MARINE MONITORING PROGRAM

Annual Report for
INSHORE SEAGRASS MONITORING
2018-19

JAMES COOK UNIVERSITY AUSTRALIA

TropWATER

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Front cover image: Seagrass on the reef flat at Green Island © Dieter Tracey 2013.

The Great Barrier Reef Marine Park Authority acknowledges the continuing Sea Country management and custodianship of the Great Barrier Reef by Aboriginal and Torres Strait Island Traditional Owners whose rich cultures, heritage values, enduring connections and shared efforts protect the Reef for future generations.

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Contents

List of Figures

List of Tables

Acronyms, abbreviations and units

Authority	Great Barrier Reef Marine Park Authority
CV	coefficient of variation
DES	Department of Environment and Science, Queensland
JCU	James Cook University
km	kilometre
m	metre
MMP	Marine Monitoring Program
MTSRF	Marine and Tropical Sciences Research Facility
NRM	Natural Resource Management
Paddock to Reef program	Paddock to Reef Integrated Monitoring, Modelling and Reporting Program
QPWS	Queensland Park and Wildlife Service
Reef	Great Barrier Reef
Reef 2050 WQIP	*Reef 2050 Water Quality Improvement Plan*
Reef 2050 Plan	*Reef 2050 Long-Term Sustainability Plan*
RIMReP	Reef 2050 Integrated Monitoring and Reporting Program
SE	Standard Error
TropWATER	Centre for Tropical Water & Aquatic Ecosystem Research

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We thank the Range Control Officer (Department of Defence) for providing support and access to the Shoalwater Bay Training Area to conduct biannual monitoring; Great Adventures (part of the Quicksilver Group) for providing discounted transfers to Green Island; and Nancy Lowe of Dunk Island Water Taxi for assistance with the maintenance of light loggers at Mission Beach.

River discharge data provided by the State of Queensland (Department of Natural Resources and Mines) 2019. The conceptual diagram symbols are courtesy of the Integration and Application Network (ian.umces.edu/symbols/), University of Maryland Center for Environmental Science. Climate data courtesy of the Australian Bureau of Meteorology, and tide data courtesy Maritime Safety Queensland, Department of Transport and Main Roads.

Executive summary

The Great Barrier Reef Marine Park Authority's Marine Monitoring Program was established in 2005 to monitor the inshore health of the Great Barrier Reef (the Reef). This document reports on the long-term health of inshore seagrass meadows and presents the findings in the context of the pressures faced by the ecosystem.

Inshore seagrass meadows across the Reef continued to decline in overall condition in 2018–19, further overturning some of the recovery experienced during a period from mid 2010 to mid 2017. The condition grade for inshore seagrass meadows has remained **poor**. All regions this year have an overall seagrass condition grade of poor, including a downgrade from moderate in the Burdekin region and an improvement from very poor in the Burnett–Mary region.

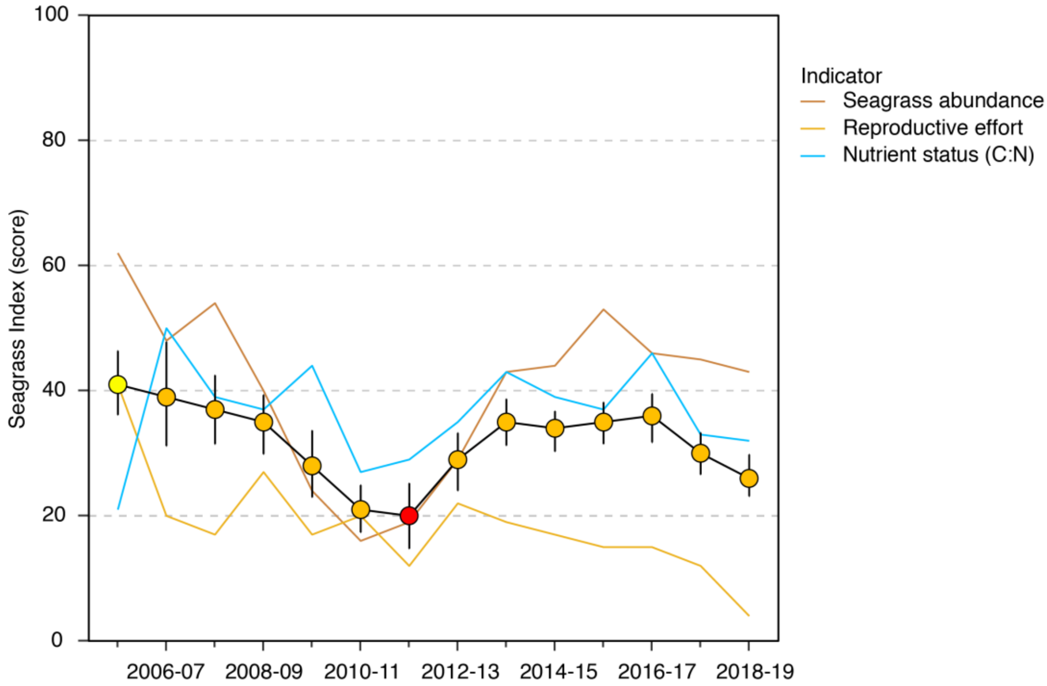


Figure 1. *Reef-wide seagrass condition index (±SE) with contributing indicator scores over the life of the MMP. The index is derived from the aggregate of metric scores for indicators of seagrass community health. Index scores scaled from 0–100 and graded: ● = very good (81-100), ● = good (61 - 80), ● = moderate (41 - 60), ● = poor (21 - 40), ● = very poor (0 - 20). NB: Scores are unitless.*

Seagrass condition is a composite of three indicators, which are measured at 30 locations (with duplicate sites nested within most locations) across the Reef. Combining these scores for all monitored seagrass meadows in a natural resource management region gives a rating for seagrass condition. Indicators are:

- seagrass abundance (per cent cover)
- reproductive effort
- leaf tissue nutrients.

Additional indicators of seagrass condition and resilience are assessed and used to assist with the interpretation of condition including:

- seagrass species composition
- relative meadow extent

- density of seeds in the seed bank.

Environmental pressures are also recorded including:

- within-canopy water temperature
- within-canopy benthic light
- sediment composition
- macroalgae and epiphyte abundance.

The majority of inshore seagrass monitoring sites are located in predominately lower littoral areas (only exposed to air at the lowest of low tides), hereafter referred to as intertidal, although eight locations also included shallow subtidal meadows. Each of the major seagrass habitat types (estuarine, coastal, reef, subtidal) were assessed in each region where possible.

Since 2010–11, seagrass abundance had been increasing at most locations, but declined in condition in the past two reporting years, including 2018–19. The decline over the past two years is a result of both recent and past events, including cyclone Debbie that crossed the coast near Airlie Beach in the Mackay–Whitsunday region in 2017 and marine heatwaves that affected all inshore seagrass meadows in 2014–15 and 2015–16. In 2018–19, heavy rainfall and above-average discharge from the Burdekin River and many of the small rivers in the Burdekin region, affected five of the six sentinel monitoring sites in this area. As a result, there was a large decline in seagrass abundance and extent in the Burdekin region, however abundance remained similar to the previous year in other regions, and overall the abundance score remained unchanged from 2018–19 for the Reef as a whole.

Reproductive effort is a measure of resilience/recovery capacity where the production of new fruits or seeds by a meadow in each season provides the basis of new propagules for recruitment in the following year. The likelihood that the meadows are able to recover is informed by the measure of reproductive effort. In addition, sexual reproduction is likely to enhance meadow scale genetic diversity, therefore increasing ‘resistance’ of the meadow to disturbance. Reproductive effort declined at over a third of sites, remaining very poor at two thirds of sites, and very poor in four regions, except the Wet Tropics and Burnett–Mary. This very poor reproductive effort was the main cause of the continued downturn in the overall condition index in 2018–19. Of particular concern is that reproductive effort remains well below historical levels and continues to decline. Furthermore, most of the reef habitat seagrass monitoring sites have almost no seed banks making them highly vulnerable to future disturbances.

Seagrass tissue nutrients indicate the availability of nitrogen relative to growth demand (i.e. carbon fixation). The leaf tissue nutrient indicator declined slightly in 2018–19. Just over 50 per cent of sites displayed symptoms of nutrient enrichment, with 18 per cent inferring elevated nitrogen (predominately coastal habitats). While 14 per cent of sites with higher than average and increasing epiphyte abundances suggest some level of increased nutrient availability.

Indicators of reduced resilience include:

- decreasing abundance at nearly a third of the meadows monitored, predominately in the Burdekin and northern Wet Tropics regions
- lower than average composition of foundational species at nearly a quarter of all sites
- declining extent at nearly a quarter of meadows, with reef habitats in the Burdekin region and estuarine habitats from Mackay–Whitsunday south showing the greatest decline

- below long-term average reproductive effort at 84 per cent of sites, with declining and very poor reproductive effort at 64 per cent of sites (reproductive structures absent from nearly half)
- declining seed banks at 38 per cent of sites, with seed banks now absent from nearly half of the sites.

Benthic light availability was lower than average at nearly half the meadows monitored (particularly across the Fitzroy, Cape York and Wet Tropics regions). Six of the 27 locations assessed had light availability below levels supportive of long-term growth.

The findings suggest seagrass in some regions, such as the Burdekin and Cape York, may be more vulnerable to adverse or severe disturbances in the future. Of greatest concern is the Burdekin region, which in 2018–19 had the greatest percentage of sites/meadows decreasing in abundance and extent, with below-average and decreasing reproductive effort, and diminishing seed banks as a consequence.

The overall decline in seagrass condition is of concern, however, declines in indicators were not consistent and there were some ‘bright spots’ of improvement. Examples include:

- increasing or stable abundances at over 60 per cent of sites, with greatest improvements in the Burnett–Mary, Fitzroy and Mackay–Whitsunday regions
- nearly a quarter of meadows continuing to expand in area or become less fragmented, while approximately a fifth remained at or near their maximum extent
- declining epiphyte loads, with below average cover at 45 per cent of sites
- increasing reproductive effort at 43 per cent of sites, particularly in the northern Wet Tropics and southern (Fitzroy and Burnett–Mary) regions.

These improvements demonstrate maintenance of or improvement in seagrass resilience in some regions, which are a consequence of variable climatic and environmental pressures. For example, benthic light availability improved in the Burnett–Mary region paving the way for some recovery.

The Reef is characterised by ongoing cumulative impacts and dynamic seagrass meadows. Intensifying pressures are slowing recovery. Water quality improvements that can be gained by land management initiatives (such as the Paddock to Reef program), will help to reduce the pressures and improve the condition and recovery capacity of seagrass meadows.

Case studies

Annual case studies are produced as part of the program every year. The case study assesses one of the metrics — leaf tissue nutrient C:N — in relation to water quality, including predicted nutrient and sediment loads reaching each of the seagrass sites and *in situ* water quality at some of the reef sites. The water quality data is available from Gruber et al. (2020). The case study also discusses the relevance of findings to the seagrass condition index and the Reef Report Card.

1 Introduction

Approximately 3,464 km^2 of inshore seagrass meadows have been mapped in Great Barrier Reef World Heritage Area (the World Heritage Area) in waters shallower than 15 m (McKenzie *et al.* 2014c; Saunders *et al.* 2015; Carter *et al.* 2016; McKenzie *et al.* 2016; C. Howley, Unpublished data) (Figure 2). The remaining modelled extent (90% or 32,335 km^2) of seagrass in the World Heritage Area is located in the deeper waters (>15 m) of the lagoon (Coles *et al.* 2009; Carter *et al.* 2016), however, these meadows are relatively sparse, structurally smaller, highly dynamic, composed of colonising species, and not as productive as inshore seagrass meadows for fisheries resources (McKenzie *et al.* 2010b; Derbyshire *et al.* 1995). Overall, the total estimated area of seagrass (34,841 km^2) within the World Heritage Area represents more than 50% of the total recorded area of seagrass in Australia (Green and Short 2003) and between 6% and 12% globally (Duarte *et al.* 2005), making the Reef's seagrass resources globally significant.

Tropical seagrass ecosystems of the Reef are a complex mosaic of different habitat types comprised of multiple seagrass species (Carruthers *et al.* 2002). There are 15 seagrass species in the Reef (Waycott *et al.* 2007) and a high diversity of seagrass habitat types. Seagrass colonise sandy or muddy areas along beaches, within reef platforms and lagoons, and can extend down to 60 m or more below Mean Sea Level (MSL).

Seagrasses in the Reef can be separated into four major habitat types: estuary/inlet, coastal, reef and deepwater (Carruthers *et al.* 2002). All but the outer reef habitats are significantly influenced by seasonal and episodic pulses of sediment-laden, nutrient-rich river flows, resulting from high volume summer rainfall. Cyclones, severe storms, wind and waves as well as macro grazers (e.g. fish, dugongs and turtles) influence all habitats in this region to varying degrees. The result is a series of dynamic, spatially and temporally variable seagrass meadows.

The seagrass ecosystems of the Reef, on a global scale, would be for the most part categorised as being dominated by disturbance-favouring colonising and opportunistic species (e.g. *Halophila* and *Halodule*), which typically have low standing biomass and high turnover rates (Carruthers *et al.* 2002, Waycott *et al.* 2007). In more sheltered areas, including reef top or inshore bays, more stable and persistent species are found, although these are still relatively responsive to disturbances (Carruthers *et al.* 2002; Waycott *et al.* 2007; Collier and Waycott 2009).

1.1 Seagrass monitoring in the Marine Monitoring Program

The vision for Great Barrier Reef Marine Park Authority (the Authority) is to have a healthy Great Barrier Reef Marine Park for future generations (Great Barrier Reef Marine Park Authority 2019). The decline of marine water quality associated with landbased run-off from the adjacent catchments is a major cause of the current poor state of many of the coastal and marine ecosystems of the Great Barrier Reef (The State of Queensland, 2017).

In response to concerns about the impact of land-based run-off on water quality, coral and seagrass ecosystems, the Reef 2050 Water Quality Improvement Plan (Reef 2050 WQIP) (Australian Government and Queensland Government 2018b) was recently updated by the Australian and Queensland governments, and integrated as a major component of Reef 2050 Long-Term Sustainability Plan (Reef 2050 Plan) (Australian Government and Queensland Government 2018a), which provides a framework for integrated management of the World Heritage Area.

A key deliverable of the Reef 2050 WQIP is the Paddock to Reef Integrated Monitoring, Modelling and Reporting Program (Paddock to Reef program), which is used to evaluate the efficiency and effectiveness of Reef 2050 WQIP implementation, and report on progress towards goals and targets (Australian Government and Queensland Government 2018b). The Marine Monitoring Program (MMP) forms an integral part of the Paddock to Reef program. The MMP has three components: inshore water quality, coral and seagrass.

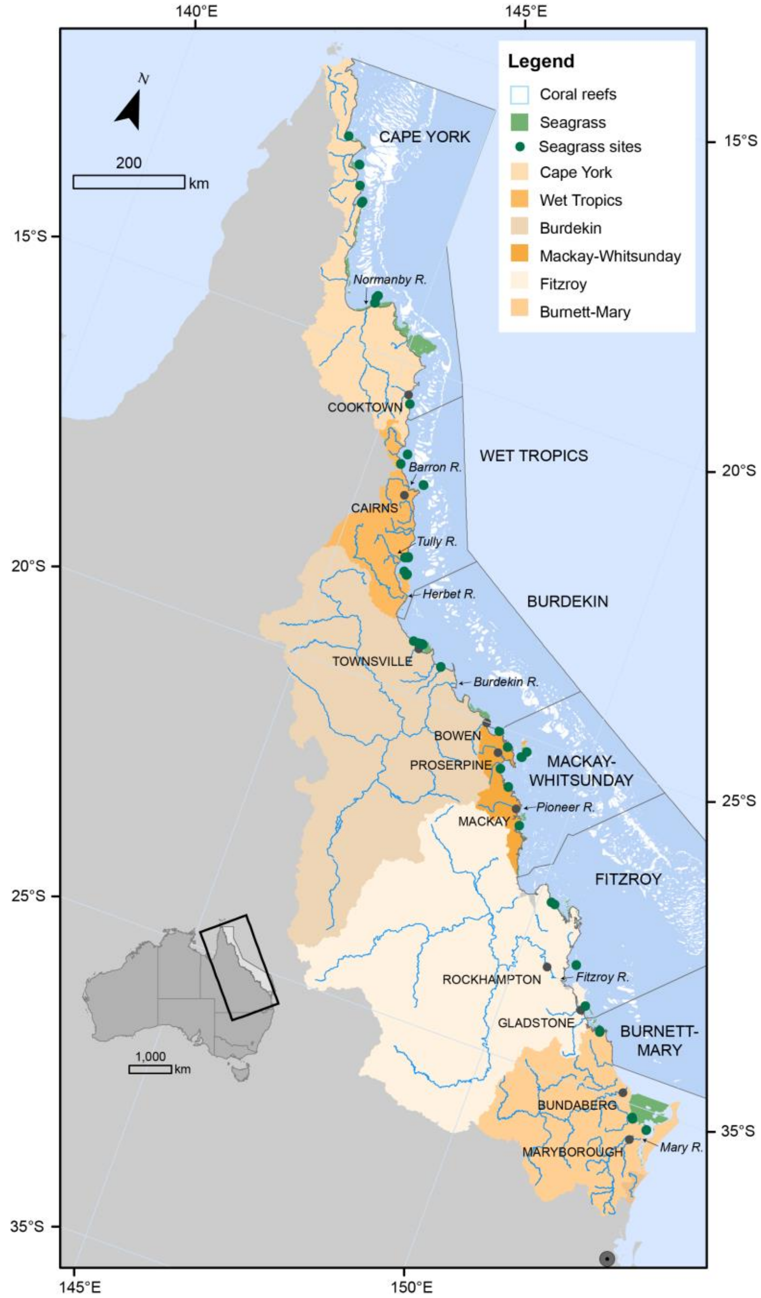


Figure 2. *Major marine ecosystems (coral reefs and surveyed seagrass meadows) in the World Heritage Area and Natural Resource Management regions (including marine) (delineated by dark grey lines) and major rivers.*

The overarching objective of the inshore seagrass monitoring program is to quantify the extent, frequency and intensity of acute and chronic impacts on the condition and trend of seagrass meadows and their subsequent recovery.

The inshore water quality monitoring component of the Marine Monitoring Program has been delivered by James Cook University (JCU) since 2005. The seagrass sub-program is also supported by contributions from the Seagrass-Watch program (Wet Tropics, Burdekin, Mackay–Whitsunday and Burnett–Mary) and Queensland Parks and Wildlife Service (QPWS).

Further information on the program objectives, and details on each sub-program are available on-line (GBRMPA 2019; http://bit.ly/2mbB8bE).

1.2 Conceptual basis for indicator selection

As seagrasses are well recognised as indicators of integrated environmental pressures, monitoring their condition and trend can provide insight into the condition of the surrounding environment (e.g. Dennison *et al.* 1997). There are a number of measures of seagrass condition and resilience that can be used to assess how they respond to environmental pressures, and these measures are referred to here as indicators. A matrix of indicators that respond on different temporal scales (Figure 3) are used including:

- plant-scale changes
- meadow-scale changes
- state change.

These indicators also respond at different temporal scales, with sub-lethal indicators responding in seconds to months, while the meadow-scale changes usually take many months to be detectable. A robust monitoring program benefits from having a suite of indicators that can indicate sub-lethal stress that forewarns of imminent loss, as well as indicators of meadow-scale changes, which are necessary for interpreting broad ecological changes. Indicators included in the MMP span this range of scales, in particular for indicators that respond from weeks (tissue nutrients, isotopes), through to months (abundance and reproduction), and even years (composition and meadow extent). Furthermore, indicators are conceptually linked to each other and to environmental drivers of concern, in particular, water quality (p 34, in Kuhnert *et al.* 2014).

Measures of Environmental stressors

Climate and environment stressors are aspects of the environment, either physio-chemical or biological that affect seagrass meadow condition. Some environmental stressors change rapidly (minutes/days/weeks/months) but can also undergo chronic shifts (years) (Figure 3). Stressors include:

- climate (e.g. cyclones, seasonal temperature)
- local and short-term weather (e.g. wind and tides)
- water quality (e.g. river discharge, plume exposure, nutrient concentrations, suspended sediments, herbicides)
- biological (e.g. epiphytes which can grow on seagrass and macroalgae)
- substrate (e.g. grain size composition)
- seagrass environmental integrators (e.g. tissue nutrients).

Indicators which respond more quickly (e.g. light) provide important early-warning of potentially more advanced ecological changes (as described below). However, a measured change in a fast-responding environmental indicator is not enough in isolation to predict whether there will be further ecological impacts, because the change could be short-term. These indicators provide critical supporting information to support interpretation of slower responding seagrass condition and resilience indicators. Epiphytes and macroalgae are an environmental indicator because they can compete with and/or block light reaching seagrass leaves, therefore compounding environmental stress.

These environmental indicators are interpreted according to the following general principles:

- Cyclones cause physical disturbance from elevated swell and waves resulting in abrasion, meadow fragmentation and loss of seagrass plants (McKenzie *et al.* 2012). Seagrass loss also results from smothering by sediments and light limitation due to increased turbidity from suspended sediments. The heavy rainfall associated with cyclones can result in flooding which exacerbates light limitation and can transport pollutants (nutrients and pesticides), resulting in further seagrass loss (Preen *et al.* 1995).

- Benthic light levels below 10 mol m^{-2} d^{-1} are unlikely to support long-term growth of seagrass, and periods below 6 mol m^{-2} d^{-1} for more than four weeks can cause loss (Collier *et al.* 2016b). However, it is unclear how these relate to intertidal habitats because very high light exposure during low tide can affect the seagrass. Therefore, it may be more informative to look at change relative to the sites.
- Water temperature can impact seagrasses through chronic effects in which elevated respiration at high temperatures can cause carbon loss and reduce growth (Collier et al 2017), while acute stress results in inhibition of photosynthesis and leaf death (Campbell *et al.* 2006; Collier and Waycott 2014)
- Daytime tidal exposure can provide critical windows of light for positive net photosynthesis for seagrass in chronically turbid waters (Rasheed and Unsworth 2011). However, during tidal exposure, plants are susceptible to extreme irradiance doses, desiccation, thermal stress and potentially high UV-A and UV-B leading to physiological damage, resulting in short-term declines in density and spatial coverage (Unsworth *et al.* 2012b).
- Sediment grain size affects seagrass growth, germination, survival, and distribution (McKenzie 2007). Coarse, sand dominated sediments limit plant growth due to increased mobility and lower nutrients. However, as finer-textured sediments increase (dominated by mud (grain size <63µm)), porewater exchange with the overlaying water column decreases resulting in increased nutrient concentrations and phytotoxins such as sulphide, which can ultimately lead to seagrass loss (Koch 2001).

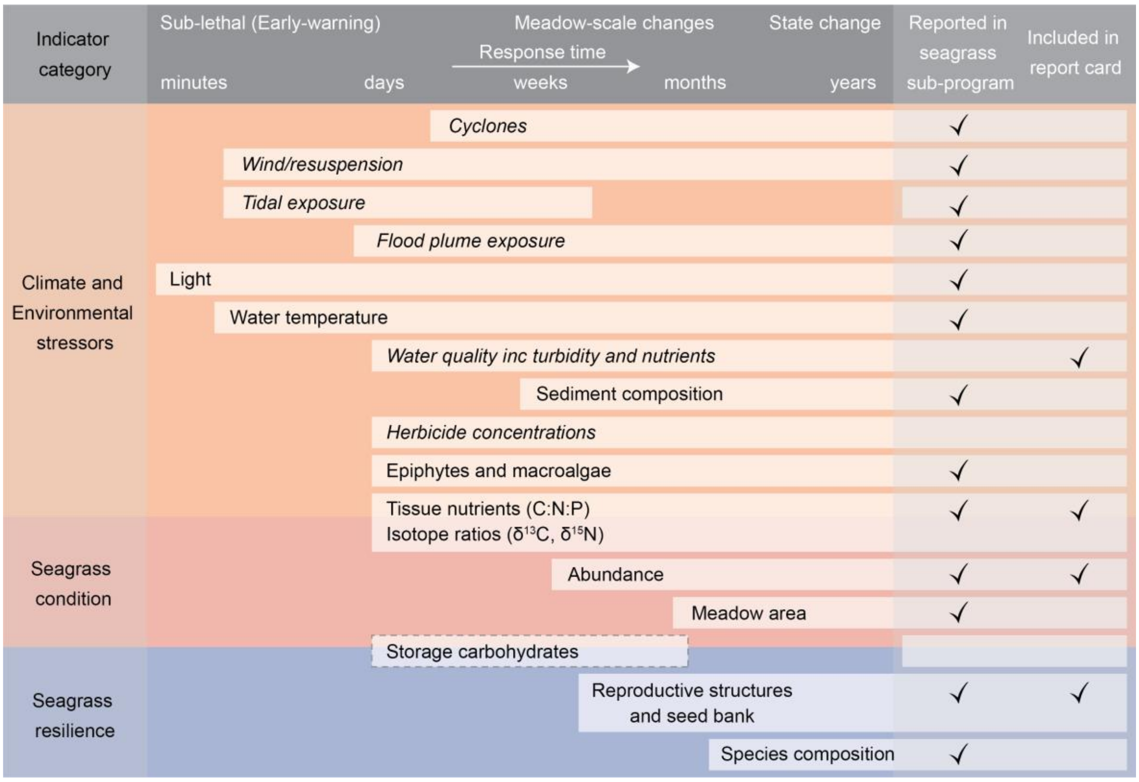


Figure 3. *Climate, environmental, seagrass condition and seagrass resilience indicators reported as part of inshore seagrass monitoring. Regular text are indicators measured in the inshore seagrass program, white box with dashed line are indicators in development, and italicised text are indicators collected in other programs or by other institutions (see Table 2 for details on data source). All indicators are shown against their response time.*

Measures of seagrass condition

Condition indicators such as meadow abundance and extent indicate the state of the plants/population and reflect the cumulative effects of past environmental conditions (Figure 3). Abundance can respond to change on time-scales ranging from weeks to months (depending on species) in the Reef, while meadow area tends to adjust over longer time-scales (months to years). Seagrass area and abundance are integrators of past conditions, and are vital indicators of meadow condition; however, these indicators can also be affected by external factors such as grazing by dugongs and turtles. Therefore, they are not suitable as stand-alone indicators of environmental change and indicators that can be linked more directly to specific pressures are needed. These condition indicators also do not demonstrate capacity to resist or recover from additional impacts (Unsworth *et al.* 2015).

Changing ratios of seagrass tissue nutrients provide an indication of seagrass condition and environmental conditions. Carbon to nitrogen (C:N) ratios have been found in a number of experiments and field surveys to be related to light levels, as leaves with an atomic C:N ratio of less than 20, may suggest reduced light availability when N is not in surplus (Abal *et al.* 1994; Grice *et al.* 1996; Cabaço and Santos 2007; Collier *et al.* 2009). Therefore, C:N ratio is reported within the seagrass component of the Marine Results report and the Reef 2050 WQIP annual report card, while other tissue nutrients are also presented as supporting information.

Measures of seagrass resilience

Ecological resilience is 'the capacity of an ecosystem to absorb repeated disturbances or shocks and adapt to change without fundamentally switching to an alternative stable state' (Holling 1973), and relates to the ability of a system to both resist and recover from disturbances (Unsworth *et al.* 2015) (Figure 4). Changes in resilience indicators show if the ecosystem is in transition (i.e. has already, or may undergo a state-change). Sexual reproduction (flowering, seed production and persistence of a seedbank) is an important feature of recovery (and therefore, of resilience) in seagrass meadows.

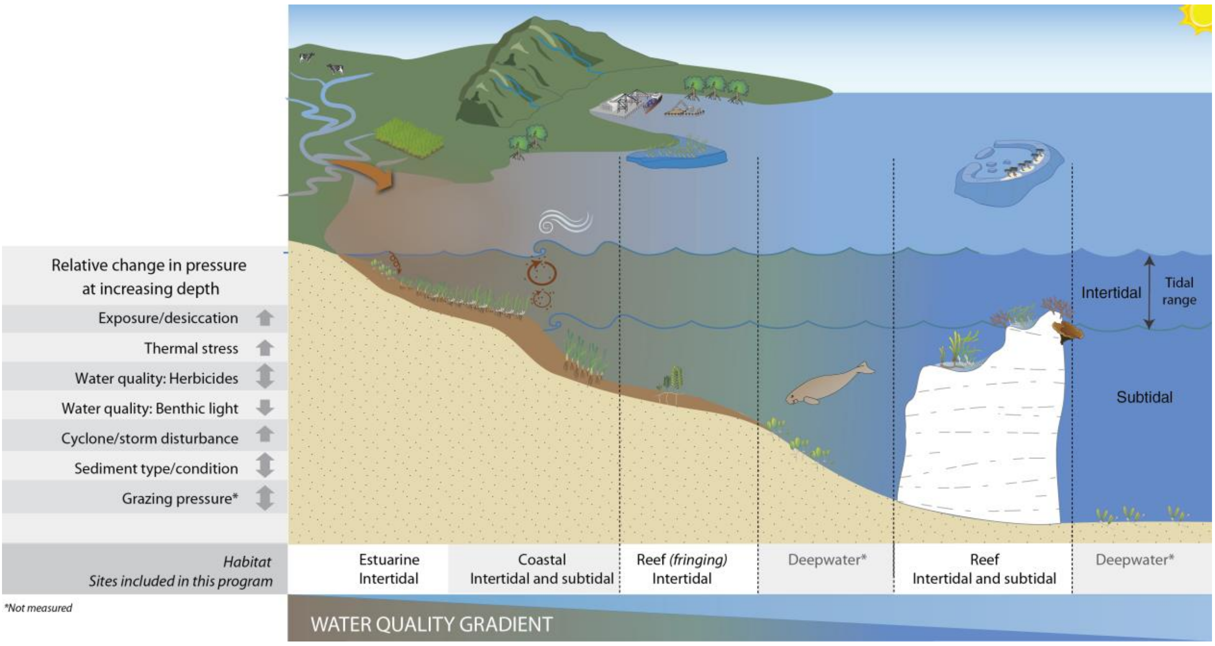


Figure 4. *General conceptual model of seagrass habitats in north east Australia and the water quality impacts affecting the habitat (adapted from Carruthers et al., 2002, and Collier et al. 2014)*

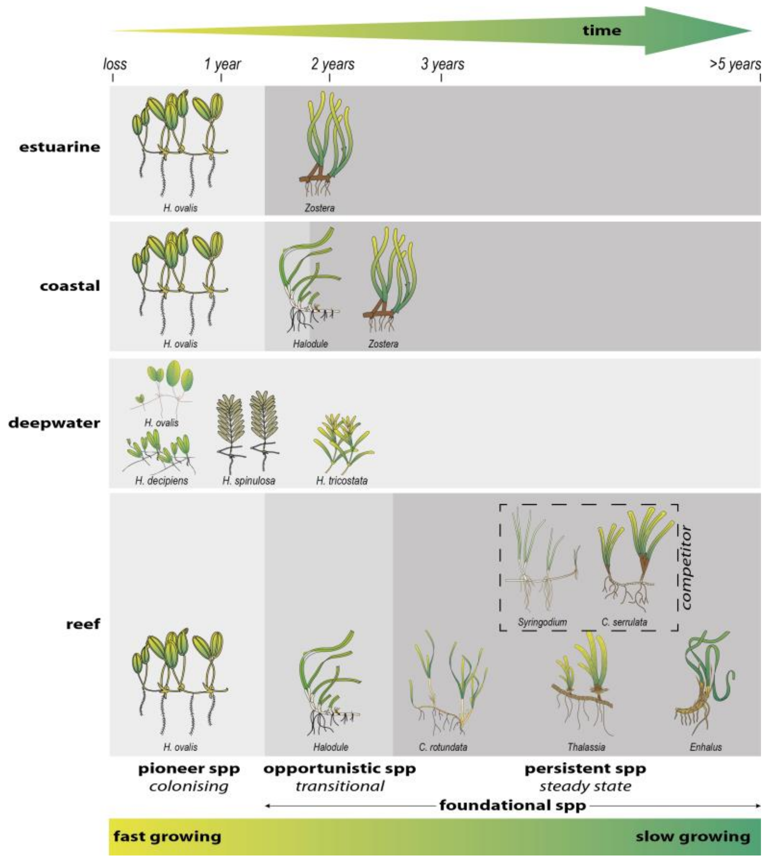


Figure 5. *Illustration of seagrass recovery after loss and the categories of successional species over time. Figure developed from observed recovery dynamics (Birch and Birch 1984; Preen et al. 1995; McKenzie and Campbell 2002; Campbell and McKenzie 2004; McKenzie et al. 2014a; Rasheed et al. 2014).*

Coastal seagrasses are prone to small scale disturbances that cause local losses (Collier and Waycott 2009), and therefore disturbance-specialist species (i.e. colonisers) tend to dominate throughout the Reef. Community structure (species composition) is also an important feature conferring resilience, as some species are more resistant to stress than others, and some species may rapidly recover and pave the way for meadow development (Figure 5).

1.3 Structure of the Report

This report presents data from the fourteenth period of monitoring inshore seagrass ecosystems of the Reef under the MMP (undertaken from June 2018 to May 2019; hereafter called 2018–19). The inshore seagrass monitoring sub-program of the MMP reports on:

- abundance and species composition of seagrass (including landscape mapping) in the late dry season of 2018 and the late wet season of 2019 at inshore intertidal and subtidal locations
- reproductive health of the seagrass species present at inshore intertidal and subtidal locations
- tissue nutrient concentrations (carbon, nitrogen and phosphorus) and epiphyte loads of foundation seagrass species (e.g. genus *Halodule, Zostera, Cymodocea*) at each inshore intertidal and subtidal location
- spatial and temporal patterns in light, turbidity and temperature at sites where autonomous loggers are deployed

- trends in seagrass condition
- seagrass community in relation to environment condition and trends
- seagrass report card metrics for use in the annual Reef Report Card produced by the Paddock to Reef program.

The next section presents a summary of the program's methods. Section 4 describes the condition and trend of seagrass in the context of environmental factors, referred to as drivers and pressures in Driver-Pressure-State-Impact-Response (DPSIR) framework.

In keeping with the overarching objective of the MMP, to *"Assess trends in ecosystem health and resilience indicators for the Great Barrier Reef in relation to water quality and its linkages to end-of-catchment loads",* key water quality results reported by Gruber *et al.* (2020) are replicated to support the interpretation of the inshore seagrass results.

2 Methods summary

In the following, an overview is given of the sample collection, preparation and analyses methods. Detailed documentation of the methods used in the MMP, including quality assurance and quality control procedures, is available in McKenzie *et al.* (2019).

2.1 Climate and environmental pressures

Climate and environmental pressures affect seagrass condition and resilience (Figure 4).

The pressures of greatest concern are:

- physical disturbance (cyclones and benthic sheer stress)
- water quality (turbidity/light and nutrients)
- water temperature
- low tide exposure
- sediment grain size/type.

The measures are either climate variables, that are generally not collected at a site-specific level, and within-canopy measures, that are recorded at each site. The data source and sampling frequency is summarised in Table 1.

2.1.1. Climate

Total daily rainfall, 3pm wind speed, and cyclone tracks were accessed from the Australian Bureau of Meteorology from meteorological stations which were proximal to monitoring locations (Table 1).

As the height of locally produced, short-period wind-waves can be the dominant factor controlling suspended sediment on inner-shelf of the Reef (Larcombe *et al.* 1995; Whinney 2007), the number of days wind speed exceeded 25 km hr^{-1} was used as a surrogate for elevated resuspension pressure on inshore seagrass meadows.

Moderate sea state with winds >25 km hr^{-1} can elevate turbidity by three orders of magnitude in the inshore coastal areas of the Reef (Orpin *et al.* 2004). To determine if the tidal exposure regime may be increasing stress on seagrass and hence drive decline, tidal height observations were accessed from Maritime Safety Queensland and duration of annual exposure (hours) was determined for each meadow (i.e. monitoring site), based on the meadows height relative to the lowest astronomical tide (Appendix 3, Table 19).

The presence of inshore seagrass meadows along the Reef places them at high risk of exposure to waters from adjacent water basins and exposure to flood plumes is likely to be a significant factor in structuring inshore seagrass communities (Collier *et al.* 2014; Petus *et al.* 2016). Hence we used river discharge volumes as well as frequency of exposure to inshore flood plumes as indicators of flood plume impacts to seagrasses.

Plume exposure is generated by wet season monitoring under the water quality sub-program (Gruber *et al.* 2020). The inshore water quality sub-program includes a remote sensing component, which describes water quality characteristics for 22 weeks of the wet season (November–April). Water quality is described as colour classes of turbid, brown primary water (class 1–4), green secondary water (class 5), and waters influenced by flood plumes (salinity <30, coloured dissolved organic matter (CDOM) threshold of 0.24 m^{-1} class 6). Colour classes are derived from MODIS True colour satellite images. Exposure to flood plumes is described in this report as frequency of exposure to primary (turbid, sediment laden) or secondary (green, nutrient rich) water during the wet season. Methods are detailed in Devlin *et al.* (2015). Flood plume mapping (Devlin *et al.* 2015) interpreted to water type and frequency of exposure at seagrass sites has been confirmed as a predictor of changes in seagrass abundance (see case study 2, in McKenzie *et al.* 2016).

2.1.2. Environment within seagrass canopy

Autonomous iBTag™ submersible temperature loggers were deployed at all sites identified in Appendix 3, Table 18. The loggers recorded temperature (accuracy 0.0625°C) within the seagrass canopy every 30–90 minutes (Table 1). iBCod™22L submersible temperature loggers were attached to the permanent marker at each site above the sediment-water interface.

Submersible Odyssey™ photosynthetic irradiance autonomous loggers were attached to permanent station markers at 20 intertidal and 4 subtidal seagrass locations from the Cape York region to the Burnett–Mary region i.e. the light loggers are deployed at one site within the locations (Appendix 3, Table 18). Detailed methodology for the light monitoring can be found in McKenzie *et al.* 2018. Measurements were recorded by the logger every 15 minutes and are reported as total daily light (mol m^{-2} d^{-1}). Automatic wiper brushes clean the optical surface of the sensor every 15 minutes to prevent marine organisms fouling.

Sediment type affects seagrass community composition and vice versa (McKenzie et al 2007, Collier et al In Prep). Changes in sediment composition can be an indicator of broader environmental changes (such as sediment and organic matter loads and risk of anoxia), and be an early-warning indicator of changing species composition. Sediment type was recorded at the 33 quadrats at each site in conjunction with seagrass abundance measures using a visual/tactile estimation of sediment grain size composition (0–2 cm below the sediment/water interface) as per standard protocols described in McKenzie *et al.* (2003). Qualitative field descriptions of sediment composition were differentiated according to the Udden-Wentworth grade scale as this approach has previously been shown to provide an equivalent measure to sieve-derived datasets (Hamilton, 1999; McKenzie 2007).

Table 1. *Summary of climate and environment data included in this report, showing historical data range, measurement technique, measurement frequency, and data source. *=variable duration of data availability depending on site*

	Data range	Method	Measurement frequency	Reporting units	Data source
Climate					
Cyclones	1968–2019	remote sensing and observations at nearest weather station	yearly	No. yr^{-1}	Bureau of Meteorology
Rainfall	1889–2019*	rain gauges at nearest weather station	daily	mm mo^{-1} mm yr-1	Bureau of Meteorology
Riverine discharge	1970–2019	water gauging stations at river mouth		L d^{-1} L yr^{-1}	DES#, compiled by Gruber *et al.* 2020
Plume exposure	2006–2019 wet season (Dec–Apr)	remote sensing and field validation	weekly	frequency of water type (1–6) at the site	MMP inshore water quality program (Gruber *et al.* 2020)
Wind	1997–2019*	anemometer at 10 m above the surface, averaged over 10 minutes, at nearest weather station	3pm wind speed	days >25 km hr^{-1}	Bureau of Meteorology
Tidal exposure	1999–2019	wave height buoys at station nearest to monitoring site	3–10 min	hours exposed during daylight	Maritime Safety Queensland, calculated exposure by MMP Inshore Seagrass monitoring
Environment within seagrass canopy					
Water temperature	2002–2019	iBTag	30–90 min	°C, temperature anomalies, exceedance of thresholds	MMP Inshore Seagrass monitoring
Light	2008–2019	Odyssey 2Pi PAR light loggers with wiper unit	15 min	daily light (Id) mol m^{-2} d^{-1} frequency of threshold exceedance (% of days)	MMP Inshore Seagrass monitoring
Sediment grain size	1999–2019	visual / tactile description of sediment grain size composition	3 mo–1yr	proportion mud	MMP Inshore Seagrass monitoring

Department of Environment and Science

2.2 Inshore seagrass and habitat condition

2.2.1 Sampling design and site selection

Sampling is designed to detect changes in inshore seagrass meadows in response to changes in water quality associated with specific catchments or groups of catchments (region) and to disturbance events. The selection of locations/meadows was based upon a number of competing factors:

- meadows were representative of inshore seagrass habitats and seagrass communities across each region (based on Lee Long *et al.* 1993, Lee Long *et al.* 1997, Lee Long *et al.* 1998; McKenzie *et al.* 2000; Rasheed *et al.* 2003; Campbell *et al.* 2002; Goldsworthy 1994)
- where possible include legacy sites (e.g. Seagrass-Watch) or former seagrass research sites (e.g. Dennison *et al.* 1995; Inglis 1999; Thorogood and Boggon 1999; Udy *et al.* 1999; Haynes *et al.* 2000; Campbell and McKenzie 2001; Mellors 2003; Campbell and McKenzie 2004; Limpus *et al.* 2005; McMahon *et al.* 2005; Mellors *et al.* 2005; Lobb 2006)
- a Minimum Detectable Difference (MDD) below 20% (at the 5% level of significance with 80% power) (Bros and Cowell 1987).

Sentinel sites were selected using mapping surveys across the regions prior to site establishment. Ideally mapping was conducted immediately prior to site positioning, however in most cases (60%) it was based on historic (>5 year) information.

Representative meadows were those which covered the greater extent within the inshore region, were generally the dominant seagrass community type and were within Reef baseline abundances (based on Coles *et al.* 2001a; Coles *et al.* 2001c, 2001b, 2001d). To account for spatial heterogeneity of meadows within habitats, at least two sites were selected at each location. If meadow overall extent was larger than ~15 hectares (0.15 km^2), replicate sites were often located within the same meadow (a greater number of sites was desirable with increasing meadow size, however not possible due to funding constraints).

From the onset, inshore seagrass monitoring for the MMP was focused primarily on intertidal/lower littoral seagrass meadows due to:

- accessibility and cost effectiveness (limiting use of vessels and divers)
- Occupational Health and Safety issues with dangerous marine animals (e.g. crocodiles, box jellyfish and irukandji)
- occurrence of meadows in estuarine, coastal and reef habitats across the entire Reef
- where possible, providing an opportunity for citizen involvement, ensuring broad acceptance and ownership of Reef 2050 Plan by the Queensland and Australian community.

Some of the restrictions for working in hazardous waters are overcome by using drop cameras, however, drop cameras only provide abundance measures and do not contribute to the other metrics (e.g. tissue nutrients, reproductive effort). Although considered intertidal within the MMP, the meadows chosen for monitoring were in fact lower littoral (rarely exposed to air).

The long-term median annual daylight exposure (the time intertidal meadows are exposed to air during daylight hours) was 1.7 (all meadows pooled) (Table 19). This limited the time monitoring could be conducted to the very low spring tides within small tidal windows (mostly 1–4 hours per day for 3–6 days per month for 6–9 months of the year). Traditionally, approaches developed for monitoring seagrass to assess changes in water quality were developed for subtidal meadows typified by small tidal ranges (e.g. Florida = 0.7 m, Chesapeake Bay = 0.6 m) and clear waters where the seaward edges of meadows were only determined by light (EHMP 2008).

Depth range monitoring in subtropical/tropical seagrass meadows has had limited success due to logistic/technical issues and non-conformism with traditional ecosystem models because of the complexity (Carruthers *et al.* 2002), including:

- a variety of habitat types (estuarine, coastal, reef and deepwater)
- a large variety of seagrass species with differing life history traits and strategies
- tidal amplitudes spanning 3.42m (Cairns) to 10.4m (Broad Sound) (www.msq.qld.gov.au; Maxwell 1968)
- a variety of sediment substrates, from terrigenous with high organic content, to oligotrophic calcium carbonate
- turbid waters nearshore to clearer further seaward
- grazing dugongs and sea turtles influencing meadow community structure and landscapes
- near-absence of shallow subtidal meadows south of Mackay–Whitsunday due to the large tides which scour the seabed.

Deepwater (>15 m) meadows across the Reef are comprised of only *Halophila* species and are highly variable in abundance and distribution (Lee Long *et al.* 1999; York *et al.* 2015; Chartrand *et al.* 2018). Due to this high variability they do not meet the current criteria for monitoring, as the MDD is very poor at the 5% level of significance with 80% power (McKenzie *et al.* 1998).

Predominately stable lower littoral and shallow (>1.5 m below lowest astronomical tide) subtidal meadows of foundation species (e.g. *Zostera, Halodule*) are best for determining significant change/impact (McKenzie *et al.* 1998). Where possible, shallow subtidal and lower littoral monitoring sites were paired when dominated by similar species.

Due to the high diversity of seagrass species it was decided to direct monitoring toward the foundation seagrass species across the seagrass habitats. A foundation species is the dominant primary producer in an ecosystem both in terms of abundance and influence, playing central roles in sustaining ecosystem services (Angelini *et al.* 2011). The activities of foundation species physically modify the environment and produce and maintain habitats that benefit other organisms that use those habitats (Ellison 2019).

Foundation species are the species types that are at the pinnacle of meadow succession. A highly disturbed meadow (due to wave/wind exposure, or low light regime) might only ever have colonising species as the foundational species, while a less disturbed meadow can have persistent species form the foundation. Also, whether *Zostera muelleri* is a foundation species is influenced by whether it grows in the tropics or in the sub-tropics, as it is more likely to form a foundation species in the sub-tropics even if it is disturbed.

For the seagrass habitats assessed in the MMP, the foundation seagrass species were those species which typified the habitats both in abundance and structure when the meadow was considered in its steady state (opportunistic or persistent) (Kilminster *et al.* 2015). The foundation species were all di-meristematic leaf-replacing forms from the following families: *Cymodocea, Enhalus, Halodule, Thalassia* and *Zostera* (Table 2).

As the major period of runoff from catchments and agricultural lands is the tropical wet season/monsoon (December to April), monitoring is focussed on the late dry (growing) season and late wet season to capture the condition of seagrass pre and post wet.

Sixty-nine sites at 30 locations were assessed during the 2018–19 monitoring period (Appendix 3, Table 18). This covered fourteen coastal, four estuarine and twelve reef locations (i.e. two or three sites at each location).

At the reef locations in the Burdekin and Wet Tropics, intertidal sites were paired with a subtidal site (Table 2). Apart from the 49 MMP long-term monitoring sites, data included nine sites from Seagrass-Watch and eight sites from QPWS to improve the spatial resolution and representation of subtidal habitats (Table 3).

A description of all data collected during the sampling period has been collated by region, site, parameter, and the number of samples collected per sampling period (Table 18). The seagrass species (including foundation) present at each monitoring site is listed in Table 2 and Table 3.

2.2.2 Seagrass abundance, composition and extent

Field survey methodology followed globally standardised protocols (detailed in McKenzie *et al.* (2003)). At each location, with the exception of subtidal sites, sampling included two sites nested within 500 m of each other. Subtidal sites were not always replicated within locations. Intertidal sites were defined as a 5.5 hectare area within a relatively homogenous section of a representative seagrass community/meadow (McKenzie *et al.* 2003).

Monitoring at sites in the late dry (September-November 2018) and late wet (March/April 2019) of each year was conducted by a qualified scientist who was trained in the monitoring protocols. In the centre of each site, during each survey, observers recorded the percentage seagrass cover within 33 quadrats (50 cm × 50 cm, placed every 5 m along three 50 m transects, located 25 m apart). The sampling strategy for subtidal sites was modified to sample along 50 m transects 2–3 m apart (aligned along the depth contour) due to logistics of SCUBA diving in waters of poor visibility.

Seagrass species were identified as per Waycott *et al.* (2004). Species were further categorised according to their life history traits and strategies and classified into colonising, opportunistic or persistent as broadly defined by Kilminister *et al.* (2015) (for detailed methods, see McKenzie *et al.* 2018).

Mapping of the meadow extent and landscape (i.e. patches and scars) within each site was also conducted as part of the monitoring in both the late dry and late wet periods. Mapping followed standard methodologies (McKenzie *et al.* 2001) using a handheld GPS on foot. Where the seagrass landscape tended to grade from dense continuous cover to no cover, over a continuum that included small patches and shoots of decreasing density, the meadow edge was delineated where there was a gap with the distance of more than 3 metres (i.e. accuracy of the GPS). Therefore, the entire 5.5 hectare site was mapped (seagrass and no seagrass).

Table 2. *Inshore sentinel seagrass long-term monitoring site details including presence of foundation (■) and other (□) seagrass species by region * = intertidal, ^=subtidal. CR = Cymodocea rotundata, CS = Cymodocea serrulata, EA = Enhalus acoroides, HD = Halophila decipiens, HO = Halophila ovalis, HS = Halophila spinulosa, HU = Halodule uninervis, SI = Syringodium isoetifolium, TH = Thalassia hemprichii, ZM = Zostera muelleri*

Region	NRM region (*Board*)	Basin	Monitoring location	Site		Latitude		Longitude		CR	CS	EA	HD	HO	HS	HU	SI	TH	ZM
Far Northern	Cape York (*Cape York Natural Resource Management)*	Jacky Jacky / Olive-Pascoe	Shelburne Bay *coastal*	SR1*	Shelburne Bay	11°	53.220	142°	54.853					□		■		■	
				SR2*	Shelburne Bay	11°	53.238	142°	54.940										
			Piper Reef *reef*	FR1*	Farmer Is.	12°	15.339	143°	14.021	■				□				■	
				FR2*	Farmer Is.	12°	15.433	143°	14.186										
		Normanby / Jeannie	Flinders Group *reef*	ST1*	Stanley Island	14°	8.563	144°	14.682	■		■		□		■	■	■	
				ST2*	Stanley Island	14°	8.533	144°	14.590										
			Bathurst Bay *coastal*	BY1*	Bathurst Bay	14°	16.068	144°	13.963	■				□		■	□	■	
				BY2*	Bathurst Bay	14°	16.049	144°	13.897										
Northern	Wet Tropics (*Terrain NRM*)	Daintree	Low Isles *reef*	LI1*	Low Isles	16°	23.110	145°	33.884					□		■		■	
				LI2^	Low Isles	16°	22.973	145°	33.854					□		■			
		Mossman / Barron / Mulgrave-Russell / Johnstone	Yule Point *coastal*	YP1*	Yule Point	16°	34.149	145°	30.756					□		■			□
				YP2*	Yule Point	16°	33.825	145°	30.568										
			Green Island *reef*	GI1*	Green Island	16°	45.709	145°	58.372	■	□			□		■		■	
				GI2*	Green Island	16°	45.696	145°	58.566										
				GI3^	Green Island	16°	45.294	145°	58.379	■	■			□		■	□	■	
		Tully / Murray / Herbert	Mission Beach *coastal*	LB1*	Lugger Bay	17°	57.645	146°	5.603					□		■			
				LB2*	Lugger Bay	17°	57.672	146°	5.626										
			Dunk Island *reef*	DI1*	Pallon Beach	17°	56.646	146°	8.452	■	■			□		■		■	
				DI2*	Pallon Beach	17°	56.734	146°	8.450										
				DI3^	Brammo Bay	17°	55.910	146°	8.417		■		□	□		■			
Central	Burdekin (*NQ Dry Tropics*)	Ross / Burdekin	Magnetic island *reef*	MI1*	Picnic Bay	19°	10.752	146°	50.480					□		■			□
				MI2*	Cockle Bay	19°	10.621	146°	49.730	■	■			□		■	□	■	
				MI3^	Picnic Bay	19°	10.888	146°	50.634		■		□	□	□	■			
			Townsville *coastal*	SB1*	Shelley Beach	19°	11.166	146°	46.272		□			□		■			■
				BB1*	Bushland Beach	19°	11.016	146°	40.951										
			Bowling Green Bay *coastal*	JR1*	Jerona (Barratta CK)	19°	25.369	147°	14.487					□		■			■
				JR2*	Jerona (Barratta CK)	19°	25.272	147°	14.435										
	Mackay–Whitsunday (*Reef Catchments*)	Proserpine / O'Connell	Lindeman Is. *reef*	LN1^	Lindeman Is.	20°	26.293	149°	1.691					□		■			
				LN2^	Lindeman Is.	20°	26.014	149°	1.923										
			Repulse Bay *coastal*	MP2*	Midge Point	20°	38.084	148°	42.107					□		■			■
				MP3*	Midge Point	20°	38.067	148°	42.282										
			Hamilton Island *reef*	HM1*	Catseye Bay - west	20°	20.636	148°	57.439					□		■	□		■
				HM2*	Catseye Bay - east	20°	20.797	148°	58.234										
		Plane	Sarina Inlet *estuarine*	SI1*	Point Salisbury	21°	23.770	149°	18.248					□		□			■
				SI2*	Point Salisbury	21°	23.719	149°	18.288										
Southern	Fitzroy (*Fitzroy Basin Association*)	Shoalwater / Fitzroy	Shoalwater Bay *coastal*	RC1*	Ross Creek	22°	22.912	150°	12.810					□		■			■
				WH1*	Wheelans Hut	22°	23.829	150°	16.520										
			Keppel Islands *reef*	GK1*	Great Keppel Is.	23°	11.776	150°	56.356					□	□	■			■
				GK2*	Great Keppel Is.	23°	11.638	150°	56.364										
		Calliope / Boyne	Gladstone Harbour *estuarine*	GH1*	Pelican Banks	23°	46.015	151°	18.059					□		□*			■
				GH2*	Pelican Banks	23°	45.884	151°	18.233										
	Burnett–Mary (*Burnett–Mary Regional Group*)	Baffle	Rodds Bay *estuarine*	RD1*	Cay Bank	24°	3.467	151°	39.333					□		□			■
				RD2*	Turkey Beach	24°	4.854	151°	39.752										
		Burrum	Hervey Bay *coastal*	BH1*	Burrum Heads	25°	11.290	152°	37.532					□		■			■
				BH3*	Burrum Heads	25°	12.620	152°	38.359										
		Mary	Hervey Bay *estuarine*	UG1*	Urangan	25°	18.053	152°	54.409					□		□			■
				UG2*	Urangan	25°	18.197	152°	54.364										

Table 3*. Additional inshore sentinel seagrass long-term monitoring sites integrated from the Seagrass-Watch (intertidal sites)* and QPWS drop-camera (subtidal sites)^ programs, including presence of foundation (■) and other (□) seagrass species. NRM region from www.nrm.gov.au. * = intertidal, ^ =subtidal.*

Region	NRM region (*Board*)	Basin	Monitoring location	Site		Latitude		Longitude		CR	CS	EA	HD	HO	HS	HU	SI	TH	ZM
Far Northern	Cape York *(Cape York Nat Res Manage)*	Lockhart	Weymouth Bay *reef*	YY1*	Yum Yum Beach	12°	34.247	143°	21.639	■	■	■		□		■		■	
			Lloyd Bay *coastal*	LR1^	Lloyd Bay	12°	47.792	143°	29.118					□	□	■			
				LR2^	Lloyd Bay	12°	49.502	143°	28.488										
		Normanby / Jeannie	Flinders Group *reef*	FG1^	Flinders Island	14°	10.9464	144°	13.522					□	□	■			
				FG2^	Flinders Island	14°	10.932	144°	13.522										
			Bathurst Bay *coastal*	BY3^	Bathurst Bay	14°	16.556	144°	17.069					□		■			
				BY4^	Bathurst Bay	14°	16.482	144°	18.006										
		Endeavour	Archer Point *reef*	AP1*	Archer Point	15°	36.508	145°	19.147	■	■	□		□		■		■	□
				AP2*	Archer Point	15°	36.533	145°	19.118										
Northern	Wet Tropics	Tully / Murray / Herbert	Rockingham Bay *reef*	GO1	Goold Island	18°	10.428	146°	9.186	■	■			□		■			
			Missionary Bay *coastal*	MS1^	Cape Richards	18°	12.950	146°	12.753					□		■			
				MS2^	Macushla	18°	12.316	146°	13.010										
Central	Burdekin (*NQ Dry Tropics*)	Ross / Burdekin	Townsville *coastal*	SB2*	Shelley Beach	19°	10.939	146°	45.767		□			□		■			■
	Mackay–Whitsunday (*Reef Catchments*)	Don	Shoal Bay *reef*	HB1*	Hydeaway Bay	20°	4.481	148°	28.943	■				□		■		■	
				HB2*	Hydeaway Bay	20°	4.292	148°	28.861										
		Proserpine	Pioneer Bay *coastal*	PI2*	Pigeon Island	20°	16.163	148°	41.585					□	□	■			■
				PI3*	Pigeon Island	20°	16.232	148°	41.850										
		Proserpine / O'Connell	Whitsunday Island *reef*	TO1^	Tongue Bay	20°	14.399	149°	0.934					□		■		■	
				TO2^	Tongue Bay	20°	14.495	149°	0.697										
		O'Connell / Pioneer	Newry Islands *coastal*	SH1*	St Helens Bch	20°	49.344	148°	50.124					□		■			■
				NB1^	Newry Bay	20°	52.057	148°	55.531		■			□	□	■	■		
				NB2^	Newry Bay	20°	52.325	148°	55.423										
		Plane	Clairview *coastal*	CV1*	Clairview	22°	6.2592	149°	31.9902					□		■			■
				CV2*	Clairview	22°	6.4932	149°	32.0748										

2.2.3 Seagrass reproductive status

Seagrass reproductive health was assessed from samples collected in the late dry 2018 and late wet 2019 at locations identified in Table 2. Samples were processed according to standard methodologies (McKenzie *et al.* 2019).

In the field, 15 cores (100 mm diameter x 100 mm depth) of seagrass were collected haphazardly within each site from an area adjacent (of similar cover and species composition) to the monitoring transects. In the laboratory, reproductive structures (spathes, fruits, female and male flowers) of plants from each core were identified and counted for each sample and species. Reproductive effort was calculated as number of reproductive structures (fruits, flowers, spathes; species pooled) per core for analysis.

Seeds banks and abundance of germinated seeds were sampled according to standard methods (McKenzie *et al.* 2019) by sieving (2mm mesh) 30 cores (50mm diameter, 100mm depth) of sediment collected across each site and counting the seeds retained in each. For *Zostera muelleri*, where the seed are <1 mm diameter, intact cores (18) were collected and returned to the laboratory where they were washed through a 710 µm sieve and seeds identified using a hand lens/microscope.

2.2.4 Seagrass leaf tissue nutrients

In the late dry season (October 2018), leaf tissue samples from the foundational seagrass species were collected from each monitoring site for nutrient content analysis (Table 2). For nutrient status comparisons, collections are made during the growth season (e.g. late dry when nutrient contents are at a minimum) (Mellors *et al.* 2005) and at the same time of the year and at the same depth at the different localities (Borum *et al.* 2004). Two to three handfuls of shoots from three haphazardly placed 0.25 m^2 quadrats were collected from an area adjacent (of similar cover and species composition) to the monitoring transects.

Species within the sample are separated, and all species (except *Halophila* spp.) were analysed for tissue nutrient content. All leaves within the sample were separated from the below ground material in the laboratory and epiphytic algae removed by gently scraping. Dried and milled leaf samples were analysed according to McKenzie *et al.* (2019). Elemental ratios (C:N:P) were calculated on a mole:mole basis using atomic weights (i.e. C=12, N=14, P=31).

2.2.5 Epiphytes and macroalgae

Epiphyte and macroalgae cover were measured according to standard methods (McKenzie *et al.* 2003). The total percentage of leaf surface area (both sides, all species pooled) covered by epiphytes and percentage of quadrat area covered by macroalgae, were measured each monitoring event. Values were compared against the Reef long-term average (1999-2010) calculated for each habitat type.

