# Whitsunday Reef Islands Initiative

Final Report

**Resilience-based mapping to support coral restoration in the Whitsundays**

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Yves-Marie Bozec[1], Devin Rowell[1], Luke Harrison[1], Johnny Gaskell, Karlo Hock[1], Dave Callaghan[2], Rebecca Gorton[3], Eva Kovacs[4], Mitchell Lyons[4], Peter J. Mumby[1], Chris Roelfsema[4].

[1] Marine Spatial Ecology Lab, School of Biological Sciences & ARC Centre of Excellence for Coral Reef Studies, The University of Queensland, Brisbane, QLD, Australia.

[2] School of Civil Engineering, The University of Queensland, Brisbane, QLD, Australia.

[3] CSIRO Oceans and Atmosphere, Hobart, TAS, Australia.

[4] Remote Sensing Research Centre, School of Earth and Environmental Sciences, The University of Queensland, Brisbane, QLD, Australia.

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Cover photograph (Johnny Gaskell): Looking over Lupton Island and Haslewood Island in the Whitsunday Group.

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| **Funding recipient information** | | |
|---|---|---|
| Organisation name: | The University of Queensland | |
| Address: | Brisbane QLD 4072 | |
| E-mail address: | sciencegrants@research.uq.edu.au | Mobile: 0733657594 |
| **Primary contact information (person who completed the report)** | | |
| Name: | Yves-Marie Bozec | |
| Position: | Research Fellow | |
| E-mail address: | y.bozec@uq.edu.au | |
| **If any contact information/key personnel has changed please provide details:** | | |
| | | |

# 1 Table of Contents

# 2 Executive Summary

The Reef Islands Initiative (RII) Whitsundays Program aims to develop a scientific, data-driven basis to support resilience-based management and guide activities of coral reef restoration in the Whitsunday Islands. To ensure that restoration activities are undertaken in the most optimal places, management decisions require baseline maps that integrate the current and historical distribution of reef habitats, their environmental regime and exposure to stress, as well as local connectivity patterns.

This report develops the underpinning environmental mapping and connectivity modelling that will act as a decision support tool to guide restoration activities within the RII Whitsundays Program. It provides high-resolution data on the physical forcing, bio-geochemical environment, ecological state and demographic connectivity of shallow coral reef habitats across the Whitsundays to help identify which sites may be suitable to the deployment of restoration activities.

A baseline map of ‘potential’ coral reef habitats was established at high-resolution (10 m) using an automated classification of satellite imagery, depth and wave exposure. Physical and biogeochemical drivers of coral populations were mapped at various resolutions (10–200 m) using the best available models (SWAN, eReefs, RECOM). Historical and recent (1993–2020) benthic data from conventional monitoring programs were collated and complemented with more recent transect and aerial (drone) surveys. The project has made these layers available in a Geographic Information system (GIS) to help reef practitioners visualise spatial patterns of environmental forcing and reef health, and to identify areas offering the best conditions for coral colonisation.

A key achievement of this 5-month project was the identification of hydrodynamic and water quality regimes that may help explain current reef state and predict the potential for natural recovery. Specifically, maps of water quality identified reef areas exposed to high suspended sediment concentrations and nutrient flux which may compromise not only coral recovery but also the success of restoration activities. Wave modelling indicated which areas are prone to cyclone-generated wave damages, bringing into consideration the risk of losing restoration benefits in the near-term.

To complement the baseline mapping, simulations of coral recovery across the Whitsundays were performed using the ecosystem model ReefMod. This allowed visualisation of the growth potential of corals under the depicted regimes of suspended sediment. Simulations also quantified the potential benefits of different strategies of coral outplanting (i.e., variable density and size of coral nubbins) which are relevant to cost-benefit analyses looking at optimal designs of nubbin deployment.

To evaluate patterns of coral connectivity, particle tracking simulations were run over multiple years (2014–2019) using large-scale (Central and Southern GBR) and local-scale (Whitsundays) circulation models (eReefs/RECOM). The simulated dispersal of larval particles identified two distinct regions located on the mid-shelf of the GBR as potential sources of coral larvae for the Whitsunday Islands. Short-distance dispersal simulations within the Whitsundays allowed evaluation of the likelihood of larval sink and source at 200-m resolution, two connectivity metrics that are relevant to the spatial prioritisation of restoration.

Finally, a prioritisation strategy was developed based on the predictions of water quality and short-distance larval dispersal. Multiple spatial prioritisations are possible following considerations on recruitment limitation, water quality, and the potential for reefs to propagate the benefits of interventions through coral connectivity. A decision-tree is provided to assist the decision of undertaking reef restoration at a given site.

Overall, this project represents a proof-of-concept of the use of high-resolution resilience-based maps to guide decision-making in reef restoration. Lessons learned from the use of the different modelling tools (SWAN, eReefs, RECOM, ReefMod) to depict fine-scale (<200 m) environmental forcing and coral demographic potential will benefit other reef restoration projects, including the Reef Restoration and Adaptation Program for the GBR. While this is a first attempt to develop a spatial prioritisation of reef restoration in the Whitsundays, there is a large scope for improvement with the help of expert knowledge from stakeholders and practitioners of coral restoration. This includes a continued refinement through the correction of model predictions and the integration of other considerations into the decision rules, such as cultural or economic values. To this view, the Whitsundays offer a concerted laboratory space for testing and improving scientific recommendations in support of reef restoration.

# 3 Introduction

## 3.1 Project Overview

The Whitsunday Islands form a group of more than 70 continental islands located ~20 km off the Central Queensland coast. The shallow coral reefs that fringe the islands display variable levels of development and condition as a result of multiple pressures, including river-runoff, tropical cyclones and, increasingly, heat stress due to anthropogenic climate warming. In 2017, many reefs of the Whitsundays were severely damaged by cyclone Debbie, and the recent large-scale events of coral bleaching may have compromised the potential for coral regeneration across the greater region. This raises concerns about the scope for natural coral recovery within the Whitsundays, a region that attracts almost half of the annual tourism visitation to the Great Barrier Reef (GBR) and holds highly significant spiritual and cultural heritage values.

The Reef Islands Initiative (RII) Whitsundays Program aims to protect and restore critical island habitats and associated values in the region. The Program seeks to develop a scientific, data-driven basis to support resilience-based management and guide restoration activities of intertidal and subtidal habitats in the face of climate change. This requires integration of existing environmental research, mapping and modelling, but also social, cultural and economic values, in order to develop baseline maps that can support adaptive management decisions and the deployment of interventions.

This project develops the underpinning mapping and connectivity modelling that will act as a decision support tool to guide restoration activities undertaken within the RII Whitsundays Program. It provides high-resolution data on the physical forcing, bio-geochemical environment, ecological state and demographic connectivity of shallow coral reef habitats (Table 1) to help identify sites which sites may be suitable to the deployment of restoration activities. Specifically, the objectives of this 5-month project were:

- Develop baseline maps of the distribution of shallow (0–10 m depth) coral reefs;
- Assess their exposure to physical and biogeochemical forcing;
- Gather and consolidate past and recent monitoring data on benthic condition;
- Establish the demographic connectivity of corals and recovery potential;
- Identify possible strategies of reef prioritisation for restoration interventions;
- Integrate all mapping products into a georeferenced database to support the planning of restoration activities under the RII Whitsundays Program.

Baseline habitat mapping was achieved at high-resolution (10 m) using satellite-bathymetry-wave-based classification of geomorphic zones and benthic colonisation (Roelfsema et al. 2020) which resulted in the definition of 'potential' coral habitats above 10-m depth.

Habitat maps were complemented with high-to-medium resolution (10–200 m) maps of the physical and biogeochemical drivers of coral populations (e.g., exposure to wave energy, suspended sediments, nutrients) using the best available predictive tools for the GBR: the SWAN model, which simulates near-reef wave generation, propagation and dissipation (Callaghan et al. 2015, 2020) and the coupled hydrodynamic-biogeochemical model eReefs (Steven et al. 2019; Baird et al. 2020).

Historical and recent (1993–2020) benthic data were gathered from monitoring programs (GBRMPA's Eye on the Reef, Australian Institute of Marine Science) and complemented with more recent (2017–2020) transect and aerial (drone) surveys (J. Gaskell unpubl. data). In addition to this database, 40 drone surveys and 10 underwater surveys were performed over the course of the project.

Simulations of particle dispersal across the Central and South GBR (1 km resolution) and within the Whitsundays region (200 m resolution) allowed assessment of larval connectivity over multiple years and the identification of reefs that are expected to experience recruitment limitations as well as reefs that have the potential to help other reefs regenerate through larval supply.

Coral habitat suitability was assessed using a semi-empirical model of coral recovery mediated by water quality (Bozec et al. 2020). This model builds on ReefMod-GBR, a complex model of coral populations that simulates individual coral colonies across the multiple environments (>3,800 reefs) of the GBR (Bozec and Mumby 2019; Bozec et al. 2020). The simplified model allowed quantifying standardised rates of coral recovery as a function of suspended sediment depicted at 200 m resolution.

The potential benefits of coral outplanting as a function of the size and density of coral outplants (i.e., nubbins) were explored by simulation using ReefMod-GBR. By simulating coral colonies individually, the model can track the fate of coral nubbins in time and space and their contributions to natural coral populations (Bozec and Mumby 2019).

Finally, spatial predictions of physical and biogeochemical forcing were combined with local connectivity patterns to develop recommendations on the spatial prioritisation of reefs for coral restoration. These recommendations were based on considerations related to water quality, larval supply and the potential of larval dispersal to propagate the benefits of local coral enhancements, considerations that were formalised into a decision-tree that may assist in selecting the appropriate reef sites for restoration interventions.

A Geographic Information System (GIS) was developed to compile and harmonise spatial data into a cohesive georeferenced database. The GIS will be available to reef practitioners to help them visualise the different layers and build their own prioritisation strategies as more information is integrated (e.g., tourist visitation, cultural values).

Table 1: Summary list of the georeferenced physical, biogeochemical, ecological and demographic data acquired or developed for reef resilience mapping in the Whitsundays.

| GIS layer | Definition and resolution | Resolution | Source |
|---|---|---|---|
| Bathymetry | - EOMAP Sentinel 2 satellite imagery of absolute bathymetry | 10 m | UQ |
| Geomorphic zonation | - Satellite imagery-based maps of reef bottom types (0-10m depth) | 10 m | UQ |
| Chronic wave environment | - Wave-energy metrics from process-based wave model (SWAN) | 10 m | UQ |
| Cyclonic wave climate | - Wave-energy metrics from simulated cyclone tracks (SWAN)<br>- Wave-energy metrics during TC Debbie (SWAN) | 10 m<br>10 m | UQ |
| Heat stress (bleaching) | - 1985-2020 Satellite-based Degree Heating Weeks (DHW) | 5 km | NOAA CRW |
| Water quality | - eReefs (GBR4)<br>- eReefs-RECOM | 4 km<br>200 m | eReefs/UQ |
| Benthic colonisation | - Satellite imagery-based maps of benthic colonisation (0–10m depth) | 10 m | UQ |
| Benthic cover | - 1993–2020 AIMS LTMP + MMP<br>- 2010–2020 Eye on the Reef Program (RHIS)<br>- 2017–2020 J. Gaskell's drones and transects | | AIMS<br>GBRMPA<br>J. Gaskell |
| Coral connectivity (2014-2019) | - Particle tracking using eReefs/Connie<br>- Particle tracking using eReefs/RECOM | 1 km<br>200 m | UQ/CSIRO |
| Habitat suitability | - Simulation of coral recovery potential | 200 m | UQ |

## 3.2 Methods

### 3.2.1 Region of interest

Maps of physical, biogeochemical and demographic forcing were developed to cover the entire region surround the Whitsundays Islands while providing fine-scale information on three focal regions: Hook Island, Whitsunday Island and the Molle Group (Figure 1).

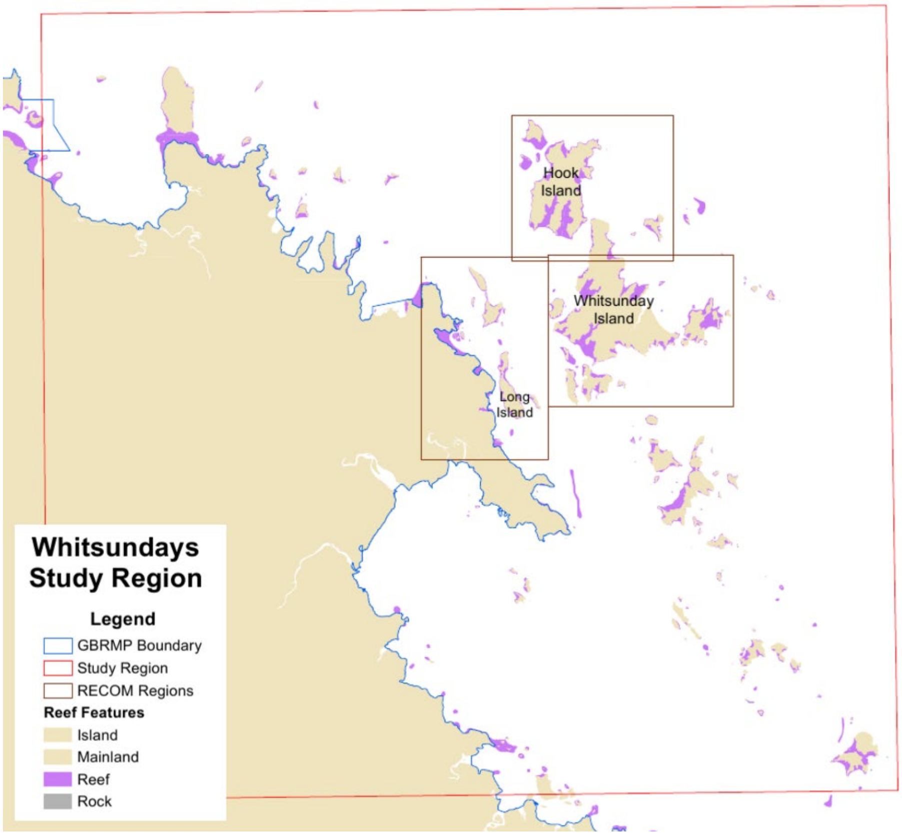


Figure 1. Study area and the three focal regions for the simulation of water quality at 200 m resolution using the CSIRO Relocatable Coastal Model (RECOM). Basemap and delimited reef areas correspond to the indicative reef boundaries defined by the Great Barrier Reef Marine Park Authority (Great Barrier Reef Marine Park Authority 2007).

### 3.2.2 Geomorphic and benthic habitat mapping

Maps of geomorphic (e.g., reef crest, slope, flat) and benthic cover (e.g., coral/algae, seagrass, sand, rubble, rock) classifications were created based on methods previously developed for mapping shallow offshore reefs on the GBR (Lyons et al. 2020; Roelfsema et al. 2020). The methods combine satellite-derived products, physical attributes (e.g., depth, slope, waves), reference data, expert knowledge, machine learning and object-based analysis. The EOMAP's Sentinel 2 satellite imagery (10 m resolution) forms the basis to derive absolute bathymetry, with sub-surface reflectance up to 20 m depth below lowest astronomical tide (LAT) depending on water clarity. This forms the base layer allowing the development of reef morphology and benthic habitat classification by combining a random forest classifier and object-based analysis in a cloud processing environment (Google Earth Engine). The methods have been well tested for the GBR 3D habitat mapping project and for reefs globally (www.allencoralatlas.org, Lyons et al. 2020) and were adjusted for nearshore reef systems such as those around the Whitsundays. Geomorphic maps were mapped up to 15 m LAT and benthic maps up to 5 m LAT, depending on water clarity.

### 3.2.3 Wave climate forcing

Wave energy predictions across the Whitsundays were derived from simulations of wave generation, propagation and dissipation using the SWAN (Simulating WAves Nearshore) model. This process-based wave model has already been applied to generate predictions of near-reef wave climates across the Great Barrier Reef (Callaghan et al. 2015, 2020;

Roelfsema et al. 2018), so that wave metrics and statistics can be easily extracted for any region of interest. Specifically, wave predictions were available for three main forcing scenarios:

- Non-cyclonic scenarios (Callaghan et al. 2015), which describe the ambient wave climate under 'normal' wind conditions (i.e., excluding tropical cyclones) at 50 m resolution;
- Tropical Cyclone (TC) scenarios (Callaghan et al. 2020), which simulate wave climates under the forcing of 6,000 synthetic TC tracks at 600 m resolution offshore of reefs. For this project, new SWAN simulations of synthetic cyclone tracks were run at 100 m resolution on reefs;
- The specific simulation at 100 m resolution of the actual track of TC Debbie (March 2017) across the Whitsundays region.

Wave metrics (Table 2) were further interpolated onto the 10-m EOMAP bathymetry grid (Roelfsema et al. 2018). Two metrics were selected for their potential to reflect mechanical stress on reefs: significant wave height (Hs, in m) and nearbed wave orbital velocity (Ub, in m/s), a measure of wave energy at the sea bed that depends on wave height but also water depth. For each metric, the mean, maximum or specific percentiles ($90^{th}$ or $95^{th}$ percentile) were extracted for each scenario. For TC Debbie, two other metrics were added: the duration (in hours) of nearbed orbital velocity exceeding 0.5 m/s and 1.0 m/s.

Table 2: Wave statistics associated to each wave modelling scenario.

| Wave modelling scenario | Significant wave height | Benthic wave velocity |
|---|---|---|
| Non-cyclonic | - Mean<br>- $90^{th}$ percentile | - Mean<br>- $90^{th}$ percentile |
| Synthetic TC | - $95^{th}$ percentile | - $95^{th}$ percentile |
| TC Debbie | - Maximum | - Maximum<br>- Duration exceeding 0.5 m/s<br>- Duration exceeding 1.0 m/s |

### 3.2.4 Thermal stress

The historical thermal stress regime of the Whitsundays was mapped using hindcast (1985-2020) daily predictions of the maximal annual degree heating weeks (DHW) available from NOAA Coral Reef Watch (Liu et al. 2017). Maximum DHW is a measure of cumulative thermal stress which is related to coral bleaching and subsequent mortality (Hughes et al. 2018).

### 3.2.5 Biogeochemical forcing

Ambient conditions of water quality at ~3-m depth were captured using retrospective spatial predictions of the eReefs coupled hydrodynamic-biogeochemical model (Steven et al. 2019; Baird et al. 2020). Daily predictions of 12 descriptors of water quality (Table 3) were extracted at different spatial resolutions during the wet season (January-April) and the dry season (July-October) of available years.

At the regional level, sediment and nutrient variables (Table 3) were extracted from the 4-km resolution model (model configuration GBR4_H2p0_B2p0_Chyd_Dnrt) over 3 years: 2017, 2018 and 2019. Variables describing the transport and re-suspension of small-sized sediment particles (dust, mud and fine sediment) were summed to obtain a global estimate of suspended sediment concentration (SSC). SSC influences the performance of early-life coral demographics (Humanes et al. 2017) which translates into slower rates of coral recovery (Bozec et al. 2020). Chronic exposure to excess nutrients from river discharge has the potential to decrease coral growth (Fabricius 2005) and increase susceptibility to thermally-induced bleaching (Wooldridge 2009) while stimulating algal productivity with detrimental effects on coral cover (Fabricius et al. 2005). Daily values were averaged over each season and years.

Table 3. List of state variables extracted from eReefs at ~3-m depth using the 4-km resolution model GBR4 (●) and the 200 m resolution RECOM models (◊). Suspended sediment variables (*) were combined together to estimate total suspended sediment concentration.

| State variable | Unit | 2017 | 2018 | 2019 |
|---|---|---|---|---|
| Temperature (temp) | degree C | ◊ | | |
| Salinity (salt) | PSU | ◊ | | |
| Photosynthetic Active Radiation (PAR) | mol photon m-2 s-1 | ◊ | | |
| Light attenuation (Kd_490) | m-1 | ◊ | | |
| Turbidity | NTU | ◊ | | |
| Dissolved Inorganic Phosphorus (DIP) | mg P.m-3 | ◊ ● | ● | ● |
| Dissolved Inorganic Nitrogen (DIN) | mg N.m-3 | ◊ ● | ● | ● |
| Nitrate (NO3) | mg N.m-3 | ◊ ● | ● | ● |
| Ammonia (NH4) | mg N.m-3 | ◊ ● | ● | ● |
| Dust* | kg.m-3 | ◊ ● | ● | ● |
| Mud* | kg.m-3 | ◊ ● | ● | ● |
| FineSed* | kg.m-3 | ◊ ● | ● | ● |

Water quality variables were also captured at 200 m resolution using the CSIRO Relocatable Coastal Ocean Model (RECOM) developed by the eReefs Environmental Modelling Suite (Baird et al. 2020). RECOM is a web-based interface allowing simulation of water circulation, biochemical cycles and sediment transport at a finer spatial resolution from a nested model with open boundaries forced by the general eReefs products (4 km or 1 km models). Because developing a model that covers the entire system (~50×50 km) is computationally demanding, three RECOM applications were implemented to focus on separate regions which, together, encapsulate the majority of the Whitsundays (Figure 1): (1) Hook Island, (2) Whitsunday Island and (3) Molle Group.

The three RECOM models were nested within the 4-km resolution model (GBR4_H2p0_B2p0_Chyd_Dnrt) and initialised using the higher resolution data available from eReefs GBR1. Water quality simulations were run from 01/11/2016 through to 31/10/2017. The same variables (Table 3) were extracted daily (at midday) at two depths, -0.5 m and -2.85 m, the latter depth being used as the representative layer for shallow reefs. Daily values were averaged in the same way as specified above, producing mean values for the 2017 wet (January-April) and dry seasons (July-October).

### 3.2.6 Historical and present-day conditions of benthic communities

Various sources of reef health data were collated to create layers describing the past and recent conditions of Whitsundays reefs:

- Participatory monitoring data from GBRMPA's Eye on the Reef program were gathered to inform about recent (2010–2020) benthic changes. Under this program, marine park rangers, tourism operators and community participants collect data using a standardised protocol enabling a rapid survey of coral reef health (RHIS – Reef Health and Impact Surveys, Beeden et al. 2014). The dataset was shared by GBRMPA and includes 77 reefs monitored in the Whitsundays during the last decade. A large spectrum of reef health information is typically recorded but for this report only the following benthic descriptors ere extracted: hard coral (HC), soft coral (SC), macroalgal (MA), rubble (RB), and *Acropora* spp. branching/plating/tabular coral (BPT). Each variable was averaged for periods 2010–2013, 2014–2017(March), 2017(April)–2020.

