonies remained healthy and bleaching impacts declined throughout the year, suggesting that this reef retains a degree of resilience to thermal stress.

Continued long-term monitoring will be important to understand the longer-term consequences of this bleaching event and the capacity of local reefs to recover. In the coming year, the Barefoot Science team will continue regular bleaching surveys at JPP, alongside environmental monitoring using temperature loggers. Integrating long-term temperature records with bleaching observations will help improve our understanding of how thermal stress, depth, and local oceanographic conditions influence coral bleaching in Raja Ampat. These efforts will contribute valuable data to regional and global monitoring efforts and support ongoing research into the resilience of coral reefs in this globally important biodiversity hotspot.

Temperature Loggers

INTRODUCTION

To support interpretation of bleaching patterns and better characterise thermal stress in the region, the temperature logger sub-project forms a critical component of our coral ecology research. Elevated SSTs are the primary driver of coral bleaching (van Oppen & Lough, 2018), with bleaching generally occurring approximately 1 - 2 °C above long term summer maxima (Veron et al. 2009). Continuous in situ temperature monitoring therefore provides essential context for understanding when and how bleaching events occur (Abdo et al. 2012; Castillo & Lima, 2010). Water temperature data collected from permanently deployed loggers enable the identification of seasonal, monthly, and short-term (daily) temperature fluctuations, as well as the duration and intensity of anomalous warming events. Such fine scale measurements are particularly valuable for assessing thermal exposure experienced directly by reef-building corals, which may differ from broader satellite-derived SST products.

This information is especially important in Raja Ampat, a region that has historically been considered a thermal refugia within the Coral Triangle (Dixon et al. 2022). However, recent projections indicate that the persistence of such refugia is likely to decline under continued warming. De Clippele et al. (2023) show that several Raja Ampat sites are identified as among the earliest reefs expected to experience Annual Severe Bleaching (ABS), making it one of the more climate-vulnerable regions in Indonesia. At the same time, substantial variation exists within the region, with some reef areas projected to experience severe bleaching much later than others. This spatial variability highlights the importance of fine-scale thermal monitoring. Long-term in situ temperature datasets in Raja Ampat remain limited, and the permanent deployment of temperature loggers therefore helps address a critical data gap and improve our ability to link observed bleaching responses to local thermal conditions.

Temperature loggers were first deployed at the Barefoot Jetty house reef, Arborek Island, in June 2024. This site serves as a long-term reference location for ongoing monitoring. A second set of loggers were subsequently deployed at Yenbuba dive site, where comparatively high levels of bleaching were observed during the 2024/2025 bleaching event. Monitoring temperature at these sites allows us to track thermal conditions associated with severe bleaching impacts and to compare temperature profiles between sites with differing bleaching responses. By comparing

temperature data across these locations, we can assess whether observed differences in bleaching severity are associated with variations in thermal exposure and evaluate the potential for early warning indicators of future bleaching events.

METHODS

Four Onset HOBO Water Temperature Pro v2 loggers were deployed at Barefoot Jetty (00°33'54.0"S, 130°31'11.8"E) at depths of 1m, 5m, 15m and 25m. All loggers were programmed to record temperate at 5-minute intervals. Installed in 2024 and maintained throughout 2025, these loggers provide a continuous temperature record across four depths for the entirety of 2025. A second set of temperature loggers were deployed at Yenbuba (0° 34'9.948"S, 130°39'29.268"E) at the same depths and logging interval. While these loggers were intended to provide a full depth profile for 2025, the 15m and 25m loggers were removed or stolen early in the year. As a result, data from the 25m logger are unavailable and data from the 15m logger are limited to less than two months and not presented in this report. Consequently, only temperature data from the 1m and 5m loggers at Yenbuba are reported for 2025, both of which provide continuous records for the full year.

All loggers were secured to metal stakes using chain and wire to minimise movement and loss. Anti-biofouling measures, including electrical tape and fabric coverings were applied to protect the sensors and communication windows. Data were downloaded multiple times throughout the year to ensure data integrity and continuity (Fig. 7).

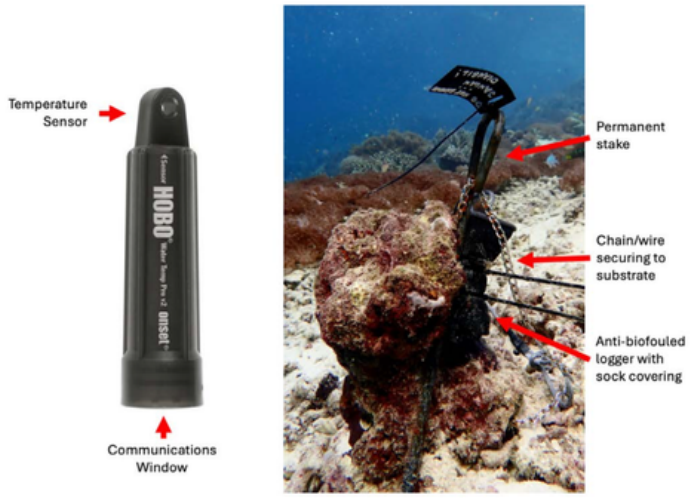


Figure 7. *Onset HOBO Water Temperature Pro v2 data logger used for reef temperature monitoring. Annotated images show logger components, attachment hardware and anti-biofouling materials*

To quantify thermal stress and enable comparison with established bleaching thresholds, in situ temperature data were analysed using the NOAA Coral Reef Watch (CRW) methodology (Strong et al. 2006). The Maximum Monthly Mean (MMM; 29.11 °C) was obtained from the CRW 5-km climatology for West Papua. Daily mean temperatures were calculated from logger data using 24-hour means. Hotspots were defined when daily mean temperatures exceeded the bleaching threshold (MMM + 1°C), and Degree Heating Weeks (DHW) were calculated as the cumulative Hotspot values ≥ 1 °C over a rolling 12-week (84-day) period (Liu et al. 2003).

RESULTS

Barefoot Jetty

Temperatures recorded by the four loggers at BFJ (1m, 5m, 15 and 25m) showed broadly similar temporal patterns throughout 2025, indicating consistent thermal conditions across depths. All loggers followed the same seasonal trend, with elevated temperatures in April – May, cooler conditions from June to September, and a secondary increase in October – November before declining again in December (Fig. 8). In situ temperatures generally ranged between 27.5 – 30.9 °C at 1m, 27.6 - 30.87 °C at 5m, 28.4 - 30.75 °C at 15m and 28.4 – 30.7 °C at 25m.

Thermal Hotspots (daily mean temperature exceeding the Maximum Monthly Mean, MMM) occurred primarily in April, May, October, and November across all depths, with May consistently showing the greatest number of Hotspot days. No Hotspot days were recorded from January to March, June to September, or in December (Table 1).

Month (2025)	***1m***	***5m***	***15m***	***25m***
Apr	12	13	13	13
May	19	20	20	20
Oct	16	18	14	12
Nov	3	6	5	3

Table 1: *Number of HotSpot days per month recorded by in situ temperature loggers at Barefoot Jetty in 2025 across four depths. Only months with HotSpot days are presented.*

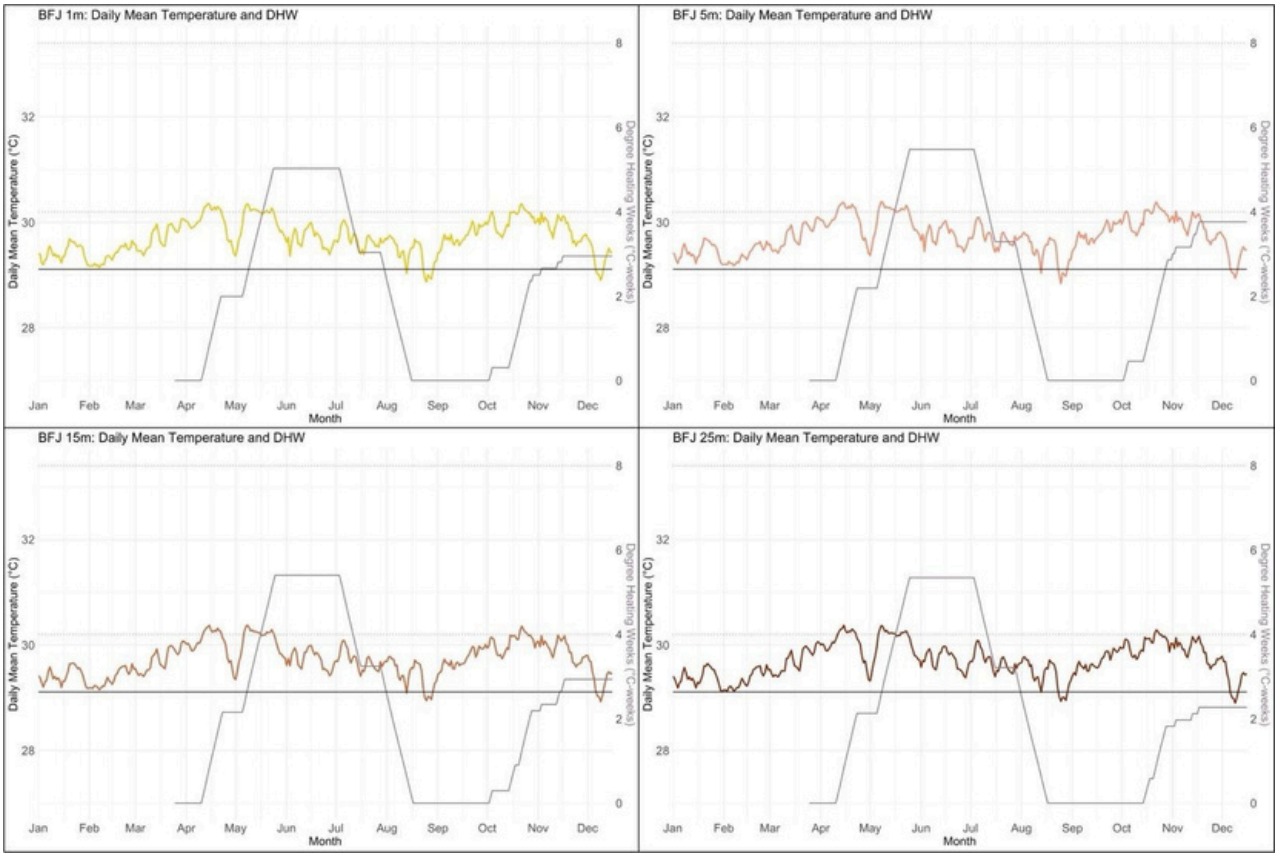


Figure 8. *Daily Mean Temperatures recorded by in situ temperature loggers at Barefoot Jetty across fourth depths (1m, 5m, 15m and 25m). The grey line represents Degree Heating Weeks (DHW), a cumulative measure of thermal stress over time.*

Cumulative thermal stress (Degree Heating Weeks, DHW) peaked in late May at all depths, exceeding the 4 °C-weeks bleaching stress threshold but remaining below 8 °C-weeks severe bleaching threshold. Peak DHW ranged from 5.03 °C-weeks at 1m to 5.48 °C-weeks at 5m, indicating similar levels of sustained thermal stress across the depth gradient (Table 2).

Depth	*Total Hotspot days*	*Max daily Hotspot (°C)*	*Peak DHW (°C-weeks)*	*Date of peak DHW*
1m	50	1.25	5.03	24 May 2025
5m	57	1.29	5.48	25 May 2025
15m	52	1.27	5.4	25 May 2025
25m	48	1.27	5.35	25 May 2025

Table 2: *Summary of thermal stress recorded by in situ temperature loggers at Barefoot Jetty in 2025 across four depths.*

Yenbuba

At Yenbuba, both temperature loggers (1m and 5m) recorded substantially greater thermal stress than at BFJ. In situ temperatures showed broadly similar temporal patterns between the two depths, although the 1m logger consistently experienced slightly warmer conditions than the 5m logger. Across 2025, temperatures at 1m generally ranged from approximately 27.75 – 32.6 °C, while temperatures at 5m ranged from approximately 27.8 – 31.4 °C. At both depths, temperatures were elevated from March through May, cooler from June to September, and showed a secondary warming phase in October to November before declining in December (Fig. 9). Despite these similar seasonal trends, the 1m logger was exposed to greater accumulated thermal stress than the 5m logger, with peak DHW values occurring in late May at both depths.

Hotspot exposure was not evenly distributed throughout the year but was concentrated primarily in two periods: March - July and October – November (Table 3). The greatest number of Hotspot days occurred in April, May and October at both depths. Overall, the two depths exhibited very similar temporal patterns in temperature and thermal stress, however the magnitude of stress was consistently great at 1m than at 5m (Table 4).

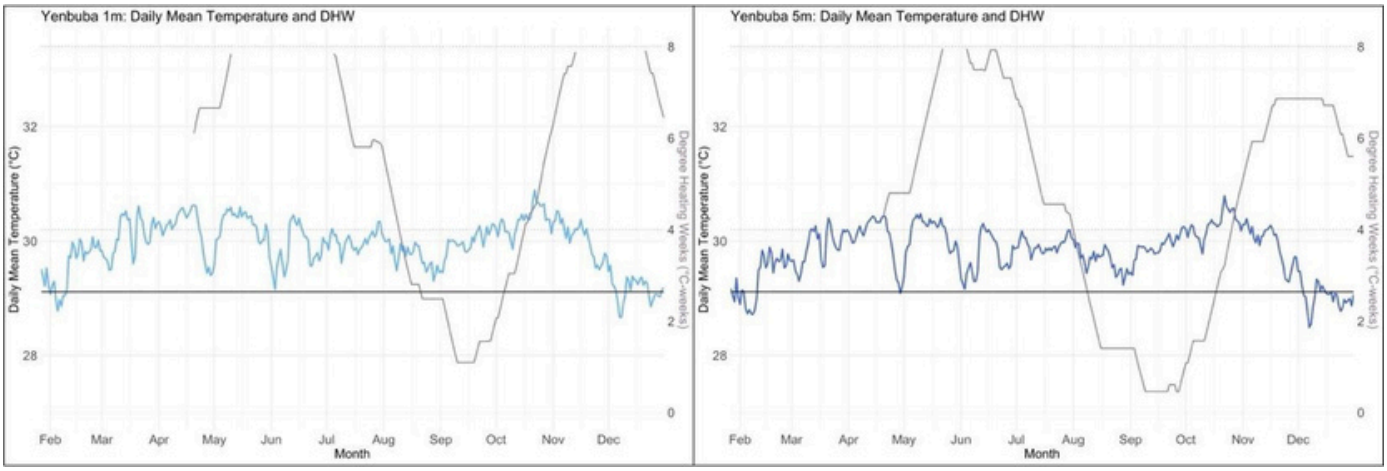


Figure 9. *Daily Mean Temperatures recorded by in situ temperature loggers at Yenbuba across two depths (1m and 5m). The grey line represents Degree Heating Weeks (DHW), a cumulative measure of thermal stress over time.*

DHW accumulated through March and early April, briefly declined in early May, then increased again to a maximum at the end of May before steadily decreasing through June – September. A secondary, smaller DHW peak occurred in October – November, after which values decreased again in December.

Month	***1m***	***5m***
Mar	14	11
Apr	22	18
May	22	20
Jun	8	6
Jul	7	3
Sep	7	4
Oct	27	24
Nov	15	12

Table 3. *Number of Hotspot days per month recorded by in situ temperature loggers at Yenbuba in 2025 across two depths. Only months with Hotspot days are presented.*

DISCUSSION

Temperature records from Barefoot Jetty (BFJ) and Yenbuba showed broadly similar seasonal patterns in 2025, indicating that both sites were influenced by the same regional climatic and oceanographic conditions. At both locations, temperatures were elevated from March to May, decreased from June to September and increased again in October – November before declining again in December.

Despite these similar temporal trends, the magnitude and duration of thermal stress differed substantially between sites. BFJ experienced comparatively moderate thermal stress. Peak cumulative thermal stress (DHW) occurred in late May across all depths but remained below 8 °C-weeks. Although conditions exceeded the 4 °C-weeks bleaching threshold, they did not reach levels associated with widespread severe bleaching (Van Hooidonk et al. 2014). Temperature ranges at BFJ were relatively consistent across depths, with all four loggers displaying similar patterns, suggesting uniform thermal exposure throughout the water column at this site.

In contrast, Yenbuba experienced more intense and prolonged thermal stress, particularly at 1m depth. Intermittent hotspot days during June, July and September, indicating sustained mid-year thermal exposure. Peak DHW values exceeded 10 °C-week at 1m, corresponding to conditions associated with a high likelihood of mass bleaching (Van Hooidonk et al. 2014). Although lower at 5m, thermal stress at this depth also reached levels linked to bleaching risk. Thermal stress at Yenbuba occurred in multiple phases throughout the year, with DHW accumulating in March – early April, peaking again in late May, and showing a smaller secondary increase in October – November. These patterns suggests that shallow reef zones at Yenbuba may be more thermally vulnerable, potentially due to great solar heating and reduced water mixing associated with weaker current exposure. Redeploying of deeper loggers in 2026 may help determine whether elevated stress is restricted to shallow habitats or reflects site-level vulnerability.

The observed differences between sites highlights the importance of fine scale, in situ temperature monitoring. Even within the same region, reefs can experience different thermal exposure profiles, which may help explain variation in bleaching severity and

Depth	Total Hotspot days	Max daily Hotspot (°C)	Peak DHW (°C-weeks)	Date of peak DHW
1m	122	1.78	10.8	30 May 2025
5m	98	1.68	8.26	24 May 2025

Table 4. *Summary of thermal stress recorded by in situ temperature loggers at Yenbuba in 2025 across two depths*

recovery patterns observed during field surveys. Continued multi-year monitoring will be essential to determine whether the 2025 patterns reflect consistent site-level differences or interannual climate variability. Linking these temperature records with ongoing bleaching assessments will strengthen our understanding of local thermal thresholds, improve predictions of future bleaching risk and support targeted, site-specific management interventions within Raja Ampat.

Photogrammetry

INTRODUCTION

To complement the coral bleaching monitoring program, a photogrammetry project was established to capture bleaching patterns across a broader spatial scale. Close-range underwater photogrammetry is an increasingly widely used method for coral reef monitoring that involves capturing overlapping photographs of the reef which are then processed to create high-resolution orthomosaic images of the benthic habitat. These models allow researchers to quantify reef structure, coral cover and colony-scale condition at sub-centimetre resolution (Remmers et al. 2024).

Compared with traditional monitoring approaches such as video-transects or point-intercept surveys, photogrammetry can document much larger, continuous areas of reef and can detect substantially greater coral abundance and species richness (Barrera-Falcon et al. 2021). This makes it particularly valuable for detecting patterns in coral bleaching across multiple coral taxa and for identifying spatial patterns in coral condition across the reef as it allows large areas of reef to be rapidly documented. The project was therefore established to expand our bleaching monitoring efforts by providing a broader spatial perspective on bleaching prevalence across two monitored reef sites. By repeating photogrammetry surveys at the same locations, changes in bleaching, mortality and reef structure can be quantified through time.

METHODS

Photogrammetry surveys were conducted at two reef sites in the Dampier Strait: JPP and Yenbuba. JPP was selected as it is the location of the long-term bleaching monitoring program, allowing photogrammetry data to complement existing colony-level observations. Yenbuba was selected due to the severe bleaching observed at this site during the recent regional bleaching event, providing an opportunity to document bleaching impacts across a wider reef area. Surveys were conducted on a fortnightly basis to document bleaching progression and early recovery through time.