2.3 Data analyses

All seagrass condition indicators had uncertainties associated with their measurements at the lowest reporting levels (e.g. percentage, count, ratio, etc.) which was presented as Standard Error (calculated from the site, day, or core standard deviations). To propagate the uncertainty (i.e. propagation of error) through each higher level of aggregation (e.g. habitat, NRM region and GBR), the square root of the sum of squares approach (using the SE at each subsequent level) was applied (Ku 1966). The same propagation of error approach was applied to the annual seagrass report card scores to calculate a more exact measure of uncertainty in the three seagrass indicators and overall index.

Results are presented to reveal temporal changes in seagrass community attributes and key environmental variables. Generalised additive mixed effects models (GAMMs) are fitted to seagrass attributes for each habitat and NRM, to identify the presence and consistency of trends, using the mgcv (Wood 2006;Wood 2014) package in R 3.4.3 (R Core Team 2014). GAMMs (Wood 2006) were used to interrogate the irregularly-spaced time-series into its trend cycles (long-term) and periodic (seasonal) components.

GAMMs are an extension of additive models, which allow flexible modelling of non-linear relationships by incorporating penalized regression spline types of smoothing functions into the estimation process. The degree of smoothing of each smooth term (and by extension, the estimated degrees of freedom of each smoother) is treated as a random effect and thus estimable via its variance as with other effects in a mixed modelling structure (Wood 2006). Results of these analyses are graphically presented in a consistent format: predicted values from the model were plotted as bold black lines, the 95% confidence intervals of these trends delimited by grey shading.

Several GAMMs were used on seagrass cover and C:N ratio to tease out trends at the habitat, regional and location scale over time. The random effects were incorporated as a nested structure of quadrat within transect within site, to account for spatial correlation. As part of our regular validation process the residuals of all models were checked for violations of the generalised model assumptions. In few instances the random effects structure caused issues and the transect level had to be omitted.

Per cent seagrass cover data GAMMs were fitted using a quasi-binomial distribution due to the proportional (bound between 0 and 1) nature of the data. Raw data at the quadrat level was used to provide the maximum resolution for modelling. However, this led to a very large proportion of zeroes in some data sets causing high heterogeneity of variance for some models. For this reason, GAMMs for reproductive effort, epiphytes, macroalgae cover are not presented and the inclusion in future reports of zero-inflated GAMMs is being investigated. C:N data models were fitted using a gamma distribution due to the strictly positive continuous nature of the data. Here the random effects consisted of species nested within site.

For the analyses of the various tissue nutrients and isotopes variables Generalised Linear Mixed Models (GLMMs) were used instead of GAMMs as these samples are only collected once a year, and due to the low frequency of sampling the use of a smoother (GAMM) is not recommended. The tissue nutrient variables (C:N, C:P, N:P, %N, %P) were analysed using the R-INLA (Rue *et al.* 2009) package with a gamma distribution and the isotopes variables ($\delta^{13}C$ and $\delta^{15}N$) with a Gaussian distribution. Similarly, to the C:N GAMMs, the random effects consisted of species nested within site.

Trend analysis was conducted to determine if there was a significant trend (reduction or increase) in seagrass abundance (per cent cover) at a particular site (averaged by sampling event) over all time periods. A Mann-Kendall test was performed using the “trend” package in R 3.2.1 (R Core Team 2014). Mann-Kendall is a common non-parametric test used to detect overall trends over time. The measure of the ranked correlation is the Kendall’s tau coefficient (Kendall-τ), which is the proportion of up-movements against time vs the proportion of down-movements, looking at all possible pairwise time-differences. As the test assumes independence between observations, data was checked for autocorrelation and if present a corrected p-value was calculated using the “modifiedmk” package (Hamed and Rao 1998).

The majority of meadows have been in a "recovery mode" since losses during the periods 2008–2009 to 2010–2011. As such, there have been periods of limited sample availability (e.g. for tissue nutrients), and the absence of data has restricted whether multivariate analysis can be undertaken routinely. Analysis is currently underway to more fully interrogate the temporal and covariate components of the data as the time series of observations lengthen.

2.4 Reporting approach

The data is presented in a number of ways depending on the indicator and section of the report:

- Report card scores for seagrass condition are presented at the start of each section. These are a numerical summary of the condition within the region relative to a regional baseline (described further below)
- Climate and environmental pressures are presented as averages (daily, monthly or annual) and threshold exceedance
- Seagrass community data such as seagrass abundance, leaf tissue nutrients are presented as averages (sampling event, season or monitoring period with SE) and threshold exceedance data
- Seagrass ecosystem data such as sediment composition, epiphyte and macroalgae are presented as averages (sampling event, season or monitoring period) and relative to the long-term
- Trend analysis (GAMM plots) are also used to explore the long-term temporal trends in biological and environmental indicators.

Within each region, estuarine and coastal habitat boundaries were delineated based on the Queensland coastal waterways geomorphic habitat mapping, Version 2 (1:100 000 scale digital data) (Heap *et al.* 2015).

Reef habitat boundaries were determined using the AUSLIG (now the National Mapping Division of Geosciences Australia) geodata topographic basemap (1:100 000 scale digital data).

2.5 Calculating report card scores

Three indicators (presented as unitless scores) are used for the seagrass component of the Marine Results report and Reef report card:

- seagrass abundance (per cent cover)
- reproductive effort
- nutrient status (leaf tissue C:N ratio).

A seagrass condition index (score) is reported for each monitoring region based on changes in each of the indicators relative to a baseline. The methods for score calculation were chosen by the Paddock to Reef Integration Team and all report card scores are transformed to a five point scale from 0 to 100 to allow integration with other components of the Reef report card (Department of the Premier and Cabinet 2014). The methods and scoring system for the report card are detailed below. *Please note that the scale from 0 to 100 is unitless and should not be interpreted as a proportion or ratio.*

2.5.1 Seagrass abundance

Seagrass abundance is measured using the median seagrass per cent cover relative to the site or reference guideline (habitat type within each NRM region). Abundance guidelines (threshold levels) were determined using the long-term (>4 years) baseline where the percentile variance plateaued (generally 15-20 sampling events), thereby providing an estimate of the true percentile value (McKenzie 2009). Guidelines for individual sites were only applied if the conditions of the site aligned with reference conditions and the site had been subject to minimal/limited disturbance for 3–5 years (see Appendix 2, Table 17).

Abundance state at each site for each monitoring event was allocated a grade:

- *very good*, median per cent cover at or above 75th percentile
- *good*, median per cent cover at or above 50th percentile
- *moderate*, median per cent cover below 50th percentile and at or above low guideline
- *poor*, median per cent cover below low guideline
- *very poor*, median per cent cover below low guideline and declined by >20% since previous sampling event).

The choice of whether the 20th or 10th percentile was used for the low guideline depended on the within-site variability; generally the 20th percentile is used, unless within-site variability was low (e.g. CV<0.6), whereby the 10th percentile was more appropriate as the variance would primarily be the result of natural seasonal fluctuations (i.e. nearly every seasonal low would fall below the 20th percentile). Details on the per cent cover guidelines can be found in Appendix 2.

A grade score from 0 to 100 (Table 4) was then assigned to enable integration with other seagrass indicators and other components of the Reef report card (Department of the Premier and Cabinet 2014). Annual seagrass abundance scores were calculated using the average grade score for each site (including all sampling events per year), each habitat and each NRM.

Table 4. *Scoring threshold table to determine seagrass abundance status. low = 10th or 20th percentile guideline. NB: scores are unitless.*

grade	percentile category	score	status
very good	75-100	100	81 - 100
good	50-75	75	61 - 80
moderate	low-50	50	41 - 60
poor	<low	25	21 - 40
very poor	<low by >20%	0	0 - 20

2.5.2 Seagrass reproductive effort

As most seagrass species of the Reef flower in the late dry season, reproductive effort is sampled during the late dry season to capture the sexual reproductive peak.

During the current monitoring period, the total number of reproductive structures per core (inflorescence, fruit, spathe, seed) was measured at each site in the late dry season (September-November 2018), and a grade score determined after normalising against the Reef habitat baseline (see Appendix 2) and using the ratio to rank the score from very good to very poor (Table 5).

Table 5. *Scores for late dry monitoring period reproductive effort average against Reef habitat baseline. NB: scores are unitless.*

grade	***Reproductive Effort*** *(monitoring period / baseline)*	***ratio***	***score***	***0-100 score***	***status***
very good	≥4	4.0	4	100	81 - 100
good	2 to <4	2.0	3	75	61 - 80
moderate	1 to <2	1.0	2	50	41 - 60
poor	0.5 to <1	0.5	1	25	21 - 40
very poor	<0.5	0.0	0	0	0 - 20

2.5.3 Seagrass nutrient status

Tissue nutrient content of seagrass leaves including carbon (C), nitrogen (N) and phosphorus (P) were measured annually. The absolute tissue nutrient concentrations (%C, %N and %P) are used to calculate the atomic ratio of nutrients in seagrass leaves (see Appendix 2). The C:N ratio was chosen for the purpose of the report card score as it is the ratio that indicates a change in either light or nitrogen availability at the meadow scale. C:N ratios were compared to a global average value of 20:1 (Atkinson and Smith 1983; Fourqurean *et al.* 1992), with values less than 20:1 indicating either reduced light or excess N is available to the seagrass. Values higher than 20:1 suggest light saturation and low nitrogen availability (Abal *et al.* 1994; Grice *et al.* 1996; Udy and Dennison 1997b). C:N ratios from the late dry season (September-November 2018) were categorised on their departure from the guideline and transformed to a score (see Appendix 2) which was then graded from very good to very poor (Table 6).

Table 6. *Scores for leaf tissue C:N against guideline to determine light and nutrient availability. NB: scores are unitless.*

grade	C:N ratio range	Score ($\overline{R}$) range and status
very good	C:N ratio >30*	81 - 100
good	C:N ratio 25-30	61 - 80
moderate	C:N ratio 20-25	41 - 60
poor	C:N ratio 15-20	21 - 40
very poor	C:N ratio <15*	0 - 20

2.5.4 Seagrass condition index

The seagrass condition index is an average score (0–100) of the three seagrass condition indicators:

- seagrass abundance (per cent cover)
- reproductive effort
- leaf tissue nutrients.

Each indicator is equally weighted, in accordance with the Paddock to Reef Integration Team's original recommendations. Until the Paddock to Reef Independent Science Panel has reviewed the findings and recommendations of the case study, the equal weighting previously used will remain. To calculate the overall score for seagrass of the Reef, the regional scores were weighted on the percentage of World Heritage Area seagrass (shallower than 15 m) within that region (Table 7). *Please note: Cape York omitted from the score in reporting prior to 2012 due to poor representation of inshore monitoring sites*.

Table 7. *Area of seagrass shallower than 15 m in each region within the boundaries of the World Heritage Area.(from McKenzie et al. 2014b; McKenzie et al. 2014c; Carter et al. 2016; Waterhouse et al. 2016).*

NRM	Area of seagrass (km^2)	Per cent of World Heritage Area
Cape York	2,078	0.60
Wet Tropics	207	0.06
Burdekin	587	0.17
Mackay–Whitsunday	215	0.06
Fitzroy	257	0.07
Burnett–Mary	120	0.03
World Heritage Area	**3,464**	**1.00**

2.5.5 Revision of seagrass scores

A comprehensive quality assurance and quality control assessment of seagrass data was undertaken this year, in parallel with generating seagrass scores through a new statistical program using 'R' scripts. This review identified errors in past data, which only appeared in the 2017-18 report card. Identification of these minor errors will be corrected for seagrass abundance and nutrient scores over the monitoring period from 2005-06 to 2017-18.

The Quality Assurance and Control assessment found that:

- Some calculations did not include all data during a transition in methodology
- Some calculations incorrectly included additional collections outside the late dry (only applies to 2017-18 published report card, errors didn't occur in previous years)
- Some data was scored against the wrong guideline (only applies to 2017-18 published report card, for sites established in 2017)
- Some sites were allocated to the wrong seagrass habitat type

Some scores and grades were affected. Most of the recalculations resulted in minor changes to scores of less than one, and no change in grades – and therefore have minimal impact on the evaluation of condition and trend. However, there were 5 corrections to indicators scores of 2 or more, and one of these resulted in a grade change for an indicator, with flow-on effects. This change relates to the 2017-18 scores for Cape York where:

- the abundance indicator was amended from 40 to 45 (and improvement in grade from poor to moderate)
- the score for the region was amended from 25 to 27 (no change in grade)
- the marine score for the Cape York score was amended from 43 to 44 (no change in grade)
- the abundance indicator for the Reef was amended from 42 to 45 (no change in grade)
- the 2017-18 seagrass score for the Reef was amended from 29 to 30 (no change in grade).

3 Drivers and pressures influencing seagrass meadows in 2018–19

The following section provides detail on the overall climate and environmental pressures during the 2018–19 monitoring period, at a relatively broad level as context for understanding trends in seagrass condition. It includes:

- climate, river discharge and flood plume exposure
- within-canopy light
- within-canopy temperature and threshold exceedance
- seagrass meadows sediment characteristics.

The ensuing section contains data on local environmental pressures and supporting data is detailed within Appendix 3 and 4:

3.1 Summary

Environmental stressors in 2018–19 were above average for rainfall and river discharge, but relatively benign for within canopy light and water temperature (Table 8). However, there was large degree of variability in rainfall and river discharge, in particular across the inshore Reef. Rivers within Cape York, the Wet Tropics, Burdekin and Mackay–Whitsunday regions exceeded their long-term medians, while they were below the long-term median in the Fitzroy and Burnett–Mary regions.

The frequency with which the sentinel seagrass sites were exposed to ‘brown’ sediment-laden (1–4) and ‘green’ phytoplankton-rich waters (5) during the wet season was also slightly elevated across the entire Reef, even in the southern regions where discharge was low (Figure 9). The presence of this coloured water is affected by resuspension-driven events as well as discharge.

Table 8. *Summary of environmental conditions at monitoring sites across the Reef in 2018-19 compared to the long-term average (range indicated for each data set). *intertidal only.*

Environmental pressure	Long-term average	2018–19
Climate		
Cyclones (1968–2018)	4	3
Daily rainfall (1960–1991)	4.0 mm d^{-1}	4.4 mm d^{-1}
Riverine discharge (1986–2018)	43,099,046 ML yr^{-1}	94,323,378 ML yr^{-1}
Wet season turbid water exposure (2003–2018)	92%	94%
Within seagrass canopy		
Within canopy temperature (±) (max) (2003–2018)*	25.7 ±0.1°C (46.6°C)	25.7 ±0.1°C (41.1°C)
Within canopy light (2008–2018) annual average	12.4 mol m^{-2} d^{-1}	12.0 mol m^{-2} d^{-1}
(min site–max site)	(2.6–20.5 mol m^{-2} d^{-1})	(3.5–22.1 mol m^{-2} d^{-1})
Proportion mud		
estuary intertidal (1999–2018)	49.2 ±2.1%	46.2 ±3.5%
coast intertidal (1999–2018)	28.4 ±2.1%	28.4 ±4.7%
coast subtidal (2015–2018)	50.6 ±1.8%	46.7 ±4.7%
reef intertidal (2001–2018)	5.0 ±1.2%	4.5 ±2.7%
reef subtidal (2008–2018)	7.2 ±0.4%	10.2 ±1.0%

Climatic and environmental pressures may have affected seagrass by reducing daily incident light reaching the seagrass canopy in some regions and habitats. Light levels, which are measured at the location level, were lower than estimated annual light requirements (10 mol m^{-2} d^{-1}) at 6 locations. The greatest deviation in benthic light from the long-term was in Cape York and in the southern Wet Tropics (Figure 8).The Burdekin region was the only region with above-average light levels in 2018–19, but this is largely attributed to high levels prior to the wet season, and some missing data (a consequence of logger losses) from the wet season when rivers across the region were in flood.

Within canopy temperatures in 2018–19 were cooler than the previous five reporting years in all regions, on average, except for the Fitzroy where they were slightly higher than average (Figure 8). The number of extreme heat days, were also lower than the previous five years, except in the Wet Tropics, where it was the second highest number of extreme days on record (Figure 11).

There were three tropical cyclones that entered the Reef in 2018–19, including tropical cyclones Owen, Penny and Trevor (see Gruber et al 2020). Cyclone Trevor was likely to have had the greatest impact, as a category 4 that crossed the Cape York coast near Lockhardt River in March 2019. It may have directly impacted Piper Reef (FR1 and FR2), but surveys in this reporting period at these sites occurred before the cyclone reached the region so the effects, if any, will be observed in 2019–20 reporting. In addition to these, there was an intense tropical low leading to major flooding in the Herbert, Black-Ross and Haughton basins including severe flooding in and around Townsville in February 2019.

3.2 Rainfall

Rainfall was above the long-term average throughout most of the central and northern GBR from the Plane basin north to the Jacky Jacky (Figure 6) (Figure 7). The largest positive deviations from the long-term averages occurred in the southern Cape York and northern Wet Tropics regions and in the smaller basins within the Burdekin region. It was slightly drier than the long-term average in the southern GBR basins.

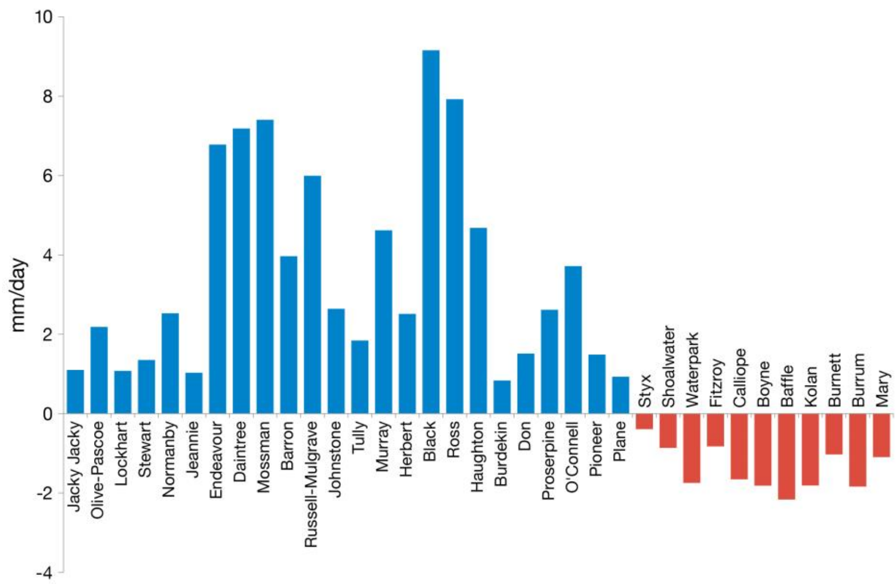


Figure 6. Difference between *annual average daily wet season rainfall (December 2017–April 2019) and the long-term average (1961–1990). Red and blue bars denote basins with rainfall below and above the long-term average, respectively. Note that the basins are ordered from north to south (left to right). Compiled by Gruber et al. 2020.*

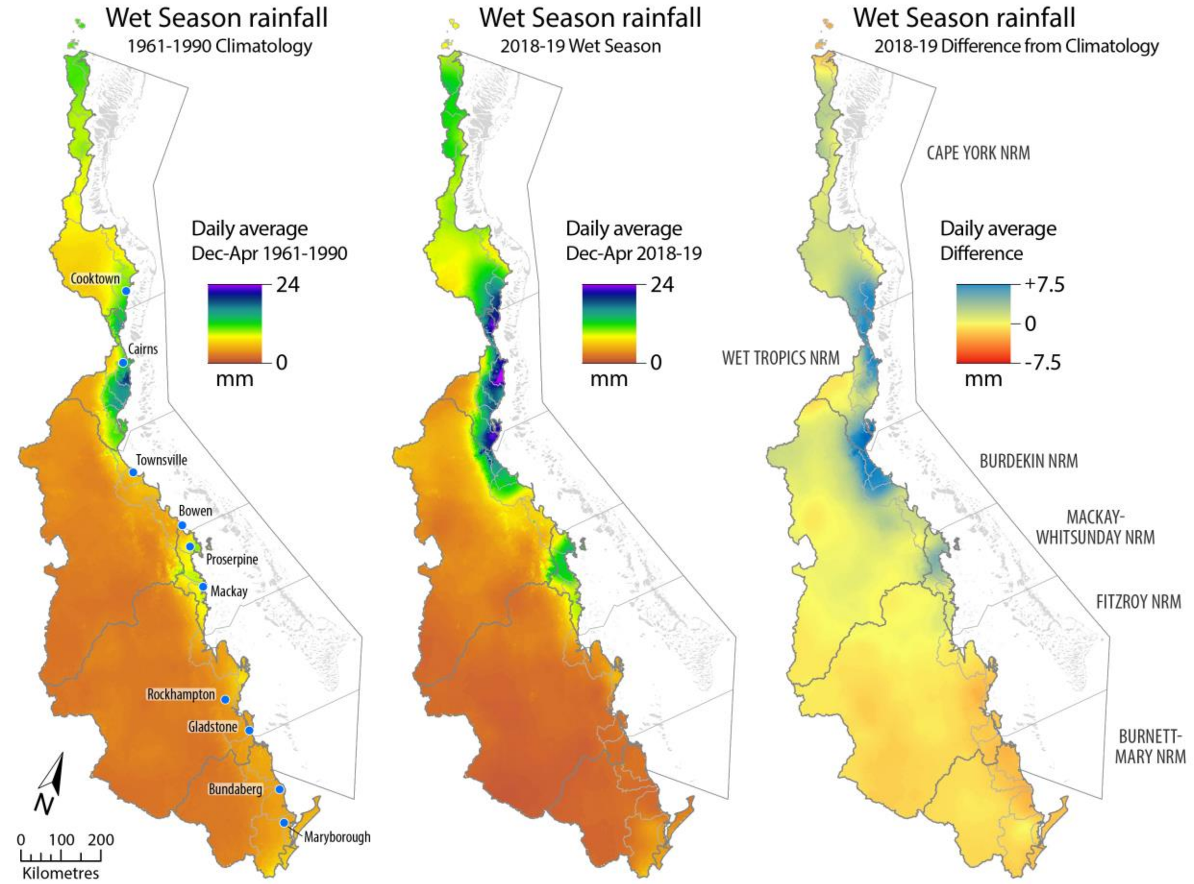


Figure 7. *Average daily rainfall (mm/day) in the Reef catchment: (left) long-term annual average (1961–1990; time period produced by BOM), (centre) 2018–19 and (right) the difference between the long-term annual average and 2018–19 rainfall patterns. From Gruber et al. 2020.*

3.3 River discharge

Annual river discharge for the entire GBR was above the long-term average in 2018-19 (Table 8). River discharge was more than 1.5 times the long-term median from most rivers in the central and northern GBR. In particular discharge from catchments in the Cape York region was the highest on record (since 2002–03), and discharge in the Burdekin region was more than three times the long-term median (Table 9). By contrast, discharge was below average in most rivers within the southern GBR in the Fitzroy in Burnett–Mary regions (Table 9).

Table 9. Annual water year discharge (ML) of the main GBR rivers (1 October 2017 to 30 September 2018, inclusive) compared to the previous seven wet seasons and long-term (LT) median discharge (1986–87 to 2018–19). Colours indicate levels above the long-term median: yellow = 1.5 to 2 times, orange = 2 to 3 times and red = greater than 3 times. Compiled by Gruber et al. 2020.

Basin	LT median	2015 - 2016	2016 - 2017	2017 - 2018	2018 - 2019
Jacky Jacky Creek	2,047,129	913,417	1,701,199	2,689,450	3,124,009
Olive Pascoe River	2,580,727	788,484	2,978,821	3,424,596	6,992,798
Lockhart River	1,634,460	499,373	1,886,587	2,168,911	4,428,772
Stewart River	674,618	311,901	685,263	826,499	3,109,052
Normanby River	4,159,062	3,407,359	3,780,651	4,333,023	12,102,053
Jeannie River	1,263,328	1,581,015	1,746,929	1,721,175	3,350,682
Endeavour River	1,393,744	1,407,701	1,665,116	1,796,913	3,847,478
Daintree River	1,512,054	1,433,059	1,590,225	1,439,220	4,752,327
Mossman River	858,320	885,529	812,585	1,069,336	1,885,921
Barron River	574,567	199,635	313,952	946,635	1,535,892
Mulgrave-Russell River	2,600,465	1,898,065	1,759,178	3,359,834	3,550,093
Johnstone River	3,953,262	2,846,943	3,348,014	4,950,329	4,774,747
Tully River	3,241,383	2,697,539	2,840,476	3,883,954	4,020,452
Murray River	380,472	301,879	293,742	521,465	519,739
Herbert River	3,556,376	1,895,526	2,248,436	6,385,655	5,707,209
Black River	208,308	109,784	64,449	386,030	965,544
Ross River	377,011	32,399	41,177	83,113	2,371,556
Haughton River	419,051	189,006	283,551	598,668	2,363,209
Burdekin River	4,406,780	1,807,104	4,165,129	5,542,306	17,451,417
Don River	508,117	173,549	1,081,946	321,875	1,356,004
Proserpine River	284,542	101,490	539,710	174,183	837,962
O'Connell River	478,097	273,420	894,975	260,937	1,223,297
Pioneer River	692,342	597,117	1,388,687	249,530	1,158,768
Plane Creek	309,931	246,274	761,503	75,052	351,879
Styx River	155,384	284,587	420,353	218,115	109,376
Shoalwater Creek	129,487	237,156	350,294	181,763	91,147
Water Park Creek	97,115	177,867	262,721	136,322	68,360
Fitzroy River	2,852,307	3,589,342	6,170,044	954,533	1,339,964
Calliope River	152,965	148,547	406,321	141,438	2,682
Boyne River	38,691	37,574	102,775	35,775	678
Baffle Creek	215,446	150,710	486,235	1,081,646	930
Kolan River	52,455	120,977	190,476	325,578	4,958
Burnett River	230,755	381,054	536,242	849,051	202,436
Burrum River	79,112	289,364	387,027	715,449	63,972
Mary River	981,183	412,160	499,295	1,630,741	658,014

3.4 Turbid water exposure and flood plume extent

The frequency of exposure to turbid water (colour classes 1–5), plume extent, and the within-canopy environmental pressures daily light and water temperature are summarised in Figure 8.

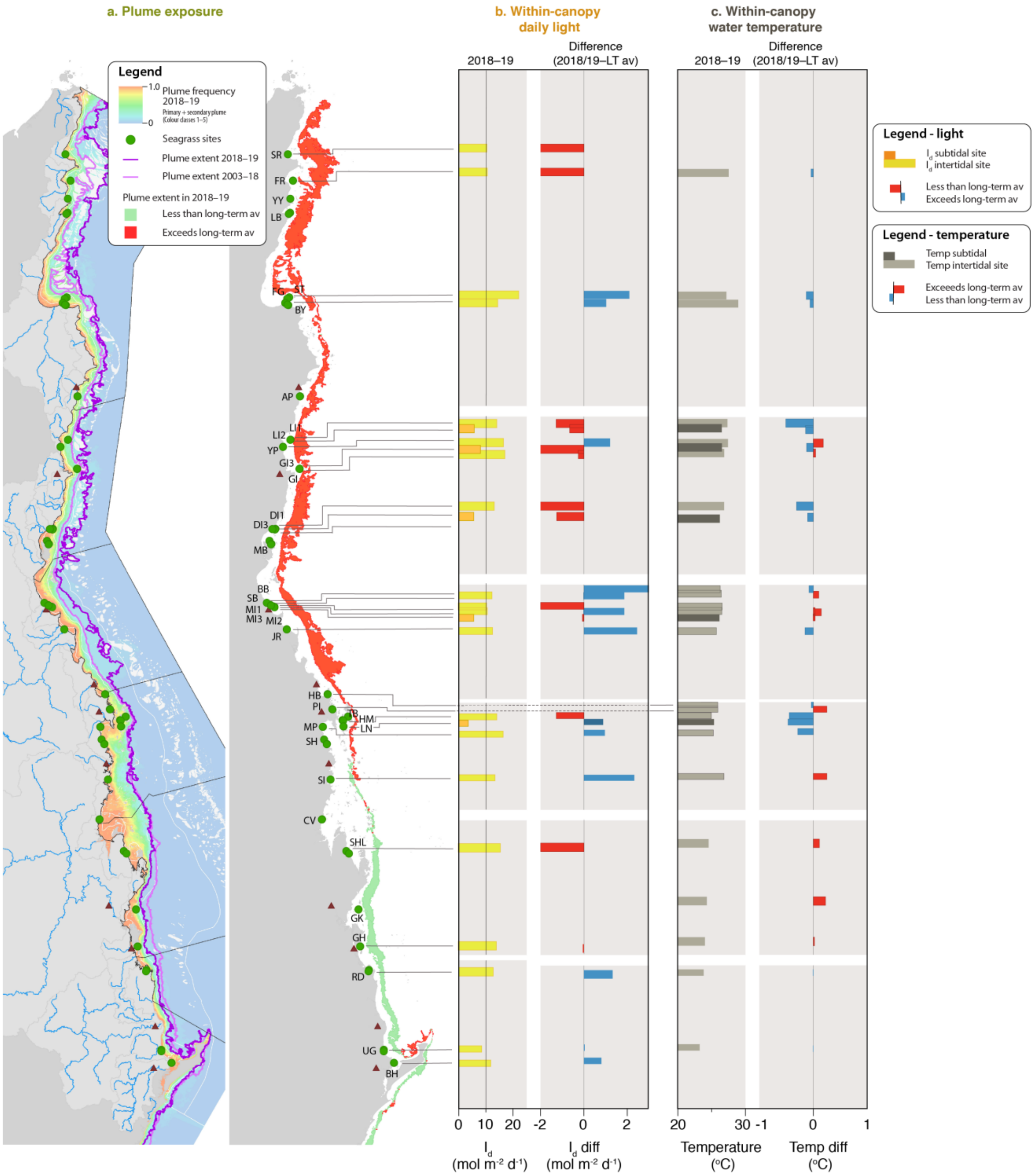


Figure 8. *Environmental pressures in the Reef during 2018–19 and relative to long-term: a. Frequency of* turbid water *(colour classes 1–5, primary and secondary water) exposure shown in the left-hand panel in the Reef from December 2018 to April 2019 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed), and right-hand panel the plume extent (10% boundary) in 2018–19 relative to the long-term average, with red showing that plumes extended further in 2018–19 and green showing they did not extend as far; b. within canopy daily light for all sites, and the deviation in daily light relative to the*

long-term average; and c. within canopy water temperature, and deviation water temperature from the long-term average.

River plumes reached all seagrass locations in 2018–19 as is characteristic of inshore conditions over the long-term (2003–18, Figure 8). However, river plumes extended further seaward (shown in red) than the long-term average and well beyond most seagrass monitoring locations throughout the northern and central GBR. This also indicates increased influence of river plumes at seagrass monitoring locations in this reporting period (Figure 8). River plumes extended less far seaward in the southern GBR (shown in green), which is indicative of lower influence of river plumes (Figure 8, panel 2).

The frequency of exposure to colour classes 1 to 4 ('brown' turbid water) during the wet season weeks (December 2018–April 2019) is typically very high in the inshore regions of the Reef, but was marginally above multiannual conditions in all regions except the Fitzroy and Burnett–Mary regions (Figure 9). The frequency of exposure to colour classes 1 to 5 (including 'green' turbid water), shows that all regions were at, or marginally above, the multiannual level of exposure. In the Burdekin region, the exposure to classes 1-5 was similar to the long-term average, despite an increase in the exposure to the brown classes (1-4) (Figure 8).

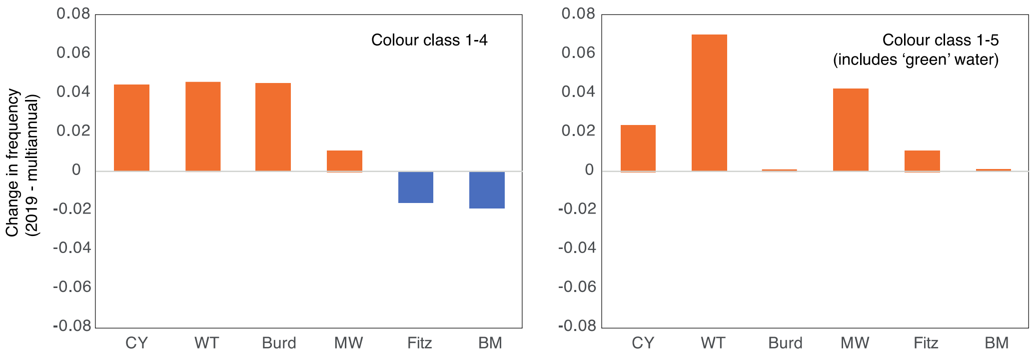


Figure 9. *Difference in the frequency of exposure to water colour classes 1 to 4 (left) and 1 to 5 (right) at seagrass monitoring sites during the wet season (December 2018–April 2019) compared to the long-term multiannual exposure (2003–2018).*

3.5 Daily incident light

Daily light in shallow habitats can be affected by water quality, depth of the site and cloudiness, which affects the frequency and duration of exposure to full sunlight at low tide (Anthony *et al.* 2004; Fabricius *et al.* 2012). Differences in I_d among seagrass meadows reported here is largely a reflection of site-specific differences in water quality, except for in reef subtidal communities where depth results in lower benthic light compared to adjacent reef intertidal communities.

Daily light reaching the top of the seagrass canopy in the Reef in 2018–19 was 12.0 mol m^{-2} d^{-1} when averaged for all sites (Table 8), compared to a long-term average of 12.4 mol m^{-2} d^{-1}. There are regional, habitat and location levels differences.

Daily light in the regions in 2018–19 from north to south were:

- Cape York (13.7 mol m^{-2} d^{-1})
- northern Wet Tropics (12.3 mol m^{-2} d^{-1})
- southern Wet Tropics (9.3 mol m^{-2} d^{-1})
- Burdekin (11.5 mol m^{-2} d^{-1})
- Mackay–Whitsunday (11.5 mol m^{-2} d^{-1})

- Fitzroy (14.3 mol m^{-2} d^{-1})
- Burnett–Mary (11.2 mol m^{-2} d^{-1})

Daily light in the habitats in 2018–19 from highest to lowest were:

- reef intertidal habitat, n = 9 (14.6 mol m^{-2} d^{-1})
- coastal intertidal locations, n = 10 (13.9 mol m^{-2} d^{-1})
- estuarine sites, n = 3 (11.6 mol m^{-2} d^{-1})
- reef subtidal sites, n = 5 (5.8 mol m^{-2} d^{-1}).

Daily light for each of the sites is presented in Figure 8. There were 6 locations in which the annual daily light level was lower than 10 mol m^{-2} d^{-1}, a light threshold that is likely to support long-term growth requirements of the species in these habitats (Collier et al 2016). Five of these were the subtidal sites, and the sixth site below 10 mol m^{-2} d^{-1} was an intertidal site at Rodds Bay in the Burnett–Mary (RD3). There were 13 locations in which daily light was lower than the long-term average, with many of these in the wet tropics (Figure 8).

Long-term trends show a peak in within canopy daily light occurs in September to December as incident solar irradiation reaches its maximum and prior to wet season conditions (Figure 10). The lowest light levels typically occur in the wet season, particularly in January to April. In 2018–19, daily light dropped rapidly from dry season maxima to wet season lows at the end of December.

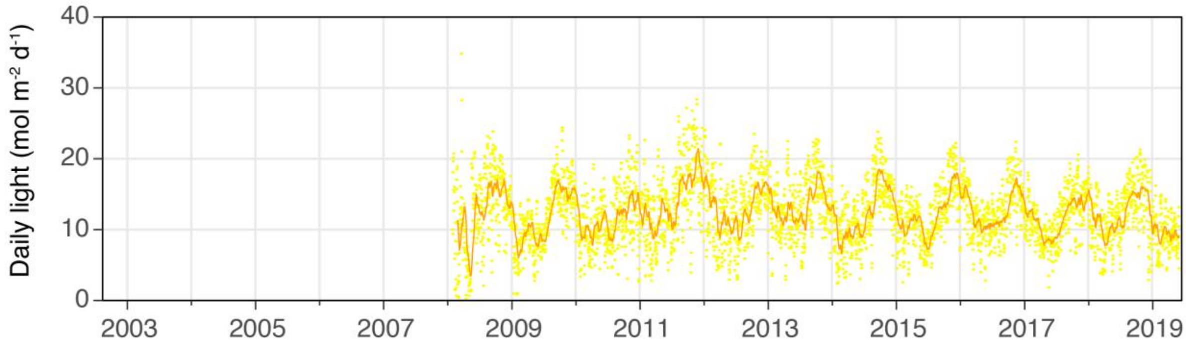


Figure 10. *Daily light for all sites combined from 2008 to 2019. In 2008–2009, light data is from the Burdekin and Wet Tropics regions only. Other regions were included from 2009–2010, with Cape York added post 2012–2013 reporting period.*

3.6 Within-canopy seawater temperature

Daily within-canopy seawater temperature across the Reef in 2018–19 was cooler than the previous four reporting periods, when there were a record number of temperature exceedances, and widespread bleaching throughout the Reef (Figure 11). The 2018–19 Reef temperature was on average (25.7 ±0.15°C) similar to the long-term (2003–18, 25.7°C) (Table 8). However, there were regional and habitat differences relative to the long-term (Figure 8).

Daily within-canopy seawater temperatures in the regions in 2018–19 (including number of days above 35°C and 40°C) from warmest to coolest difference (* = greater than 0.5°C) relative to the long-term average ($^{\uparrow}$ = greater than, $^{\updownarrow}$ = similar to long-term) were:

- Cape York (avg = 27.5°C, max = 37.7°C, days>35°C = 33)$^{\uparrow}$*
- Burnett–Mary (avg = 23.6°C, max = 38.3°C, days>35°C=9)$^{\uparrow}$
- northern Wet Tropics (avg = 27.0°C, max = 41.0°C, days>35≤40°C = 56, days>40°C = 2)$^{\updownarrow}$
- Mackay–Whitsunday (avg = 25.5°C, max = 40.2°C, days>35≤40°C =64, days$_{>40°C}$ =1)$^{\updownarrow}$

- Fitzroy (avg = 24.3°C, max = 41.1°C, days$_{>35\leq40°C}$ =54, days$_{>40°C}$ =3)↨
- Burdekin (avg = 26.4°C, max = 39.4°C, days$_{>35°C}$ =44)↨
- southern Wet Tropics (avg = 26.8°C, max = 35.1°C, days$_{>35°C}$ =1)↨.

Daily within-canopy seawater temperatures in the habitats in 2018–19 from warmest to coolest difference (* = 0.2°C cooler) relative to the long-term were:

- reef intertidal habitat (avg = 26.3°C, max = 40.2°C)
- estuarine sites (avg = 23.9°C, max = 39.2°C)
- coastal intertidal sites (avg = 25.9°C, max = 41.1°C)*
- reef subtidal sites (avg = 26.2°C, max = 33.0°C)*

The hottest seawater temperature recorded at inshore seagrass sites along the Reef during 2018–19 was 41.1°C in the Fitzroy region, and all regions except the Burdekin and Burnett–Mary, had at least one day above 40°C (Figure 11). Extreme temperature days (>40°C) can cause photoinhibition but when occurring at such low frequency, they were unlikely to cause burning or mortality. Subtidal temperatures remained below 35°C in 2018–19 (Figure 12), however were above the long-term average in the southern Wet Tropics and Burdekin regions.

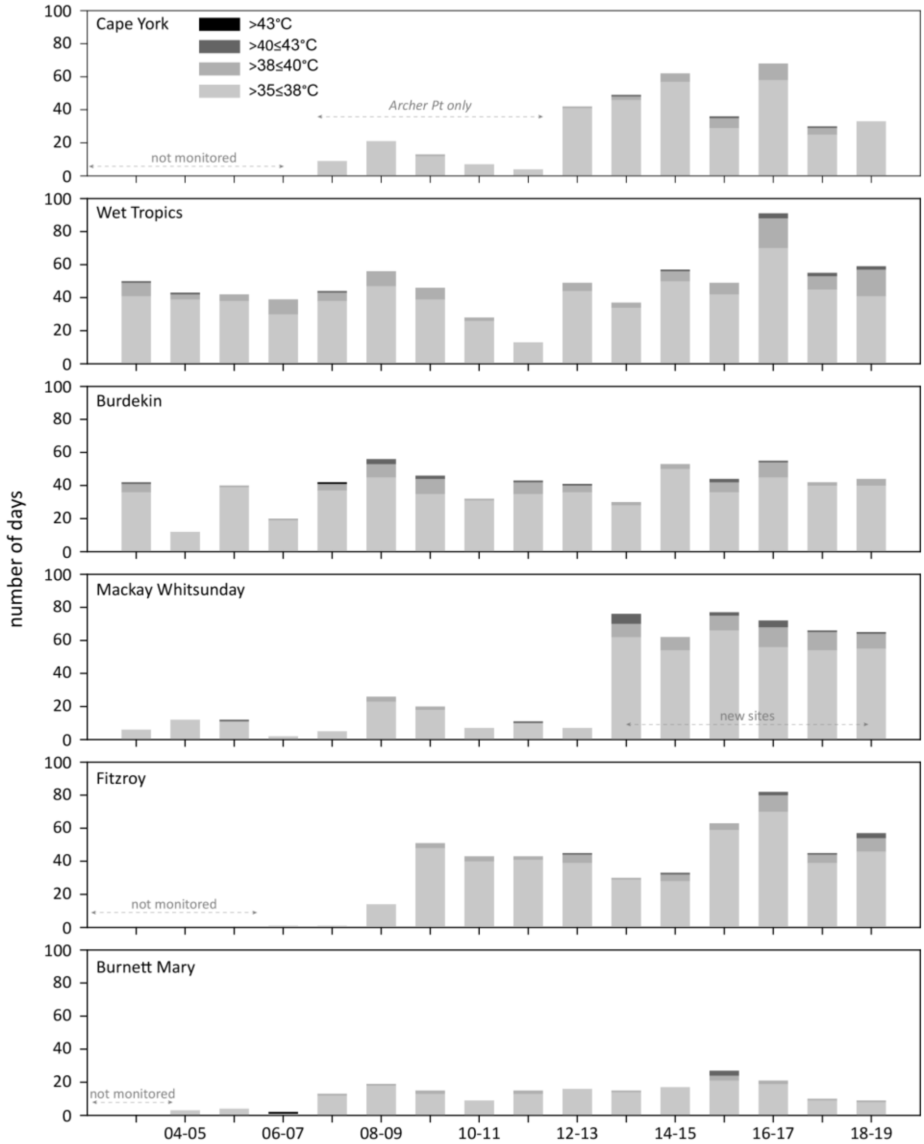


Figure 11. *Number of days when inshore intertidal sea temperature exceeded 35°C, 38°C, 40°C and 43°C in each monitoring period in each NRM region. Thresholds adapted from Campbell et al. 2006; Collier et al. 2012a.*

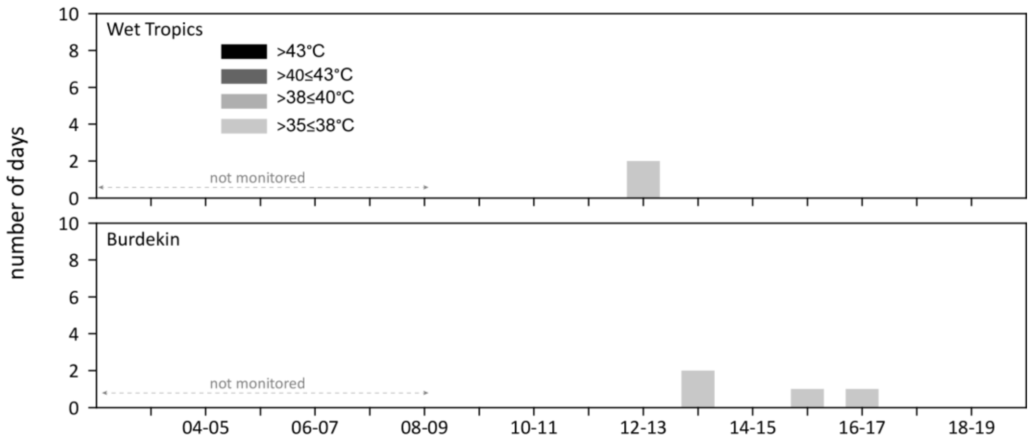


Figure 12. *Number of days when inshore subtidal sea temperature exceeded 35°C, 38°C, 40°C and 43°C in each monitoring period in each NRM region. Thresholds adapted from Campbell et al. 2006; Collier et al. 2012a*.

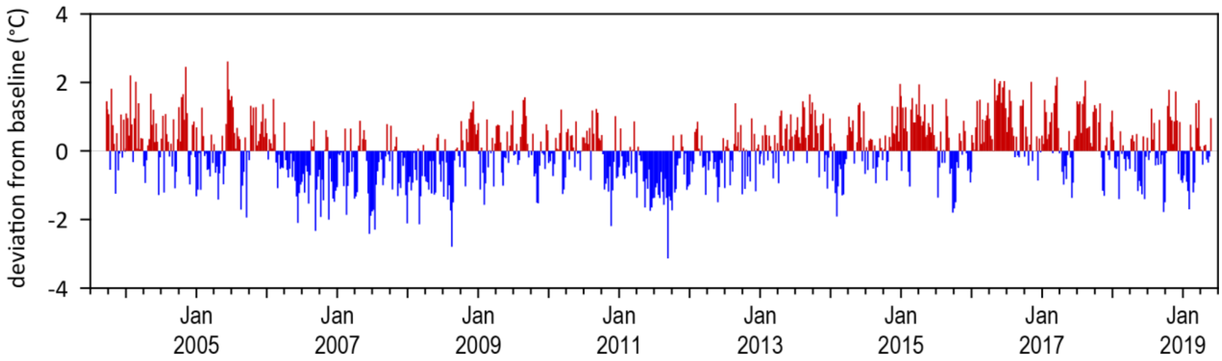


Figure 13. *Inshore intertidal sea temperature deviations from baseline for Reef seagrass habitats from 2003 to 2019. Data presented are deviations from 13-year mean weekly temperature records (based on records from September 2003 to June 2018). Weeks above the long-term average are represented as red bars and the magnitude of their deviation from the mean represented by the length of the bars, blue bars represent weeks with temperatures lower than the average and are plotted as negative deviations.*

3.7 Seagrass meadow sediments

Coastal subtidal and estuarine seagrass habitats across the Reef had a greater proportion of fine sediments (i.e. mud) than other habitats (Table 10). Sediments at intertidal coastal habitats were predominately medium and fine sands, while reef habitats (intertidal and subtidal) were dominated by medium sands (Table 10).

Table 10. *Long-term average (±SE) sediment composition for each seagrass habitat (pooled across regions and time) monitoring within the Reef (1999–2018). *only 4 years of data.*

Habitat	Mud	Fine sand	Sand	Coarse sand	Gravel
estuarine intertidal	49.2 ±2.1	19.7 ±2.0	28.4 ±1.8	0.2 ±0.5	2.6 ±1.1
coastal intertidal	28.4 ±2.1	32.3 ±2.4	34.3 ±2.5	0.3 ±0.5	4.7 ±1.2
coastal subtidal*	50.6 ±1.8	12.7 ±0.5	19.4 ±2.2	12.6 ±1.3	4.7 ±0.0
reef intertidal	5.0 ±1.2	7.1 ±1.7	48.6 ±2.8	17.3 ±1.8	21.9 ±2.3
reef subtidal	7.2 ±0.4	12.8 ±1.1	64.8 ±6.9	1.9 ±0.7	13.3 ±6.9

Since monitoring was established, the composition of sediments has fluctuated at all habitats, with the proportion of mud declining below the long-term average at estuary and coastal habitats immediately following periods of physical disturbance from storms (e.g. cyclones in 2006 and 2011). Conversely, the proportion of mud increased above the long-term average at reef (intertidal and subtidal) habitats during periods of extreme climatic events (e.g. cyclones and/or flood events). It should be noted that the increase in proportion of mud in reef subtidal habitats may be a sampling artefact, due primarily to the increased number of sites monitored post 2014-15. Finer-textured sediments (i.e. mud) tend to have higher fertility, allowing rhizome elongation, and greater levels of anoxia. Although anaerobic conditions may stimulate germination in some species, the elevated sulfide levels generally inhibit leaf biomass production in more mature plants. Only seagrass species adapted for growth in anaerobic mud sediments (e.g. *Zostera*) are able to persist if sufficient light for photosynthesis is available. During the 2018–19 monitoring period there were small fluctuations in the contribution of mud sediments to sediment type relative to the previous year (Figure 14).

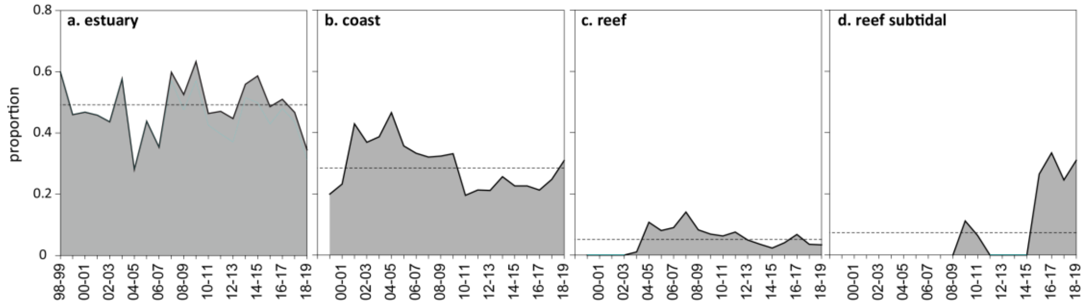


Figure 14. *Proportion of sediment composed of mud (grain size <63µm) at Reef seagrass monitoring habitats from 1999–2019.*

4 Seagrass condition and trend

The following results section provides detail on the overall seagrass responses for the 2018–19 monitoring period, in context of longer-term trends. It is structured as a Reef-wide summary: overall condition and trend for each habitat type is presented separately, including:

- a summary of the key findings from the overall section including a summary of the report card score
- seagrass abundance and extent
- seagrass species composition based on life history traits
- seagrass reproductive effort and seed banks
- seagrass leaf tissue content (C:N, N:P and C:P ratios)
- epiphyte and macroalgae abundance
- linkage back to broad-scale environmental pressures.

Detailed results for each region are presented in the next section. Supporting data identified as important in understanding any long-term trends is detailed within Appendices 3 and 4.

4.1 Reef-wide seagrass condition and trend

Inshore seagrass meadows across the Great Barrier Reef (the Reef) continued to decline in overall condition in 2018–19, further overturning some of the recovery experienced during a period from mid 2010 to mid-2017. The condition grade for inshore seagrass meadows has remained **poor** (Figure 15).

In summary, the decline was primarily due to a reduction in reproductive effort as opposed to abundance (which remained unchanged) or tissue nutrients:

- Seagrass abundance remained unchanged from 2017-18 to 2018–19, with declines experienced over the previous two periods abating. Seagrass abundance (per cent cover) at meadows monitored in the MMP declined from 2005–2006 until 2012–2013, caused by multiple years of above-average rainfall, and resultant discharges of poor quality water, followed by extreme weather events, after which abundances increased (Figure 15, Figure 17b). Based on the average score against the seagrass guidelines (determined at the site level), the abundance of inshore seagrass in the Reef over the 2018–19 period remained in a moderate grade (Figure 15).
- The 2018–19 year was the fifth consecutive year of declining reproductive effort (Figure 15). Reef-wide reproductive effort in 2018–19 remained very poor (Figure 15). Low reproductive effort will hinder replenishment of the depauperate seed banks, and seed banks are therefore likely to remain low in coming years. Most meadows can be considered vulnerable to further disturbances because of their limited capacity to recover from seed (i.e. low resilience).
- The regression in tissue nutrients follows an improving trend since 2010–2011 (Figure 15). The seagrass leaf tissue nutrient indicator (C:N ratio) decreased in late 2017 from the previous year, but remained unchanged in 2018-19 in a poor state for the twelfth consecutive year (Figure 15). This indicates an elevation in the availability of nitrogen at some locations, relative to the rate at which the leaves are growing and incorporating carbon. In most locations, $\delta^{15}N$ values suggest diverse sources of nitrogen affecting nitrogen availability.

Trends in seagrass abundance and tissue nutrients demonstrate that until 2016–2017, the system was on a recovering trajectory. However, since 2017–18, declines in abundance, tissue nutrients and continued very low reproductive effort throughout most of the Reef, may signal that inshore seagrass resilience has decreased and recovery processes may be further hampered following future disturbances.

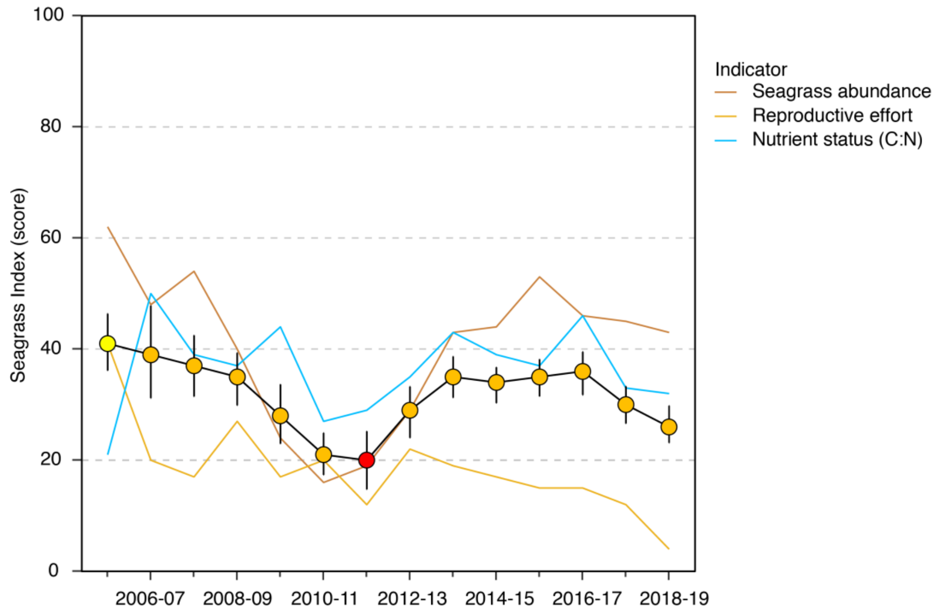


Figure 15. *Reef-wide seagrass condition index (±SE) with contributing indicator scores over the life of the MMP. The index is derived from the aggregate of metric scores for indicators of seagrass community health. Index scores scaled from 0–100 and graded: ● = very good (81-100), ● = good (61 - 80), ● = moderate (41 - 60), ● = poor (21 - 40), ● = very poor (0 - 20). NB: Scores are unitless.*

4.2 Trends in seagrass condition indicators between regions

The Reef-wide score for seagrass is derived from the average of seagrass indicator scores in each of six Regions, weighted by seagrass area. In 2018–19 all but one NRM region (Wet Tropics) declined in seagrass condition (Figure 16), although trends in indicators between the six Regions are not uniform:

- The seagrass abundance score was poor in the 2018–19 monitoring period in all regions except the Burdekin, which remained moderate (Figure 16). There were increases in the abundance score compared to the previous year in the Wet Tropics, Fitzroy and Burnett–Mary NRM regions, but they remained in the poor category. Furthermore, the score declined from moderate in the Cape York region.
- Reproductive scores were moderate in the Wet Tropics (improving from very poor due primarily to Yule Point) in 2018–19, and very poor in the other regions (Figure 16). Reproductive effort declined in the Burdekin and Mackay–Whitsunday and increased in the Wet Tropics, but was relatively stable in all other regions (Figure 16).
- Seagrass nutrient status scores (using only C:N) reduced in all regions except the Wet Tropics, furthermore the score was poor in all regions except for the Burdekin which was moderate, albeit only just (Figure 16). The C:N score was the lowest since monitoring began in the Fitzroy and Burnett–Mary regions, but this has been influenced by changes in sites in both regions.

Inshore seagrass condition scores across the Regions reflect a system that is being impacted by heatwaves, cyclones, and elevated discharge from rivers. Regional differences in condition and indicator scores appear due to the legacy of significant environmental conditions in 2016–2017 (e.g. cyclone Debbie in Mackay–Whitsunday, above-average riverine discharge throughout the southern and central Reef, and a marine heatwave in the northern and central Reef) and/or less favourable environmental conditions in 2018–19.

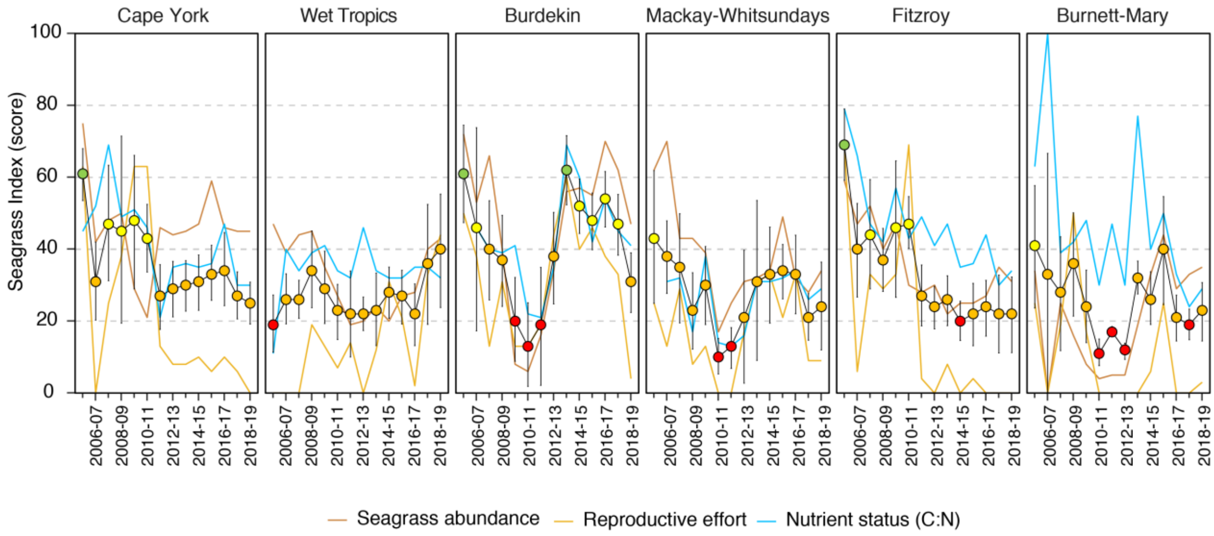


Figure 16. *Seagrass condition index (±SE) with contributing indicator scores for each NRM region over the life of the MMP. The index is derived from the aggregate of metric scores for indicators of seagrass community health. Values are indexed scores scaled from 0–100 and graded: ● = very good (81-100), ● = good (61 - 80), ● = moderate (41 - 60), ● = poor (21 - 40), ● = very poor (0 - 20). NB: Scores are unitless. Scores reflect amendments outlined in Section 2.5.5*

The long-term trends in the seagrass condition index, and the raw data for each of the indicators are shown in Figure 17. Generalised additive models are presented for per cent cover and tissue nutrients to show long-term trends in these indicators. These models could not be constructed on the reproductive data due to the large number of zeroes. Instead, reproductive effort is displayed as mean and standard errors, which highlights the large seasonal variability in reproductive effort.