- Longer time-series (1993–2020) of benthic data were obtained for 3 reefs of the Whitsundays from the Australian Institute of Marine Science (AIMS) Long Term Monitoring Program (LTMP). This dataset was complemented by 7 other reefs monitored by AIMS from 2005 under the Marine Monitoring Program (MMP) run in partnership with GBRMPA (Thompson et al. 2018). The cover of HC, SC and MA assessed on transects at 5–6 m depth was averaged for each reef (2–3 sites per reef). Data at shallower depth (2 m) were not integrated to this report.

- More recent monitoring data (2017–2020) were gathered by J. Gaskell under a citizen science program initiated in the aftermath of cyclone Debbie. In late 2017 and 2018, around 30 expeditions were conducted at sites that were more likely to have been protected from waves generated by the storm. Surveys produced semi-quantitative data including the percentage cover of hard corals, soft corals, macroalgae (seaweed), as well as qualitative estimates of coral and fish diversity for 50 sites. These sites were not selected randomly but following the expectation of healthy coral cover that attracts tourism visitation. Other sites surveyed in 2019 and early 2020 were also integrated to the GIS.

To complement the database of current reef health, drone and underwater surveys were performed as part of the project on 40 key sites of the Whitsundays. When undertaken at low tide, drone surveys are effective at getting an overview of coral cover of a large section of reef within 0–3 m depth (up to 3–6 m depth under optimal conditions, Figure 2). Their advantage is to quickly map a large area of reef, locate healthy sites with high coral cover but also identify zones of macroalgal overgrowth. Google Maps or Queensland Globe can be used to identify landmarks as point of reference for stitching drone images. The optimal time for reef drone photography is when the sun does not reflect off the water back to the camera, ie mid-morning or mid-afternoon. This has to coincide with low tide heights (<0.8m) for capturing the clearest images enabling coral identification. Another important limiting factor is wind speed as the drone cannot operate above 8 knots, a significant constraint when surveying the exposed sides of an island. This makes the period extending from June to December optimal for operating drone photography in the Whitsundays (higher water clarity, relatively calm weather, favorable tides). Underwater surveys were undertaken on a limited number (10) of sites to document shallow (0–5 m depth) coral condition associated to drone footage.

Figure 2: Screenshot from a drone video footage of the fringing reef off Whitsunday Island. Native resolution images enable determination of the dominating taxa (here, soft corals with sparse hard colonies of *Porites* spp.).

Some data could not be obtained within the timeline of the project:

- a 1999–2018 monitoring program led by James Cook University (JCU) on inshore reef (fish and benthos) communities (Williamson et al. 2019) at fished sites vs. no-take reserves. This program includes 43 reef sites distributed in the northern Whitsundays (around Hayman, Hook and Whitsunday Islands) and provides a detailed assessment of the response of reef communities to cyclone Debbie. The leading team (Daniella Ceccarelli & David Williamson, JCU) has been contacted and the integration of the benthic data into the GIS is underway.
- monitoring data collected by Reef Check Australia on Daydream Is. (since 2013), Hayman Is. (Blue Pearl Bay, since 2001) and Hook Is. (Luncheon Bay since 2013). Access to the database would require completion of a data sharing agreement with Reef Check Australia.

### 3.2.7 Connectivity and dispersal of corals

#### *3.2.7.1 Large-scale coral connectivity*

Region-wide connectivity of corals was estimated using the particle-tracking simulation tool CONNIE3 (visual interface available at http://www.csiro.au/connie/; see also Condie et al. 2012, Condie and Condie 2016) to simulate the dispersal of coral larval particles. CONNIE3 is based on eReefs hydrodynamic forcing and includes a three-dimensional structured mesh that simulates both horizontal and vertical mixing of ocean currents, as well as the influence of tides and wind, at 1 km resolution. The locations of the particles were resolved in continuous space at 1-hour intervals. Larval dispersal was simulated over six spawning seasons for which GBR1 was available and larval particles were released from 6 pm to midnight on dates when the mass spawning of broadcast-spawning corals was estimated to have occurred in the region (2014 November 12; 2015 November 2; 2016 November 17; 2017 December 8; 2018 December 1; 2019 November 16; Hock et al. 2019, C. Doropoulos pers. comm.).

For each of the six spawning events, a total of 1,000 particles were seeded within each indicative reef boundary outline (GBRMPA 2007): 1,544 reef polygons in total (bounding box: 146.810; -22.479; 151.695; -18.505), including 221 in the Whitsundays region (bounding box: 148.364; -20.599; 149.185; -19.988). For each reef polygon, particles were released to the surface layer of GBR1 and remained at the surface to reflect positive buoyancy of spawning slicks during dispersal. Larval particles were tracked at hourly intervals during 120 days, with settlement allowed to start 6 hours after release. Larval particles were considered to have settled on a reef polygon when located within 1 km of that polygon. Settled particles were removed from the simulation. Each particle from a source reef that settled at a sink reef would contribute to the directed connectivity link between these two reefs. This contribution was adjusted relative to the expected post-spawning survival and competency of *Acropora* larvae at the time of settlement (Connolly and Baird 2010), resulting in an estimate of the density of settled larvae as a measure of connectivity strength between the two reefs. Connectivity strengths were further adjusted proportionally to the area of source and sink polygons to reflect the importance of reef size in demographic connectivity (Bozec et al. 2020). Conventional connectivity matrices were developed from the pairwise source-sink connectivity strengths (1,544 sources × 1,544 sinks, one matrix for each year between 2014–2019).

#### *3.2.7.2 Fine-scale coral connectivity*

A reliable representation of coral connectivity within the Whitsundays requires resolving larval dispersal and exchanges at a fine resolution due to the narrow convolutions of dry areas (islands) and because most reef areas are small. We used the RECOM models to simulate particle tracking at 200 m resolution in all three subregions separately (with no cross-boundary flow) for each year between 2014–2018. Particle tracking was carried out from 1st of September till the 31st of January. This allowed for a minimum of 1 month 'warm up period' before the release of any particles.

Coral larval particles were released between 6 pm and 6 am at a rate of one particle per minute for the two nights of the expected larval spawning event, and tracked until the end of the time period. Particle position was recorded every 30 minutes, along with the age of each particle. All these points were then compared to the reef polygons identified by the newly produced geomorphic map. Were considered as potential reefs all polygons classified either as 'reef slope, 'back reef slope', 'sheltered reef slope, 'inner reef flat', 'outer reef flat', 'reef crest', or 'small reef'. As a result, each modelled area had between 250-450 polygons (1,055 polygons in total). Reef polygons traversed by each larval particle were recorded. Each particle was assessed on the likelihood of propagation for four different coral species based on empirical rates of larval mortality and acquisition/loss of competence (Connolly and Baird 2010). Estimates of propagation likelihoods were then summed for each reef-to-reef connection and placed into a connectivity matrix.

### 3.2.8 Modelling of the growth potential of corals

Growth potential of corals was evaluated at 200 m resolution by running simulations of a semi-empirical model of coral recovery mediated by water quality (Bozec et al. 2020). The model builds on ReefMod-GBR, a complex model of coral populations that simulates individual coral colonies across the multiple reef environments of the GBR (Bozec and Mumby 2019; Bozec et al. 2020). It integrates empirical dose-response relationships between suspended sediment (SSC) and processes of survival, growth and development of coral larvae and recruits (Humanes et al. 2017) to predict rates of natural recovery in terms of annual increments in percentage coral cover. This enables simulation of recovery trajectories of coral cover under routine (i.e., annual averages) concentrations of suspended sediment (Figure 3).

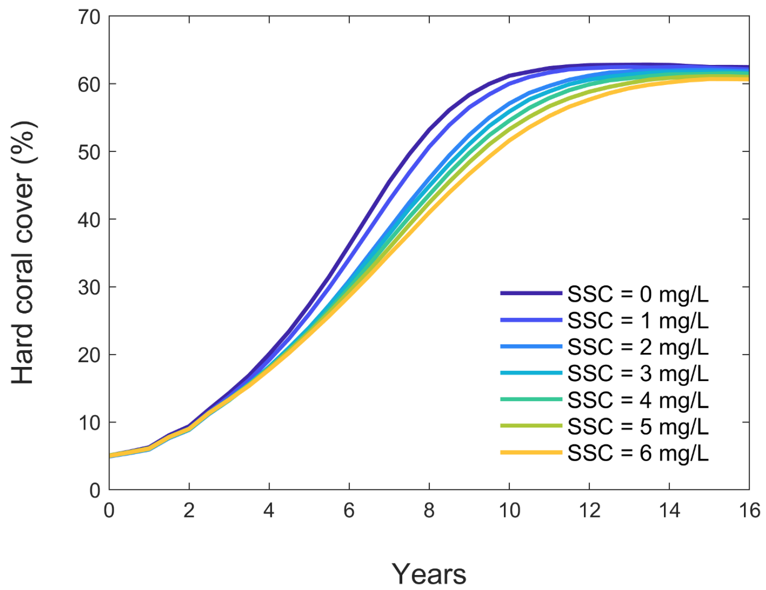


Figure 3: Theoretical curves of coral recovery simulated for different routine concentrations of suspended sediment (SSC).

Each of the 1,055 reef polygons identified in the geomorphic layer was assigned the value of SSC predicted by RECOM in the closest 200 m pixel during the 2017 wet and dry seasons. Seasonal values were averaged to produce a mean annual estimate of SSC. Natural coral recovery was then simulated for each reef polygon based on the predicted mean SSC and from a hypothetical 5% hard coral cover. Because growth increments of coral cover differ with the initial value of coral cover (i.e., growth curves are nonlinear for a given SSC, Figure 3), simulating coral recovery from a fixed initial cover for all reef polygons allows generating standardised growth increments that only reflect the influence of suspended sediment. The resulting growth potentials can then be mapped to visualise how habitat suitability varies across the region as a result of variations in water quality.

### 3.2.9 Simple simulations of the benefits of coral restoration

Potential benefits of scenarios of coral outplanting were explored by simulation using ReefMod-GBR (Bozec et al. 2020), a model of coral populations that simulate the fate of individual corals across the GBR (Figure 4). ReefMod-GBR integrates the effects of water quality (suspended sediments, chlorophyll), thermal stress (bleaching) and tropical storms on the demographics of different groups of coral species. Because the model simulates coral colonies individually, it is particularly suitable for assessing the benefits of coral outplanting on a reef: the deployment of coral outplants can be simulated at different densities and sizes, and their growth and survival can be tracked individually in time and space (Bozec and Mumby 2019). The model is currently used as simulation tool for exploring the efficacy of restoration strategies across the GBR under the Reef Restoration and Adaptation Program (https://gbrrestoration.org/).

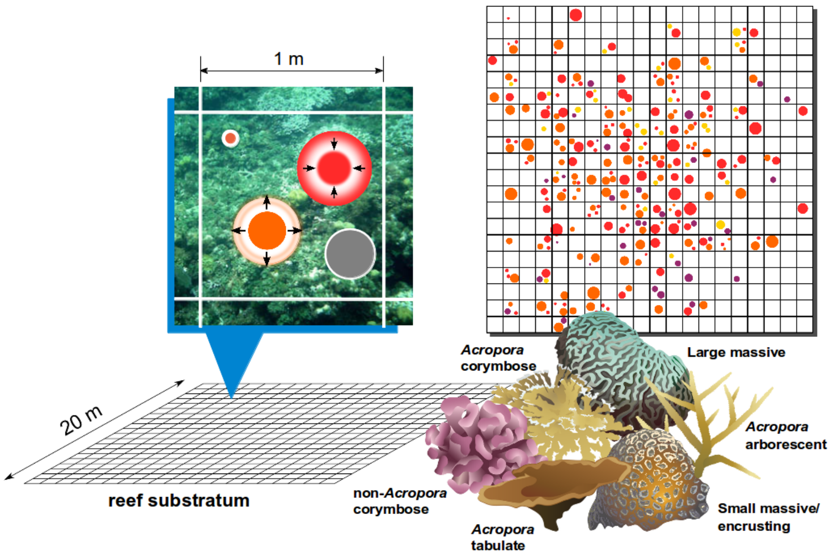


Figure 4: Schematic representation of the spatially-explicit model of coral demographics, ReefMod-GBR (Bozec et al. 2020). Individual coral colonies are typified by circular areas of variable size. Corals settle, grow, shrink and die in a virtual 20m × 20m environment as they do *in situ*. Demographic rates are specific to six modelled coral groups.

Different strategies of coral deployment can be explored, whereby strategies refer to the use of different sizes (diameter) of corals (as nubbins, juveniles or adults), different densities of outplants, but also environmental characteristics that are likely to influence the success of the intervention at local (e.g., larval retention, water quality and exposure to acute stress) or regional scales (e.g., importance of the selected reef for supplying coral larvae to other reefs).

We use the model to simulate the benefits of different outplanting scenarios on a hypothetical reef, whereby each scenario refers to the deployment of coral fragments (nubbins) of different diameters at different densities on a given year. The reef was simulated with a starting coral cover of 5% equally distributed among six coral groups (Figure 4) until full recovery was achieved for increased exposure to suspended sediment (Figure 3). Coral outplanting was modelled as the addition of *Acropora* corymbose corals (radial extension ~3 cm/year) of increasing diameter sizes (from 2 to 10 cm) on the modelled reef space (20 m × 20 m horizontal space). Corals were deployed at year 0, 1 and 2 at different densities (from 0.2 to 2 coral nubbins per m$^2$). The benefits of restoration were assessed five years after the start of outplanting as the difference in the percent coral cover (all hard corals) achieved at that time between the restoration scenario and the baseline scenario (no outplanting) (Figure 5). As such, this benefit corresponds to the additional percent coral cover generated by the survival and growth of coral outplants. Simulations assume constant recruitment at rates modelled on a mid-shelf reef (Bozec et al. 2020) in the absence of observations representative of inshore reefs.

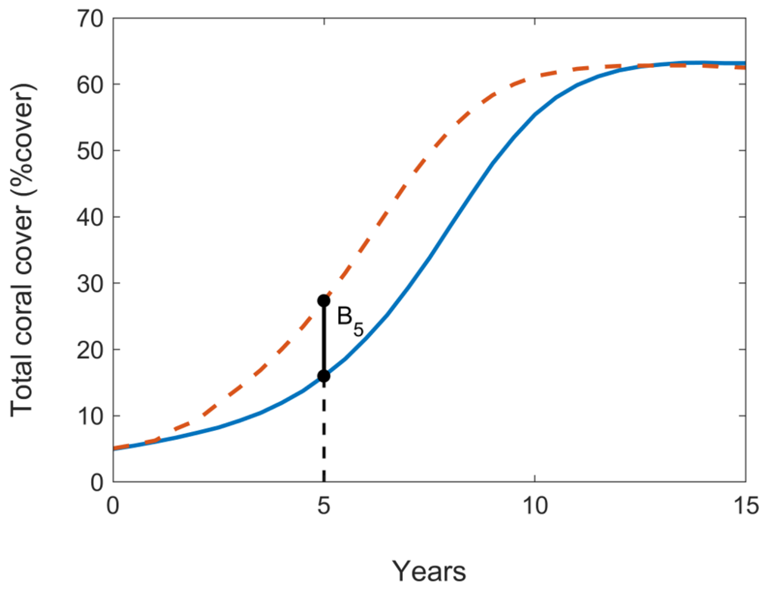


Figure 5: Calculation of the net benefit of a scenario of coral outplanting (red dashed curve) relative to a scenario without coral outplants (blue curve). Benefit is expressed as the net difference between the total coral cover predicted by the two scenarios at year 5. Here, coral nubbins were outplanted at years 1, 2 and

## 3.3 Project milestones

| Contract Deliverable (refer to Contract Milestones) | Status | Outline the work completed or provide an explanation if not 100% complete, and next steps (if applicable) | If not complete, propose revised date |
|---|---|---|---|
| Concerted list of key 'sites' for RECOM applications | yes | Agreed on three regions (Hayman-Hook, Whitsunday Is., Molle Group) to be modelled at high-resolution (200 m) with RECOM to simulate water quality (eReefs downscaled) and particle tracking (fine-scale coral connectivity). | |
| Geomorphic baseline map | yes | Developed a map of subtidal geomorphic classes derived from satellite imagery, bathymetry and wave energy modelling. | |
| Regional maps of environmental variables | yes | Developed maps of non-cyclonic (routine) wave climate and wave metrics during representative cyclones (synthetic tracks) and cyclone Debbie (2017 reconstructed track).<br>Developed maps of water quality variables (12 variables) from eReefs-GBR4. | |
| Reef regional connectivity | yes | Simulated coral particle dispersal at a regional level (Central and Southern GBR, including the Whitsundays) using CONNIE3/eReefs-GBR1. | |
| Reef habitat suitability maps | yes | Simulated coral recovery trajectories for 1,055 potential reef habitats mediated by suspended sediment using ReefMod-GBR. | |
| Drone image analysis completed | yes | Completed 40 drone surveys and 10 underwater sampling of benthic reef communities | |
| Benthic composition | yes | Developed a benthic map (corals, algae, seagrass, sand etc.) from satellite imagery | |

| | | | |
|---|---|---|---|
| map | | | |
| Maps of current reef condition | yes | Collated data on current (2017-2020) reef condition (486 reef sites) from conventional monitoring programs (AIMS, Eye On the Reef), participatory surveys (Gaskell unpublished data) and surveys performed over the course of the project. | |
| RECOM completed, including reef connectivity | yes | Simulation of water quality and particle tracking (coral larval connectivity) completed for the three RECOM models. | |
| Simulations of coral restoration | yes | Explored by simulation (ReefMod-GBR) different strategies of coral outplanting to predict restoration benefits from variable size and density of coral nubbins. | |
| Key recommendations for spatial prioritisation of coral restoration | yes | Developed a framework to identify priority reefs for restoration, based on expectation of suitable habitats for coral development and short-distance dispersal of coral larvae. | |

# 4 Results

## 4.1 Geomorphic zonation

Figure 6 shows the mapping of subtidal geomorphic classes at 10-m resolution from the object-based analysis combining satellite products, depth and wave climate. This hierarchical classification identified 7 categories of coral habitat: 'reef slope', 'back reef slope', 'sheltered reef slope', 'inner reef flat', 'outer reef flat', 'reef crest', and 'small reef'. Geo-referenced polygons in these categories were subsequently considered as potential reef habitats, i.e., areas with a significant amount of hard substratum providing space for coral settlement and growth.

This product must be considered as a draft classification of reef habitats and would require some form of validation from *in situ* observations or local expert knowledge before it is formally used to support a spatial prioritisation of reef restoration.

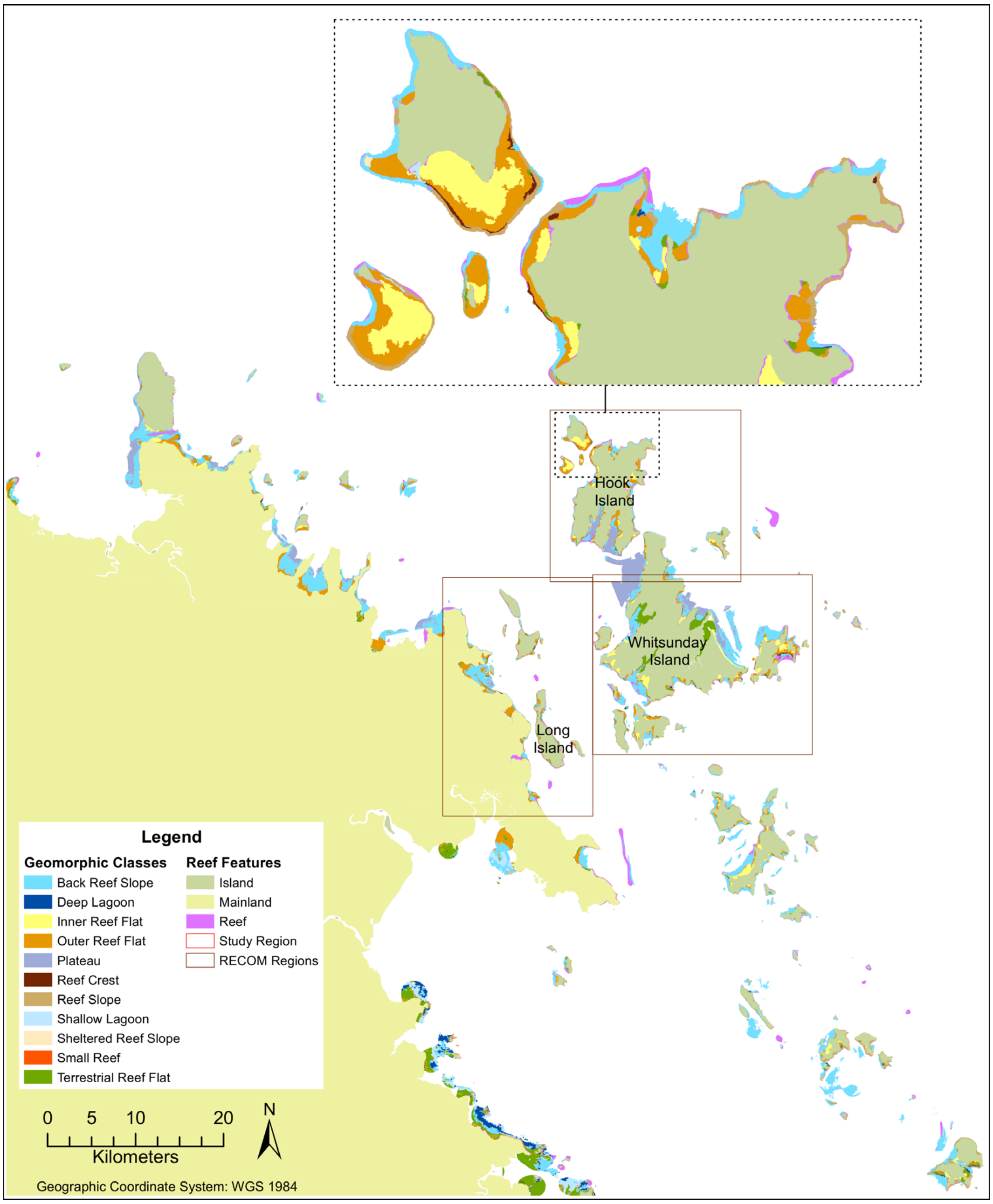


Figure 6. Geomorphic map of the Whitsundays.