At JPP, surveys were conducted within a 10 x 10 m permanent monitoring plot located on the southern protected side of the seamount at approximately 7–11 m depth. The plot is marked with permanent stakes at each corner to ensure the same reef area is surveyed. At Yenbuba, surveys were conducted within a 12 x 12 m plot located in front of the Yenbuba jetty at approximately 12–16 m depth, also marked with permanent corner stakes.

During each survey, transect lines were laid between the corner stakes to define the survey area. Five depth markers were placed within the plot to assist with spatial reference and model scaling during image processing. The depth of each corner stake and each depth marker was recorded prior to image collection to provide reference measurements for the photogrammetry models. Image collection was

conducted using three GoPro cameras mounted on a straight metal rig spaced 90 cm apart. Cameras were set to capture one image per second in linear mode. Divers swam approximately 1m above the reef surface, completing 1m spaced S-shaped flybys across the survey plot to ensure sufficient image overlap. Flybys were conducted first parallel to the reef slope and then perpendicular to the slope to maximise coverage and improve image alignment during processing.

Following each survey, images were uploaded and survey metadata were recorded. Photogrammetry models were constructed using Agisoft Metashape, which processes overlapping photographs to produce high-resolution orthomosaic images representing the reef surface in a two-dimensional spatial plane. The depth markers and recorded depth measurements provide reference points to assist with model scaling and spatial accuracy. Due to limited internet bandwidth and computing capacity at the field site, model construction and orthomosaic generation were undertaken by collaborators at the National Oceanic and Atmospheric Administration (NOAA). Analysis of the orthomosaic imagery is currently ongoing and will provide further insight into spatial patterns of bleaching, mortality and reef recovery across both monitoring sites.

RESULTS

Photogrammetry surveys were conducted fortnightly at both monitoring sites to document reef condition during and following the recent bleaching event. Image datasets were collected from August, October, November and December 2024, capturing the peak of the recent bleaching event, and January to July 2025, documenting bleaching response and early recovery. Surveys resumed in December 2025 and will continue into 2026.

At the time of writing, orthomosaic models have been successfully produced for December 2024 and January - March 2025. These orthomosaics provide high-resolution imagery of the survey plots and allow detailed visual assessment of coral condition across the monitored reef areas. Temporal changes in coral condition are illustrated in Figures 10 & 11, which show orthomosaic imagery from Yenbuba in January and March 2025, respectively, alongside zoomed in subsections of the same reef area. Visual comparison of these timepoints indicates a reduction of the extent of visibly bleached coral tissue between January and March, suggesting early signs of recovery at the colony level.

Further orthomosaic generation and quantitative analysis are currently ongoing in collaboration with NOAA, with the long-term aim of building and analysing future models on site at Barefoot Conservation.

DISCUSSION AND FUTURE DIRECTIONS

The photogrammetry component of this project has established a detailed visual record of reef condition across both monitoring sites during and following the recent bleaching event. Although quantitative analysis is still ongoing, the orthomosaic imagery provides clear evidence of temporal changes in coral condition and supports observations from colony-level surveys, including signs of recovery following peak bleaching. The use of permanent monitoring plots allows direct comparison of the same reef area through time, providing a strong foundation for tracking changes in coral condition at both the colony and community scale. As additional orthomosaics are generated, quantitative analysis will be undertaken to measure changes in coral cover, bleaching extent and mortality across both monitoring sites.

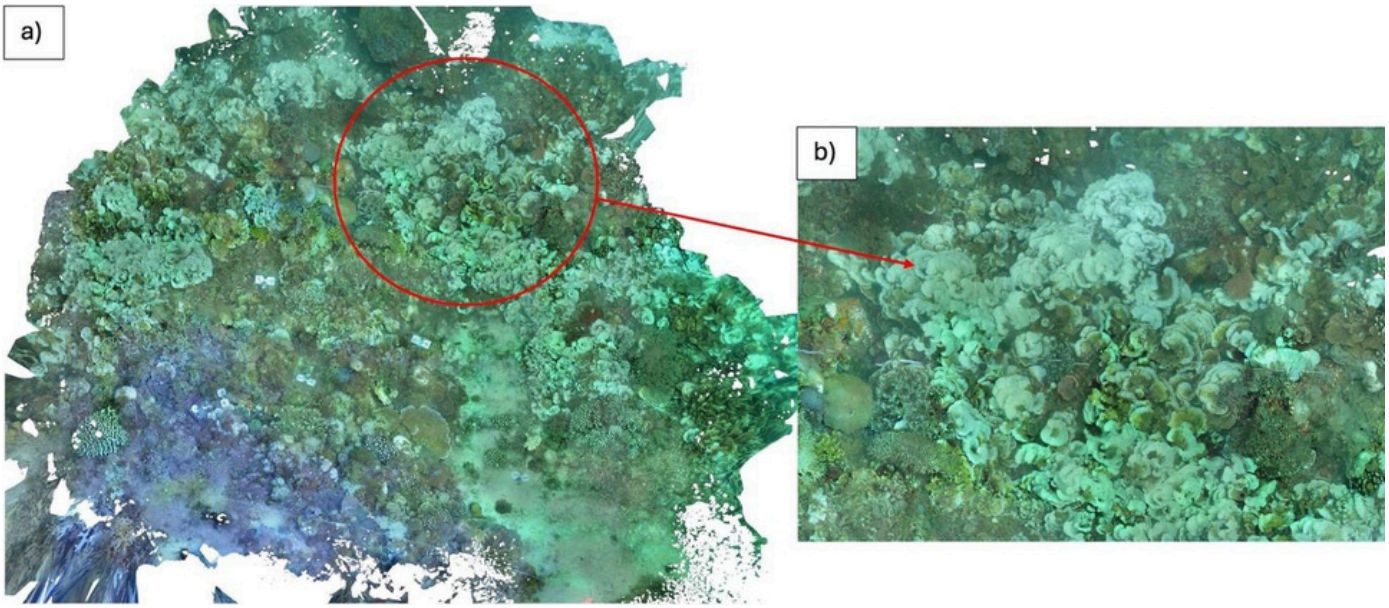


Figure 10. *Orthomosaic of 12 metre by 12 metre plot at Yenbuba imaged on Jan 24, 2025. a) Wide view of the entire plot. b) Narrow view of a smaller section of the plot that experienced bleaching*

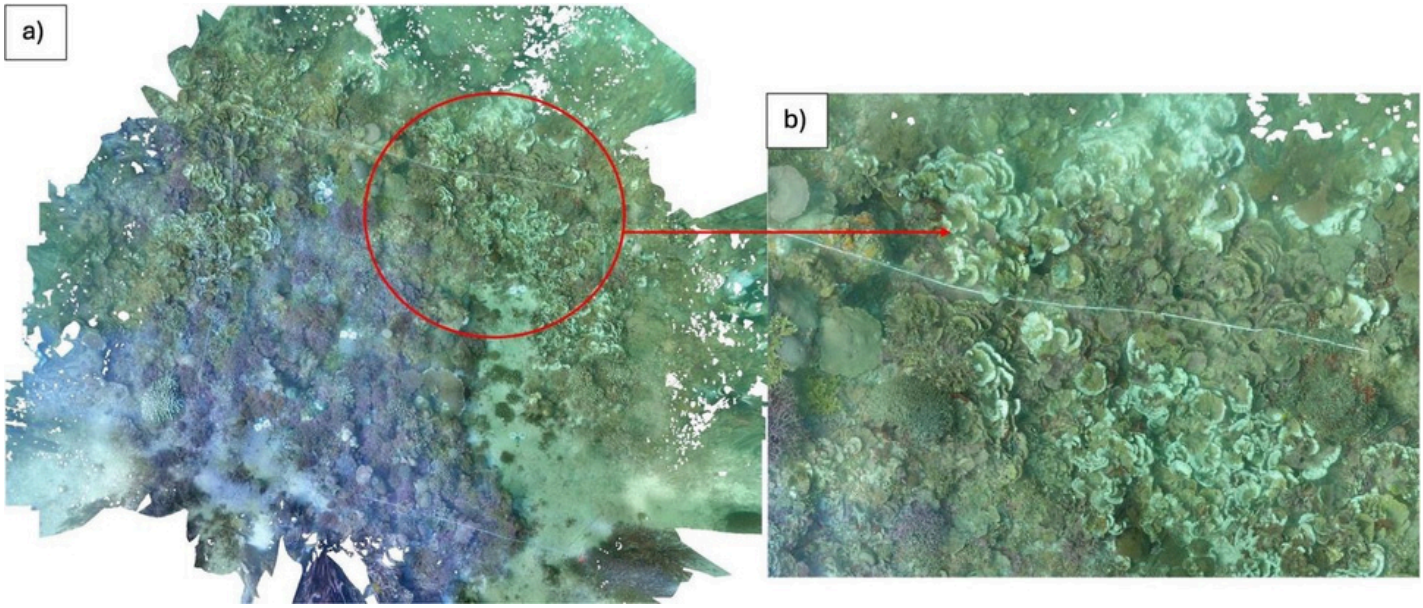


Figure 11. *Orthomosaic of 12 metre by 12 metre plot at Yenbuba imaged on March 28, 2025. a) Wide view of the entire plot. b) Narrow view of a smaller section that experienced bleaching.*

Future work will focus on expanding temporal coverage and developing analytical workflows to full utilise the photogrammetry dataset. Integrating these spatial data with colony-level observations and in-situ temperature records will provide a more comprehensive understanding of how thermal stress is impacting reef systems in Raja Ampat. Continued monitoring will be critical not only for assessing the long-term recovery trajectory following the 2024-2025 bleaching event, but also for detecting and responding to future bleaching events.

Coral Spawning

INTRODUCTION

Coral spawning is the primary mode of sexual reproduction for most scleractinian corals and typically occurs via broadcast spawning. In this strategy, colonies release buoyant egg-sperm bundles into the water column, where external fertilisation occurs (Harrison et al. 1984). Many broadcast-spawning species reproduce synchronously during mass spawning events, which typically take place once or twice per year depending on geographic location (Babcock et al. 1986; Penland et al. 2004). The environmental cues that trigger these events, such as temperature, lunar cycles, and night time duration after full moon can vary among regions and remain poorly understood in some areas (Keith et al. 2016; Sakai et al. 2020). Detailed data on multi-species synchronous spawning of scleractinian corals across the Indo Pacific remain limited (Baird et al. 2021). In Raja Ampat, in particular, there is currently no published literature, and only limited anecdotal information documenting the timing or occurrence of mass coral spawning. This represents a significant knowledge gap for one of the most biodiverse reef systems globally.

Understanding when and how corals spawn in this region is fundamental to interpreting coral biology and resilience within one of the world's most diverse reef systems. As corals face increasing pressures from climate change and local human activities, improved knowledge of their reproductive timing and success can inform more effective management, support targeted restoration efforts, and enhance predictions of how these reefs may respond to future disturbances.

The objective of this project is to identify the likely spawning periods for scleractinian corals in Raja Ampat and to determine which species participate in these mass spawning events. To guide this effort, we draw on established observations from the Banda Sea (Maluku, Indonesia), where coral spawning timing has been documented in detail (Novriansyah et al. 2023). Given the geographic proximity and broadly shared oceanographic influences between the Banda Sea and Raja Ampat, similar seasonal patterns are expected. Based on documented spawning trends across the wider Coral Triangle, we anticipate two major spawning periods: one occurring around April and a second between October and December (Bachtiar, 2001; Chelliah et al. 2015; Guest et al. 2005; Yusuf et al. 2013).

METHODS

Coral spawning monitoring was conducted at Barefoot Jetty, Arborek Island (00°33’54.0"S, 130°31’11.8"E). The primary method involved deploying egg traps to detect spawning activity. Egg traps consisted of an inverted funnel fitted with a plastic collection bottle and were secured to coral colonies or adjacent substrate using rope, ensuring stability while minimising disturbance.

Egg traps were deployed around full moon periods in April, October and November 2025. During April and October, 24 traps were deployed across multiple coral genera, with two replicate colonies per genus. In November, 12 traps were deployed across six coral genera. Coral genera were identified visually in situ, and replicate colonies were selected to be as similar as possible in morphology. Traps were monitored daily for eight days during each deployment period. The presence of gamete bundles (eggs and/or sperm) was recorded and photographed, after which gamete bundles were released back into the water column.

In addition to egg trap monitoring, targeted night dives were conducted during the October and November full moon periods to directly observed coral setting or spawning behaviour. No traps were deployed in December, however, night dive observations continued following the full moon.

RESULTS

Egg trap observations

Coral spawning activity was detected during two of the three egg trap deployment periods (Fig. 12). During the April 2025 deployment, spawning activity was recorded in three *Acropora* colonies. All three colonies released gamete bundles two days after the full moon. In October 2025, spawning was detected in a single *Acropora millepora* colony, which released gamete bundles one day after the full moon (Fig. 13). No spawning activity was recorded in egg traps during the November 2025 deployment period.

Night dive observations

Targeted night dives conducted during the October 2025 full moon period resulted in the observation of coral setting behaviour. At approximately 19:00, one *Acropora* colony located on one of the coral nursery frames was observed setting, indicating imminent spawning activity (Fig. 14a).

In December 2025, spawning was directly observed during a night dive at 19:21. A colony of *Montipora verucossa* was observed spawning at a depth of approximately 9m, adjacent to coral frames at the Barefoot Conservation nursery site (Fig. 14b). Spawning had commenced prior to arrival, and the active release of gametes was observed for approximately 10 minutes.

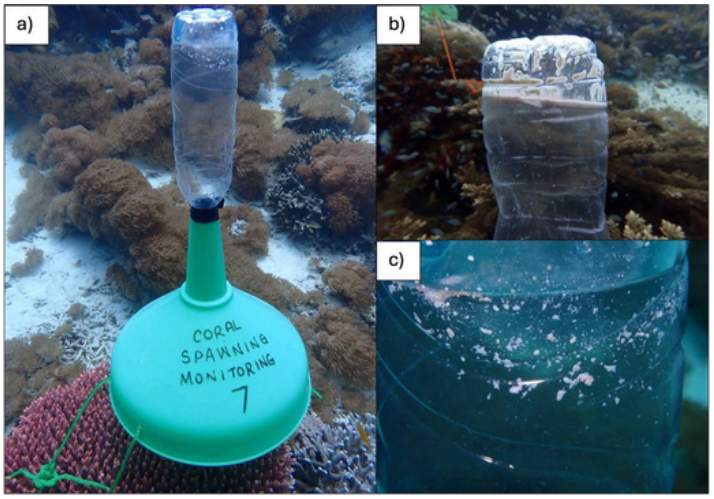


Figure 12. *Evidence of coral spawning recorded using egg traps at Barefoot Jetty, Arborek Island. a) Egg trap deployed on a coral colony, showing the collection bottle containing gamete bundles. b & c) Close up views of coral gamete bundles retained within the egg traps.*

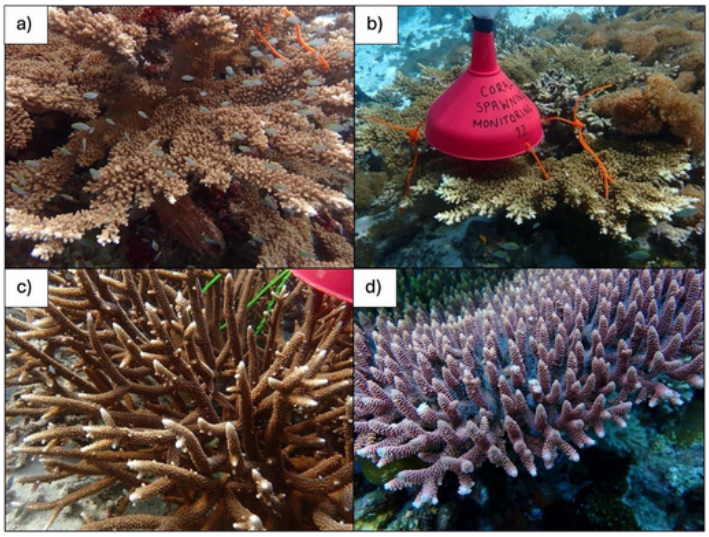


Figure 13. *Coral colonies in which spawning was detected using egg traps at Barefoot Jetty, Arborek Island. Shown are three Acropora colonies that spawned during the April 2025 full moon period (a – c) and one Acropora colony that spawned during the October 2025 full moon (d).*

DISCUSSION

This study provides the first structured observations of coral spawning activity at Arborek Island in Raja Ampat, addressing a key knowledge gap for a region where spawning timing remains poorly documented. The spawning detected indicates that broadcast spawning corals in Raja Ampat reproduce across multiple lunar periods, consistent with observations in other regions in the Coral Triangle.

Egg trap observations indicate *Acropora* species spawning during both the April and October full moon periods is consistent with observations from the Banda Sea and other parts of the Coral Triangle, where *Acropora* and other broadcast spawning genera are known to reproduce across several months rather

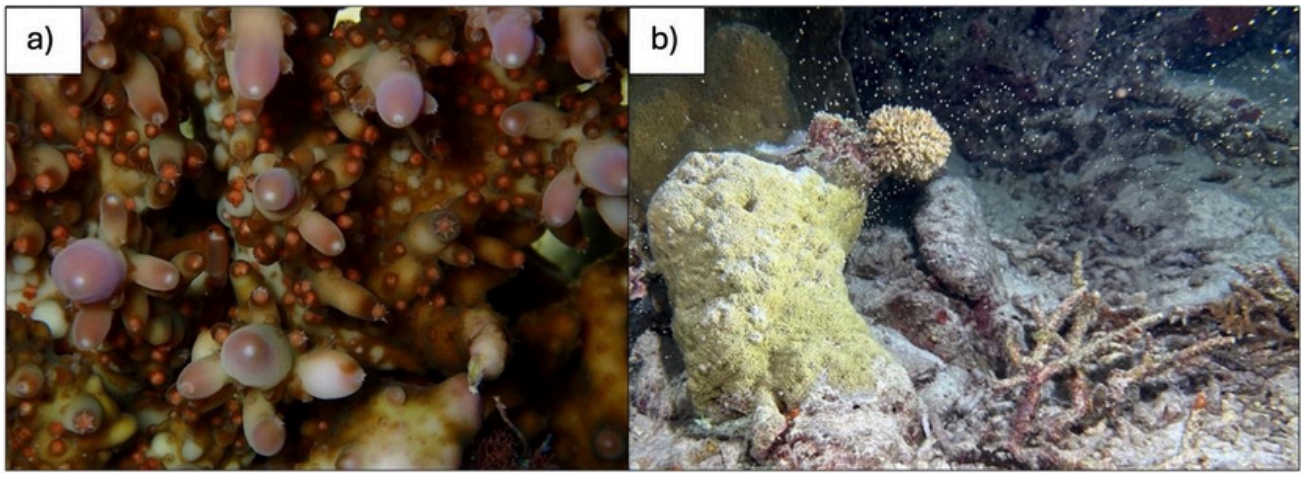


Figure 14. *Coral reproductive activity observed during night dives at Barefoot Jetty, Arborek Island. a) Coral setting observed on Acropora sp. in October 2025. b) Active coral spawning observed in December 2025 by Montipora verucossa, with gamete bundles being released into the water column.*

than a single annual event (Bachtiar, 2001; Guest et al. 2005; Novriansyah et al. 2023). The absence of spawning detections in eggs traps during November does not necessarily indicate a lack of reproductive activity, but may reflect species-specific spawning schedules, limited sample sizes, or methodological constraints.