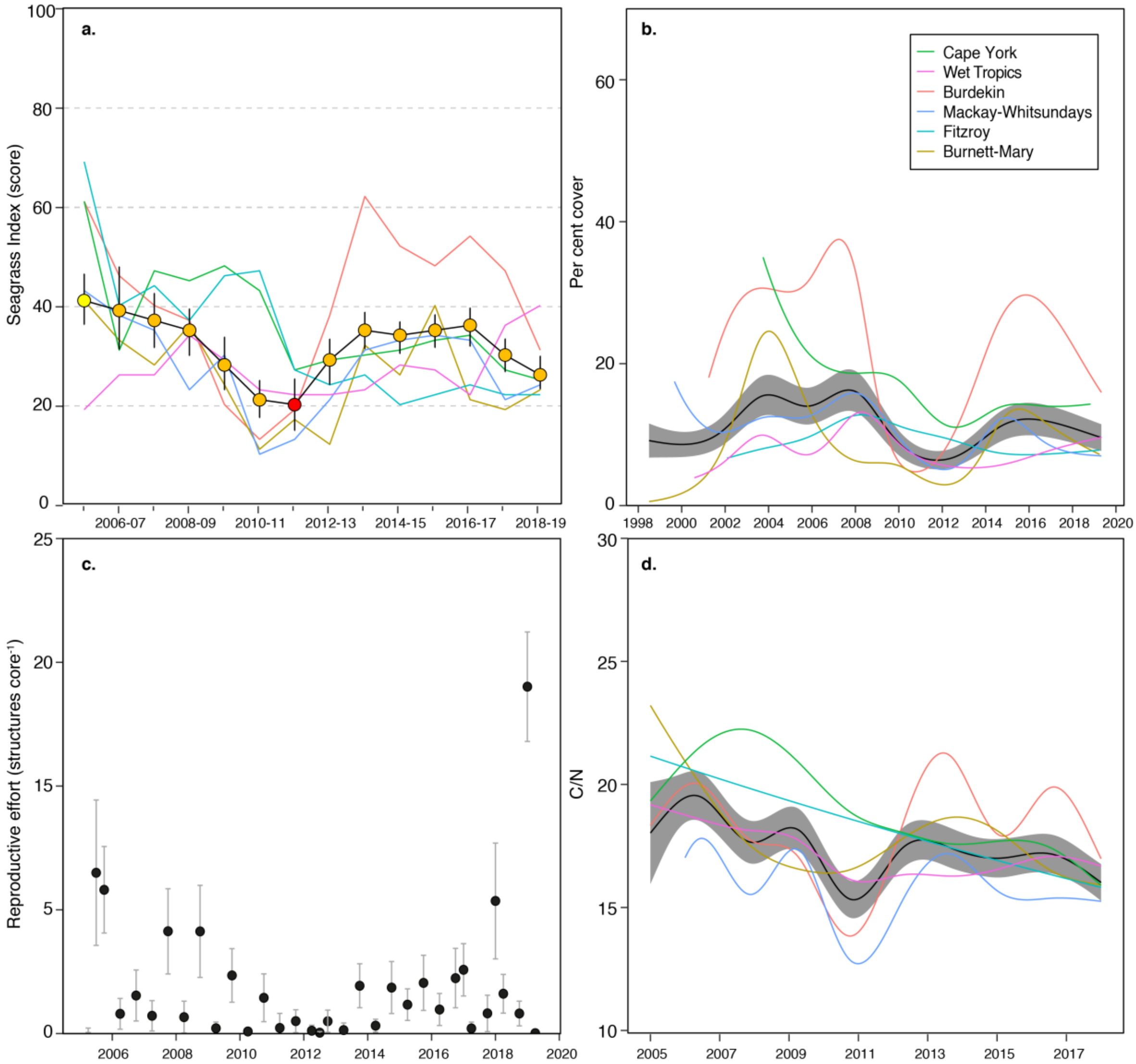


Figure 17. *Trends in the seagrass condition index and indicators used to calculate the index including: a. Reef-wide seagrass index (circles) and regional trends (lines); b. trends in seagrass abundance (per cent cover) represented by a GAM plot as dark lines with shaded areas defining 95% confidence intervals of those trends (Reef), and coloured lines representing NRM trends; c. reproductive structures (gam is not possible due to high count of zeroes); and d. tissue nutrient content represented by a GAM plot as dark lines with shaded areas defining 95% confidence intervals of those trends (Reef), and coloured lines representing NRM trends.*

4.3 Trends in seagrass condition indicators by habitat type

4.3.1 Seagrass abundance, composition and extent

Seagrass abundance scores have fluctuated since monitoring was established. An examination of long-term abundances across the Reef indicates:

- no significant trends at 74% of long-term monitoring sites, however 5% of sites significantly increased in abundance and 13% decreased (Appendix 4, Table 20)
- the rate of change in abundance was higher at sites increasing (0.9 ±0.5%, sampling event^{-1}) than decreasing (-0.3 ±0.1% sampling event^{-1}) (Appendix 4, Table 20)

- the most variable Reef seagrass habitat in abundance (since 2005) was intertidal estuary (CV=73.7%), followed by intertidal reef (CV=52.1%), intertidal coastal (CV=45.5%) and lastly subtidal habitats (reef CV=44.2% and coastal CV=31.2%).

Since 1999, the median percentage cover values for the Reef were mostly below 25% cover, and depending on habitat, the 75th percentile occasionally extended beyond 50% cover (Figure 18). These long-term percentage cover values were similar to the Reef historical baselines, where surveys from Cape York to Hervey Bay (between November 1984 and November 1988) reported most (three-quarters) of the per cent cover values fell below 50% (Lee Long *et al.* 1993). The findings negate the assumption that seagrass meadows of the Reef should have abundances closer to 100% before they are categorised as good.

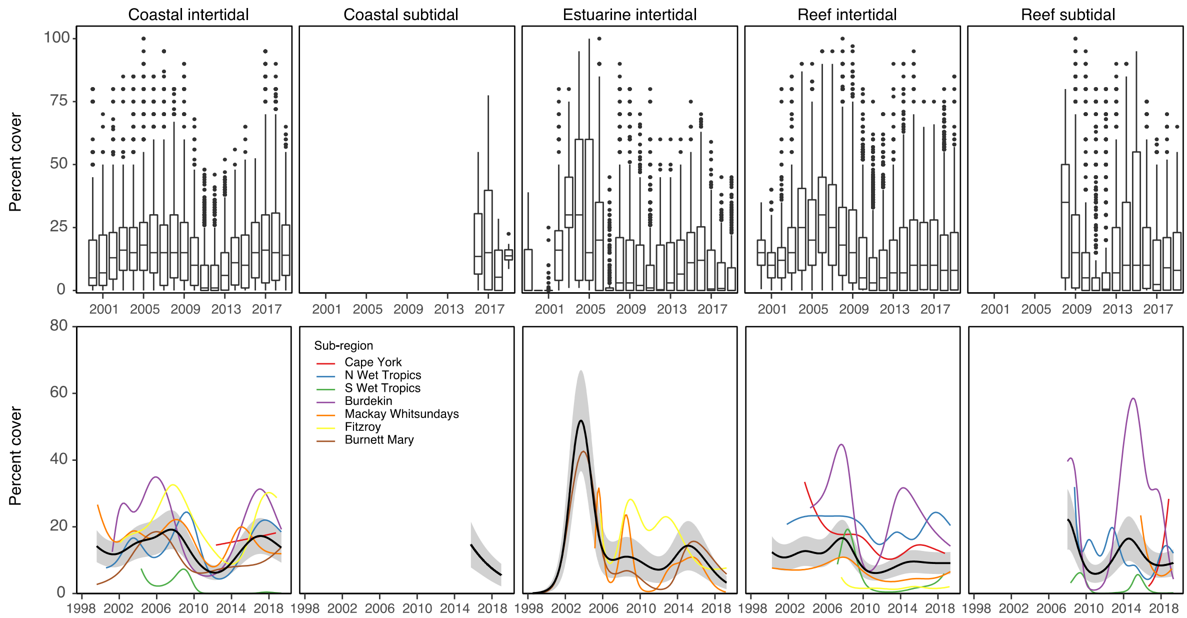


Figure 18. *Seagrass per cent cover measures per quadrat from meadows monitored from June 1999 to May 2019 (sites and habitats pooled). The box represents the interquartile range of values, where the boundary of the box closest to zero indicates the 25th percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75th percentile. Whiskers (error bars) above and below the box indicate the 90th and 10th percentiles, and the dots represent outlying points. GAM plots (bottom), also showing trends for each NRM.*

In 2018–19, coastal sites had the highest average abundance of the habitat types (Figure 18). Over the past decade, the patterns of seagrass abundance in each Reef habitat have been similar between coastal and reef sites; gradually increasing from 2001 to 2008 (with a mild depression in 2006-07 as a consequence of cyclone Larry), then declining from 2009 to 2011 due to above average rainfall and river discharge (Figure 17). The extreme weather events of early 2011 (e.g., cyclone Yasi) resulted in further substantial decline in inshore seagrass meadows throughout much of the Reef.

Estuarine habitats, which are monitored only in the southern Reef, reached record per cent cover in 2002 to 2003, but have remained low since 2005–06. Trends have fluctuated at a site level in estuary habitats, most often at smaller localised scales where there have been some acute event related changes (McKenzie *et al.* 2012).

Post 2011, seagrasses have progressively recovered, although by 2016–2017 still remained below the 2008 levels, except in coastal sites which have recovered (Figure 17).

In 2018–19, Reef-wide relative meadow extent was similar to the previous year, however these remain lower than the baseline (2005), 2014 and 2015 (Figure 19). Since the MMP was established in 2005, meadow extent across inshore monitoring sites declined in early 2011, recovering within 3–4 years (Figure 19). Similar to seagrass abundance, this decline in relative extent was a consequence of extreme weather and associated flooding. Since 2014, the meadows monitored across the Reef have varied in extent within and between years. The changes in extent over the last three years appear a consequence of severe weather events (e.g. cyclones) and regional climate (frequency of strong wind days).

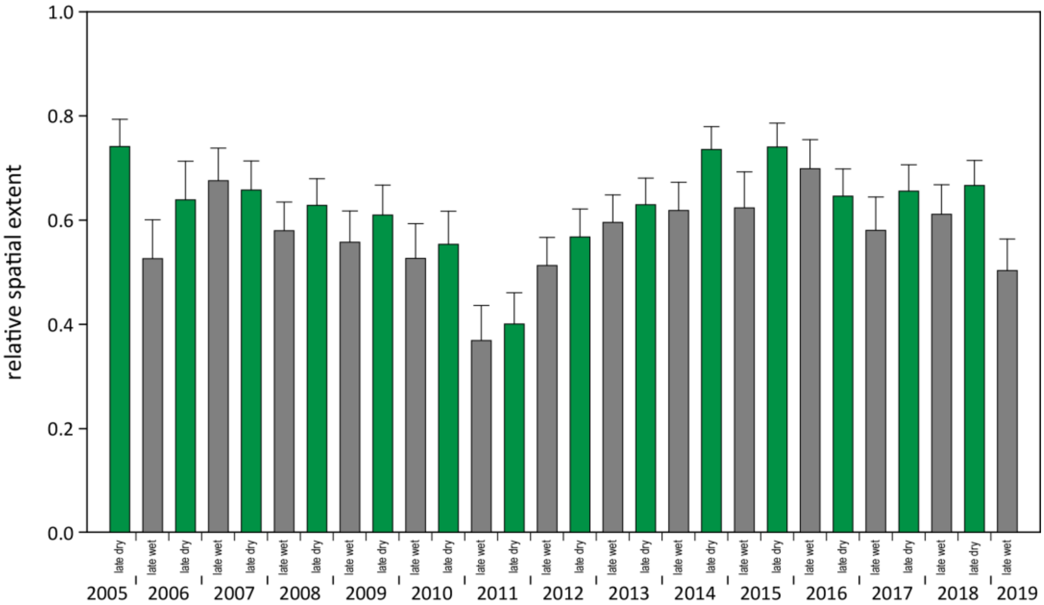


Figure 19. *Average relative spatial extent of seagrass distribution at monitoring sites across inshore Reef (locations, habitats and NRM regions pooled).*

After the extreme weather events in 2009 to 2011 that caused widespread declines in seagrass extent (Figure 19) and abundance, there was increasing proliferation of species displaying colonising traits, such as *Halophila ovalis,* at coast and reef sites (Figure 20). Over the 2018–19 monitoring period, the proportion of species displaying colonising traits remained around or lower than the Reef-wide average for each habitat type in coastal and estuarine habitats in favour of species displaying opportunistic or persistent traits (*sensu* Kilminster *et al.* 2015). The displacement of colonising species is a natural part of the meadow progression expected during the recovery of seagrass meadows. This is a positive sign of recovery for these habitats/meadows.

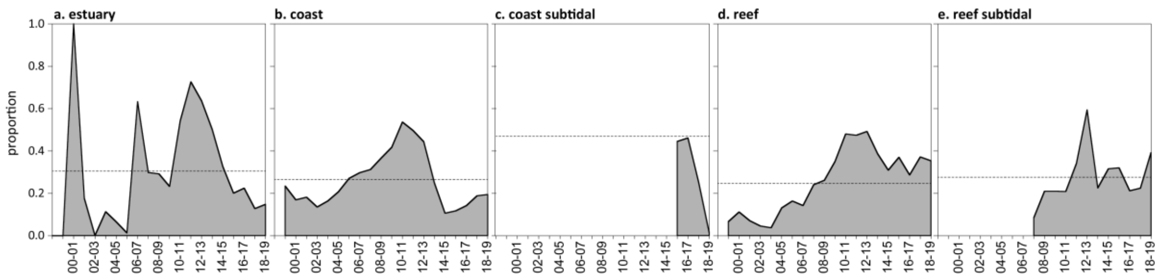


Figure 20. *Proportion of total seagrass abundance composed of species displaying colonising traits (e.g.* Halophila ovalis*) in: a) estuary intertidal, b) coastal intertidal, c) coast subtidal, d) reef intertidal and e) reef subtidal habitats (sites pooled) for the Reef (regions pooled) each monitoring period. Dashed line illustrates Reef average proportion of colonising species in each habitat type.*

4.3.2 Seagrass reproductive status

Seagrass reproductive effort remained very low in reef intertidal and subtidal habitats and in estuarine habitats. By contrast, there were increases in reproductive effort in coastal habitat during the dry season, albeit with a high degree of variability depending on site. This resulted in the reproductive effort score remaining very poor in the Reef.

Reproductive effort had gradually been increasing at estuary, coastal and reef subtidal habitats since 2011, however, this year it decreased significantly in estuaries and remained low in subtidal reefs. This occurred in conjunction with declining seagrass per cent cover in estuarine and reef habitats. Reproductive effort at reef intertidal habitats declined in 2014 and has remained very low since. Contrarily, reproductive effort in coastal habitats reached historically high levels in 2018–19 due to a record number of reproductive structures in the northern Wet Tropics, Burdekin and Mackay–Whitsunday regions. Despite these decreases in reproductive effort, seed banks continued to increase at subtidal reef habitats in 2018–19, a legacy of higher reproductive effort in the previous year. Coastal seed banks have continued to increase and remain high, but at estuary and intertidal reef habitats remain small or near absent.

Since the implementation of the MMP, the maximum reproductive effort and the inter-annual variability in reproductive effort has differed between habitats, and varied within and between years. Reef habitats, both intertidal and subtidal reef sites, had the lowest reproductive effort and smallest seed banks of all habitats (Figure 21, Figure 22).

Reproductive effort has been historically higher in estuary and coastal habitats but gradually decreased from 2006 to 2011 (in concert with decreasing seagrass cover) and has been increasing since. This increase continued in 2018–19 at coastal habitats, however, reproductive effort decreased significantly in estuaries. The historically high reproductive effort in coastal habitats is due to a record number of reproductive structures in the northern Wet Tropics (Yule Point), Burdekin (Bushland Beach and Jerona) and Mackay–Whitsunday (Midge Point). The decline in estuary habitats was most likely due to the declines in seagrass per cent cover. By contrast, reproductive effort at reef intertidal habitats declined in 2014 and has remained very low since.

Seed banks across the inshore Reef meadows were higher in late dry and greater in coastal than reef or estuarine habitats over the long-term (>10 years) (Figure 22). Coastal seed banks declined between 2008 and 2011, and have subsequently increased, but remain below the 2007–2008 levels. Seed banks continued to increase at subtidal reef habitats in 2018–19, but remain low or near absent at estuary and intertidal reef habitats, respectively.

The small seed banks could have been caused by reduced reproductive success (failure to form seeds) or loss of seed bank (germination or grazing). The low reproductive effort and low density of seeds in the seed bank in intertidal reef habitats in all regions (except Burnett–Mary, where no reef sites are monitored), indicates a low seed production rate and vulnerability of these habitats to future disturbances, as recovery may be hampered.

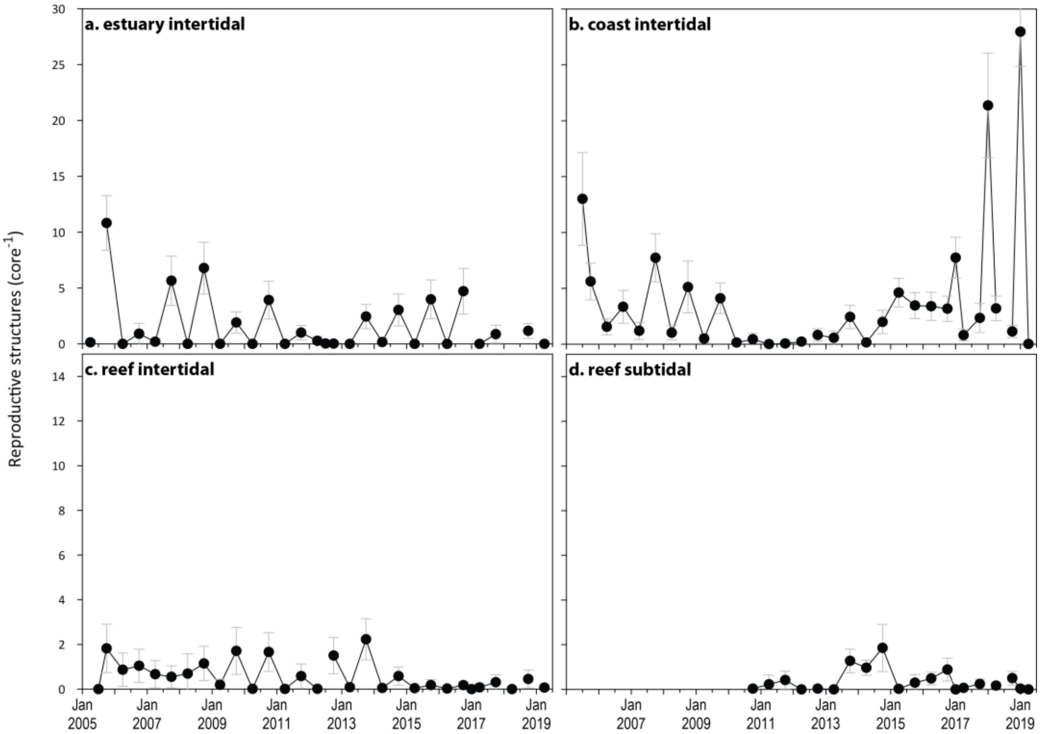


Figure 21. *Seagrass reproductive effort (number of reproductive structures produced by all seagrass species) during the late dry of each monitoring period for a) estuary intertidal; b) coast intertidal; c) reef intertidal; d) reef subtidal.*

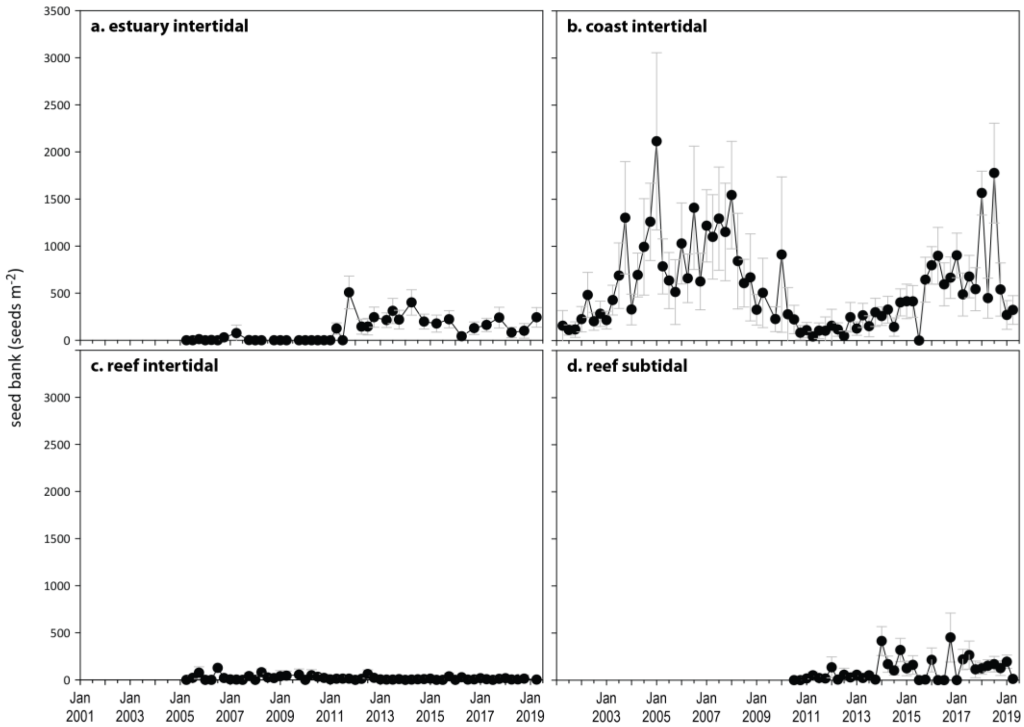


Figure 22. *Average seeds banks (seeds per square metre of sediment surface, all sites and species pooled) in Reef seagrass habitats: a) estuary intertidal; b) coast intertidal; c) reef intertidal; d) reef subtidal.*

4.3.3 Seagrass leaf tissue nutrients

In 2018–19, the ratio of carbon (C) to nitrogen (N) was below the guideline value of 20 in all habitats except reef subtidal habitat. The C:N ratio is used as an indicator of water quality and seagrass condition because elevated carbon (and elevated C:N) suggests high light availability, while elevated N (lower C:N), indicates elevated nitrogen supply rates relative to growth requirements. Therefore, in all habitats other than reef subtidal, there was an oversupply of N relative to growth requirements.

In 2018–19, C:N ratio of seagrass leaves decreased at approximately a third of sites from the previous period, but this was not significant due to variation in this trend among regions and sites, and the number of sites remaining above the threshold of 20 was the lowest in 5 years. The lowest C:N values on average continue at Hamilton Island (10.1), Yule Point (12.8), and Shelburne Bay (12.9).

Tissue nutrients are measured in the late dry (~October 2018) of the reporting period, and are therefore related to the previous water quality reporting year (1st October 2017–31st September 2018). Despite river discharge around the long-term median for most rivers, secchi depth across the inshore Reef declined in 2017–18 and did not meet guidelines, but dissolved and particulate nitrogen were relatively stable (Gruber et al 2020). Site-specific changes in C:N are likely related to local conditions, in particular localised variations in benthic light.

Seagrasses are passive indicators of $\delta^{15}N$ enrichment, as they integrate the signature of their environment over time throughout their growth cycle. $\delta^{15}N$ values can indicate the source of nitrogen. Very low (~0‰) or negative values of $\delta^{15}N$ can indicate nitrogen sourced from nitrogen fixation (Peterson and Fry 1987; Owens 1988); which can supply one third to one half of seagrass demand (O'Donohue *et al.* 1991). Low to moderate values (i.e. $\delta^{15}N$ >0 - ~3‰) indicate internal sources from remineralisation (Peterson and Fry 1987; Owens 1988) and N fertilizer, produced by industrial fixation of atmospheric nitrogen (Udy and Dennison 1997a). Higher values (>3‰) can indicate septic and aquaculture sources (Jones *et al.* 2001) and further biological fractionation results in sewage nitrogen having a $\delta^{15}N$ signature greater than 9 or ~10‰ (Lajtha and Marshall 1994; Udy and Dennison 1997b; Dennison and Abal 1999; Costanzo *et al.* 2001; Jones *et al.* 2018). In general, $\delta^{15}N$ in Reef seagrass tissues are variable but low (Figure 23), suggesting multiple sources of nitrogen. There is currently no indication or concern that anthropogenic sources are strongly influencing seagrass N supply.

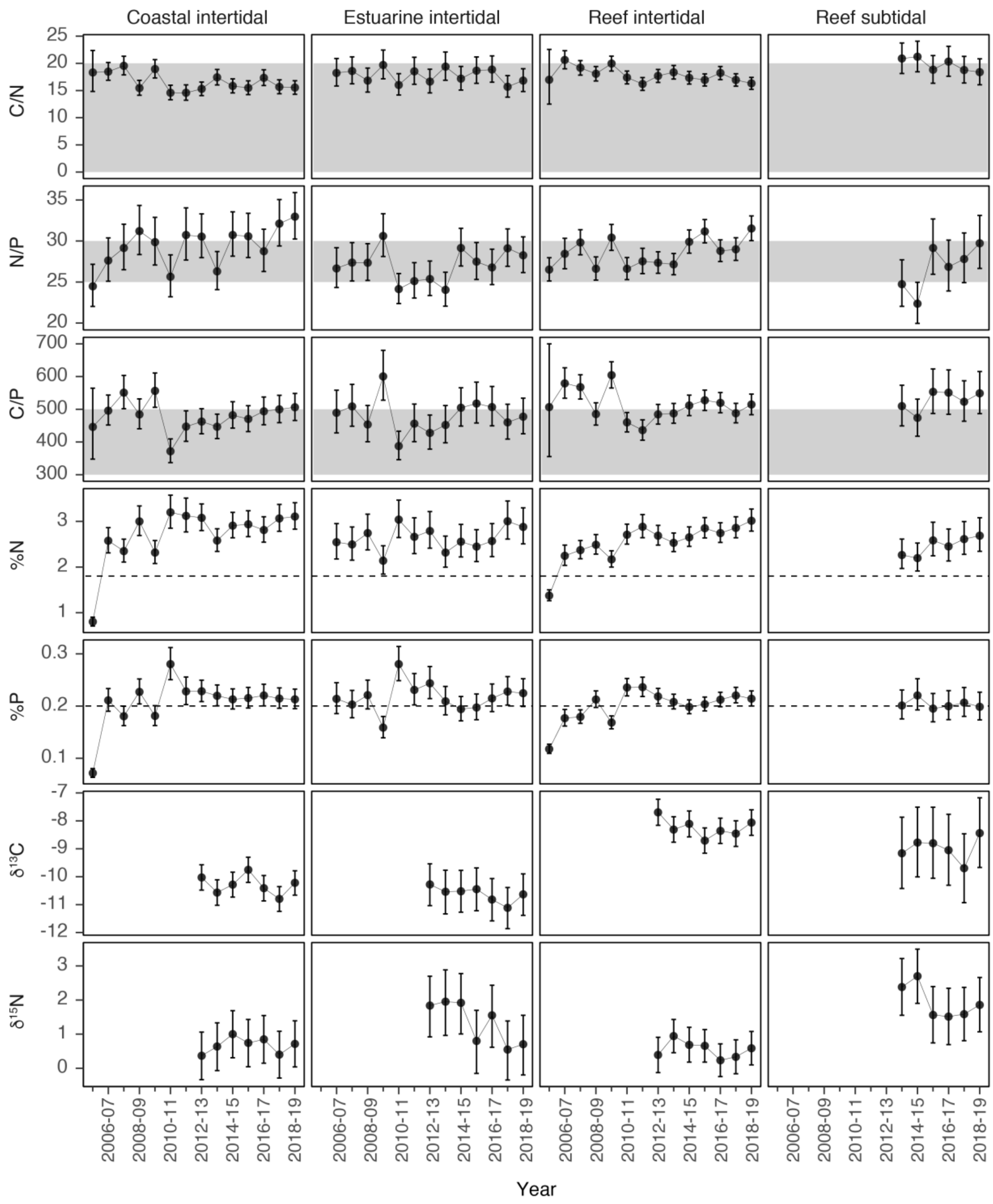


Figure 23. *Seagrass leaf tissue nutrient elemental ratios (C:N:P) and concentrations (%N,%P, $\delta^{13}C$ and $\delta^{15}N$) for each seagrass habitat each year (± SE) (foundation species pooled). Horizontal shaded bands or dashed lines represents the accepted seagrass guideline values, where: C:N ratios within the band may indicate reduced light availability and/or N enrichment; N:P ratios above the band indicate P limitation, below indicate N limitation and within indicates replete, and; C:P ratios within the band may indicate nutrient rich habitats (large P pool). Dashed lines in %N and %P indicate global median values of 1.8% and 0.2% for tissue nitrogen and phosphorus, respectively (Duarte 1990).*

4.3.4 Epiphytes and macroalgae

Epiphyte cover on seagrass leaves during 2018–19 was below or at the Reef-wide long-term average in all habitats except reef subtidal, where it has remained above average over the past four years due to high epiphyte cover at all sites but in the wet season of 2018–19 it dropped to below the average (Figure 24).

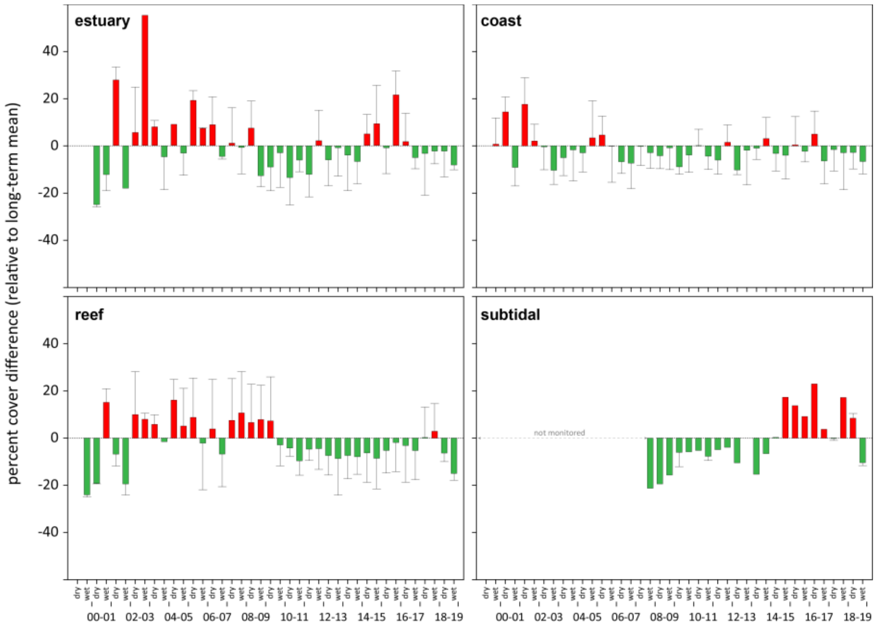


Figure 24. *Epiphyte abundance (per cent cover) relative to the long-term average (the zero axis) for each Reef seagrass habitat (sites pooled, ± SE). Reef long-term average; estuarine = 25.1±5.6% coastal=17.8±3.7%, reef = 22.8±4.2%, subtidal= 20.6±3.1%.*

Macroalgae abundance was generally low and stable at Reef seagrass habitats, with little change this year (Figure 25). Macroalgae abundance has been higher than long-term average at some reef intertidal habitats over the last 2 years, but this has been highly variable.

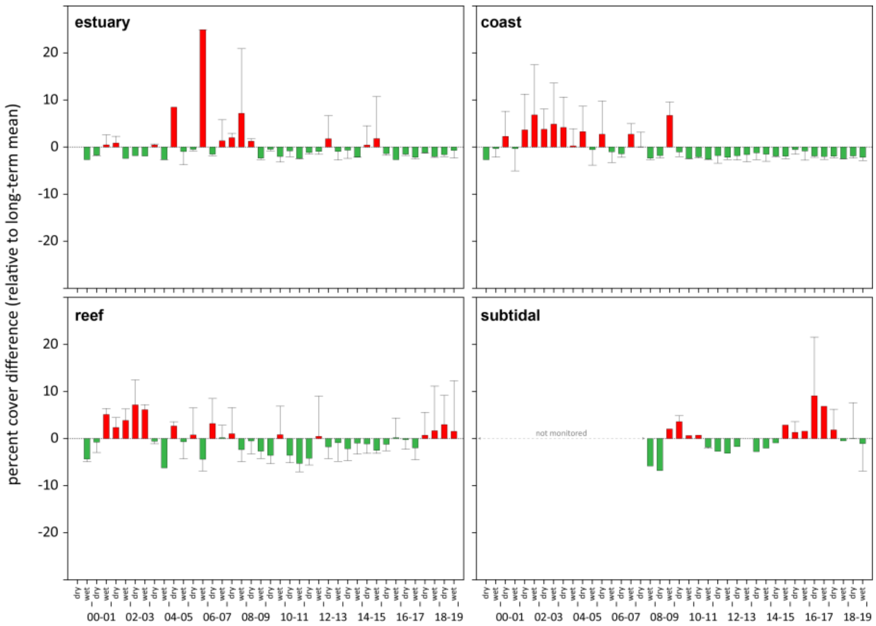


Figure 25. *Macroalgae abundance (per cent cover) relative to the long-term average for each inshore Reef seagrass habitat. (sites pooled, ± SE). Reef long-term average; estuarine = 2.3±1.0%, coastal=2.5±1.2%, reef = 6.9±1.9%, subtidal = 6.6±2.0%.*

5 Regional Reports

This section presents detailed results on the condition and trend of indicators within Regions, and relates the results to local environmental factors including:

- annual daytime tidal exposure at each monitoring site
- daily light each monitoring location
- sediment grain size composition at each monitoring site
- tables detailing statistical analysis.

5.1 Cape York

5.1.1 2018–19 Summary

Seagrass meadows across the Cape York NRM region in 2018–19 remained similar to 2017-18 overall condition, with slight improvements in abundance being offset by slight declines in reproductive effort and tissue nutrient condition indicators:

- abundance score was moderate
- tissue nutrient score was poor
- reproductive effort score was very poor.

On average, seagrass abundance marginally increased relative to the previous period. Seagrass abundance (per cent cover) increased at half or remained similar at 40% of sites across all habitats, predominately in coastal and subtidal reef meadows. The only declines occurred in meadows located in the north of the region.

Seagrass leaf tissue nutrient concentrations in 2018–19 corresponded with the higher ‘green’ water exposure, indicating that the availability of nitrogen (N), particularly in coastal habitats, has increased relative to the demand for carbon for growth. However, the N source appears primary natural fixation rather than anthropogenic, and levels are not of concern as they do not appear to have significantly influenced epiphytic and macroalgae abundances.

The capacity for the meadows to recover across the Cape York region is variable between habitats. The large seed banks which persist at intertidal coastal meadows could aid recovery in the short term, if environmental conditions are favourable for germination, but the low reproductive effort may limit replenishment and maintenance of the bank in the future. The lack of seeds in most intertidal reef meadows currently limits recovery, and the decreased reproductive effort may weaken capacity in the future.

Lastly, the region experienced above average elevated within-canopy water temperatures for the seventh consecutive year, which may have exacerbated chronic stress conditions in the intertidal meadows, further impacting growth.

An assessment of long-term trends in other Cape York habitats is affected by changes in the number, onset and duration of monitoring at individual sites. An examination of the long-term trend shows seagrass per cent cover progressively decreased at intertidal reef habitats across Cape York from 2003 to 2012, with relatively little improvement since. Coastal intertidal and subtidal habitats monitored since 2012 and 2015 respectively, generally showed no significant trend. Similarly, meadow extent across the region has been relatively stable since 2012.

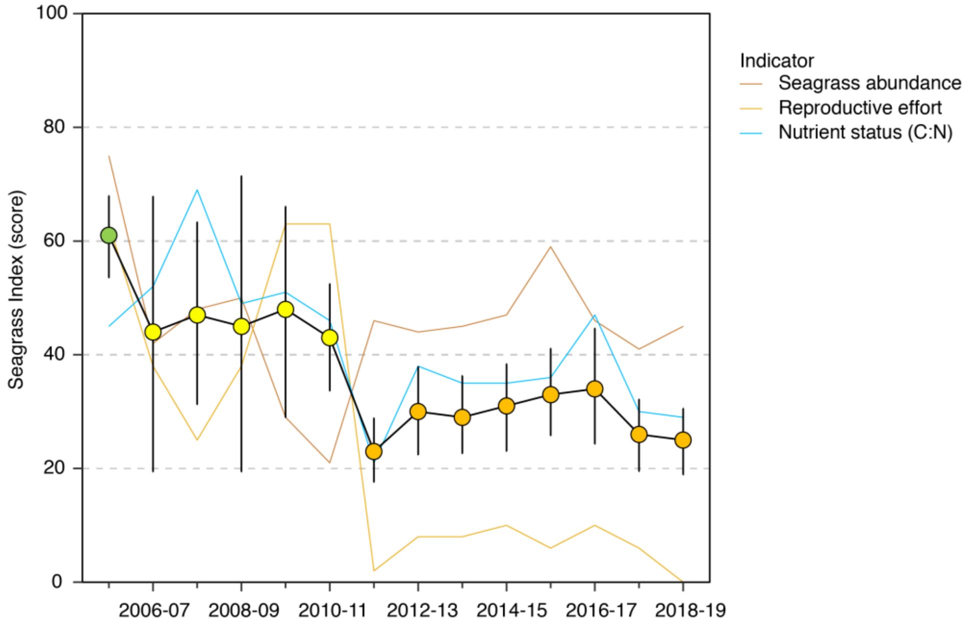


Figure 26. *Seagrass condition index (±SE) with contributing indicator scores for the Cape York NRM region (averaged across habitats and sites). Index scores scaled from 0-100 and graded: ● = very good (81-100), ● = good (61 - 80), ● = moderate (41 - 60), ● = poor (21 - 40), ● = very poor (0 - 20). NB: Scores are unitless.*

5.1.2 Climate and environmental pressures

Multiple large flood events influenced Cape York marine waters during the 2018–19 wet season. Major flood events were associated with cyclone Penny in late December 2018, with extensive sustained rainfall in late January and cyclone Trevor in mid- to late March.

River discharge during the 2018–19 wet season was more than double the long-term average from most basins within the Cape York NRM region, and more than three times the long-term median in the Normanby and Endeavour Rivers (Figure 27). The extent of river influence on the Reef (using model tracers), and the exposure levels and risk from turbid primary ('brown', sediment laden colour classes one to four) and secondary water type ('green', phytoplankton rich water, colour class five) using MODIS satellite products is detailed in Gruber et al (2020). The inshore waters of Cape York had predominantly secondary water type ('green', phytoplankton rich water), and some brown turbid water exposure through the wet season (December-April; Figure 27). Shelburne Bay sites (SR1 and SR2) had the highest exposure to turbid primary water, consistent with previous years. The frequency of exposure to both primary and secondary water ranged from 50% to 100% of wet season weeks at seagrass monitoring sites (Figure 27).

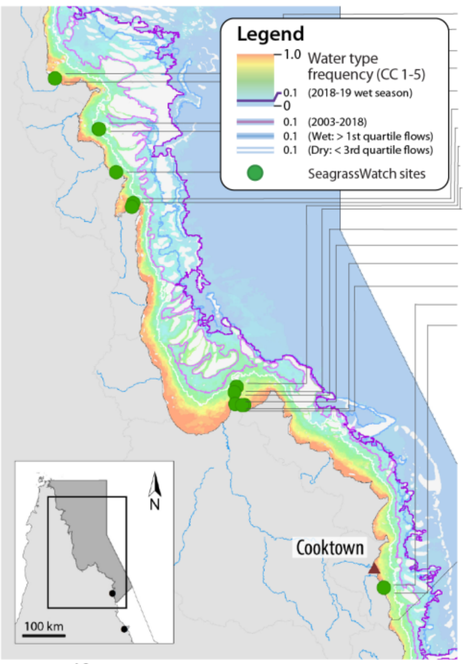
Daily incident light (I_d, mol m^{-2} d^{-1}) reaching the top of the seagrass canopy is generally very high at all Cape York sites (long-term average = 16.5 mol m^{-2} d^{-1}) (Figure 99). However in 2018–19, daily incident light (13.7 mol m^{-2} d^{-1}) was below the long-term average (Figure 27). This was most likely a consequence of the greater duration meadows were exposed to brown and or green water, however the shorter/incomplete logging duration (approximately two thirds of data missing) at reef intertidal sites due to instrument failure may have also contributed.

2018–19 was the seventh consecutive year intertidal within-canopy temperatures were above the long-term average and the second highest average annual temperatures (27.5°C) since 2006 (Figure 27). Maximum within-canopy temperatures exceeded 35°C for a total of 33 days during 2018–19 (Figure 27), with the highest temperature recorded at 37.7°C (ST2, 3pm 17Feb19).

Daily tidal exposure (hours water has drained from the meadow) was below the long-term average for the second consecutive year (Figure 27, Figure 91), which may have provided some respite from the elevated temperatures.

In the Cape York NRM region, reef habitats remain dominated by sands and coarser sediments, while coastal habitats contained a greater proportion of mud (Appendix 3, Figure 106, Figure 107).

a.



b.

Wk	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22
SR1	1		1	5		2	2				4	1	5	1	2		1		5	1	1	
SR2	1		1	5		2	2				4	1	5	1	2		1		5	1	1	
FR1	6	6	6			5	5				6	5	6	6	6		5		5	5	5	
FR2	6	6	6			5	5				6	5	6	6	6		5		5	5	5	
YY1	5	6	5		5	5	5			5	5	4	5	5	5		1		5	5	5	
LR1	5	5	5		5	4	5			7	6	5	5	5	5	7	5		4	5	5	
LR2	5	5	5		5	4	5				5	5	5	5	5		3		4	4	4	
ST2	5	6	5	6	5	5	5			6	3	5	5	5	5		5	5	5	5	5	5
ST1	5	6	5	6	5	5	5			6	3	5	5	5	5		5	5	5	5	5	5
FG2	5	6	5	6	5	5	5			6	3	5	5	5	5		4	5	5	5	5	5
FG1	5	6	5	6	5	5	5			6	3	5	5	5	5		4	5	5	5	5	5
BY2	5	5	5	5	4	5	1				4	4	4	4	5		4	4	4	4	4	4
BY1	5	5	5	5	4	5	1				4	4	4	4	5		4	4	4	4	4	4
BY4	5	5	5	5	5	5	5			5	4	5	5	5	5		5	5	5	5	5	5
BY3	5	5	5	5	5	5	5				4	5	5	5	5		5	5	5	5	5	5
AP1	5	6	5	5		5	5				4	4	4	5	5	4	5	4	4	5	5	5
AP2	5	6	5	5		5	5				4	4	4	5	5	4	5	4	4	5	5	5

1 - 4 = turbid primary water (red/brown), 5 = nutrient/chlorophyll-enriched secondary water (green), 6 = some freshwater/CDOM influence tertiary water (blue) , 7 = no plume influence (white), blank = no data, clouds (grey)

c.	Long-term average	2018-19
Rainfall (1961-1990)	9.2 mm d^{-1}	10.7 mm d^{-1}
River discharge (1970-2018)	15,265,122 L yr^{-1}	41,707,172 L yr^{-1}
Turbid water exposure (2006-2018)	83%	87%
Daytime tidal exposure (1999-2018)	61 hr yr^{-1}	44 hr yr^{-1}
Within-canopy temperature-intertidal (2006-2018)	26.8°C (41.6°C)	27.5°C (37.7°C)
Within-canopy light	16.5 mol m^{-2} d^{-1}	13.7 mol m^{-2} d^{-1}

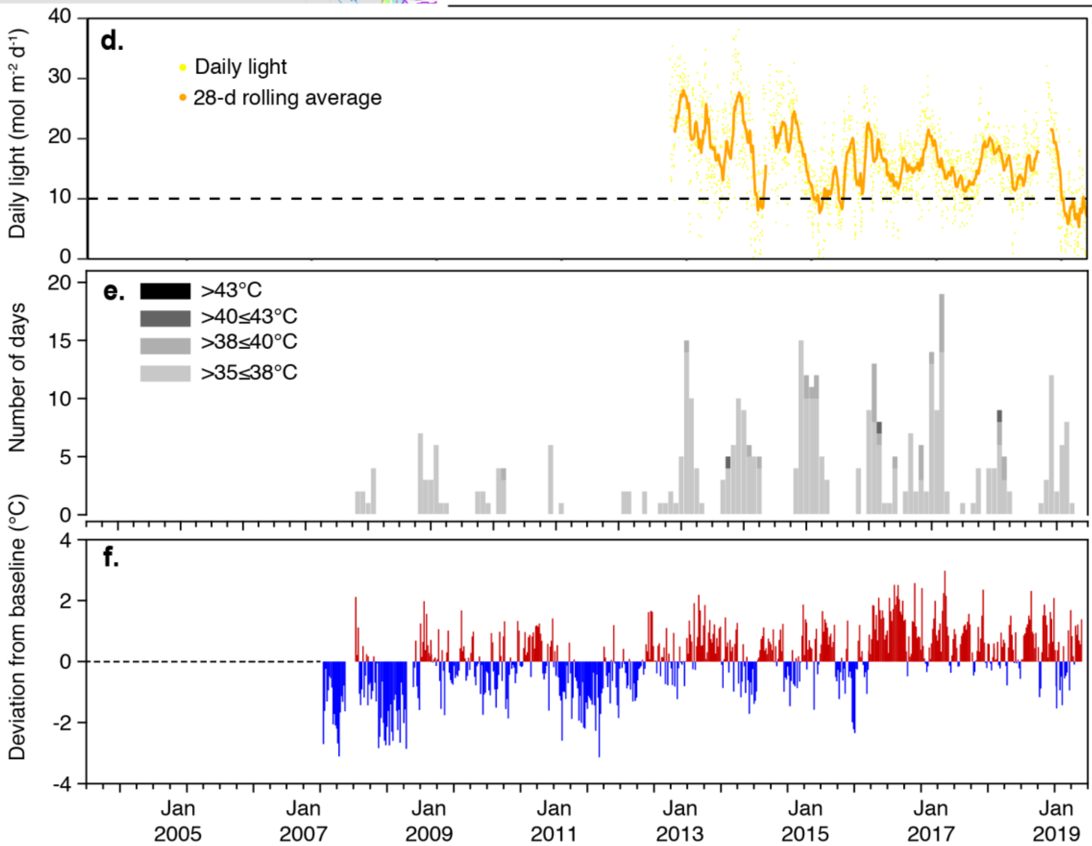


Figure 27. *Environmental pressures in the Cape York region including: a. frequency of exposure to turbid water (colour classes 1-5) (from Gruber et al. 2019b), b. wet season water type at each site; c. average conditions over the long-term and in 2018–19; d. daily light and*

the 28-day rolling mean of daily light for all sites; e. number of day temperature exceeded 35°C, 38°C, 40°C and 43°C, and; f. deviations from 13-year mean weekly temperature records.

5.1.3 Inshore seagrass and habitat condition

There are 17 seagrass monitoring sites in Cape York from 9 locations (Table 11). Four seagrass habitat types were assessed across the region in 2018–19, with data from 10 of the 17 long-term monitoring sites (Table 11, Table 18).

Table 11. *List of data sources of seagrass and environmental condition indicators for each seagrass habitat type in the Cape York NRM region. For site details see Table 2 and Table 3. Open square indicates not measured in 2018-19. † drop camera sampling (QPWS), *Seagrass-Watch.*

Habitat	Site		abundance	composition	extent	reproductive effort	seed banks	leaf tissue nutrients	meadow sediments	epiphytes	macroalgae
coastal intertidal	BY1	Bathurst Bay	■	■	■	■	■	■	■	■	■
	BY2	Bathurst Bay	■	■	■	■	■	■	■	■	■
	SR1	Shelburne Bay	■	■	■	■	■	■	■	■	■
	SR2	Shelburne Bay	■	■	■	■	■	■	■	■	■
coastal subtidal	BY3†	Bathurst Bay	□	□							□
	BY4†	Bathurst Bay	□	□							□
	LR1†	Lloyd Bay	□	□							□
	LR2†	Lloyd Bay	□	□							□
reef intertidal	AP1	Archer Point	□	□							□
	AP2	Archer Point	□	□							□
	FR1	Farmer Is. (Piper Reef)	■	■	■	■	■	■	■	■	■
	FR2	Farmer Is. (Piper Reef)	■	■	■	■	■	■	■	■	■
	ST1	Stanley Island (Flinders Group)	■	■	■	■	■	■	■	■	■
	ST2	Stanley Island (Flinders Group)	■	■	■	■	■	■	■	■	■
	YY1*	Yum Yum Beach (Weymouth Bay)	□	□			□		□	□	□
Reef subtidal	FG1†	Flinders Island (Flinders Group)	■	■							■
	FG2†	Flinders Island (Flinders Group)	■	■							■

5.1.3.1 Seagrass index and indicator scores

In the 2018–19 monitoring period, the seagrass condition index score for the Cape York region reduced slightly since the previous monitoring period, but the overall grade remained **poor** (Figure 28). The reduction was due to lower scores in reproductive effort and tissue nutrients, but there were small gains in the abundance score.

The greatest score reduction occurred in reproductive effort, which received the lowest grade of zero for the second time since monitoring began. The previous occurrence in 2006–07 was based only on Archer Point as other sites had not yet been commissioned, making 2018–19 the first time that there had been no reproductive structures at all sites (Figure 28). Other counts with zero reproductive effort have been observed in the wet season but wet season data is not used in the metric, because it is a time that counts are typically low. In 2018–19, there were no reproductive structures recorded at any of the sites. Tissue nutrients remained poor and at the second lowest level recorded in Cape York.

Overall, the Cape York seagrass condition index remains well below the 2005–06 baseline and in 2018–19 was the lowest score since the addition of new sites in 2012–13.

An examination of the long-term trends across the Cape York NRM region needs to be interpreted carefully as new sites were included in 2012–13, which are associated with consistently lower abundance and tissue nutrients compared to the highest levels recorded for the region. Archer Point, which was the only location monitored prior to 2012–13, is now only monitored as part of the Seagrass-Watch due to logistical complications (Figure 28).

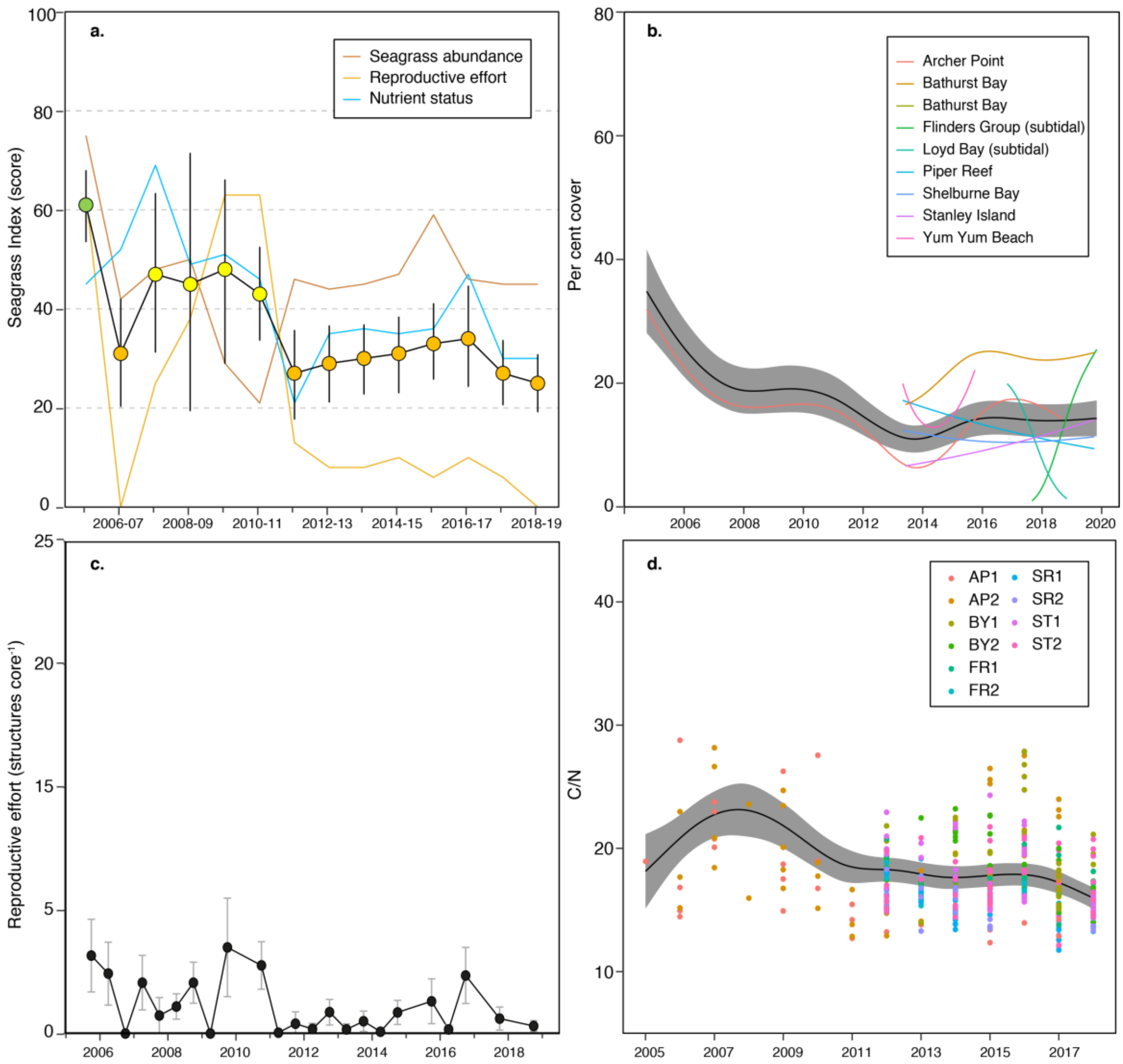


Figure 28. *Temporal trends in the Cape York seagrass condition index and the indicators used to calculate the index: a. seagrass condition index (circles) and indicator trends (lines); b. GAM plots of seagrass abundance (per cent cover) trends for each location (coloured lines) and the region (black line with grey shaded area defining 95% confidence intervals); c. average number of reproductive structures (±SE) (GAM not possible due to high count of zero values); and d. elemental ratios (atomic) of leaf tissue C:N nutrient content at each site (coloured circles) and regional trend represented by a GAM plot as dark line with shaded areas defining 95% confidence intervals of the trend.*

5.1.3.2 Seagrass abundance, composition and extent

The increase in seagrass abundance in 2018–19 appears a consequence of improvements in per cent cover at 50% of sites across all habitats; except the intertidal reef habitat at Piper Reef where one site (FR2) slightly decreased (Figure 29).

On average, seagrass abundance marginally increased relative to the previous period. Seagrass abundance (per cent cover) increased at half or remained similar at 40% of sites across all habitats, predominately in coastal and subtidal reef meadows. The only declines occurred in meadows located in the north of the region.

An examination of the long-term trend in seagrass abundance shows seagrass per cent cover progressively decreased at intertidal reef habitats across Cape York from 2003 to 2012, with relatively little improvement since (i.e. no trend) (Figure 29, Table 20). Coastal intertidal and subtidal habitats which have only been monitored since 2012 and 2015 respectively, generally showed no trend (Figure 29, Table 20).

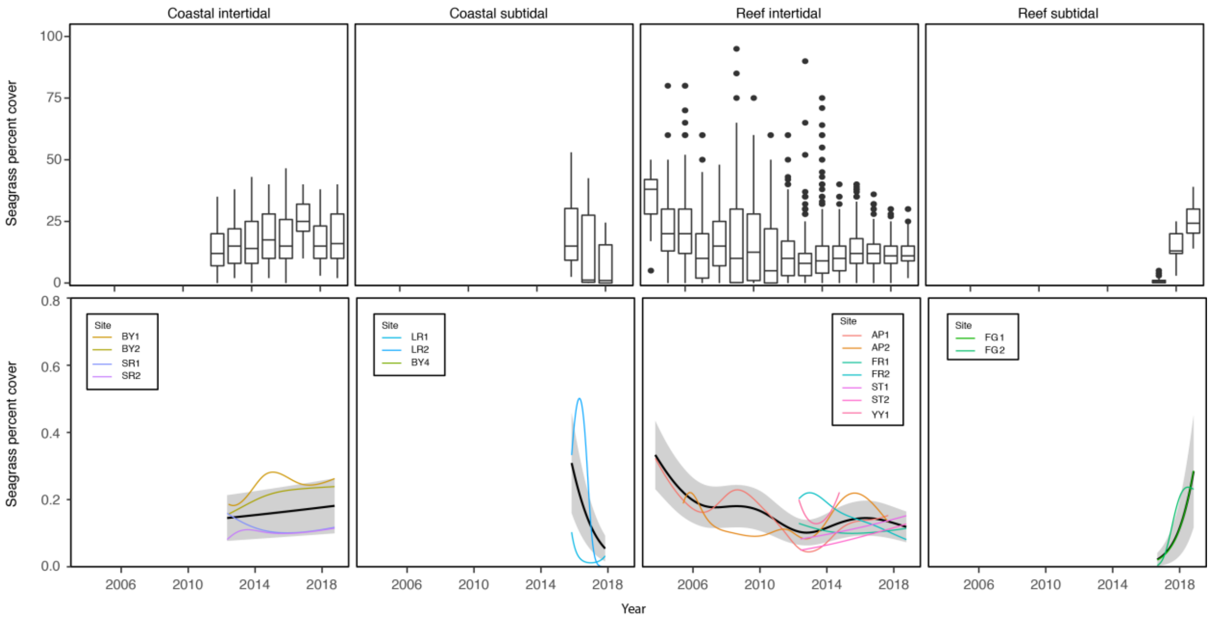


Figure 29. *Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends for each habitat monitored in the Cape York region from June 2005 to May 2019. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates the 25th percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75th percentile. Whiskers (error bars) above and below the box indicate the 90th and 10th percentiles, and the dots represent outlying points. GAMM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.*

In 2018–19, intertidal coastal seagrass habitats in the Cape York NRM region were composed of species displaying colonising traits at a Reef-average level (Figure 30). Subtidal reef habitats increased their composition of colonising species from the previous monitoring period; conversely, intertidal reef habitats reduced in composition of species displaying colonising traits (Figure 30).

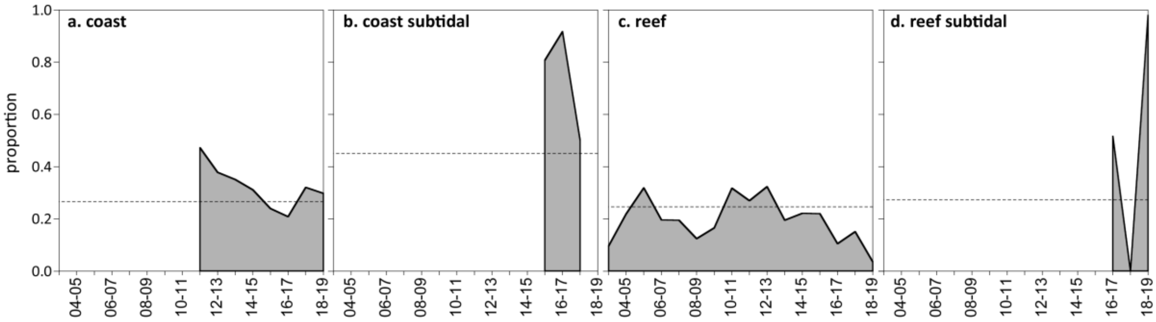


Figure 30. *Proportion of seagrass abundance composed of species displaying colonising traits at inshore habitats in the Cape York region.The dashed line represents Reef long-term average for each habitat type.*

Seagrass spatial extent mapping was conducted within meadows to determine if changes in abundance were a consequence of the meadow landscape changing and to indicate if plants were allocating resources to colonisation (asexual reproduction). Prior to 2012, the only meadow extent mapping in the Cape York region was conducted at reef intertidal meadows at Archer Point. The meadows within monitoring sites on the reef flat at Archer Point have fluctuated within and between years (Figure 31), primarily due to changes in the landward edge and appearance of a drainage channel from an adjacent creek (data not presented). As of 2012–13, additional reef and coastal meadows in the Cape York region were included. Overall, meadow extent has been relatively stable since 2012 (Figure 31).

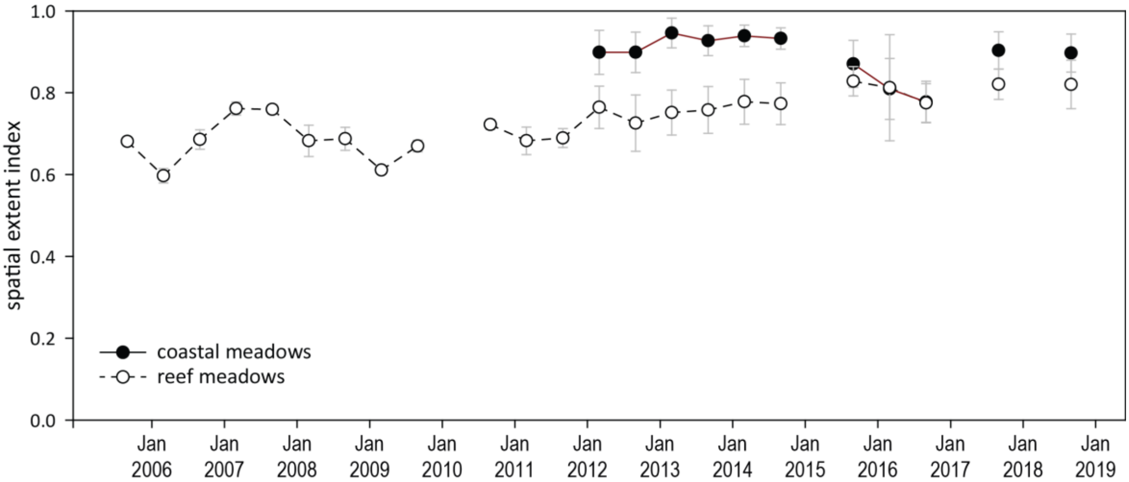


Figure 31. *Change in spatial extent of seagrass meadows within monitoring sites for each intertidal coastal and reef habitat and monitoring period across the eastern Cape York NRM region.*

5.1.3.3 Seagrass reproductive status

Total reproductive effort is only monitored at intertidal meadows in Cape York. Reproductive effort declined at reef habitats in 2018–19, and remained low at coastal habitats across the region (Figure 32). Seed banks are also only measured at intertidal sites across Cape York and are dominated by *Halodule uninervis*. Seed density has declined at coastal habitats in 2018–19 but remains at relatively high levels compared to coastal sites in other regions, and remains much higher than those found in reef habitats. At reef sites, there has been few or no seeds recorded since 2013, and these meadows may have poor recovery rates if there is substantial decline in seagrass abundance.