## 4.2 Wave exposure

### 4.2.1 Non-cyclonic wave climate

Non-cyclonic predictions of wave height and nearbed orbital velocity are available in the GIS tool for the entire Whitsundays, but here only a close-up on Hook Island is provided (Figure 7) to demonstrate the capacity of high-resolution (10 m) wave modelling to support a spatial prioritisation for restoration. These two metrics are generally correlated and provide a complementary picture of the typical wave climate across the region in the absence of cyclones (i.e., the routine exposure to waves). While wave height is an intuitive metric of wave exposure, it does not necessarily translate into hydrodynamic stress above the reef substratum. Nearbed wave orbital velocity captures the hydrodynamic force transferred to the seabed depending on the reef profile and depth.

This is a direct measure of the amount of stress that corals are virtually exposed to, which makes it relevant to integration into decision planning for coral restoration. For example, coral transplantation might target low-energy environments to decrease the risk of detachment or fragmentation caused by shear-stress. In addition, macroalgal productivity is often stimulated in high-energy environments in the presence of excess nutrients.

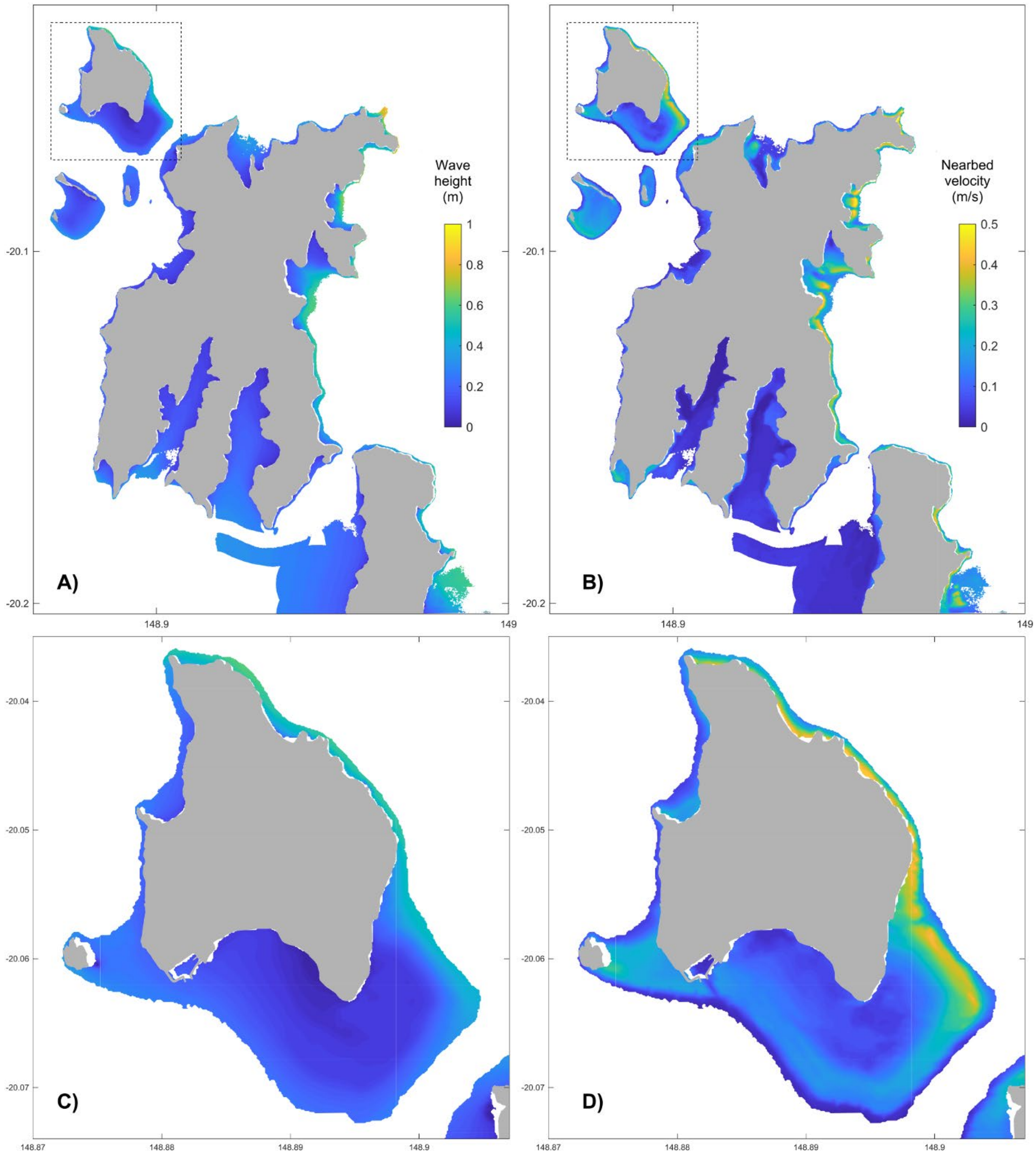


Figure 7: SWAN nearshore predictions of mean wave height (A, C) and nearbed wave velocity (B, D) at 10 m resolution under typical non-cyclonic conditions. Mean values were obtained by averaging all the predicted values over the timeframe of the simulation.

The wave statistics mapped in Figure 7 correspond to the average values of wave height and orbital velocity calculated over the simulated scenario. The GIS tool also integrates the mapping of the 90th percentiles of these two metrics (i.e., the values below which 90% of the distribution of predictions for a given 10-m pixel is found).

### 4.2.2 Cyclonic wave climate

Predictions of wave height and nearbed orbital velocity under multiple scenarios of tropical cyclones (simulation of thousands of synthetic cyclone tracks over 150 years) inform about the likelihood of storm-generated wave impacts across the Whitsundays reefs (Figure 8). Values were mapped at 10-m resolution and represent the 95th percentiles of each metric. Based on this information, potential refugia from tropical storms can be identified and elected as reliable sites for coral restoration.

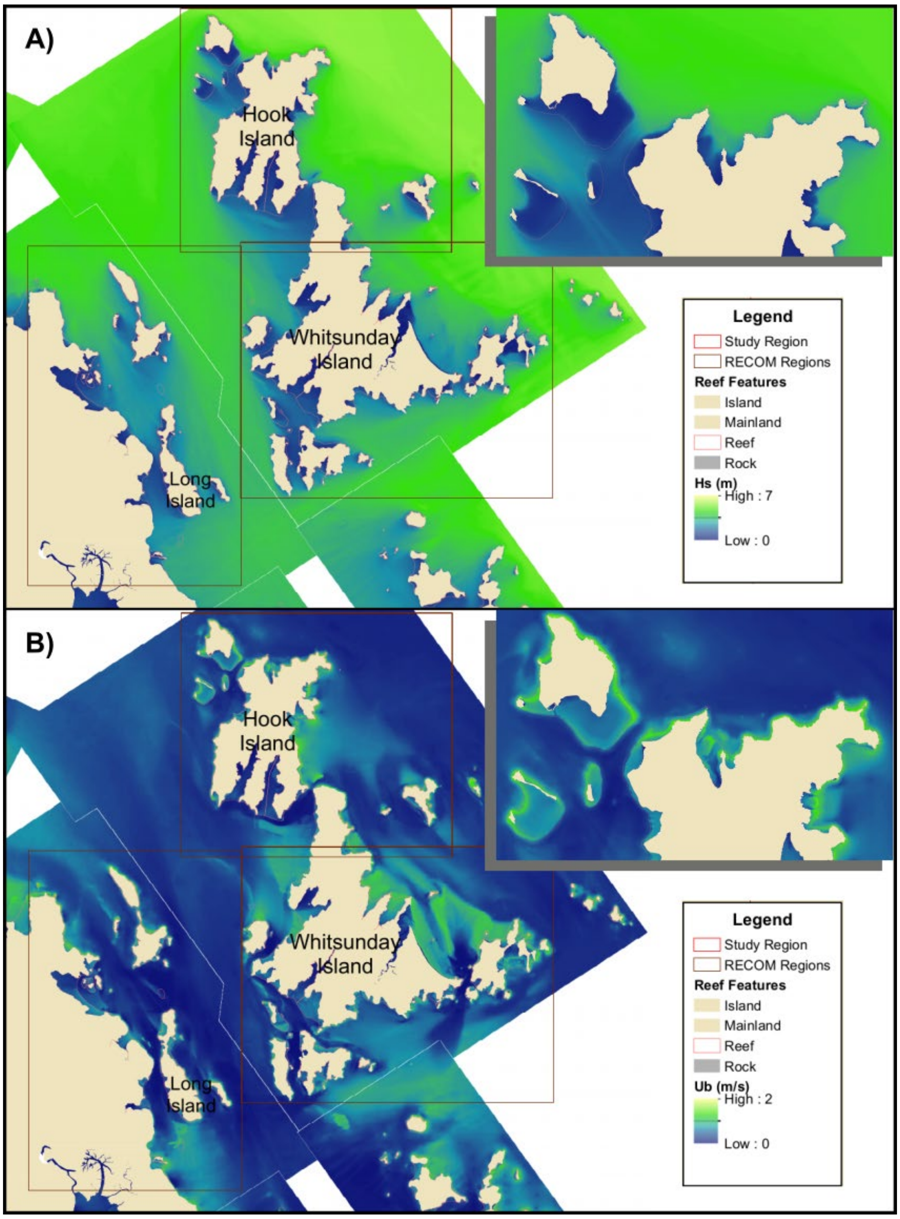


Figure 8: SWAN predictions of wave height (A) and nearbed wave velocity (B) at 10-m resolution during cyclonic conditions. Values represent the 95th percentiles of the simulated TC tracks (values capped for better contrasts).

Other metrics can be extracted, such as specific return periods expressed as the exceedance value of wave height or orbital velocity after 10, 20, 50 or 100 years (not available in the GIS tool). Compared to percentiles of wave metrics, return periods provide more tangible information about the risk of damaging TC waves over time.

### 4.2.3 Wave energy during TC Debbie

Reconstruction of the wave forcing during the crossing of cyclone Debbie (25th-28th of March 2017) shows which areas may have sustained the strongest hydrodynamic stress (Figure 9). This information can assist the interpretation of monitoring surveys with maximum orbital velocity being expected to correlate with recent reductions of coral cover observed in the region (e.g., (Williamson et al. 2019), this report). Moreover, these predictions of cyclone exposure can guide future monitoring by ranking reef areas along a gradient of expected damages.

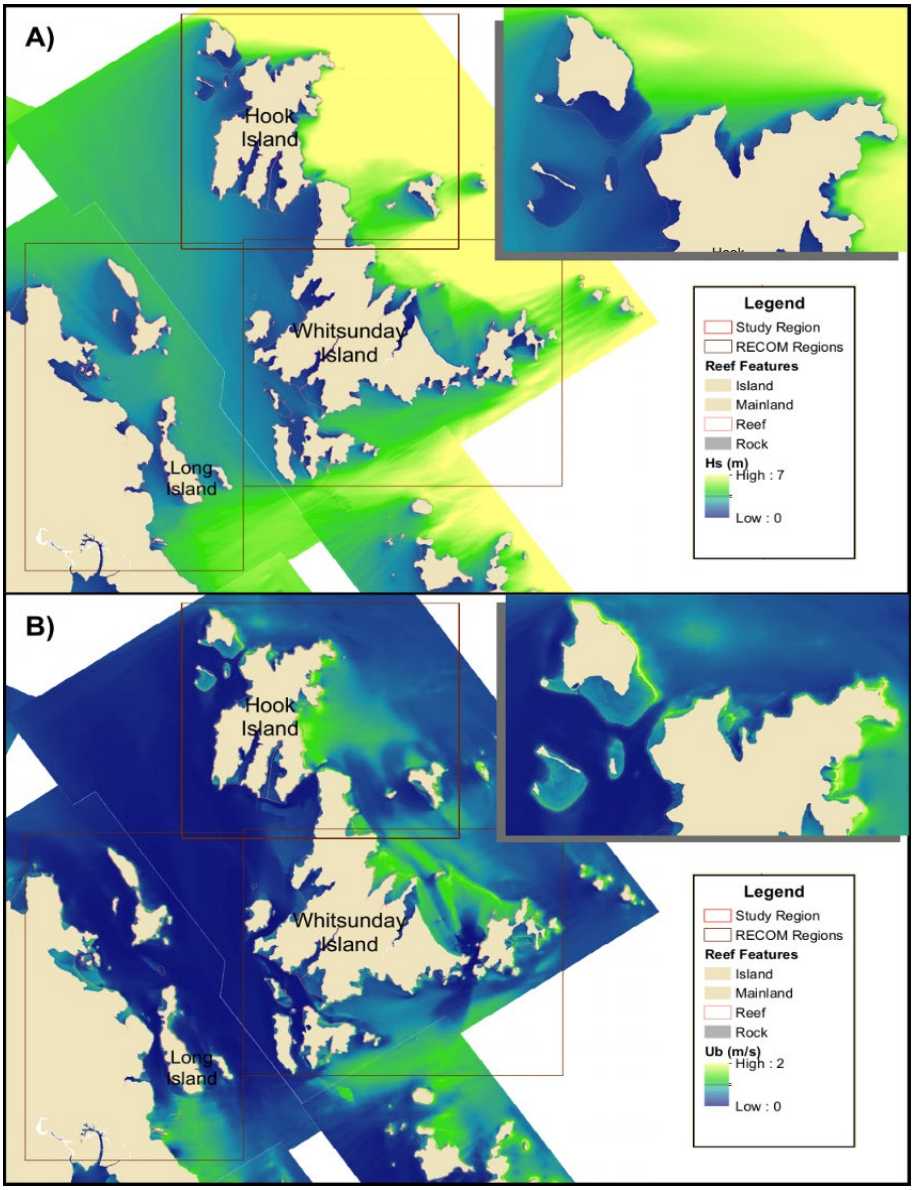


Figure 9: Maximum wave height (A) and nearbed wave velocity (B) at 10-m resolution during the crossing of category 4 cyclone Debbie in March 2017.

Other wave metrics associated to TC Debbie have been integrated to the GIS tool. These represent the duration in number of hours of nearbed orbital velocity exceeding two arbitrary thresholds: 0.5 m/s (Ubt1) and 1.0 m/s (Ubt2). The rationale behind these metrics is that the duration of stress exposure could be more correlated with coral damages than the maximum stress intensity experienced by a reef.

### 4.2.4 Considerations on wave climate modelling

These wave predictions should be considered as a proxy of the actual wave climate in the Whitsundays. The wave modelling was done at the scale of the entire GBR (Callaghan et al. 2020) with as much information that can be used in the calibration process afforded by model scale and data availability. For example, there is currently no wave measurements available in locations to calibrate the tropical cyclone modelling. However, the model integrates satellite-derived depth data to ensure the wave generation was reasonable. One important implication is that absolute predictions of wave metrics may be off but their relative variations – both in space and time – are useful. This means there is reasonable expectation that locations with lower values than others for any of those metrics will prove to be correct. The magnitude of this difference has uncertainty and without significant data collection, its value is unknowable.

Another limitation is in the summary statistics (mean, percentiles), given the considerable variability of the modelled processes (waves occurs from calm to cyclonic). This characterisation is necessary to take terabytes of information into something manageable, but the reef experiences it all. As our knowledge progresses, other wave-related metrics and statistics may prove better proxies of the hydrodynamic stress experienced by corals. The final but critical limitation is bathymetry. While there has been considerable improvement in the high-resolution mapping of seascape topography, many wave processes are strongly influenced by depth and depth gradients in numerous ways. Bathymetry will always be a significant source of uncertainty in wave predictions.

## 4.3 Water quality

### 4.3.1 Suspended sediment

The spatial footprint of suspended sediment concentration (SSC) predicted by eReefs GBR4 during 2017–2019 revealed the influence of river sources bringing terrestrial inputs to the system (Figures 10-11). SSC values were substantially higher during the wet season: on average, ~5 times higher than the dry season at a given location. The influence of river flows is also reflected by greater sediment concentrations close to the mainland, with mean SSC values above 20 mg/L during the summer runoff period at the mouth of the Proserpine river (~20.5°S, 148.7°E, in Repulse Bay). In 2017, high SSC values (2–3 mg/L) were obtained at a greater distance from the shore (up to Hook Is. and Whitsunday Is.) likely reflecting the influence of river floods associated to TC Debbie.

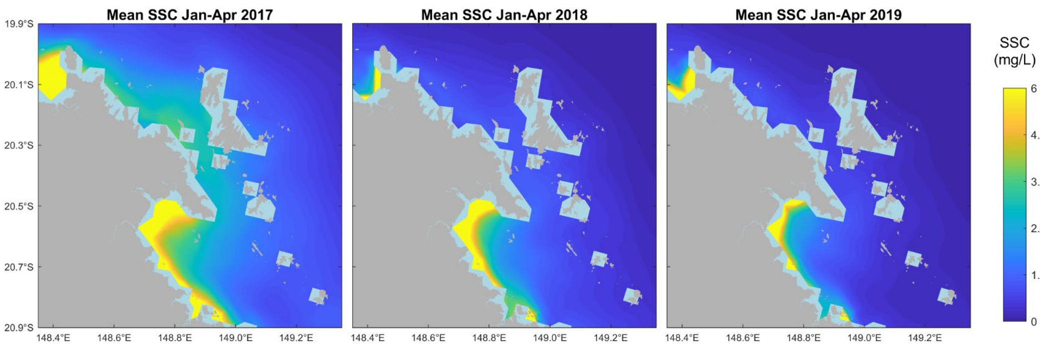


Figure 10. Predictions of suspended sediment concentration (SSC) at 4 km resolution using eReefs during the 2017–2019 wet seasons (daily values averaged from January through April, inclusive). SSC values were capped to 6 mg/L.

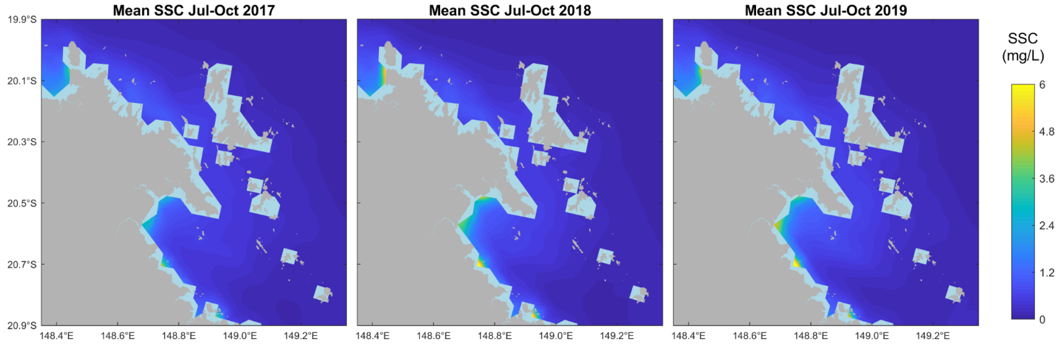


Figure 11. Predictions of suspended sediment concentration (SSC) at 4 km resolution using eReefs during the 2017–2019 dry seasons (daily values averaged from July through October, inclusive). SSC values were capped to 6 mg/L.

Finer scale predictions from the three RECOM models in 2017 reveals zones of high SSC that were not apparent at 4 km resolution (Figure 12). Very high exposure to SSC characterises the entire Molle Group (3–7 mg/L in summer, 1–3 mg/L in winter) while Hook Is. appears globally less exposed, except the south-west area of the island which sustains relatively high sediment concentrations (3–4 mg/L in summer 2017). The zone between Hook and Whitsunday islands exhibits very high SSC values (~8 mg/L in summer 2017) which could be due to wind-driven resuspension.

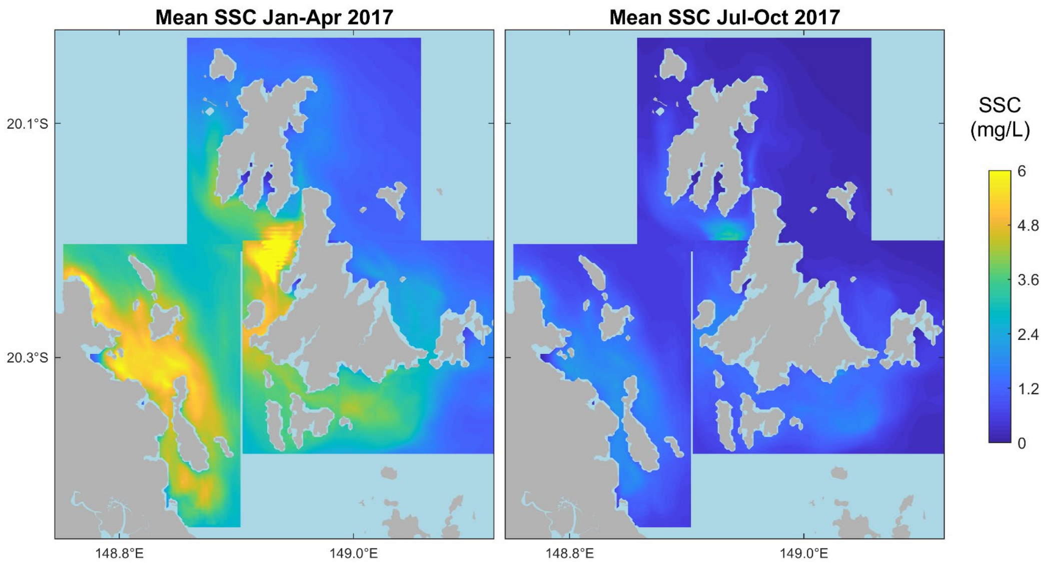


Figure 12. Predictions of suspended sediment concentration (SSC) at 200 m resolution using RECOM over Hook Island, Whitsunday Island and Molle Group during the 2017 seasons.

### 4.3.2 Nutrients

Seven nutrient variables simulated by eReefs were extracted (ammonia, nitrate, dissolved inorganic nitrogen, dissolved inorganic phosphorus) but results are presented here for dissolved inorganic nitrogen (DIN) only (Figures 13-14). The spatial footprint of DIN reveals similar patterns than suspended sediments: higher nutrient concentrations overall during the summer runoff season, with peak values found in coastal waters, especially at the mouth of the Proserpine river.