Night dive observations provided additional insights into spawning behaviour that may not be captured by egg traps alone. The observation of *Acropora sp.* setting in October and *Montipora sp.* spawning in December confirms that reproductive activity occurred across multiple months. These findings suggest an extended or multi-phase spawning season, consistent with patterns reported from equatorial reefs where coral reproduction can occur over a broader temporal window rather than being restricted to a single synchronous event (Richmond & Hunter, 1990).

Collectively, these results suggest that coral spawning in Raja Ampat occurs across multiple months and involves different taxa spawning at different times. This pattern is consistent with observations from other highly diverse reef systems and highlights the importance of identifying species-specific spawning windows to better understand reproductive dynamics. While these observations provide valuable initial insight, they represent a limited subset of the coral community. Egg trap deployments were restricted to a small number of coral genera relative to the exceptionally high diversity present in Raja Ampat, and spawning activity in other taxa may not have been captured during the monitoring periods. In addition, the timing and detectability of spawning events may vary among species and years, highlighting the need for continued and expanded monitoring.

Future efforts will therefore focus on extending monitoring across lunar cycles, including targeted night dive observations to improve detection of spawning and setting behaviour. Continued multi-year monitoring will be essential to refine spawning windows, capture internal variability and assess how ongoing thermal stress and bleaching events may affect reproductive success and recovery potential in Raja Ampat.

Cyanobacteria Project Report



INTRODUCTION

Cyanobacteria is one of the most diverse phylums of life on Earth, and as an evolutionary group is responsible for the presence of Oxygen on Earth and the existence of life as we know it. Despite this, cyanobacteria has been known to pose a threat to human and animal life since the late 1870s, when a report detailing its dangers was published in the journal Nature (Francis, 1878). Cyanobacteria are a type of photosynthetic microbe found in almost every environment on Earth. Often termed "blue-green algae", they are technically not, as they are prokaryotes (have no organelles) and not part of the evolutionary group of eukaryotes, which can be classed as algae. On coral reefs, filamentous cyanobacteria are of particular interest. This is the type of cyanobacteria addressed in this report. These cyanobacteria are able to form microbial mats which proliferate exponentially under the right conditions. As these mats attach to the ocean floor (substrate) and grow rapidly, they can compete with hard corals and other important, reef-building species, and threaten the healthy growth and turnover of coral reefs around the world. Kuffner et al. (2006) found evidence that algae and cyanobacteria use tactics beyond space occupation to inhibit coral recruitment. On reefs experiencing phase shifts or temporary algal blooms, the restocking of adult coral populations may be slowed due to recruitment inhibition by cyanobacteria, thereby perpetuating reduced coral cover and limiting coral community recovery. Cyanobacterial mats act as a poison for scleractinian corals and are able to kill live coral tissue (Titlyanov, 2007).

In Raja Ampat, the rapid growth of tourism since 2017 has placed strain on local sanitation systems. Most communities in West Papua rely on basic sanitation, such as leach pit toilets, which were sufficient for local populations but are inadequate for the sudden and considerable influx of tourists. Concerns have been raised about the seepage of these inappropriate and overloaded systems into the marine environment, as they introduce excess nutrients and pathogens to coral reefs. Changes in land use that lead to alter dissolved organics, iron, and phosphorus input enhance proliferation of noxious blooms of cyanobacteria (Albert, 2005). Nutrient upload, the introduction of nutritive chemical elements and compounds into an ecosystem, can stimulate cyanobacteria growth. Similarly to plants and algae, cyanobacteria require nutrients to grow. The primary nutrients required by cyanobacteria are Nitrogen and Phosphorus, both of which are present at high levels in sewage/blackwater.

In Raja Ampat, we have noticed an increase in cyanobacterial mat presence on the reef in recent years. This appears to be closely tied with the rapid increase in tourism in the region, but may also be precipitated by rising sea temperatures, which accelerate cyanobacterial cell division. As nutrient upload is the main predictor of cyanobacterial biomass, we can infer that the observed increase in cyanobacteria on Raja Ampat's reefs is due to greater levels of nutrients entering the ocean. As well as providing an indicator for harmful levels of nutrients in the water, cyanobacteria also poses a direct threat to other organisms on the reef. Cyanobacteria produce toxins which are harmful to animal and plant life. These cyanotoxins bioaccumulate in target fishery species which can pose a threat to human health when consumed. As well as harming humans, cyanotoxins enable cyanobacterial mats to compete with young coral colonies establishing on the reef, hindering hard coral growth.

The cyanobacteria monitoring project continued in 2025 with the objective of assessing cyanobacterial mat proliferation at two monitoring sites around Arborek Island, Barefoot Jetty and Manta Sandy. The study aimed to determine whether cyanobacteria cover at these sites is increasing, decreasing, or remaining stable over time. These sites were selected due to potential anthropogenic influences, including the recent construction of a new toilet system at Barefoot Camp that may affect nutrient input near Barefoot Jetty, and the high tourism activity at Manta Sandy. We hope that the Cyanobacteria Monitoring Project produces findings which lead to improved wastewater management guidance for tourism operations in Raja Ampat. By addressing this issue, we aim to mitigate cyanobacteria proliferation, protect reef ecosystems, and enhance human health in the region.

METHODS

To establish a baseline understanding of cyanobacterial mat proliferation, reef monitoring surveys were conducted bimonthly over a one-year period. Data collection focused on documenting the presence and coverage of cyanobacterial mats to assess temporal changes and potential environmental drivers.

1. Site Selection

1.1 Barefoot Jetty

The Barefoot Jetty survey site (0°33'54.0"S 130°31'12.0"E) begins at the Barefoot jetty and extends eastwards toward the eastern tip of Arborek Island (Fig. 15).



Figure 15. *Map of two key cyanobacteria monitoring sites. Scale shown in bottom left-hand corner. Image taken from Google Earth.*

This site was selected to monitor potential environmental impacts associated with Barefoot Camp. In particular, monitoring aims to assess whether the newly installed toilet system effectively prevents nutrient discharge, such as nitrate, into the surrounding reef environment. As Barefoot Camp hosts up to 25 volunteers at a time, monitoring potential environmental impacts from these activities is considered important. The reef at this site is predominantly composed of rubble, with scattered hard coral bommies and soft corals. The survey area also includes part of the Barefoot Conservation restoration zone, where coral nurseries and outplanted corals aim to stabilise the rubble substrate and promote hard coral recruitment. This stabilisation may influence cyanobacterial mat development by either providing shelter for cyanobacteria or allowing other benthic organisms to compete for available space. The site frequently experiences strong currents, particularly near the eastern tip of the reef where water flow increases during falling tides. Overall, Barefoot Jetty represents a relatively degraded reef system with high rubble composition, relatively low hard coral cover, strong current exposure, and proximity to human activity at Barefoot Camp.

1.2 Manta Sandy

Manta Sandy was selected as a key site representing a reef affected by both historical degradation and high levels of tourism activity. The site experienced significant damage from historic blast fishing and is now one of the most frequently visited dive sites in Raja Ampat. The benthic composition at Manta Sandy is largely dominated by macroalgae and soft corals, with relatively low hard coral cover and very limited exposed abiotic substrate such as rubble, sand, or rock within the survey area. This substrate composition contrasts with Barefoot Jetty and provides an opportunity to compare cyanobacterial proliferation across reefs with different benthic structures.

Manta Sandy receives over 120 divers per day during peak tourism seasons (BLUD records), with numerous dive boats and liveaboard vessels operating in close proximity to the site. High levels of tourism activity may contribute to ecosystem disturbance through diver presence and potential nutrient inputs associated with boat operations, such as wastewater discharge. Overall, Manta Sandy represents a historically degraded reef system currently experiencing high tourism pressure.

2. Survey design

Cyanobacteria survey methodology was designed to align with methodologies used by other monitoring groups in Raja Ampat. Cyanobacteria surveys were conducted at 8m for each sites. Each survey consists of a 50m photo transect. An underwater camera (OM Systems TG-7, PT 059 housing) and rig were used to take photos each metre (50 photos per survey) along the transect at a set distance from the substrate, in order to capture a standardised area of the reef. While moving along the transect, the position of the rig over the transect also alternated sides, in order to maximise the area of reef covered. The rig was positioned at the 0.9m mark for each photo, so that the distance (metre mark) along the transect could be seen in each photo, with the lateral edge of the rig being approximately 10cm away from the transect on alternating sides (Fig. 16). The TG-7 camera was used in underwater wide mode, with white balance set at depth to ensure photos were taken with clear imaging of the substrate.

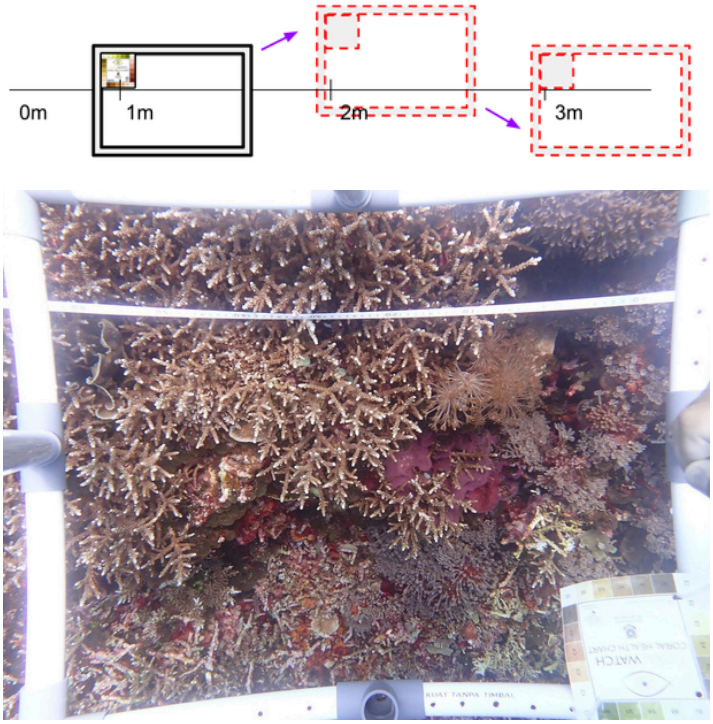


Figure 16. *Rig positioning (above) and photo example (below) for cyanobacteria monitoring survey.*

3. ReefCloud Analysis

All photoquadrats from cyanobacteria monitoring surveys were uploaded to ReefCloud (Australian Institute of Marine Science) for artificial intelligence (AI) assisted analysis. Using ReefCloud accelerated post-dive photo-analysis, improving accuracy of substrate identification while enabling us to keep up with the quantity of data collected during these surveys. ReefCloud uses machine-learning and pattern-recognition to semi-automate identification of substrate types. 30 points were randomly placed on each image, and the substrate beneath each point was identified. Initially this was carried out by project scientists and volunteers, but over time the AI was able to increasingly successfully identify substrates. Recently, the AI has been able to identify substrates with over 95% accuracy, with final checking by humans still remaining to ensure validity.

We created a label set with 44 different substrate types to classify bottom composition, including soft coral, hard coral, rock and rubble, and cyanobacteria on macroalgae, soft coral, hard coral, rock, rubble and sand (Fig. 17). This label set is available for use or inspection on request.

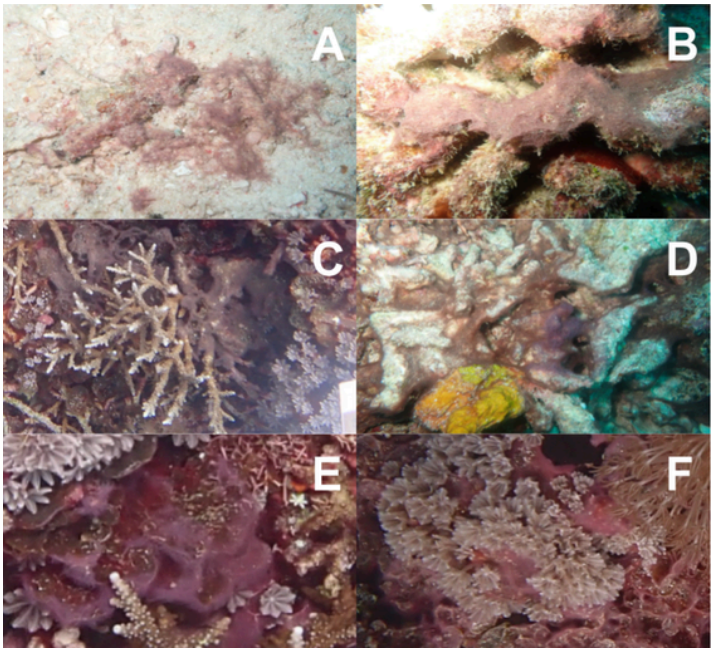


Figure 17. *Cyanobacterial mats growing on different substrates. A - Sand, B - Rock, C - Hard Coral, D - Rubble, E - Macroalgae, F - Soft Coral*

RESULTS

Baseline Bottom Composition

Figure 18 contains the benthic substrate composition at Barefoot Jetty (8m) and Figure 19 contains the benthic substrate composition at Manta Sandy (8m) across the monitoring period. Both sites were dominated by soft coral; however, Manta Sandy maintained consistently higher soft coral cover (60.35%) compared to Barefoot Jetty (49.81%).

Hard coral cover was substantially higher at Manta Sandy (31.43%) than at Barefoot Jetty (16.91%), indicating greater reef framework development at Manta Sandy. In contrast, Barefoot Jetty exhibited a more unconsolidated substrate, with higher rubble (42.32%) and consistently high sand cover (20.64%), whereas rubble at Manta Sandy are lower (25.58%) and sand was minimal (≤2.71%). Cyanobacteria cover was low at both sites and declined through time at Manta Sandy (4.88% to 1.04%), while remaining consistently low with minor fluctuation at Barefoot Jetty (2.66% to 0.12%, increasing slightly to 1.77% in December).

In comparison, monitoring in 2024 recorded considerably higher cyanobacteria cover at Manta Sandy (8% at 8m), while Barefoot Jetty remained below 1%. During the 2025 monitoring period, cyanobacteria cover at Manta Sandy was lower, ranging from 4.52% to 1.04%, while Barefoot Jetty showed slightly higher but still relatively low values, ranging from 2.66% to 0.12% and increasing to 1.77% in December.

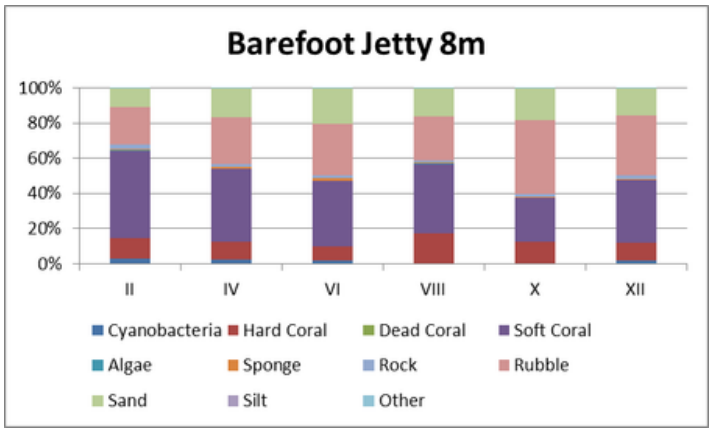


Figure 18. *Baseline bottom composition - Figure showing the relative frequency of different benthic types / substrates at Barefoot Jetty (8m). Cyanobacteria is displayed at the bottom of each bar for viewing ease.*

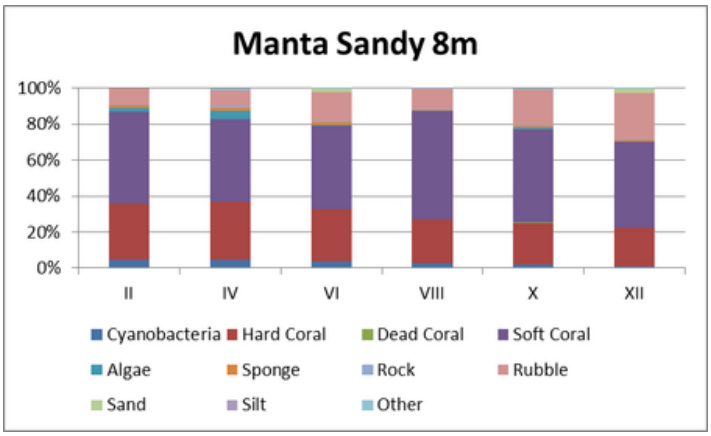


Figure 19. *Baseline bottom composition - Figure showing the relative frequency of different benthic types / substrates at Manta Sandy (8m). Cyanobacteria is displayed at the bottom of each bar for viewing ease.*

Cyanobacteria growth over time

Figure 20 shows the temporal trend of cyanobacteria percentage cover at Manta Sandy (8 m) and Barefoot Jetty (8m). Cyanobacteria cover at Manta Sandy was consistently higher than at Barefoot Jetty throughout the monitoring period, beginning at 4.52% in February, peaking at 4.88% in April, and then declining steadily to 1.04% by December. In contrast, Barefoot Jetty exhibited lower initial cyanobacteria cover (2.66% in February) and a gradual decrease to very low levels between August and October (0.16–0.12%), followed by a slight increase to 1.77% in December. Compared to 2024, when cyanobacteria cover at Manta Sandy reached approximately 8% at 8m while Barefoot Jetty remained below 1%, the current monitoring period suggests an overall reduction in cyanobacteria cover at Manta Sandy, while Barefoot Jetty showed slightly higher but still relatively low levels of cyanobacteria.

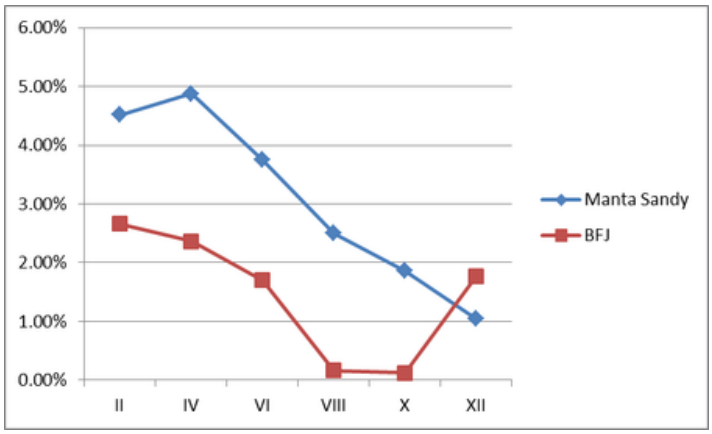


Figure 20. *Cyanobacterial mat coverage over time - Figure showing mat coverage change over time for both sites.*

Substrate Susceptibility

Figure 21 shows the distribution of cyanobacterial cover across different benthic substrate types at both monitoring sites. Cyanobacteria were most frequently associated with rubble, accounting for 56% of the total observed cyanobacterial coverage. Soft coral substrates represented the second highest proportion, supporting 22% of cyanobacterial cover. Smaller proportions were observed on hard coral (9%), algae (8%), and rock (5%). Overall, cyanobacteria were predominantly found on unconsolidated substrates, particularly rubble, while relatively lower occurrences were observed on more stable substrates such as hard coral and rock. This pattern is consistent with observations from 2024, where rubble was also identified as the most frequent substrate associated with cyanobacterial cover.