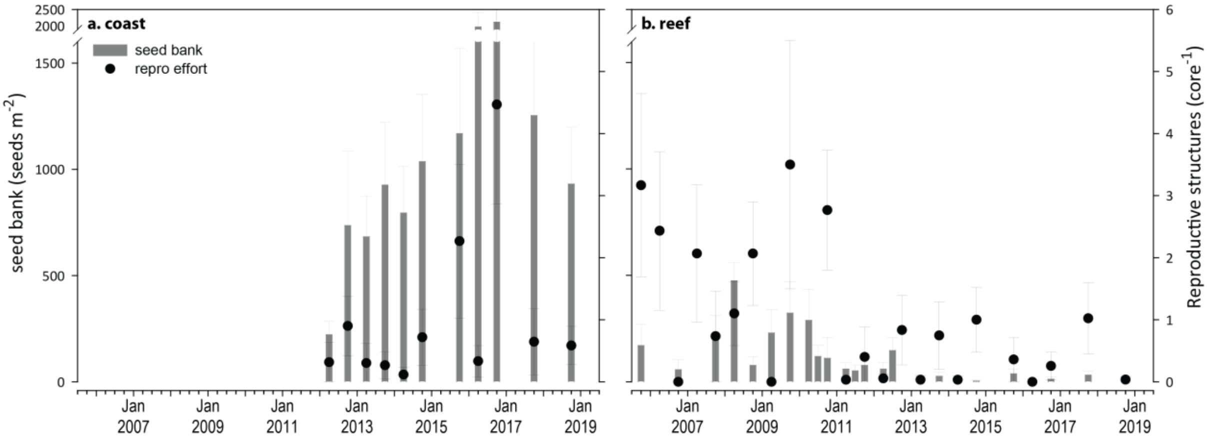


Figure 32. *Seed banks and reproductive effort at inshore intertidal coastal (a) and reef (b) habitats in the Cape York region(species and sites pooled). Seed banks (bars ± SE) presented as the total number of seeds per m² sediment surface. Reproductive effort for late dry season (dots ± SE) presented as the average number of reproductive structures per core.*

5.1.3.4 Seagrass leaf tissue nutrients

Seagrass leaf molar C:N ratios in 2018–19 remained similar to the previous year and within range of those observed since the introduction of additional sites in 2012–13 (Figure 33). However; there was a very small decline in reef intertidal habitats resulting in a small decline in the tissue nutrient score. Leaf N:P ratios and %N remained above guideline and global median in 2018–19 for the second year in a row (Figure 33), indicating that nitrogen remains high in the seagrass habitats of Cape York, but the low and/or negative $\delta^{15}N$ (Figure 33) suggests this is not an anthropogenic source of N.

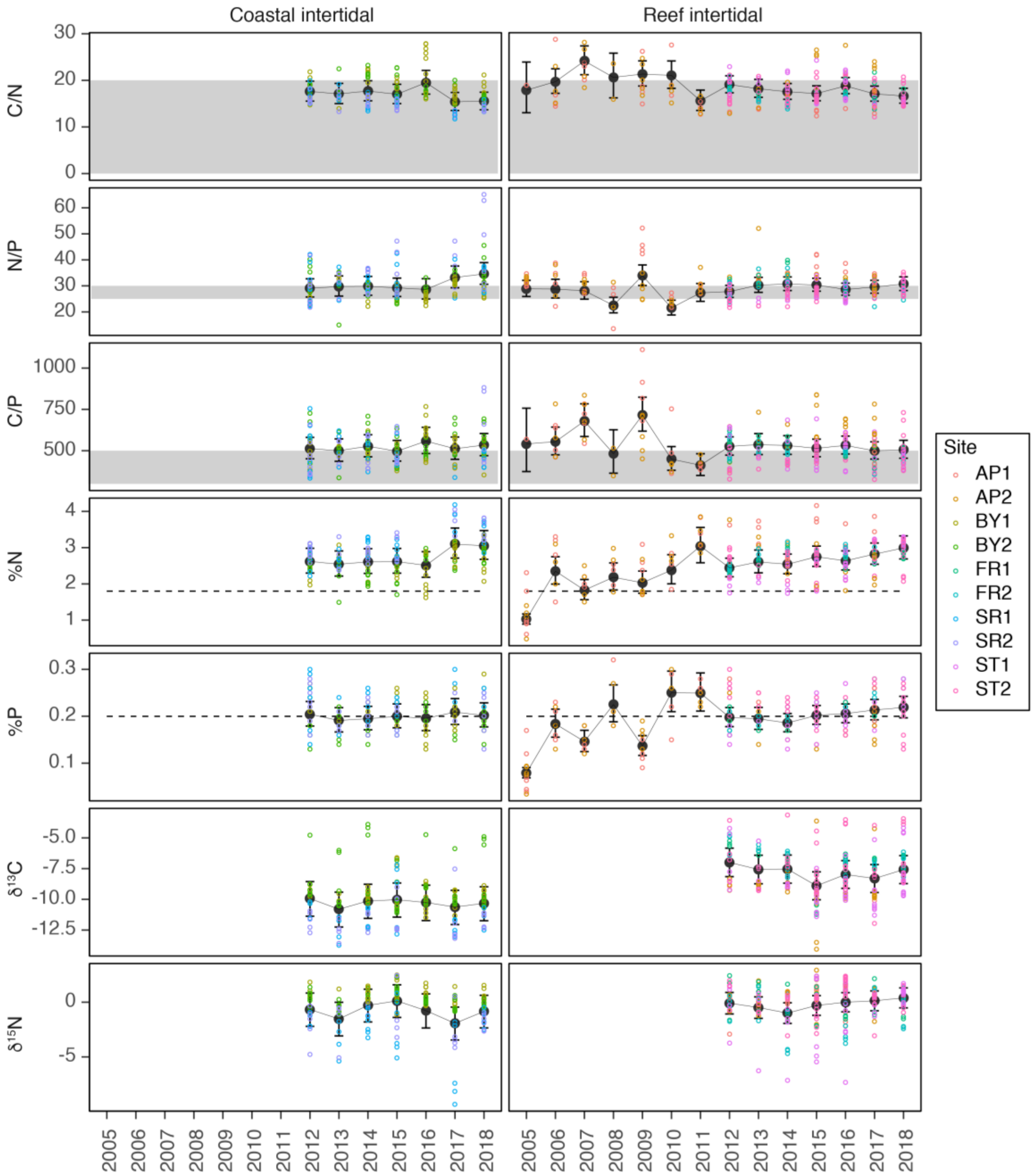


Figure 33. *Seagrass leaf tissue nutrient elemental ratios (C:N:P) and concentrations (%N,%P, $\delta^{13}C$ and $\delta^{15}N$) for each habitat in the Cape York NRM region (± SE) (foundation species pooled). Horizontal shaded bands or dashed lines represent the accepted seagrass guideline values, where: C:N ratios within the band may indicate reduced light availability and/or N enrichment; N:P ratios above the band indicate P limitation, below indicate N limitation and within indicates replete, and; C:P ratios within the band may indicate nutrient rich habitats (large P pool). Dashed lines in %N and %P indicate global median values of 1.8% and 0.2% for tissue nitrogen and phosphorus, respectively (Duarte 1990).*

5.1.3.5 Epiphytes and macroalgae

Epiphyte cover on seagrass leaf blades remained below the long-term average at coastal habitats, and fluctuated close to the long-term average at reef habitats (Figure 34).

Per cent cover of macroalgae was variable between locations, and remained above the Reef long-term average for reef habitats in the central and north of the region for the sixth consecutive year (Figure 34). Macroalgae cover at coastal sites has varied little and this year remained near to the Reef-wide long-term average (Figure 34).

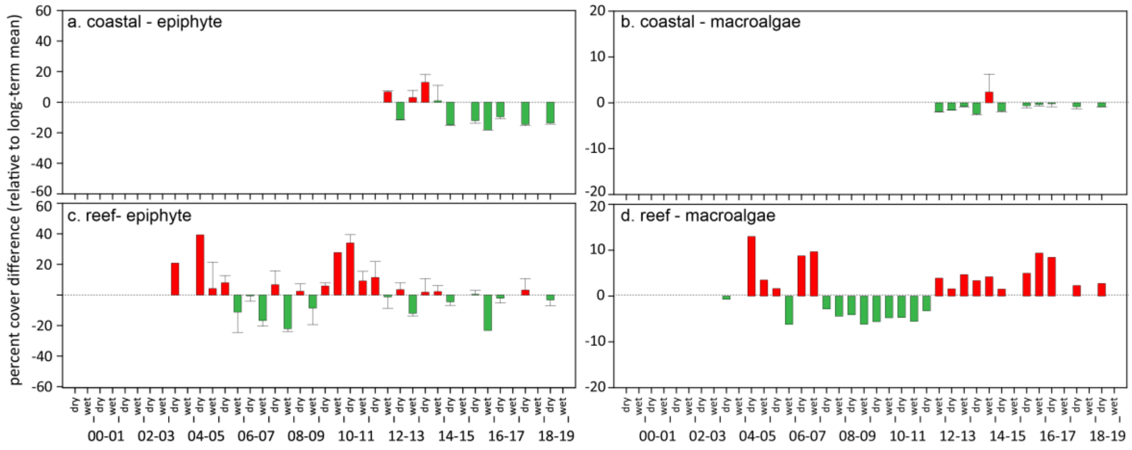


Figure 34. *Deviations in mean epiphyte and macroalgae abundance (per cent cover) at monitoring habitats in the Cape York region, relative to the Reef long-term average (sites pooled, ±SE). Green bars indicate positive deviations for condition, red bars negative.*

5.2 Wet Tropics

5.2.1 2018–19 Summary

Seagrass meadows within the Wet Tropics showed an overall improvement in seagrass condition index in 2018–19, but they remain in a vulnerable state, particularly in the southern Wet Tropics region. Although the status of seagrass condition in the northern Wet Tropics NRM region increased to the highest score since monitoring was established, it remained **moderate**. Similarly, seagrass condition improved from very poor to **poor** in the southern Wet Tropics (Figure 35), although the combined regional condition was **poor** (Figure 16). Contributing indicators in the north were:

- abundance score was moderate
- reproductive effort score was good
- tissue nutrient score was poor.

Contributing indicators in the south were:

- abundance score was poor
- reproductive effort score was very poor
- tissue nutrient score was poor.

Seagrass abundance decreased slightly in 2018-19 relative to the previous period in the northern Wet Tropics sites, which is likely to have been affected by above-average discharge from rivers influencing the northern Wet Tropics and lower than long-term average light levels for the second consecutive period. In the south, the seagrass abundance score increased to the highest level observed for southern sites, which may have been facilitated by average river discharges from most of the rivers influencing the south and cooler water temperature for the second year in a row.

An examination of temporal trends in seagrass abundance across the region shows a high degree of variability reflecting a complex range of environmental and biological processes. In the north, 40% of reef sites have significantly declined in abundance over the long-term, while no trend was apparent for the remaining sites or habitats. In the south, only sites in coastal habitats have significantly declined over the long-term. The declines are a consequence of the significant losses that occurred from 2009 to 2011, the result of multiple years of above-average rainfall and severe weather events. Recovery of seagrass meadows post 2011 has been challenged, particularly in the south, by unstable substrates (legacy of cyclone Yasi), chronic poor water quality (high turbidity, light limitation, elevated temperatures), and limited recruitment capacity.

While meadows in the north have maintained a healthy seed bank and reproductive effort reached record high levels during 2018–19, in the south reproductive structures remain at very low levels and seed banks absent. This has limited recovery in the south to relying on expansion of remnant plants or recruitment from elsewhere (e.g., vegetative fragments).

Leaf tissue nutrients (C:N) have remained relatively unchanged in the north for a number of years, and suggest an excess of nitrogen relative to photosynthetic C uptake (C:N <20), which is consistent with the high frequency of exposure to secondary water particularly in coastal habitat. Nutrient status therefore remained poor. In the south, the nutrient status indicator increased at coastal and reef intertidal sites, but declined at reef subtidal sites resulting in the 2018–19 score remaining poor.

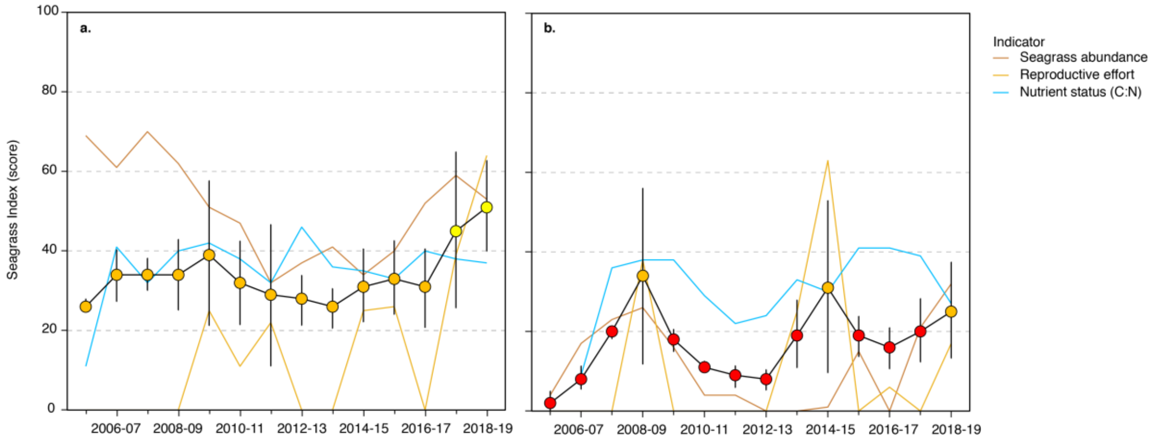


Figure 35. *Report card of seagrass index and indicators for the northern (a.) and southern (b.) Wet Tropics NRM region (average across habitats and sites). Values are indexed scores scaled 0–100 and graded: ● = very good (81-100), ● = good (61 - 80), ● = moderate (41 - 60), ● = poor (21 - 40), ● = very poor (0 - 20). NB: Scores are unitless.*

5.2.2 Climate and environmental pressures

Cyclone Owen affected the Wet Tropics region when it crossed back into the Reef near Cardwell as a tropical depression in December 2018 (Gruber et al 2020).

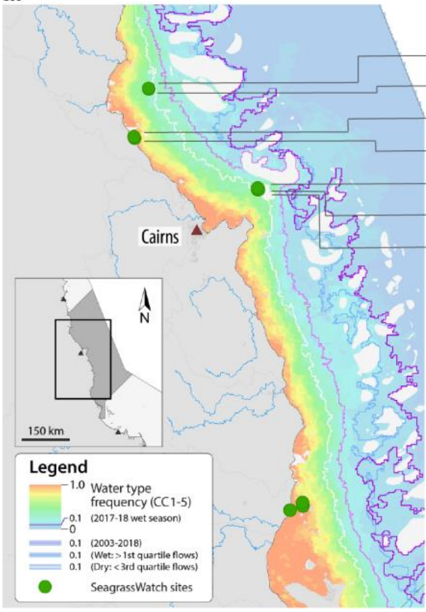
Annual rainfall, river discharge and exposure to primary (‘brown’ sediment laden) or secondary (‘green’, phytoplankton rich) turbid water were above the long-term average across the northern Wet Tropics during 2018–19 (Figure 36). Furthermore, benthic light levels (12.0 mol m^{-2} d^{-1} in 2018–19) were lower than the long-term average in the northern Wet Tropics (12.7 mol m^{-2} d^{-1}) (Figure 36). Sites were primarily exposed to ‘green’ water (class 5), which allows more light to reach the seagrass habitats than ‘brown’ water (Gruber *et al.* 2020), and therefore although light levels were lower than usual, they were still, on average, above 10 mol m^{-2} d^{-1} long-term light threshold (Collier et al 2016).

Intertidal within-canopy temperatures in the northern Wet Tropics were above the long-term average and this year were the sixth highest average annual temperature (27.0°C) since 2003 (Figure 36). Maximum intertidal within-canopy temperatures exceeded 35°C for a total of 58 days during 2018–19, with the highest temperature recorded at 41°C (YP2, 2pm 04Dec18).

This was the second year since 2014–15 where annual subtidal within-canopy temperatures in the north were below the long-term average and the fifth lowest average annual temperature (26.5°C) since 2008. The maximum subtidal temperature recorded this year was 31.7°C (LI2, 6pm 21Feb19), below temperatures expected to stress seagrass.

Daily tide exposure in the north was below the long-term average for the second consecutive year (Figure 36, Figure 92, Figure 93), which may have provided some respite from the elevated temperatures, particularly in coastal habitats.

a.



b.

Wk	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22
LI2	7	6	6	7	6	6	6	6			4	5	5	5	6		5	5	5	5	6	6
LI1	7	6	6	7	6	6	6	6			4	5	5	5	6		5	5	5	5	6	6
YP2	5	5	5			5	5	5			4	4	5	5	5		5	4	5	5	5	5
YP1	4	5	5			5	5	5			4	4	5	5	5		5	4	4	4	5	5
GI3	7	6	6		5	6	6	6		6	5	5	5	5	6		5	5	5	6	6	7
GI2	7	6	6		5	6	6	6		6	5	5	5	5	6		5	5	5	6	6	7
GI1	7	6	6		5	6	6	6		6	5	5	5	5	6		5	5	5	6	6	7

1 - 4 = turbid primary water (red/brown), 5 = nutrient/chlorophyll-enriched secondary water (green), 6 = some freshwater/CDOM influence tertiary water (blue) , 7 = no plume influence (white), blank = no data, clouds (grey)

c.

	Long-term average	2018-19
Rainfall (1961-1990)	12.2 mm d^{-1}	15.6 mm d^{-1}
River discharge (1970-2018)	7,986,614 L yr^{-1}	11,746,653 L yr^{-1}
Turbid water exposure (2006-2018)	46%	61%
Daytime tidal exposure (1999-2018)	138 hr yr^{-1}	116 hr yr^{-1}
Within-canopy temperature - intertidal (2003-18)	26.9ºC (41.5ºC)	27.0ºC (41.0ºC)
- subtidal (2008-18)	26.8ºC (36.7ºC)	26.5ºC (32.7ºC)
Within-canopy light	12.7 mol m^{-2} d^{-1}	12.0 mol m^{-2} d^{-1}

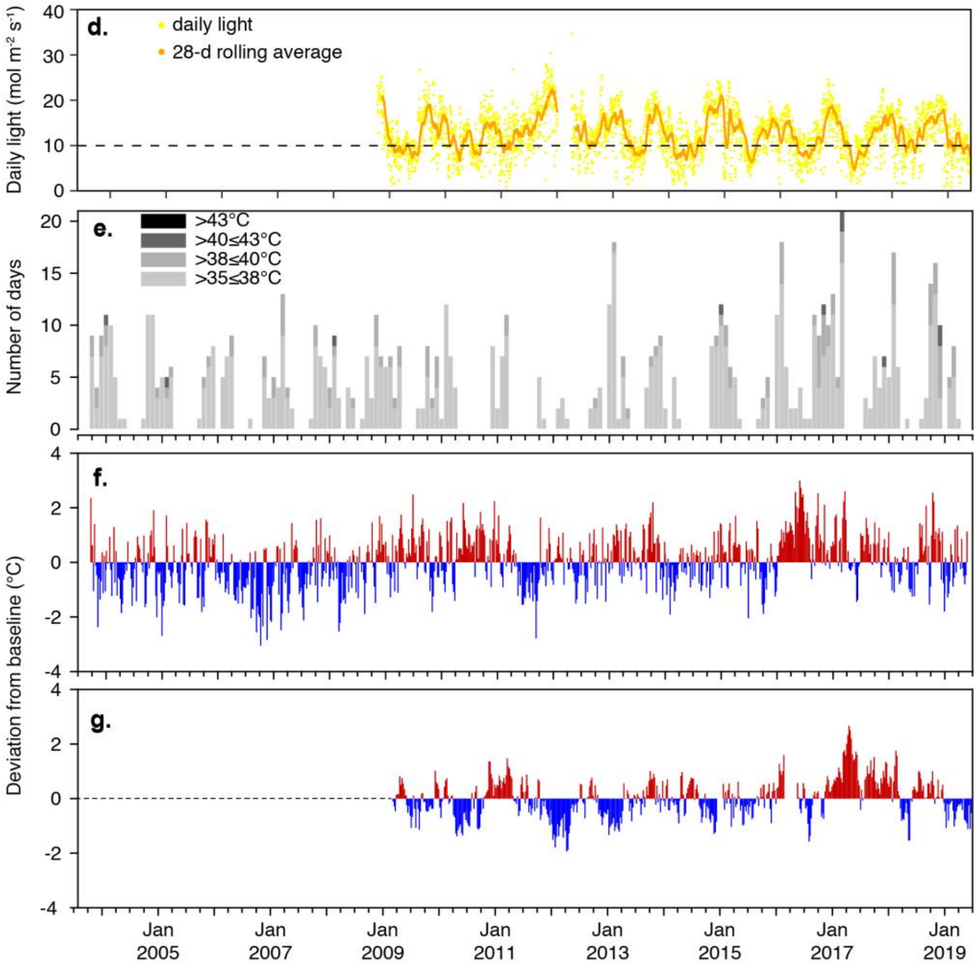


Figure 36. *Environmental pressures in the northern Wet Tropics region including: a. frequency of exposure to turbid water (colour classes 1–5) (from Gruber et al. 2019b); b. wet season water type at each site; c. average conditions over the long-term and in 2018–19; d. daily light and the 28-day rolling mean of daily light for all sites; e. number of days temperature exceeded 35°C, 38°C, 40°C and 43°C; f. intertidal temperature deviations from 13-year mean weekly records, and; g. subtidal temperature deviations from 13-year mean weekly records.*

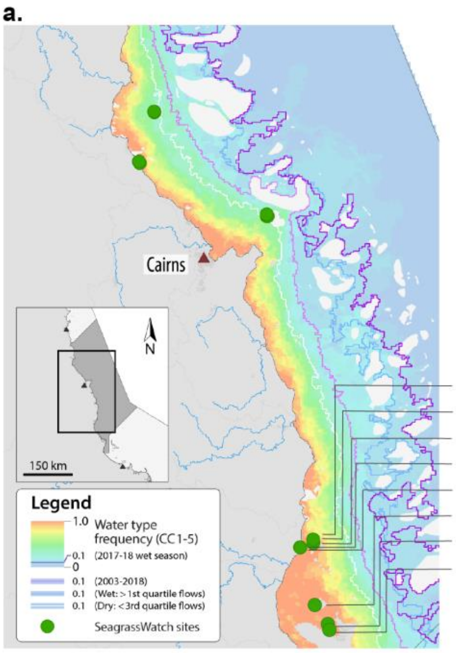
Annual rainfall and river discharge was slightly above-average across the southern Wet Tropics during 2018–19, while exposure to 'brown' or 'green' turbid water for the wet season was similar to the long-term average with 100% frequency of exposure (Figure 37). Coastal sites at Lugger Bay (LB1 and LB2) and Missionary Bay (MS1 and MS2) experienced the highest exposure to 'brown' turbid water, while the remaining reef sites were exposed predominately to 'green' water. Light levels are only measured at Dunk Island in the southern Wet Tropics, where they were lower than the long-term average. At the subtidal site, the annual average (5.5 mol m^{-2} d^{-1}) was below both acute (6 mol m^{-2} d^{-1}) and long-term light thresholds (10 mol m^{-2} d^{-1}), particularly during the wet season (Figure 37, Figure 101).

In the southern Wet Tropics, within-canopy temperatures in 2018–19 remained similar to the long-term average for the second year since 2014–15 (Figure 37). Maximum intertidal within-canopy temperatures exceeded 35°C for only one day during 2018–19, with the highest temperature recorded at 35.1°C (DI2, 4pm 19 Feb 19). The maximum subtidal within-canopy temperature recorded during 2018-19 was 31.5°C (1pm 19 Dec 18).

Daily tide exposure was above the long-term average for the first time in three years (Figure 36, Figure 92, Figure 93), which may have exerted additional stress on plant growth by carbon limitation.

Overall, the inshore seagrass habitats throughout the southern Wet Tropics experienced much greater environmental pressures in 2018–19 than those in the northern Wet Tropics, and the previous monitoring period.

In 2018–19, sediments appeared similar to the long-term and the proportion of fine sediments (i.e. mud) was well below the Reef-wide long-term average across all habitats (Figure 108, Figure 109). Across the Wet Tropics region, coastal sediments were composed primarily of fine sand, while reef habitats were composed of sand and coarser sediments (Figure 108, Figure 109). Subtidal reef sediments were predominately sand, which in the southern region often included coarser grains (Figure 110).



b.

	Long-term average	2018-19
Rainfall (1961-1990)	7.8 mm d^{-1}	10.5 mm d^{-1}
River discharge (1970-2018)	7,178,231 L yr^{-1}	10,247,400 yr^{-1}
Turbid water exposure (2006-2018)	99%	100%
Daytime tidal exposure (1999-2018)	37 hr yr^{-1}	48 hr yr^{-1}
Within-canopy temperature - intertidal (2006-18)	26.9°C (41.2°C)	26.9°C (35.4°C)
- subtidal (2007-18)	25.9°C (32.2°C)	26.2°C (31.5°C)
Within-canopy light	11.2 mol m^{-2} d^{1}	9.3 mol m^{-2} d^{1}

c.

Wk	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22
DI3	5	5	4	5	4	5	5	5			4	4	5	5	5	5	5	5	4	5	5	5
DI1	5	5	4	5	4	5	5	5			4	4	5	5	5	5	5	5	4	5	5	5
DI2	5	5	4	5	4	5	5	5			4	4	5	5	5	5	5	5	4	5	5	5
LB1	5	4	4	5	4	4	4	5			4	4	4	5	5	5	5	5	4	4	4	5
LB2	5	4	4	5	4	4	4	5			4	4	4	5	5	5	5	5	4	4	4	5
GO1	5	5	4	5	5	5	5	5			4	4	5	5	5	5	5		5	5	5	5
MS2	3	4	4	5	4	4	5	5			4	4	5	4	4	5	5		5	5	5	5
MS1	1	5	4	5	1	4	5	5			4	4	5	4	5	5	5		4	5	5	5

1 - 4 = turbid primary water (red/brown), 5 = nutrient/chlorophyll-enriched secondary water (green), 6 = some freshwater/CDOM influence tertiary water (blue) , 7 = no plume influence (white), blank = no data, clouds (grey)

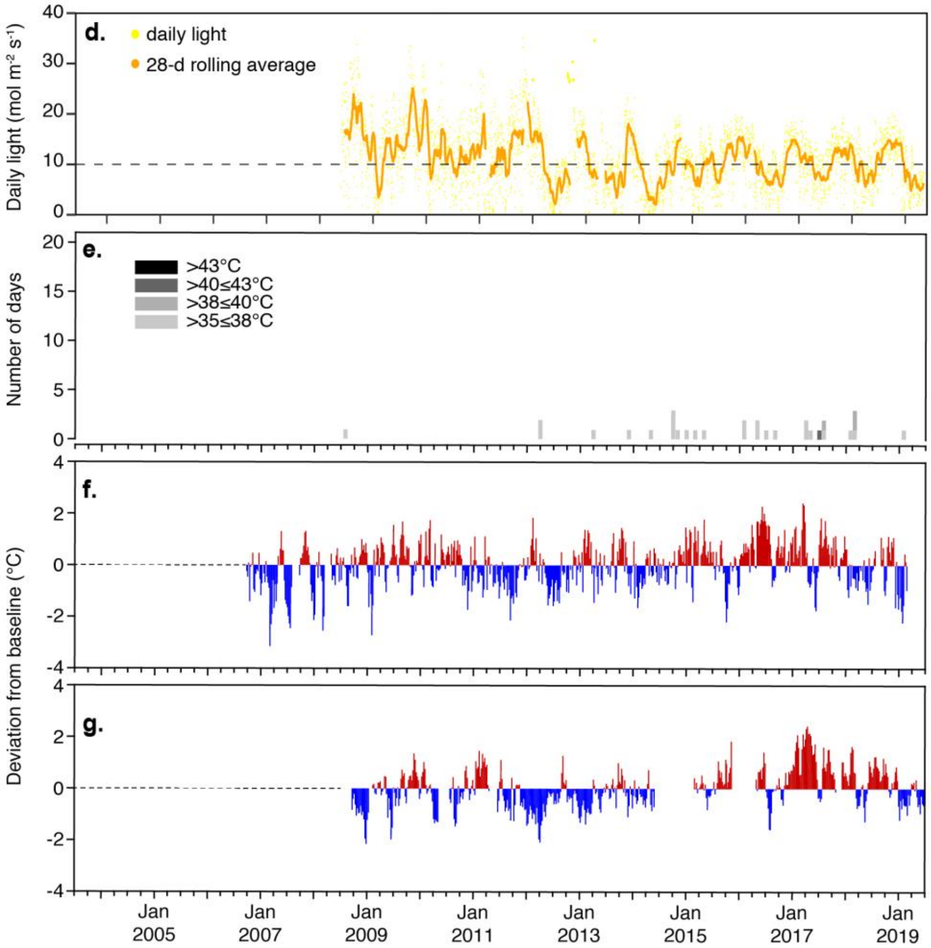


Figure 37. *Environmental pressures in the southern Wet Tropics region including: a. frequency of exposure to turbid water (colour classes 1–5) (from Gruber et al. 2019b); b. wet season water type at each site; c. average conditions over the long-term and in 2018–19; d. daily light and the 28-day rolling mean of daily light for all sites; e. number of days temperature exceeded 35°C, 38°C, 40°C and 43°C; f. intertidal temperature deviations from 13-year mean weekly records, and; g. subtidal temperature deviations from 13-year mean weekly records.*

5.2.3 Inshore seagrass and habitat condition

There are 12 seagrass monitoring sites in the Wet Tropics from 7 locations (Table 12). Three seagrass habitat types were assessed across the region (Table 12).

Table 12. *List of data sources of seagrass and environmental condition indicators for each seagrass habitat type in the Wet Tropics NRM region. † drop camera sampling (QPWS), *Seagrass-Watch. For site details see* Table 2 *and* Table 3*.*

Sub region	Habitat	Site		abundance	composition	distribution	Reproductive effort	seed banks	Leaf tissue nutrients	Meadow sediments	Epiphytes	Macroalgae
north	coastal intertidal	YP1	Yule Point	■	■	■	■	■	■	■	■	■
		YP2	Yule Point	■	■	■	■	■	■	■	■	■
	reef intertidal	LI1	Low Isles	■	■	■	■	■	■	■	■	■
		GI1	Green Island	■	■	■	■	■	■	■	■	■
		GI2	Green Island	■	■	■	■	■	■	■	■	■
	reef subtidal	LI2	Low Isles	■	■	■	■	■	■	■	■	■
		GI3	Green Island	■	■	■	■	■	■	■	■	■
south	coastal intertidal	LB1	Lugger Bay	■	■	■	■	■	■	■	■	■
		LB2	Lugger Bay	■	■	■	■	■	■	■	■	■
	coastal subtidal	MS1†	Missionary Bay	■	■							■
		MS2†	Missionary Bay	■	■							■
	reef intertidal	DI1	Dunk Island	■	■	■	■	■	■	■	■	■
		DI2	Dunk Island	■	■	■	■	■	■	■	■	■
		GO1*	Goold Island	□	□			□		□	□	□
	reef subtidal	DI3	Dunk Island	■	■	■	■	■	■	■	■	■

5.2.3.1 Seagrass index and indicator scores

In the 2018-19 monitoring period, the seagrass condition index for the overall Wet Tropics region increased to the highest score since reporting was established, but the overall grade remained poor (Figure 16). The increase was due to improved scores in two indicators: abundance and reproductive effort. The only indicator to change grade from the previous year was reproductive effort, which increased from very poor to poor. Examination of the sub-regional scores highlights the differences between seagrass condition in the north and south of the Wet Tropics (Figure 35).

In the northern Wet Tropics, the seagrass condition index increased to the highest score since reporting was established, improving to a moderate grading (Figure 38). Similar to the overall NRM regional grade, the increase appears primarily due to improved reproductive effort scores.

The seagrass abundance score has progressively improved since 2013–2014, peaking in 2017-18, and although declining slightly remains graded as moderate in 2018–19 (Figure 38). The long-term trend in seagrass per cent cover is variable between monitoring locations (Table 20), but closely reflects the sub-regional scores with improved cover from 2015.

Reproductive effort has fluctuated the most of the three condition indicators, and in 2018–19 was the highest score since monitoring was established (Figure 38). Due to the variable nature of sexual reproduction in seagrass systems, no long term trends are apparent.

In contract, seagrass leaf nutrient (C:N) status has varied the least of all indicators, and although declined marginally in 2018–19, has remained in a poor grade (Figure 38). Examination of the long-term trend in nutrient status, suggests a significant increase for a period between 2006 and 2009 (Figure 38).

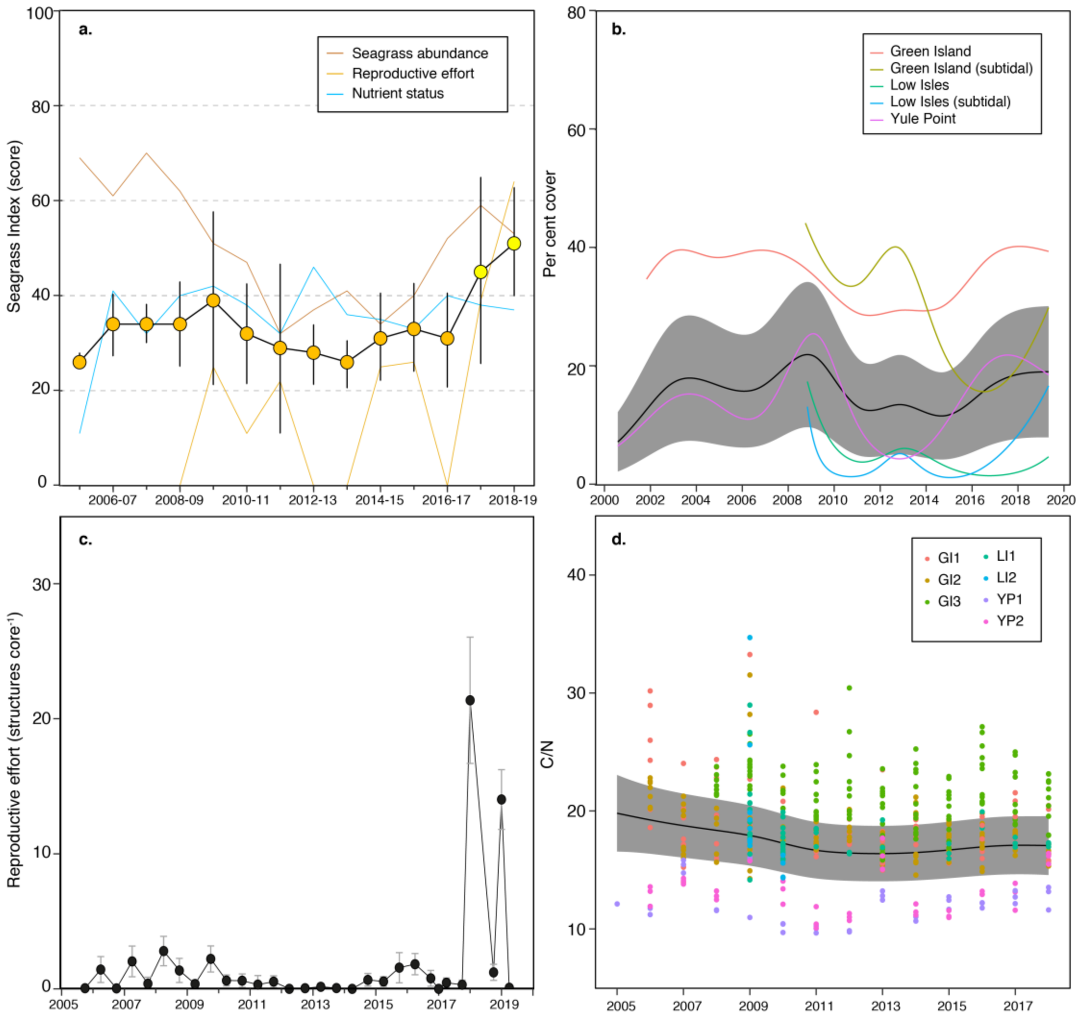


Figure 38. *Temporal trends in the northern Wet Tropics seagrass condition index and the indicators used to calculate the index: a. seagrass condition index (circles) and indicator trends (lines); b. GAM plots of seagrass abundance (per cent cover) trends for each location (coloured lines) and the region (black line with grey shaded area defining 95% confidence intervals); c. average number of reproductive structures (±SE) (GAM not possible due to high count of zero values); and d. elemental ratios (atomic) of leaf tissue C:N nutrient content at each site (coloured circles) and regional trend represented by a GAM plot as dark line with shaded areas defining 95% confidence intervals of the trend.*

In the southern Wet Tropics, the seagrass condition index improved from very poor to poor in 2018–19, a consequence of improved abundance and reproductive effort scores (Figure 39).

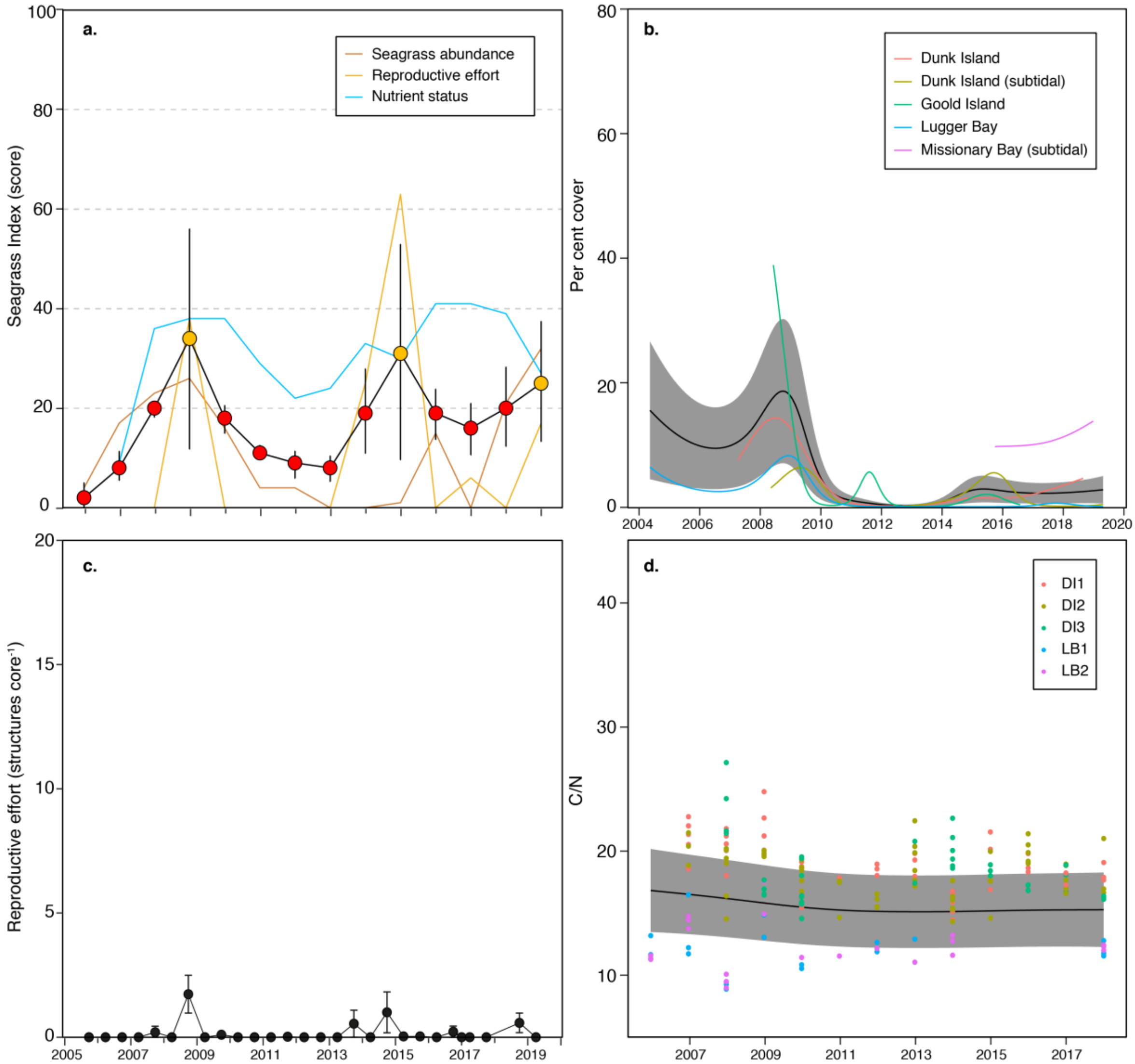


Figure 39. *Temporal trends in the southern Wet Tropics seagrass condition index and the indicators used to calculate the index: a. seagrass condition index (circles) and indicator trends (lines); b. GAM plots of seagrass abundance (per cent cover) trends for each location (coloured lines) and the region (black line with grey shaded area defining 95% confidence intervals); c. average number of reproductive structures (±SE) (GAM not possible due to high count of zero values); and d. elemental ratios (atomic) of leaf tissue C:N nutrient content at each site (coloured circles) and regional trend represented by a GAM plot as dark line with shaded areas defining 95% confidence intervals of the trend.*

5.2.3.2 Seagrass abundance, community and extent

Seagrass meadows are more abundant across all habitats in the northern than the southern Wet Tropics (Figure 40, Figure 41). In the northern Wet Tropics, seagrass abundance over the long-term is higher at intertidal reef (28.0 ±2.2%) than subtidal reef (17.0 ±2.5%) or coastal habitats (14.5 ±1.6%). In 2018–19, although seagrass abundances remained steady at 14% of sites, the increase in abundance observed at approximately 43% of sites, was offset by declines experienced at the remaining sites, resulting in a slight decrease in abundance overall.

Although seagrass losses have occurred at the local level (e.g. individual sites) for some period over the duration of the monitoring, complete loss has not occurred at the habitat level. Nevertheless, abundance has fluctuated between and within years. For example, seagrass cover at coastal habitats differs between seasons (9.7 ±1.3% in the dry and 19.8 ±2.1% in the late dry-monsoon) and years (from 3.9% to 24.9% annual average).

In the southern Wet Tropics, although seagrass abundance is similarly higher at intertidal reef (5.1 ±1.1%) than subtidal reef (1.9 ±0.8%) or coastal habitats (1.9 ±0.6%), the abundances are a mere tenth of those to the north. This is a consequence of periods of complete loss occurring at all habitats for at least 3-6 months since early 2011. At coastal habitats in Lugger Bay, complete loss has been sustained for periods of years. Although recovery is very slow, isolated seagrass shoots appeared at Lugger Bay sites in 2016–17, and by 2018–19 small patches had established. Abundances similarly improved at the reef intertidal habitats, but remain well below historical levels.

An examination of temporal trends in seagrass abundance across the Wet Tropics NRM region show no significant trend over the long-term (Table 20). In the northern Wet Tropics, changes in seagrass abundance were variable among habitats, with 29% of sites significantly declining over the long-term, while no trend was apparent for the remaining sites. The declines in the north are all in reef habitats; 33% of intertidal and 50% subtidal. In the southern sub-region, 25% of sites have significantly declined over the long-term, but these only occurred at coastal sites (Lugger Bay). No long-term trend was apparent in the reef habitats of the southern sub-region.

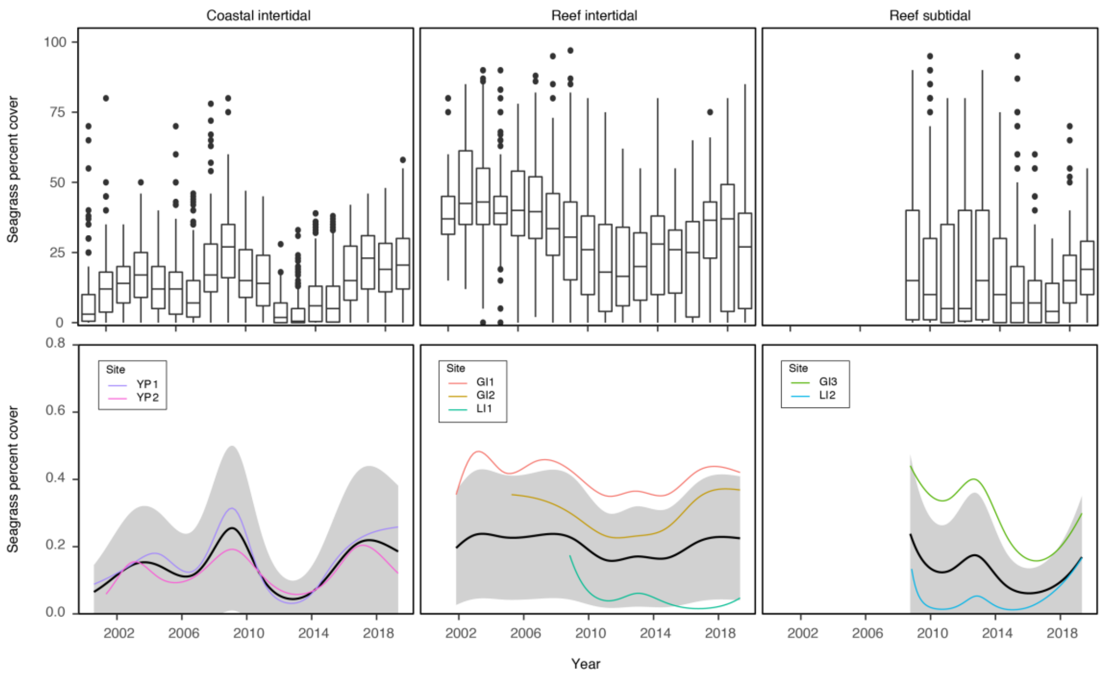


Figure 40. *Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the northern Wet Tropics NRM region from 2001 to 2019. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates the 25th percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75th percentile. Whiskers (error bars) above and below the box indicate the 90th and 10th percentiles, and the dots represent outlying points. GAMM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.*

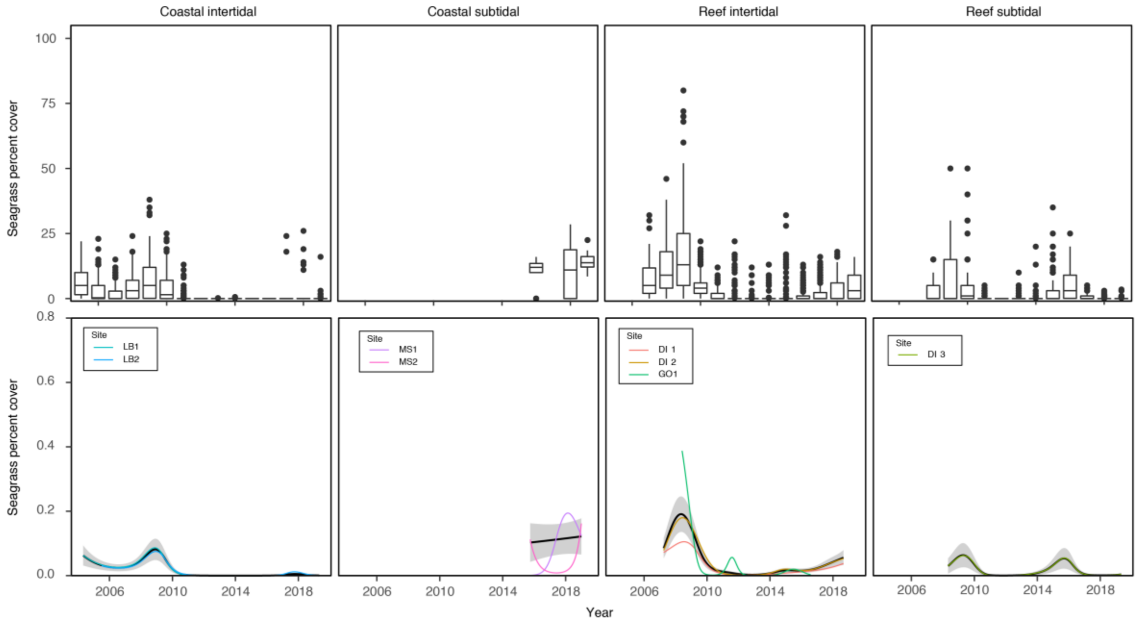


Figure 41. *Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the southern Wet Tropics NRM region from 2001 to 2019. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates the 25th percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75th percentile. Whiskers (error bars) above and below the box indicate the 90th and 10th percentiles, and the dots represent outlying points. GAMM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.*

The proportion of seagrass species displaying colonising traits varied across habitats in the northern Wet Tropics (Figure 42). In 2018–19 the proportion decreased slightly at coastal intertidal habitats (Yule Point), suggesting some recovery and reduced physical disturbance. Between 2010 and 2014, all habitats were either dominated or had higher than the Reef-average of species displaying colonising traits. Post 2014, the composition of species displaying colonising traits has fallen below the Reef-wide average, with the exception of reef subtidal habitats.

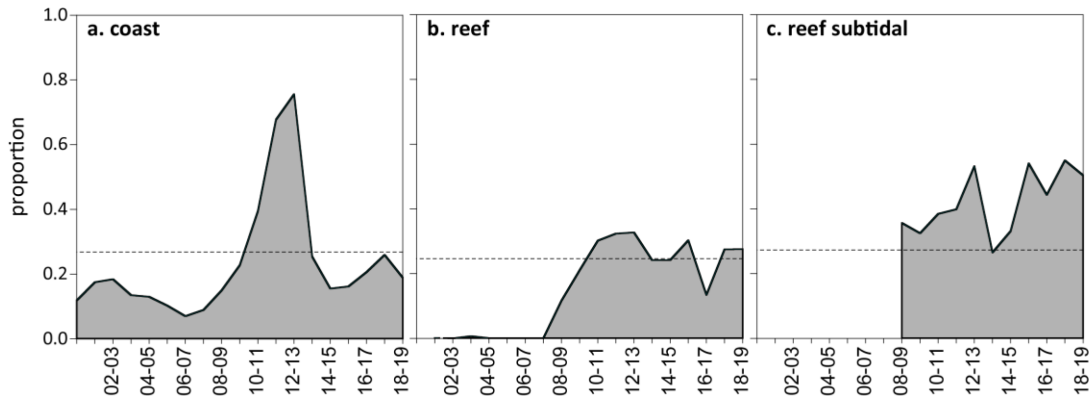


Figure 42. *Proportion of seagrass abundance composed of colonising species at inshore habitats in the northern Wet Tropics region, from the 2000–2001 to the 2018–19 reporting periods. The dashed line represents the Reef-wide average for each habitat type.*

In the southern Wet Tropics, the proportion of seagrass species displaying colonising traits has similarly varied across habitats (Figure 43). In 2018–19 the proportion of seagrass species displaying colonising traits increased at coastal intertidal habitats, suggesting increased levels of physical disturbance. However, at all other habitats the proportion of colonisers remained below the Reef-wide average indicating a recovery trajectory.

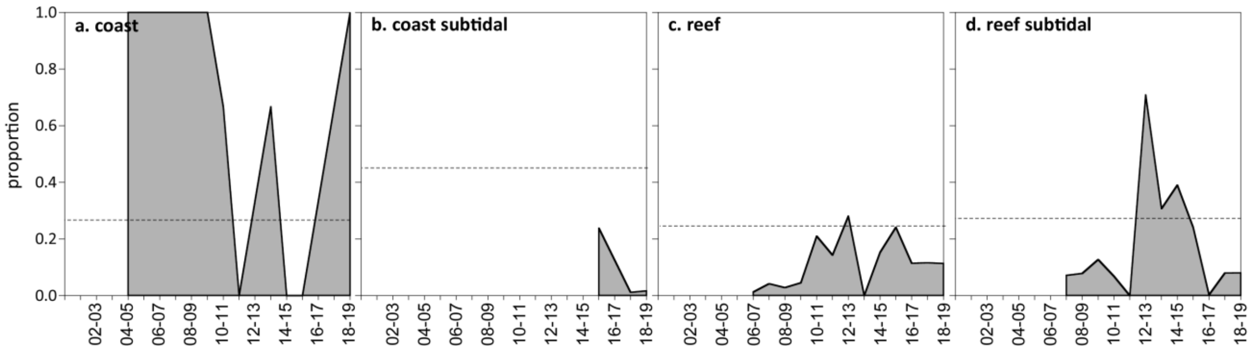


Figure 43. *Proportion of seagrass abundance composed of colonising species at inshore habitats in the southern Wet Tropics region, from the 2000–2001 to the 2018–19 reporting periods. The dashed line represents the Reef-wide average for each habitat type.*

Seagrass meadow extent within all monitoring sites has fluctuated within and between years (Figure 44). At intertidal coastal and reef habitats in the northern Wet Tropics, meadow extent has gradually improved since 2011 and although relatively stable on reefs since 2015, has increased to the greatest extent at coastal habitats. Subtidal reef meadows in the north had been increasing in extent from 2015, but decreased in early 2019 to the smallest area since 2014.

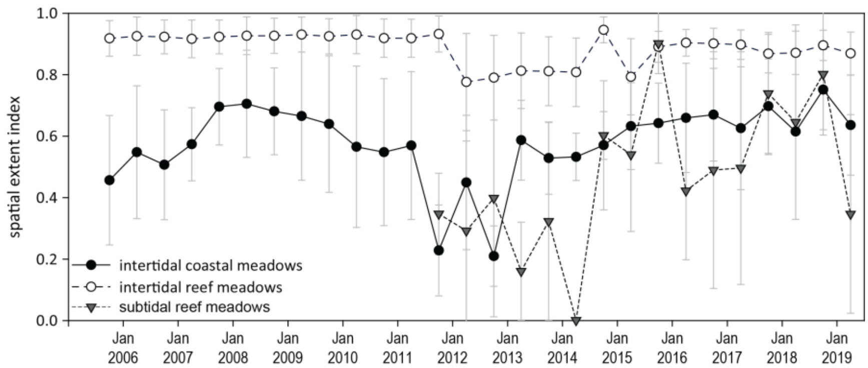


Figure 44. *Change in relative spatial extent (±SE) of seagrass meadows within monitoring sites for each habitat and monitoring period across the northern Wet Tropics NRM region.*

In the southern Wet Tropics, all seagrass meadows were lost in early 2011 as a consequence of cyclone Yasi (Figure 45). Since then, intertidal reef meadows have progressively improved, with the greatest extent since 2011 measured in 2018–19. At intertidal coastal habitats, the meadows have not improved greatly, but a few isolated patches which colonised in mid-2018 appear to have established. The greatest fluctuation in extent has occurred in subtidal reef meadows, which established in 2014, but after rapidly expanding, have sharply declined. In 2018–19, only a few small isolated patches of seagrass remained of the subtidal reef meadows.

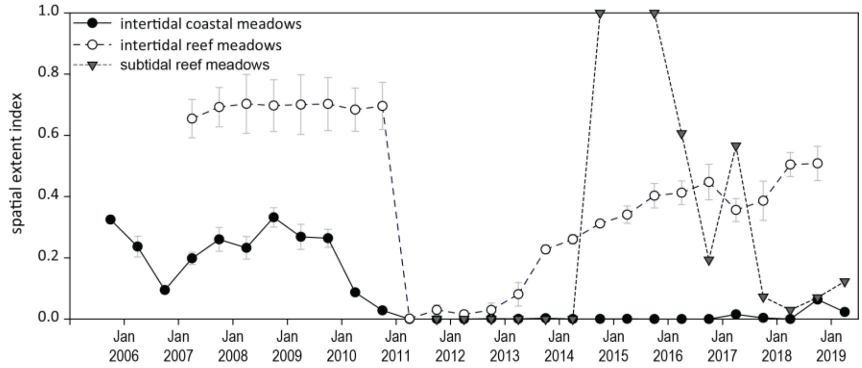


Figure 45. *Change in relative spatial extent (±SE) of seagrass meadows within monitoring sites for each habitat and monitoring period across the southern Wet Tropics NRM region.*

5.2.3.3 Seagrass reproductive status

Reproductive effort varies across habitats in the Wet Tropics, and is higher in the northern sub-region than the south. In the northern Wet Tropics, reproductive effort peaked during 2018–19 in coastal intertidal habitats (Yule Point) (Figure 46). The density of seeds in the coastal seedbank remained higher on average than it has been since 2011, although below historical peaks. At intertidal and subtidal reef habitats reproductive effort remained low but increased slightly from the previous period. To date, seed banks have remained very low across the region in reef habitat (Figure 46). Some possible explanations for the low seed bank include failure to set seed, particularly in low density dioecious species (Shelton 2008), or rapid loss of seeds after release from germination or grazing (Heck and Orth 2006).

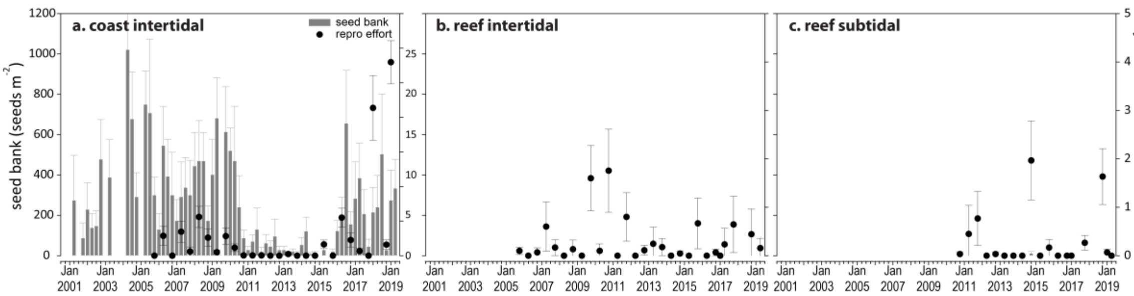


Figure 46. *Reproductive effort and seed banks for inshore intertidal coast and reef habitats in the northern Wet Tropics region, 2001–2019.Seed banks presented as the total number of seeds per m^2 sediment surface (bars ±SE), and reproductive effort presented as the average number of reproductive structures per core (species and sites pooled) (dots ±SE).*

In the southern Wet Tropics, sexually reproductive structures and seed banks were absent from seagrass in the coastal intertidal and reef subtidal habitats (Figure 47). The absence of reproductive structures and seed banks may render the seagrass at risk from further disturbances, as recovery potential remains extremely low without a seed bank. However, the second highest level of reproductive effort recorded in reef intertidal habitats occurred in conjunction with small increases in abundance and extent (Figure 47).