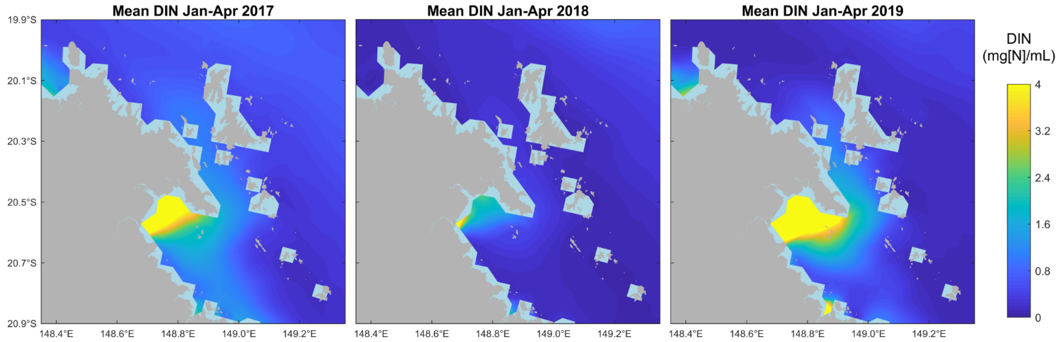


Figure 13. Predictions of dissolved inorganic nitrogen (DIN) concentration at 4 km resolution using eReefs GBR4 during the 2017–2019 wet seasons. DIN values were capped to 4 mg[N]/mL.

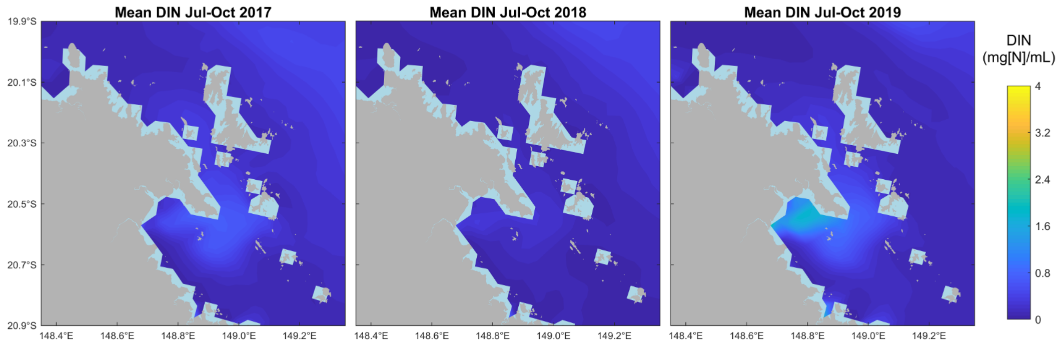


Figure 14. Predictions of dissolved inorganic nitrogen (DIN) concentration at 4 km resolution using eReefs GBR4 during the 2017–2019 dry seasons. DIN values were capped to 4 mg[N]/mL.

Finer scale predictions from RECOM in 2017 confirm these seasonal differences with higher nutrient concentrations during summer (Figure 15). High DIN values concentrated around Hook island, especially in the south of the island (inside the Nara and Macona inlets) and along the east side, possibly a result of wind-driven resuspension between Hook and Whitsunday islands and transport by currents through the Hook Passage.

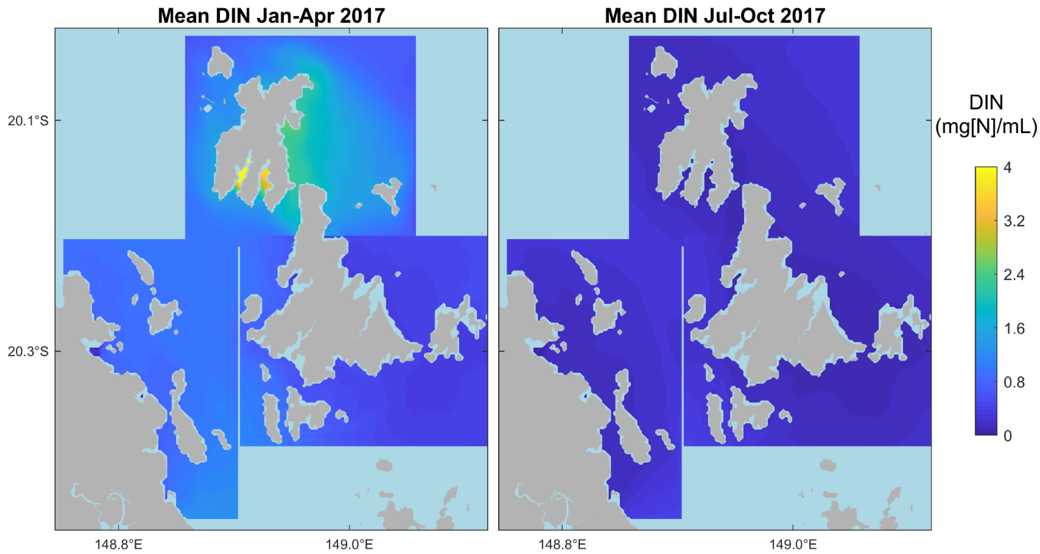


Figure 15. Predictions of dissolved inorganic nitrogen (DIN) concentration at 200 m resolution using RECOM during the 2017 wet and dry seasons.

### 4.3.3 Considerations on water quality layers

The accuracy of water quality predictions depends on a number of factors that, in the absence of validation with *in situ* measurements, need to be carefully considered before integration to the decision process.

Perhaps the most important limitation of the fine-scale predictions provided by RECOM is that a single year (2017) was simulated during the project, due to slow and labour-intensive computation (see 8-Lessons learned). The 2017 water year is strongly influenced by Cyclone Debbie which generated considerable rainfall and river discharge (Gruber et al. 2020). For this reason, it is not representative of an average water quality regime, but rather captures the spatial footprint of river discharges that can be achieved after extreme wet seasons. One advantage is that the spatial footprint of sediment and nutrient concentrations is exacerbated, allowing finer segregation of reef environments within a given area. This also enables identification of reef habitats that may experience the best water quality overall, i.e., those expected to escape flood events of similar magnitude than experienced in 2017.

Moreover, fine-scale predictions of RECOM may be highly sensitive to the complexity of the seascape and island shorelines which affect the accuracy of hydrodynamic transport. In addition, the extent to which river flows influence the concentration of sediment and nutrient in coastal waters depends on how well river flows dynamics are captured by the source catchment model.

Finally, because the three RECOM models were simulated separately, there is no continuation of fine-scale hydrodynamic transport across the boundaries of the modelled regions, although all boundary conditions were forced by the 4-km model (eReefs GBR4). This means that patterns of SSC and DIN close to the boundaries of each RECM region need to be carefully interpreted. Future developments may proceed to the cross-boundary interpolation of SSC and DIN predictions.

## 4.4 Benthic mapping

### 4.4.1 Benthic classification

Figure 16 shows the mapping of benthic cover classification (coral/algae, rock, rubble, sand, seagrass) at 10-m resolution. This information, when crossed with the geomorphic map (Figure 6), provides additional information on the available space for coral/algal colonisation (i.e., excluding sand, seagrass and rubble). As for the geomorphic map, this draft classification of benthic covers requires validation with *in situ* observations or local expert knowledge before it is formally used to support a spatial prioritisation of reef restoration.

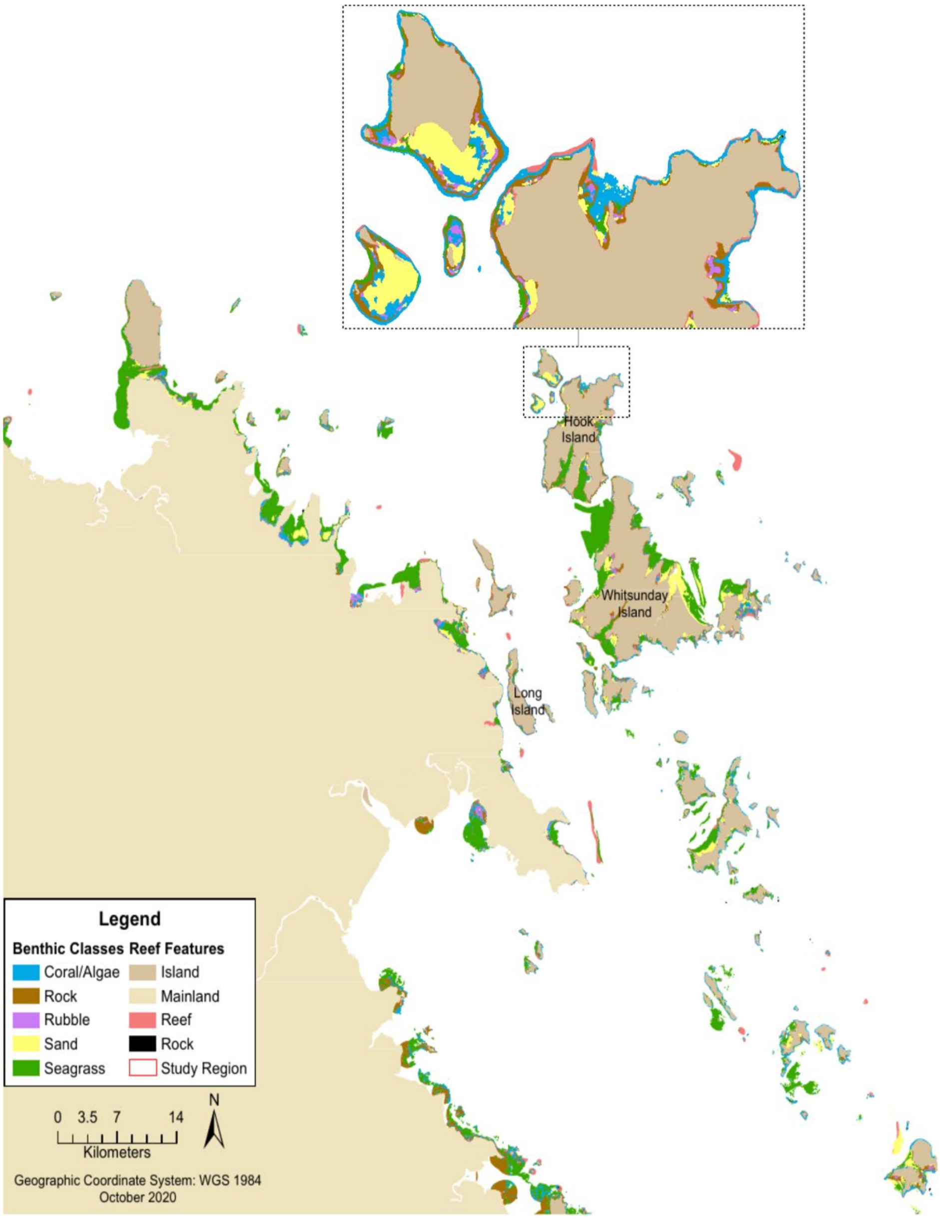


Figure 16. Benthic classification map of the Whitsundays.

### 4.4.2 Drone surveys

A total of 40 drone surveys were conducted from August and December 2020 to document the extent and condition of reefs around Hook Island, Whitsunday Island and across the Molle Group (figure 17). Another 10 drone surveys were performed during this period but due to wind impacting the surface, the footage was unusable for data collection. This footage produced a database of 300 aerial photographs that can assist the process of site selection for reef restoration (Figure 18 and 19). In addition, aerial imagery provides reference points for future assessment of reef health and constitutes a first step towards the development of high-definition (centimeters) mapping of reefs in the Whitsundays.

| Island | Location | Date Surveyed | GPS | Available Photos / Videos | Average depth | Hard Coral Cove | Soft Coral Cove | Total Coral Cove % | Algae Cove |
|---|---|---|---|---|---|---|---|---|---|
| Whitsunday | Silica Bay | Saturday, 10 October 2020 | -20.306201, 149.032634 | UW | 3m | 5 | 2 | 7 | 90 |
| Whitsunday | Chance Bay South | Saturday, 10 October 2020 | -20.303661, 149.053927 | UW / Aerial | 3m | 5 | 5 | 10 | 80 |
| Whitsunday | Chance Bay East | Saturday, 10 October 2020 | -20.300803, 149.057237 | UW | 2m | 20 | 15 | 35 | 30 |
| Haslewood | Teague Island | Saturday, 10 October 2020 | -20.301018, 149.072246 | UW | 2m | 10 | 50 | 60 | <5 |
| Haslewood | Whites Bay | Saturday, 10 October 2020 | -20.281763, 149.093425 | UW | m | 30 | 15 | 45 | 10 |
| Hook Island | Luncheon Bay Mid | Sunday, 11 October 2020 | -20.065453, 148.948885 | UW / Aerial | 1.5m | 10 | 15 | 25 | <5 |
| Hook Island | Luncheon Bay West | Sunday, 11 October 2020 | -20.064969, 148.946385 | UW | 3m | 2 | 5 | 7 | <5 |
| Hook Island | Butterfly Bay | Sunday, 11 October 2020 | -20.073838, 148.923230 | UW | 2m | 12 | 25 | 37 | <5 |
| Hook Island | Cockatoo Point | Sunday, 11 October 2020 | -20.078708, 148.900975 | UW | 3m | 25 | 40 | 65 | <5 |
| Border Island | Catarran Bay | Sunday, 18 October 2020 | -20.156484, 149.032155 | UW / Aerial | 2m | 3 | 8 | 11 | 20 |
| Deleraine Island | West | Sunday, 18 October 2020 | -20.161637, 149.074395 | UW | 3m | 5 | 5 | 10 | 60 |
| Esk Island | East Side | Sunday, 18 October 2020 | -20.234433, 149.042195 | UW | 3m | 2 | 10 | 12 | 40 |
| Hook Island | North West | Sunday, 18 October 2020 | -20.068492, 148.907319 | Aerial | 3m | 15 | 5 | 20 | <5 |
| Hook Island | Stonehaven Bay | Sunday, 18 October 2020 | -20.106163, 148.904282 | Aerial | 2m | 20 | 15 | 35 | <5 |
| Hook Island | West Wall 1 | Sunday, 18 October 2020 | -20.112477, 148.888541 | Aerial | 2m | 10 | 25 | 35 | <5 |
| Langford Spit | West | Sunday, 18 October 2020 | -20.086728, 148.874345 | Aerial | 3m | 15 | 15 | 30 | 20 |
| Haymen | East Channel | Sunday, 18 October 2020 | -20.070627, 148.890883 | Aerial | 2m | 10 | 50 | 60 | 10 |
| Hook Island | West wall 2 | Monday, 19 October 2020 | -20.161932, 148.882238 | Aerial | 1m | 3 | 5 | 8 | <5 |
| Hook Island | Nara entrance West | Monday, 19 October 2020 | -20.162358, 148.893443 | Aerial | 4m | 6 | 9 | 15 | <5 |
| Hook Island | False Nara East | Monday, 19 October 2020 | -20.164951, 148.886391 | Aerial | 2m | 5 | 15 | 20 | 40 |
| Hook Island | Macona East Entry | Monday, 19 October 2020 | -20.177186, 148.930088 | Aerial | 2m | 15 | 25 | 40 | 10 |
| Hook Island | South East | Monday, 19 October 2020 | -20.165499, 148.943426 | Aerial | 2m | 10 | 35 | 45 | <5 |
| Whitsunday | North West | Monday, 19 October 2020 | -20.155253, 148.960538 | Aerial | 2m | 5 | 5 | 10 | 20 |
| Whitsunday | Cairn Beach | Monday, 19 October 2020 | -20.160563, 148.956487 | Aerial | 3m | 15 | 55 | 70 | <5 |
| Whitsunday | Cairn Beach South | Monday, 19 October 2020 | -20.175547, 148.956414 | Aerial | 2m | 20 | 40 | 60 | <5 |
| Whitsunday | North CID | Monday, 19 October 2020 | -20.223410, 148.946159 | Aerial | 2m | 15 | 5 | 20 | |
| Whitsunday | Cid Harbour | Monday, 19 October 2020 | -20.275059, 148.927034 | Aerial | 2m | 5 | 30 | 35 | 30 |
| Shute Harbour | Islands | Tuesday, 20 October 2020 | -20.298858, 148.792255 | Aerial / UW | 2m | 15 | 60 | 75 | <5 |
| Shute Harbour | Mainland | Tuesday, 20 October 2020 | -20.290094, 148.795783 | UW | 2m | 3 | 2 | 5 | 50 |
| North Molle | West | Saturday, 26 September 2020 | -20.228367, 148.816468 | Aerial | 2m | 6 | 3 | 9 | 10 |
| North Molle | North | Saturday, 26 September 2020 | -20.213344, 148.809608 | Aerial | 2m | 3 | 8 | 11 | <5 |
| North Molle | East | Saturday, 26 September 2020 | -20.236341, 148.830308 | Aerial | 2m | 2 | 3 | 5 | 20 |
| North Molle | South | Tuesday, 20 October 2020 | -20.243225, 148.828009 | Aerial | 3m | 40 | 8 | 48 | <5 |
| Daydream Island | East | Friday, 14 August 2020 | -20.259049, 148.814606 | Aerial | 2m | 4 | 15 | 19 | 20 |
| South Molle Island | South | Tuesday, 20 October 2020 | -20.280617, 148.841504 | Aerial | 2m | 8 | 5 | 13 | 50 |
| South Molle Island | East | Tuesday, 20 October 2020 | -20.264401, 148.849671 | Aerial / UW | 1m | 25 | 23 | 47 | 10 |
| Hook Island | Cockatoo South | Thursday, 5 November 2020 | -20.084766, 148.903665 | Aerial / UW | 2m | 15 | 32 | 47 | <5 |
| Hook Island | Stonehaven Bay | Thursday, 5 November 2020 | -20.095956, 148.906290 | Aerial / UW | 2m | 8 | 9 | 17 | <5 |
| Black Island | East Side | Thursday, 5 November 2020 | -20.082146, 148.894853 | Aerial / UW | 2m | 10 | 55 | 65 | <5 |
| Hook Island | South West Saba | Friday, 18 December 2020 | -20.131002, 148.944858 | Aerial | 1m | 7 | 8 | 15 | 5 |
| Hook Island | Mackerel Bay | Friday, 18 December 2020 | -20.088007, 148.951958 | Aerial | 2m | 8 | 30 | 38 | <5 |
| Hook Island | Saba Bay | Friday, 18 December 2020 | -20.105821, 148.945407 | Aerial | 3m | 6 | 18 | 25 | <5 |
| Hook Island | Wrasse Bay | Friday, 18 December 2020 | -20.061406, 148.960551 | Aerial | 2 | 5 | 3 | 8 | <5 |
| Hook Island | Mantaray Bay Point | Friday, 18 December 2020 | -20.060918, 148.955180 | Aerial / UW | 2m | 5 | 14 | 19 | <5 |
| Hook Island | Butterfly Bay | Friday, 18 December 2020 | -20.069074, 148.921877 | Aerial | 2m | 15 | 25 | 40 | <5 |
| Whitsunday Island | Luke Bay | Friday, 18 December 2020 | -20.185113, 148.982653 | Aerial | 3m | 8 | 4 | 12 | 10 |
| Whitsudnays Island | NE Point | Friday, 18 December 2020 | -20.160946, 148.978571 | Aerial | 2m | 7 | 30 | 37 | <5 |
| Whitsudnays Island | Peter Bay | Friday, 18 December 2020 | -20.200119, 148.978867 | Aerial | 2m | 10 | 35 | 45 | <5 |
| Hayman Island | Blue Pearl Bay | Friday, 18 December 2020 | -20.041325, 148.881652 | Aerial / UW | 3m | 12 | 5 | 17 | <5 |
| Hayman Island | East Side | Friday, 18 December 2020 | -20.042101, 148.891124 | Aerial | 2m | 3 | 5 | 10 | <5 |

Figure 17: Data collected using drone (n=40) and underwater (n=10) surveys performed during the project.

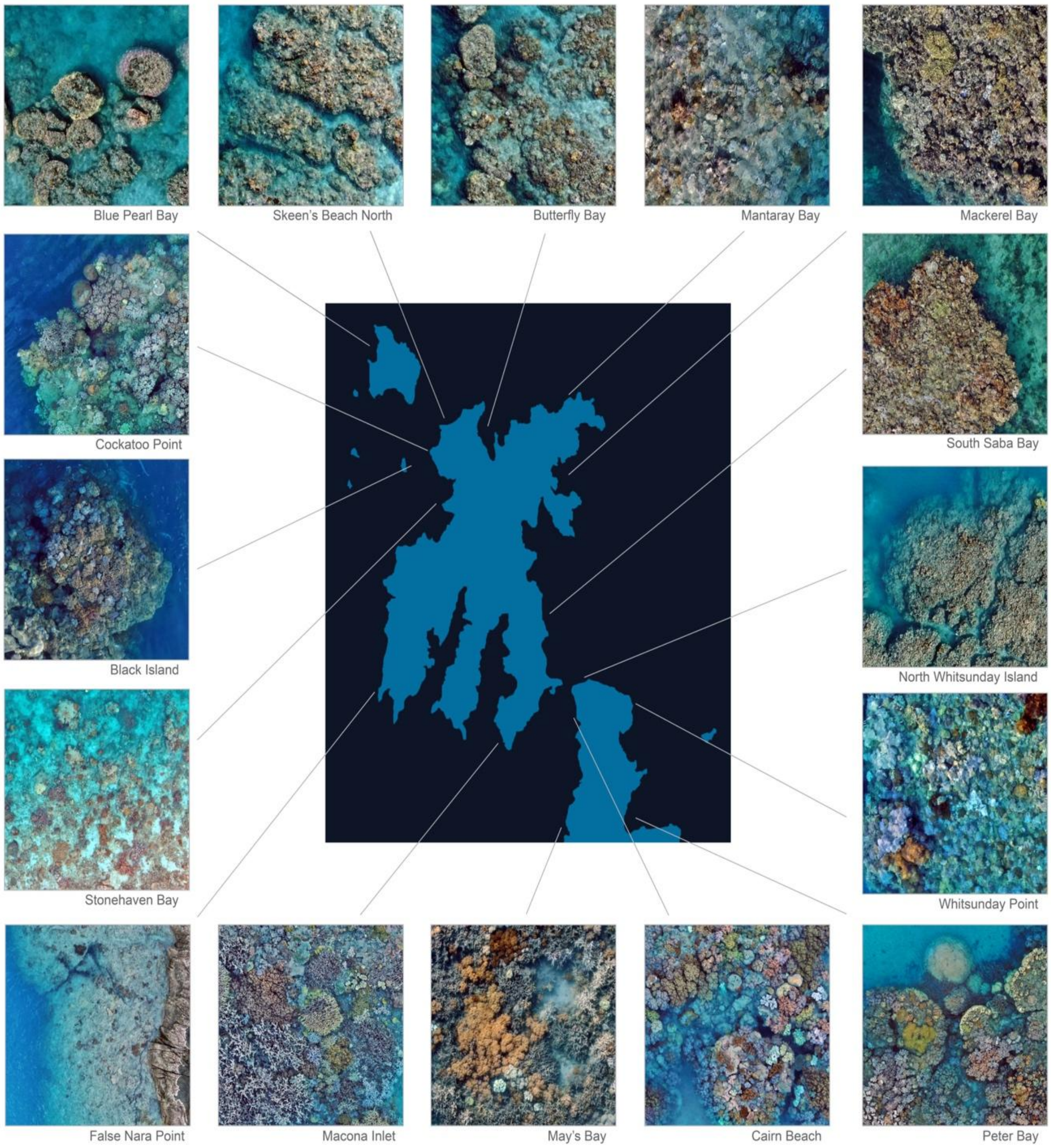


Figure 18: Drone surveys at Hook Island and Northern Whitsunday Island.