DISCUSSION

Monitoring cyanobacterial mat coverage and benthic composition at Barefoot Jetty and Manta Sandy provides insight into the dynamics of cyanobacteria proliferation around Arborek Island. The benthic composition observed during the monitoring period

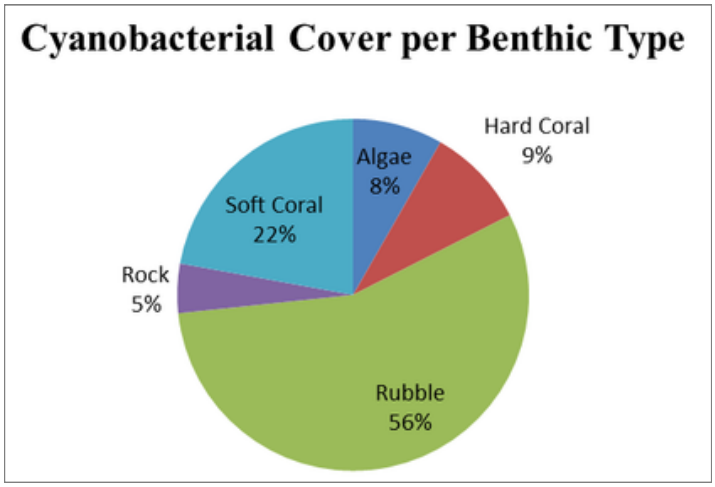


Figure 21. *Cyanobacterial cover per benthic type - Figure showing the relative frequency of Cyanobacterial cover per benthic type at both sites*

shows clear differences between the two sites. Manta Sandy was dominated by soft coral and relatively higher hard coral cover, while Barefoot Jetty was characterized by a more unconsolidated substrate with higher proportions of rubble and sand. These differences in substrate composition may influence the availability of suitable surfaces for cyanobacterial growth and may contribute to variation in cyanobacterial cover between sites.

Compared with observations from 2024, cyanobacteria coverage at Manta Sandy was considerably lower during the 2025 monitoring period. In 2024, cyanobacterial cover at Manta Sandy reached approximately 8% at 8m depth, whereas in 2025 it ranged from 4.52% in February and gradually declined to 1.04% by December. This reduction may indicate that cyanobacterial proliferation at the site is not persistent over long periods and may fluctuate depending on environmental conditions. Changes in water movement, seasonal variability, or other ecological factors may contribute to this decline, although the precise drivers remain unclear.

At Barefoot Jetty, cyanobacteria levels remained relatively low overall but were slightly higher than recorded in 2024. While cyanobacteria remained below 1% at this site in the previous year, the 2025 monitoring period recorded an initial value of 2.66% in February, followed by a decline to very low levels before increasing slightly to 1.77% in December. Although these values remain relatively low compared to Manta Sandy, this increase highlights the importance of continued monitoring at this site, particularly given its proximity to Barefoot Camp. The recently installed toilet system at Barefoot Camp was designed to prevent nutrient discharge into the surrounding reef environment, and continued monitoring will help confirm whether nutrient inputs remain effectively controlled.

Analysis of cyanobacterial distribution across substrate types showed that rubble was the most frequent substrate associated with cyanobacterial growth, accounting for 56% of total observed cyanobacterial cover. This pattern is consistent with findings from 2024, where rubble was also identified as the dominant substrate supporting cyanobacterial mats. Rubble provides unstable and exposed surfaces that may be easily colonized by opportunistic organisms such as cyanobacteria. In contrast, more consolidated substrates such as hard coral and rock supported smaller proportions of cyanobacterial cover.

The results suggest that degraded reef areas with higher rubble composition may be more susceptible to cyanobacterial proliferation. Although both monitoring sites showed relatively low overall cyanobacterial coverage during the 2025 monitoring period, the continued presence of cyanobacteria on rubble substrates highlights the potential vulnerability of disturbed reef habitats. Understanding how substrate composition interacts with environmental conditions will be important for predicting future cyanobacterial dynamics in the region.

Tourism activity may also play a role in influencing cyanobacterial presence at key dive sites. Manta Sandy receives high numbers of divers and liveaboard vessels throughout the tourism season. While cyanobacteria coverage declined during the monitoring period, continued observation is necessary to determine whether tourism-related pressures such as boat activity or wastewater discharge contribute to changes in reef condition. Long-term monitoring will be essential to distinguish natural variability from potential anthropogenic impacts.

FUTURE DIRECTIONS

Moving forward, the cyanobacteria monitoring project will continue to track changes in cyanobacterial coverage at the key monitoring sites of Barefoot Jetty and Manta Sandy. Extending this time series will allow for improved understanding of long-term trends and seasonal variability in cyanobacteria proliferation around Arborek Island.

In future monitoring efforts, we plan to reintroduce water quality measurements as part of the project. While water quality sampling was not conducted during the 2025 monitoring period, previous monitoring efforts highlighted the importance of understanding nutrient levels and other water quality parameters in relation to cyanobacterial growth. Reestablishing water quality monitoring will help assess whether nutrient inputs or other environmental factors may influence cyanobacterial dynamics at the study sites.

Another important goal for the coming year is to expand the monitoring program to include additional sites around Arborek Island. By increasing the number of monitored locations, we aim to develop a more comprehensive understanding of cyanobacteria distribution across reefs surrounding the island. Expanding the spatial coverage of surveys will allow comparisons between sites with different environmental conditions, levels of tourism activity, and benthic compositions, ultimately providing a clearer picture of reef health across the Arborek reef system.

The continued use of ReefCloud for benthic image analysis will remain an important component of the monitoring program. Improvements in automated substrate classification may increase the efficiency and consistency of data analysis and allow monitoring efforts to scale up as additional survey sites are incorporated.

Through continued monitoring and expanded data collection, the cyanobacteria project aims to better understand the drivers of cyanobacterial proliferation in the region and provide useful insights for reef management in Raja Ampat. Long-term monitoring will be essential for identifying patterns of reef change and supporting conservation efforts that help protect the ecological integrity of reefs surrounding Arborek Island.

Marine Debris Project Report



INTRODUCTION

Marine debris pollution is a growing global issue that threatens the health of marine ecosystems worldwide. Plastics are the most common form of marine litter and found in environments ranging from densely populated coastlines to some of the most remote regions on Earth (Barnes et al. 2010). Their durability allows them to persist in the marine environment for hundreds of years, where they gradually break down into smaller fragments and microplastics and continue to accumulate over time (Wayman & Niemann, 2021). Plastic pollution poses serious risks to marine life through entanglement, ingestion, habitat degradation and disruption of food webs (Derraik, 2002; Rochman et al. 2013).

Although Raja Ampat is renowned for its exceptional marine biodiversity and relatively remote location, it is not immune to the impacts of marine debris. Studies have identified Raja Ampat and the surrounding waters of West Papua as areas of marine debris accumulation, driven by major currents and circulation patterns that transport and concentrate floating waste throughout the region (Purba et al. 2021). Communities throughout Raja Ampat regularly observe large quantities of debris washing ashore, while floating accumulations are frequently encountered on the sea surface.

As part of our ongoing commitment to marine conservation, Barefoot Conservation conducts a Marine Debris project that includes regular beach clean-up activities and debris monitoring. Our project aims to quantify the amount and composition of marine debris accumulating on Arborek shorelines, providing valuable data on pollution trends within the region. These data are shared with the local government and other environmental organisations to support evidence-based management, raise awareness of marine pollution and inform future mitigation strategies.

METHODS

The Marine Debris project aims not only to remove waste from Arborek's coastline but also to monitor the quantity and composition of debris accumulating on the island. During 2025, beach clean-ups were conducted twice weekly across four designated monitoring sites around the island (Fig. 22). The sites collectively cover the majority of Arborek's shoreline, with individual site lengths ranging from approximately 200 – 300m. This sampling design resulted in each site being surveyed and cleaned on a fortnightly basis.

Beach clean-ups were carried out by Barefoot Conservation staff and volunteers. Following collection, all debris was transported to Barefoot camp for processing.

Waste was weighed, sorted into 14 predefined categories, and counted to quantify both the abundance and composition of marine debris. Categories included broad material types such as plastic, glass, metal and cardboard, as well as specific items commonly encountered on Arborek, including plastic cups and plastic ice-pop containers. After processing, recyclable materials such as cardboard, glass and metal were disposed of through appropriate waste management alongside camp waste. The remaining debris was transported to a secure landfill site in Waisai, outside the Raja Ampat MPA.



Figure 22. *Arborek Island with four beach clean survey sites labelled*

RESULTS

A total of 2,888kg (2.9 tonnes) of marine debris was collected and removed from Arborek shorelines during 2025 (Fig. 23), almost double the amount that was recorded in 2024 (1,351kg). Debris accumulation was highest in January and February, followed by April and June. The elevated quantities recorded during January and February coincide with the Northwest Monsoon (November – March), which is characterised by occasional strong winds and ocean swell across the region (Mangubhai et al. 2012). Previous studies have shown that floating marine debris accumulates in western Papua during this period, particularly between December and February (Purba et al. 2021). These seasonal current patterns likely contribute to the increased volumes of debris washing ashore on Arborek during the early months of the year.

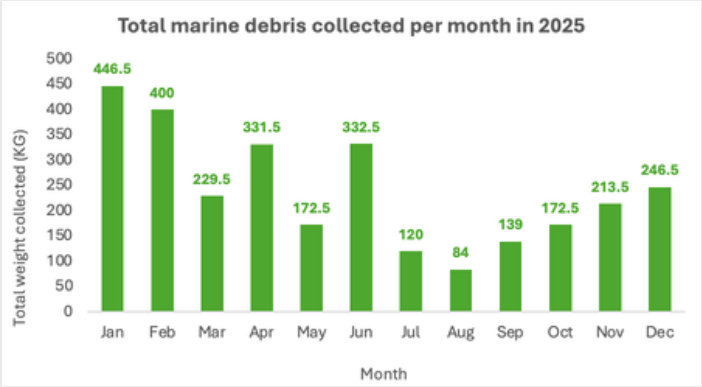


Figure 23. *Total weight of waste collected each month on Arborek Island during beach cleans, presented from January - December 2025*

The increase in quantities observed in April may reflect changing wind and current conditions during the transition between monsoon seasons, which can redistribute debris throughout the region. A further peak in June coincides with the onset of the Southeast Monsoon, when changing wind and current patterns have been shown to transport debris towards West Papua (Purba et al. 2021). Together, these findings suggest that the seasonal pattern of debris accumulation observed on Arborek is strongly influenced by monsoon-driven oceanographic processes.

Debris accumulation varied considerably among monitoring sites (Fig. 24). Site 4 recorded the highest quantity of debris collected throughout the year. Located on the western side of Arborek, this site encompasses an extensive mangrove area which may act a natural trap for floating debris. Site 2 recorded the second highest total, followed closely by Site 3, both of which are situated along the northern shoreline of the island. In contrast, Site 1 recorded the lowest quantity of debris. This site is located on the north-eastern side of Arborek between the village's

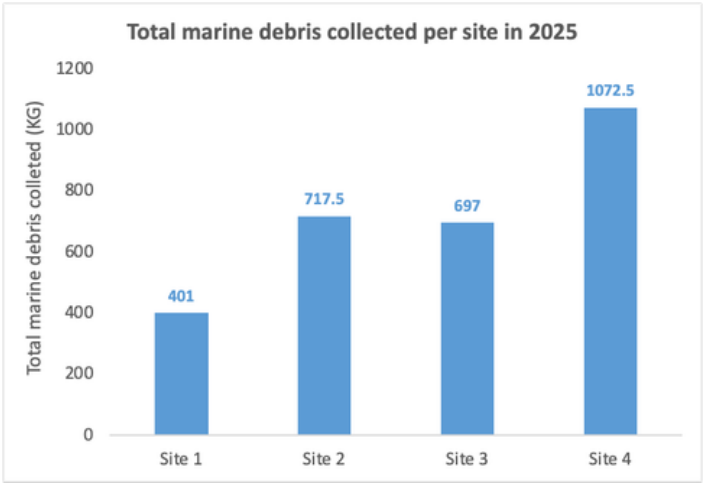


Figure 24. *Total weight of waste collected across four sites on Arborek Island during beach cleans*

main jetty and Barefoot Conservation's jetty and covers a smaller section of coastline than the other monitoring sites.

A total of 77,798 debris items were collected across all monitoring sites during 2025. Hard plastics were the most common debris category, followed by styrofoam, plastic cups, plastic bottles, plastic wrappers, straws and soft plastics (Fig. 25). Collectively, these plastic-based materials comprised more than 80% of all debris collected, indicating that plastic pollution remains the primary contributor to shoreline litter on Arborek. In contrast, non-plastic materials such as rope and line, cardboard, fabrics, metal and glass represented a comparatively small proportion of total items.

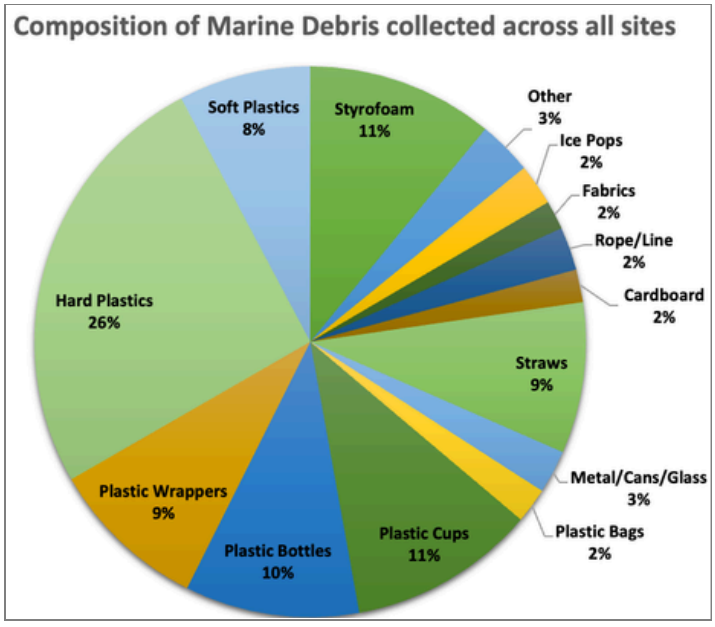


Figure 25. *Relative proportions of marine debris collected during 2025 beach cleans on Arborek Island, pooled across all sites*

DISCUSSION AND FUTURE DIRECTIONS

The results of this project demonstrate that marine debris continues to be a significant issue on Arborek. Plastic-based materials accounted for the vast majority of debris collected, reflecting the persistent nature of plastic pollution in the marine environment and highlighting the need for improved waste management and waste reduction strategies both within and beyond Raja Ampat.

Continued monitoring will provide valuable long-term data on the quantity and composition of marine debris impacting Arborek, helping to demonstrate the extent of the issue and provide evidence to support waste management initiatives and pollution reduction efforts. In 2026, Barefoot Conservation will expand the Marine Debris Project through a new collaboration with Soul Initiatives. This partnership will enable collected marine debris to be recycled, reducing the volume of waste sent to landfill. By combining debris removal, monitoring and recycling, the project aims to contribute to more sustainable waste management solutions within Raja Ampat.



Crown-of-Thorns Starfish Project Report



INTRODUCTION

Crown-of-Thorn Starfish (COTS) are echinoderms and one of the major predators of corals. COTS are endemic to the tropical coral reefs throughout the Indo-Pacific region (Deaker and Byrne, 2022). They play an important role in promoting coral diversity by keeping fast-growing coral species in check, allowing slow-growing corals space to live and grow. However, abnormally high numbers of COTS on coral reefs can lead to devastating impacts, as they may shift their feeding from fast-growing corals to slow-growing species and contine feeding until much of the coral on the reef has been consumed. COTS are already recognised as one of the major drivers of coral mortality throughout the Indo-Pacific (Castro-Sanguino, et al. 2023).

The increase in tourism in Raja Ampat may have indirectly contributed to increasing COTS populations, likely linked to the Terrestrial Run-Off Hypothesis. This hypothesis suggests that nurtient-rich runoff entering the ocean promotes phytoplankton growth. Phytoplankton are an important food source for COTS larvae, and their abundance can increase following terrestrial runoff during periods of high rainfall (Deaker and Byrne, 2022). The increasing number of COTS is becoming a growing concern, particularly as coral reefs in Raja Ampat experienced mass bleaching events at the end of 2024 and still require time to recover. Coral growth can be outpaced by predation from even 'average' sized COTS, while the ecologically sustainable density threshold is estimated at 3 ha^{-1} (Westcott et al, 2020). In addition to the progressive build-up of COTS densities and nutrient loading, overfishing of natural COTS predators has also contributed to conditions that facilitate increases in COTS populations (Milne et al. 2023).

The Crown-of-Thorn Starfish (COTS) Cull Surveys were established by the Barefoot Science Team in April 2025 following COTS monitoring conducted at fives sites around Dampier Strait in March 2025. The results indicated that estimated COTS numbers were showing signs of an early outbreak. The survey currently includes six sites around Arborek Island and four additional sites outside Arborek Island. The initial objective of this survey was to control COTS numbers around Arborek Island and the wider Dampier Strait area and, hopefully, help prevent a second outbreak from occurring. The results are subsequently reported to the Raja Ampat Marine Park Authorities (BLUD UPTD).

METHODS

Locations for the COTS cull were selected based the locations of Reef Check surveys, with six core sites around Arborek Island (Fig. 26) and four sites nearby Arborek Island (Fig. 27). The sites chosen for the COTS culls represent a range of coral reef conditions and levels of diversity, as well as differing exposure to currents and varying levels of anthropogenic impacts.

COTS cull dives occured several times per week, depending on the number of COTS sightings observed during dives conducted at survey sites. Each dive was conducted along a section of reef estimated to be 100m in length, with a survey width of approximately 5m, depending on topography and visibility, up to 10m.

Each dive was led by two dive guides or scientists tand supported by at least 4 volunteers. Participants were divided into two teams, with each teams consisting of atleast three people. Divers were assigned different roles to support the survey, with one person operating the injector gun, one person carrying the gaff to remove COTS, and one person recording obvservations on a slate.



Figure 26. *COTS Cull locations on Arborek Island (Source: Google Earth)*



Figure 27. *COTS Cull locations near Arborek Island (Source: Google Earth)*

Divers within each team were positioned at different depths and spaced 2–5 m apart, depending on reef structure, currents, and visibility. They searched thoroughly for COTS beneath corals, inside crevices, and within cracks in the reef structure. Whenever a COTS was found, the diver signalled the rest of the team. The gaff holder removed the COTS from its location if necessary and measured its arm-to-arm diameter. The slate holder recorded the depth at which the COTS was found, any feeding scars observed in the surrounding area, and the behaviour of the individual. Once the diameter had been measured, the injector gun operator administered 20mL of vinegar solution into the polian vesicle or at the base of an arm. For COTS measuring 40 cm or greater in diameter, 40mL of vinegar solution was administered. The slate holder also recorded whether the injection was successfully delivered.