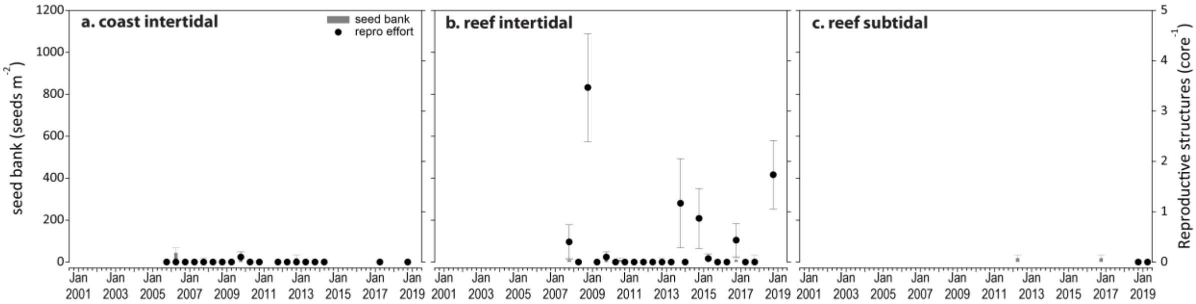


Figure 47. *Reproductive effort and seed banks for inshore intertidal coast and reef habitats in the southern Wet Tropics region, 2001–18.Seed banks presented as the total number of seeds per m^2 sediment surface (bars ±SE), and reproductive effort presented as the average number of reproductive structures per core (species and sites pooled) (dots ±SE).*

5.2.3.4 Seagrass leaf tissue nutrients

Seagrass leaf tissue molar C:N ratios of the foundation seagrass species (in the late dry season 2017) have remained relatively stable across the northern Wet Tropics over the last few years (Figure 48). At intertidal coastal and reef habitats, the ratio has remained below the guideline value (20) and C:N ratios at the coastal sites were lower than other habitats in the north (Figure 48). This indicates that nitrogen loads are in excess of growth requirements, due possibly to elevated N or light limitation. High N:P ratios and %N in coastal habitats (Figure 48) also provides evidence of excess nitrogen loads at these sites, however, both values appear to have been declining over the past 4 to 5 years. Seagrasses in subtidal reef habitats had higher leaf molar C:N ratios than those in intertidal habitats, and higher leaf C:P ratios (Figure 48). The slightly higher $\delta^{15}N$ values at coastal habitats suggests multiple sources of nitrogen, possibly including some anthropogenic point sources (Figure 48).

In the southern Wet Tropics, similar to the northern sub-region, C:N ratios at coastal sites were lower than other habitats, indicating nitrogen loads in excess of growth requirements; similarly supported by higher N:P ratios and %N (Figure 49). At the reef habitats, seagrass leaf tissue molar C:N ratios of the foundation seagrass species (in the late dry season 2018) have remained relatively stable below the guideline value (20) over the last few years (Figure 49). Similar to the north, this indicates that nitrogen loads are in excess of growth requirements, due possibly to elevated N and/or light limitation. The low C:N ratios combined with %N and N:P values marginally above the global guidelines, suggest some level of excess N. Also, lower $\delta^{13}C$ suggests some degree of light limitation, particularly in coastal and subtidal reef habitats (Figure 49). The range $\delta^{15}N$ values below 4 across all habitats suggests multiple sources of nitrogen, possibly including some anthropogenic point sources (Figure 49). However, there is currently no indication or concern that anthropogenic point sources are strongly influencing seagrass N supply overall.

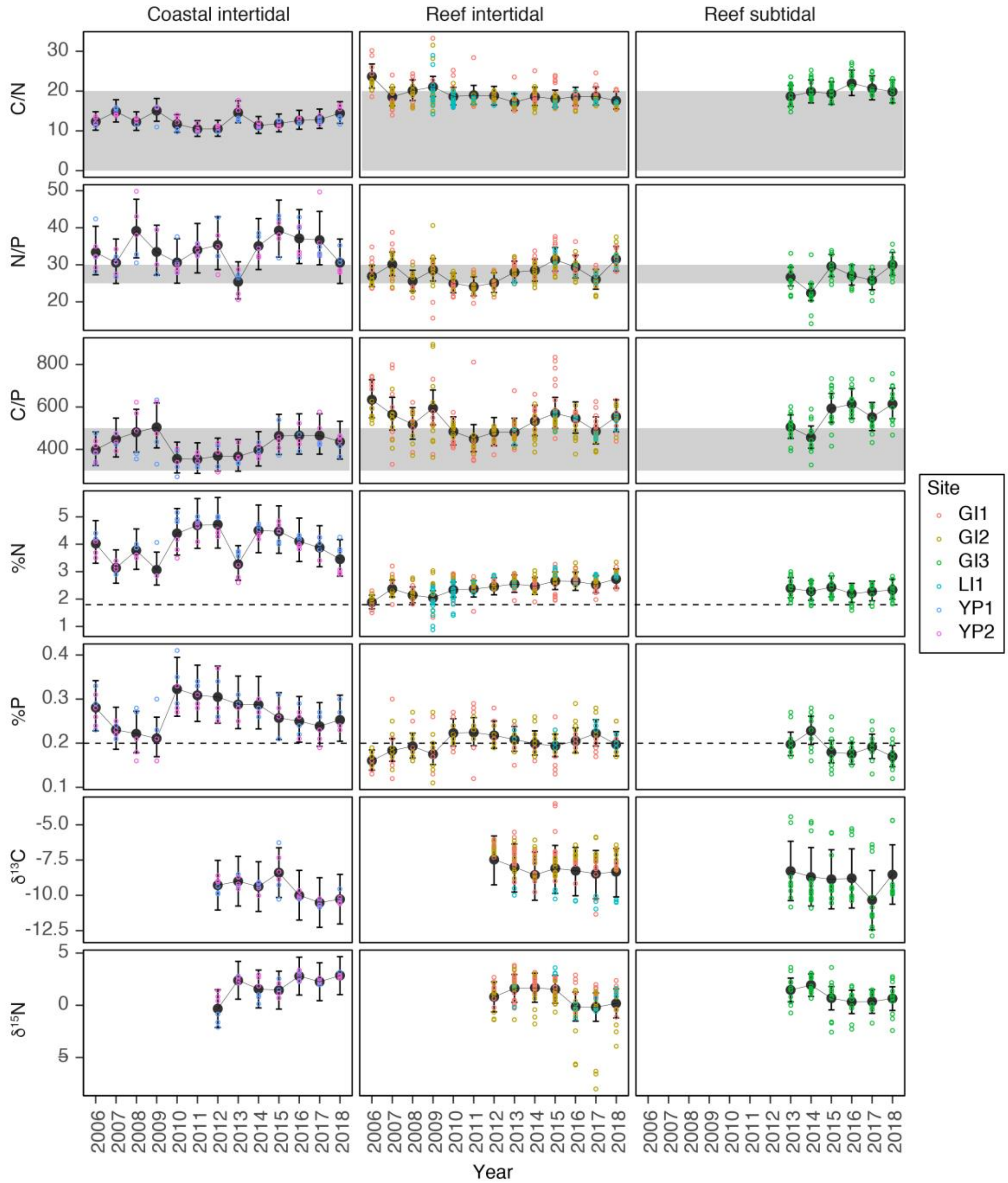


Figure 48. *Seagrass leaf tissue nutrient elemental ratios (C:N:P) and concentrations (%N, %P, $\delta^{13}C$ and $\delta^{15}N$) for each habitat in the northern Wet Tropics region (± SE) (foundation species pooled). Horizontal shaded bands or dashed lines represents the accepted seagrass guideline values, where: C:N ratios within the band may indicate reduced light availability and/or N enrichment; N:P ratios above the band indicate P limitation, below indicate N limitation and within indicates replete, and; C:P ratios within the band may indicate nutrient rich habitats (large P pool). Dashed lines in %N and %P indicate global median values of 1.8% and 0.2% for tissue nitrogen and phosphorus, respectively (Duarte 1990).*

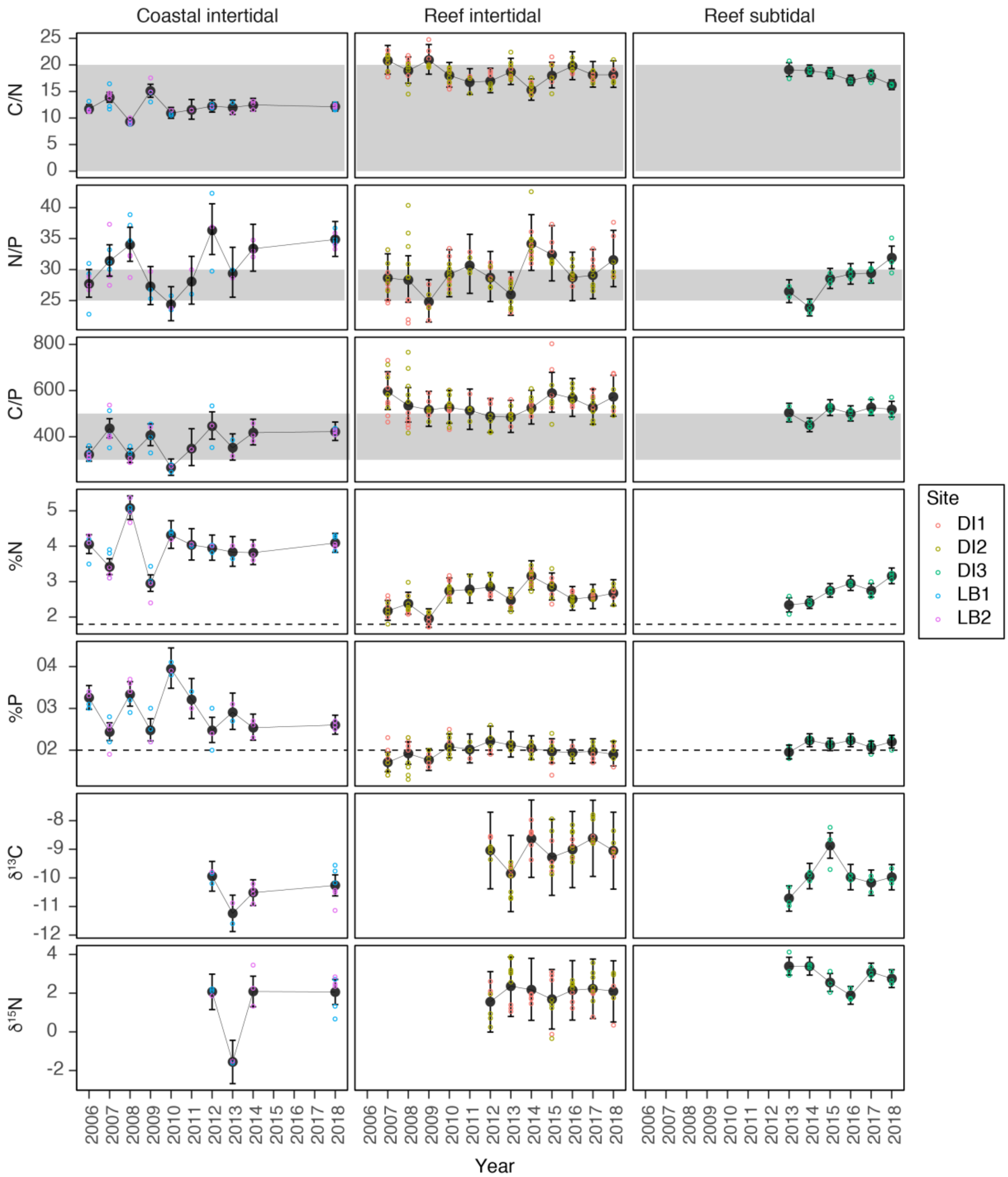


Figure 49. *Seagrass leaf tissue nutrient elemental ratios (C:N:P) and concentrations (%N, %P, δ^{13}C and δ^{15}N) for each habitat in the southern Wet Tropics region (± SE) (foundation species pooled). Horizontal shaded bands or dashed lines represents the accepted seagrass guideline values, where: C:N ratios within the band may indicate reduced light availability and/or N enrichment; N:P ratios above the band indicate P limitation, below indicate N limitation and within indicates replete, and; C:P ratios within the band may indicate nutrient rich habitats (large P pool). Dashed lines in %N and %P indicate global median values of 1.8% and 0.2% for tissue nitrogen and phosphorus, respectively (Duarte 1990).*

5.2.3.5 Epiphytes and macroalgae

Epiphyte cover on seagrass leaf blades has historically been higher in the wet season across all habitats in the Wet Tropics region (Figure 50). Epiphyte cover remained above the Reef-wide long-term average across all intertidal habitats in the northern Wet Tropics in 2018–19 (Figure 50), but below average in the subtidal habitats.

Macroalgae cover was lower than the Reef long-term average in all habitats in the wet season (Figure 50). Macroalgae cover was also below the Reef long-term average in the dry season, except in reef intertidal habitats where it was slightly above-average.

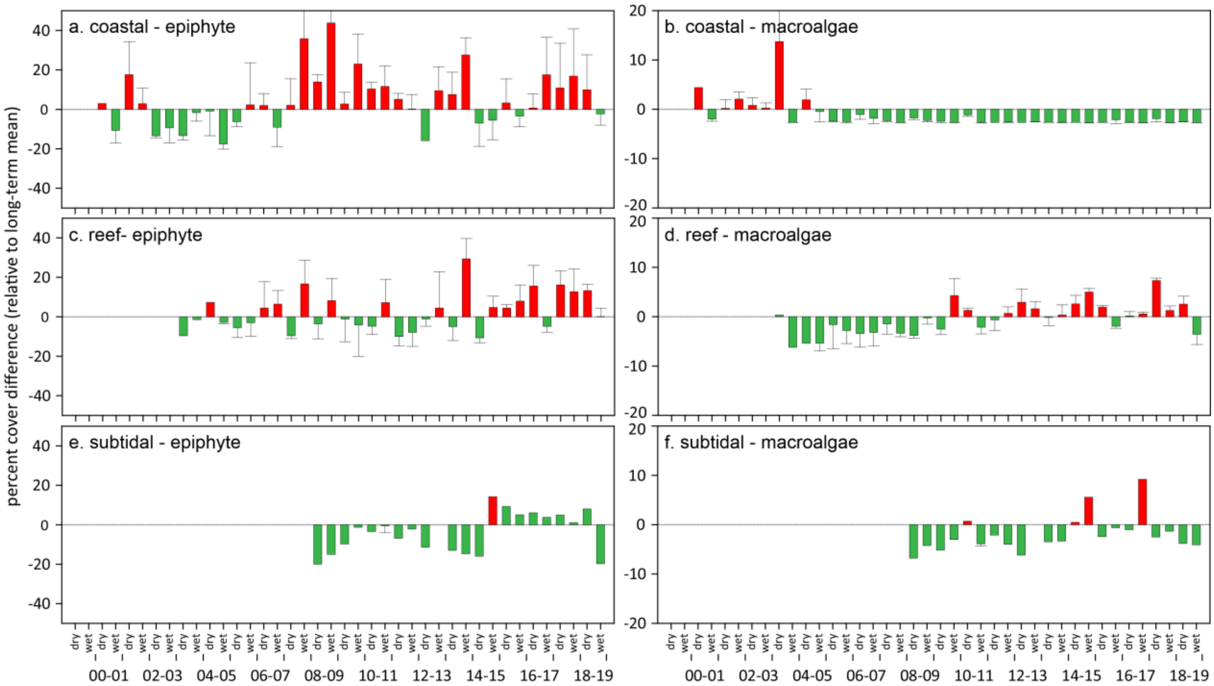


Figure 50. *Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average for each inshore seagrass habitat in the northern Wet Tropics region, 2001–2019 (sites pooled, ±SE). Red/green words*

In the southern Wet Tropics, epiphyte cover in 2018–19 was below the Reef long-term average in the wet season, but above the Reef long-term average in the dry season at reef habitats (Figure 50).

Macroalgae cover continued to remain below the Reef long-term average for the tenth year at intertidal reef habitats, and was also below average at subtidal reef habitats. Macroalgae cover remained near absent at coastal habitats.

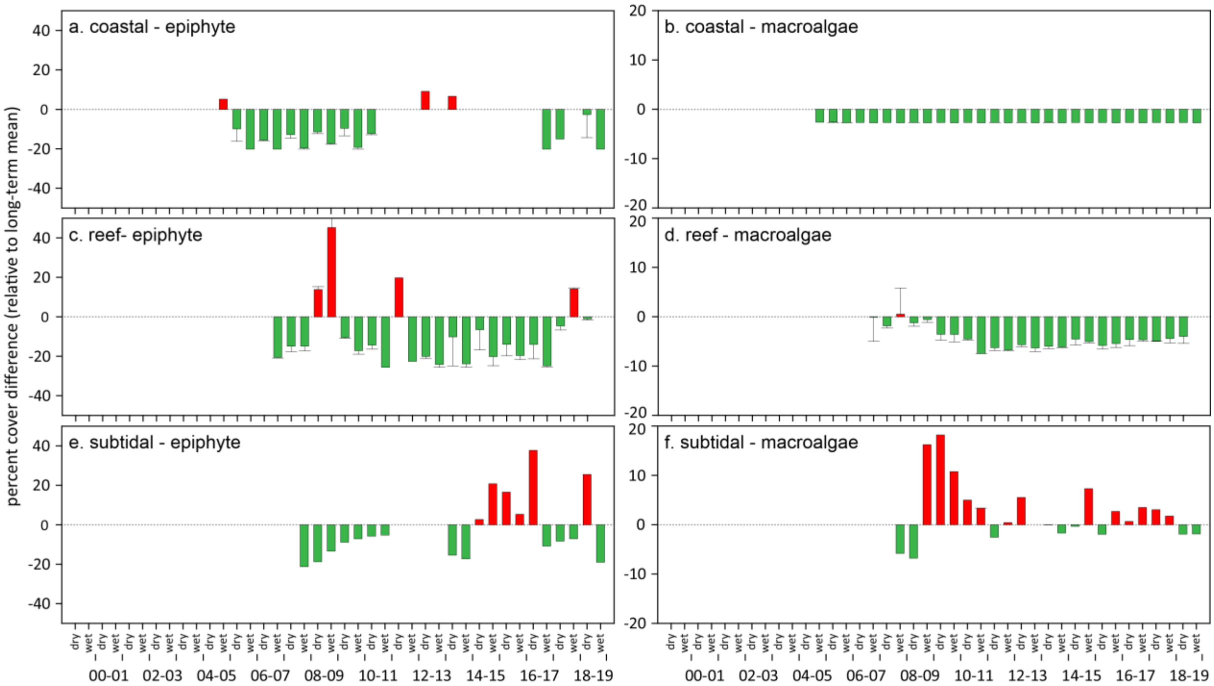


Figure 51. *Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average for each inshore seagrass habitat in the southern Wet Tropics region, 2001–19 (sites pooled, ±SE).*

5.3 Burdekin

5.3.1 2018–19 Summary

Seagrass meadows across the Burdekin NRM region declined in overall condition in 2018–19 from moderate to **poor** (Figure 52). All condition indicators declined:

- abundance score was moderate
- reproductive effort score was very poor
- tissue nutrient score was moderate.

Seagrass abundance decreased relative to the previous period, due to declines in per cent cover at three quarters of sites, with the largest declines found in reef subtidal habitats. The declines in abundance were likely the result of river discharge events from the Burdekin River as well as unusually large events from the smaller rivers discharging into Cleveland Bay. Sediment loads in the discharge and resuspension elevates turbidity and reduces benthic light during the wet season. However, these events followed a period of unusually high light levels during the growing season (September to November) of 2018, leading to above-average light levels on average for the whole year. There was a return to average water temperature following four consecutive years when intertidal within-canopy temperatures were above the long-term average.

Reproductive effort was low across Burdekin region habitats in 2018–19. Reproductive effort and seed banks declined at coastal and reef subtidal sites, particularly in the wet season, and remained low at reef intertidal sites. There has been continued decline in this indicator score since 2013–14, with a large decline from poor to very poor 2018–19. Seed densities in the seed bank of the coastal intertidal and reef subtidal habitat were the lowest recorded since 2014 and 2015, respectively.

There has been a declining trend in the tissue nutrient indicator score since 2013–14 except for a small rise in 2016–17. In 2018-19, small declines in C/N occurred in all habitats. This was due to elevated nitrogen content in leaves at reef subtidal habitats and reduced carbon in leaves at coastal and reef intertidal habitats.

Over the past decade, seagrass meadows of the Burdekin region have demonstrated high resilience particularly through their capacity for recovery. This may reflect a conditioning to disturbance (high seed bank, high species diversity), but also reflects the nature of the disturbances which are episodic and dominated by wind events and Burdekin River flows.

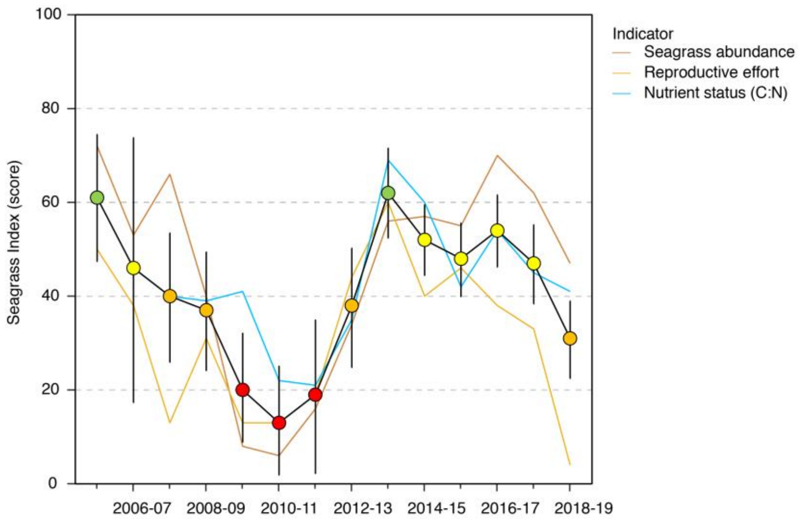


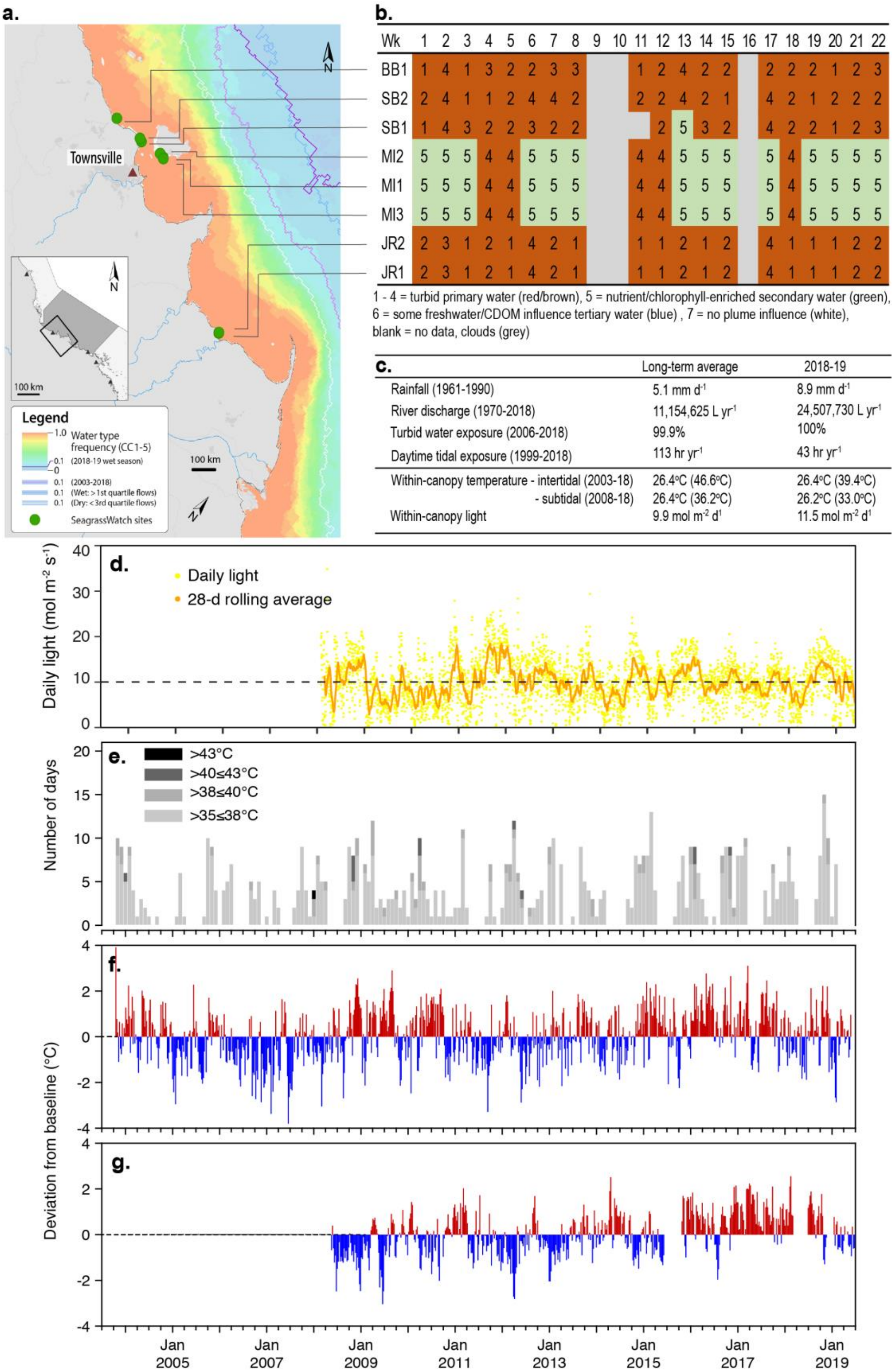
Figure 52. *Report card of seagrass status indicators and index for the Burdekin NRM region (averages across habitats and sites). Values are indexed scores scaled from 0–100 and graded: ● = very good (81-100), ● = good (61 - 80), ● = moderate (41 - 60), ● = poor (21 - 40), ● = very poor (0 - 20). NB: Scores are unitless.*

5.3.2 Climate and environmental pressures

In 2018–19, rainfall and river discharge were above the long-term median for all of the basins in the Burdekin region (Figure 53, Table 9). In most rivers, the discharge events were more than 3 times the long-term median. The most significant and unusually high discharge events (occurring only a few times in every 100 years) occurred in February 2019 in the Ross River, which passes through the city of Townsville and discharges into Cleveland Bay, and the Haughton River discharging into Bowling Green Bay. The Burdekin River was also 3 times above the long-term median discharge. The volume of water and sediment from the Burdekin River far exceeded each of the small rivers by almost an order of magnitude, but such an event is more common for this river (occurring about 10 times in every 100 years). All seagrass monitoring sites in the Burdekin Region are affected by these rivers and the sediment and nutrient loads they carry.

Exposure of inshore seagrass to turbid waters during the wet season was at the long-term average. All sites monitored throughout the region were exposed to ‘brown’ or ‘green’ turbid water for the entire wet season (100% frequency of exposure). Coastal sites (BB, SB and JR) experienced the highest exposure to ‘brown’ turbid, sediment laden, waters (94–100% of wet season weeks categories 1–4, 100% of categories 1–5), while the remaining reef sites were exposed predominately to ‘green’, phytoplankton rich waters for 100% of weeks (categories 1–5) (Figure 53).

Daily light levels in the Burdekin region are below 10 mol m^{-2} d^{-1} on average. In 2018–19, they were higher than average due to high light levels during the latter part of 2018 when there was a prolonged high light period before the wet season rainfall and river discharge. During the wet season when there were flood events, some light loggers failed to record. From those sites that we have data, we can see an extended period of low light during the wet season (Figure 102). These sites were at reef intertidal sites which tend to have the higher light levels amongst sites in the region, therefore leading to an over inflation of the region-wide light average for the wet season, and an over inflation in the regional average for the year. Light limitation during the wet season is very likely to have contributed to reductions in seagrass abundance and the score for the region.



Wk	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22
BB1	1	4	1	3	2	2	3	3			1	2	4	2	2		2	2	2	1	2	3
SB2	2	4	1	1	2	4	4	2			2	2	4	2	1		4	2	1	2	2	2
SB1	1	4	3	2	2	3	2	2				2	5	3	2		4	2	2	1	2	3
MI2	5	5	5	4	4	5	5	5			4	4	5	5	5		5	4	5	5	5	5
MI1	5	5	5	4	4	5	5	5			4	4	5	5	5		5	4	5	5	5	5
MI3	5	5	5	4	4	5	5	5			4	4	5	5	5		5	4	5	5	5	5
JR2	2	3	1	2	1	4	2	1			1	1	2	1	2		4	1	1	1	2	2
JR1	2	3	1	2	1	4	2	1			1	1	2	1	2		4	1	1	1	2	2

	Long-term average	2018-19
Rainfall (1961-1990)	5.1 mm d^{-1}	8.9 mm d^{-1}
River discharge (1970-2018)	11,154,625 L yr^{-1}	24,507,730 L yr^{-1}
Turbid water exposure (2006-2018)	99.9%	100%
Daytime tidal exposure (1999-2018)	113 hr yr^{-1}	43 hr yr^{-1}
Within-canopy temperature - intertidal (2003-18)	26.4°C (46.6°C)	26.4°C (39.4°C)
- subtidal (2008-18)	26.4°C (36.2°C)	26.2°C (33.0°C)
Within-canopy light	9.9 mol m^{-2} d^{1}	11.5 mol m^{-2} d^{1}

Figure 53. *Environmental pressures in the Burdekin region including: a. frequency of exposure to turbid water (colour classes 1–5) (from Gruber et al. 2019b); b. wet season water type at each site; c. average conditions over the long-term and in 2018–19; d. daily light and the 28-day rolling mean of daily light for all intertidal sites; e. number of days intertidal site temperature exceeded 35°C, 38°C, 40°C and 43°C, and; f. deviations from 13-year mean weekly temperature records.*

This year intertidal within-canopy temperatures were similar to the long-term average (Figure 53). Maximum intertidal within-canopy temperatures exceeded 35°C for a total of 44 days during 2018-19, with the highest temperature recorded at 39.4°C (MI2, 2pm 03Dec18). 2018–19 was the first time in four years annual subtidal temperatures were below the long-term average. Maximum subtidal temperature during 2018–19 was 33.0°C (midday 04Dec18).

Daily tide exposure was below the long-term average for the third consecutive year at all sites (Figure 53, Figure 94, Figure 95), which may have provided some respite from the elevated temperatures.

The proportion of mud at Jerona (Barratta Creek) coastal meadows was much higher than Townsville meadows (Bushland Beach and Shelley Beach) and has remained well above the Reef long-term average (Figure 111). Post 2011, Townsville coastal meadows have been dominated by fine sediments, although the proportion of mud increased at Bushland Beach 2017-18, it has since returned to its previous state (Figure 111). Conversely, reef habitats, which were dominated by coarser sediment from 2012 to 2017, have since gradually increased in composition of fine sand and mud. More fine sediments were present at the Cockle Bay (MI2) than the Picnic Bay (MI1) reef habitat meadows (Figure 112, Figure 113).

5.3.3 Inshore seagrass and habitat condition

There are 8 seagrass monitoring sites in the Burdekin from 5 locations (Table 13). Three seagrass habitat types were assessed across the region in 2018–19 (Table 13, Table 18).

Table 13. *List of data sources of seagrass and environmental condition indicators for each seagrass habitat type in the Burdekin NRM region. *Seagrass-Watch. For site details see Table 2 and Table 3.*

Habitat	Site code and location		seagrass abundance	seagrass composition	seagrass distribution	reproductive effort	seed banks	leaf tissue nutrients	meadow sediments	epiphytes & macroalgae
coastal intertidal	SB1	Shelley Beach (Townsville)	■	■	■	■	■	■	■	■
	SB2*	Shelley Beach (Townsville)	■	■			■		■	■
	BB1	Bushland Beach (Townsville)	■	■	■	■	■	■	■	■
	JR1	Jerona (Barratta CK, Bowling Green Bay)	■	■	■	■	■	■	■	■
	JR2	Jerona (Barratta CK, Bowling Green Bay)	■	■	■	■	■	■	■	■
reef intertidal	MI1	Picnic Bay (Magnetic Island)	■	■	■	■	■	■	■	■
	MI2	Cockle Bay (Magnetic Island)	■	■	■	■	■	■	■	■
reef subtidal	MI3	Picnic Bay (Magnetic Island)	■	■	■	■	■	■	■	■

5.3.3.1 Seagrass index and indicator scores

In the 2018-19 monitoring period, the seagrass condition index for the Burdekin region declined from moderate to **poor** (Figure 54). Over the previous four monitoring periods the index has changed little, increasing and subsequently decreasing, but at a relatively insignificant level. The large change in 2018–19 is most likely the result of the region-wide above average rainfall and river discharge during the wet season. All indicators declined, but the largest declines occurred in seagrass abundance and reproductive effort, which declined by a whole grade to moderate and very poor, respectively (Figure 54). These are measured before and after the wet season, therefore those scores reflect the effect of wet season river discharges, while tissue nutrients are measured in the previous growing season.

Examination of contributing seagrass condition indicators over the long-term, show declines from 2009–2011 as a consequence of the years of above-average rainfall and severe weather, proceeded by rapid recovery. Based on those previous trends, the seagrass meadows in 2018–19 would appear to be in a vulnerable state and at risk of further decline (Figure 54).

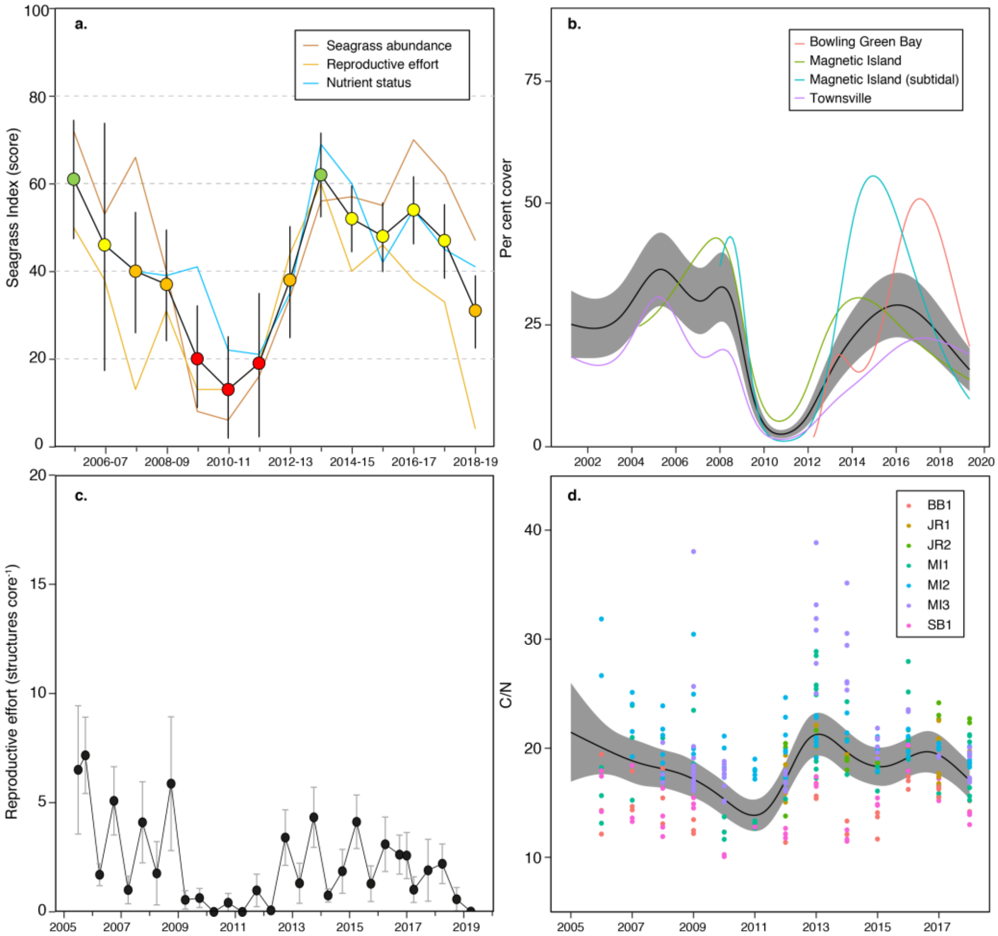


Figure 54. *Temporal trends in the Burdekin seagrass condition index and the indicators used to calculate the index: a. seagrass condition index (circles) and indicator trends (lines); b. GAM plots of seagrass abundance (per cent cover) trends for each location (coloured lines) and the region (black line with grey shaded area defining 95% confidence intervals); c. average number of reproductive structures (±SE) (GAM not possible due to high count of zero values); and d. elemental ratios (atomic) of leaf tissue C:N nutrient content at each site (coloured circles) and regional trend represented by a GAM plot as dark line with shaded areas defining 95% confidence intervals of the trend*

5.3.3.2 Seagrass abundance, composition and extent

Over the duration of the MMP, seagrass abundance in the Burdekin region has shown a pattern of loss and recovery. Losses occurred as a result of multiple consecutive years of above-average rainfall (river discharge) and severe weather (cyclone Yasi) between 2008–09 and 2010–11. From 2011, seagrass rapidly recovered, however since 2014, seagrass abundance has progressively declined at reef (intertidal and subtidal) habitats. In 2018–19, the largest declines occurred in reef subtidal and coastal intertidal habitats. Three-quarter of Burdekin region sites declined in abundance in 2018–19, which was predominately at reef subtidal and coastal intertidal habitats.

An examination of the long-term abundances across the Burdekin region indicates no significant trend, although significant trends were detected at two of the five coastal sites. One site (SB2), which has been monitored for nearly two decades (since 2001), showed a decreasing trend (Table 20). The other site (JR2), near Jerona (Barratta Creek, Bowling Green Bay), has only been monitored since 2012, and not surprisingly showed a significant increasing trend in abundance, as this coincides with the main recovery period after the regional losses.

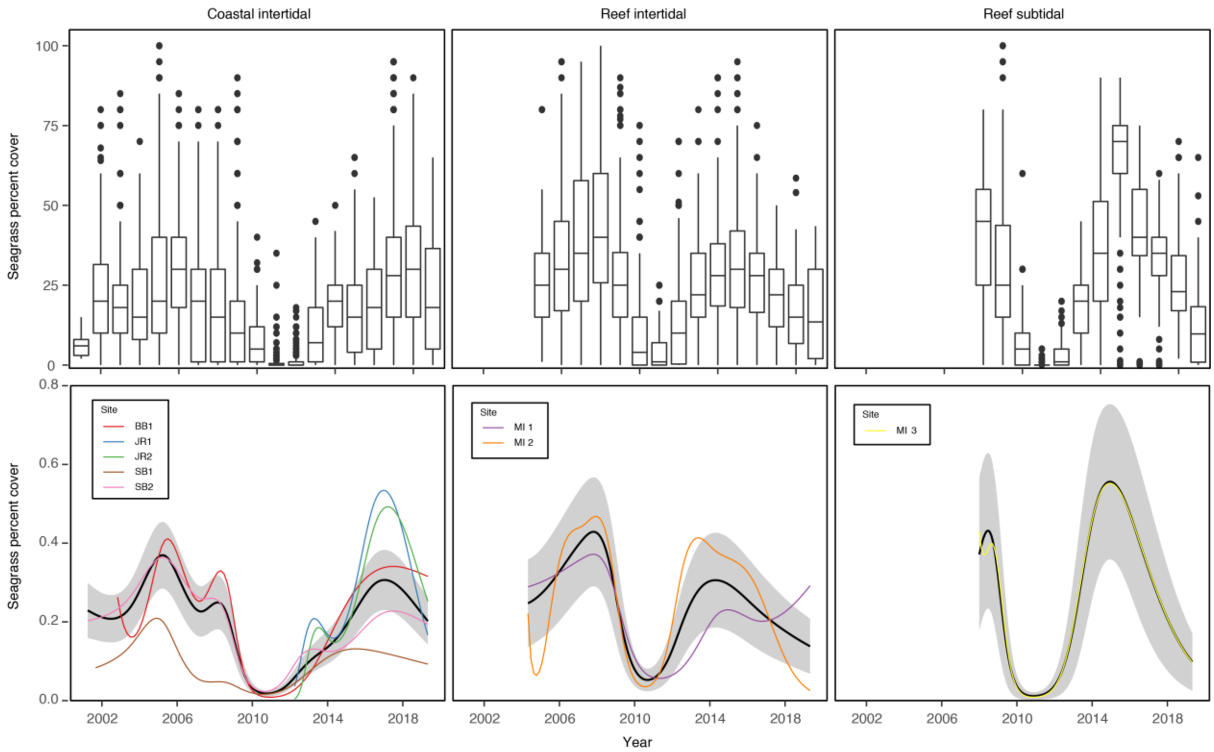


Figure 55. *Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the Burdekin NRM region from 2001 to 2019. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates the 25th percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75th percentile. Whiskers (error bars) above and below the box indicate the 90th and 10th percentiles, and the dots represent outlying points. GAMM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.*

This year, as it has been since 2014–2015, a low proportion of species displaying colonising traits are present in all habitats (e.g. *Halophila ovalis*). Instead these habitats are dominated by opportunistic species (*H. uninervis, Z. muelleri, C. serrulata*) in coastal and reef sites or persistent species in intertidal reef habitat (*T. hemprichii*). Opportunistic and persistent foundation species also have a capacity to resist stress (survive, through reallocation of resources) caused by acute disturbances (Collier *et al.* 2012b), and therefore, current species composition provides greater overall resilience in Burdekin meadows. However, following on from seagrass loss observed during this reporting period, there may be an increase in the proportion of colonising species during future surveys.

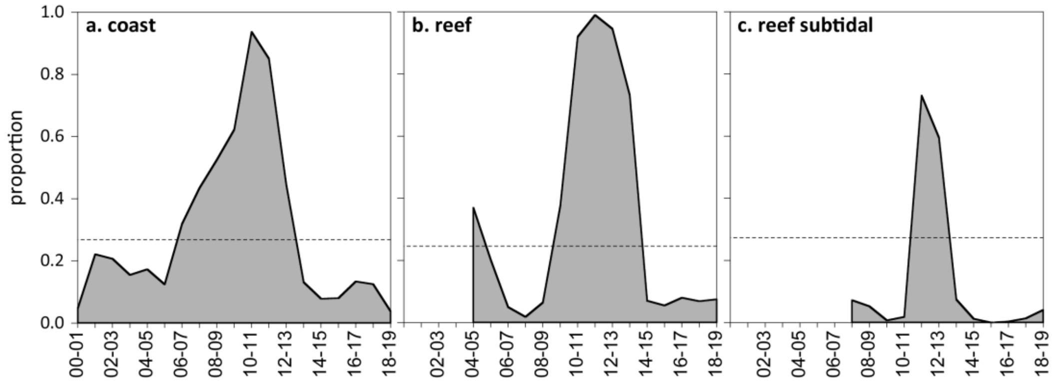


Figure 56. *Proportion of seagrass abundance composed of colonising species at inshore habitats in the Burdekin region, 2001–2019. Grey area represents Reef long-term average proportion of colonising species for each habitat type.*

Seagrass meadow extent declined in all habitat types in 2018–19 (Figure 57). This follows small declines in intertidal coastal and subtidal reef habitats since 2017. In the two to three years prior to 2011, significant changes occurred across the region with all seagrass meadows reducing in size and changing in landscape from continuous, to patchy, to isolated patches and finally to isolated shoots with the loss of meadow cohesion (Figure 57). That trend was also replicated at the bay-wide scale in Cleveland Bay, with considerable loss of meadow area and meadow fragmentation (Petus *et al.* 2014). This was caused by the high rainfall and riverine discharge that affected much of the Reef.

Since 2011, meadow extents have increased in both coastal and reef habitats to pre-2009 levels (Figure 57) and predominately remained stable until 2017–18. In early 2014, subtidal seagrass extent declined to the lowest in 2 years but subsequently recovered within 6 months to its maximum extent. In 2018–19, the subtidal reef seagrass extent again declined, but unlike in 2014, the coastal intertidal meadows have also declined in extent due to a proliferation of scarring and fragmentation.

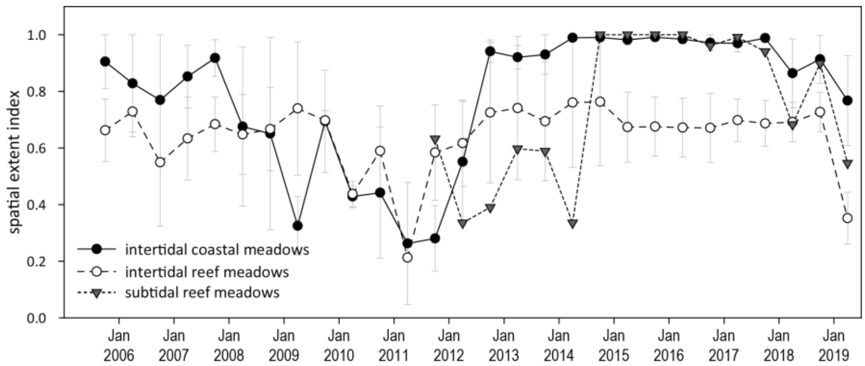


Figure 57. *Change in spatial extent of seagrass meadows within monitoring sites for each inshore intertidal habitat and monitoring period across the Burdekin region, 2005–2018.*

5.3.3.3 Seagrass reproductive status

Reproductive effort is typically variable across Burdekin region habitats, but in 2018–19, reproductive effort was very low in all habitats, particularly during post-wet season surveys. Coastal habitats had been on an increasing trajectory since 2012, with high reproductive effort contributing to a developing seed bank, but this year reached low levels not seen since 2014. At reef subtidal habitats seed densities sharply declined, and with no reproductive structures observed in 2018–19, there is no sign of replenishment. At reef intertidal habitats, reproductive effort and seed banks remained low but within range of what is typically observed in this habitat (Figure 58). Seagrass meadows of the region are dynamic, going through periods of decline and recovery. Recovery from seed is one of the important ways that meadows can recovery following disturbances, and a low seed bank raises concern for their current level of resilience to any disturbances that may arise into the immediate future.

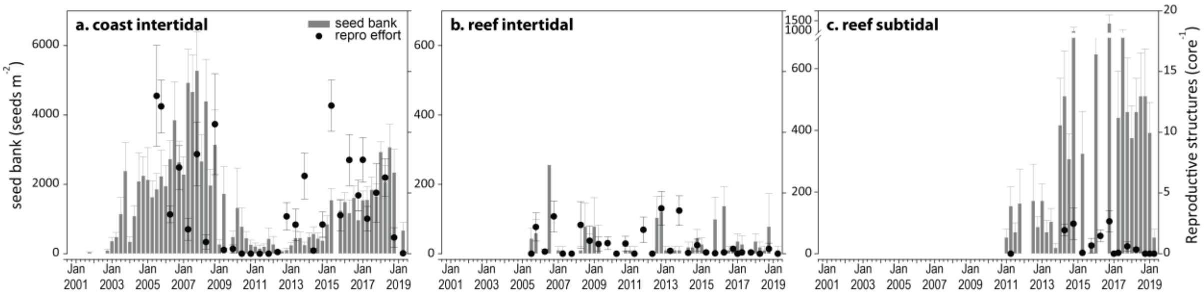


Figure 58. *Reproductive effort at inshore intertidal coast and reef and subtidal reef habitats in the Burdekin region. Seed bank presented as the total number of seeds per m^2 sediment surface (bars ±SE), and late dry season reproductive effort presented as the average number of reproductive structures per core (species and sites pooled) (dots ±SE). NB: Y-axis scale for seed banks differs between habitats.*

5.3.3.4 Seagrass leaf tissue nutrients

Seagrass leaf tissue molar C:N ratios declined slightly in 2018–19 (Figure 59). This follows a declining trend in reef subtidal habitats since the high levels observed in 2012–13. At coastal intertidal and reef intertidal habitats C:N varies slightly, but usually in a manner that reflects local processes, in particular nitrogen loads and light levels (see Case Study 1). The C:N ratios declined to a low in 2010–11 following extreme weather events, then recovered to a maxima in 2016–17 as seagrass recovered. A decline in C:N indicates that nitrogen is in supply at a rate in excess of growth requirements (i.e. C limited). The tissue nutrients are measured during the growing season in September to November, so a further reduction in the score is likely to occur in the following reporting period as a reflection of the 2018–19 wet season. The trends at reef intertidal habitats are similar to those observed in coastal habitats. The %P and N:P values were relatively stable in both of the intertidal habitats, but an ongoing trend for increasing N:P at reef subtidal habitats since 2013–14 has continued in 2018–19 (Figure 59).

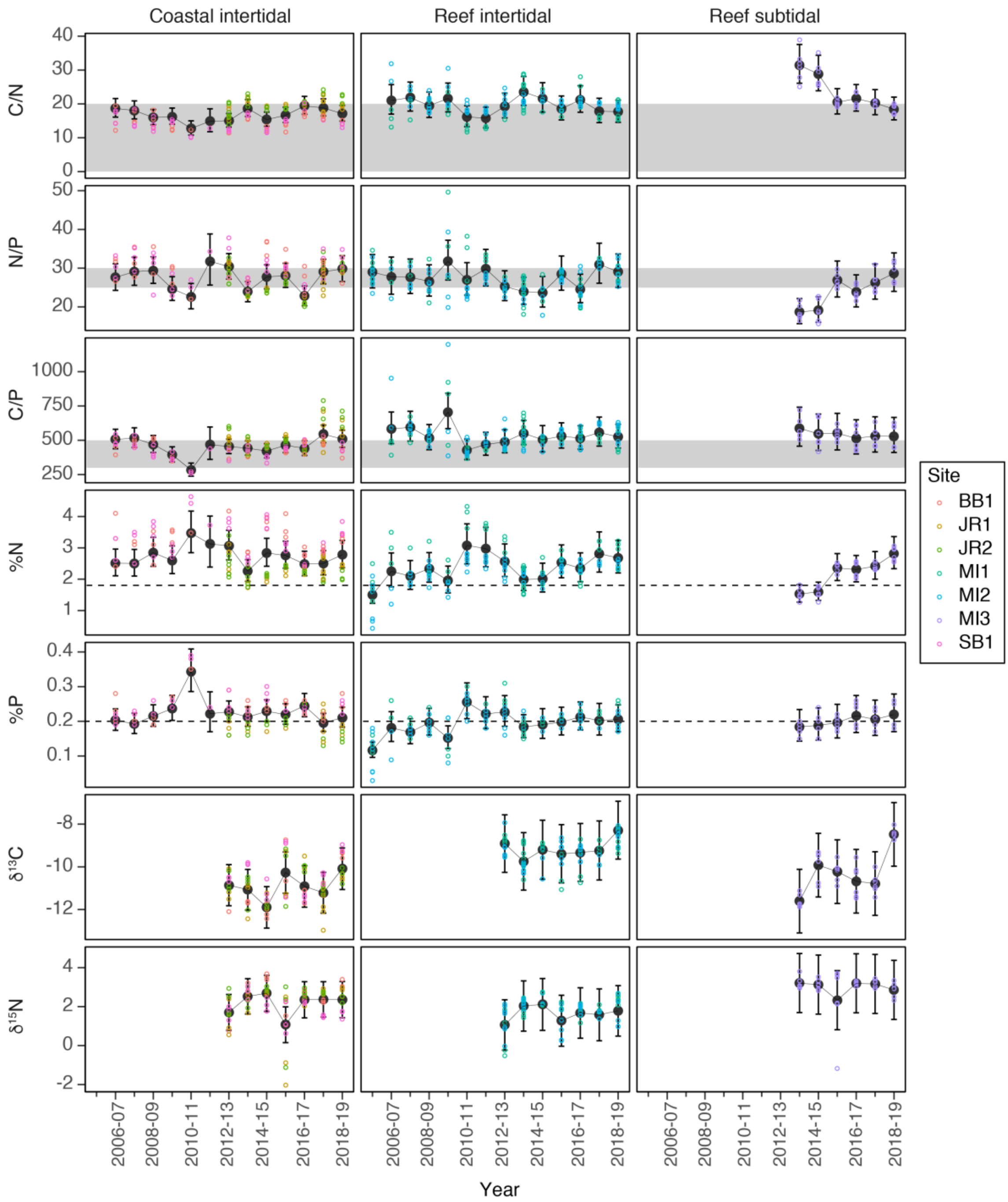


Figure 59. *Seagrass leaf tissue nutrient elemental ratios (C:N:P) and concentrations (%N, %P, δ^{13}C and δ^{15}N) for each habitat in the Burdekin region (± SE) (foundation species pooled). Horizontal shaded bands or dashed lines represents the accepted seagrass guideline values, where: C:N ratios within the band may indicate reduced light availability and/or N enrichment; N:P ratios above the band indicate P limitation, below indicate N limitation and within indicates replete, and; C:P ratios within the band may indicate nutrient rich habitats (large P pool). Dashed lines in %N and %P indicate global median values of 1.8% and 0.2% for tissue nitrogen and phosphorus, respectively (Duarte 1990).*

5.3.3.5 Epiphytes and macroalgae

Epiphyte cover on seagrass leaf blades differs between the wet and dry season at coastal sites, but there is not a strong seasonal trend in other habitats. (Figure 60). In reef habitats, epiphyte abundance had been above the Reef-wide long-term average for the last few years, but declined in 2018–19 in both the dry and wet in intertidal habitats and in the wet season for subtidal habitats. Declines in epiphyte abundance have been observed following previous above-average discharge (e.g. 2010–11), which may reflect their sensitivity to turbidity and light limitation. It might also reflect declining salinity for sites affected by freshwater, including those reef sites at Magnetic Island, which can be affected by very localised discharge. Both epiphytes and macroalgae cover can increase following nutrient enrichment (Cabaço *et al.* 2013; Nelson 2017); however, due to complex ecological and biological factors (e.g. grazing Heck and Valentine 2006), their abundance may not necessarily correlate to nutrient loading.

Macroalgae abundance has remained low and below the long-term average at coastal habitats, and returned to levels at/or around the long-term mean however abundances have increased over the last two monitoring periods at reef (intertidal and reef) habitats.

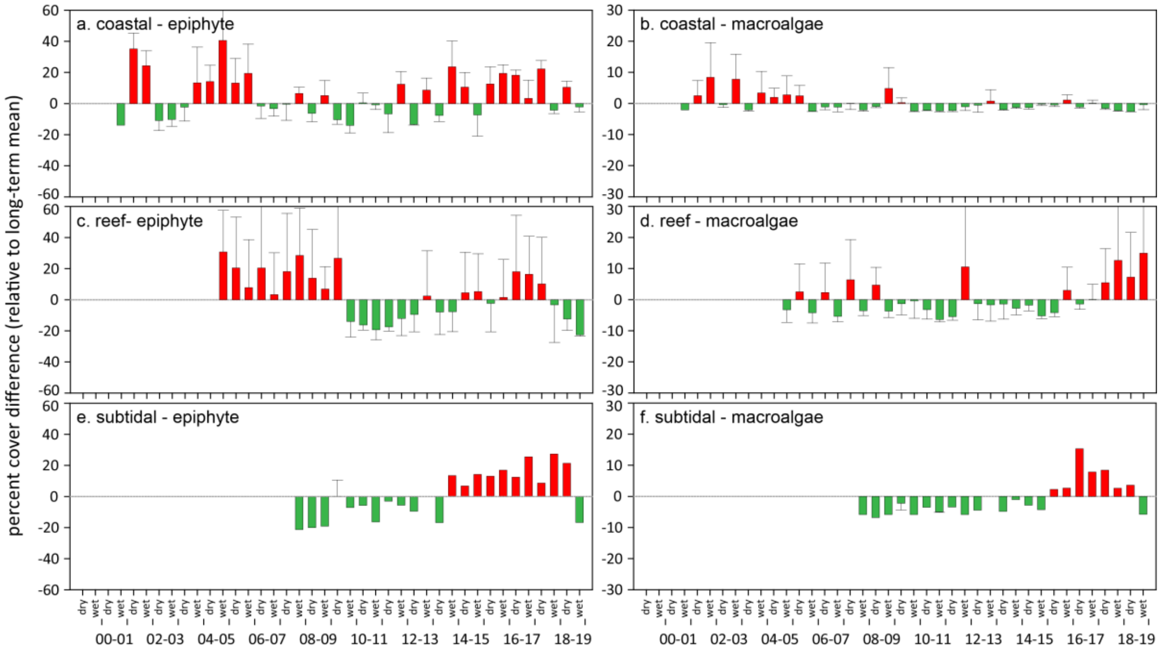


Figure 60. *Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term Reef average for each inshore seagrass habitat in the Burdekin region (sites pooled, ±SE). Red/green text*

5.4 Mackay–Whitsunday

5.4.1 2018–19 Summary

Inshore seagrass meadows across the Mackay–Whitsunday NRM region were relatively stable in overall condition in 2018–19, with the condition grade remaining **poor** (Figure 61). There was a small increase in seagrass abundance and the tissue nutrient score in 2018–19, but these gains were offset by further reduction in reproductive effort. Indicators for the overall condition score were:

- abundance score was poor
- reproductive effort score was very poor
- tissue nutrient score was poor.

Two-thirds of sites increased in abundance, including all subtidal sites. The greatest losses occurred in estuarine habitats where all sites decreased in abundance and extent. Other losses observed occurred in 33% and 25% of coastal and reef intertidal sites, respectively. The long-term trend indicates a declining trajectory with a region struggling to recover from losses in the years leading up to 2010–11 and in early 2017.

Seagrass reproductive effort declined at coastal habitats, and remained virtually absent in reef habitat. Reproductive effort at the estuarine site is highly variable both inter-annually and seasonally, but there are usually some reproductive structures observed in the dry season. In 2018–19 reproductive structures were observed in estuarine habitats in the dry season, but at the second lowest level ever recorded (with the lowest being zero). Despite the decline in reproductive effort, seeds are persisting within the seed bank of all habitats, which provides an indication of ability to recover from future impacts.

Despite a small increase in the leaf tissue nutrient score, tissue nutrients remained relatively unchanged in 2018–19, compared to 2017–18. In all habitats, the score remains below the threshold of 20, indicating that nitrogen occurs in excess of growth requirements at the Mackay–Whitsunday sites.

The Mackay–Whitsunday regional seagrass condition had been improving since 2010–2011, when it reached its lowest level since monitoring commenced. However, by 2016–2017, the recovery trend abated as a consequence of cyclone Debbie, and the region continues to struggle to recover from the legacy of the disturbance events. Moderate rainfall and discharge events, as well as near average water temperatures in 2018–19 are conditions that could support the onset of recovery in coming years.

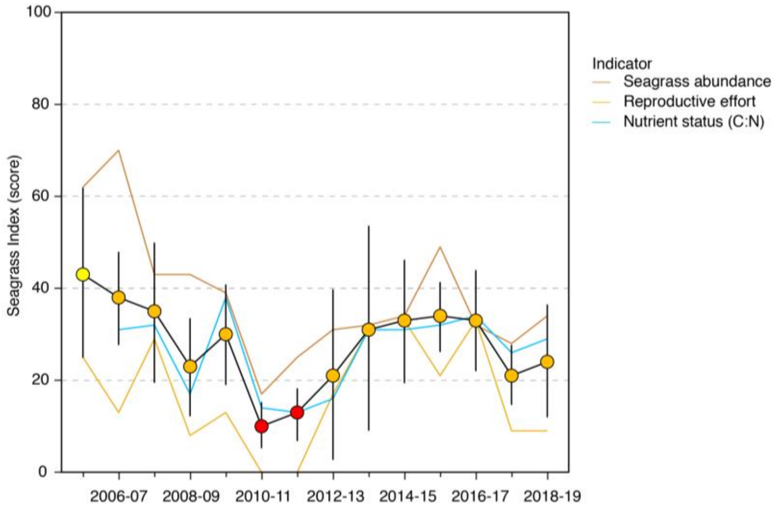


Figure 61. *Report card of seagrass status indicators and index for the Mackay–Whitsunday NRM region (averages across habitats and sites). Values are indexed scores scaled from 0–100 and graded: ● = very good (81-100), ● = good (61 - 80), ● = moderate (41 - 60), ● = poor (21 - 40), ● = very poor (0 - 20). NB: Scores are unitless.*

5.4.2 Climate and environmental pressures

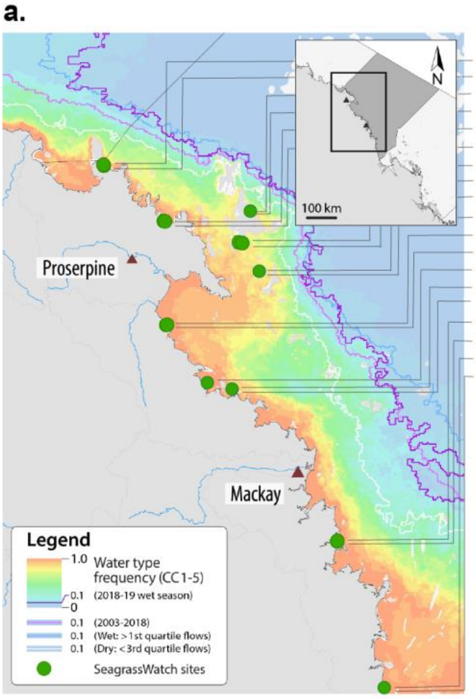
The 2018–19 monitoring period in the Mackay–Whitsunday region was characterised by rainfall and discharge that was above the long-term average for northern rivers, and below average in the southern rivers (Figure 7, Table 9, Figure 53). The majority of the rainfall and discharge occurred in a single event in late January-early February 2019 (Gruber *et al.* 2020).