Overall, the surveys indicated that post-cyclone coral recovery is very slow across the surveyed islands. Moreover, the dominance of soft corals was observed at almost every site. A key finding of these surveys was not only identifying reef sites with very low coral cover, but also identifying sites that exhibit high levels of hard coral cover which are potentially the strongholds for the region. Excluding known coral-rich areas, three sites that stood out as unexpected 'hard coral havens' were at the southern end of Macona Inlet, SE North Molle Island and the Island around Shute Harbour (Figure 20). Conversely, some areas where high cover of hard corals was expected, based on surveys performed two years ago, now appear as degraded (e.g., False Nara Inlet, Figure 21).

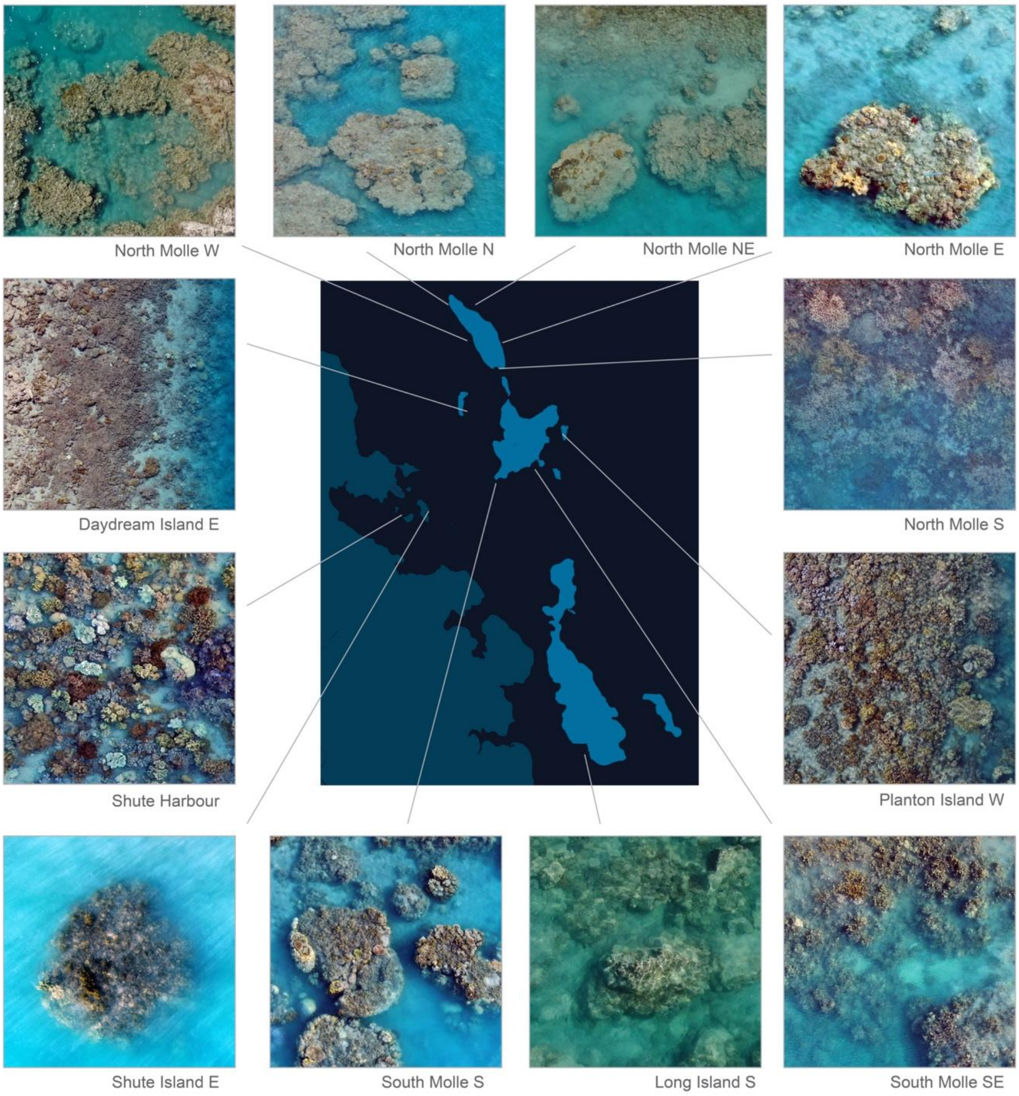


Figure 19: Drone surveys in the Molle Group.

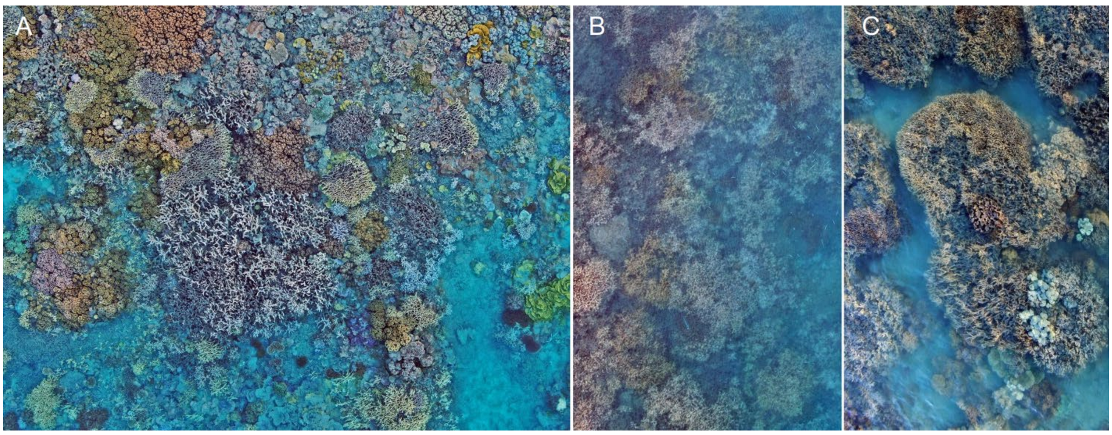


Figure 20: Coral healthy sites identified by drone mapping. A: Macona Inlet Entrance; B: South-East of North Molle Island; Shute Harbour.

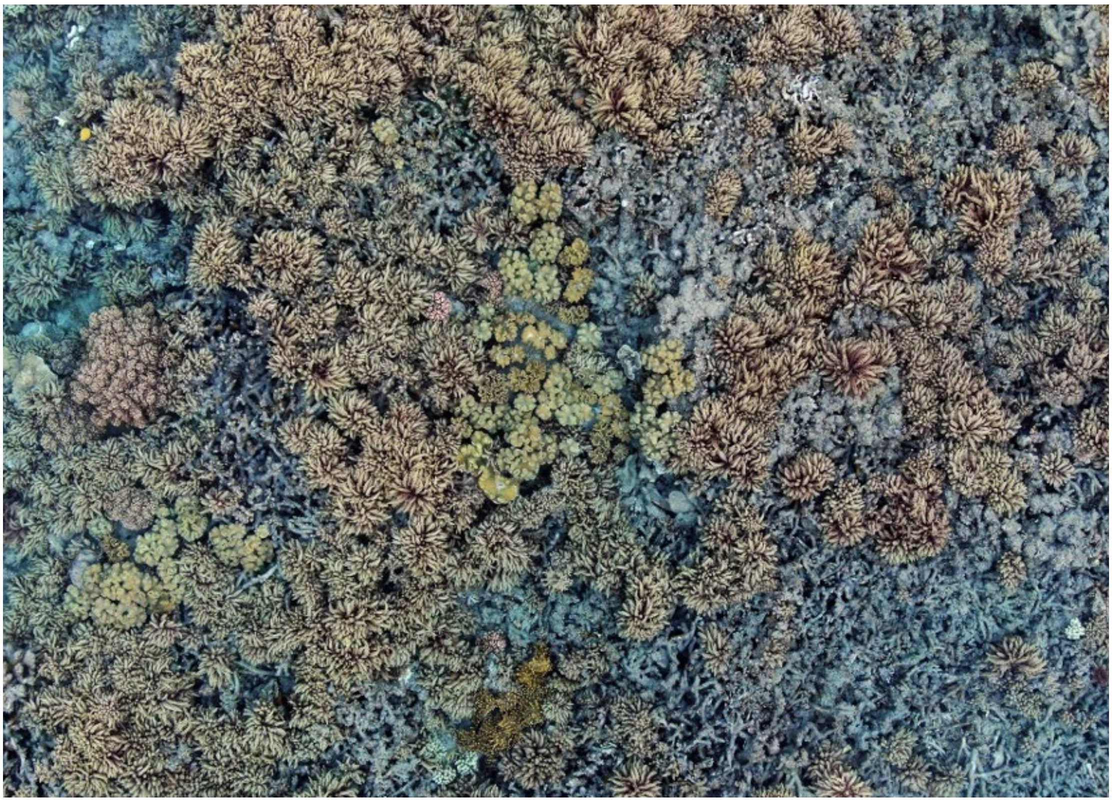

Figure 21: Coral degradation at False Nara Inlet. This site was until recently dominated by a mix of soft and hard branching Acropora corals. A brown algae now dominates the site.

While the period extending from June to December is optimal for operating drone photography in the Whitsundays (higher water clarity, relatively calm weather, favorable tides), drone surveys faced an unexpected above average number of windy days during the survey period, potentially as a result of La Niña climate conditions. This made it very difficult to find days where the wind was below 5 knots during the spring tides. To overcome this, many surveys could only occur very early in the morning, during cloud cover, at low tide on the leeward side of the Larger Islands. In some cases, this meant that a bathyscope had to be used to ground truth the observations.

### 4.4.3 Benthic communities

#### *4.4.3.1 Historical baselines in benthic communities*

The first ecological descriptions of reef communities in the Whitsundays (van Woesik 1993, van Woesik and Done 1997, van Woesik et al. 1995, 1999, DeVantier et al. 1998) have revealed significant reef development both inshore (i.e., near the mainland) and offshore (i.e., around continental islands). Yet, the depth range of the coral zone was variable depending on the ambient turbidity, with maximal extensions (10–12 m depth) mostly occurring in clear waters. Moreover, benthic communities displayed clear depth-stratification, and dominance of fast-growing corals, in particular acroporids (Acropora spp. and Montipora spp.), was restricted to the reef crest and upper slope due to their light requirements (DeVantier et al. 1998, Thompson et al. 2018). Distance to the mainland was reflected by changes in the composition of benthic communities, with higher density of acroporids being found offshore (van Woesik and Done 1997, van Woesik et al. 1999). The lack of large stands of branching *Acropora* corals and massive *Porites* near the river mouths suggested that some inshore reefs have lost their reefbuilding capacity due to increased terrestrial runoff and high capacity of sediment resuspension due to strong tidal currents (van Woesik et al. 1999).

#### 4.4.3.2 Long-term trends of benthic cover

Monitoring data collected by AIMS provide a long-term perspective of ecological changes that occurred across the Whitsundays (Figure 22). Early LTMP data collected in 1993 (Hayman and Langford Islands) and 1995 (Border Island) revealed moderate levels (~20% to ~40%) of hard coral cover that remained fairly stable until 2010. Similar levels and trends of hard coral cover were reported by the monitoring of management zones since 1999 (Williamson et al. 2019). This relative stability suggests that, during this period, reefs may have escaped major disturbances, such as the 1998 and 2002 widespread bleaching events (Thompson and Dolman 2010; Thompson et al. 2014; Williamson et al. 2019).

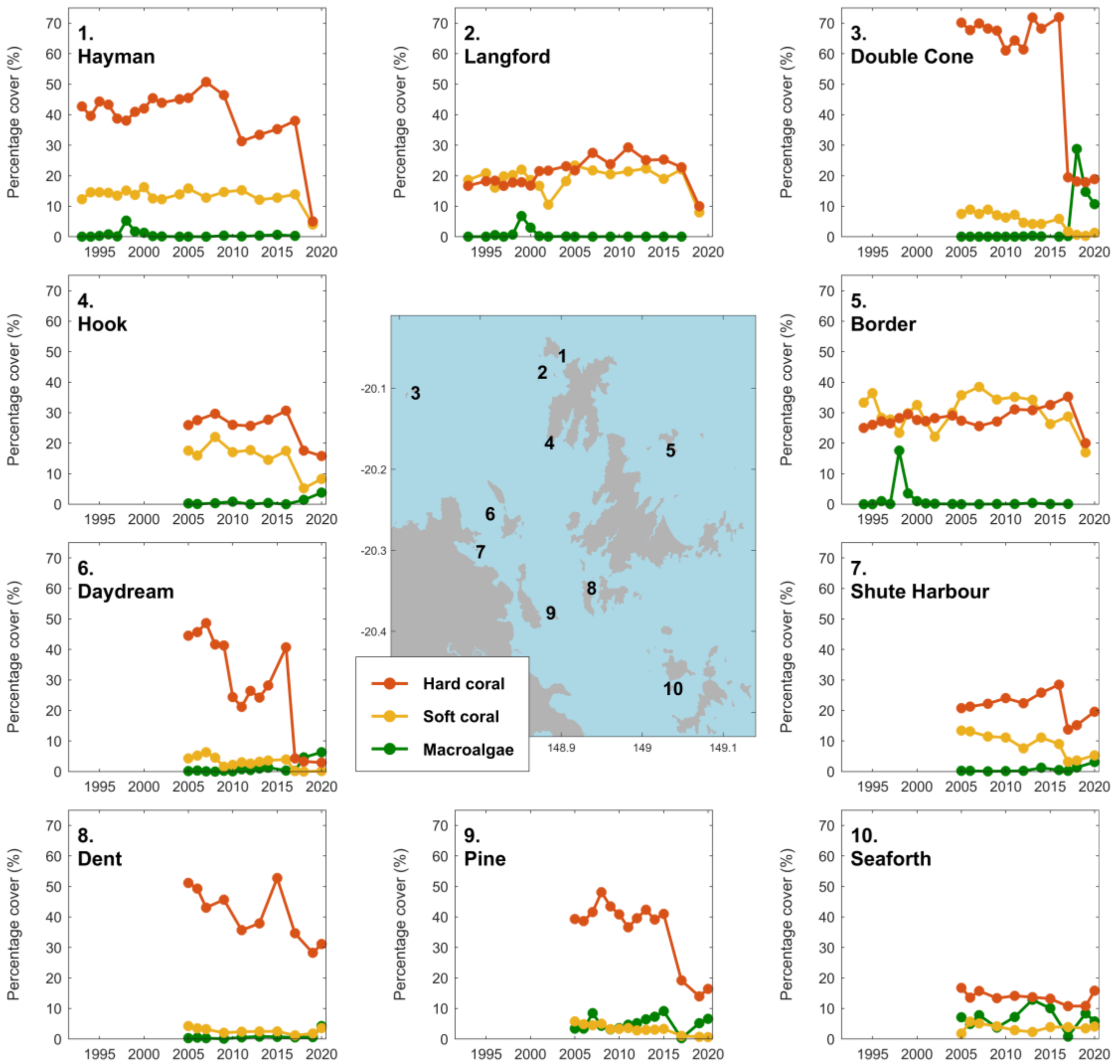


Figure 22: Long-term changes (1993–2020) in benthic communities surveyed by AIMS in the Whitsundays region under the Long Term Monitoring Program (LTMP, reefs 1, 2 and 5) and the Marine Monitoring Program (MMP). Benthic covers averaged across multiple sites (2-3 sites per reef). MMP data selected at 5 m depth to align with LTMP surveys (6 m depth). LTMP reefs were surveyed in early March 2017 just before the crossing of cyclone Debbie.

Seven other reefs were surveyed from 2005 under the MMP (Figure 22). Hard coral cover was within the same range of values (~20–40%) until 2010, with the notable exception of Double Cone Island exhibiting a high 70% cover. Cyclone Ului (category 5) in 2010 caused moderate impacts on some reefs (mostly Hayman, Daydream and Dent Island, (Thompson et al. 2018) quickly recovered in the following years. The other monitoring program (Williamson et al. 2019) only reported a marginal coral decline (-5%) in the northern Whitsundays (post-cyclone survey performed in 2012). Predominant wind and wave direction generated by this fast-moving (yet powerful) cyclone was from the south and southeast, which could explain the lack of significant damages on these northerly exposed reefs (Williamson et al. 2019).

In 2017, Cyclone Debbie generated considerable impacts across the Whitsundays, causing a 40-95% reduction in hard coral cover at 5–6 m depth on all reefs surveyed by AIMS (Thompson et al. 2018) except Seaforth Island in the far south (Figure 22). Similar impacts (55% coral reduction on average) were reported by the other monitoring program (Williamson et al. 2019). In the aftermath of Cyclone Debbie, hard corals were reduced to a low 5–20% cover on AIMS survey sites (except at Dent Is.). Half of the monitored sites of the other program (northern Whitsundays) exhibited less than 15% hard coral cover (Williamson et al. 2019). While some of these impacts may have been confounded with the 2017 marine heatwave (Thompson et al. 2018; Williamson et al. 2019), wave damages and river discharges generated by Cyclone Debbie may have been the strongest disturbance event for Whitsundays corals in the past three decades.

Soft corals were an important component of benthic communities on many reefs monitored by AIMS, especially in the northeast region. Impacts of Cyclone Debbie on soft corals were of similar magnitude than impacts on hard corals (50–70 % reductions). Macroalgal cover remained generally low during 1993–2020 at the monitored sites although showing moderate increases following coral reductions. Higher levels of macroalgal cover in the southern region (Pine and Seaforth) could be a response of increased nutrient availability (Thompson et al. 2018). The monitoring of management zones during 1999–2018 revealed consistently high covers of macroalgae on fished reefs, although a confounding effect of wave exposure cannot be excluded as macroalgal cover was the highest on the eastern side of Whitsunday Island (Williamson et al. 2019; Ceccarelli et al. 2020).

#### *4.4.3.3 Current condition of benthic communities*

The collation of benthic surveys from AIMS, Eye on the Reef (RHIS), local participatory monitoring (J. Gaskell, unpublished data) in the aftermath of Cyclone Debbie (April 2017–2020), complemented with drone imagery and underwater surveys performed during this project (Figure 17), draws a detailed picture of present-day reef health across the Whitsundays (Figures 23-25) from 486 reef sites surveyed across a range of depths (1–6 m, average depth: 2.8 m) in different reef habitats (reef flat, crest and slope).

Present-day hard coral cover was estimated to be 11% on average, with only 9 sites (2% of all sites) exhibiting cover values above 40% (Figures 23). Half of the reef sites were below 8% hard coral cover. There were no clear spatial patterns of hard coral cover across the region.

Soft coral cover was estimated to be 11% on average, with 17 sites above 40% (Figure 24). Half the reef sites were below 5% soft coral cover. Higher cover values of soft corals were generally observed around Hook and Whitsunday Islands.

Macroalgal cover was estimated to be 14% on average, with 53 sites (11% of all sites) exhibiting macroalgal cover values above 40% (Figures 25). Half the reef sites were below 5% macroalgal cover. The distribution of macroalgae seems to follow a North-

South gradient with generally higher macroalgal cover towards the south of the Whitsundays.

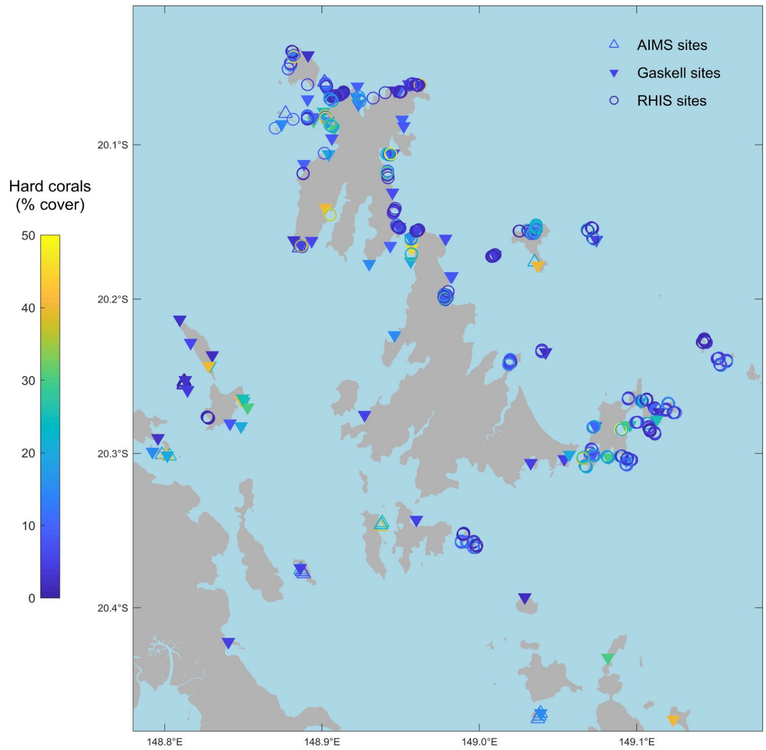


Figure 23: Hard coral cover in the Whitsundays post-cyclone Debbie (from April 2017 to December 2020). Symbols refer to sites surveyed by each monitoring program. Color code proportional to the percentage cover (sites with no data were surveyed before the selected period).