RESULTS

1. Total COTS found Throughout 2025

Figure 28 shows the total number of Crown-of-Thorns Starfish (COTS) culled at selected sites around Arborek Island and four additional core sites between April and December 2025. The highest number of COTS culled was recorded at West Arborek, with 161 individuals removed across 23 dives, followed by Juan's Reef with 142 COTS culled from 20 dives, and Isaia's Garden with 108 COTS culled from 18 dives. Juan's Bay recorded 75 COTS culled from 9 dives, followed by Kakawa Corner with 74 COTS culled from 3 dives, Arborek Tip/Barefoot Jetty with 68 COTS culled from 15 dives, and Manta Sandy with 42 COTS culled from 5 dives. Lower numbers were recorded at South Arborek, with 12 COTS culled from 3 dives, and Southwest Arborek, with 10 COTS culled from 2 dives. Main Jetty recorded the lowest number of COTS culled, with 8 individuals removed across 4 dives.

The frequency of dives conducted at each site depended on site conditions and observations made during previous surveys. The high number of dives conducted at West Arborek, Juan's Reef, and Isaia's Garden was driven by the presence of numerous COTS feeding scars at these sites, as well as frequent reports of COTS sightings during other non-COTS-cull dives.

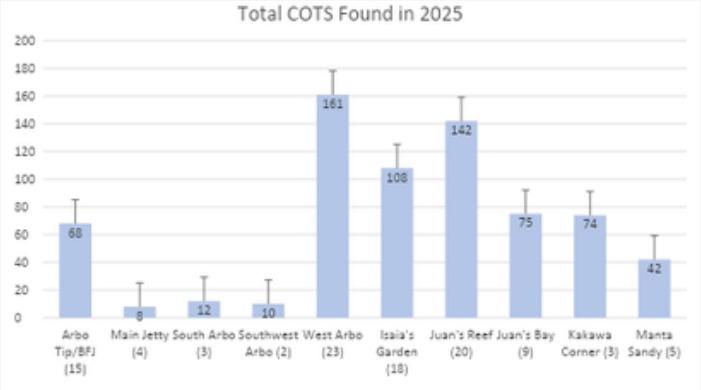


Figure 28. *Total COTS Culled at all core sites in 2025*

2. COTS on Arborek Island

Figure 29 shows the number of COTS found and culled at sites around Arborek Island throughout 2025. West Arborek showed a consistent presence of COTS throughout the year, with the highest number recorded in September, when 48 COTS were culled, and the lowest number recorded in August, when 5 COTS were culled. Isaia's Garden showed a notable increase in the number of COTS culled in July, as the search area was expanded into very shallow reef areas, which were found to be among the most heavily infested sections of the site. The total number of COTS culled across the six core dive sites on Arborek Island was 367 individuals. Among these sites, Main Jetty recorded the lowest total number of COTS culled throughout the survey period.

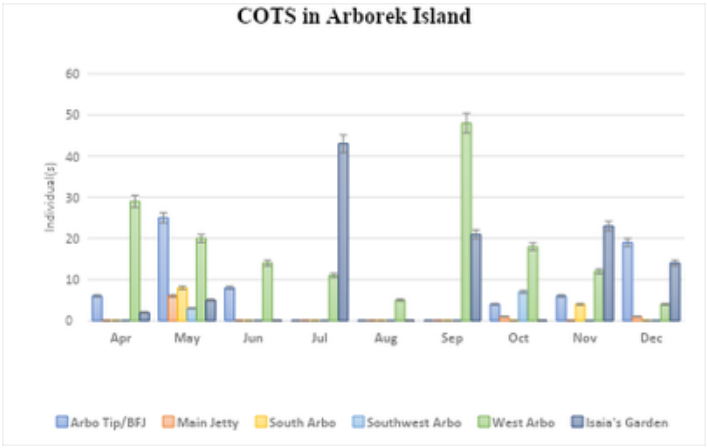


Figure 29. *COTS Culled at sites on Arborek Island throughout 2025*

3. COTS around Arborek Island

Figure 30 shows the number of COTS found and culled at the four additional core sites throughout 2025. Juan's Reef showed a consistent presence of COTS, with individuals being discovered and culled in every month surveyed. The highest number of COTS recorded at any of these sites was at Kakawa Corner in December, when 58 individuals were culled. This site had only been surveyed since November, following Divemaster trainee mapping dives that identified the area as a priority for COTS monitoring and removal. Juan's Bay showed the second most consistent occurrence of COTS, with individuals being culled during most survey months, except for March, July, and December. Manta Sandy showed fluctuations in the number of COTS culled during the final three months of the year, with the highest number recorded in November. Many of these individuals were found while exploring the area behind the waiting area adjacent to the manta cleaning station.

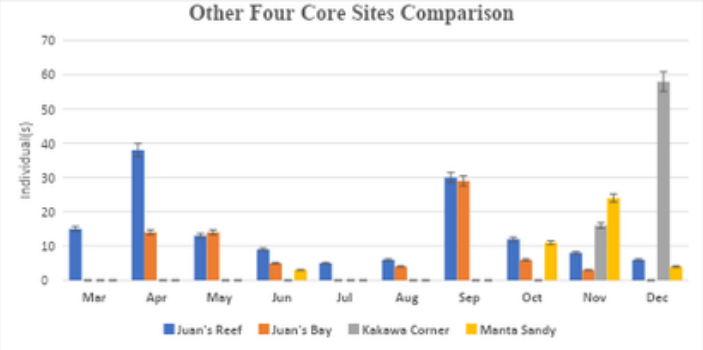


Figure 30. *COTS Culled at sites further afield from Arborek Island*

4. *Comparison of April - July and August - December Period*

Figures 31 and 32 present comparisons of the number of COTS culled across survey sites during two survey periods: April–July 2025 and August–December 2025, respectively. The data show that during the April–July period, West Arborek and Juan's Reef recorded COTS removals every month, with the highest numbers at both sites occurring in April, followed by a gradual decline in subsequent months. The lowest numbers of COTS culled during this period were recorded at Southwest Arborek and Manta Sandy, with 3 individuals removed at each site. The greatest fluctuation occurred at Isaia's Garden, where the number of COTS culled increased from 0 in June to 43 in July. Kakawa Corner was not surveyed during this period.

In comparison, Figure 32 shows that during the August–December period, West Arborek and Juan's Reef continued to record COTS removals every month, with the highest numbers at both sites occurring in September. Main Jetty recorded the lowest number of COTS culled during this period, with only 2 individuals removed, followed by South Arborek with 4 individuals and Southwest Arborek with 7 individuals. The highest number of COTS culled during the August–December period was recorded at Kakawa Corner in December, where 58 individuals were removed from just two survey dives.

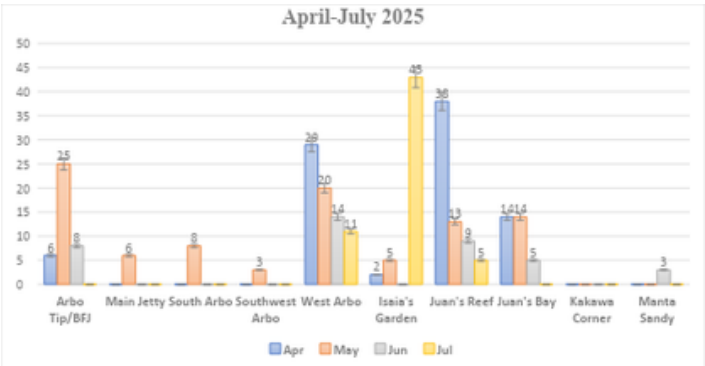


Figure 31. *COTS Culled April to July at all sites during 2025*

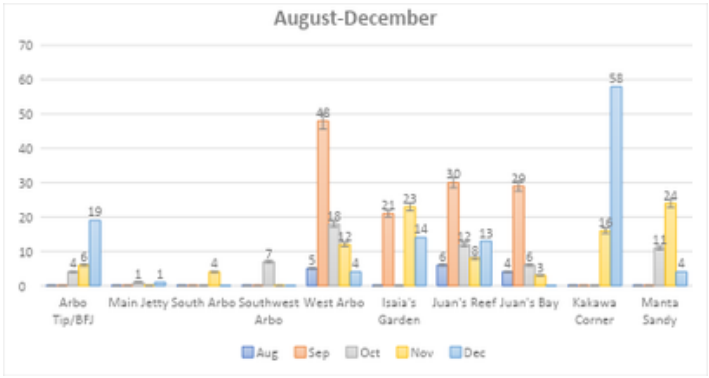


Figure 32. *COTS Culled August to December at all sitesduring 2025*

DISCUSSION

The Crown-of-Thorns Starfish (COTS) Monitoring and Control Program at Barefoot Conservation provides valuable insight into the abundance and distribution of COTS throughout Raja Ampat, particularly around Arborek Island and its surrounding reefs. The mass bleaching event at the end of 2024 highlighted concerns regarding the long-term resilience and sustainability of Raja Ampat's coral reefs. Although Raja Ampat is recognised as having some of the most resilient coral reef ecosystems in the world, its reefs were still significantly affected by the bleaching event. Recovery of coral reef ecosystems is a slow process, and COTS are among the largest and most efficient coral predators. As one of the major contributors to coral mortality, high COTS abundance can place additional pressure on reefs that are already in a degraded state. Furthermore, the history of a COTS outbreak in Raja Ampat during 2017–2018 reinforces the importance of increasing and strengthening monitoring and culling efforts around Arborek Island.
Based on the initial density data, COTS abundance appears to be increasing, which is also supported by the growing number of sightings reported during routine dives. Using the density threshold applied on the Great Barrier Reef, where densities greater than 15 individuals ha^{-1} may indicate outbreak conditions, many of the core monitoring sites around Arborek Island exceed this threshold and could be considered to be experiencing outbreak-level densities.

Analysis of the COTS cull data throughout 2025 showed that West Arborek recorded the highest number of COTS removed, with 161 individuals culled across 23 dives. In contrast, Main Jetty recorded the lowest number, with 8 individuals culled across 4 dives. The highest monthly total was recorded at Kakawa Corner in December, where 58 COTS were culled during only two survey dives.

The results suggest that April and September were periods during which many of the core sites recorded relatively high numbers of COTS removals. Elevated

numbers of COTS were also recorded across multiple sites during May and December. Increasing the frequency of monitoring and culling dives at core sites may improve understanding of COTS distribution patterns and population dynamics. Continued monitoring may also provide information on the timing of spawning events, which could contribute to more effective management strategies and help reduce the likelihood of future outbreaks.

FUTURE DIRECTIONS

Moving forward, the COTS Monitoring and Control Project will continue to track changes in COTS distribution and density across the ten core monitoring sites around Arborek Island and, where possible, surrounding areas. The project will also continue to actively manage sites affected by high COTS densities and, when necessary, intervene directly through targeted culling efforts. These actions aim to reduce the risk of future COTS outbreaks while improving understanding of COTS spatial distribution patterns and identifying potential hotspots of COTS abundance over the long term.

Future monitoring and control efforts will also seek to identify and analyse the Gonadosomatic Index (GSI), the percentage of gonad weight relative to total body weight, as an indicator of reproductive condition and spawning activity. Understanding when spawning events are likely to occur may help guide management actions aimed at reducing larval recruitment and future population increases, while also improving knowledge of COTS reproductive ecology in Raja Ampat.

Another important goal for the coming year is to strengthen cooperation with BLUD, the authority responsible for overseeing and managing the Raja Ampat Marine Park. Increased collaboration will support the reporting and management of COTS sightings and control efforts, while also assisting the development of regulations and management strategies that may directly or indirectly reduce the risk of COTS population growth and expansion throughout Raja Ampat.

Through the continuation of monitoring and control activities, the COTS Monitoring and Control Project aims to improve understanding of COTS ecology and distribution in the Raja Ampat region, where information remains limited. Long-term monitoring efforts will be essential for understanding spatial distribution patterns, identifying areas most affected by COTS, and determining reproductive cycles and spawning periods. This knowledge will contribute to the protection and long-term sustainability of Raja Ampat's coral reef ecosystems.





Manta Ray Project Report

INTRODUCTION

The Raja Ampat archipelago, West Papua, Indonesia, is home to large populations of both reef manta rays (*Mobula alfredi*) and oceanic manta rays (*Mobula birostris*). Over a six-year study period, 588 individual oceanic manta rays have been identified, while long-term monitoring over 15 years has documented 1,375 individual reef manta rays (Beale et al. 2018; Setyawan et al. 2020). Protection for manta rays in Raja Ampat began in 2007 with the implementation of marine protected area (MPA) networks, followed by the declaration of all regency waters as Southeast Asia's first shark and ray sanctuary in 2012, and full national-level protection for both species in 2014 (Decree No. 4/KEPMEN-KP/2014; Setyawan et al. 2022). As a result of these measures, together with the influence of ENSO events, estimated *M. alfredi* population sizes have increased significantly over the past decade in both the Dampier Strait and Southeast Misool (Setyawan et al. 2022).

Within the Dampier Strait, the patch reefs east of Arborek Island represent an important habitat for reef manta rays, where feeding, cleaning, and mating activities are regularly observed. The largest aggregation was recorded in early 2019 at Manta Ridge, when 112 individual manta rays were observed chain feeding, surface feeding, and visiting cleaning stations over several hours (Setyawan et al. 2020). Seasonal peaks in *M. alfredi* aggregations occur during the northwest monsoon (November to March), coinciding with the reversal of the South Equatorial Current (SEC) and the inflow of warm waters into Raja Ampat from the Halmahera and Seram Seas. During this period, the eastward surface flow of the SEC, combined with strong tidal currents in the Dampier Strait, generates local eddies that concentrate zooplankton and support surface-feeding aggregations of reef manta rays (Mangubhai et al. 2012; Setyawan et al. 2020).

According to the IUCN Red List of Threatened Species, the reef manta ray is listed as Vulnerable (VU), while the oceanic manta ray is listed as Endangered (EN) (Marshall et al. 2022), meaning their global populations are decreasing. Manta rays are threatened by targeted commercial fishing for their gill plates used in traditional Chinese medicine (TCM), fisheries bycatch, plastic pollution, entanglement, boat strikes, and unsustainable tourism practices (O'Malley et al. 2017; Germanov et al. 2019; Fernando and Stewart, 2021; Gómez-García et al. 2021; Strike et al. 2022). Their K-selected life history makes them highly vulnerable to population declines (Marshall and Bennett, 2010; Dulvy et al. 2014). In Raja Ampat, the rapid growth of tourism has become a major threat to manta rays.

According to the Statistics of Raja Ampat Regency, tourist numbers have increased drastically over the last five years, with 33,277 visitors recorded in 2024.

In the Dampier Strait, Manta Sandy and Manta Ridge, both manta ray cleaning stations, have implemented standard operating procedures (SOPs) and codes of conduct for interactions with manta rays for both divers and snorkelers. These regulations were established in 2017 through collaboration between government and non-government stakeholders. Key regulations include mandatory reservations via the WhatsApp group "Booking Manta Sandy" at least three days prior to arrival; a maximum of four speedboats allowed at one time; limits of 20 SCUBA divers in the water at Manta Sandy and 10 at Manta Ridge at any one time; and a strict prohibition on active interactions with manta rays, such as chasing, touching, blocking their pathways, crossing the line, or entering the cleaning station area. However, despite the established regulations, irresponsible tourism practices and behavioural misconduct at cleaning stations remain ongoing concerns (Fig. 33), which may drive them away from their aggregation areas.

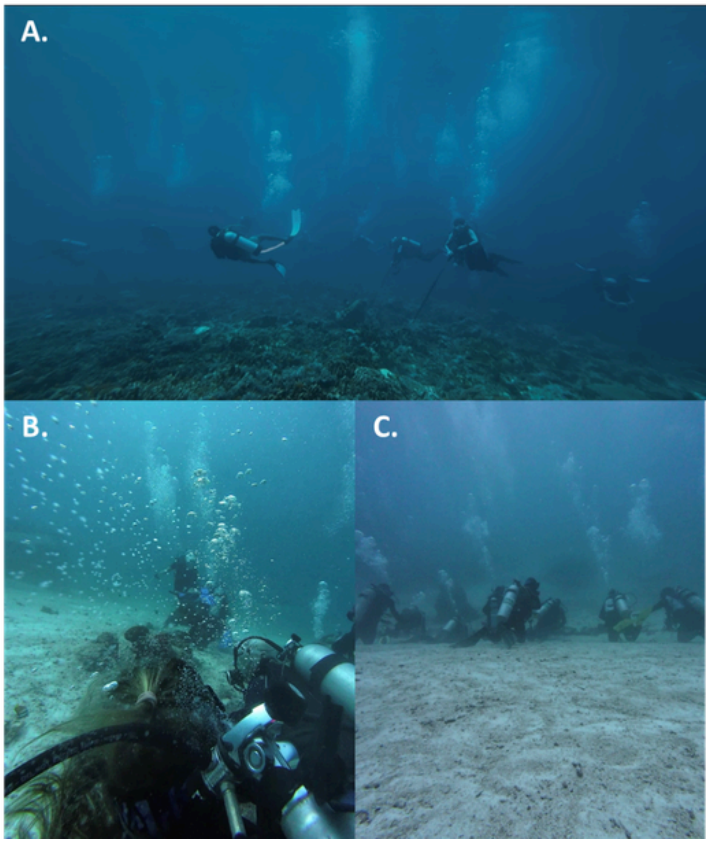


Figure 33. *Irresponsible tourism practices and behavioural misconduct at cleaning stations. (A) The number of divers exceeded the maximum limit at Manta Ridge; (B) Multiple divers were seen deploying reef hooks within the cleaning station at Manta Sandy; (C) Diver numbers at Manta Sandy reached up to 30 at one time*

The Manta Ray Project at Barefoot Conservation was established to support the long-term monitoring of both reef manta rays and oceanic manta rays in the Dampier Strait, Raja Ampat. Monitoring is conducted through systematic field surveys that document all manta ray sightings, with photo-ID integrated as a core component to enable individual identification and tracking. These combined datasets are used to assess population structure, site fidelity, and population demographics over time. The ID catalogue currently comprises 386 identified individuals and continues to expand through ongoing monitoring efforts. All records are submitted to Elasmobranch Institute Indonesia to support their broader research initiatives. The project also delivers a manta talk to volunteers, followed by a manta ID workshop where they are trained in photo-ID techniques. After completing the workshop, volunteers assist in data collection alongside the manta scientist during field surveys. Manta presentations are also delivered aboard the Ratu Laut liveaboard to promote responsible interactions in line with the Manta Trust's Code of Conduct.

METHODS

Surveys were conducted at two key manta ray cleaning stations within the Dampier Strait: Manta Sandy and Manta Ridge. Additional surveys were undertaken along the sloping reefs surrounding Arborek Island, including West Arborek and East Arborek (Arborek Tip), which have been observed to function as manta ray feeding grounds. Occasional sightings were also recorded during other field surveys, recreational dives, and opportunistic encounters at Barefoot Jetty, JPP, Main Jetty, Isaias Garden, Juan's Reef, and Lalosi (Fig. 34).

The study was conducted over a one-year period (January–December 2025), with surveys carried out on a weekly basis using SCUBA diving or snorkelling, and occasional drone surveys. During each encounter with *M. alfredi*, an attempt was made to collect a photograph of the ventral surface for photo-identification. However, due to environmental conditions (e.g., low visibility, strong currents) or short encounter durations, photo-ID images were not always successfully obtained. In such cases, other observational data were recorded and entered into a Microsoft Excel spreadsheet.