Exposure of inshore seagrass to turbid waters during the wet season was at the long-term average (Figure 62). Exposure to 'brown' or 'green' turbid water was variable among seagrass habitats (Figure 62). Estuarine and coastal sites were not only exposed to turbid waters for the entire wet season, but were the only habitats exposed to 'brown' sediment laden waters. Estuarine sites in Sarina Inlet (SI1 and SI2), were exposed to 'brown' turbid water for 90% of the wet season, resulting in marginally lower benthic light (Figure 9, Figure 62). Reef habitats fringing the mainland (HB1 and HB2) and located on open water islands (HM1 and HM2, LN1 and LN2) were only exposed to 'green' water but at high frequency (90 – 100% of wet season weeks) and experienced average benthic light (Figure 9, Figure 62).

Within-canopy light was slightly lower than the long-term average for all sites combined within the region (Figure 9, Figure 62, Figure 103). At a site level, benthic light was lower than average at Hamilton Island (HM), but greater than average at all other sites; however, there are periods where no data is available at all sites in the region in 2018–19, so annual averages need to be compared carefully. In general, benthic light levels were around average.

2018–19 was the sixth consecutive year intertidal within-canopy temperatures were above the long-term average, but the difference was marginal (Figure 62). Maximum intertidal within-canopy temperatures exceeded 35°C for a total of 65 days during 2018–19, with the highest temperature recorded at 39.6°C (MP2, 14Mar19). 2018–19 was the second full year of subtidal monitoring with an annual average temperature of 25.3°C, and maximum of 31.9°C (4pm 20Mar19). Daily tide exposure was above the long-term average in 2018–19 for the first time in three years (Figure 62, Figure 96), which may have exacerbated the stresses from the marginally higher water temperatures experienced at intertidal sites.

The proportion of fine grain sizes decreases in the sediments of the seagrass monitoring sites/meadows with distance from the coast/river mouths in the Mackay–Whitsunday region. Estuarine sediments were composed of a greater proportion of finer sediments, and in 2018–19 the proportion of mud was lower than 2017–18 and lower than the Reef-wide long-term average (Figure 114). Coastal habitat meadows had less mud than estuarine habitats over the long term, but fluctuate within and between both meadows and years. In 2018–19 some sites/meadows continued to contain a higher proportion of mud (e.g. PI2 and MP2) than the Reef long-term average (Figure 115). Reef habitats were composed predominately of fine to medium sand, however after cyclone Debbie in early 2017, one of the meadows has maintained a proportion of mud above the Reef long-term average due to heavy scouring of original base sediment and only deposition of fine sediments (predominantly mud) to date (Figure 116).



b.

Week	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22
HB2	6	5	5	5	5	5	5	5			5	5	5	5	5	5	5	5	5	5	5	5
HB1	5	5	5	5	5		5	5			4	5	5	5	5	5	5	5	5	5	5	5
TO1	6	6	5	5	5	5	5	5			5	5	5	5	5	5	6	5	5	5	5	
TO2	6	6	5	5	5	5	5	5			5	5	5	5	5	5	6	5	5	5	5	
PI2	5		5	5	5	5	5	5			4	5	5	5	5	5	5		5	5	5	5
PI3	5		5	5	5	5	5	5			4	5	5	5	5	5	5		5	5	5	5
HM1	6		5	5	5	5	5	5			5	5	5	5	5	5	6	5	5	5	5	
HM2	6		5	5	5	5	5	5			5	5	5	5	5	5	6	5	5	5	5	
LN2	6		5	5	5	5	5	5			5	5	5	5	5	5	6	5	5	5	5	
LN1	6		5	5	5	5	5	5			5	5	5	5	5	5	6	5	5	5	5	
MP3	3		1	2	2	4	1	4			1	1	2	5	4	4	5	1	2	1	4	
MP2	3		1	1	2	4	1	4			1	1	2	5	4	4	5	1	2	1	4	
SH1	3		2	4	1	2	2	2			4	2		3	4	4	5	2	4	2	4	
NB1	5	7	4	5	4	5	4	4			4	4			5	5	5	5	5	4		
NB2	5	7	4	5	4	4	4	4			4	4	5		5	5	5	5	5	4		
SI2	1		1	4	1	4	1	1	5		2	2	2	2	4	1	5	3	4	1	2	1
SI1	1		1	4	1	4	1	1	5		2	2	2	2	4	1	5	3	4	1	2	1
CV1	2		2		2	4	1	4	4		1	2	2		2	2	4	1	3	1	4	
CV2	2		2		2	4	1	4	4		1	2	2		2	2	4	1	3	1	4	

1 - 4 = turbid primary water (red/brown), 5 = nutrient/chlorophyll-enriched secondary water (green), 6 = some freshwater/CDOM influence tertiary water (blue) , 7 = no plume influence (white), blank = no data, clouds (grey)

c.	Long-term average	2018-19
Rainfall (1961-1990)	7.7 mm d^{-1}	9.9 mm d^{-1}
River discharge (1970-2018)	1,920,296 L yr^{-1}	3,681,282 L yr^{-1}
Turbid water exposure (2006-2017)	94%	95%
Daytime tidal exposure (1999-2018)	49 hr yr^{-1}	66 hr yr^{-1}
Within-canopy temperature - intertidal (2003-2018)	25.3°C (42.7°C)	25.5°C (40.2°C)
- subtidal (2017-2018)	25.9°C (32.0°C)	25.3°C (32.0°C)
Within-canopy light	11.9 mol m^{-2} d^{-1}	11.5 mol m^{-2} d^{-1}

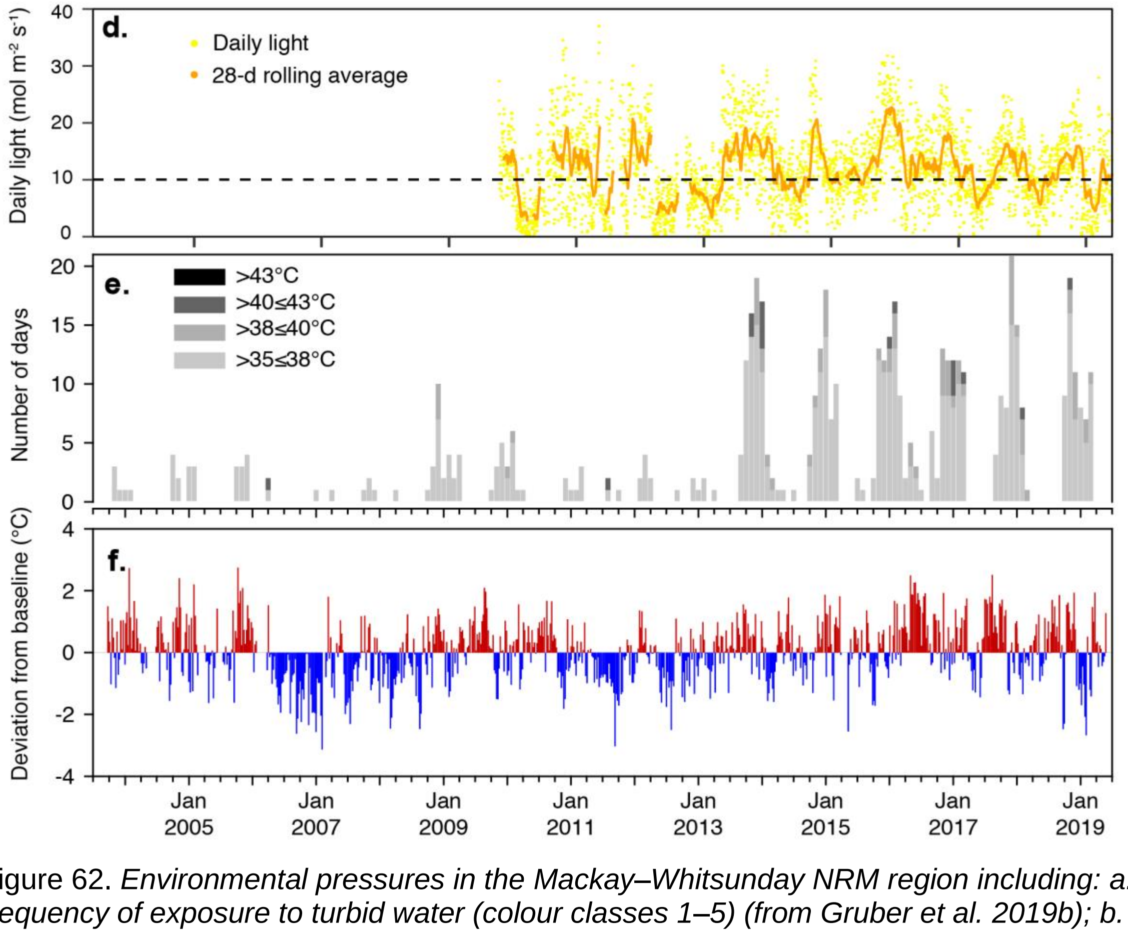


Figure 62. *Environmental pressures in the Mackay–Whitsunday NRM region including: a. frequency of exposure to turbid water (colour classes 1–5) (from Gruber et al. 2019b); b. wet season water type at each site; c. average conditions over the long-term and in 2018–19; d. daily light and the 28-day rolling mean of daily light for all sites; e. number of day temperature exceeded 35°C, 38°C, 40°C and 43°C, and; f. deviations from 13-year mean weekly temperature records.*

5.4.3 Inshore seagrass and habitat condition

There are 16 seagrass monitoring sites in Mackay–Whitsundays from 10 locations (Table 14). Five seagrass habitat types were assessed across the region this year (Table 14, Table 18).

Table 14. *List of data sources of seagrass and environmental condition indicators for each seagrass habitat type in the Mackay–Whitsunday NRM region. † drop camera sampling (QPWS), *Seagrass-Watch. For site details see Table 2 and Table 3.*

Habitat	Site		abundance	composition	distribution	reproductive effort	seed banks	leaf tissue nutrients	meadow sediments	epiphytes	macroalgae
estuary intertidal	SI1	Sarina Inlet	■	■	■	■	■	■	■	■	■
	SI2	Sarina Inlet	■	■	■	■	■	■	■	■	■
coastal intertidal	MP2	Midge Point	■	■	■	■	■	■	■	■	■
	MP3	Midge Point	■	■	■	■	■	■	■	■	■
	PI2*	Pioneer Bay	■	■			■		■	■	■
	PI3*	Pioneer Bay	■	■			■		■	■	■
	SH1*	St Helens	■	■			■		■	■	■
	CV1*	Clairview	■	■			■		■	■	■
	CV2*	Clairview	■	■			■		■	■	■
coastal subtidal	NB1†	Newry Bay	■	■							■
	NB2†	Newry Bay	■	■							■
reef intertidal	HM1	Hamilton Island	■	■	■	■	■	■	■	■	■
	HM2	Hamilton Island	■	■	■	■	■	■	■	■	■
	HB1*	Hydeaway Bay	■	■			■		■	■	■
	HB2*	Hydeaway Bay	■	■			■		■	■	■
reef subtidal	LN1	Lindeman Is	■	■	■	■	■	■	■	■	■
	LN2	Lindeman Is	■	■	■	■	■	■	■	■	■
	TO1†	Tongue Bay	■	■							■
	TO2†	Tongue Bay	■	■							■

5.4.3.1 Seagrass index and indicator scores

In the 2018–19 monitoring period, the Mackay–Whitsunday region seagrass condition index increased slightly from the previous year, but remained graded as **poor** (Figure 63).

Overall, the Mackay–Whitsunday seagrass index had been improving since 2010–11, when it reached its lowest level since monitoring commenced. In 2016–17 the improving trend abated and abundance declined as a consequence of cyclone Debbie, but this year has increased slightly (Figure 63). However, the 2018–19 increase has been slightly offset by one of the condition indicators.

The reproductive effort in 2018–19 was the lowest since 2011 (Figure 63). This appears a legacy of losses experienced from the impacts of cyclone Debbie and associated flooding.

An examination of the long-term trends across the Mackay–Whitsunday NRM region using GAM plots suggests seagrass abundance (per cent cover) and reproductive effort have been declining since 2015–16 (Figure 63).

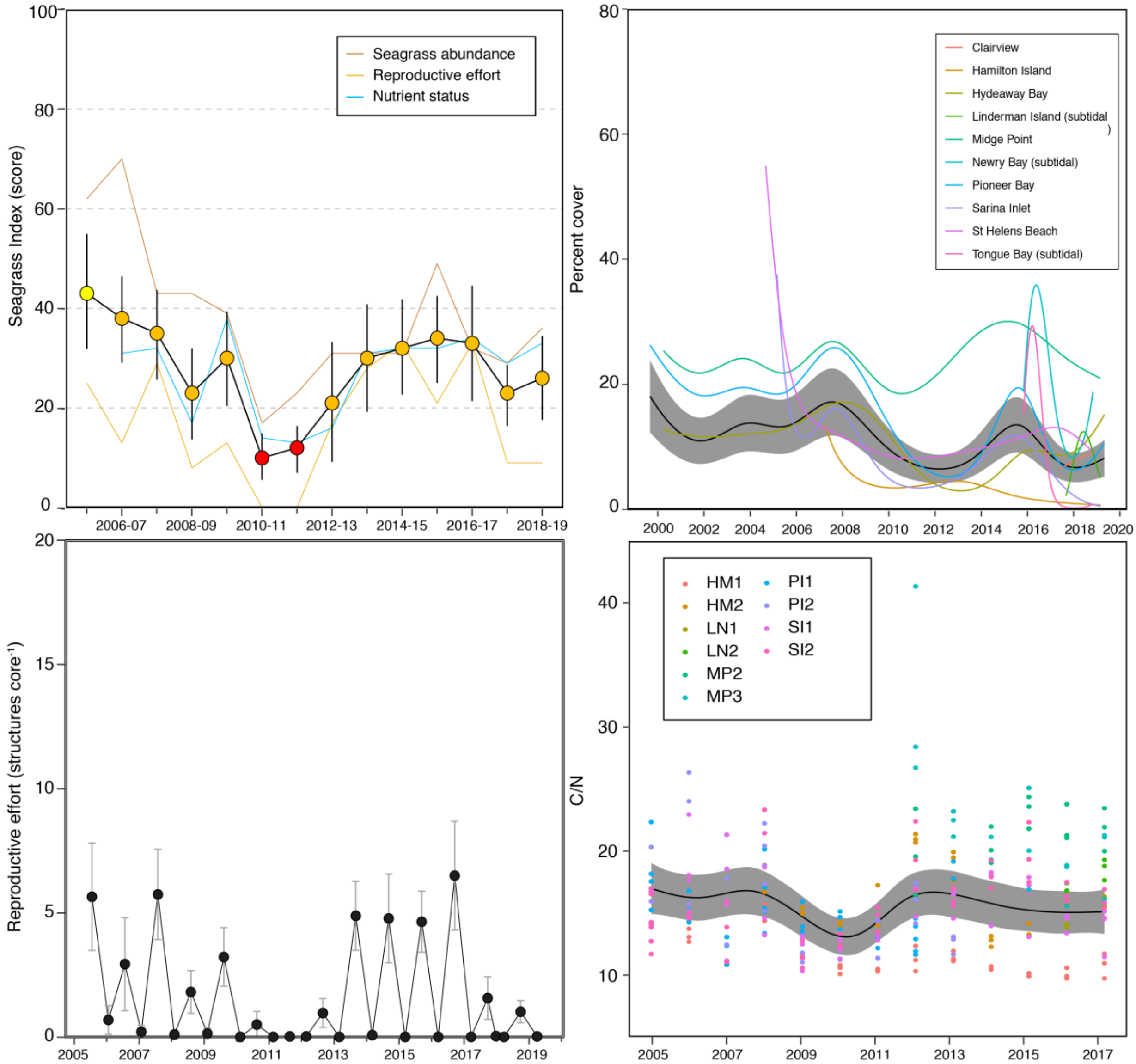


Figure 63. *Temporal trends in the Mackay–Whitsunday seagrass condition index and the indicators used to calculate the index: a. seagrass condition index (circles) and indicator trends (lines); b. GAM plots of seagrass abundance (per cent cover) trends for each location (coloured lines) and the region (black line with grey shaded area defining 95% confidence intervals); c. average number of reproductive structures (±SE) (GAM not possible due to high count of zero values); and d. elemental ratios (atomic) of leaf tissue C:N nutrient content at each site (coloured circles) and regional trend represented by a GAM plot as dark line with shaded areas defining 95% confidence intervals of the trend*

5.4.3.2 Seagrass abundance, community and extent

Seagrass abundance increased in 2018–19 at two thirds of sites across the region, relative to the previous period, reversing the legacy of the losses experienced in early 2017 as a consequence of the impacts of cyclone Debbie and associated flooding (Figure 64). Although some losses were observed (33% and 25% of coastal and reef intertidal sites, respectively), the greatest continued in estuarine habitats where all sites decreased in abundance.

Seagrass abundance (per cent cover) in the Mackay–Whitsunday region in 2018–19 was higher in coastal habitats (intertidal = 13.1 ±1.2%, subtidal = 19.0 ±2.9%) than reef habitats (intertidal = 7.6 ±124%, subtidal = 4.9 ±0.7%) or estuarine (0.9 ±0.5%), respectively. As a consequence of the recovering abundances, seagrass per cent covers were not greatly different between seasons across all habitats (e.g. coastal intertidal, late dry = 14.5 ±1.2%, late monsoon = 14.6 ±1.5%).

Seagrass abundance at estuary and coastal intertidal habitats has fluctuated greatly between and within years over the long-term, with some sites experiencing total or near total loss followed by recovery (Figure 64). The long-term trend indicates a declining trajectory (Table 20) with a region struggling to recover from losses in the years leading up to 2010–11 and in early 2017.

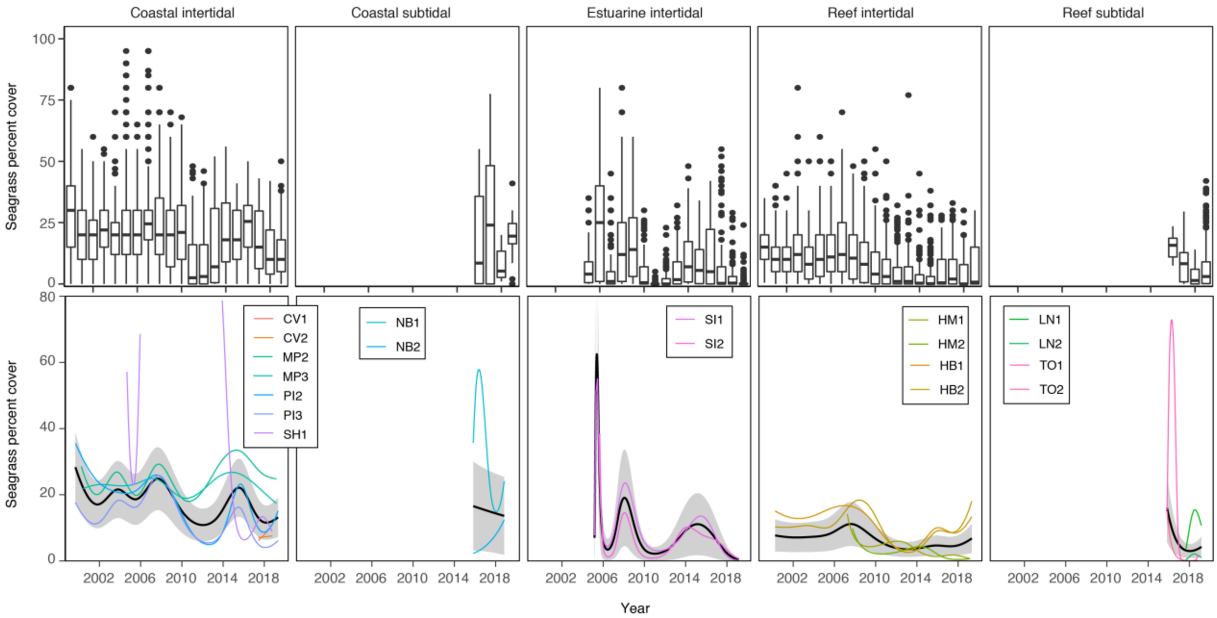


Figure 64. *Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the Mackay–Whitsunday NRM region from 1999 to 2019. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates the 25th percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75th percentile. Whiskers (error bars) above and below the box indicate the 90th and 10th percentiles, and the dots represent outlying points. GAMM plots (bottom), show trends for each habitat and coloured lines represent individual site trends*

The most common seagrass species across all habitats in the Mackay–Whitsunday NRM region were *Halodule uninervis* and *Zostera muelleri*, mixed with the colonising species *Halophila ovalis*.

Colonising species dominated intertidal meadows across the Mackay–Whitsunday region in the first few years following the extreme weather in 2011. In the previous two years, there has been a dramatic reduction in colonising species in estuarine and coastal intertidal habitats. In all habitats except reef, opportunistic foundational species (*H. uninervis* and *Z. muelleri*) now dominate (Figure 65), suggesting meadows may have an improved ecosystem resistance to tolerate disturbances (Figure 65). In contrast, in intertidal reef habitats (Hamilton Island), colonising species have been steadily increasing since 2006 and remained above the Reef long-term average over the last few years. In 2018–19, the only notable change was an increase of colonising species in subtidal reef meadows, where the composition has increased above the Reef long-term average (Figure 65).

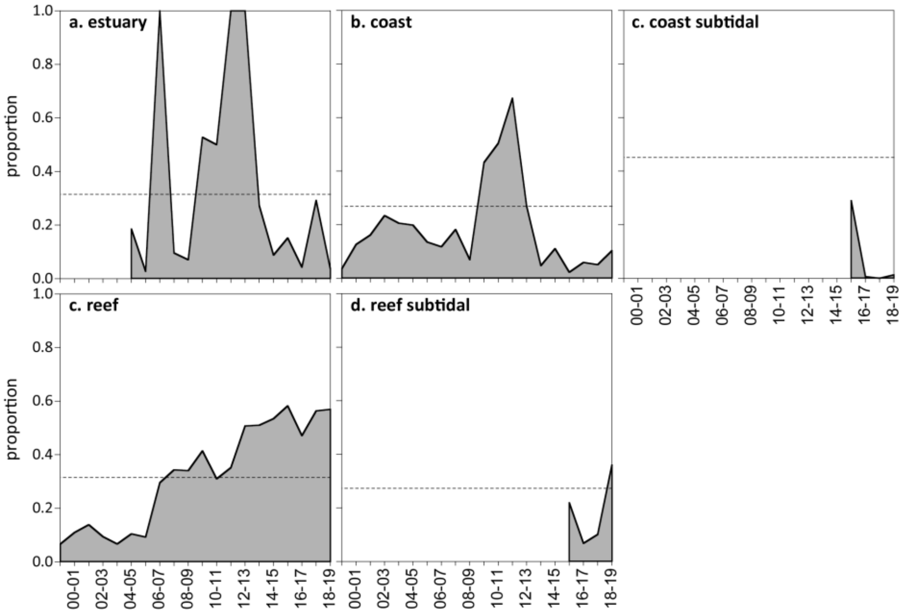


Figure 65. *Proportion of seagrass abundance composed of colonising species at inshore intertidal habitats in the Mackay–Whitsunday region, 1999–2019. Grey area represents Reef long-term average proportion of colonising species for each habitat type*.

Seagrass meadow landscape mapping was conducted within all sentinel monitoring sites in October 2017 and April 2018 to determine if changes in abundance were a consequence of the meadow landscape changing (e.g. expansion or fragmentation) and to indicate if plants were allocating resources to colonisation (asexual reproduction). Over the past 12 months, spatial extent improved slightly at reef intertidal meadows following the declines experienced in 2016–2017 as a consequence of the destructive effects of cyclone Debbie. At coastal meadows, extent remained steady, but at estuarine meadows extent declined greatly relative to the previous monitoring period, due to increased fragmentation (Figure 66).

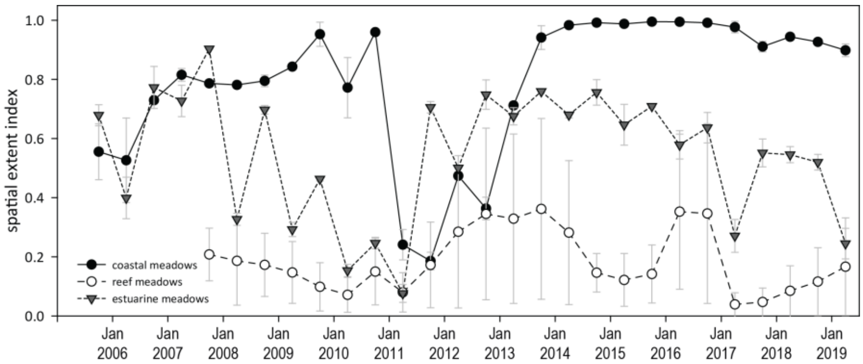


Figure 66. *Change in spatial extent of seagrass meadows within monitoring sites for each inshore intertidal habitat and monitoring period across the Mackay–Whitsunday NRM region.*

5.4.3.3 Seagrass reproductive status

Reproductive effort was highly variable and highly seasonal in the Mackay–Whitsunday region (Figure 67). Reproductive effort declined slightly in coastal habitats, relative to the previous period, although the density of seeds in the seedbank remained steady. At the estuary meadow (Sarina Inlet), reproductive effort was near absent, but seed banks increased slightly relative to the previous year. In contrast, reproductive effort and the seeds density continued to remain very low at reef sites in 2018–19, which appears typical for reef habitat meadows (Figure 67).

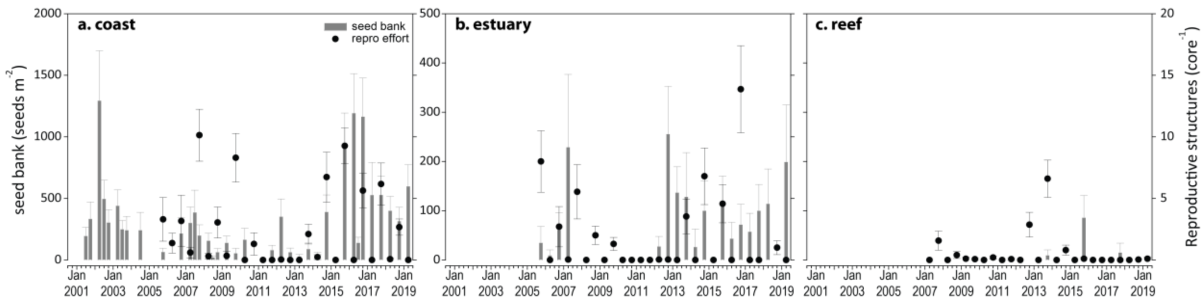


Figure 67. *Seed bank and reproductive effort at inshore intertidal coast, estuary, and reef habitats in the Mackay–Whitsunday region, 2001–2019. Seed bank presented as the total number of seeds per m^2 sediment surface and reproductive effort presented as the average number of reproductive structures per core (species and sites pooled). NB: Y-axis scale for seed banks differs between habitats.*

5.4.3.4 Seagrass leaf tissue nutrients

Seagrass leaf molar C:N ratios were unchanged compared to the previous year, remaining below 20 (Figure 68), indicating a surplus of N relative to photosynthetic C incorporation. N:P ratios continued to increase across all habitats, and %N remained above the global median, indicating surplus availability of N across the region. The moderate and fluctuating $\delta^{15}N$ (e.g. increasing at reef habitats), suggests some influence of an anthropogenic source of N at some sites (e.g., Hamilton Island) (Figure 68).

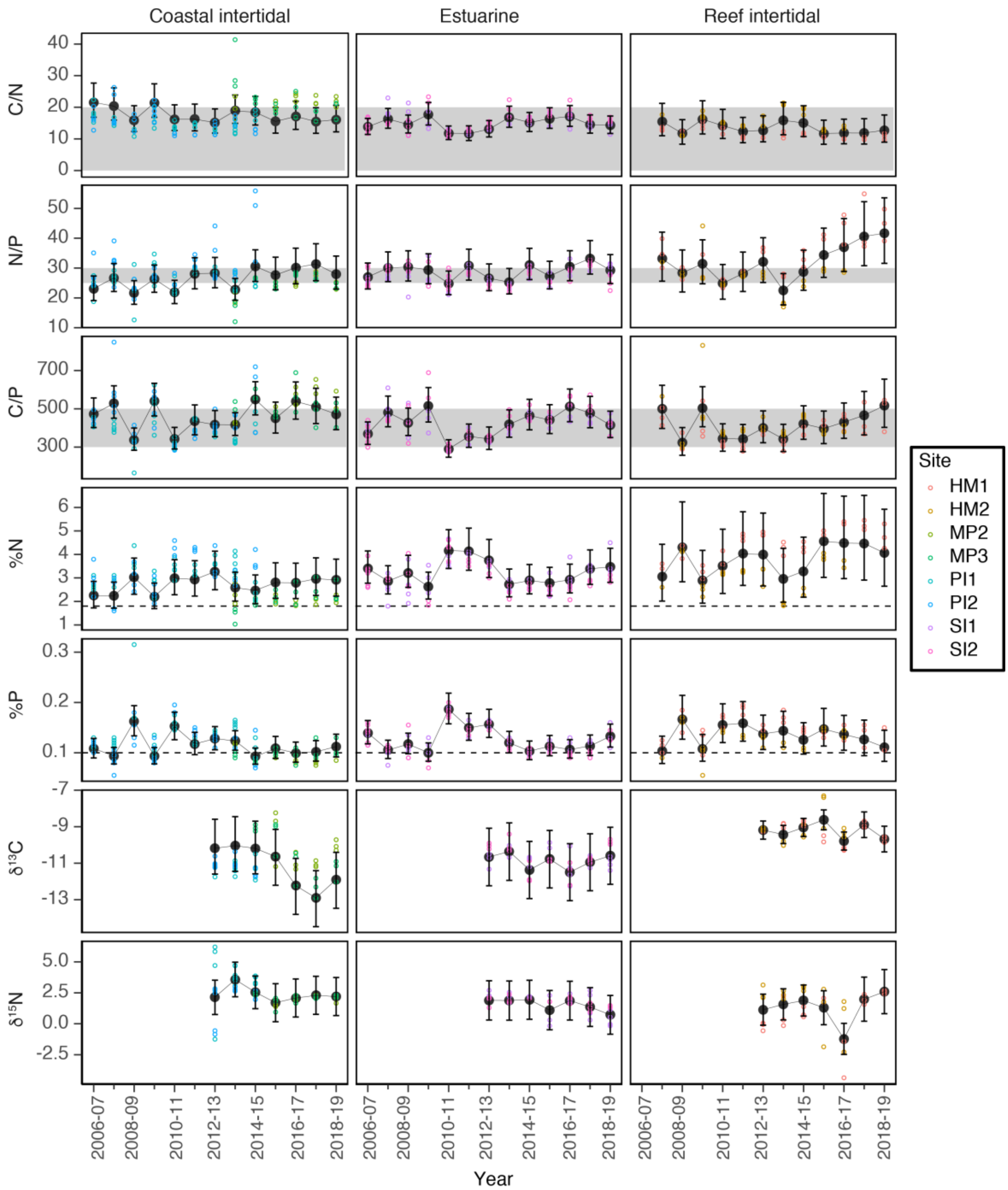


Figure 68. *Seagrass leaf tissue nutrient elemental ratios (C:N:P) and concentrations (%N, %P, $\delta^{13}C$ and $\delta^{15}N$) for each habitat in the Mackay–Whitsunday region (± SE) (foundation species pooled). Horizontal shaded bands or dashed lines represents the accepted seagrass guideline values, where: C:N ratios within the band may indicate reduced light availability and/or N enrichment; N:P ratios above the band indicate P limitation, below indicate N limitation and within indicates replete, and; C:P ratios within the band may indicate nutrient rich habitats (large P pool). Dashed lines in %N and %P indicate global median values of 1.8% and 0.2% for tissue nitrogen and phosphorus, respectively (Duarte 1990).*

5.4.3.5 Epiphytes and macroalgae

Epiphyte cover on seagrass leaf blades in 2018–19 has remained below the Reef-wide long-term average at estuarine and reef habitats since early 2017, and decreased slightly at coastal habitats relative to the previous reporting year (Figure 69).

Percentage cover of macroalgae remained unchanged, at or below the Reef-wide long-term average for estuarine and coastal habitats throughout 2018–19 (Figure 69). At reef meadows, however, macroalgae cover increased above the Reef-wide long-term average in 2018–19, and was the highest cover observed in over a decade (Figure 69).

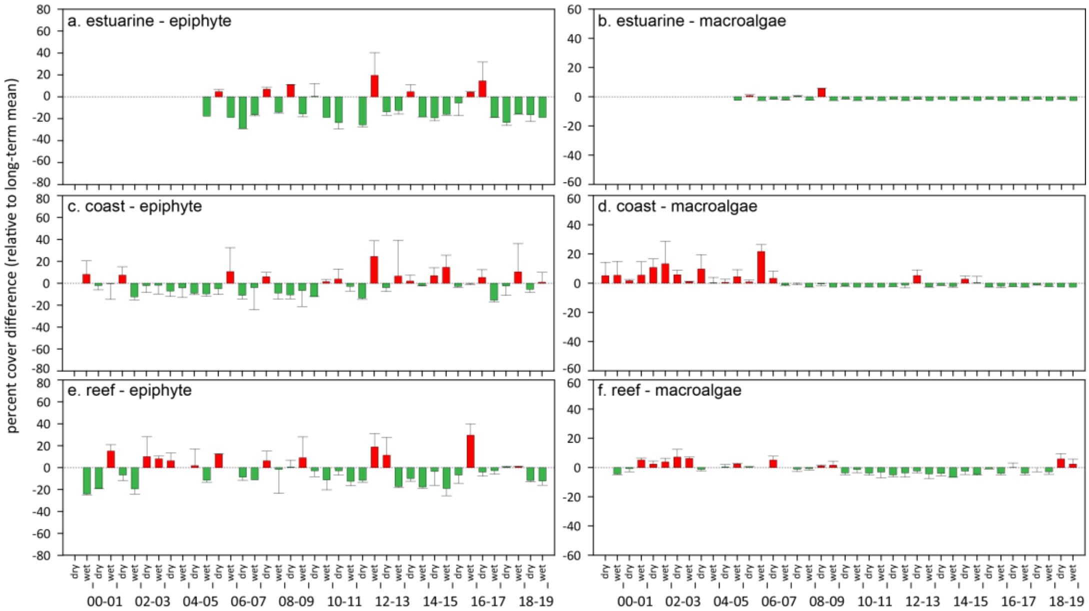


Figure 69. *Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average for each inshore intertidal habitat in the Mackay–Whitsunday region, 1999–2018 (sites pooled, ±SE). Red/green text*

5.5 Fitzroy

5.5.1 2018–19 Summary

Overall, the Fitzroy regional seagrass condition score remained graded as **poor** in 2018–19 (Figure 70). There were no substantial changes from the previous year in any of the indicators, where the:

- abundance score was poor
- reproductive effort score was very poor
- tissue nutrient score was poor.

Approximately two-thirds of sites improved in abundance this year, however half of the coastal sites decreased relative to the previous period. Abundances at the coastal intertidal sites in Shoalwater Bay remain near record high levels. However, meadow abundance in estuarine habitat remains very low, owing to a wave of mud and burrowing shrimp activity moving through the area. The poor condition at the estuarine sites is consistent with findings for the Pelican Banks meadow and the Gladstone Healthy Harbour Partnership score for the whole region, both rated as poor in 2018. Abundances remain very low at the reef intertidal sites, with little variability among years except in the degree of fragmentation as shown by the seagrass extent. However, a reduction in the proportion of colonising species in 2018–19 indicates that the reef meadows have been relatively stable. The long-term trend in seagrass abundance (per cent cover) across the region reveals a significant decrease, primarily driven by estuary and reef habitats.

Reproductive effort remains well below historical peaks for all habitats in the region. However, the consistent presence of some reproductive structures and a persistent seed bank in both coastal and estuarine habitats indicates some resilience and capacity to recover from any future events. Of concern is that reproductive effort at reef sites remains very low to absent, and there is no seed bank despite an increase in the proportion of *H. uninervis*, a species that can contribute to the seed bank.

The seagrass leaf nutrient status remained relatively stable overall in 2018–19, with a slight increase at estuarine sites, and a slight decrease at coastal and reef sites. Seagrass leaf molar C:N ratios continue to indicate a surplus of N relative to photosynthetic C incorporation (i.e. C:N is less than 20) at most sites; however, there is no indication of elevated N across the region. This is supported by continuing low epiphyte and macroalgae cover.

Environmental pressures were similar to the long-term average levels for the region. River discharge was below average, but benthic light levels were slightly lower than average. Average annual water temperature was around the average, but there were a number of high temperature days, including three days when temperature exceeded 40°C, a threshold likely to impart stress on all species, and in particular on *Zostera muelleri*.

Inshore seagrass meadows across the region remain in the early stages of recovering from multiple years of climate related impacts which, similar to Mackay–Whitsunday, are more recent than in other regions. The coastal habitats have been improving, while other habitats demonstrate a legacy of reduced resilience.

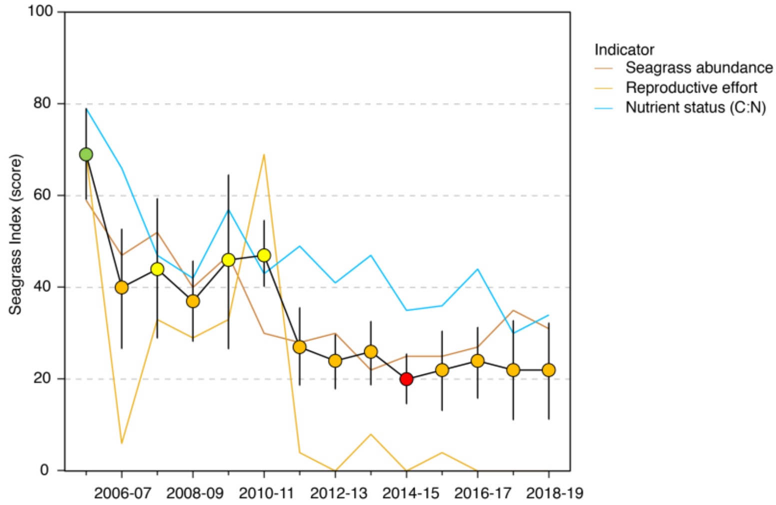


Figure 70. *Report card of seagrass status index and indicators for the Fitzroy NRM region (averages across habitats and sites). Values are indexed scores scaled from 0–100 and graded: ● = very good (81-100), ● = good (61 - 80), ● = moderate (41 - 60), ● = poor (21 - 40), ● = very poor (0 - 20). NB: Scores are unitless.*

5.5.2 Climate and environmental pressures

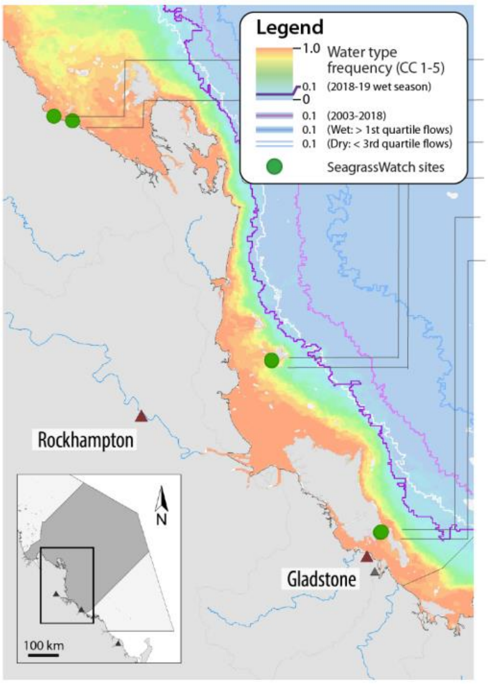
Rainfall and river discharge in 2018–19 were well below the long-term average for the Fitzroy region (Figure 71). Exposure of inshore seagrass to turbid waters during the wet season was similar to the long-term average, with the coastal and estuarine sites exposed to highly turbid 'brown' water in most weeks. By contrast, the reef sites were exposed only to 'green' water which has lower light attenuation.

Annual within-canopy light availability was similar in 2018–19 to the long-term average for the region (Figure 9, Figure 71). The most notable change in benthic light levels occurred at Shoalwater Bay, where benthic light levels (15.5 mol m^{-2} d^{-1}) were below the long-term average (18.4 mol m^{-2} d^{-1}). Despite this, light levels at Shoalwater Bay were the highest among all sites in the region because they are very shallow and frequently expose to full sunlight (Figure 104). Predicted daytime tidal exposure was considerably greater than the long-term average for the region, which increases the risk of desiccation stress, but can provide windows of light for photosynthesis (Figure 97).

2018–19 was the first year that within-canopy temperatures were similar to the long-term average after five consecutive years where they were above the long-term average (Figure 71). Maximum intertidal within-canopy temperatures exceeded 35°C for a total of 56 days during 2018–19, with the highest temperature recorded in the region at 40.3°C (RC1, 1pm 13Feb19). Daily tidal exposure was above the long-term average in 2018–19 for the first time in three to four years (Figure 62, Figure 96), which may have exacerbated stresses experienced at intertidal sites.

The proportion of fine grains in meadow sediments generally decreases with distance from the coast/river mouths. Estuarine sediments were composed primarily of finer sediments, with the mud portion around the Reef-wide long-term average, although one site (GH1) continued to be much muddier this year (Figure 117). Coastal and reef habitat sediments are dominated by fine sand/sand, but the proportion of mud in coastal habitats continued to increased greatly in 2018–19 above the Reef long-term average (Figure 118, Figure 119).

a.



b.

Wk	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22
RC1	1		2	4	2	1	2	1			1	1	2	1	1	1	4	4	4	2	4	
WH1	1	5	1	4	1	1	1	1			1	1	2	2	1	1	5	2	4	1	4	
GK2	5	5	5	5	5	5	5	5	5		5	5	5	5	5	5	6	5	5	5	5	5
GK1	5	5	5	5	5	5	5	5	5		5	5	5	5	5	5	6	5	5	5	5	5
GH2	4		2	4	4	2	4	1	3		5	2	4	4	4	2	5	1	3	2	4	4
GH1	4		2	4	4	2	4	1	3		5	2	4	4	4	2	5	1	3	2	4	4

1 - 4 = turbid primary water (red/brown), 5 = nutrient/chlorophyll-enriched secondary water (green), 6 = some freshwater/CDOM influence tertiary water (blue) , 7 = no plume influence (white), blank = no data, clouds (grey)

c.

	Long-term average	2018-19
Rainfall (1961-1990)	2.9 mm d^{-1}	2.1 mm d^{-1}
River discharge (1970-2018)	3,270,564 L yr^{-1}	1,502,831 L yr^{-1}
Turbid water exposure (2006-2018)	97.8%	99.0%
Daytime tidal exposure (1999-2018)	108 hr yr^{-1}	148 hr yr^{-1}
Within-canopy temperature - intertidal (2006-2018)	24.2°C (41.6°C)	24.3°C (41.2°C)
Within-canopy light	14.8 mol m^{-2} d^{1}	14.3 mol m^{-2} d^{1}

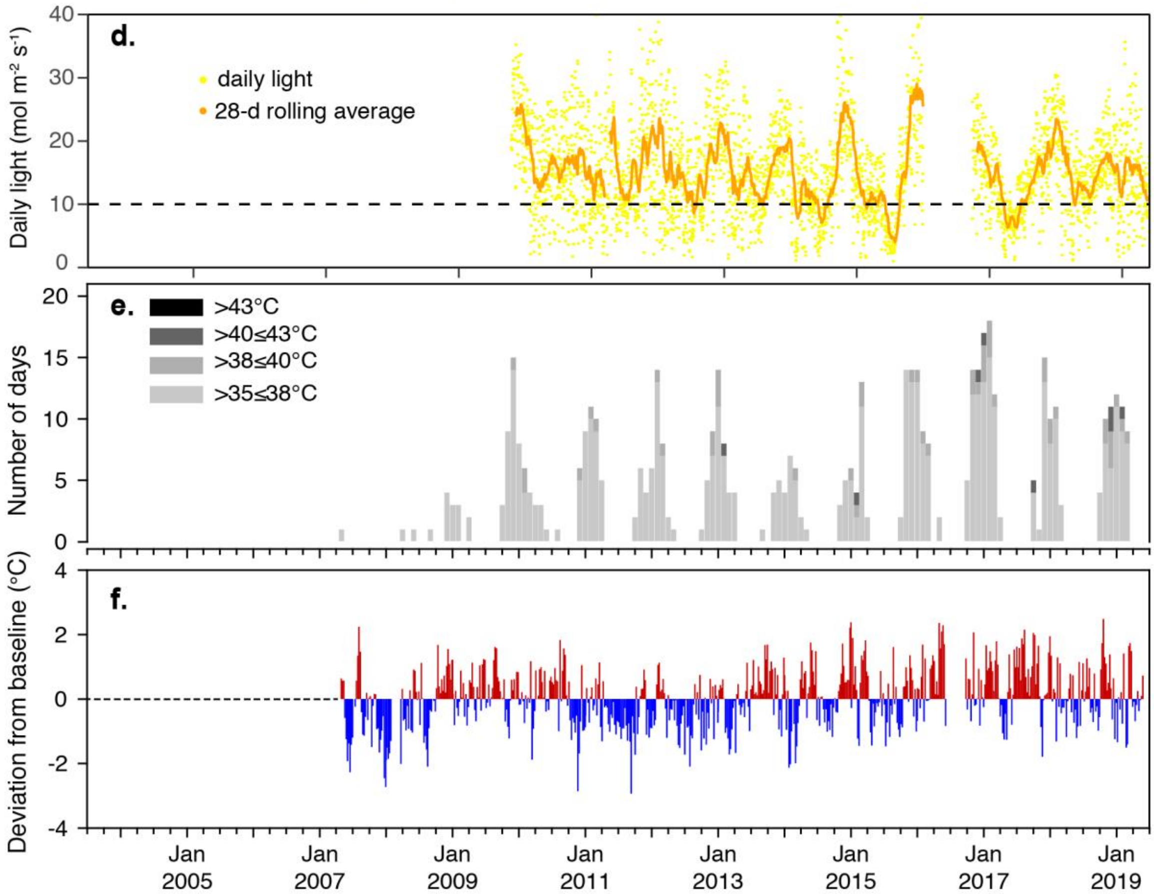


Figure 71. *Environmental pressures in the Fitzroy region including: a. frequency of exposure to turbid water (colour classes 1–5) (from Gruber et al. 2019b); b. wet season water type at each site; c. average conditions over the long-term and in 2018–19; d. daily light and the 28-day rolling mean of daily light for all sites; e. number of day temperature exceeded 35°C, 38°C, 40°C and; 43°C, and f. deviations from 12-year mean weekly temperature records.*

5.5.3 Inshore seagrass and habitat condition

There are 6 seagrass monitoring sites in the Fitzroy from 3 locations (Table 15). Three seagrass habitat types were assessed across the region in 2018–19, with data from 6 sites (Table 15).

Table 15. *List of data sources of seagrass and environmental condition indicators for each seagrass habitat type in the Fitzroy NRM region. For site details see* Table 2 *and* Table 3*.*

Habitat	Site		abundance	composition	distribution	reproductive effort	seed banks	leaf tissue nutrients	meadow sediments	epiphytes	macroalgae
estuary intertidal	GH1	Gladstone Harbour	■	■	■	■	■	■	■	■	■
	GH2	Gladstone Harbour	■	■	■	■	■	■	■	■	■
coastal subtidal	RC1	Ross Creek (Shoalwater Bay)	■	■	■	■	■	■	■	■	■
	WH1	Wheelans Hut (Shoalwater Bay)	■	■	■	■	■	■	■	■	■
reef intertidal	GK1	Great Keppel Is.	■	■	■	■	■	■	■	■	■
	GK2	Great Keppel Is.	■	■	■	■	■	■	■	■	■

5.5.3.1 Seagrass index and indicator scores

In the 2018–19 monitoring period, the seagrass condition index remained relatively stable and was graded as **poor** (Figure 72).

The abundance score declined marginally, which was offset by a marginal improvement in the tissue nutrients score and the reproductive effort score remaining at zero (Figure 72).

Reproductive effort has remained low since 2011–2012, and fluctuations in the seagrass condition index over the last 7 monitoring periods have been primarily driven by fluctuations in abundance and tissue nutrient status.

Of particular concern is that seagrass abundance (per cent cover) has significantly decreased since 2008 at estuarine sites in Gladstone Harbour, due predominantly to a mud wave that has moved through and recovery is expected (Figure 72, Table 20).

Long-term trends using GAM plots also suggests tissue nutrient elemental C:N has been declining since 2005 across the region (Figure 72).

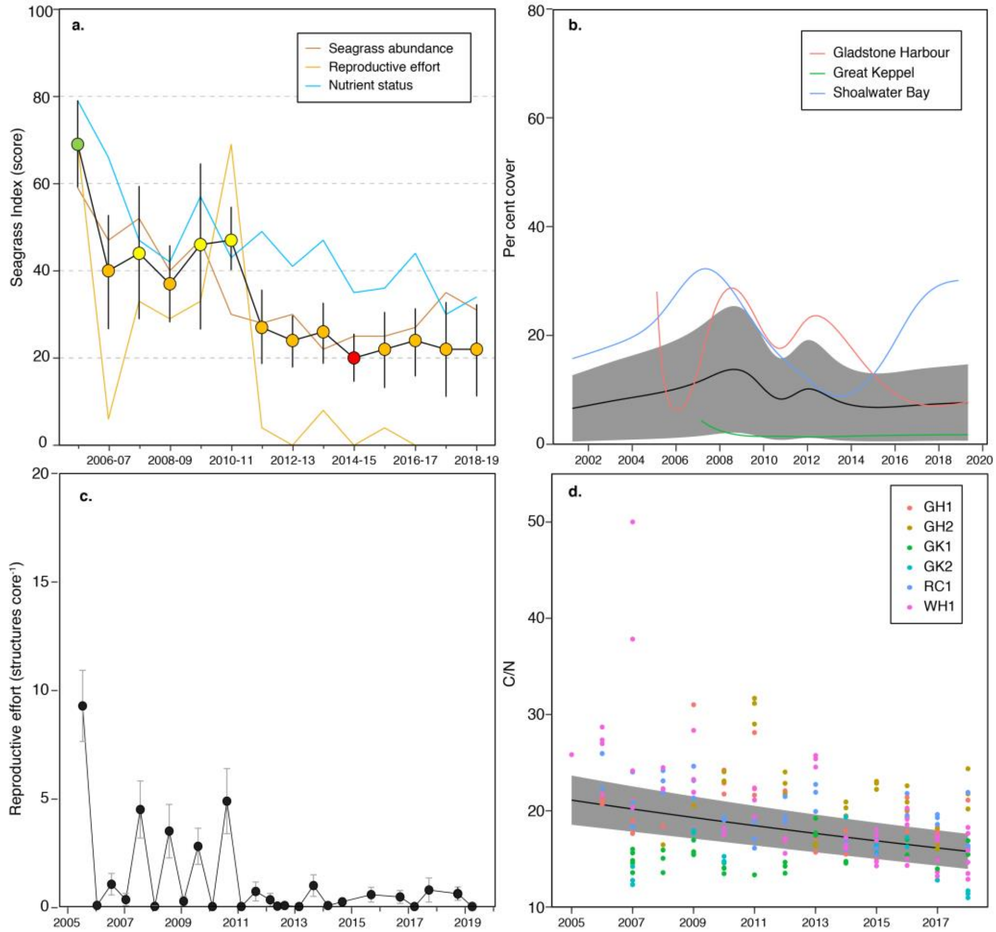


Figure 72. *Temporal trends in the Fitzroy seagrass condition index and the indicators used to calculate the index: a. seagrass condition index (circles) and indicator trends (lines); b. GAM plots of seagrass abundance (per cent cover) trends for each location (coloured lines) and the region (black line with grey shaded area defining 95% confidence intervals); c. average number of reproductive structures (±SE) (GAM not possible due to high count of zero values); and d. elemental ratios (atomic) of leaf tissue C:N nutrient content at each site (coloured circles) and regional trend represented by a GAM plot as dark line with shaded areas defining 95% confidence intervals of the trend*

5.5.3.2 Seagrass abundance, composition and extent

In 2018-19, approximately two third of sites improved in abundance, however half of the coastal sites decreased relative to the previous period. Seagrass abundances (per cent cover) in the Fitzroy region in 2018-19 were significantly higher in coastal (23.5 ±1.0%) and estuarine (8.3 ±2.0%) habitats, than reef (2.9 ±0.7%) (Figure 73). With the exception of reef habitats, there was little difference in seagrass abundance between the seasons. In reef habitats, abundances were higher in the late dry than the late wet season (5.5 ±0.9% and 0.2 ±0.1%, respectively).

Seagrass abundance at estuary and coastal intertidal habitats has fluctuated greatly between years over the life of the monitoring, with some sites experiencing total or near total loss followed by recovery (Figure 73). In 2018–19, half of the coastal sites decreased in abundance relative to the previous period, with all remaining sites, including all reef sites, increasing or remaining stable (Figure 73).

Examination of the long-term trend in seagrass abundance (per cent cover) across the region reveals a significant decrease (Figure 72, Table 20). These decreases have primarily occurred in the estuary and reef habitats, although two thirds of all monitoring sites in the region (including coastal) show no significant trend (Table 20).

The low seagrass abundance in the estuarine habitat appears a legacy of decline in 2016–17, the result of a mud wave traversing across the meadow. As the mud wave dissipated in 2018–19, meadow integrity (e.g. reduced scarring) improved.

In the north of the region, coastal sites receive low river discharge, however, the meadows were still exposed to turbid 'brown' sediment laden waters for much of the year. There turbid waters could be partly the result of wind-driven resuspension, but appear mainly the consequence of the extreme tidal movement in Shoalwater Bay (some of the highest along the Queensland coast).

Seagrasses in Shoalwater Bay are able to persist on the large intertidal banks, where periods of shallowing water provide some respite from the highly turbid waters. However, these periods of shallowing water and carbon limitation (when exposure to air coincides with low spring tides) not only stress plants with desiccation, but also fluctuating water temperatures.

Maximum water temperatures exceeded 35°C for a total of 54 days in Shoalwater Bay during 2018–19, with a highest temperature of 40.3°C. The high temperatures are particularly stressful for *Z. muelleri* communities which dominate the coastal habitats as it has a thermal optima for overall net primary productivity of 24°C and above 35°C net productivity goes into deficit, i.e. it loses energy (Collier *et al.* 2017). This is in stark contrast to other tropical species (*H. uninervis* and *C. serrulata*), which must exceed 40°C for respiration rates and photoinhibition to cause the plants to lose energy for pulsed exposure (Collier *et al.* 2017). Similarly, water temperature exceeded 35°C (max 37.4) on 8 days at Pelican banks in Gladstone Harbour and this was likely to have placed a substantial stress on these *Z. muelleri* dominated communities.

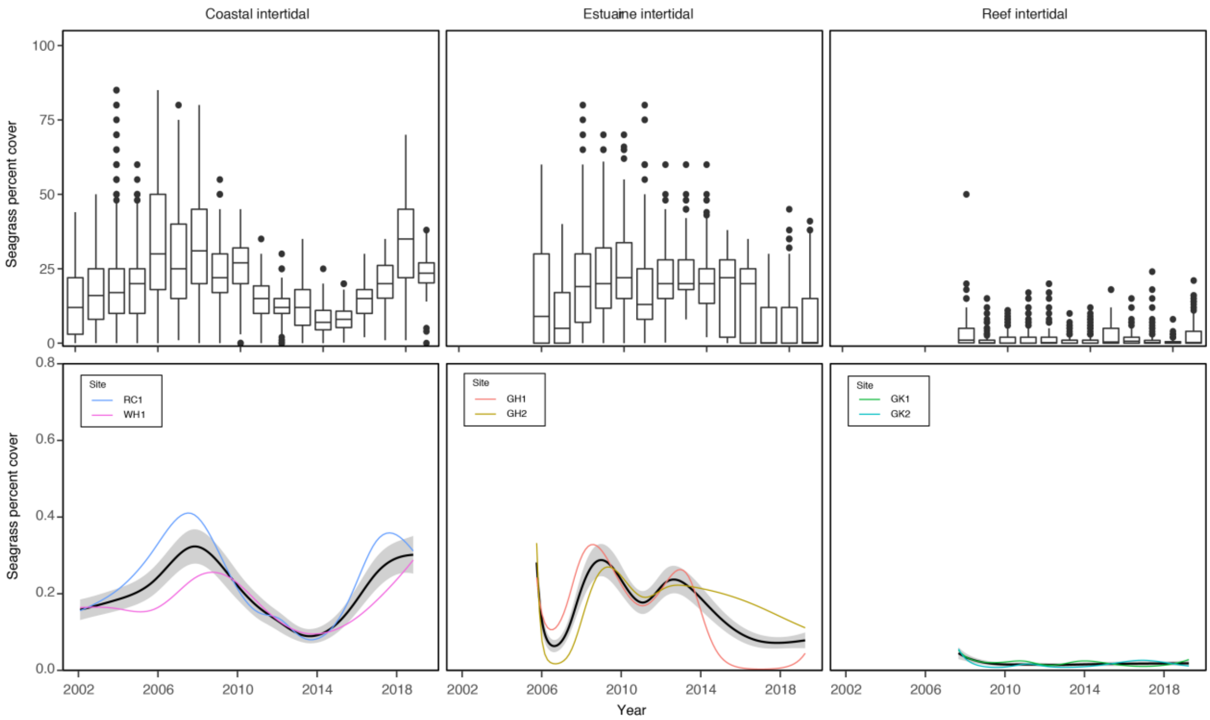


Figure 73. *Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the Fitzroy NRM region from 2002 to 2019. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates the 25th percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75th percentile.*

Whiskers (error bars) above and below the box indicate the 90th and 10th percentiles, and the dots represent outlying points. GAMM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.

Coastal meadows in Shoalwater Bay (Ross Creek and Wheelans Hut) had an increased proportion of colonising species (*H. ovalis*) after 2011 but remained dominated (>0.5) by the opportunistic species *Z. muelleri* and *H. uninervis* (Figure 74). In 2018–19, the proportion of these opportunistic species increased at both the coastal and estuarine sites (Figure 74) which continued to be dominated by *Zostera muelleri*. Colonising species, however, continued to dominate the reef habitat sites (well above the Reef-wide long-term average), which appears a direct relationship with decreased abundances over the last few years (Figure 74).

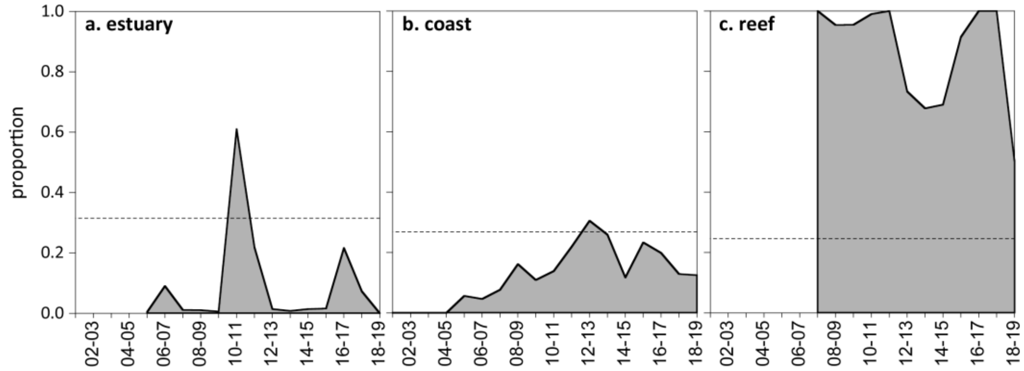


Figure 74. *Proportion of seagrass abundance composed of colonising species in inshore intertidal habitats of the Fitzroy region, 2001–2019. Grey area represents Reef long-term average proportion of colonising species for each habitat type.*

The extent of the coastal meadows within monitoring sites in Shoalwater Bay has remained stable at or near the maximum since monitoring commenced in 2005. The extent of the estuarine meadows has fluctuated since 2016 when there was a large reduction in one of the sites due to extensive scarring and sediment deposition. This year the scarring had abated and the meadow was showing signs of recovering, e.g. shoot extension and improved meadow cohesion. Conversely, meadows on the reef flat at Great Keppel Island remained highly fragmented after the 2016 losses and show little sign of recovery, e.g. unstable sediments.