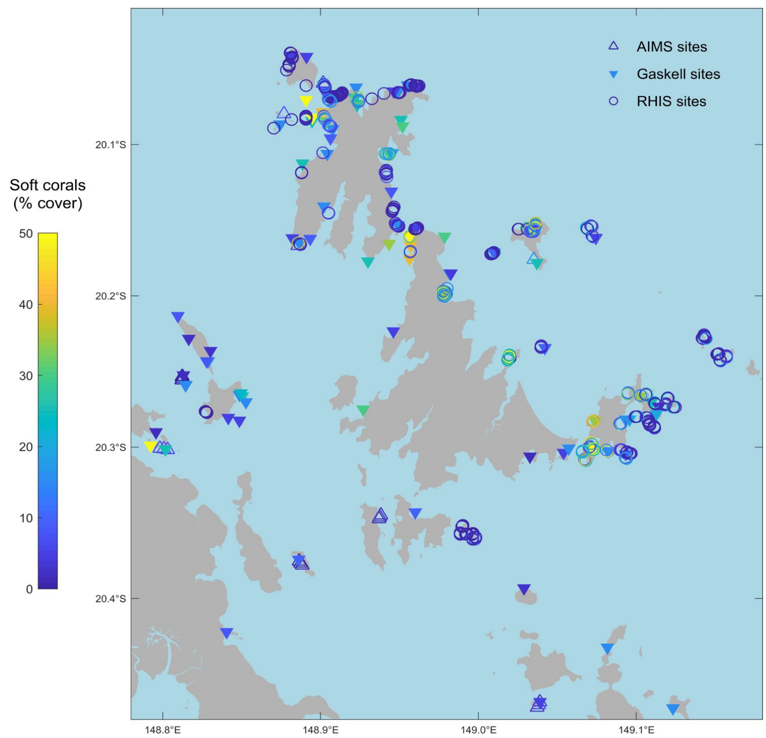


Figure 24: Soft coral cover in the Whitsundays post-cyclone Debbie (from April 2017 to December 2020). See Figure 23 legend.

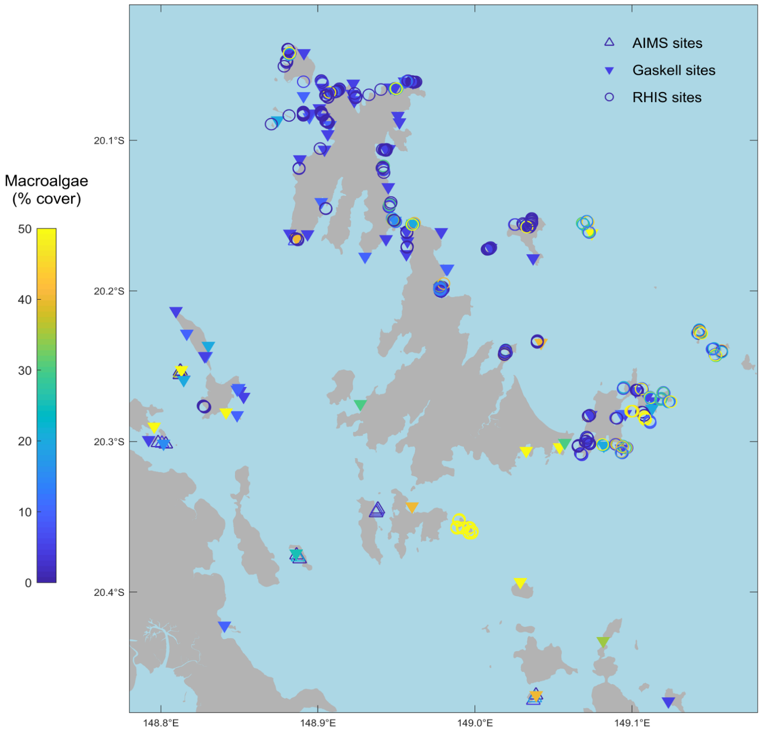


Figure 25: Macroalgal cover in the Whitsundays post-cyclone Debbie (from April 2017 to December 2020). See Figure 23 legend.

While these observations are necessary to identify and locate the needs for coral restoration, it must be emphasised that degraded reefs following severe disturbances often undergo a phase during which not much coral recovery is visible. Natural recovery can be delayed for several years, especially in the presence of unconsolidated rubble fields. Reefs extensively damaged by cyclone Debbie should be monitored for tracking recovery abilities (i.e., coral recruitment, natural stabilisation of rubble), as early signs of recovery may change restoration needs.

## 4.5 Larval dispersal and connectivity

### 4.5.1 Long-distance dispersal of coral larvae

Particle tracking using CONNIE3 (with currents simulated by the hydrodynamic model eReefs-GBR1) was used to indicate the location of source reefs that had the potential to supply coral larvae to the Whitsundays during 2014–2019 (Figure 26). Among the 1,544 reef polygons from which larval particles were released (including 221 polygons within the Whitsundays), reefs contributing to larval supply to the Whitsundays varied between years due to vagaries of ocean currents. Recurrent patterns of large-scale dispersal emerge in individual years, with coral larvae coming mostly from mid-shelf reefs lying either in the North-West (2015, 2018 and 2019) or in the East-South-East (2016, 2017) up to 300 km away from the Whitsundays. More diffuse larval dispersal may have occurred in 2014 with larvae coming from both regions but with lower levels of supply overall. This indicates that the Whitsundays would mostly receive external larvae from long-distance dispersal rather than from nearer mid-shelf reefs located 30-50 km away (e.g. Hardy Reef). Importantly, these estimates of larval trajectories should be considered only as *potential* since they do not account for the actual coral brood stock of source reefs. As such, the connectivity patterns only reflect dispersal possibilities driven by prevailing currents; for instance, a coral-depleted reef will obviously not contribute much in larval supply although depicted as an important source from dispersal simulations.

Across all years, the Whitsundays region was an important source of larval supply through reef retention and short-distance dispersal. Importantly, these estimates of larval output are proportional to the area of reef polygons. While reefs of the Whitsundays are generally small (mean polygon area: 4.7 ha) compared to the surrounding reefs (mean polygon area: 58.8 ha), the amount of larvae potentially retained within the region is substantial due to high survival rates over short dispersal periods. This suggests that inter-reef supply within the region is at least as important as external sources in providing coral larvae for Whitsundays reefs.

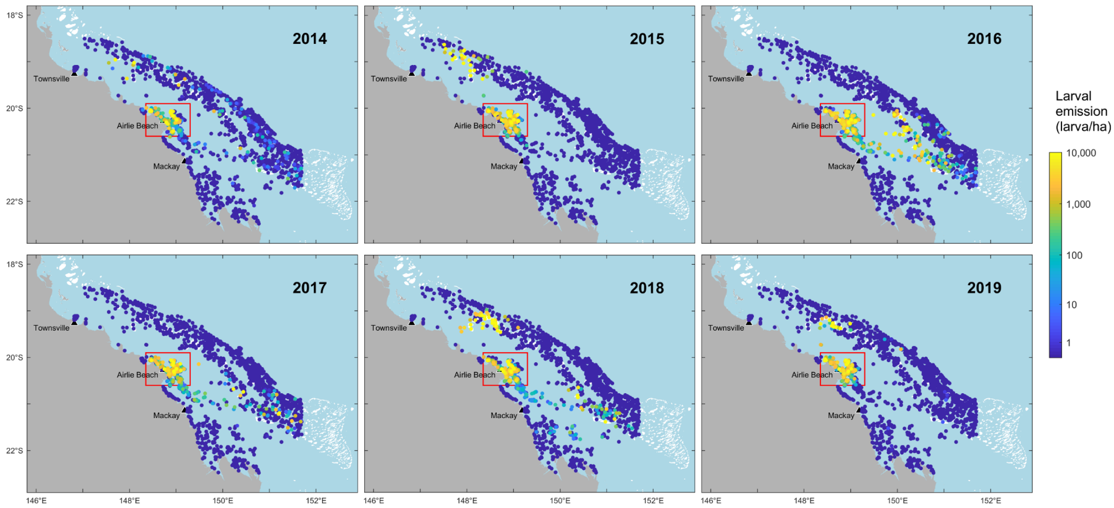


Figure 26. Source reefs of coral larvae to the Whitsundays as indicated by annual emission of coral larval particles estimated from CONNIE3 particle tracking. Larval emission (logarithmic scale) is expressed as the number of particles released per hectare of source reef that arrived within the boundaries of the Whitsundays region (red box). This number is indicative and depends on the amount of particles released per polygon. Here, 1,000 particles were released from 1,544 reef polygons (including 221 in the Whitsundays) defined by the GBRMPA indicative reef boundaries (GBRMPA 2007). Polygons without estimate of larval output were not included in simulations.

For many reefs of the Whitsundays, the amount of larval particles received from outside and within the region varied substantially between years (Figure 27), reflecting the inter-annual variability of the main sources of particle emission. As such, high interannual variability in coral recruitment is expected to occur across the Whitsundays; fluctuations in the size of coral brood stock across reefs will make predictions of coral recruitment and recovery potential even more challenging. Importantly, these predictions do not account for the settlement potential of the receiving reefs. Settlement success depends on the availability of suitable space and on factors affecting post-settlement survival. For instance, a receiving reef covered by a thick layer of sediment, or dominated by macroalgae or soft corals, is unlikely to experience high rates of coral settlement, even if high larval supply is predicted from the dispersal patterns.

Some reefs, however, show consistency over time in the amount of received larval particles. Overall, reefs in the north and east of the Whitsundays tend to receive more consistent supply than reefs in the south. This pattern may result in spatial differences in the ability of coral populations to recover after disturbance. Importantly, fluctuations in larval supply are not clearly related to the previously described temporal patterns of long-distance dispersal, possibly due to the influence of local retention and short dispersal within the Whitsundays. While this influence seems to mainly benefit reefs located in the north and east of the region, the underlying mechanisms (i.e., local retention or short-distance dispersal from reefs of the southern Whitsundays) remain unclear at this stage.

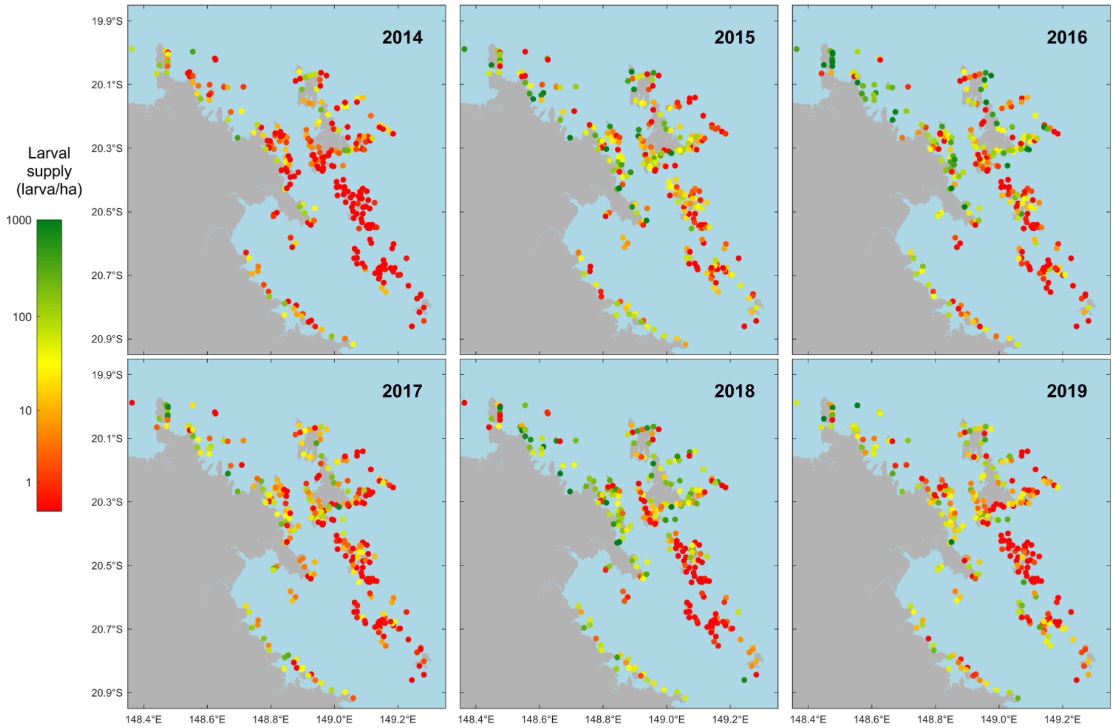


Figure 27. Cumulative annual larval supply in the Whitsundays as estimated by particle tracking simulation CONNIE3 (2014–2019). Each dot represents a reef area as defined by the indicative reef boundary (n=221 reef polygons). The color scale is proportional (on a logarithmic scale) to the incoming number of larvae after dispersal from 1,544 reefs of the GBR, including the Whitsundays.

Predictions of larval supply to Whitsundays reefs were averaged over the six years of particle tracking (Figure 28) and separated following the location of source reefs (i.e., larval supply from reefs located outside vs. within the Whitsundays). Results indicate that both external and within-region supply tend to be higher in the east and north of the Whitsundays, including reefs around Hayman and Hook islands. Reefs in the south (i.e., the south-western flank of Whitsunday Island, Dent and Hamilton islands, the Lindeman Group) exhibit very low levels of external larval supply, possibly because larval particles were already captured by reefs positioned further upstream along their dispersal path. This ‘shadow’ effect, whereby upstream reefs prevent larval supply from reaching downstream reefs, needs to be cautiously considered when interpreting patterns of larval dispersal within the complex landscape of the Whitsundays.

As a result, reefs in the south might be supplied by surrounding reefs via short-distance dispersal. Because reefs in the far south-east of the Whitsundays did not receive larval particles from other reefs in the region, it can be postulated that dispersal within the Whitsundays is mostly directional and that prevailing currents during the spawning season follow a north-westerly direction. This suggests that southern reefs of the Whitsundays may be prone to infrequent and low levels of coral recruitment, potentially affecting their capacity to recover after disturbance. Coral recovery on those reefs might be reliant on the health of coral populations located further south.

Conversely, reefs located in the north and east would appear to exhibit more consistent rates of larval supply from both short- and long-distance dispersal. Note that this does not necessarily imply faster rates of coral recovery since the success of recruitment would also depend on the local environment affecting larval settlement and post-settlement coral demographics. Nonetheless, from the current batch of dispersal simulations it can be anticipated that these reefs offer the best scope for recovery since they might be less reliant on the health of surrounding coral populations.

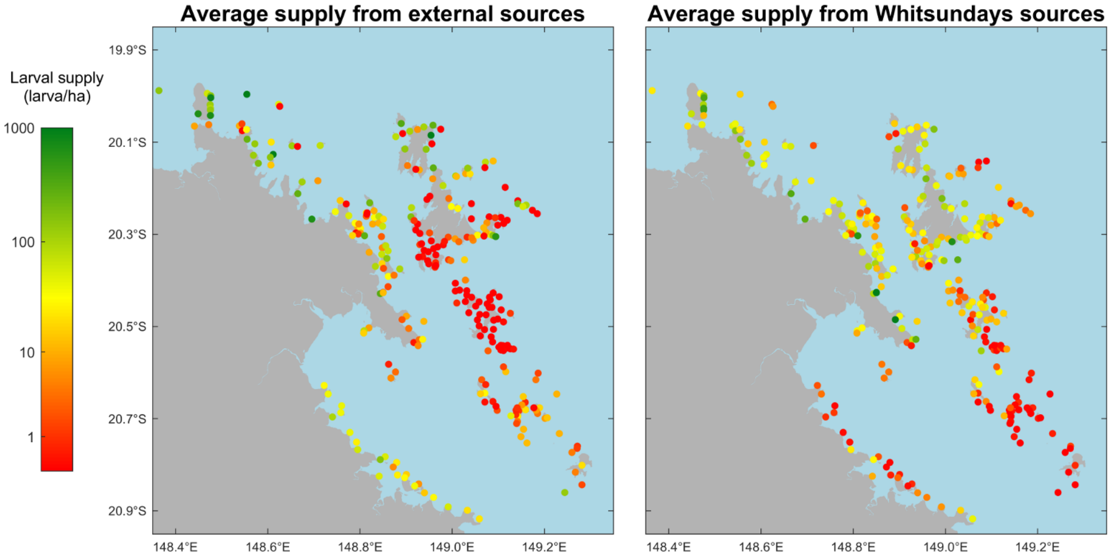


Figure 28: Patterns of larval supply as estimated by particle tracking simulation using eReefs-GBR1/CONNIE averaged over 2014–2019. Left: incoming larvae from source reefs located outside the Whitsundays (n=1,323 source polygons). Right: incoming larvae from source reefs located within the Whitsundays (n=221 source polygons). Connectivity strength expressed as cumulative number of larvae released/received per hectare of reef habitat.

### 4.5.2 Short-distance, fine-scale predictions of coral connectivity

RECOM particle tracking simulations performed at 200 m resolution between reef polygons defined by the geomorphic map (Figure 6) provided two metrics of short-distance larval exchange:

- the likelihood of larval sink, estimated by the number of source reefs that a reef receives larval particles from (Figure 29), which identifies potential 'super-receivers' of coral larvae – these reefs may have greater recovery abilities as a connection with more source reefs increases the chance to receive more larvae;
- the likelihood of larval source, estimated by the number of sink reefs that a reef supplies larval particles to (Figure 30), which identifies potential 'super-spreaders' of coral larvae – these reefs may promote coral recovery on other reefs.

These metrics were calculated for four species characteristics of larval dispersal (larval mortality and duration of the competency period) and averaged over the five spawning seasons (2014–2018). Here, we focus on metrics obtained for the two species exhibiting the most different dispersal abilities (Figures 29-30): *Acropora millepora* and *Goniastrea retiformis*. Importantly, particle tracking was simulated within each RECOM separately (i.e., with no cross-boundary exchanges) so that connectivity metrics are likely underestimated, especially near model boundaries. Unlike particle tracking with CONNIE3/eReefs-GBR1, a particle that passed over a reef was not considered as settled but would continue its dispersal until the end of the simulation (i.e., upstream reefs do not create a 'shading effect' on downstream reefs).

The likelihood of larval sink was highly heterogeneous within each sub-region (Figure 29). This suggests important differences in larval supply among reefs separated by short distances. Larval particles with the shortest period of competency and survival (species *G. retiformis*) had greater capacity to settle close to an emitting reef, generating a greater number of super-receivers within each sub-region. In general, the same reefs were identified as low receivers for both species, an indication of their relative isolation from dispersal routes (or possible boundary effects where applicable).

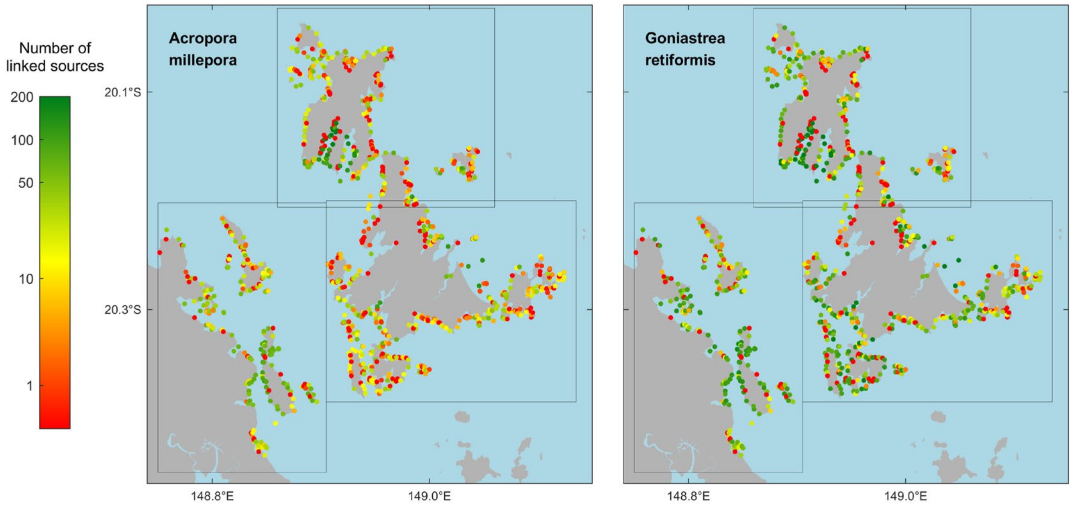


Figure 29: Likelihood of larval sink as the number of source reefs each reef receives larval particles from (i.e., number of links from source reefs), based on RECOM particle tracking (200-m resolution). Reef polygons (1,055 in total) were defined based on the satellite-derived geomorphic map (Figure 6). Particle dispersal was simulated within the each RECOM (i.e., without cross-border exchanges) so that connectivity patterns only reflect short-distance dispersal (i.e., within each delimited region). Numbers were averaged over the five spawning seasons (2014–2018).

A greater homogeneity was obtained in the likelihood of larval sources (Figure 30) which characterises the potential of reefs to spread larvae to surrounding reefs. Larval particles with the shortest period of competency (*G. retiformis*) generated the greatest amount of super-spreaders reefs due a greater capacity of larvae to be distributed over short distances. Inversely, a slower acquisition of competency (*A. millepora*) delays settlement, which tends to decrease the likelihood of dispersal and settlement at short distances from sources. For these latter larval traits, Hook Island exhibited the greatest potential for super-spreaders to emerge.

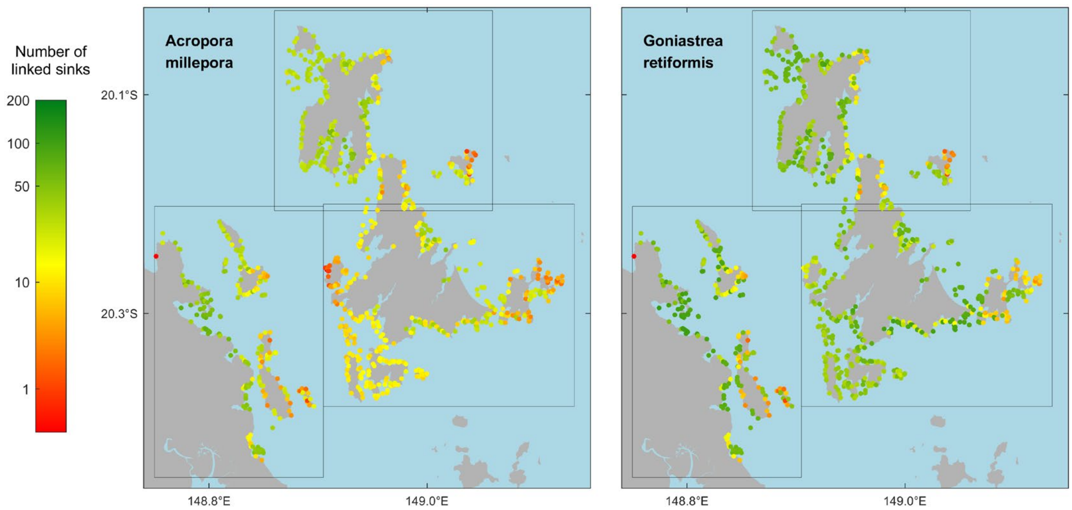


Figure 30: Likelihood of larval source expressed as the number of sink reefs each reef sends larval particles to (i.e., number of links to sink reefs), based on RECOM particle. High values indicate reefs that supply coral larvae to many other reefs. See Figure 29 legend.