When a clear image was available, the distinctive and unique patterns of spots, patches, and shading on the ventral surface (Fig. 35), along with other identifying features such as damaged pectoral fins, missing tails, or scars, were used to distinguish individuals (Marshall et al. 2011; Gómez-García et al. 2021). Sex was determined by the presence (males) or absence (females) of claspers on the pelvic fins. Female sexual maturity was assessed based on the presence of mating scars or visible pregnancy, while male maturity was estimated from the length and extent of clasper calcification and the development of the clasper glands, as described by Marshall and Bennett (2010). Additional metadata for each sighting included date, time, location, tidal condition, colour morph, estimated size (disc width), injuries, and observed behaviours (feeding, cleaning, cruising, or courtship). Tourism pressure at cleaning stations was also monitored. The number of divers, speedboats, and liveaboards present during each survey was recorded, along with any observed behavioural misconduct.



Figure 34. *Location of survey sites. Red points indicate cleaning stations, yellow points indicate feeding sites that are regularly surveyed, and green points indicate locations of opportunistic sightings*

Photo-ID images were manually matched to the existing ID catalogue to determine whether the individual was resighted or newly sighted. Newly sighted individuals were assigned a unique identification number and added to the project database. The photographer who obtained the photo-ID was given the opportunity to name the individual.

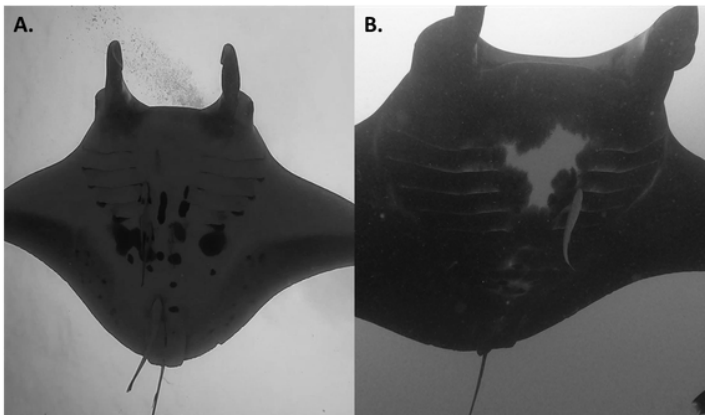


Figure 35. *Manta rays are photographed during surveys to capture their unique ventral spot patterns used for individual identification. (A–B) Ventral surfaces of different colour morphs: (A) Chevron; (B) Melanistic*

RESULTS

A total of 272 manta ray sightings were recorded throughout the year, consisting of *Mobula alfredi* (n = 264; 97%) and *Mobula birostris* (n = 8; 3%). *M. alfredi* was present year-round, with the lowest numbers recorded in September (n = 3) and October (n = 2)(Fig. 36). Sightings increased from November onwards, reaching a peak in December (n = 118). The December peak coincided with a period of intensified survey effort, including more frequent dives and the addition of drone surveys, which may have influenced the overall number of sightings. In contrast, *M. birostris* was recorded infrequently, with occasional sightings during January–March and December.

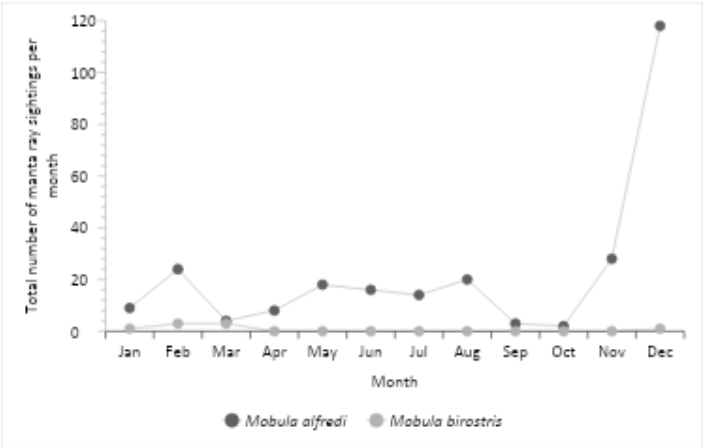


Figure 36. *Total number of manta ray sightings of both Mobula alfredi and Mobula birostris recorded across months in 2025*

Manta ray observations were recorded across multiple sites. Among these, West Arborek made up 33.1% (n = 90) of the total sightings, followed by Manta Ridge, which contributed a further 20.2% (n = 55). The largest aggregation was documented in December, during which 39 *M. alfredi* were observed surface feeding for several hours at West Arborek (Fig. 37). At Manta Ridge, aggregations of *M. alfredi* were also routinely observed during the peak season, particularly on falling tides. *M. birostris* were recorded at Blue Magic, Manta Ridge, and West Arborek. At Blue Magic, individuals were observed cleaning at the top of the seamount, which functions as a cleaning station for oceanic manta rays. This observation aligns with findings by Setyawan et al. (2024), who reported that oceanic manta rays in Raja Ampat use cleaning stations as intensively as reef manta rays. Blue Magic is located close to the mouth of the Dampier Strait, where depths exceed 1,000 m. Such proximity to deep waters may facilitate oceanic manta rays' access to off-shelf feeding grounds during nighttime (Setyawan et al. 2024). In contrast, *M. birostris* were mainly observed cruising at Manta Ridge and West Arborek.

It was possible to determine the sex in 62.5% of *M. alfredi* sightings (n = 165), while 37.5% (n = 99) remained of unknown sex (Table 5). Of the total sightings, 42% (n = 111) were identified as female and 20.5% (n = 54) as male. In terms of colour morph, 52.3% (n = 138) of sightings were classified as chevron, whereas 47.7% (n = 126) were melanistic. A higher proportion of male sightings were chevron (64.8%; n = 35), while sightings of unknown sex were predominantly melanistic (54.5%; n = 54). Female sightings were more evenly distributed between colour morphs, with chevron slightly more common than melanistic (52.3% and 47.7%, respectively). The higher proportion of melanistic sightings within the unknown-sex category may reflect observational bias, as the darker ventral coloration can make claspers more difficult to identify, particularly under low-visibility conditions.

In 2025, 111 individuals of *M. alfredi* were photo-identified across Manta Sandy, Manta Ridge, and Arborek Island, representing 42.0% of the 264 total *M. alfredi* sightings recorded. Of these, 32 were newly identified, whereas 79 were resighted from previous records. Fifteen individuals were classified as the most frequently sighted across the multi-year dataset (2022–2025) and are presented in Table 6. All individuals listed in Table 6 were females, comprising both melanistic (M) and chevron (C) colour morphs. The highest cumulative number of sightings was recorded for individuals MR-0324 (F, M) and MR-0298 (F, C), each with 10 sightings over the four-year period.



Figure 37. *Mobula alfredi feeding aggregation observed at West Arborek in December 2025 (photograph by Riccardo Zini)*

Sex	Number of Sightings	Chevron	Melanistic
Female	111 (42%)	58 (52.3%)	53 (47.7%)
Male	54 (20.5%)	35 (64.8%)	19 (35.2%)
Unknown	99 (37.5%)	45 (45.5%)	54 (54.5%)
Total	264	138 (52.3%)	126 (47.7%)

Table 5. *Sighting records of reef manta rays (Mobula alfredi) observed across study sites in 2025, showing the distribution by sex and colour morph*

MR-0324 (F, M), MR-0010 (F, C), and MR-0137 (F, M) were resighted consistently from 2022 to 2025, indicating continuous annual presence throughout the study period. MR-0291 (F, C) was resighted seven times in 2023 but only once in 2025. Similarly, MR-0064 (F, M) was resighted six times in 2023, with fewer resightings in other years. MR-0226 (F, M) was recorded exclusively in 2025, with four resightings and no records during the 2022–2024 period. Although survey effort was by no means constant across years, multi-year resightings were nevertheless recorded for a subset of individuals.

FUTURE DIRECTIONS

In 2026, the project will strengthen its collaboration with the Elasmobranch Institute Indonesia by sharing photo-ID and sightings data, as well as integrating the existing project database with their broader dataset. Survey efforts will also be intensified, especially during the high season (November–March). In addition, the project plans to initiate the use of drones as a complementary survey method, which may help document manta ray sightings and aggregation sizes that may not be fully captured during in-water observations.

Manta ray presentations will continue to be delivered to the Ratu Laut liveaboard, with efforts to expand to other liveaboard operators. These sessions aim to increase awareness and education about manta rays and promote the code of conduct for responsible interactions, while also encouraging guests to share photo-ID images and video footage from manta ray encounters during their trips.

ID (Sex, Colour Morph)	Total number of sightings per year				Total
	2022	2023	2024	2025	
MR-0324 (F, M)	2	5	1	2	10
MR-0298 (F, C)		5	1	4	10
MR-0064 (F, M)	1	6		2	9
MR-0291 (F, C)		7		1	8
MR-0010 (F, C)	1	4	1	1	7
MR-0013 (F, C)		4	2	1	7
MR-0137 (F, M)	2	3	1	1	7
MR-0067 (F, M)	1	5			6
MR-0005 (F, C)		3		3	6
MR-0026 (F, M)	1	2		2	5
MR-0309 (F, C)		3		2	5
MR-0208 (F, C)		4	1		5
MR-0009 (F, M)		2	1	2	5
MR-0078 (F, C)	1	2		2	5
MR-0226 (F, M)				4	4

Table 6. *Annual sightings of the most frequently photo-identified Mobula alfredi individuals recorded across Manta Sandy, Manta Ridge, and Arborek Island (2022–2025)*

Reef Restoration Project Report



INTRODUCTION

Raja Ampat is considered one of the world's most important coral reef systems, hosting over 500 coral species, which represent nearly 75% of all known coral species. On Arborek Island, coral reefs are a key source of income for the local community through tourism and fishing, thereby supporting local livelihoods. However, these reefs have experienced multiple pressures, including historical blast fishing, anchor damage, and crown-of-thorns starfish (COTS) outbreaks. More recently, the rapid growth of tourism has added further pressure on the reef ecosystem, including the construction of homestays, unsustainable diving practices, and poor wastewater management. As a result, the condition of coral reefs in Arborek, particularly at frequently visited dive sites, has been declining over the past three years, with a consistent increase in rubble cover (see Barefoot Science Report, 2024). These ongoing declines highlight the urgent need for coral restoration efforts to facilitate reef recovery and support long-term ecosystem sustainability.

The restoration project was established in late 2021 and continues to be carried out with three major aims: (1) to restore an acutely degraded section of the Arborek reef that was not naturally recovering; (2) to educate and engage local children about corals and how to preserve them; and (3) to improve restoration science best practices by undertaking research on different techniques and addressing key knowledge gaps. The initial phases of the project showed significant progress. In 2022, corals in rope nurseries reached an overall survival rate of 85%. After 12 months of growth, 2023 marked the transition to the outplanting phase, during which over 500 corals were transplanted onto the reef. Approximately 70% of these corals successfully attached and survived well, with less than 5% mortality and minimal signs of disease or predation, although around 25% became dislodged. Over the following years, 10 stabilization frames and 8 coral ropes were deployed to stabilize the loose substrate. Together, these structures supported coral growth, promoted natural settlement, and reduced rubble from falling down the slope.

Despite these positive outcomes, coral bleaching emerged as a threat in Raja Ampat. In response, restoration efforts were largely paused in 2024 to avoid additional stress, with the focus shifting towards maintaining nursery corals and monitoring the progress of the restoration work. Monitoring showed steady improvement over time, with continued growth of transplanted corals on the frames. In one case, coral cover reached 86%, nearly covering the entire structure. At the same time, the rubble beneath the frames showed signs of consolidation, supporting the project's aim of stabilizing loose substrate and promoting reef recovery (Barefoot Science Report, 2024).

In 2025, several adjustments were made to the design of the stabilization frames. One of the main issues observed was the presence of coral-killing sponges (CKS), which overgrew several coral colonies, particularly on the frames. This led to the hypothesis that coral tissue loss caused by predation from Drupella snails may have encouraged sponge growth by creating conditions that favour their expansion. In response, targeted removal of CKS was carried out to prevent further overgrowth on neighbouring coral colonies, and Drupella snails were removed to reduce ongoing coral mortality. Although CKS appeared to preferentially affect certain coral species, initial monitoring was conducted to assess potential differences in survival and resistance among species. However, further monitoring is required to confirm these patterns and to better inform future restoration strategies.

As part of the third objective, a collaborative research project was conducted with a student from the University of Papua to compare the growth rates of three *Acropora* species across two nursery methods: hexagonal structures (commonly known as "coral spiders") and rope nurseries. The study aimed to assess both the effectiveness of the different methods and species-specific growth performance. Overall, this year helped shape a clearer and more practical approach to coral propagation and outplanting moving forward.

METHODS

To support the continued success of the restoration efforts, aerial imagery was used to track changes in substrate cover and evaluate the effectiveness of the stabilization frames at four key time points: November 2024, March 2025, July 2025, and November 2025. Photos were taken using an Olympus TG-7 camera and analysed using Coral Point Count with Excel (CPCE) software. A total of 50 random points were specified for each image, and all frame photos were analysed at each time point. Percent cover of hard coral, dead coral, and rubble was then averaged across all frames. The classification of coral health status was determined based on the categories established by the Coral Reef Rehabilitation and Management Program – Coral Triangle Initiative (COREMAP–CTI, 2017), as follows: live coral cover < 19% (low), 19% ≤ live coral cover ≤ 35% (moderate), and live coral cover > 35% (high) (Giyanto et al. 2017).

To evaluate differences in growth and survival rates across nursery methods and species, fragments of three *Acropora* species (*Acropora parilis, Acropora yongei*, and an unidentified *Acropora sp.*) were used. Fragments of each species were attached to one spider and one rope, resulting in three spiders and three ropes in total, with 12 fragments attached to each structure.

The coral spiders were constructed using a steel rebar frame, coated with a mixture of epoxy resin and hardener, followed by a layer of sand. This coating process was repeated twice to enhance durability, and the structures were then deployed at a depth of 8m. For the coral rope setup, polypropylene ropes were used, with coral fragments attached at 30cm intervals and suspended horizontally in the water column. Coral fragments were measured weekly from the base to the furthest branch tip over a two-month period (September–October 2025) using calipers. Survival and growth rates were calculated following Ricker (1975) and Effendi (1997), respectively.

Nursery maintenance was carried out regularly to provide optimal conditions for coral growth and survival. Cleaning was conducted one to two times per week and involved the removal of dead tissue, predators such as Drupella snails and COTS, as well as biofouling organisms and competitors including algae, sponges, *Xenia* soft coral, fire coral, hydroids, tunicates, and cyanobacteria. Special attention was given to the removal of CKS from affected colonies. Sponges were carefully removed using pliers, collected in mesh bags, and transported to land for burial. This was done to prevent reintroduction or further spread in the reef, where they could otherwise continue to overgrow and damage coral colonies.

RESULTS

Figure 38 shows dynamic changes in benthic cover on stabilization frames over time. Average hard coral cover across all frames increased from 56.7% in November 2024 to 59.6% in March 2025, before declining to 55.4% in July 2025 and further to 51.7% in November 2025. This decline was followed by a consistent increase in dead coral cover, from 2.9% in November 2024 to 11.3% in November 2025. Rubble cover showed a fluctuating pattern, decreasing from 25.0% in November 2024 to 19.2% in March 2025 and further to 17.1% in July 2025, before increasing again to 25.8% in November 2025. Although hard coral cover remains classified as being in “high” condition, the increasing proportion of dead coral indicates ongoing mortality. This mortality is likely lined to predation by Drupella snails, which weakens coral colonies by

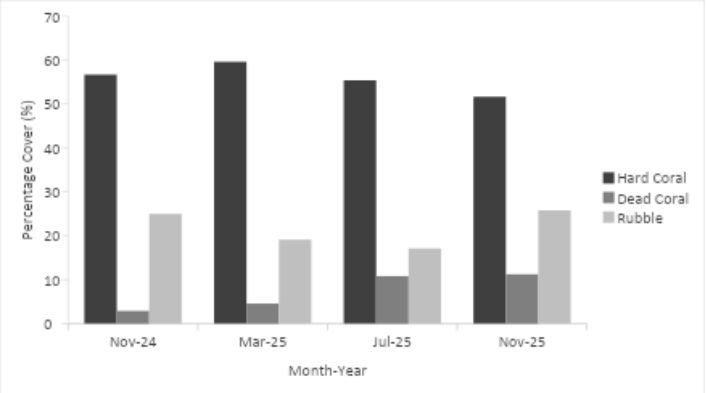


Figure 38. *Temporal variation in hard coral, dead coral, and rubble cover on stabilization frames during the study period*

removing tissue at their base. Although these predators can be removed promptly when found, this predation leaves behind tissue loss that is highly susceptible to colonization by encrusting sponges, which is consistent with Aerts (2000), who reported that tissue damage, such as lesions, can increase coral susceptibility to such overgrowth. Encrusting coral-killing sponges at the restoration site subsequently overgrow these areas and expand onto live coral tissue (Fig. 39). In some cases, these sponges develop branching extensions that allow them to reach and overgrow adjacent coral colonies. This is becoming a concern, as controlling sponge overgrowth is more difficult than removing mobile predators. To address this issue, some affected coral colonies were removed. Where parts of the colony, particularly the branch tips, had not yet been overgrown by sponges, these sections were cut and transferred to rope nurseries to support continued growth rather than being entirely removed.

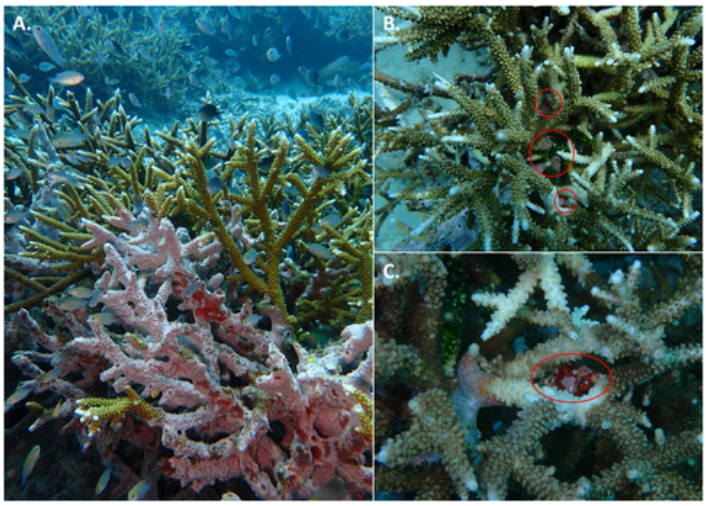


Figure 39. *Overgrowth of encrusting coral-killing sponges on Acropora colonies in stabilization frames, following tissue loss caused by Drupella predation. (A) Half of the colony is overgrown by sponges; (B) Drupella snails feeding on live coral tissue; (C) Sponges quickly colonizing areas of recent tissue loss*

The increase in rubble cover in November 2025 is likely attributed to the removal of dead coral from the frames, which subsequently accumulated as rubble on the substrate. In addition, some of the coral ropes deployed to stabilize loose substrate provided less stability when placed on the steepest parts of the sloping reef, where strong currents, particularly during full moon periods, caused sediment resuspension and movement of the structures. This reduced their ability to secure loose substrate, further contributing to rubble accumulation. Additionally, we suggest that closer spacing of frames may provide better support as corals grow, reducing the risk of collapse. Based on this, additional stabilization frames will be deployed in the coming year and positioned more closely together to fill gaps between existing frames, allowing neighbouring colonies to fuse and form a more stable, continuous framework.