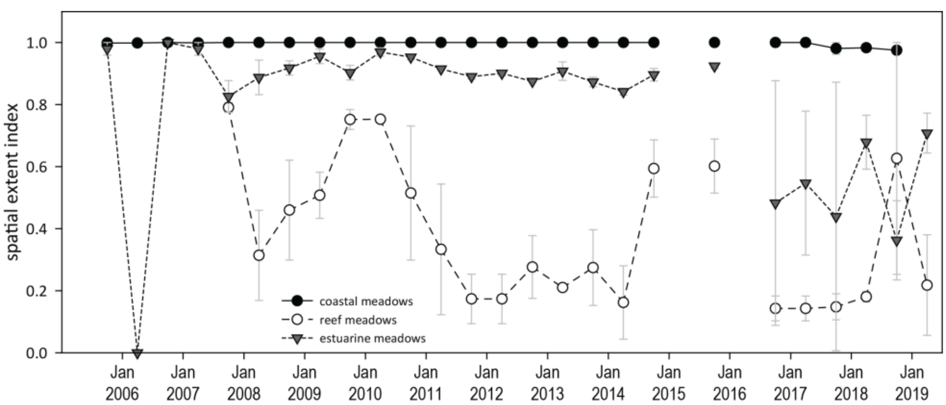


Figure 75. *Change in spatial extent of seagrass meadows within monitoring sites for each inshore intertidal habitat across the Fitzroy NRM region, 2005–2019.*

5.5.3.3 Seagrass reproductive status

Reproductive effort has varied inconsistently among habitats in the Fitzroy region over the life of the MMP (Figure 76). Reproductive effort is higher in the late dry season and remained steady at coastal and estuary sites in 2018–19 (Figure 76). A seed bank has also persisted at coastal and estuary sites since 2012. Reproductive effort has remained very low at reef sites, and seed banks remain absent (Figure 76). This limits the meadow capacity to recover following further disturbance.

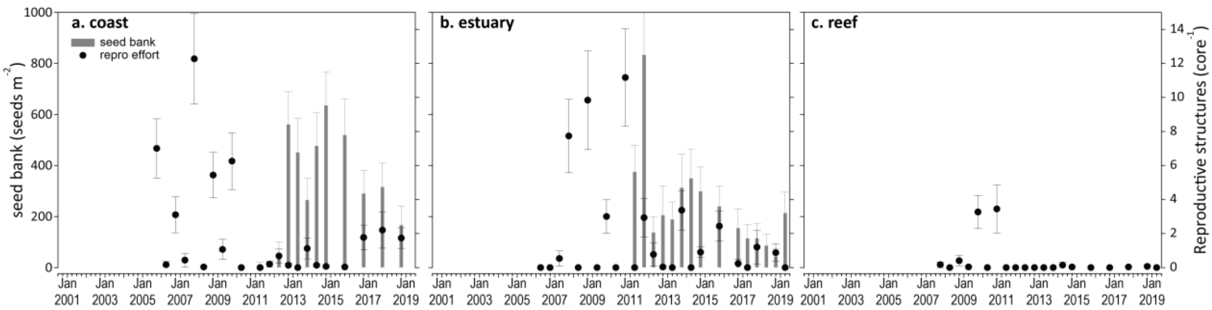


Figure 76. *Reproductive effort for inshore intertidal coastal, estuary and reef habitats in the Fitzroy region, 2005–2019. Seed bank presented as the total number of seeds per m^2 sediment surface and late dry season reproductive effort presented as the average number of reproductive structures per core (species and sites pooled).*

5.5.3.4 Seagrass leaf tissue nutrients

Seagrass leaf molar C:N ratios marginally declined across all habitats in 2018–19 relative to the previous year, remaining at or below 20 (Figure 68), indicating a surplus of N relative to photosynthetic C incorporation. N:P ratios marginally increased across all habitats, which combined with C:N indicates sufficient availability of N across the region relative to seagrass growth requirements. There is no indication of elevated N, despite %N remaining above the global median. The low δ^{15}N (e.g. decreasing at reef habitats), suggests negligible influence of an anthropogenic source of N (Figure 68).

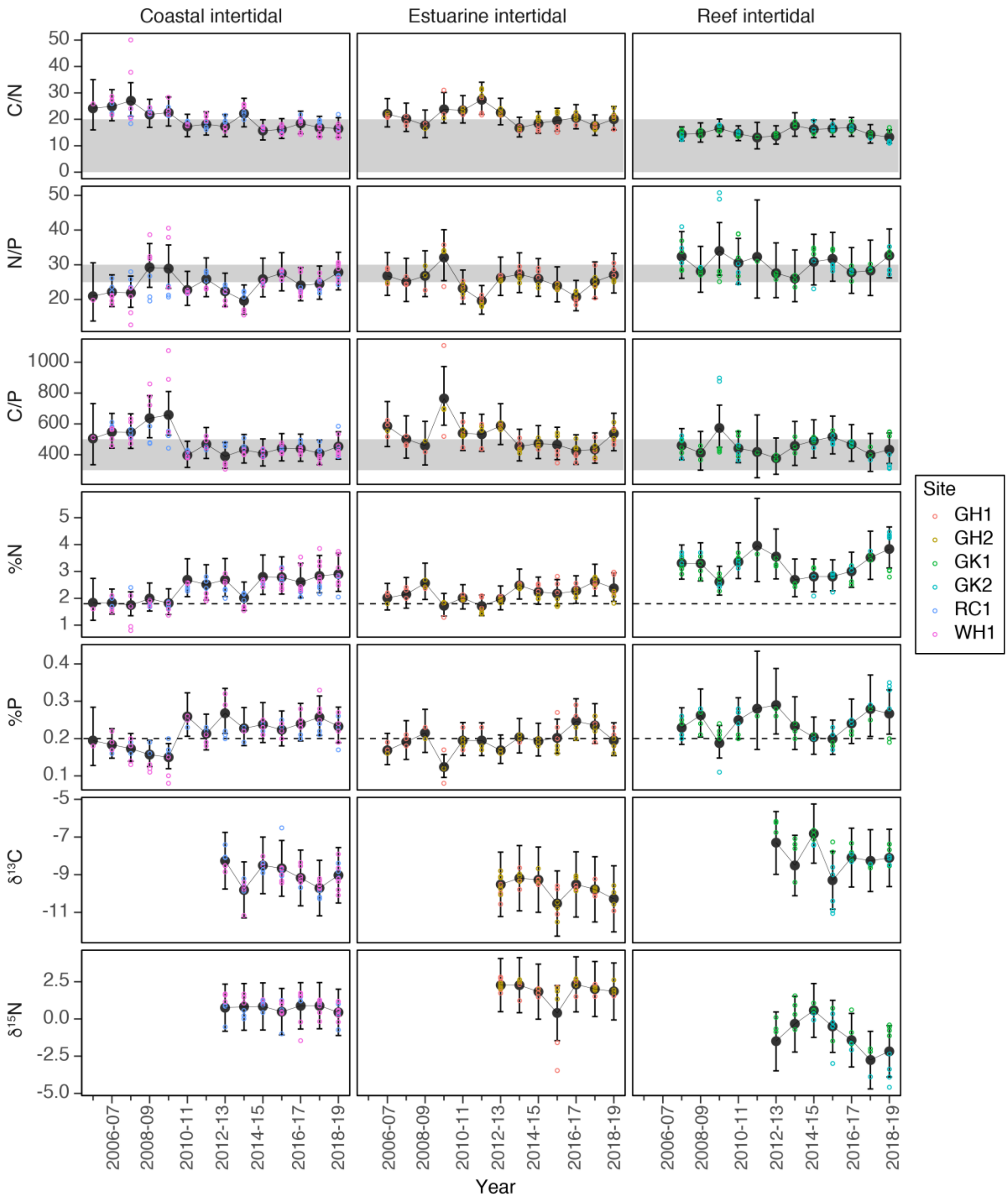


Figure 77. *Seagrass leaf tissue nutrient elemental ratios (C:N:P) and concentrations (%N, %P, $\delta^{13}C$ and $\delta^{15}N$) for each habitat in the Fitzroy region (± SE) (foundation species pooled). Horizontal shaded bands or dashed lines represents the accepted seagrass guideline values, where: C:N ratios within the band may indicate reduced light availability and/or N enrichment; N:P ratios above the band indicate P limitation, below indicate N limitation and within indicates replete, and; C:P ratios within the band may indicate nutrient rich habitats (large P pool). Dashed lines in %N and %P indicate global median values of 1.8% and 0.2% for tissue nitrogen and phosphorus, respectively (Duarte 1990).*

5.5.3.5 Epiphytes and Macroalgae

Epiphyte cover on the leaves of seagrass across the Fitzroy region either remained below the Reef-wide long-term average for the fifth consecutive year (estuarine and reef habitats), or declined (coastal habitat) in 2018–19 compared to the previous reporting year (Figure 78).

Macroalgae cover remained very low and unchanged at all habitats in the Fitzroy region, with the exception of a minor increase in the late wet 2018 at the reef habitat (Figure 78).

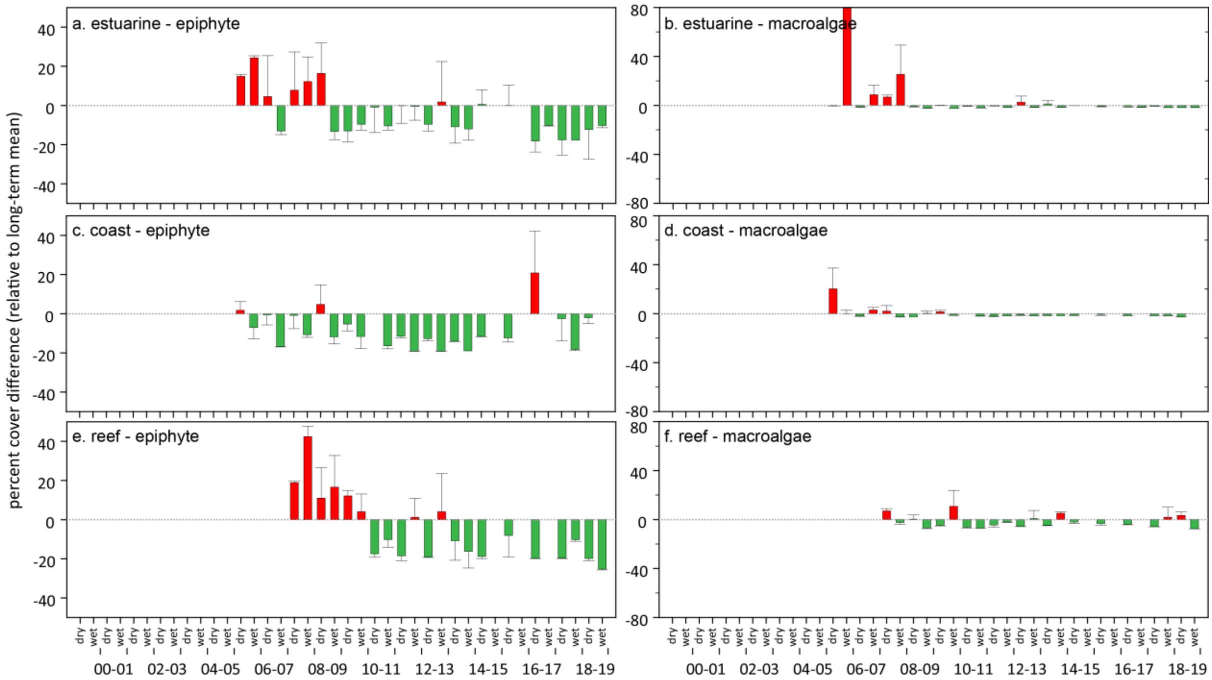


Figure 78. *Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average (2005-2018) for each inshore intertidal seagrass habitat in the Fitzroy region, 2005–2019 (sites pooled, ±SE).*

5.6 Burnett–Mary

5.6.1 2018–19 Summary

Inshore seagrass meadows across the Burnett–Mary NRM region increased in overall condition in 2018–19 and the index score rose from very poor to a **poor** grade (Figure 79). The scores of each of the indicators increased marginally but the grades for each of the indicators remained unchanged. Contributing indicators to the overall score were:

- abundance score was poor
- reproductive effort score was very poor
- tissue nutrient score was poor.

Seagrass abundance increased overall, but there are location-specific variations in the trends in the region. Abundance increased at Rodds Bay and Burrum Heads, but at Burrum Heads, there was a large increase in the proportion of colonising species. Abundance and meadow extent declined at Urangan and this was associated with increased activity of burrowing shrimps and the presence of fine mud.

The increased seed banks coupled with improved abundances in meadows in the north of the region may indicate an improved resilience with greater capacity for recovery if threatened by larger scale disturbances. However reproductive effort continues to remain very low across habitats in the south of the region, possibly limiting replenishment of seed bank.

In late 2018, seagrass leaf tissue nutrient concentrations and ratios continue to indicate surplus (elevated) availability of N in estuarine and coastal meadows; from natural N-fixation rather than anthropogenic sources. Although N availability may be high, it does not appear to have influenced epiphyte and macroalgae abundances which remain low across the region.

Rainfall and river discharge were below average, and yet all sites were exposed to high levels of turbidity, predominantly ‘brown’ water, for all weeks (100%) during the wet season. Within-canopy temperatures in 2018–19 were around the long-term average. This follows after five consecutive years where they were above the long-term average.

The marginal increase in Burnett–Mary region seagrass condition index in the 2018–19 follows the declines in 2016–17 and 2017-18, from the highest score in 10 years, and appears predominately driven by marginal improvments in all indicators.

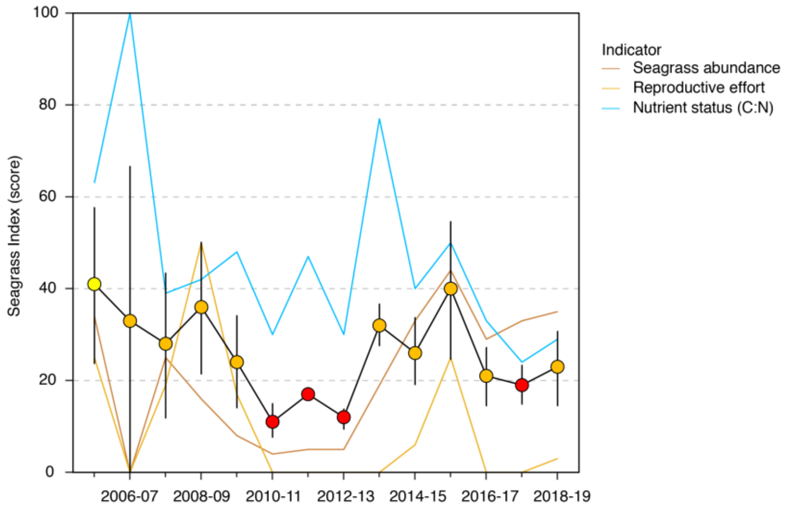


Figure 79. *Report card of seagrass index and indicators for the Burnett–Mary region (averages across habitats and sites). Values are indexed scores scaled from 0–100 and*

graded: ● = very good (81-100), ● = good (61 - 80), ● = moderate (41 - 60), ● = poor (21 - 40), ● = very poor (0 - 20). NB: Scores are unitless.

5.6.2 Climate and environmental pressures

During 2018–19, rainfall and river discharge in the Burnett–Mary region were below average (Figure 80, Table 9). But despite this, monitoring sites were exposed to turbid water, predominantly ‘brown’ turbid water for 100% of the wet season. The most significant flow events from the largest rivers last occurred in October 2017 (Gruber *et al.* 2019b).

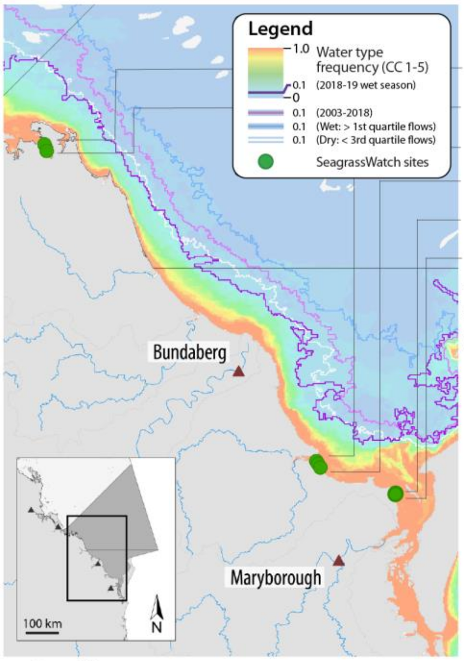
Within-canopy light was lower than the long-term average for the region as a whole (Figure 80, Figure 98). However, due to relocation of RD2 to RD3 and the recent addition of light monitoring to the Burrum Heads stes, it is difficult to assess trends in light levels at this time.

Within-canopy temperatures in 2018–19 were at the long-term average following on five consecutive years when they were above the long-term average (Figure 80). Maximum intertidal within-canopy temperatures exceeded 35°C for a total of 9 days during 2018–19, with the highest temperature recorded at 38.3°C (UG2, 1pm 20Dec18).

Although daily tidal exposure was below the long-term average for the region (Figure 80), levels of exposure differed with meadows in the north exposed for longer than those in the south (Figure 98). The less than long-term average exposure may have reduced the risk of temperature and desiccation stress in the south, but may also increase the risk of light limitation in the turbid water areas.

Sediments in the estuary seagrass habitats of the Burnett–Mary region are generally dominated by mud, but in 2018–19 the meadows in the south became more sandy with lower mud contact, while meadows in the north remained relatively stable, albeit with seasonal variability (Figure 120). Coastal meadows in 2018–19 continued to be dominated by fine sand with little change from the previous year (Figure 121).

a.



b.

Wk	1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20	21	22
RD1	5		1	3	1	1	4	1	3		4	1	5	2	4	2	4	1	4	4	5	4
RD2	4		1	1	1	1	1	1	2		4	1	4	2	4	1	3	1	3	2	4	4
BH1	3	5	1	4	4	4	4	1	4		3	1	5		4	1	4	3	4	1	4	4
BH3	2	2	1	2	1	2	2	1	1	2	2	1			2	1	1	1	1	1	1	2
UG1	5	5	2	3	2	3	4	1	4	3	3	2	1	2	4	2	4	2	3	2	4	4
UG2	5	5	2	3	2	3	4	1	4	3	3	2	1	2	4	2	4	2	3	2	4	4

1 - 4 = turbid primary water (red/brown), 5 = nutrient/chlorophyll-enriched secondary water (green), 6 = some freshwater/CDOM influence tertiary water (blue) , 7 = no plume influence (white), blank = no data, clouds (grey)

c.

	Long-term average	2018-19
Rainfall (1961-1990)	3.3 mm d^{-1}	2.1 mm d^{-1}
River discharge (1970-2018)	1,558,952 L yr^{-1}	930,309 L yr^{-1}
Turbid water exposure (2006-2018)	99.8%	100%
Daytime tidal exposure (1999-2018)	114 hr yr^{-1}	90 hr yr^{-1}
Within-canopy temperature - intertidal (2003-2018)	23.5°C (40.9°C)	23.5°C (38.3°C)
Within-canopy light	12.5 mol m^{-2} d^{1}	11.2 mol m^{-2} d^{1}

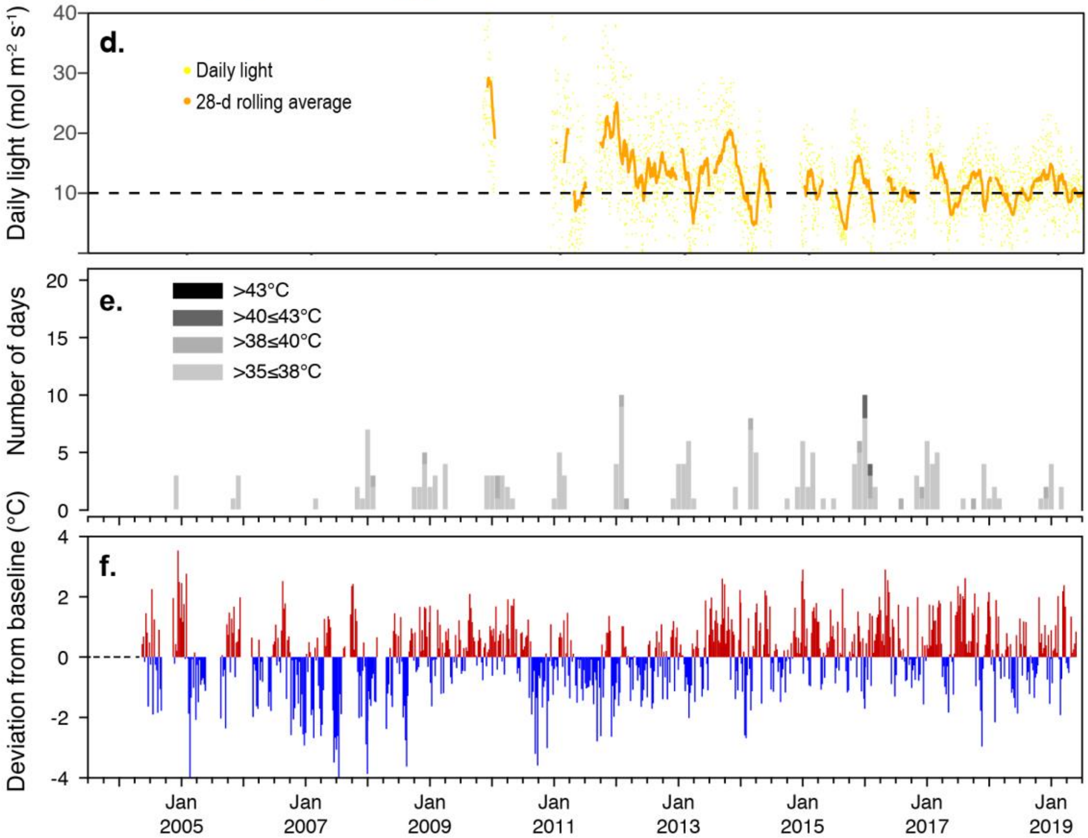


Figure 80. *Environmental pressures in the Burnett–Mary region including: a. frequency of exposure to turbid water (colour classes 1–5) (from Gruber et al. 2019b); b. wet season water type at each site; c. average conditions over the long-term and in 2018–19; d. daily light and the 28-day rolling mean of daily light for all sites; e. number of day temperature exceeded 35°C, 38°C, 40°C and 43°C, and; f. deviations from 13-year mean weekly temperature records.*

5.6.3 Inshore seagrass and habitat condition

There are 6 seagrass monitoring sites in the Burnett–Mary from 3 locations (Table 16). Only estuarine and coastal habitats were assessed across the Burnett–Mary region in 2018–19 (Table 16).

Table 16. *List of data sources of seagrass and environmental condition indicators for each seagrass habitat type in the Burnett–Mary NRM region. For site details see Table 2 and Table 3.*

Habitat	Site		abundance	composition	distribution	reproductive effort	seed banks	leaf tissue nutrients	meadow sediments	epiphytes & macroalgae
estuary intertidal	RD1	Rodds Bay	■	■	■	■	■	■	■	■
	RD2	Rodds Bay	■	■	■	■	■	■	■	■
	UG1	Urangan	■	■	■	■	■	■	■	■
	UG2	Urangan	■	■	■	■	■	■	■	■
coastal intertidal	BH1	Burrum Heads	■	■	■	■	■	■	■	■
	BH3	Burrum Heads	■	■	■	■	■	■	■	■

5.6.3.1 Seagrass index and indicator scores

In the 2018–19 monitoring period, the Burnett–Mary region seagrass condition index increased from very poor to a poor grade (Figure 81). The increase this year follows two years of decline since the seagrass index peaked in 2015–2016, predominately driven by declining nutrient status and very low reproductive effort (Figure 81).

Over the long term, seagrass abundance regionally has fluctuated greatly (e.g. periods of loss and subsequent recovery). Increases between 2012 and 2016 placed the meadows on a pathway towards recovery. The long-term trend suggests that the losses observed in 2016–2017 and 2018–19 may not be part of a declining trend (Table 20), despite reduction in the abundance score.

Similarly, an examination of the long term trends across the Burnett–Mary region using GAM plots suggests tissue nutrient elemental C:N has no discernible trend since 2005 (Figure 81).

Reproductive effort, however, appears generally low with occasional increases in the number of reproductive structures corresponding to increased seagrass abundance (Figure 81).

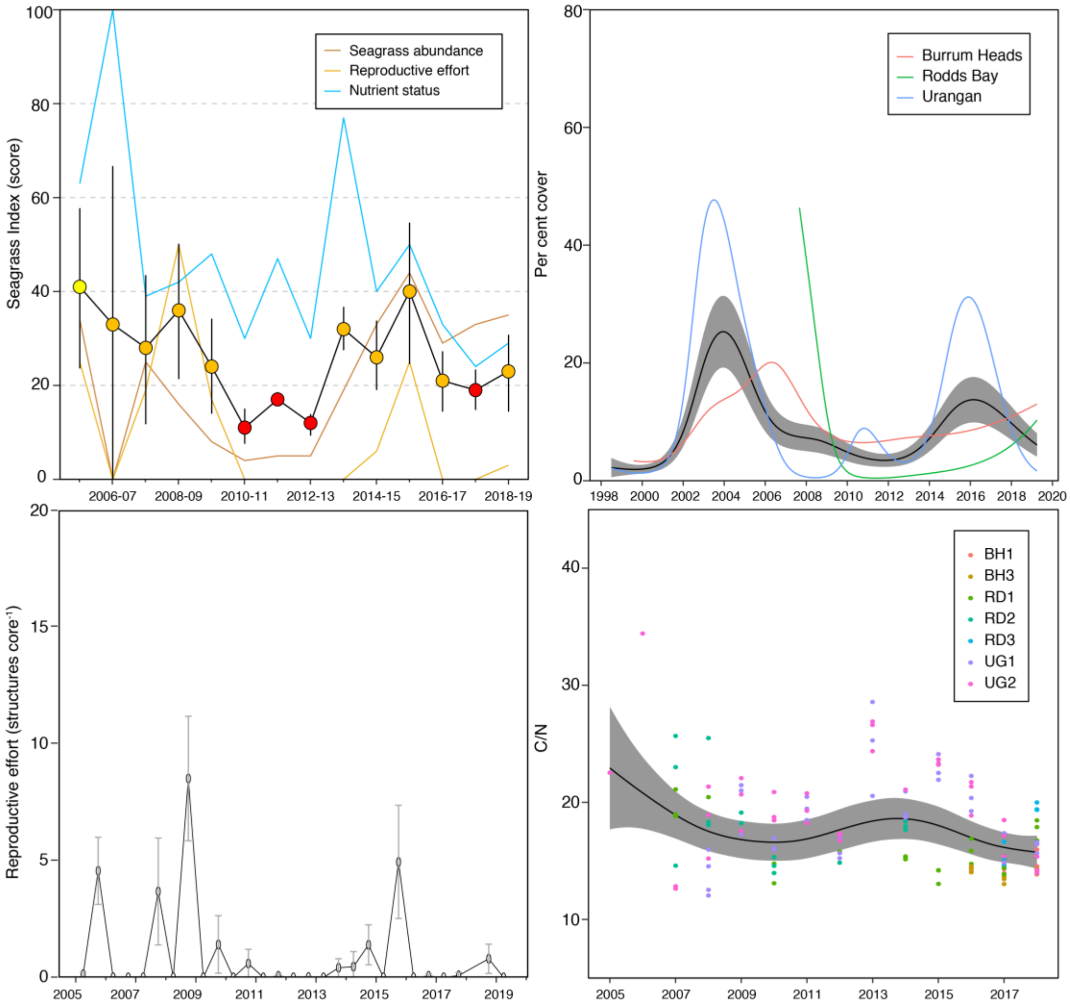


Figure 81. *Temporal trends in the Burnett–Mary seagrass condition index and the indicators used to calculate the index: a. seagrass condition index (circles) and indicator trends (lines); b. GAM plots of seagrass abundance (per cent cover) trends for each location (coloured lines) and the region (black line with grey shaded area defining 95% confidence intervals); c. average number of reproductive structures (±SE) (GAM not possible due to high count of zero values); and d. elemental ratios (atomic) of leaf tissue C:N nutrient content at each site (coloured circles) and regional trend represented by a GAM plot as dark line with shaded areas defining 95% confidence intervals of the trend.*

5.6.3.2 Seagrass abundance, composition and extent

Seagrass abundances (per cent cover) across the Burnett–Mary region in 2018-19 were greater in coastal than estuarine habitats (14.7 ±0.6% and 8.2 ±1.2%, respectively), however estuarine abundances were higher in the late dry than the late wet season (11.7 ±1.4% and 4.8 ±1.0%, respectively). Two thirds of monitoring sites increased in abundance in 2018–19 relative to the previous period, with the remaining third continuing to decline. Only the estuarine meadows at Urangan declined in 2018–19.

Since monitoring was established, the estuarine meadows have come and gone on an irregular basis. The only site to significantly decline over the long-term, was in the north of the region in the Rodds Bay estuary (RD2), however this decline was due to changes in the intertidal bank topography which rendered the site no longer suitable for ongoing monitoring. In the south, both an estuary and a coastal site have significantly increased over the long-term, while no trend is apparent at the remaining monitoring sites (Table 20).

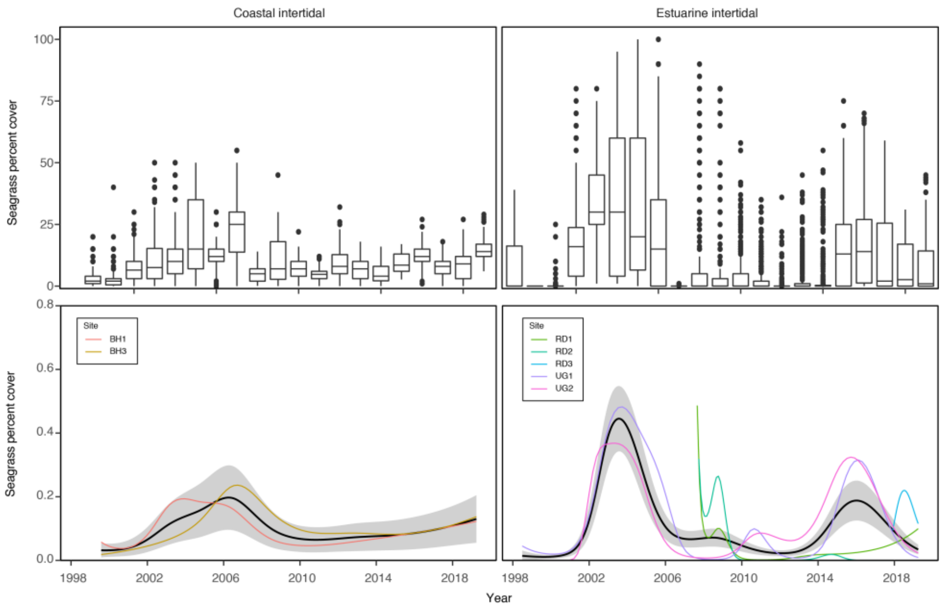


Figure 82. *Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the Burnett–Mary NRM region from 1999 to 2019. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates the 25th percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75th percentile. Whiskers (error bars) above and below the box indicate the 90th and 10th percentiles, and the dots represent outlying points. GAMM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.*

The estuarine and coastal seagrass habitats have been dominated by *Zostera muelleri* with varying components of *Halophila ovalis*. In 2018–19, the proportion of colonising species increased compared to the previous monitoring year, and in the coastal meadows exceeded the Reef long-term average (Figure 83). An increase in the proportion of colonising species in the meadows suggests some level of physical disturbance which may reduce ability to tolerate/resist major disturbances in future.

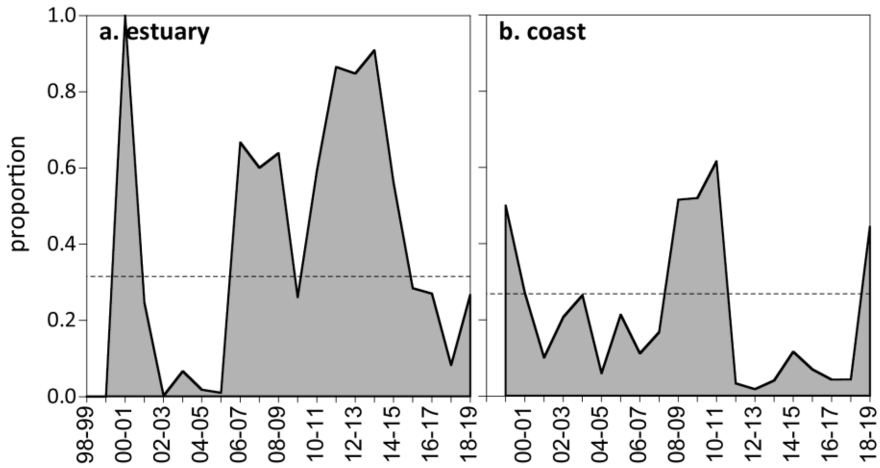


Figure 83. *Proportion of seagrass abundance composed of colonising species at: a. estuary and b. coastal habitats in the Burnett–Mary region, 1998–2019. Dashed line represents Reef long-term average proportion of colonising species for each habitat type.*

Over the last 12 months meadow extent has changed little at coastal meadows relative to the previous year (Figure 84). Estuarine meadows, however, declined in extent. This decline was restricted to meadows in the south (Urangan) which have fluctuated greatly with periods of decline, absence and recovery over the life of the MMP.

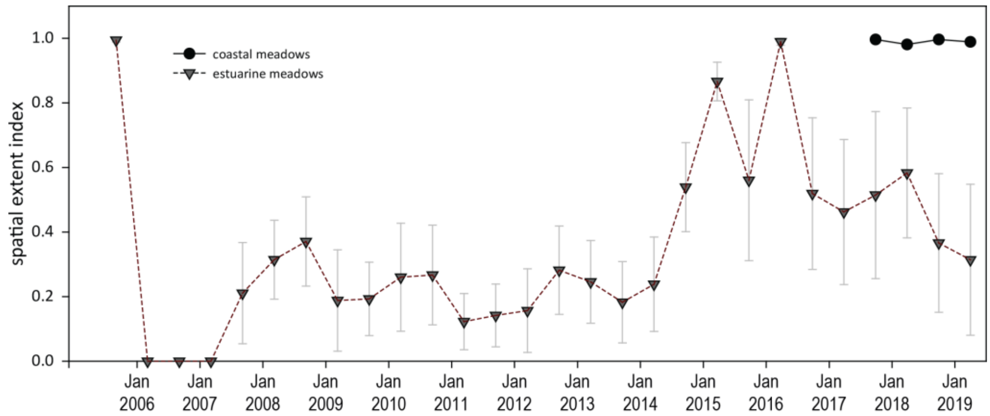


Figure 84. *Change in spatial extent of estuary seagrass meadows within monitoring sites for each habitat and monitoring period across the Burnett–Mary NRM region.*

5.6.3.3 Seagrass reproductive status

Seagrass reproductive effort remained at zero across coastal habitats this year, but increased at estuarine sites in the dry season compared to the previous monitoring period (Figure 85). A seed bank persists at all meadows monitored across the region, which was greater at estuary sites in 2018–19 than the previous period (Figure 85). This may indicate the meadows have a greater capacity to recover from the declining abundances, provided conditions are favourable.

The apparent disconnect between reproductive effort and seed densities may be an artefact of the sampling frequency and the somewhat stochastic triggers and possibly short flowering period.

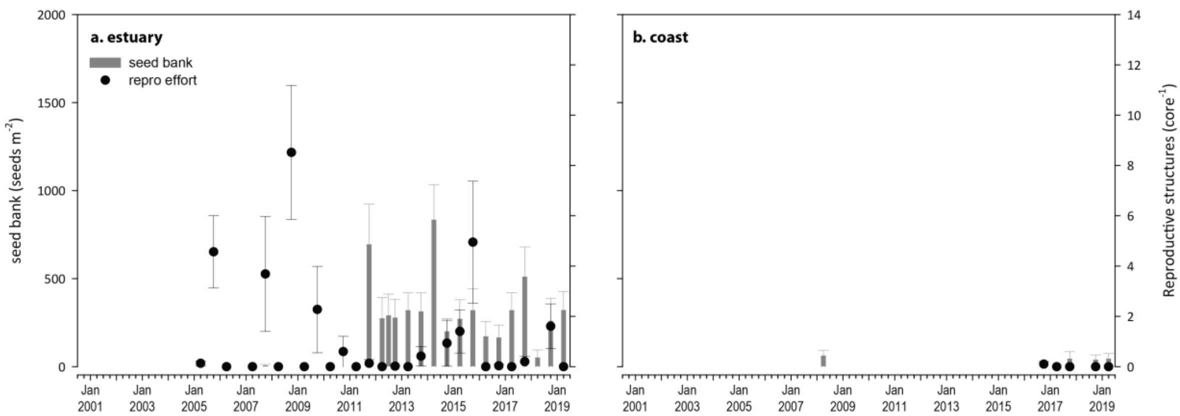


Figure 85. *Burnett–Mary estuary seed bank and reproductive effort. Seed bank presented as the total number of seeds per m^2 sediment surface and reproductive effort presented as the average number of reproductive structures per core (species and sites pooled).*

5.6.3.4 Seagrass leaf tissue nutrients

In 2018, *Zostera muelleri* leaf tissue molar C:N, C:P and N:P ratios remained steady at the estuary sites compared to the previous year (Figure 86). This indicates a surplus of N relative to photosynthetic C incorporation, and equally high P availability. The marginally less negative $\delta^{13}C$ indicates improved light availability, particularly in the meadows to the north of the region (Rodds Bay).

At the coastal sites, seagrass (*Halodule uninervis* and *Zostera muelleri*) leaf molar C:N ratios were similar to the previous year, remaining below 20 (Figure 86), indicating a surplus of N relative to photosynthetic C incorporation. N:P ratios remained very high (above 40), which is higher than global median %N, indicating surplus (elevated) availability of N (Figure 86).

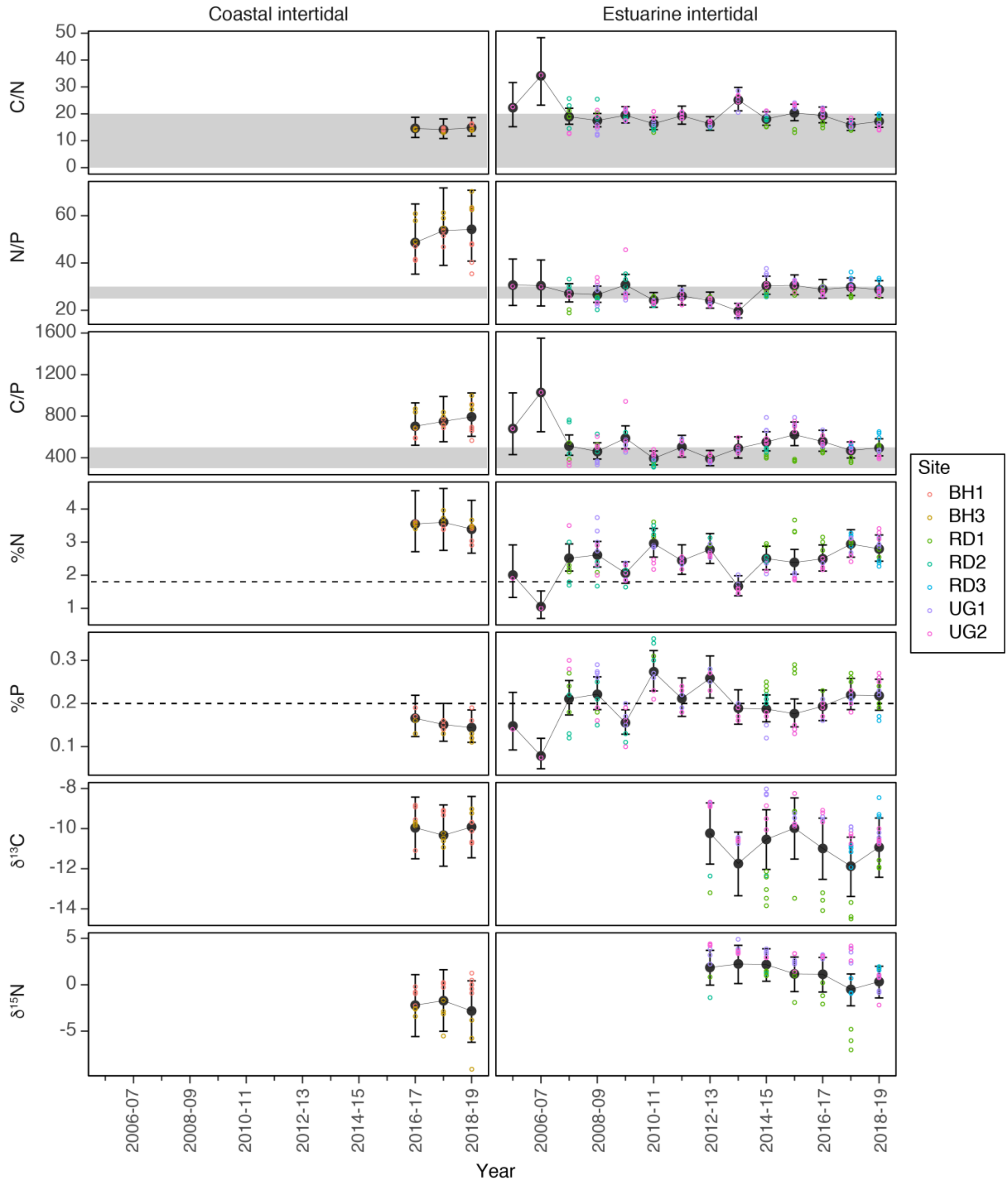


Figure 86. *Seagrass leaf tissue nutrient elemental ratios (C:N:P) and concentrations (%N, %P, $\delta^{13}C$ and $\delta^{15}N$) for each habitat in the Burnett–Mary region (± SE) (foundation species pooled). Horizontal shaded bands or dashed lines represents the accepted seagrass guideline values, where: C:N ratios within the band may indicate reduced light availability and/or N enrichment; N:P ratios above the band indicate P limitation, below indicate N limitation and within indicates replete, and; C:P ratios within the band may indicate nutrient*

rich habitats (large P pool). Dashed lines in %N and %P indicate global median values of 1.8% and 0.2% for tissue nitrogen and phosphorus, respectively (Duarte 1990).

5.6.3.5 Epiphytes and macroalgae

Epiphyte cover on seagrass leaf blades in 2018–19 remained higher than the long-term average for the fifth consecutive year at estuarine habitats (Figure 87). Alternatively, at coastal habitats, the epiphyte abundance has remained below the long-term average for the third consecutive year (Figure 87).

Per cent cover of macroalgae has remained low and below the long-term average at across all habitats monitored (Figure 87), with the exception of a slight incease in estuarine habitats in the late wet 2019.

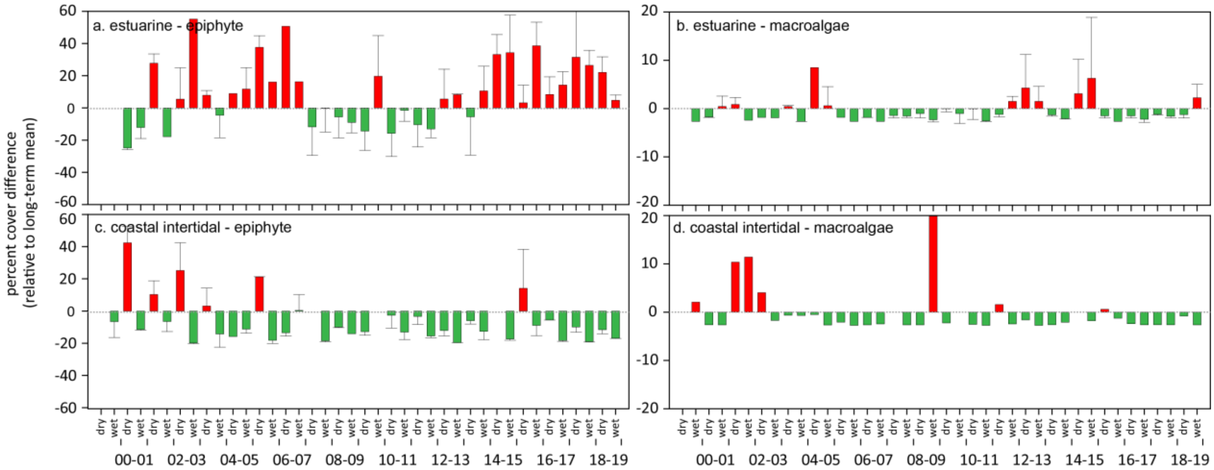


Figure 87. *Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average for each seagrass habitat in the Burnett–Mary NRM region (sites pooled, ±SE).*

6 Discussion

6.1 Seagrass resilience

While seagrass meadows of the Reef are inherently dynamic, poor recovery rates at many locations and poor resilience (e.g. reproductive effort and seed density), indicate that future recovery capacity following impacts is compromised. This, coupled with intensifying disturbances continued to present a concerning outlook.

Throughout the inshore Reef, the rate of seagrass recovery since 2011 has been protracted in some locations compared to previous reports (e.g. Birch and Birch, 1984; Campbell and McKenzie 2004b), particularly at reef habitats. Low reproductive effort may be a contributing factor. At some of the reef sites reproductive structures are never observed for some species, while at others there is some reproductive effort but seed banks are not forming or persisting either because no seeds are being produced, or seeds are lost through other processes, such as predation (Orth *et al.* 2006). The presence of seeds is fundamental to building resilience at reef sites, as without them the meadows remain vulnerable to large disturbances and would need to rely on recruitment of propagules from other meadows. This external recruitment process may operate at timescales ranging up to centuries or millennia depending on whether the propagules are reproductive or through clonal expansion (Grech *et al.* 2016, McMahon *et al.* 2014). Absence of a seed bank at some sites and poor reproductive effort across the Reef, has left many of the MMP meadows vulnerable to further environmental perturbations.

Recovery of seagrass meadows proceeding slower than expected might also be due to the frequent and repeated disturbances occurring over the decade. The capacity of seagrass meadows to naturally recover requires environmental conditions that will enable expansion, sexual reproduction and seed bank formation, including optimum conditions of light and nutrient availability and the absence of major physical disturbances such as cyclones or even excessive sediment resuspension. For example, the low and variable light availability across the Reef habitats in 2014–15, 2016–17, 2017-18 and 2018–19 appears to have slowed and abated recovery, which in turn may reduce capacity to produce viable seed banks in some locations (van Katwijk *et al.* 2010). Continued strategic monitoring through world-leading monitoring networks such as such as Seagrass-Watch (Duffy *et al.* 2019), as well as integration with complementary monitoring programs through the Reef Integrated Monitoring and Reporting Program (RIMReP), will enable continued assessment of their trajectories.

6.2 Seagrass ecosystem service provisioning

The ecosystem services provided by seagrass ecosystems makes them a high conservation priority (Cullen-Unsworth and Unsworth 2013; Unsworth *et al.* 2018a). Certain seagrasses are the primary food for marine green turtles and dugongs, which are seagrass specialists (Read and Limpus 2002; Arthur *et al.* 2008; Marsh *et al.* 2011). Seagrass form highly productive habitats for a large number of invertebrates, fish and algal species (Carruthers *et al.* 2002), which are important to commercial (e.g. prawns) and subsistence fisheries (Coles *et al.* 1993; Cullen-Unsworth and Unsworth 2013). Seagrass also produce natural biocides and improve water quality by controlling pathogenic bacteria to the benefit of humans, fishes, and marine invertebrates such as coral (Lamb *et al.* 2017). Nutrient cycling in seagrass meadows makes them one of the most economically valuable ecosystems in the world (Costanza *et al.* 1997) and the retention of carbon within their sediments contributes significantly to Blue Carbon sequestration (Fourqurean *et al.* 2012; Unsworth *et al.* 2012a; Duarte and Krause-Jensen 2017; Macreadie *et al.* 2017). Inshore seagrass meadows, which are the focus of this monitoring program, represent only 10% of the total seagrass area estimated within the World Heritage Area (McKenzie *et al.* 2010b), but many of these services are more pronounced in the inshore meadows.

Much of the connectivity in reef ecosystems depends on intact and healthy non-reef habitats, such as seagrass meadows (Waycott *et al.* 2011). These non-reef habitats are particularly important to the maintenance and regeneration of populations of reef fish such as Emperor fish (*Lethrinus spp*) and Tuskfish (*Choerodon spp*) (Cullen-Unsworth *et al.* 2014).

In addition, the incorporation of carbon within seagrass tissues can affect local pH and increase calcification of coral reefs, thereby mitigating some effects of ocean acidification (Fourqurean *et al.* 2012; Unsworth *et al.* 2012a). Therefore, monitoring changes in seagrasses meadows not only provides an indication of coastal ecosystem health, but also improves our capacity to predict changes to resources upon which coastal communities depend (Heck *et al.* 2008).

Chronic declines in inshore water quality in the Reef since European settlement have contributed to major ecological shifts in a few Reef marine ecosystems (De'ath and Fabricius 2010; Roff *et al.* 2013). The scientific evidence is clear: initiatives that will halt and reverse the effects of climate change at a global level and effectively improve water quality at a regional scale are the most urgent to improve the Region's long-term outlook (GBRMPA 2019). Flood waters deliver terrestrially sourced pollutants (e.g. sediments, nutrients, pesticides) dispersing them over the sensitive ecosystems including seagrass meadows (summarised in Schaffelke *et al.* 2013).

6.3 Emerging priorities for management

As seagrasses within the World Heritage Area provide considerable ecosystem services, they are high on the list of ecosystems to prioritise for management response (Unsworth *et al.* 2018b). Practicable conservation opportunities exist and have been implemented, which can make substantial and quantifiable improvements to seagrass condition. Management initiatives that target reversing wider-scale catchment degradation and poor water quality (i.e. Paddock 2 Reef), will have the greatest benefit to inshore seagrass by reducing overall stress and improving resilience. In particular, reducing suspended particulate matter loads (SPM) is important as these present the greatest risk to inshore seagrasses through impacts to water clarity (Waterhouse et al 2017, Bainbridge et al 2018). Minimising localised pressures from coastal and urban runoff will also reduce cumulative stress. This program can provide additional guidance on management via the following tasks:

1. Through the data collected in this monitoring program we are building information about recovery rates, and are better positioned now, than ever before, to develop accurate recovery models. These are needed to enable managers to decide when and what management interventions may be required to enhance recovery. These models also need to be tailored to this program, and the data produced from this program, so that we can provide quantitative estimates of recovery in annual reports.
2. Updating risk assessments within an adaptive management framework will enable us to know whether the program continues to be measuring and reporting against the the most important stressors. Furthermore, we need to update pressures thresholds (e.g. light and temperature) to consider variation arsing from site-specific factors including species differences and habitat differences, or cumulative pressures. These will enable information on pressures from site-level monitoring to be used for predicting conditions across the seagrass meadows of the Reef. For example see Lawrence (2019).
3. Scaling the site-level monitoring undertaken in this program to provide information on seagrass condition across the Reef underpins the information needs for Outlook and the Scientific Consensus statement. Initiatives such as the reef Integrated Monitoring and Reporting Program can help with this to some extent, but only if resources are made available for monitoring over a broader scales and for data integration.
4. Adaptive management also requires that indicators and metrics are assessed and possibly revised as new information is available. This will ensure that managers are being provided with the most relevant assessment of Reef condition and resilience. As a case study to this report we have assessed the tissue nutrient indicator (See

Appendix 1). The reproductive metric also requires re-assessment, as the importance of sexual reproduction for resilience varies amongst species (Kilminster et al 2015) but the existing metric only accounts for this through habitat-specific guidelines. That assessment will be presented in the 2019-20 annual report as a case study. Together these will be used to update the seagrass Index.

5. Active restoration or enhancement of resilience may be required in some locations (van Oppen *et al.* 2017) and as such, restoration strategies to enhance resilience and promote recovery may be a timely exploration. In the first instance, the basis of poor and variable reproductive effort should be investigated as a matter of priority because the absence of these in most reef habitats and some coastal and estuarine sites inhibits natural recovery. For example, the lack of flowers and fruits may be due to sampling artefact (e.g. timing and frequency of sampling may miss short flowering periods), but this cannot be the only explanation as seeds of some species can persist for many years and would be detected in seed banks if they were produced. There may be communities unable to reproduce due to their effective population size being reduced to a critical threshold. This is known to have happened for *Cymodocea serrulata* and *Syringodium isoetifolium* on Green Island and *Thalassia hemprichii* at Magnetic Island where some meadows are made up of a single clone (and therefore a single sex as these species are dioecious) leading to their inability to set seed (McKenzie *et al.* 2014c; Collier *et al.* 2016c). Improved understanding of what affects reproduction can help in the development of recovery models (see task 1 above), for assessing risk (task 2) and for assessment of metrics (task 4), and to underpin management strategies that enhance resilience.

7 Conclusion

This year inshore seagrass meadows across the Reef declined for the second consecutive year in overall condition, with the condition grade remaining **poor**. Declines were primarily as a result of continued exposure to brown and green waters and the legacy of severe climate events in the previous year, which has reduced resilience and increased vulnerability of seagreass to future disturbances.

In 2018–19, the inshore seagrass of the Reef was graded in a **poor** condition in all NRM regions: the northern section of the Wet Tropics NRM was graded as moderate however offset by the southern section which was graded as poor. The southern Wet Tropics and Burnett–Mary were the only regions to improve their grade, from very poor in the previous year to poor. Seagrass condition in the Fitzroy has been poor, or very poor for an extended period including abundance and reproduction indicators.

The Reef occurs in a climate belt where variable rainfall patterns and cyclones, and increasingly in recent years — marine heatwaves — creates frequent disturbances moving up and down the 2,300 kilometre coastline creating complex and varied environmental conditions (Babcock *et al.* 2019). Climatic conditions in 2018–19 were above the long-term average. For example, rainfall and river discharge across all basins from the central to far northern Reef regions during the 2018–19 wet season exceeded their long-term medians, while below median discharges occurred in the southern Reef regions.

Three cyclones crossed the inshore areas of the Reef in 2018–19, which may have elevated exposure to 'brown' sediment-laden (1–4) and 'green' phytoplankton-rich waters (5), and exacerbated inshore turbidity and disturbance, particularly in the far northern region. The most significant environmental conditions affecting inshore seagrasses in 2018–19 were lower benthic light availability across nearly half the meadows monitored, with light levels lower than annual light requirements (10 mol m^{-2} d^{-1}) at six locations, and lower than the long-term average at 13 locations. The legacy effects of severe climate events in previous years (e.g. cyclone Debbie and associated flooding), coupled with elevated temperatures continues to be reflected in the condition of some meadows.

Tropical seagrasses of the Reef are a mosaic of different habitat types with multiple seagrass species assemblages. At a habitat level, those in poorest condition were seagrass within reef habitats, specifically intertidal and subtidal reef habitats which have consistently had seagrass with very poor reproductive effort and low or no seeds in the seed banks. Seagrass within subtidal reef habitats have shown variable or little sign of recovery in abundance following 2011.

Trends

Seagrass meadows of the Reef are dynamic, with large changes in abundance being seemingly typical (e.g. Birch and Birch 1984; Preen *et al.* 1995; Campbell and McKenzie 2004; Waycott *et al.* 2007), but the timing and mechanisms that cause their dynamism (i.e. declines and subsequent recovery) are complex.

Declines in seagrass abundance occurring in 2006 and then from 2009 to 2012 (from Cooktown south) abated in late 2012 and seagrass condition, although remaining poor, had been improving until 2017 (Figure 88). More specifically, although some locations in the Wet Tropics and Burdekin regions experienced declines in early 2006 as a consequence of cyclone Larry, most recovered within 1–2 years; with the exception of the coastal sites in southern Wet Tropics where recovery was protracted.

In late 2008, locations in the northern Wet Tropics and Burdekin regions were in a moderate state of health with abundant seagrass and seed banks. In contrast, locations in the southern Mackay–Whitsunday and Burnett–Mary regions were in a poor state, with low abundance, reduced reproductive effort and small or absent seed banks (Figure 88).

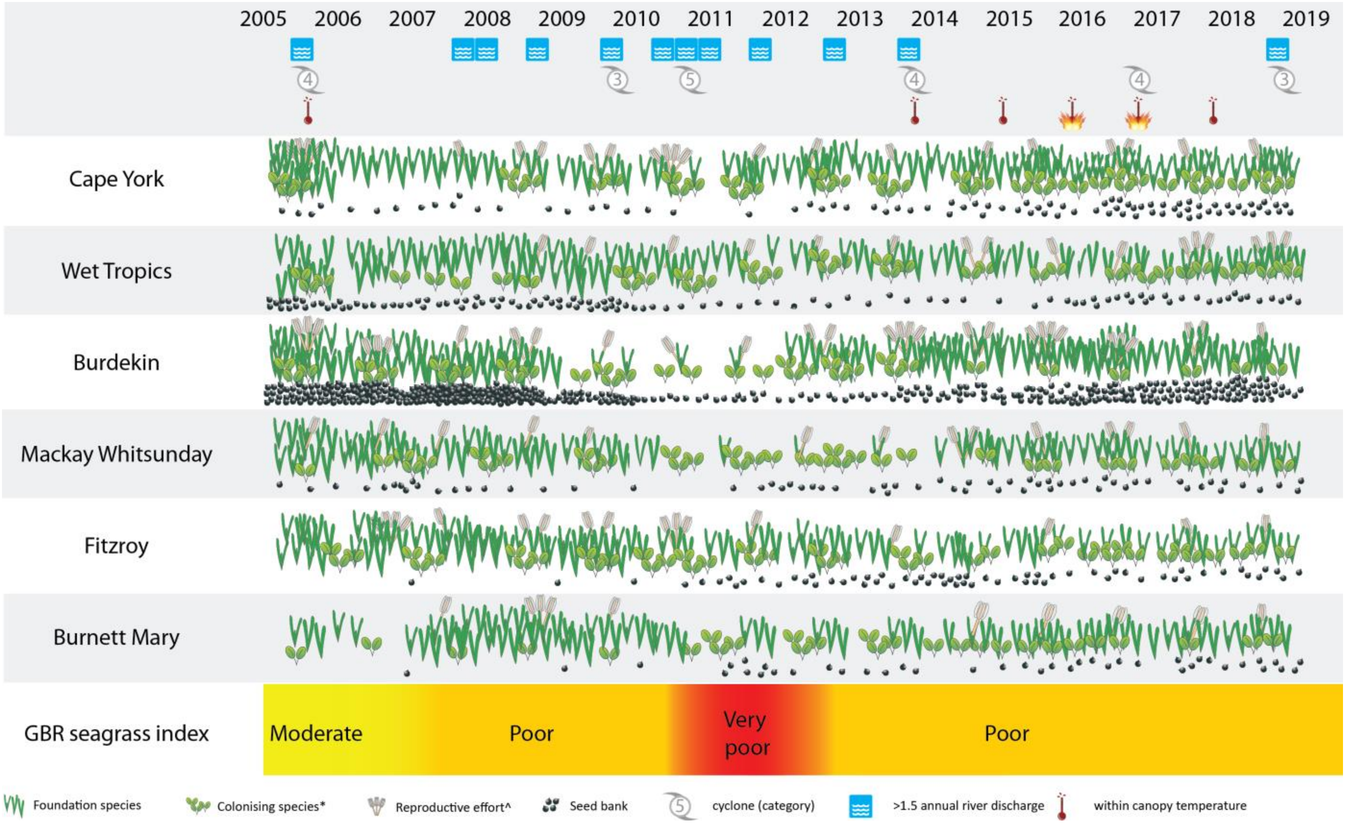


Figure 88. *Summary of inshore seagrass state illustrating pressures, abundance of foundation / colonising species, seed banks and reproductive effort from 2005 to 2019. * colonising species are represented by the genus Halophila, however, Zostera and Halodule can be both colonising and foundational species depending on meadow state. ^ not conducted in 2005.*

In 2009 with the onset of the La Niña, the decline in seagrass state steadily spread across the Burdekin region and to locations within the Fitzroy and Wet Tropics where discharges from large rivers and associated catchments occurred (McKenzie *et al.* 2010a; McKenzie *et al.* 2012). The only locations of better seagrass state were those with relatively little catchment input, such as Gladstone Harbour and Shoalwater Bay (Fitzroy region), Green Island (northern Wet Tropics), and Archer Point (Cape York) (McKenzie *et al.* 2012).