### 4.5.3 Considerations on coral connectivity predictions

Connectivity links in both sets of simulations should be viewed as *potential* larval exchanges between reefs, not as *realised* patterns of larval dispersal. Results of particle tracking indicate which connectivity links (and their strength) are possible between polygons identified as 'reefs', based on the predicted velocity and direction of ocean currents at the time of coral spawning for each simulated year. This implicitly assumes that every source polygon is occupied by healthy coral populations and that receiving reef polygons are suitable for coral settlement. However, actual (i.e., realised) demographic connectivity will depend on the brood stock of corals and the amount of larvae produced on each source reef, the changing environmental conditions along dispersal paths, such as changes in temperature and salinity, as well as whether the receiving reef offers adequate conditions for coral recruitment (availability of free hard substratum) and post-settlement survival.

These limitations must be carefully considered before integrating connectivity features in restoration planning. First step is to ensure that the connected polygons are correctly classified as suitable habitats for corals; connectivity properties can be easily re-assessed after exclusion of non-suitable reef habitats. If the restoration strategy is to target reefs that exhibit high dispersal potential, an efficient spatial prioritisation would require evaluating the scope for coral settlement on reef habitats supplied by the restored sites; restoring a super-spreader reef that supplies habitats overgrown by macroalgae might not enhance coral recruitment on those habitats. If the strategy is to exclude reefs with high larval supply based on the assumption these reefs have greater potential for recovery, attention should be paid to the current health of their larval sources because recovery may be delayed until the brood stocks of source reefs first recover.

In any case, the natural variability of larval supply deserves careful consideration, and despite the inclusion of multiple spawning seasons, this variability is probably underestimated. Moreover, using different larval trait characteristics produce notable differences in the predicted patterns of dispersal. With this in mind, more attention should be paid to reefs that consistently indicate (i.e., over time and/or between species) high/low levels of larval supply or dispersal, because connectivity predictions for these reefs might be less uncertain.

Here, two modeling methods were used to resolve coral connectivity at different resolutions. Each method captures different properties of ocean circulation and particle behaviour to provide a complementary assessment of coral larval dispersal in support of coral restoration. Connectivity at 1 km resolution identifies long-dispersal routes that can bring coral larvae to the Whitsundays. It also informs about the relative importance of short- vs long-distance dispersal in the making of larval supply across the Whitsundays.

This representation can be refined using the connectivity patterns resolved at 200 m resolution. Here, the selected method of particle tracking does not generate a 'shading effect' of upstream reefs, which makes it more suitable for assessing fine-scale larval connectivity across complex seascapes of reef habitats. In particular, it allows assessing larval dispersal across the successive habitats of a same reef zonation (e.g., from the reef slope to back reef habitats). One important limitation is that running particle tracking at 200 m resolution for a large region like the Whitsundays is computationally demanding, and required subdividing the system in three separate sub-regional models. A drawback is the lack of connections between each modelled sub-region. Note that fine-scale connectivity was analysed in terms of number of links (sources/sinks) instead of connectivity strength (percentage likelihood of a link). Assessing source and sink potentials using connectivity strength produced similar patterns and conclusion than the number of links.

## 4.6 Growth potential of corals

Spatial variations in the growth potential of corals were obtained by simulating coral cover increments as a response to suspended sediment concentrations predicted at 200 m resolution for the 2017 wet and dry seasons (Figure 12). Simulations were performed for the 1,055 reef polygons identified by the geomorphic classification (Figure 6). The simulated growth rates (standardised from an initial 5% coral cover) allow visualising the recovery potential of corals in response to water quality (Figure 31).

Importantly, the modelled response is representative of *Acropora* dominated communities that are typically observed on offshore environments of the GBR (Bozec et al. 2020). Relationships between suspended sediment and coral growth for inshore coral communities, where turbid-tolerant species are more abundant, are yet to be developed. However, growth potential can inform the spatial prioritisation of coral outplanting, which often relies on the deployment of acroporid corals.

Another important limitation is that only one year of suspended sediment was available for simulating coral growth at 200 m resolution. The 2017 water year is representative of an extreme wet season as Cyclone Debbie generated large river floods and, most likely, an intense resuspension of deposited sediment. Therefore, this map should be viewed as a pessimistic representation of the potential of coral growth, especially in areas that would be lightly exposed to river discharges during an average, moderate wet season. As previously noted, this representation is useful for restoration planning as it indicates which reef environments are expected to escape extreme flood events.

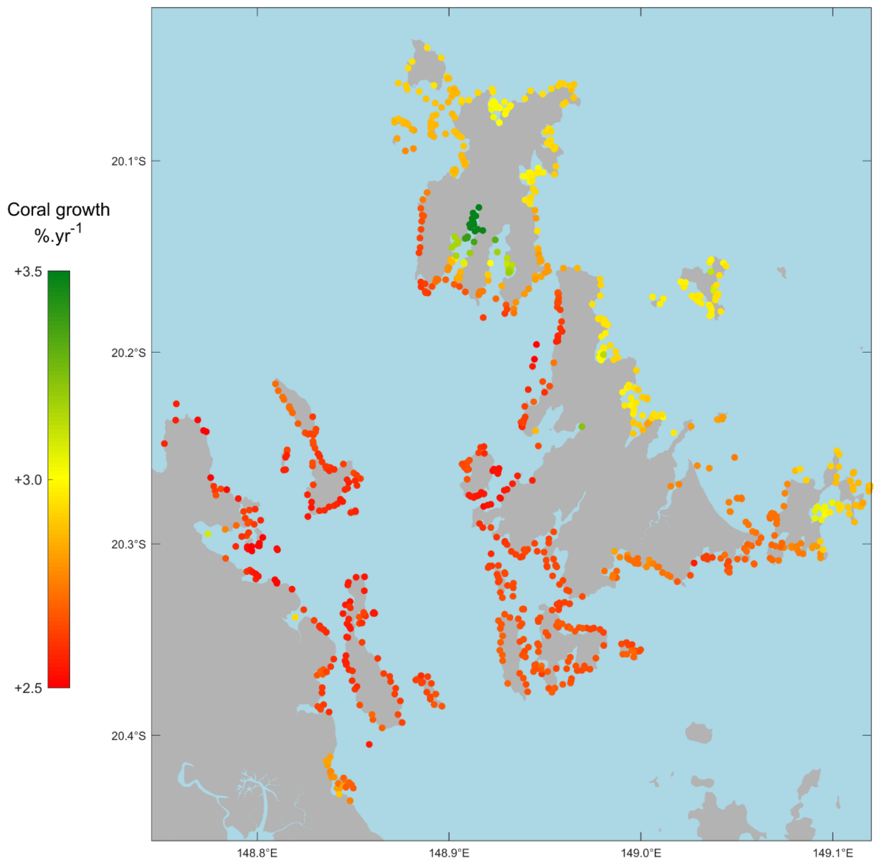


Figure 31: Simulated growth potential of corals as one-year increment of hard coral cover function of suspended. Growth predictions are based on the mean annual SSC during 2017 predicted by RECOM (see Figure 12). Because coral cover growth depends on the initial value of coral cover, growth predictions were standardized for a hypothetical 5% hard coral cover on all reefs.

Finally, it must be emphasised that the predicted growth rates are only function of suspended sediment concentrations and do not account for temporal and spatial variations in the success of coral recruitment, the ability of corals to growth and survive under competitive interactions with algae and soft corals, the influence of wave energy, etc. Future model developments under the Reef Restoration and Adaptation Program (https://gbrrestoration.org/) will attempt at integrating these processes for better predictions of coral habitat suitability across the GBR.

## 4.7 Exploration of coral outplanting scenarios

Simulated scenarios of coral outplanting at different sizes and densities provide a first set of expectations about the benefits of outplanting under increased suspended sediment concentrations (Figure 32). We simulated the deployment of coral nubbins of the group '*Acropora* corymbose' at years 0, 1 and 2, and tracked the associated benefits on total coral cover until full recovery was achieved (~15-20 years) in the absence of perturbation (cyclone, bleaching). Restoration benefit was measured as the difference in total coral cover (6 coral groups combined) between the restoration and baseline scenarios predicted at year 5, i.e., 5 years after the first deployment (Figure 5). Note this assumes no external disturbance (cyclone, bleaching), nor that the reef is recruitment limited. The only forcing variable was suspended sediment which retards coral recovery (Figure 3).

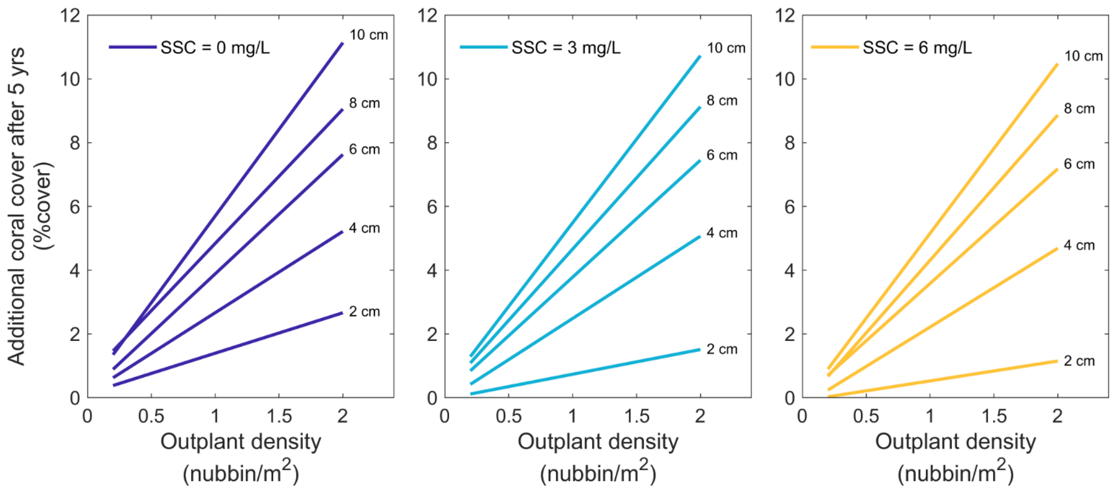


Figure 32: Simulated benefits of coral outplanting for hypothetical reefs under increasing suspended sediment concentration (SSC): 0 mg/L (left); 3 mg/L (middle); 6 mg/L (right). Benefits are expressed as the net difference in total coral cover between the restoration scenario and a scenario without outplanting after 5 years of recovery (see methods in Figure 5). Coral nubbins were deployed at year 0, 1 and 2, at increasing densities (0.1 to 2.0 nubbins/m$^2$) and diameters (from 2 to 10 cm). In a given scenario, all deployed nubbins had the same diameter size.

Simulations show that the benefits of coral outplanting after 5 years increase linearly with the deployed densities of coral nubbins. For a given nubbin density, total coral cover benefits increased non-linearly with the size of coral nubbins (slopes becoming steeper as nubbin diameter increases). Suspended sediment had limited impact on the restoration benefit *per se*, except for the lowest nubbin sizes (≤4 cm). This is because in the model SSC only affect the demography of juvenile (higher mortality and slower growth of juveniles when exposed to high SSC).

These results should not be considered as benchmarks. In the real world, benefits of coral outplanting will depend on many other local factors such as, for example, rates of recruitment, conditions for growth (light, water flow, abundance of competitors) and exposure to disturbances (storms, heatwaves, diseases, predation) which were not considered in simulations. However, these predictions are useful in providing expectations about the relative benefit that different outplanting strategies could achieve, all other drivers of coral recovery being equal. They offer a quantitative framework for comparing the relative efficiency of different strategies based solely on the density and size of outplants. Future model developments will allow for a more flexible parameterisation of coral outplant vs. native corals (e.g., growth, survival).

Simulation outputs can be used to derive a general equation of the expected benefit of coral outplanting after 5 years of recovery for a given value of nubbin density and diameter in a given SSC environment:

$$B_{5yrs}(\%coralcover) = 3.34 \times density + 0.61 \times diameter - 0.13 \times SSC - 3.25 \quad \text{(Eq 1)}$$

Note this equation was obtained assuming 3 deployment of coral outplants (on year 0, 1 and 2), using nubbins of corymbose-acroporid coral types (e.g., *Acropora millepora*, *Acropora valida*, *Acropora humilis*) and that the modelled reef had constant larval supply. Because benefits are additive, equation 1 allows extrapolating the benefits of more complex scenarios that combine different outplant diameters and densities (Table 4).

Table 4: Example calculation using Equation 1 to predict the net benefit of coral outplanting from a mix of nubbin sizes. This calculation assumes additivity of benefits associated to each nubbin diameter and density of deployment, so that the total net benefit can be estimated as the sum of the benefits predicted for density-at-size nubbins. This assumes a deployment at year 0, 1, and 2 with the same design (i.e., same density/size) for each deployment.

| Diameters (cm) | Densities (nubbin/m$^2$) | Benefits at year 5 (% cover) SSC = 0 mg/L | Benefits at year 5 (% cover) SSC = 4 mg/L |
|---|---|---|---|
| 4 | 0.5 | +0.9 | +0.3 |
| 5 | 0.7 | +2.1 | +1.6 |
| 7 | 0.2 | +1.7 | +1.2 |
| 10 | 0.8 | +5.5 | +5.0 |
| **Total** | **2.2** | **+10.2** | **+8.1** |

## 4.8 GIS tool

The GIS tool was created with ArcMap 10.8 and shared with all the stakeholders involved in the project. It is accessible upon request to the Great Barrier Reef Foundation.

# 5 A decision framework for the prioritisation of reef restoration

Prioritising reefs for restoration not only requires integration of a large amount of spatial information but also a definition of the targeted spatial scale of restoration (i.e., landscape/island/regional) as this will strongly influence the spatial strategy and choice of interventions. The decision framework proposed here relies on the following considerations:

- Reefs in the Whitsundays appear globally depleted following severe impacts of cyclone Debbie (possibly combined with recent bleaching events);
- Regional loss of coral brood stock may impair short-distance larval dispersal to support coral recovery;
- Biogeochemical forcing influenced by river inputs and wind-driven resuspension may compromise coral recovery due to (1) high suspended sediment that impede early-life coral demographics and (2) nutrient-enhanced algal productivity that out-compete corals;

Based on these considerations, the decision framework was developed with the objective of identifying and selecting sites where reef restoration would facilitate cross-scale coral recovery (at reef, island and regional levels). With this objective in mind, three selection criteria were considered relevant:

(1) The avoidance of non-favourable conditions for coral development that cannot be easily mitigated by management. Suspended sediment impede coral recovery (Figure 3) by reducing the success of coral fertilisation, larval settlement and the survival and growth of coral recruits (Jones et al. 2015; Humanes et al. 2017). Yet, reducing inputs of terrigenous sediment through efficient management of river catchment is unlikely to improve water quality in the short term: reef environments may remain turbid for several decades due to the resuspension of deposited sediment under the action of waves and tidal/wind-driven currents. Exposure to suspended sediment during the 2017 wet season was represented for each nominal reef habitat identified by the satellite-based geomorphic layer (Figures 33, 34, 35). A greater potential of coral growth would be expected on reefs with minimal exposure to suspended sediments.

(2) The consideration of sub-optimal conditions for coral development that could be improved through appropriate management. Because macroalgae can have detrimental effects on coral colonisation, coral restoration might be avoided in areas of high macroalgal productivity. Yet, the expansion of seaweed can possibly be mitigated with targeted interventions (reduction of nutrient loads from regulated agricultural practices, protection of fish herbivores, on-site removal of seaweed), which, if deemed affordable and efficient, can increase the scope for coral restoration. We approximated the risk of seaweed expansion by estimating the flux of nutrient at the seabed, obtained by multiplying orbital velocity with the average nutrient concentrations (DIN) during the 2017 wet season (Figures 33, 34, 35). Because high water velocity increases nutrient uptake by macroalgae, nutrient flux may be a good predictor of seaweed productivity. This does not imply, however, that abundant seaweed is observed where nutrient flux is high, as other factors influence seaweed populations (e.g., herbivore biomass, irradiance).

(3) The consideration of coral larval connectivity (Figures 33, 34, 35) from the standpoint of likelihood of larval sink (scope for coral recruitment) or larval source (scope for larval supply to other reefs) – this choice would lead to different restoration strategies :

- The likelihood of larval sink gives an indication on the potential of a reef to recover through coral recruitment. Reefs poorly connected to larval sources are more likely to experience low rates of larval supply; their recovery abilities may be impaired due to limited recruitment, especially in areas of high algal productivity as macroalgal expansion impedes coral recruitment by reducing the space for larval settlement. Active coral restoration might be needed in these areas to increase recovery capacities. Inversely, reefs connected with many source reefs are less likely to be recruitment-limited and may recover naturally. However, active coral restoration would boost their recovery capacities.

- The likelihood of larval source gives an indication on the potential of a reef's coral brood stock to supply coral larvae to surrounding reef habitats thus contributing to their recovery. Source reefs that are connected to multiple sink reefs may be prioritised for restoration considering that a restored brood stock would benefit non-restored reefs through increased larval supply.

Combining likelihoods of larval source and sink may help identify spatial prioritisations that make the best use of larval connectivity for spreading the benefits of restoration across the seascape. For example, rather than restoring actively a reef that is recruitment-limited, it could be more efficient to boost the recovery of its larval sources to achieve greater coral recruitment on that reef. Note that the present connectivity metrics were calculated assuming all the geomorphic polygons were correctly classified as reefs and that they are suitable for coral colonisation. Yet, a reef that is linked to many larval sources could be misrepresented as a likely larval sink (good potential for recovery) if most of the identified sources are not suitable reef habitats. Conversely, a reef that supplies larvae to many non-suitable reef habitats would not be considered as an effective super-spreader. However, connectivity metrics can be easily re-calculated after validation of the geomorphic classification (i.e., suppressing misclassified polygons) and the deletion of dispersal links from/to non-suitable reef habitats.

As previously mentioned, this assessment of habitat suitability for corals must be considered with caution as only one year of water quality was simulated. Had multiple years of water quality been available, some of the reefs appearing as 'non-suitable' (high suspended sediment or macroalgal productivity) could have displayed more favourable conditions. Moreover, this assessment is representative of an extreme wet season influenced by a powerful cyclone (Cyclone Debbie). While further work would be needed for more robust predictions of water quality at high resolution, the present assessment can be used to identify reefs that are the most likely to escape extreme flood events.

Routine (non-cyclonic) orbital velocity (Figure 7) was used to derive spatial predictions of nutrient uptake and estimate the potential for macroalgal productivity, but it can also be used to infer the risk of wave damages to corals. An important limitation is that a quantitative link between orbital velocity and coral damage (colony dislodgement and fragmentation) is currently lacking, and this prevents the definition of threshold velocity values above which significant coral damage is expected. A conservative approach would tend to avoid reefs displaying the highest orbital velocity for specific restoration interventions, such as the outplanting of fragile coral morphologies. Note, however, that water flow is important for coral physiology (removal of deposited sediment, flow-related feeding and oxygen uptake) so that reef environments with the lowest orbital velocity are not necessarily the best for coral growth. Indeed, in the past, some reef areas exposed to moderate currents seem to have flourished in the Whitsundays despite high levels of turbidity levels (van Woesik et al. 1999). With a tidal range up to 4 m, tidal currents are strong in this region and could be a good predictor of coral performance in combination with orbital velocity. Bottom orbital velocity reflects the wave-driven motion of particles moving back and forth over short distances (< tens of metres) whereas tidal and wind-driven current velocities relate to the net motion of particle transport over longer distances. While assessing mean current velocities would require re-running RECOM at

200 m resolution, a possible faster alternative is to use the SLIM model applied to GBR[1] which achieves similar resolution close to reefs and coastlines (Thomas et al. 2014).

Lastly, the other layers acquired in this project can be integrated to complement this framework. Observations of coral cover after Cyclone Debbie allow setting restoration needs based on the current level of degradation of coral populations. The historical baselines of coral cover can be used to corroborate the habitat/benthic cover maps and the spatial predictions of water quality. Current observations of macroalgae can be combined to the spatial predictions of nutrient flux to substantiate the risk of macroalgal productivity. Predictions of cyclone-driven wave exposure can assist in selecting reefs with the lowest risk of cyclone damages to ensure the long-term benefit of interventions.

These recommendations are a first attempt at developing a decision framework for the spatial prioritisation of reef restoration based on the best available models. They can be refined with expert knowledge and *in situ* assessments, and the connectivity metrics can be easily recalculated as the list of suitable habitats evolves. While different objectives may lead to different prioritisations, there are some general rules that should be followed during the process of selection (Edwards and Gomez 2007). A decision-tree was developed to assist this process (Figure 36), with a particular emphasis on the spatial data that can be help navigating the different steps of the selection.