Figure 40 shows that coral spiders outperformed coral ropes in both growth and survival. Coral spiders had a higher growth rate (0.13 cm/2 months) than coral ropes (0.04 cm/2 months). Coral fragments on spiders showed consistent growth over time, with a survival rate of 97% (35 out of 36 fragments survived). This may be due to the more rigid and stable structure of coral spiders, which keeps fragments relatively stationary and reduces physical stress. In contrast, coral ropes showed a slightly lower survival rate (92%), with three fragments detached and some broken. The ropes were installed in areas exposed to stronger currents, which likely contributed to these losses. Fragments that were not securely attached may have loosened over time and eventually detached, which may have contributed to the lower growth rate.

The growth rates of coral fragments showed clear differences between nursery methods (coral spiders and coral ropes) as well as among species. *Acropora parilis* showed higher growth on coral spiders (0.17 cm/2 months) than on coral ropes (0.06 cm/2 months), which suggests better performance on more stable substrates. *Acropora yongei* showed similar growth rates across both methods (0.15 cm/2 months), indicating a consistent growth response regardless of substrate type. This pattern aligns with prior studies (Anggara et al. 2022; Tangian et al. 2024), that report relatively stable growth of this species under different environmental conditions. A different trend was observed in *Acropora sp.*, where growth was positive on coral spiders (0.07 cm/2 months) but negative on coral ropes (-0.10 cm/2 months). This indicates higher sensitivity to environmental factors, especially in areas exposed to stronger currents. The decline on ropes likely reflects some level of tissue loss or stress, possibly linked to less stable attachment and continued fragment movement. However, it is also important to note that the potential influence of rope material on coral growth remains unclear, and this will require further investigation in the coming years.

Overall, this study highlights how species respond differently to substrate stability and environmental conditions. *Acropora yongei* appears the most consistent, while *Acropora parilis* and *Acropora sp.* are more affected by stability. This provides useful guidance for species selection in coral nurseries and suggests that more stable structures may be better suited for movement-sensitive species.

FUTURE DIRECTIONS

In 2026, further studies will focus on coral-killing sponges in our restoration site. This will include collecting and sending sponge samples to a laboratory for species identification, as well as monitoring their growth rates and substrate preferences, particularly which coral species are most susceptible. In addition, coral species that are less preferred as substrates by these sponges and more tolerant of environmental pressures in the restoration area will be selected for propagation in the nursery. Coral fragments currently maintained on nursery ropes will be outplanted onto new frames. These frames will be strategically placed in adjacent areas between existing structures to help stabilize the substrate and prevent rubble from falling down the slope. More frequent removal of Drupella snails will also be carried out, as they are a major coral predator within the restoration site.

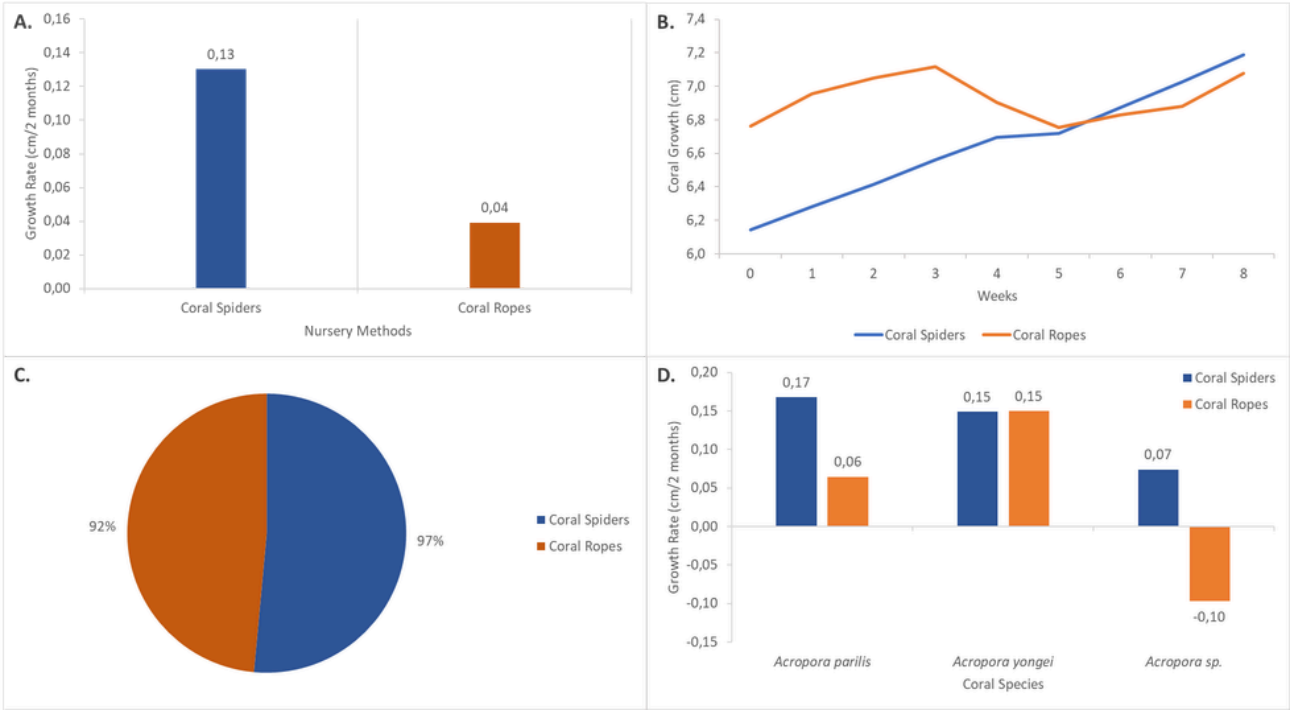


Figure 40. *Comparison of growth rates of three Acropora species (Acropora parilis, Acropora yongei, and an unidentified Acropora sp.) across two nursery methods: coral spiders dan coral ropes. (A) Fragment growth rates under each method; (B) Average weekly growth over the observation period; (C) Fragment survival rates under each method; (D) Species-specific growth rates across both methods*

Reef Check Project Report



INTRODUCTION

Long-term reef monitoring around Arborek Island has been conducted since 2016, highlighting the importance of tracking changes in coral reef health over time. The Reef Check methodology has been used to assess coral reef ecosystem health by focusing on key indicators such as fish abundance, benthic coverage, and invertebrate abundance. The fish and invertebrates indicators were selected based on their ecological and economic value, as well as their sensitivity to anthropogenic impacts. To support this initiative, volunteer divers have been trained to collect data on coral reef health, forming teams of citizen scientists. Their involvement plays a crucial role in expanding the scope and frequency of monitoring efforts, enabling the collection of large-scale datasets over time.

Nine core survey sites and four additional sites were selected to represent a range of reef habitats and levels of anthropogenic impact. Barefoot Jetty and Main Jetty were designated as “impacted sites” due to their direct and sustained pressures associated with human activities such as recreational diving, tourism, and boat traffic. In 2025, these sites were monitored at least once every three months. The impacted sites were surveyed at least four times throughout the year and at two depths. The data collected are reported annually to local policy makers in Waisai, the Raja Ampat Marine Park Authority and Management Unit (BLUD UPTD), and Reef Check Indonesia, contributing to informed decision-making and the sustainable management of coral reef ecosystems.

METHODS

A 100m line transect was deployed along the reef and divided into four 20m segments, separated by 5m gaps, following the Reef Check methodology. The gaps provide separation between survey segments, allowing them to be treated as independent replicates and improving the consistency and comparability of data collection. The Belt Transect method was used for fish survey, covering a 5m wide area, extending 2.5m on each side of the transect line and 5m above the substrate. The survey targeted nine indicator fish groups selected based on their ecological importance and sensitivity to human impacts. Following the fish survey, benthic cover composition was assessed using the Point Intercept Transect (PIT) method. Observers recorded the substrate type directly beneath the transect line at 0.5m intervals. Accounting for the 5m gaps between each 20m segment, a total of 160 data points were collected per 100m line transect. Following the substrate survey, an invertebrate survey was conducted using a belt transect with the same width as the fish survey. Divers swam in an S-shaped pattern within the transect area to search for and record the nine targeted indicator invertebrates.

RESULTS

1. Temporal variation in fish abundance at impacted sites in 2024 and 2025

Figure 41 presents fish abundance data collected at Barefoot Jetty in 2024 and 2025, while Figure 42 presents fish abundance data collected at Main Jetty over the same period. At Barefoot Jetty, the mean abundance of butterflyfish, an important indicator of coral reef health, increased from 3.84 in 2024 to 4.19 in 2025. This increase may reflect the higher hard coral cover recorded in 2025 (Fig. 45), as hard corals provide an important food source for obligate corallivorous fish such as butterflyfish. As butterflyfish are not typically targeted by fisheries, changes in their abundance are more likely to reflect habitat condition than fishing pressure. Parrotfish abundance also increased from 2.63 in 2024 to 4.13 in 2025. Similarly, grouper abundance increased from 0.78 in 2024 to 1.25 in 2025, while snapper abundance increased from 0.97 to 1.38 over the same period. Bumphead parrotfish, which were not recorded in 2024, were observed at a mean abundance of 0.66 in 2025. In contrast, the mean abundance of Haemulidae declined from 1.00 in 2024 to 0.63 in 2025.

At Main Jetty, butterflyfish abundance increased, with mean abundance rising from 3.88 in 2024 to 6.81 in 2025. This increase may reflect the higher hard coral cover recorded in 2025 (Fig. 46), as hard corals provide an important food source for butterflyfish.

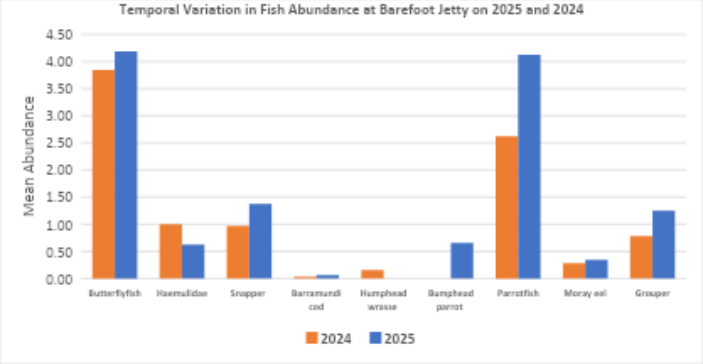


Figure 41. *Temporal Variation in Fish Abundance at Barefoot Jetty from 2024-2025*

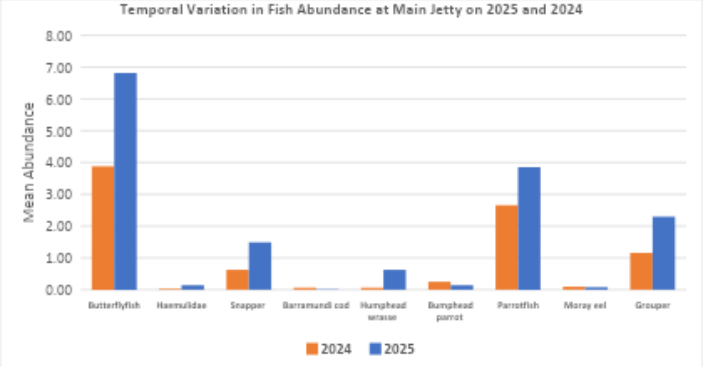


Figure 42. *Temporal Variation in Fish Abundance at Main Jetty from 2024-2025*

Parrotfish were the second most abundant indicator group at Main Jetty, with a mean abundance of 3.86 in 2025, up from 2.66 in 2024. Snapper and grouper also showed notable increases in abundance, with snapper increasing from 0.63 in 2024 to 1.49 in 2025 and grouper increasing from 1.16 to 2.30 over the same period. Groupers are typically associated with structurally complex reef habitats, using coral formations and reef crevices for shelter and hunting. The increase in grouper abundance may reflect changes in habitat structure and coral cover at the site, although additional data would be required to confirm this relationship. In contrast, bumphead parrotfish abundance declined from 0.25 in 2024 to 0.14 in 2025, while humphead wrasse abundance increased from 0.06 to 0.63 over the same period. The relatively low abundance of both species is not unexpected, as bumphead parrotfish are listed as Vulnerable and humphead wrasse as Endangered on the International Union for Conservation of Nature (IUCN) Red List. Their low numbers may reflect the continued pressures faced by these species, including habitat degradation and overfishing.

2. Temporal variation in invertebrates abundance at impacted sites in 2024 and 2025

Figures 43 and 44 show trends in the abundance of selected invertebrate indicator groups across two impacted sites between 2024 and 2025. At Barefoot Jetty, the mean abundance of Diadema sea urchins increased from 0.47 in 2024 to 0.72 in 2025. In contrast, other sea urchin groups, including pencil sea urchins and collector sea urchins, were not recorded during the 2025 survey period. Giant clam abundance also increased, with mean abundance rising from 0.38 in 2024 to 0.53 in 2025. As filter feeders, giant clams play an important role in maintaining water quality within reef ecosystems. Banded coral shrimp were not recorded during the 2025 survey period, while sea cucumber and lobster abundance remained unchanged between 2024 and 2025. The mean abundance of Crown-of-Thorns Starfish (COTS) declined from 0.09 in 2024 to 0.06 in 2025. This reduction coincided with the implementation of COTS culling efforts during 2025. The decline in COTS abundance may have reduced predation pressure on corals, potentially contributing to the increase in hard coral cover observed at the site. However, additional monitoring would be required to confirm this relationship.

At Main Jetty, the mean abundance of Diadema sea urchins declined from 0.06 in 2024 to 0.03 in 2025, while pencil sea urchins and collector sea urchins remained absent during both survey periods. This trend highlights the continued low abundance of sea urchins at the site, which may be of concern given

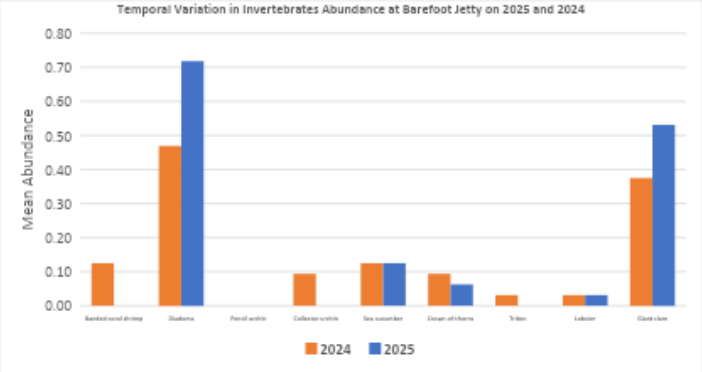


Figure 43. *Temporal Variation of Invertebrates Abundance at Barefoot Jetty from 2024-2025*

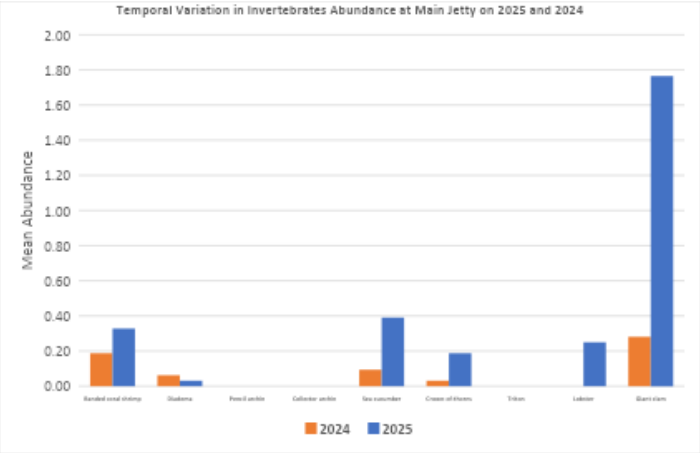


Figure 44. *Temporal Variation of Invertebrates Abundance at Main Jetty from 2024-2025*

their important ecological role in controlling algal growth on coral reefs.

In contrast, giant clam abundance increased substantially, rising from 0.28 in 2024 to 1.77 in 2025. Banded coral shrimp abundance also increased from 0.19 in 2024 to 0.33 in 2025, whereas at Barefoot Jetty this indicator group declined from 0.13 in 2024 to no recorded individuals in 2025. Lobsters, which were not recorded in 2024, were observed at a mean abundance of 0.25 in 2025. Similarly, sea cucumber abundance increased from 0.09 in 2024 to 0.39 in 2025. In contrast to Barefoot Jetty, the mean abundance of Crown-of-Thorns Starfish (COTS) increased from 0.03 in 2024 to 0.19 in 2025. The increase in CoTS abundance warrants continued monitoring, as elevated CoTS densities can increase predation pressure on corals and may contribute to declines in hard coral cover if populations continue to increase.

3. Proportion of benthic components at surveyed sites

Figure 45 shows the mean benthic cover recorded at Barefoot Jetty in 2024 and 2025. Hard coral cover increased from 17% in 2024 to 20% in 2025. Recently killed coral remained stable at 1%, while rock cover

declined from 13% in 2024 to 11% in 2025 and rubble cover decreased slightly from 30% to 29%. The reduction in rock and rubble cover may reflect the inclusion of restoration areas within the survey extent, where coral restoration activities have contributed to increases in hard coral cover.

Figure 46 shows the mean benthic cover recorded at Main Jetty in 2024 and 2025. Hard coral cover increased from 27% in 2024 to 28% in 2025. Recently killed coral cover also increased, from 1% to 2%, while rock cover declined from 10% to 9%. Rubble cover remained relatively stable between years. The increase in recently killed coral may indicate elevated coral mortality during the 2025 monitoring period and highlights the importance of continued monitoring at this site.

Figures 47 and 48 show the proportional composition of benthic categories at Barefoot Jetty and Main Jetty, respectively. Barefoot Jetty recorded the lowest hard coral cover, at 17% in 2024, while Main Jetty recorded the highest hard coral cover, at 28% in 2025. Soft coral cover at Barefoot Jetty declined from 23% in 2024 to 19% in 2025, whereas Main Jetty showed an increase from 8% to 10% over the same period.

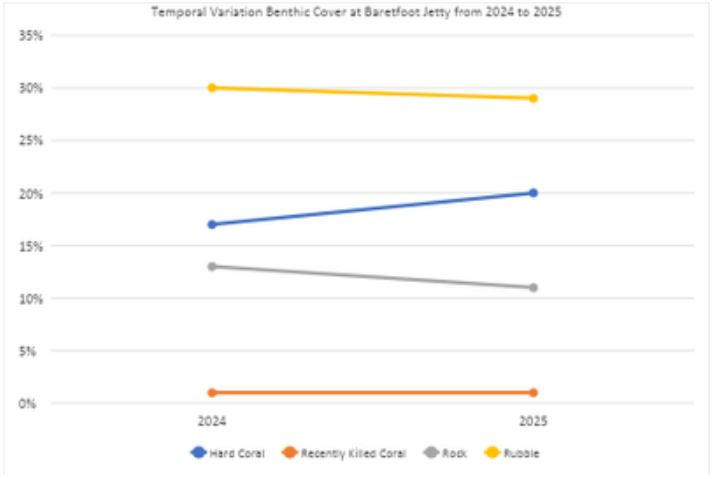


Figure 45. *Temporal Variation of Benthic Cover at Barefoot Jetty from 2024 to 2025*

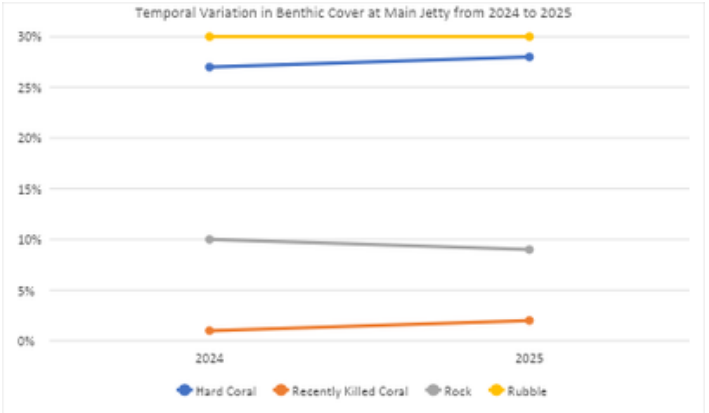


Figure 46. *Temporal Variation of Benthic Cover at Main Jetty from 2024 to 2025*

Both sites continued to be dominated by rubble, which accounted for 29% of benthic cover at Barefoot Jetty and 44% at Main Jetty in 2025.

FUTURE DIRECTIONS

Based on the data collected in 2025, the Reef Check Project will continue to focus on monitoring and assessing coral reef health throughout 2026, while also contributing to the reporting of Crown-of-Thorns Starfish (COTS) sightings to support early detection and management of increasing COTS populations. Observations from the monitoring program suggest that increasing tourism activity may place additional pressure on coral reef ecosystems over time. Continued monitoring will therefore focus on assessing trends in key indicator species, including species that have shown increases in abundance as well as those displaying signs of decline. Tracking these changes will help improve understanding of reef ecosystem condition and identify potential emerging threats. Efforts will also be made to ensure greater consistency in survey methodology. Surveys will continue to be conducted at the same locations using GPS coordinates to identify transect starting points. Where possible, permanent markers or stakes will be installed to maintain consistency in transect placement and improve the comparability of monitoring data over time.

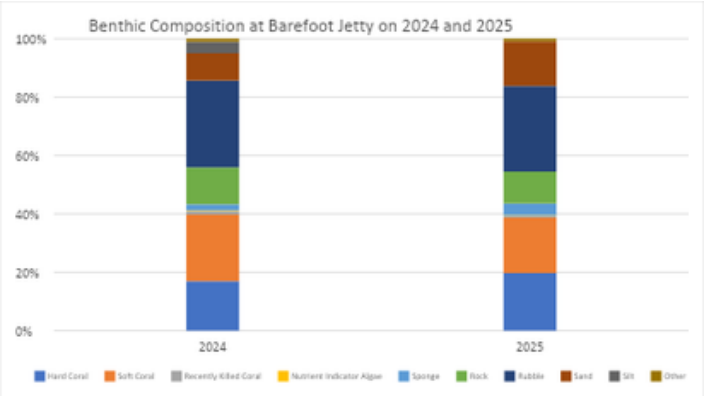


Figure 47. *Proportion of Benthic Composition at Barefoot Jetty from 2024 to 2025*

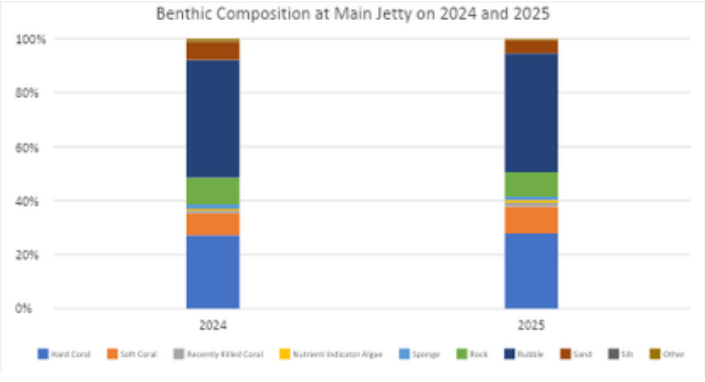
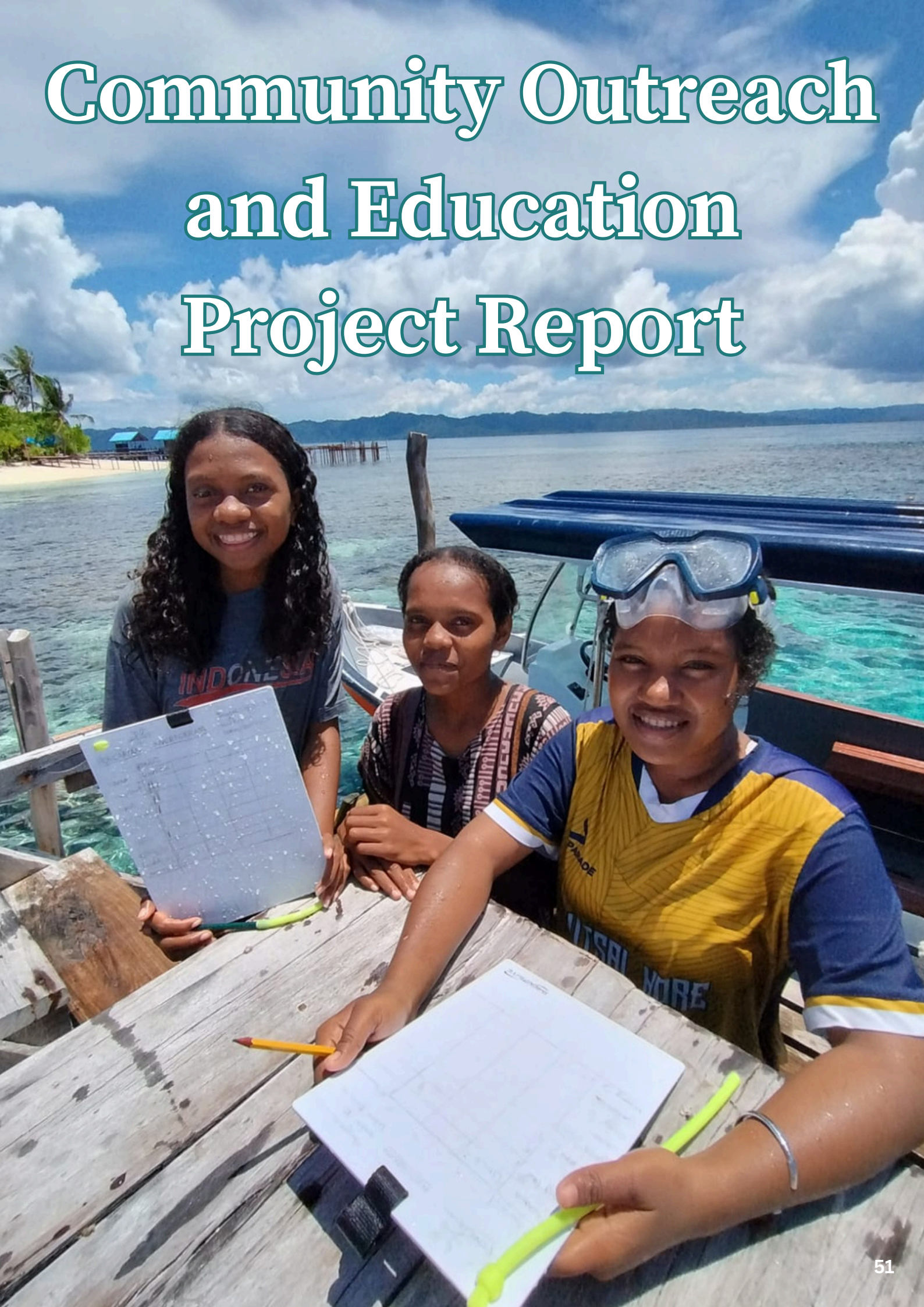


Figure 48. *Proportion of Benthic Composition at Main Jetty from 2024 to 2025*

Community Outreach and Education Project Report



TEACHING ENGLISH AT ARBOREK SCHOOL

Barefoot Conservation volunteers and Indonesian staff teach English at Arborek School every Thursday. The lessons focus on helping the children improve their English skills, which can create more opportunities for them in the future. The students are divided into three groups based on their grade levels: the Youngest Class (Grades 1–2), Middle Class (Grades 3–4), and Oldest Class (Grades 5–6). Before each lesson, volunteers and staff prepare a lesson plan that includes a recap of the previous lesson and an introduction to new topics. Classes are designed to be fun and interactive, incorporating games, songs, and group activities to keep students engaged. To support learning, volunteers use flashcards and visual aids, as well as real-life objects related to the lesson topic. This approach helps students connect new vocabulary with real objects, making lessons more engaging and easier to remember. Over time, the students have become more confident using English. Many can now have simple conversations with Barefoot volunteers, both during lessons and in everyday interactions around the village.



Figure 49. *Barefoot Conservation volunteers teaching at Arborek School*

WASTERWATER TREATMENT PROJECT

Wastewater management has become an increasing concern in Raja Ampat as tourism and homestay development continue to expand due to the potential impacts of untreated wastewater on surrounding marine ecosystems. Following local water quality assessments in 2024 and discussion among local stakeholders, Barefoot Conservation has been working with collaborators to develop a wastewater treatment system that is affordable, durable, low-maintenance and suitable for local conditions. The system is designed to break down waste and remove excess nutrients and pathogens before wastewater reaches the environment (Fig. 50). In July 2025, a large-scale prototype was installed at Barefoot Conservation (Fig. 51), and a smaller system is being implemented at Kalabia Homestay (Fig. 52). The system is functioning well, with healthy growth of vetiver grass indicating successful nutrient uptake within the treatment process.

To better understand the situation, surveys were conducted at homestays around Arborek Island, revealing that many still rely on older systems where wastewater is discharged directly into the ground, often close to the shoreline. Given the porous nature of coastal soils, untreated wastewater may eventually enter surrounding reef ecosystems. While challenges such as maintaining sufficient rainwater for flushing and storm impacts on vetiver grass propagation have been encountered, many homestay owners expressed interest in adopting a similar wastewater treatment system. Building on these early successes, Barefoot Conservation plans to support the wider rollout of wastewater treatment systems across Arborek in the coming years. Future work will also include water quality monitoring at multiple stages of the treatment process and within the marine environment to assess system effectiveness and measure environmental improvements following implementation.

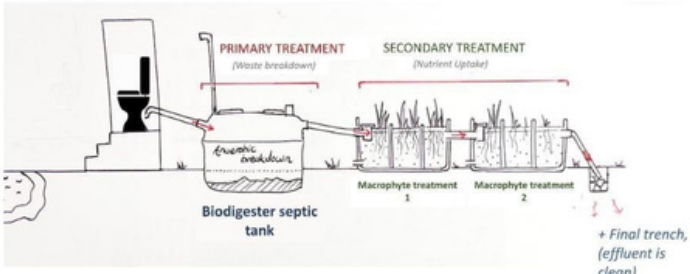


Figure 50. *Diagram of wasterwater treatment system installed at Barefoot Conservation, showing the sequence of treatment stages*



Figure 51. *Wastewater treatment system installed at Barefoot Conservation*



Figure 52. *Installation of reed bed tanks at Kalabia homestay on Arborek Island*

WASTEWATER MANAGEMENT MEETING WITH ENVIRONMENTAL AGENCY

In May 2025, Barefoot Conservation's Science Team attended a stakeholder meeting at the Raja Ampat Regency Environmental Agency Office in Waisai alongside local NGOs, government representatives, resort operators and members of the Homestay Association. The meeting focused on the current condition of Raja Ampat's coastal waters, with data presented by government scientists highlighting the increasing pollution from wastewater and marine debris, much of which is associated with tourism activities. The meeting also provided an opportunity to discuss innovative wastewater management solutions, including the design for a Eco-Toilet developed by Jack from Raja Ampat Eco Lodge and a modified version adapted by Josie Chandler from Barefoot Conservation for use in remote island communities. Barefoot Conservation shared progress on its pilot wastewater treatment project and offered support to homestay owners on Arborek Island interested in implementing similar systems to improve environmental sustainability and reduce impacts on surrounding marine ecosystems.

UNIVERSITY OF PAPUA STUDENT INTERNSHIP PROGRAM

From June to November 2025, Barefoot Conservation hosted two students from the University of Papua, Hilda and Margareth, to conduct their undergraduate research on Arborek Island. Hilda's research focused on comparing the growth rates of three *Acropora* species using Coral Spider and rope restoration methods, while Margareth studied the habitat characteristics and abundance of the walking shark (*Hemiscyllium freycineti*). Under the mentorship of our Science Team, the interns were involved in all stages of the research process, including study design, fieldwork, data analysis, and report writing. They also earned their Advanced Open Water Diver certification and gained practical experience through scientific diving activities, including coral bleaching surveys, COTS removals, and manta ray photo-identification. They also took part in community outreach initiatives, including English lessons at the local school and beach clean-up. This internship program reflects Barefoot Conservation's commitment to building local capacity in marine conservation and creating pathways for young Papuan scientists to gain practical experience in the field. We look forward to continuing this internship program in the years to come.



Figure 53. *Highlights from the University of Papua Student Internship*

OCEAN GUARDIANS DAY

In June 2025, Barefoot Conservation staff and interns from the University of Papua, Hilda and Margareth, organized an Ocean Guardians Day for the children of Arborek. The event aimed to introduce marine conservation concepts in a fun and engaging way, while helping children learn more about the ocean and marine life around them. The children were divided into small groups and rotated through four learning stations. At the first station, they learned about the differences between organic and non-organic waste and how they affect the ocean through a simple experiment. Using a basin filled with seawater, they observed how organic waste breaks down over time while non-organic waste remains in the environment for much longer. The second station challenged the children to match picture cards of marine animals with their correct English names. At the third station, they learned about hard and soft corals, the importance of healthy coral reef ecosystems, and the threats that reefs face. The session included a coral ID activity and a movement-based game. The final station introduced Reef Check target fish species and provided a land-based fish survey simulation, where they practiced identifying and counting fish along a belt transect and recording their observations on data slates. The day ended with team games, including tug of war and a sack race relay, followed by prize giving and a handprint mural titled "We Are the Ocean Guardians." It was a fun day of learning, teamwork, and discovering more about the marine environment around their island.



Figure 54. *Children participating in Ocean Guardians Day at Barefoot Conservation*

CORAL BLEACHING EDUCATION WITH LOCAL CHILDREN

In April 2025, Barefoot Conservation hosted a Coral Bleaching Event for the children of Arborek Island, with support from volunteers. The event aimed to increase understanding of global warming and its impacts on coral reefs, particularly coral bleaching. Participants were introduced to Reef Check survey methodology, including fish, substrate and invertebrate identification, as well as common marine species they could see during the dive. One of the interns also provided a brief introduction to how the cyanobacteria data is being collected and analysed. To conclude the event, staff and volunteers accompanied the children on a snorkeling session around the jetty, giving them the opportunity to observe their new knowledge on the reef.



Figure 55. *Children participating in Coral Bleaching Event at Barefoot Conservation*

COTS WORKSHOP WITH BLUD STAFF

In April 2025, Barefoot Conservation's Science Team met with BLUD and other key stakeholders in Raja Ampat at the BLUD Office in Waisai to discuss the increasing abundance of Crown of Thorns Starfish (COTS) and the importance of continued management efforts. The meeting aimed to raise awareness of the potential impacts of the COTS outbreak on coral reef ecosystems and to encourage collaboration between organisations involved in reef conservation. The science team shared the culling methods currently used by Barefoot Conservation, highlighting their effectiveness and safety compared to previously used approaches. In addition, discussions focused on the importance of collecting standardised data during culling to improve understanding of COTS populations and support more effective management strategies in the future.



Figure 56. *Barefoot Conservation staff hosting a COTS workshop with BLUD*

COTS WORKSHOP WITH RAJA AMPAT SEA CENTRE

In September 2025, Barefoot Conservation hosted a visit from the Raja Ampat Sea Centre to share knowledge and experience from our Crown of Thorns Starfish (COTS) management project. The science team began by providing an overview of the project, including the rationale behind the COTS control efforts and the culling methods currently used by Barefoot Conservation. Following the presentation, the teams travelled to Juan's Reef, one of the sites most affected by COTS, to conduct a practical training session. Participants had the opportunity to trial the culling methods and equipment while learning field protocols, safety procedures and data collection techniques used during monitoring and culling dives. To support the expansion of COTS management efforts in Raja Ampat, Barefoot Conservation provided Raja Ampat Sea Centre with two complete culling kits for use in their future conservation activities.



Figure 57. *Barefoot Science Staff conducting a COTS training workshop for Raja Ampat Sea Centre staff*

DISCOVER SCUBA DIVING WITH ARBOREK LOCALS

In 2025, Barefoot re-started it's Discover Scuba Diving (DSD) program to help encourage the young generation of Arborek to adventure under the water and begin to hone their scuba diving skills. The DSD program was done in conjunction with the Divemaster training program, assisted by our staff, to safely bring new divers to explore the underwater reefs in which they call home.

With each DSD experience, up to four young boys or girls, typically aged between fourteen and twenty, would begin with a briefing led by our Divemasters in training. After gearing up, the excited divers would head off for a skills session in shallow water on a one-to-one ratio with their Divemaster buddy. To finish the experience, the divers headed out to an exciting dive site around Arborek for their first real dive. Our new ocean warriors would always be all smiles as they encountered colourful reef fish, turtles and the occasional manta ray. A truly unforgettable experience for Arborek's next generation. Barefoot looks forward to continuing this program into 2026 and bringing more new divers into their ocean home.

We would like to thank all of our Divemasters in training for their fantastic contribution to the Discover Scuba Diving Program.





Figure 58. *Discover Scuba Dive training with Arborek Locals conducted by Barefoot staff and Divemasters in training*

Acknowledgements & Partners

Our special thanks goes out to, first & foremost, Simon Barden, for founding Barefoot Conservation back in 2012 and managing the organisation on a daily basis, jumping hurdles and seeking out exciting opportunities to increase our impact every step of the way.

Secondly, we would like to thank our Head of Science, Josie Chandler, for her inexhaustible drive to establish new and level up exciting science projects at Barefoot Conservation while involving the local community wherever possible, fueled by a passion for marine conservation which inspires the team every day.

None of the projects would have been possible without the Barefoot Conservation management team on site, led by our determined and relentlessly hardworking Head of Operations, Iris Uijttewaal. The team works long days every day to make sure data is being collected, science is moving forward and volunteers are being trained to a high quality standard. Thank you both current and previous team members (either voluntarily or on payroll) for your relentless dedication to marine conservation and community development the past year. Special thanks go to Matthew Perrodou for his tireless commitment to the Marine Debris Project and Ocean Warriors Programme. The local staff are the absolute backbone of Barefoot Conservations, without skippers, compressor operators, cooks and maintenance work Barefoot Conservation camp would simply not be operational. Thank you Pak Manto, Papa Ribka, Papa Klara, Sakarius, Maikel, Teni and Kelvin. We look forward to continue to conduct marine science research, collect data, expand community projects and education in 2026.

Partners involved are:

- The Regional Public Service Agency Regional Technical Implementing Unit (BLUD UPTD) in the Management of the Conservation Area (KKP) of the Raja Ampat Islands
- POKJA Manta - BLUD UPTD Raja Ampat, Konservasi Indonesia, Waisai
- Universitas Papua – Fakultas Perikanan dan Ilmu Kelautan
- NOAA Coral Reef Watch, Reef Check Indonesia, CoralWatch and Raja Ampat Manta Conservation Research
- Jack Burns and Dr Phil Dustan
- Child Aid Papua, Sawinggrai Village
- The Community of Arborek Island



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