1 - see https://www.slim-ocean.be/index.php/great-barrier-reef/

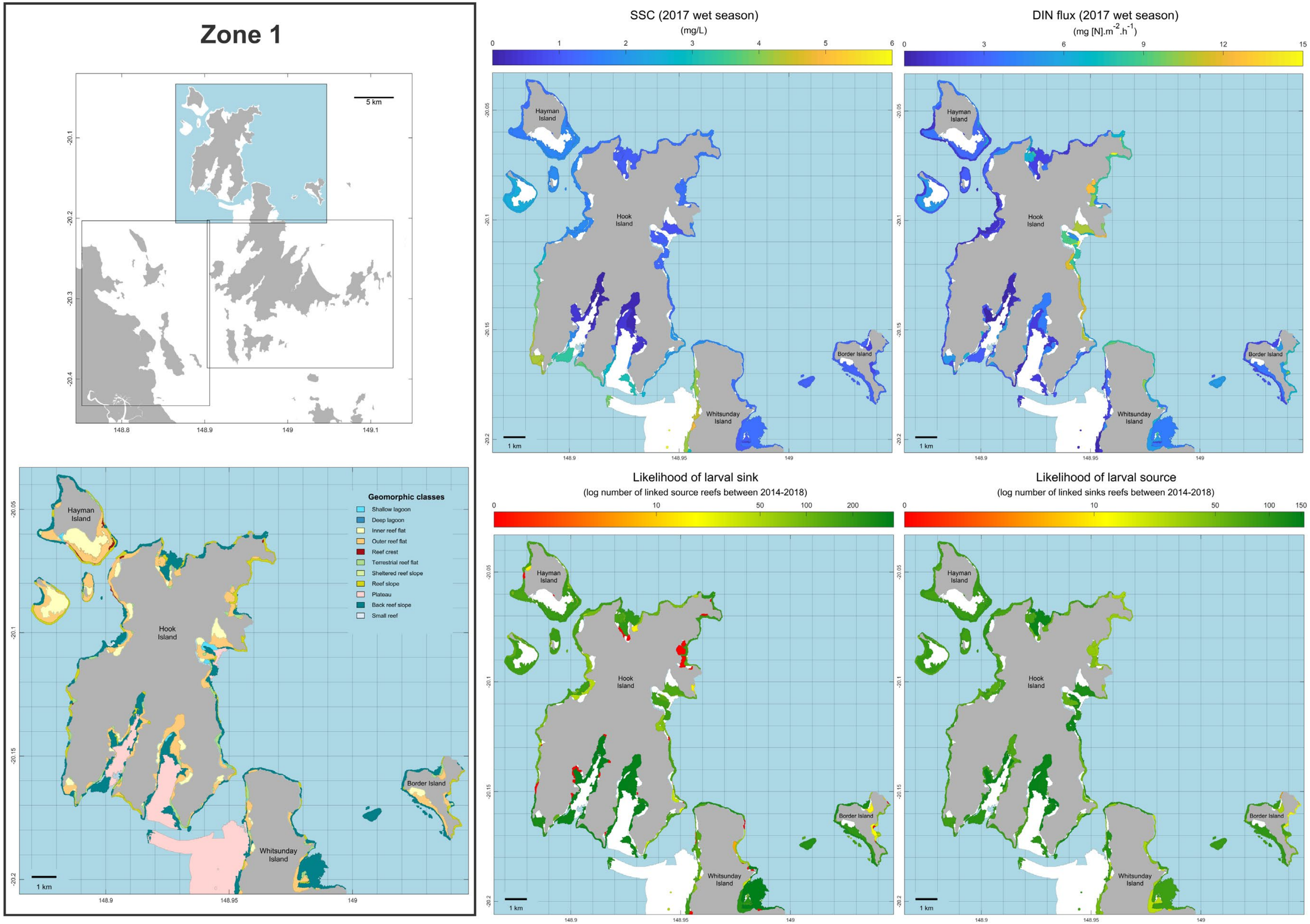


Figure 33: Crossing of spatial information to support reef prioritisation in the Hook Island region (RECOM zone 1). Each geomorphic reef polygon was coloured following the predicted value of suspended sediment and nutrient flux (proxy of macroalgal productivity) of the 2017 wet season, and the likelihoods of larval sink and source determined at 200 m resolution.

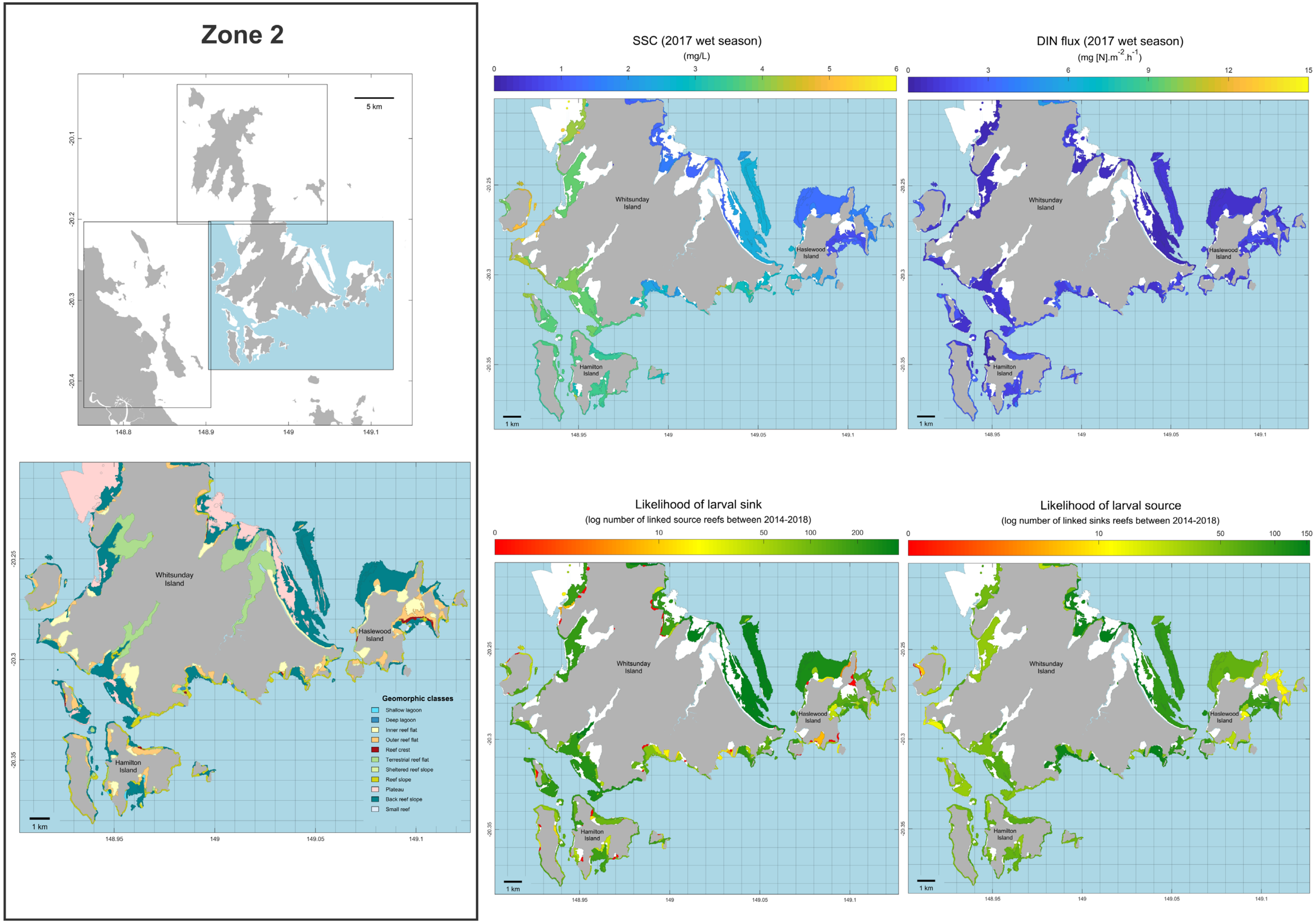


Figure 34: Crossing of spatial information to support reef prioritisation in the Whitsunday Island region (RECOM zone 2). See legend in Figure 33.

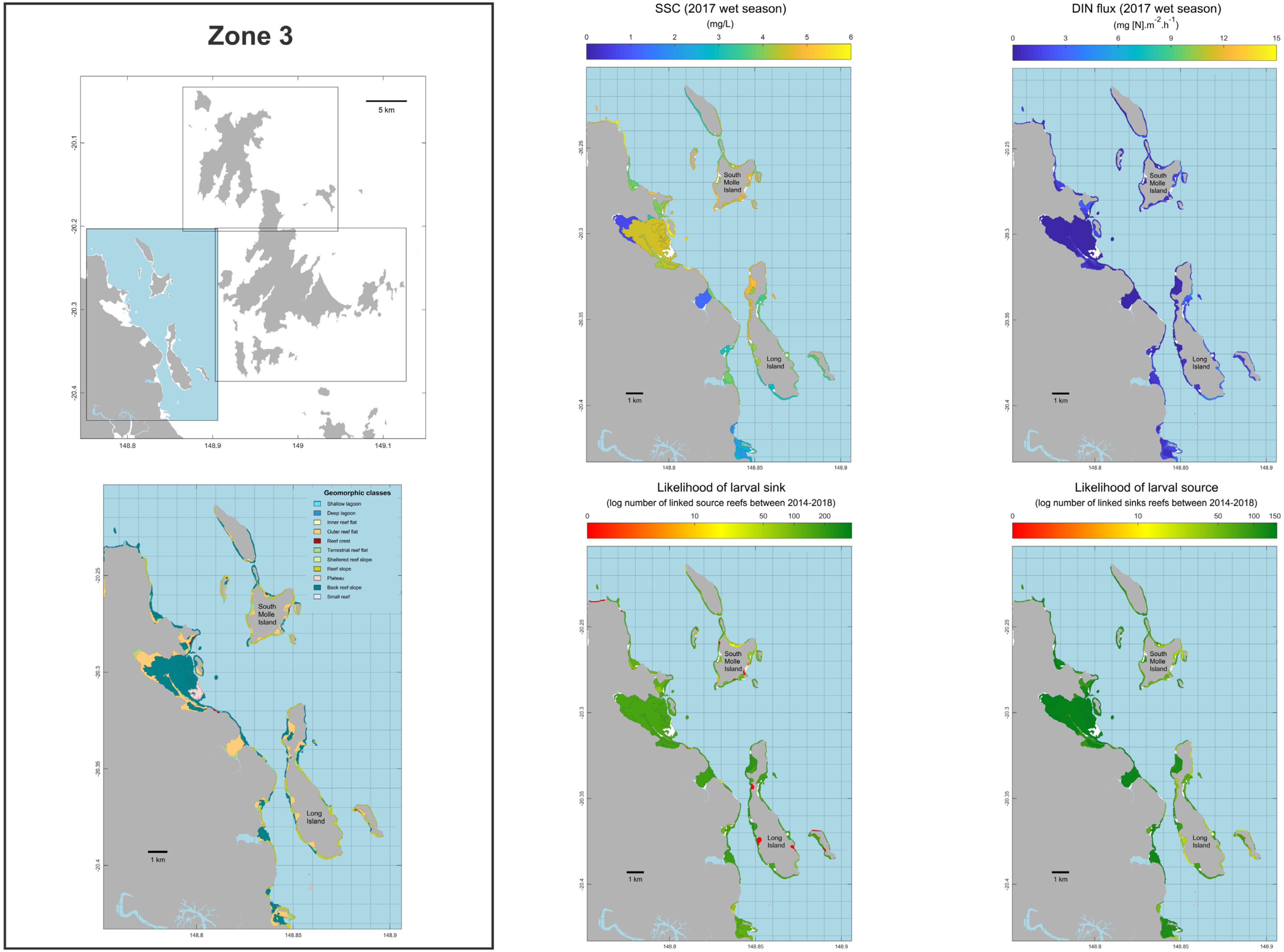


Figure 35: Crossing of spatial information to support reef prioritisation in the Molle Groupd (RECOM zone 3). See legend in Figure 33.

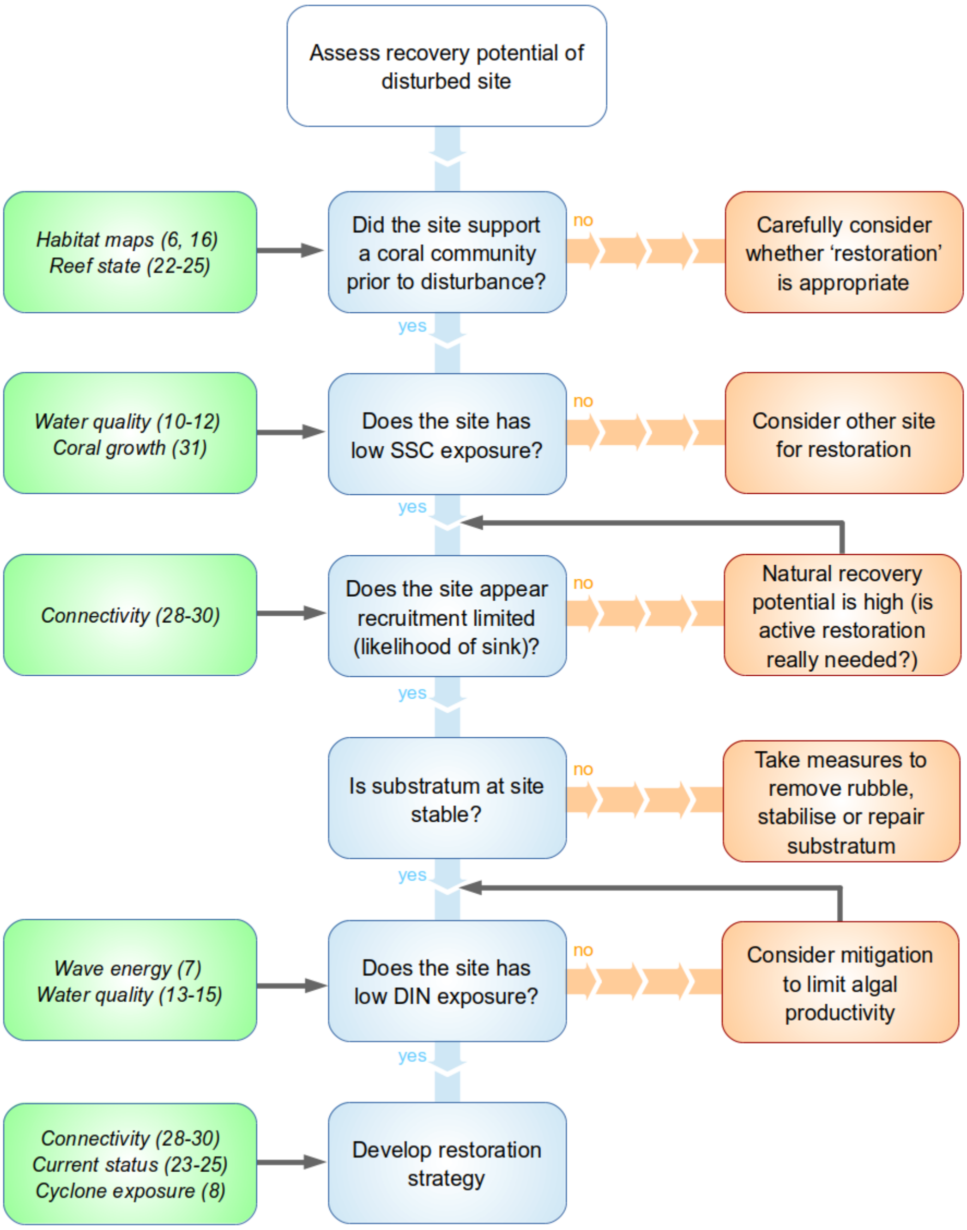


Figure 36: Decision-tree to assist the decision of undertaking reef restoration on a site (adapted from Edwards and Gomez 2007). Maps produced in this report are referenced (figure number) in support of the different decision steps.

# 6 References

Baird ME, Wild-Allen KA, Parslow J, Mongin M, Robson B, Skerratt J, Rizwi F, Soja-Woźniak M, Jones E, Herzfeld M, others (2020) CSIRO Environmental Modelling Suite (EMS): scientific description of the optical and biogeochemical models (vB3p0). Geosci Model Dev 13:4503–4553

Beeden R, Turner M, Dryden J, Merida F, Goudkamp K, Malone C, Marshall P, Birtles A, Maynard J (2014) Rapid survey protocol that provides dynamic information on reef condition to managers of the Great Barrier Reef. Environ Monit Assess 186:8527–8540

Bozec Y-M, Hock K, Mason RA, Baird ME, Castro-Sanguino C, Condie SA, Puotinen M, Thompson A, Mumby PJ (2020) Cumulative impacts across Australia's Great Barrier Reef: A mechanistic evaluation. bioRxiv

Bozec Y-M, Mumby PJ (2019) Detailed description of ReefMod-GBR and simulation results. Technical Appendix B2. In: Anthony K.R.N., Condie S.A., Bozec Y.-M., Harrison D., Gibbs M., Baird M.E., Mumby P.J., Mead D. (eds) Reef Restoration and Adaptation Program: Modelling Methods and Findings. A report provided to the Australian Government by the Reef Restoration and Adaptation Program. Australian Institute of Marine Science, Townsville, Australia, pp 72–113

Callaghan DP, Leon JX, Saunders MI (2015) Wave modelling as a proxy for seagrass ecological modelling: Comparing fetch and process-based predictions for a bay and reef lagoon. Estuar Coast Shelf Sci 153:108–120

Callaghan DP, Mumby PJ, Mason MS (2020) Near-reef and nearshore tropical cyclone wave climate in the Great Barrier Reef with and without reef structure. Coast Eng 157:103652

Ceccarelli DM, Evans RD, Logan M, Mantel P, Puotinen M, Petus C, Russ GR, Williamson DH (2020) Long-term dynamics and drivers of coral and macroalgal cover on inshore reefs of the Great Barrier Reef Marine Park. Ecol Appl 30:e02008

Condie S, Condie R (2016) Retention of plankton within ocean eddies. Glob Ecol Biogeogr 25:1264–1277

Condie SA, Hepburn M, Mansbridge J (2012) Modelling and visualisation of connectivity on the Great Barrier Reef. 9–13

Connolly SR, Baird AH (2010) Estimating dispersal potential for marine larvae: dynamic models applied to scleractinian corals. Ecology 91:3572–3583

DeVantier L, De'Ath G, Done T, Turak E (1998) Ecological assessment of a complex natural system: a case study from the Great Barrier Reef. Ecol Appl 8:480–496

Edwards AJ, Gomez ED (2007) Reef restoration concepts and guidelines: making sensible management choices in the face of uncertainty.

Fabricius K, De'ath G, McCook L, Turak E, Williams DM (2005) Changes in algal, coral and fish assemblages along water quality gradients on the inshore Great Barrier Reef. Mar Pollut Bull 51:384–398

Fabricius KE (2005) Effects of terrestrial runoff on the ecology of corals and coral reefs: review and synthesis. Mar Pollut Bull 50:125–146

Great Barrier Reef Marine Park Authority (2007) Great Barrier Reef (GBR) Features (Reef boundaries, QLD Mainland, Islands, Cays, Rocks and Dry Reefs).

Gruber R, Waterhouse J, Logan M, Petus C, Howley C, Lewis S, Tracey D, Langlois L, Tonin H, Skuza M, others (2020) Marine Monitoring Program: Annual report for inshore water quality monitoring 2018-19.

Hock K, Doropoulos C, Gorton R, Condie SA, Mumby PJ (2019) Split spawning increases robustness of coral larval supply and inter-reef connectivity. Nat Commun 10:3463

Hughes TP, Kerry JT, Baird AH, Connolly SR, Dietzel A, Eakin CM, Heron SF, Hoey AS, Hoogenboom MO, Liu G, others (2018) Global warming transforms coral reef assemblages. Nature 556:492

Humanes A, Fink A, Willis BL, Fabricius KE, de Beer D, Negri AP (2017) Effects of suspended sediments and nutrient enrichment on juvenile corals. Mar Pollut Bull 125:166–175

Jones R, Ricardo G, Negri A (2015) Effects of sediments on the reproductive cycle of corals. Mar Pollut Bull 100:13–33

Liu G, Skirving WJ, Geiger EF, De La Cour JL, Marsh BL, Heron SF, Tirak KV, Strong AE, Eakin CM (2017) NOAA Coral Reef Watch's 5km satellite coral bleaching heat stress monitoring product suite version 3 and four-month outlook version 4. Reef Encount 32:39–45

Lyons M B, Roelfsema CM, Kennedy EV, Kovacs EM, Borrego-Acevedo R, Markey K, Roe M, Yuwono D M, Harris D L, Phinn S R, Asner GP, Li J, Knapp D E, Fabina N S, Larsen K, Traganos D, Murray N J (2020) Mapping the world's coral reefs using a global multiscale earth observation framework. Remote Sens Ecol Conserv 6:557–568

Roelfsema C, Kovacs E, Ortiz JC, Wolff NH, Callaghan D, Wettle M, Ronan M, Hamylton SM, Mumby PJ, Phinn S (2018) Coral reef habitat mapping: A combination of object-based image analysis and ecological modelling. Remote Sens Environ 208:27–41

Roelfsema CM, Kovacs EM, Ortiz JC, Callaghan DP, Hock K, Mongin M, Johansen K, Mumby PJ, Wettle M, Ronan M (2020) Habitat maps to enhance monitoring and management of the Great Barrier Reef. Coral Reefs 39:1039–1054

Steven AD, Baird ME, Brinkman R, Car NJ, Cox SJ, Herzfeld M, Hodge J, Jones E, King E, Margvelashvili N (2019) eReefs: An operational information system for managing the Great Barrier Reef. J Oper Oceanogr 12:S12–S28

Thomas CJ, Lambrechts J, Wolanski E, Traag VA, Blondel VD, Deleersnijder E, Hanert E (2014) Numerical modelling and graph theory tools to study ecological connectivity in the Great Barrier Reef. Ecol Model 272:160–174

Thompson A, Costello P, Davidson J, Logan M, Coleman G, Gunn K (2018) Marine Monitoring Program: Annual report for inshore coral reef monitoring 2016-2017.

Thompson A, Schroeder T, Brando VE, Schaffelke B (2014) Coral community responses to declining water quality: Whitsunday Islands, Great Barrier Reef, Australia. Coral Reefs 33:923–938

Thompson AA, Dolman AM (2010) Coral bleaching: one disturbance too many for near-shore reefs of the Great Barrier Reef. Coral Reefs 29:637–648

Williamson D, Ceccarelli D, Jones G, Russ G (2019) Assessing the ecological effects of management zoning on inshore reefs of the Great Barrier Reef Marine Park: Reef 2050 Integrated Monitoring and Reporting Program milestone report 2.

van Woesik R (1993) Coral assemblages on continental islands in the southern Great Barrier Reef, Australia. 40 pp

van Woesik R, De Vantier L, Glazebrook J (1995) Effects of Cyclone Joy on nearshore coral communities of the Great Barrier Reef. Mar Ecol Prog Ser 128:261–270

van Woesik R, Done TJ (1997) Coral communities and reef growth in the southern Great Barrier Reef. Coral Reefs 16:103–115

van Woesik R, Tomascik T, Blake S (1999) Coral assemblages and physico-chemical characteristics of the Whitsunday Islands: evidence of recent community changes. Mar Freshw Res 50:427–440

Wooldridge SA (2009) Water quality and coral bleaching thresholds: Formalising the linkage for the inshore reefs of the Great Barrier Reef, Australia. Mar Pollut Bull 58:745–751