By 2010, seagrasses of the Reef were in a poor state with declining trajectories in seagrass abundance, reduced meadow extent, limited or absent seed production and increased epiphyte loads at most locations. These factors would have made the seagrass populations particularly vulnerable to large episodic disturbances, as demonstrated by the widespread and substantial losses documented after the floods and cyclones of early 2011.

Following the extreme weather events of early 2011, seagrass habitats across the Reef further declined, with severe losses reported from the Wet Tropics, Burdekin, Mackay–Whitsunday and Burnett–Mary regions. By 2011–12, the onset of seagrass recovery was observed across some regions, however a change had occurred in which colonising species dominated many habitats. The majority of meadows appeared to allocate resources to vegetative growth rather than reproduction, indicated by the lower reproductive effort and seed banks. In 2016–2017, recovery had slowed or stalled across most of the regions.

The Wet Tropics and Fitzroy regions have shown the most protracted recovery rates, though the causes for this differ between the regions. In the Fitzroy region declines up to early 2011 were more moderate than in other regions, but the estuarine intertidal and coastal intertidal habitats declined further in 2013–2015, and recovery had since been slow except in coastal habitats. Abundance in the Wet Tropics declined in early 2011, and recovery has been delayed. In the southern Wet Tropics, it appears that sediment scouring caused by cyclone

Yasi in 2011 altered bed elevation and substrate composition. The growth substrate is not routinely measured however it does appear that seed banks are recovering at Lugger Bay and the sediments becoming less gravelly at Dunk Island. As a consequence the seagrass has been able to establish in patches and hopefully will continue to recover. By contrast, slow recovery in the northern Wet Tropics reef sites (Low Isles intertidal and subtidal and Green Island subtidal) may be affected by water quality.

There was increasing evidence that water quality degradation within the seagrass meadows of the inshore Reef prior to the episodic disturbances of 2011 may have reduced their resilience. Light availability is one of the primary driving factors in seagrass growth and persistence (Collier and Waycott 2009; Brodie *et al.* 2013;Collier *et al.* 2012b). Seagrasses can survive in highly turbid sites if restricted to shallow areas where light reaches the canopy around low tide (Petrou *et al.* 2013). Conversely, infrequent low tide exposure occurring in summer months when water can be very turbid, coincident with high water temperatures, drives faster rates of decline (Collier *et al.* 2016a).

From 2009, reduced canopy light to low and limiting light levels was reported in seagrass meadows across the Reef, and, coincident with this, nutrients (N and P) increased relative to plant requirements. Conditions in the years leading up to 2011 were extremely turbid and were correlated with seagrass decline (e.g. Collier *et al.* 2012b; Petus *et al.* 2014). Since then, there have been periods of low light and exceedance of light thresholds, but the low light levels have not been as extreme (as low light, or for as long). The meadows have continued in a state of recovery, and the biological processes of recovery appears to complicate the response to environmental stressors.

Conditions conducive for seagrass growth, with a reduction in disturbance events are needed for the Reef's inshore seagrass meadows to improve from their current poor state and weakened.

To maintain a comprehensive understanding of seagrass condition and trend, and the factors driving change in these systems improved interdisciplinary and multidisciplinary ecosystem science on resilience and recovery need to be maintained and strengthened. In conjunction with that research, it is important to continue to refine the MMP, particularly as, after 15 years of monitoring, there is substantially more information now that can be used to address emerging priorities, namely: 1. recovery models, 2. risk assessment and scaling of pressures data, 3. scaling of seagrass condition data, 4. re-assessment of metrics; and, 5. an assessment of what affects seagrass reproductive effort to inform points 1 to 4.

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Appendix 1 Case study

Leaf tissue nutrient C:N ratio in relation to water quality: Model assessment and implications for report card metrics

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Executive summary

Seagrass leaf tissue C:N is one of three metrics that form the seagrass health Index for annual reporting through the Paddock to Reef Program. Leaf tissue C:N is a measure that integrates concentrations of nitrogen in the water column and sediment, as well as photosynthetic carbon uptake and light availability. The suitability of leaf tissue nutrient as a metric is reported from a retrospective analysis, which aimed to:

1. summarise leaf tissue C:N from 2005 to 2018. C:N was measured in all foundational species[1] in the late dry season. The median C:N for seagrasses of the Reef was 17.3, but there was variability among the different seagrass species.
2. test whether C:N varies in a consistent and predictable way related to past water quality. There is no on-going nutrient monitoring at seagrass monitoring sites, so the annual dissolved inorganic nitrogen (DIN) exposure was estimated from a 2-dimensional loading model, and benthic light reaching seagrass leaves was measured using *in situ* light loggers and summarised as an annual average. There was a weak relationship between relative DIN exposure and C:N for all species pooled, whereby C:N ratios decreased as DIN exposure increased. There was also a relationship between light and C:N ratios, where greater light availability correlated with higher C:N ratios, although this trend varied by habitat and sub-region. These trends were the most distinct in coastal habitats where there is high inter-annual variability in relative DIN exposure and light, but in estuarine and reef habitats, the response to DIN exposure was affected by the light history of the site. These relationships were also less distinct when testing on species individually, and when comparing against *in situ* water quality monitoring data (available for reef sites only).
3. test whether C:N is an early-warning indicator of changes in seagrass condition. There was no consistent trend in the C:N indicator and future changes in seagrass per cent cover, indicating that it is not an early-warning indicator when applied over the time-scales tested here. The inconsistent and unpredictable responses of C:N to the water quality variables may be related to the lack of variation in the water quality variables when summarised over annual time-scales. Furthermore, our models may also have been unable to account for the complexity of environmental conditions, and the dynamic nature of seagrass meadows in the Reef.
4. explore the effect of including C:N as a metric in the seagrass health index. In general, the C:N metric elevates the seagrass health score, and as such down-weighting or removing the C:N score reduces the overall index, especially in the Fitzroy and Burnett-Mary regions.

[1] A foundation species is the dominant primary producer in an ecosystem both in terms of abundance and influence, playing central roles in sustaining ecosystem services (Angelini et al. 2011). For the seagrass habitats assessed in this study, the foundation seagrass species were those species which typified the habitats both in abundance and structure when the meadow was considered in its steady state (opportunistic or persistent) (Kilminster et al. 2015). The foundation species were all di-meristematic leaf-replacing forms from the following families: *Cymodocea, Enhalus, Halodule, Thalassia* and *Zostera*.

Introduction

Water quality in the inshore Reef has declined since European settlement as a result of increasing loads from land-based run-off of sediment, nitrogen and phosphorus (Schaffelke et al. 2017). This is a major cause of concern for many of the inshore coastal and marine ecosystems (Waterhouse et al. 2017). The Great Barrier Reef Marine Monitoring Program (MMP) tracks the health of coral and seagrass as key habitats of these inshore ecosystems. Seagrasses are well-recognised indicators of environmental pressures such as water quality, and are vitally-important component of the inshore ecosystems of the Reef (McKenzie et al. 2020).

Water quality can affect seagrasses through multiple impact pathways. Suspended particulate matter (SPM), including sediment, plankton and detritus, attenuates light as it passes through the water column and reduces the light reaching benthic communities such as seagrasses (Waycott et al. 2009; Kirk 2011; Bainbridge et al. 2018). Less light is therefore available for photosynthetic carbon fixation, which can result in lower internal carbon concentrations and reduced seagrass leaf tissue carbon content. Persistently low carbon fixation and the ensuing carbon deficit leads to depletion of internally stored carbon reserves and to physiological and structural modifications of the seagrass plant, such as leaf senescence and changes in leaf dimensions (Collier et al. 2009a; Collier et al. 2012a; McMahon et al. 2013; O'Brien et al. 2018). Shoot loss and mortality can result from persistently low light conditions (Chartrand et al. 2016). As the SPM settles on seagrass leaves and on the benthic substrate, it can also affect oxygen diffusion into leaves and affect the biogeochemical condition of the sediment (Brodersen et al. 2015; Brodersen et al. 2017) both of which exacerbate the impact of light reduction (Bainbridge et al. 2018). Being able to detect risk of habitat loss due to low benthic light, and being able to identify the causes of loss if it occurs, are important prerequisites for adaptive management.

The effect of nutrient concentrations on seagrasses depends on the duration and concentration of exposure, as well as other local habitat conditions (Cabaço et al. 2013). When seagrasses are nutrient-limited, a pulse of nutrients can enhance seagrass growth (Udy et al. 1999b). If the pulse of nutrients isn’t used for growth, which could occur if they are limited by other resources including light, then there is luxury consumption and the nutrient content of their tissues increases (Collier et al 2014; Romero et al 2007). Luxury uptake can be a means of buffering against variations in nutrient availability, particularly from variable water column nutrient concentrations (Romero et al 2007). Elevated tissue nutrient concentrations within seagrass leaves can therefore indicate the long-term relative availability of nutrient species (Fourqurean et al 1997, Udy et al 1999), and tissue nutrient content can be used as an indicator of nutrient enrichment for a particular nutrient species, such as nitrogen or phosphorus.

Excessive luxury consumption occurs when nutrient availability is particularly high and can be detrimental to seagrasses above species-specific thresholds (Cabaço et al. 2013), but damaging luxury uptake has not been observed in the Reef. Elevated nutrients can also trigger ecological effects that are harmful to seagrasses such as blooms of epiphytic algae and macroalgae, which reduce gas diffusion and the acquisition of light and carbon by seagrass leaves (Cambridge et al. 1986; Frankovich and Fourqurean 1998; van Katwijk et al. 2010b). Elevated nutrients can also contribute to eutrophication, which has many detrimental outcomes for seagrasses, including sediment anoxia, shifts in primary producer communities, and alterations to biogeochemical cycles (Burkholder et al. 2007). Ecological shifts within seagrass communities due to nutrient loading have not been reported in seagrass meadows of the Reef. However, nutrients play an important role in producing SPM (Bainbridge et al 2012), thereby affecting benthic community composition through complex pathways including light limitation (Bainbridge et al 2018).

There are a number of seagrass plant-scale, or meadow-scale indicators that are sensitive to changing water quality including both light and nitrogen loads, including variables such as shoot density, shoot length, and lower depth limit. Early-warning physiological indicators that can forewarn of imminent risk to seagrass meadow condition are also useful (Schliep et al. 2015) but these tend to be less consistent in their responses (McMahon et al. 2013). There are a few notable exceptions, including plant nutrient content as ratios, and the isotopic signature of nutrients (Fourqurean et al. 1997; McMahon et al. 2013). They are pressure-specific indicators (McMahon et al. 2013; Roca et al. 2016), responding to nutrient availability and to light levels prior to morphological responses such as shoot loss and mortality (Grice et al. 1996a, McMahon et al 2013). As such, they have been used to monitor and forecast seagrass responses to changing water quality in some regions of the world (Fourqurean et al. 2003; Marbà et al. 2013; Roca et al 2013).

There has been no ongoing measure of water or sediment quality at the Marine Monitoring Program (MMP) seagrass monitoring sites, though there have been brief periods (e.g. a few years) in which both were measured at some sites. These periods highlighted that measuring sediment and water column nutrient concentrations is time consuming and expensive. Furthermore, estimating rates of nutrient uptake and assimilation from concentration data is difficult due to the complexity of nutrient pools and processes (e.g. McKenzie et al 2008, McKenzie et al 2009). Instead, seagrass tissue nutrient content (carbon, nitrogen and phosphorus) has been measured at Reef inshore seagrass monitoring sites each year during the growing season since the inshore seagrass MMP inception in 2005. These have been used as a proxy of changing water quality, and are also interpreted as an early-warning of changes to seagrass condition (as outlined in McMahon et al 2013, Roca et al 2016).

The ratio of carbon to nitrogen content (C:N) in seagrass tissue is a measure that integrates concentrations of nitrogen in the water column and sediment, as well as photosynthetic carbon uptake. It is therefore an indirect indicator of benthic light levels. C:N has been scored as one of three metrics in the MMP, weighted equally to seagrass abundance as per cent cover, and seagrass reproductive effort (see section 2.5.4 in main report). After 15 years of measurement, we are able to examine tissue C:N ratios in relation to water quality. This has been undertaken as a collaboration with the MMP water quality sub-program and as a result of emerging tools including water quality classification based on colour in remotely-sensed imagery and *in situ* measured water quality at some reef sites near to seagrass monitoring sites.

The aims of this analysis were to test:

- whether the leaf C:N baseline values vary between Reef seagrass species and habitats and how they compare to the global guideline
- whether leaf tissue nutrients, specifically C:N, vary in a consistent and predictable way to reconstruct past water quality in the Reef, and are therefore a time-integrated indicator of water quality
- whether C:N ratios vary in a consistent and predictable way to future seagrass abundance and are therefore an early-warning indicator of changes in seagrass condition
- the effect of including C:N ratios on the seagrass report card score.

Methods

Data variables and availability

Seagrass tissue nutrients

Seagrass tissue nutrient data as C:N ratios were compiled for all sites and all years of the inshore seagrass MMP (2005–2019) where available (Tables 1 and 2). Data were compiled for each species separately, but only for species that are considered foundational at the site. Data gaps occur during times when seagrass was absent, or when there was insufficient seagrass for collection, and depending on site commencement (Table 1). Tissue nutrients were measured only in foundation species (i.e. not the non-leaf replacing colonising *Halophila* spp.). Phosphorus (P) concentration, and the ratio of tissue C:P and N:P is also measured and reported in the MMP, but analysis of those data is not included here because they are not indicators in the seagrass health metric. All data are presented in this report, but statistical analysis was carried out on a sub-set of data as described in the statistics section below.

Benthic light

Benthic light data were obtained from *in situ* PAR logger data at the seagrass monitoring sites (see methods of this report) and averaged for each year leading up to the dry season sampling (i.e. November to October of the following year) (Table 1).

Exposure to river loads

Single yearly values of exposure to sediments and nitrogen for each site and monitoring year were extracted from a 2-dimensional loading model (Waterhouse et al. 2018) consisting of dissolved inorganic nitrogen (DIN) (Table 1). The loading model is calculated as an annual load dispersed during the wet season (December to April), when the majority of the rain falls throughout the Reef catchment area. These DIN values are not actual concentrations but provide an indication of the influence of the rivers on the inshore marine ecosystem and therefore reflect the relative magnitude of the loads prior to measurement of seagrass tissue nutrients. Exposure was calculated for each site from the model, except for sites that were not within the model boundary (e.g. too close to land), in which case exposure from the nearest adjacent pixel was taken.

In situ water quality

In situ water quality data were compiled for a small selection of locations where measurements are taken nearby to the seagrass site (Table 2). All of these locations with available *in situ* water quality data are classified as seagrass reef habitat. The water quality stations ranged in distance from ~2 km (e.g. Green Island) to ~8 km (e.g. Low Isles and Hamilton Island) away from the seagrass site. *In situ* water quality data were averaged for the previous wet season (December to April) to match the time-frames of the C:N data (i.e. annual).

Table 1. *Variables used for statistical modelling of C:N ratio in relation to water quality*

Dataset	Variable Name	Description
MMP inshore seagrass monitoring data (this report)	C:N No units	Leaf tissue nutrient C:N ratio averaged for each seagrass monitoring site (collected and analysed once a year during the Late Dry sampling, usually September or October). Measured as %C and %N, then ratio is calculated as the atomic ratio (see QA/QC document for detailed methods). Only data for foundational species have been included in this analysis (see section 2.2.1 McKenzie et al 2020 for definition)
	Light (mol m^{-2} d^{-1})	Mean yearly daily light from Odyssey Photosynthetically Active Radiation (PAR) loggers measuring PAR every 15 minutes, summed to total daily light, and an average daily light for the previous year leading up to the tissue nutrient sample (November Y-1 to October Y) taken at each sampling location (mol m^{-2} d^{-1}). For visual presentation of the model, the minimum, mean and maximum were selected for plotting (Table 5)
	f_DL	Mean daily light split into 2 categories, f_DL1 is assigned to data points with Light < median Light for the Habitat and region, f_DL2 is assigned to data point with Light > median Light (see Tables 6 and 7)
	NRM_Sub-region	Natural Resource Management (NRM) boundaries as defined by the NRM2014 Land shapefile © State of Queensland Department of Natural Resources, Mines and Energy. Northern and southern tropics are further distinguished for this report as north and south of Russell-Mulgrave River mouth
	Habitat	Habitat type as estuarine, coastal, reef, and subtidal if it never exposes (see this report, and Carruthers et al. 2002)
MMP water quality data from the loading model (Gruber et al 2020)	DIN exposure (relative units)	Modelled mean dissolved inorganic nitrogen (DIN) load for the wet season of the previous MMP sampling year for each seagrass monitoring sites (Waterhouse *et al.* 2018)
MMP water quality data from in situ monitoring stations (Gruber et al 2020)	Secchi (m)	Mean Secchi depth measured during the wet season of the previous MMP sampling year for selection of seagrass monitoring locations
	TDN (µmol L^{-1})	Mean total dissolved nitrogen measured during the wet season of the previous MMP sampling year for selection of seagrass monitoring locations
	TDP (µmol L^{-1})	Mean total dissolved phosphorus measured during the wet season of the previous MMP sampling year for selection of seagrass monitoring locations
	DIN (µmol L^{-1})	Mean dissolved inorganic nitrogen measured during the wet season of the previous MMP sampling year for selection of seagrass monitoring locations
	DIP (µmol L^{-1})	Mean dissolved inorganic phosphorus measured during the wet season of the previous MMP sampling year for selection of seagrass monitoring locations
	Chl (mg L^{-1})	Mean chlorophyll *a* measured during the wet season of the previous MMP sampling year for selection of seagrass monitoring locations
	TSS (mg L^{-1})	Mean total suspended solids measured during the wet season of the previous MMP sampling year for selection of seagrass monitoring locations

Table 2. *Seagrass monitoring locations, and number of data points at each location where a nearby (up* to ~8 *km)* in situ *water quality monitoring station was available for statistical modelling*

Seagrass Location (sites)	Water quality station	Years available	Number of data points
Dunk Island (DI1/DI2/DI3)	Dunk Island South East (TUL5)	2015-2018	35
Green Island (GI1/GI2/GI3)	Green Island (C11)	2006-2018	87
Great Keppel Island (GK1/GK2)	Humpy Island	2007-2010, 2014	21
Hamilton Island (HM1/HM2)	Pine Island (WH14)	2007-2018	31
Low Isles (LI1/LI2)	Port Douglas (C4)	2009-2013, 2015-2017	44
Magnetic Island (MI1/MI2/MI3)	Geoffrey Bay	2006-2017	102

Table 3. *Seagrass leaf tissue nutrient data count of C:N, DIN loads, daily light, region and habitat for all species combined, and when analysing species H. uninervis or Z. muelleri in isolation. Black font indicates data that was included in statistical analysis, and grey font was not further analysed due to insufficient data and/or limited range in relative DIN exposure.*

	coastal	estuarine	reef intertidal	reef subtidal
All foundational species				
Burdekin	74	0	92	51
Burnett–Mary	0	82	0	0
Cape York	151	0	123	0
Fitzroy	41	48	48	0
Mackay–Whitsunday	85	60	52	0
Northern Wet Tropics	58	0	148	64
Southern Wet Tropics	0	0	92	38
Halodule uninervis				
Burdekin	50	0	60	29
Burnett–Mary	0	0	0	0
Cape York	70	0	43	0
Fitzroy	5	0	23	0
Mackay–Whitsunday	28	7	29	0
Northern Wet Tropics	58	0	69	32
Southern Wet Tropics	0	0	63	28
Zostera muelleri				
Burdekin	24	0	0	0
Burnett–Mary	0	82	0	0
Cape York	17	0	0	0
Fitzroy	36	48	25	0
Mackay–Whitsunday	57	53	23	0
Northern Wet Tropics	0	0	0	0
Southern Wet Tropics	0	0	0	0

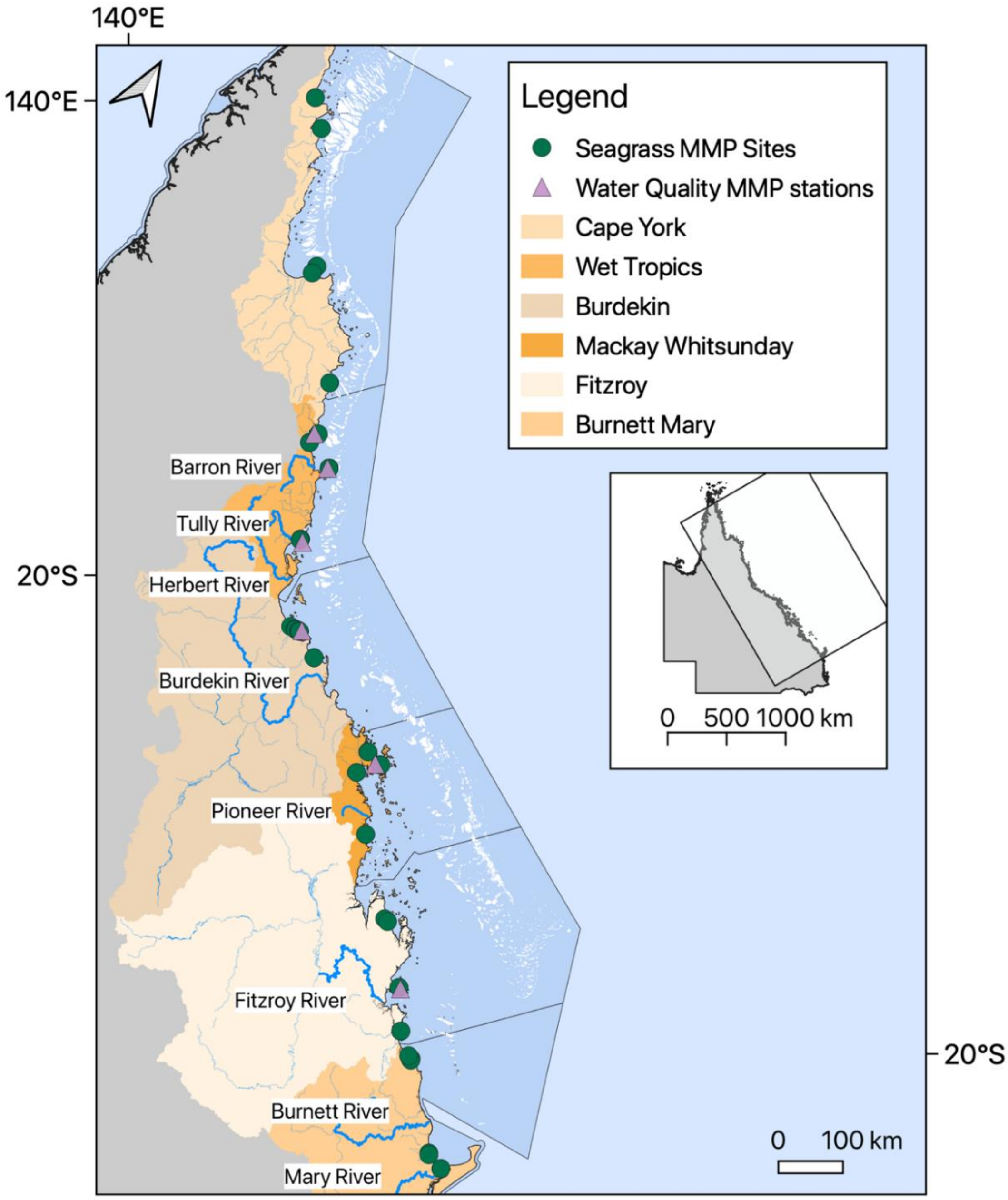

Figure 1. *Maps showing location of the MMP seagrass sites (circles) and water quality stations (purple triangles).*

Statistical analysis

Three modelling frameworks were explored to assess the influence of water quality on leaf C:N ratio and a fourth analysis testing the effectiveness of C:N as an early-warning indicator. These were:

(i) a parametric Generalized Linear Model (GLM) on a Gamma family with link log

(ii) non-parametric statistical models (Tree analysis)

(iii) explorative correlations of *in situ* water quality and C:N at reef sites

(iv) exploratory analysis of C:N in relation to changes in seagrass per cent cover.

All analysis was conducted in R Studio version 1.1.463.

1. Generalized linear models

A GLM model on the C:N was run with data for all species pooled. The addition of Location, Sites and Species as random effects were investigated but had to be dropped because of too many convergence issues resulting in over-estimated confidence interval estimations, in particular for high DIN range. This is mostly due to the unbalanced nature of this dataset amongst the random effects variables. The fixed factor estimates were still very similar between the two models and the only potential incidence is a slight under-estimation of the prediction confidence intervals but this should not alter the main trends for our interpretation in the context of this case study.

The models tested included the predictor variables (all continuous variables showed low levels of correlation):

- Habitat (as factor)
- NRM_sub-region (as factor)
- DIN (as a linear variable)
- Light (as a linear variable)

The interactive effects of all predictor variables were tested up to the maximum of 4-way interactions. The dredge function (package MuMIn) was used to determine the best model to describe C:N i.e. the function runs all combination of interactions and provides the optimum model. The optimum model was based on the Akaike Information Criterion (AIC), which describes goodness of fit for the model and also includes a penalty for an increasing number of predictor variables and interactions. The optimum model for describing C:N for all species pooled based on the outcome of this criteria is model 1 (M1):

CN = DIN + Habitat + Light + NRM_subregion + DIN:Habitat + DIN:Light + DIN:NRM_subregion + Habitat:Light + Habitat:NRM_subregion + DIN:Habitat:Light

“:” defines interaction term

Burnett-Mary, Cape York, Fitzroy and reef subtidal sites were removed from the analyses due to insufficient data and/or limited range in DIN; however, reef subtidal habitats were retained when investigating responses of *H. uninervis* only, as it is the most common species in reef subtidal habitats. The final model included the regions: Burdekin, Mackay-Whitsundays, Northern Wet Tropics and Southern Wet Tropics. The Burdekin region and coastal habitat were set as the references for all of the models.

To simplify and avoid 3-dimensional graphical representation of M1, the mean daily light variable was categorised into three groups (minimum, mean, maximum) calculated for each Region and Habitat type combination. The three scenario predictions of M1 were then calculated and plotted against the individual data points for the 8 Region and Habitat combinations.

GLM models were also run for individual species only (*Halodule uninervis, Cymodocea serrulata, Thalassia hemprichii* and *Zostera muelleri*). There was insufficient data for *C. serrulata* and *T. hemprichii* to interpret model outputs so they are not shown here. Due to the low significance level of the Light variable we tried categorical transformations based on either two or three categories. The transformation leading to the lowest AIC was a two category variable named f_DL for factor of daily light. Category 1 and 2 were defined as light levels that were below the median (f_DL1) and above the median (f_DL2) i.e. low and high light of each region and habitat type. For *Z. muelleri*, the regions Fitzroy and Burnett-Mary were included because that species is the most common in those regions. Model selection for the two species followed the criteria above for M1.

The model for *H. uninervis* (M_Hu) was:

CN = DIN + f_DL + Habitat + NRM_subregion + DIN:f_DL + DIN:Habitat + DIN:NRM_subregion + Habitat:f_DL + Habitat:NRM_subregion + DIN:f_DL:Habitat + DIN:Habitat:NRM_subregion

The model for *Z. muelleri* was (M_Zm) was:

C:N = DIN + Habitat + f_DL + NRM_subregion + DIN:f_DL + DIN:Habitat + DIN:NRM_subregion + Habitat:f_DL + f_DL:NRM_subregion + Habitat:NRM_subregion + DIN:f_DL:NRM_subregion + DIN:Habitat:NRM_subregion + f_DL:Habitat:NRM_subregion + DIN:f_DL:Habitat:NRM_subregion

2. Non-parameteric models (Tree analysis)

Tree structured regression and classification models (Trees, using "partykit" package in R) were used to explore specific DIN and Light values that were associated with significant splits of the C:N values. If identifiable, these splits could be used to assess report card grading levels, and aid in the interpretation of inter-annual trends; if DIN or Light exceeds a particular threshold, then C:N is expected to change and vice versa. The model M_HuTree was:

C:N = DIN + Light

As this is a non-parametric test there is no need for data distribution specification or model selection (i.e. no interaction terms). This was carried out for *H. uninervis* only as it has the most data available due to its very wide geographical spread. We tried incorporation of additional covariates to the tree model such as Species, Habitat and Region. The outputs did not show clear and consistent trends when comparing amongst models. Furthermore, the large number of nodes made it very difficult to interpret and therefore results are not shown here.

3. Correlation analysis of in situ water quality data

Exploration of *in situ* water quality sampling was undertaken by plotting various water quality variables (all data points at the nearby location in the year prior to C:N measurement being taken) against seagrass C:N ratio in the late dry season. A simple linear regression (gaussian distribution) was used to check for significant trend. No further analyses were conducted at this stage, because no apparent correlations were observed.

4. Exploratory analysis of C:N and future seagrass cover

Finally, an exploratory analysis of C:N in relation to future change in seagrass per cent cover (all species pooled) was explored. This was to test whether C:N is an early-warning indicator of seagrass condition. To do this, the C:N in year x, was plotted against the change in seagrass %cover calculated as:

$$\text{Change in total cover} = \%C_{(x+1)} - \%C_{(x)} \qquad \text{Equation 1}$$

Where %C is the average per cent cover of all seagrass species at the site.

A GLM model was used to investigate the influence of the lagged C:N ratio on the change in seagrass cover by Habitat and NRM Region. Model selection using the dredge function was applied and the optimal model (gaussian distribution) was as follows:

Change in cover = CN_lag + Habitat + Region + CN_lag:Habitat + CN_lag:Region + Habitat:Region + CN_lag:Habitat:Region

Where CN_lag is the CN of year x and other factors are described in Table 1.

Results

Long-term baseline summary of leaf C:N in seagrasses of the Reef

The median C:N of all species pooled in the Reef was 17.3 (Table 4). However, C:N varies among species, with *T. hemprichii* and *H. uninervis* having the lowest median C:N (16.2 and 16.3), and *C. serrulata* the highest (21.9, Table 4). Median C:N exceeds the global guideline value of 20 in *C. serrulata*, and *S. isoetifolium* but not in the remaining species (Table 4, Figure 2) highlighting that species composition will affect scores based on the guideline of 20. C:N of species also varies among habitats, particularly for two of the species most broadly distributed throughout the Reef: *Z. muelleri* and *H. uninervis* (Figure 3). C:N is higher in *H. uninervis* from reef habitats (both intertidal and subtidal), while for *Z. muelleri* C:N is lowest in reef habitats (intertidal only as it is not found in subtidal reef habitats) (Figure 4).

Table 4. *The mean, median, 25th and 75th percentiles (PCTL) of C:N ratios for all species pooled in the Reef, and for the six species assessed in the MMP. Samples were collected from 2005 to 2018.*

Species	Mean	Median	25th PCTL	75th PCTL
Reef pooled	17.7	17.3	15.1	20.0
C. rotundata	18.6	18.0	16.8	19.7
C. serrulata	22.3	21.9	20.0	23.9
H. uninervis	16.4	16.3	13.4	18.7
S. isoetifolium	21.3	21.4	19.9	22.9
T. hemprichii	16.6	16.2	15.4	17.4
Z. muelleri	18.7	18.2	15.7	21.1

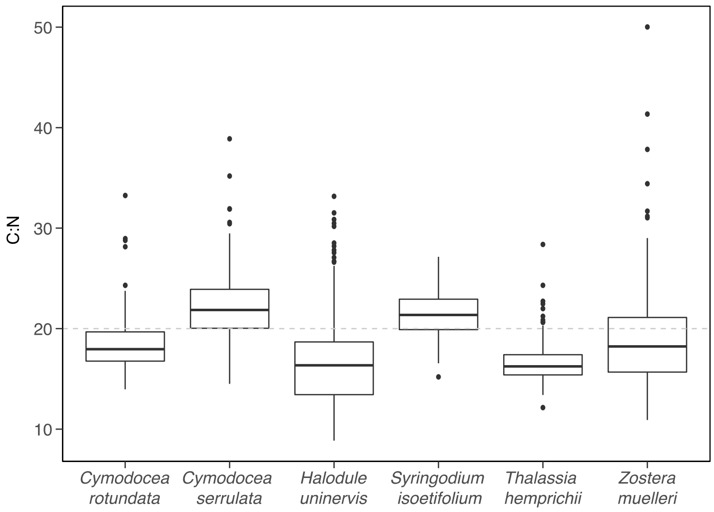

Figure 2. *Boxplots showing distribution of C:N data for each species for samples collected from 2005 to 2018. Dashed line indicates the accepted GBR seagrass guideline value based on the global median value (Atkinson and Smith 1983; Fourqurean et al 1992). The box represents the interquartile range of values, where the boundary of the box closest to zero indicates the 25th percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75th percentile. Whiskers (error bars) above and below the box indicate the 90th and 10th percentiles, and the dots represent outlying points.*

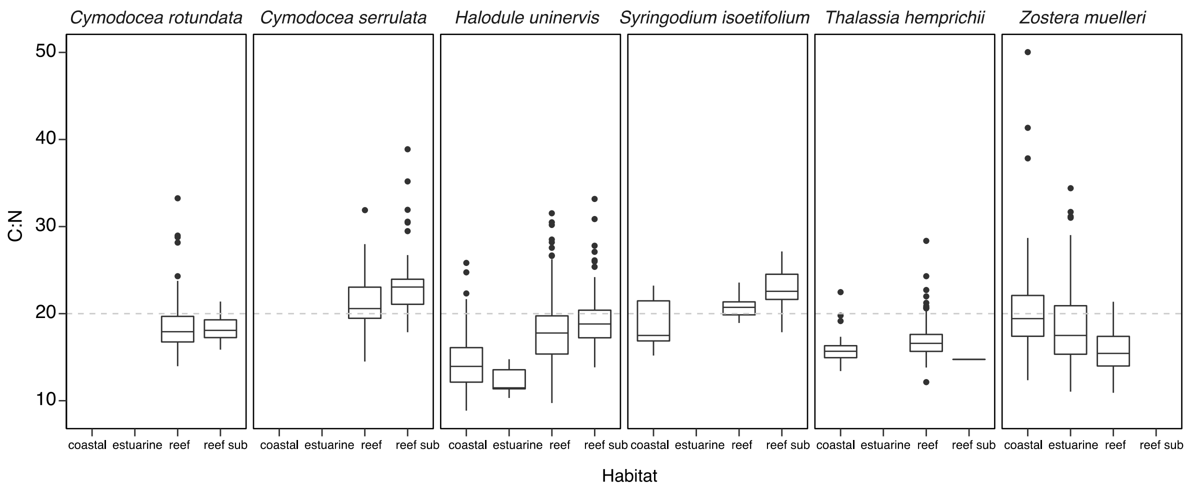

Figure 3. *Boxplot showing distribution of C:N data for each Reef seagrass species based on each Reef habitat type. Dashed line indicates the accepted seagrass guideline value based on global median value (Atkinson and Smith 1983; Fourqurean et al 1992)*

1. Generalised linear models of C:N

Changes in C:N (species pooled) in relation to DIN loads and daily light

There was a weak non-significant inverse relationship between relative DIN exposure and C:N for all species pooled, whereby C:N ratios decreased as DIN exposure increased. There was also a relationship between light and C:N ratios, where greater light availability correlated with higher C:N ratios, although this trend varied by habitat and sub-region (Figure 4, see Supplementary Materials for output summary details). Relationships between light and C:N were especially clear in coastal intertidal habitats where wide ranges of both relative DIN exposure and light levels occur. In estuarine and reef intertidal habitats, light availability affected the response of C:N to relative DIN loads in more complex ways as described in greater detail below.

Relationships between C:N, light, and relative DIN exposure varied with habitat type. Ratios of C:N increased with light availability in coastal intertidal habitats (Figure 4; Table A1) in the three regions included in the analysis. This finding is compatible with the conceptual understanding of how light affects C:N based on previous experimental and field observations; as light and photosynthesis increase, relatively more carbon is incorporated into seagrass tissue. There is high inter-annual variability in light in coastal habitats because it can be influenced by riverine discharge, wind-driven resuspension and clouds.

There was a negative relationship between DIN exposure and C:N in coastal habitats of the Burdekin and Mackay-Whitsunday regions. Again, this is compatible with previous experimental and field observations; as nitrogen availability increases, then there is relatively more uptake of N, which lowers the C:N ratio. In general, there is high inter-annual variability in predicted relative DIN loads in coastal habitats. However, there was no detectable effect of relative DIN load on C:N in the northern Wet Tropics for coastal habitat due to the low variability in relative DIN loads predicted for that region from the loading model compared to the two other regions. Overall, these results are consistent with what we would expect in habitats that are subjected to a variable light and nutrient regime. However, there was no interaction between light and relative DIN load, represented by the fact that the slope of change of the C:N with DIN does not significantly change with the light (i.e. the prediction lines for the 3 scenarios are parallel); which is an unexpected finding.

The analysis of estuarine intertidal habitat was limited to the Mackay-Whitsunday region when all species were pooled, but it is dominated by *Z. muelleri*. Ratios of C:N showed contrasting relationships with DIN loads, which depended on light availability (Table A1; Figure 4; Table 5). When subject to low light, the C:N significantly decreased with increasing relative DIN load, which indicated luxury nitrogen uptake. Daily light alone does not seem to have any significant effect on C:N, which may be due to a relatively more limited light range compared to coastal intertidal sites in the same region (Table 5)

In reef intertidal habitats there was a negative effect of relative DIN on C:N when light was high (an interactive effect): C:N was at its highest under high light but low relative DIN loads, but as the DIN loads increased C:N declined particularly in the Burdekin region. This was not observed when light was low, which is the opposite of what occurred in the estuarine habitat. The reef intertidal sites are further seaward and get consistently more light (16 mol m^{-2} d^{-1}) than the coastal and estuarine sites (11.8 mol m^{-2} d^{-1} and 11.3 mol m^{-2} d^{-1}).

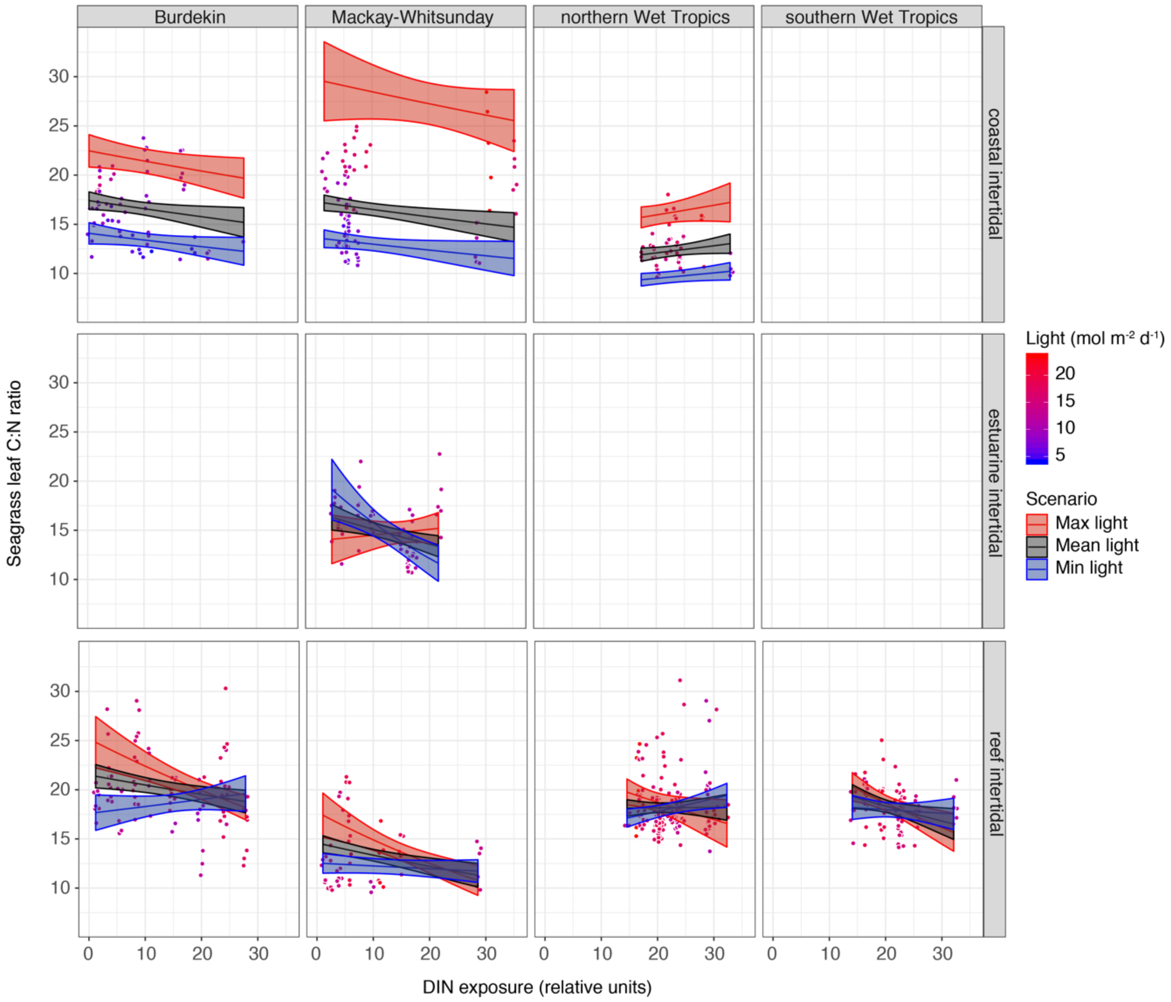

Figure 4. Graphical representation of the GLM results for C:N in relation to relative DIN exposure, mean light, region and habitat for all species pooled (model M1). To simplify the representation, daily light is plotted in 3 categories (minimum, mean, maximum for each habitat and region), but it was used as a linear data set for the modelling. The gradient scale of Light refers to the light levels of the point data, while the Scenarios refer to the 3 daily light categories (Table 5). Shaded areas indicate the 95% prediction confidence intervals from the model for each of the scenarios.

Table 5. *The light levels corresponding to the minimum, mean, and maximum for each sub-region and habitat used for graphical representation (Figure 4) of the continuous variable of light (mol m^{-2} d^{-1}) in M1 of C:N for all species.*

NRM_subregion	Habitat	Light (mol m^{-2} d^{-1})		
		Minimum	Mean	Maximum
Burdekin	coastal intertidal	4.0	8.9	14.8
Burdekin	reef intertidal	9.4	14.5	18.5
Mackay–Whitsunday	coastal intertidal	5.5	11.1	23.6
Mackay–Whitsunday	estuarine intertidal	9.2	11.3	13.2
Mackay–Whitsunday	reef intertidal	13.2	17.0	22.0
northern Wet Tropics	coastal intertidal	10.2	15.7	22.1
northern Wet Tropics	reef intertidal	13.6	17.7	23.5
southern Wet Tropics	reef intertidal	13.2	17.0	20.3

In conclusion, habitat strongly affects the response of C:N (all species pooled) to light and DIN loads. In coastal habitats where there is high variability in light and relative DIN loads, the C:N ratio responds strongly to both. In estuary and reef habitats, C:N ratio responds to DIN when subject to the light environment they most frequently encounter i.e. low and high light respectively. However, the variation in response among habitats may also be affected by the species composition, therefore, we also tested the response of C:N in the most common species: *H. uninervis* and *Z. muelleri*.

Changes in C:N (H. uninervis) in relation to relative DIN exposure and light

For *H. uninervis*, the most widespread seagrass species across MMP monitoring sites on the Reef, C:N was affected by light and DIN exposure most clearly in the Burdekin region (Table A3, Table A4, Figure 5). There was a negative effect of DIN exposure on C:N particularly when f_DL was in the middle of the range for light levels including f_DL2 in the reef subtidal and f_DL1 and f_DL2 in the coastal habitats. The factor for light is based on being above or below the median for each habitat within region (Table 6).

In coastal habitats, ratios of C:N in *H. univervis* were greater under high light conditions compared to low light conditions. Burdekin and Mackay-Whitsunday coastal habitats showed a negative relationship between C:N and DIN loading, whereby C:N ratios decreased with increasing DIN loading. Similar patterns were observed for *H. univervis* in reef intertidal and subtidal habitats of the Burdekin and Mackay–Whitsunday regions. For reef habitats of the Wet Tropics region, the effect of light availability on C:N ratios was not significant or was driven by a limited range of data points. The high variation in C:N ratio for those instances are making it very difficult to detect clear patterns and would require additional data to be validated.

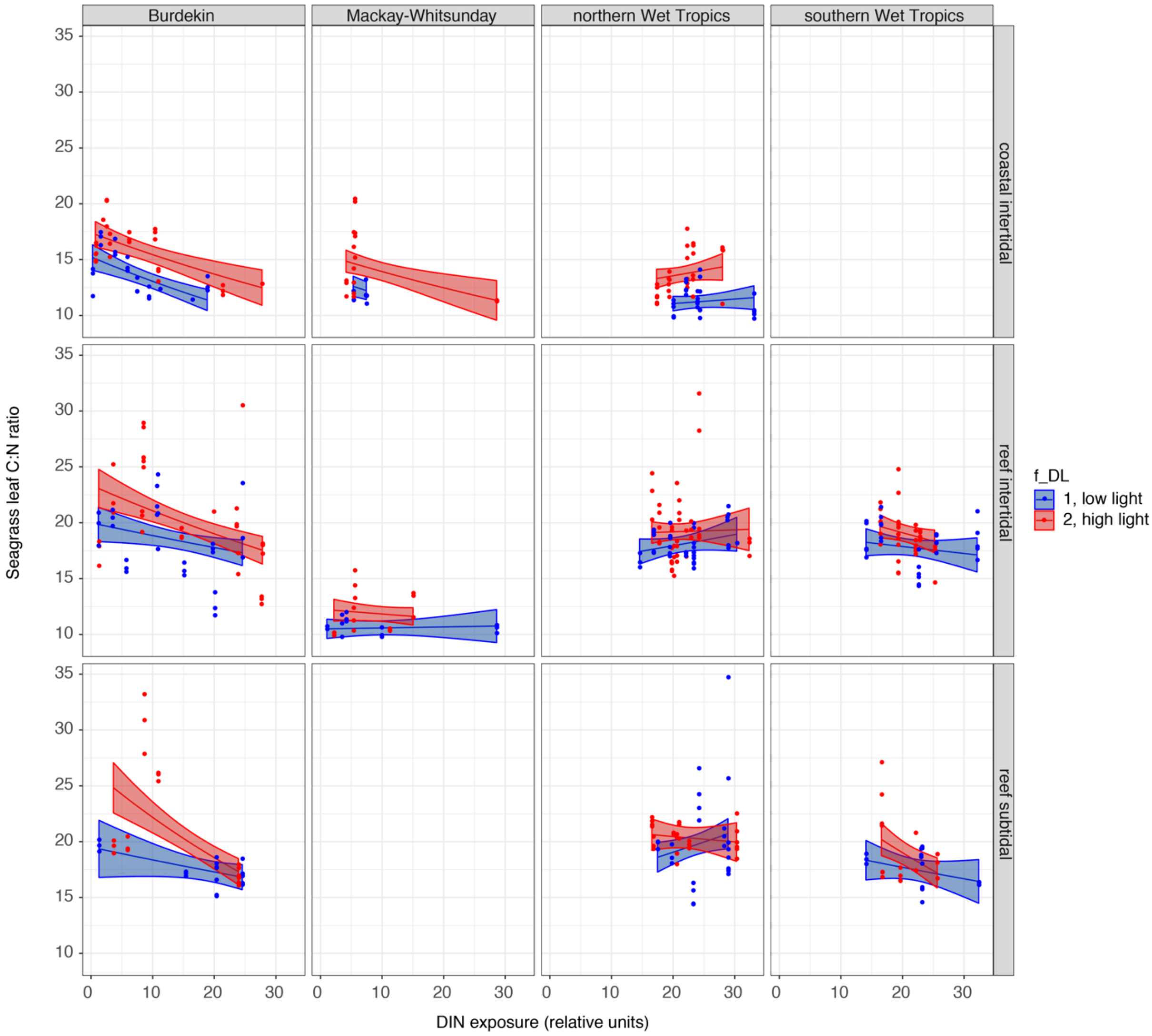

Figure 5. *Graphical representation of the GLM results for C:N of H. uninervis in relation to relative DIN loads, mean daily light, region and habitat (model M1). To simplify the representation, daily light was used in the model as a factor in two categories below (f_DL1, low light) and above the median (f_DL2, high light) for each habitat within region (Table 6). Shaded areas indicate the 95% prediction confidence intervals from the model for both light categories.*

Table 6. *The minimum, median and maximum daily light (mol m^{-2} d^{-1}) used to set the two factors for f_DL for each sub-region and habitat in M_Hu of C:N. F_DL1 is the mean annual daily light when below the median, but greater than the minimum, and f_DL2 includes mean daily light that is equal to or greater than the median.*

NRM subregion	Habitat	Light (mol m^{-2} d^{-1})		
		Min	**Median**	**Max**
Burdekin	coastal intertidal	4.0	7.6	14.8
Burdekin	reef intertidal	9.4	14.4	18.5
Burdekin	reef subtidal	4.5	5.2	7.0
Mackay–Whitsunday	coastal intertidal	5.5	7.8	11.4
Mackay–Whitsunday	reef intertidal	12.0	16.7	22.0
northern Wet Tropics	coastal intertidal	10.2	16.1	22.1
northern Wet Tropics	reef intertidal	11.6	16.6	23.5
northern Wet Tropics	reef subtidal	5.6	10.5	16.0
southern Wet Tropics	reef intertidal	13.2	17.4	20.3
southern Wet Tropics	reef subtidal	4.6	6.9	11.1

Changes in C:N (Z. muelleri) in relation to DIN loads and daily light

Z. muelleri is mostly found in the southern regions of the Reef and the effects on C:N were very complicated due to a large number of interactions, including 3-way interactions (Table A5, Figure 6). This makes it very difficult to interpret C:N using this species. Ratios of C:N were significantly higher in low light in estuarine habitats of the Burnett-Mary, and significantly higher in high light in coastal habitats of the Mackay-Whitsundays. Furthermore, DIN exposure had a significant positive effect in some cases (estuarine habitat, Burnett-Mary and coastal habitat Fitzroy), but it must be noted that there was very low range in DIN exposure in those habitats. These results go against the expected effect of light and excess nitrogen and highlight the complexity of interpreting C:N ratio across such a wide variety of species, habitats and locations.

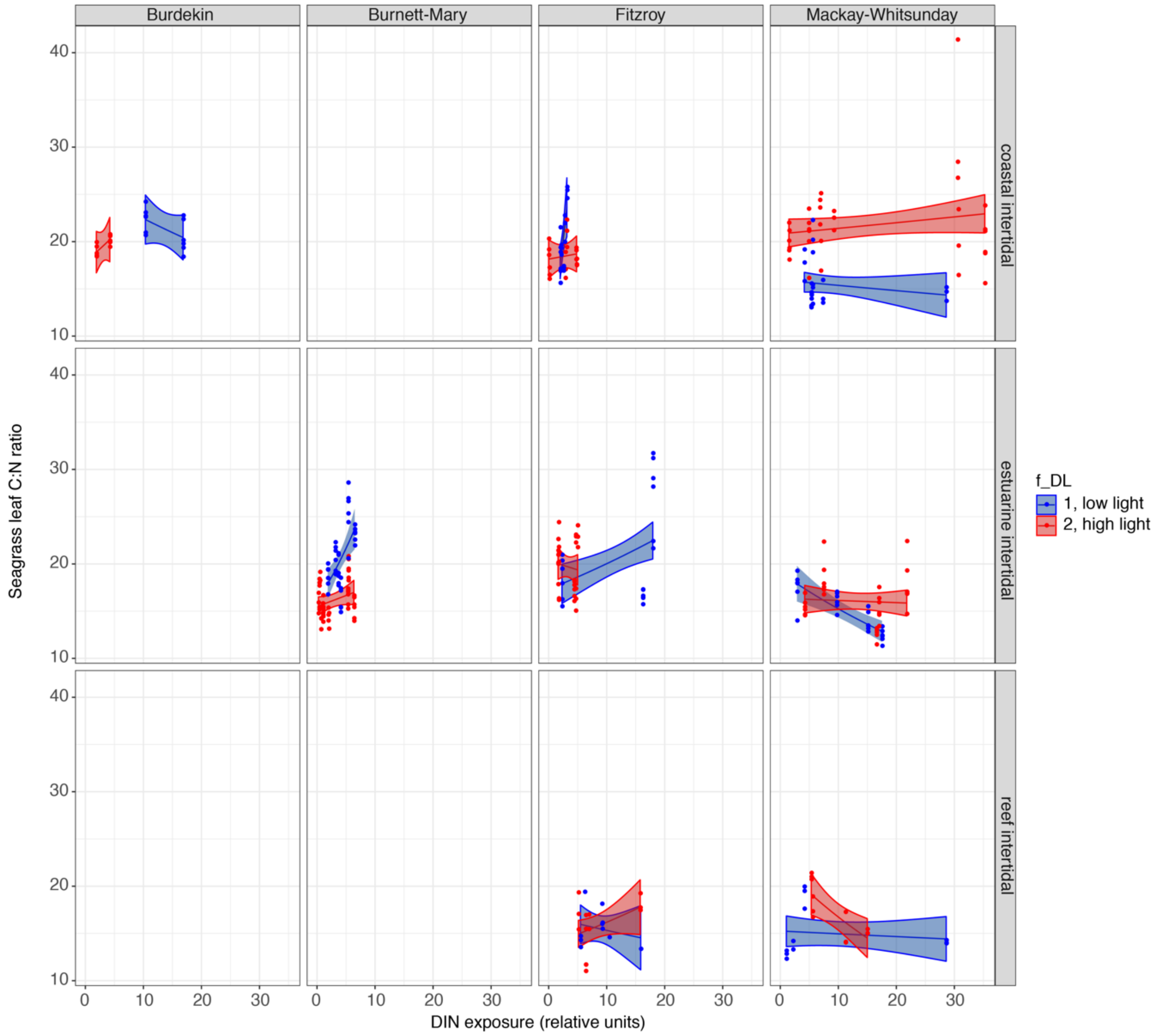

Figure 6. *Graphical representation of the GLM results for C:N of Z. muelleri in relation to relative DIN loads, mean daily light, region and habitat (model M1). To simplify the representation, daily light was used in the model as a factor in two categories below (f_DL1, low light) and above the median (f_DL2, high light) for each habitat within region (Table 7). Shaded areas indicate the 95% prediction confidence intervals from the model for both light categories.*

Table 7. *The light levels corresponding to f_DL for each sub-region and habitat for graphical representation of daily light (mol m^{-2} d^{-1}) variable when converted to a factor for analysis in M_Zm of C:N for the species* Z. muelleri*.*

NRM_subregion	Habitat	Light (mol m^{-2} d^{-1})		
		Min	Median	Max
Burdekin	coastal intertidal	8.0	12.3	14.8
Burnett-Mary	estuarine intertidal	8.0	11.2	27.6
Fitzroy	coastal intertidal	13.7	19.4	25.5
Fitzroy	estuarine intertidal	9.5	10.5	14.5
Fitzroy	reef intertidal	6.3	16.1	21.4
Mackay-Whitsunday	coastal intertidal	5.5	11.6	23.6
Mackay-Whitsunday	estuarine intertidal	9.2	10.7	13.2
Mackay-Whitsunday	reef intertidal	13.2	17.1	22.0

2. Trees analysis of seagrass leaf C:N, DIN loads and daily light

Trees analysis was used to explore whether specific thresholds for light and relative DIN exposure can be identified (Figure 7).

C:N of *H. uninervis* was affected by relative DIN exposure. Below relative DIN exposure of 7.8, C:N was 15.4 on average (node 2, Figure 7). Between relative DIN load of 7.8 and 17.1 C:N was 19.0 (node 9, Figure 7). When relative DIN loads were greater than 17.1, which was the majority of data points, C:N was further affected by Light (nodes 6, 7 ,8). Within this category, the response to Light was not clear. The lowest C:N occurred when Light was between 14.2 and 16.1 (node 6) and C:N increased when Light was either below or above this range (nodes 7 and 8). Furthermore, the range in C:N in this category overlaps with the C:N in the very low relative DIN loads category (node 2). This means it is not possible to use the DIN threshold or Light to predict whether changes in C:N are due to daily light or DIN with this simple model.

Many other Tree analyses were conducted, including for other species, combined species, and by focussing on certain habitats or regions but these results are not shown here. Unfortunately, the results were highly sensitive to the data selection for the model and thresholds associated with the splits varied widely. This is not surprising, given the variability in response to C:N based on species, habitat and region as described above from GLMs and other factors that are hard to capture and model. Therefore, the Trees analysis cannot provide thresholds that can be used for interpretation of future C:N monitoring.

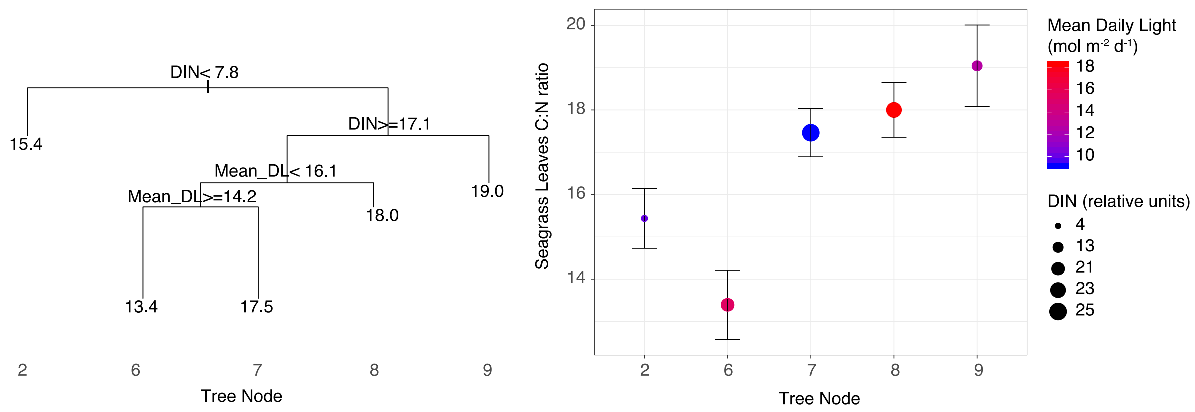

Figure 7. *Trees analysis of C:N in relation to daily light (Light) and relative DIN exposure for H. uninervis .*

3. Exploratory analysis of seagrass leaf C:N and in situ water quality

This analysis was undertaken in order to assess whether the *in situ* water quality data, which has a high degree of precision, could provide an alternative to modelled nutrient exposure particularly with respect to nitrogen. However, *in situ* water quality data are only available for reef sites and only represent 'point-in-time' measurements. The nutrient variables (TDN, TDP and DIP) did not correlate to C:N with R^2 values ranging from 0.022 to 0.11 (Figure 8). Furthermore due to the large amount of explained and heterogenous variance in C:N with the different covariate even the significant p values returned are probably unreliable. The strongest linear relationship (explaining only 12% of the variance, R^2=0.12) was a positive effect of Secchi depth on C:N. However, it is important to note that this is influenced by a few high Secchi depth values from the Green Island water quality monitoring sites in 2006, 2008, 2009 and 2018. Within the normal range of Secchi depths recorded (0-12m), there is no significant trend in C:N in relation to Secchi depth.

There are a number of considerations when looking at the *in situ* water quality data in relation to seagrass leaf C:N. Firstly, the water quality and seagrass sites are up to ~8km apart from each other and there is not a unique water quality sample for both intertidal and subtidal habitats. For parameters that affect benthic light levels (TSS, Chl and Secchi), these will have different effects on seagrass growing subtidally or intertidally. Secondly, most of the water quality sampling is conducted in the wet season (and this analysis was based only on wet season data). For both of these reasons, the water quality data can only be used as broadly representative of changing spatial-temporal patterns in water quality in relation to seagrass nutrient availability. In addition, *in situ* sampling is only conducted in proximity to seagrass 'reef' sites, and not the coastal or estuarine sites, which had some of the stronger patterns in the GLMs particularly at coastal sites. No further analysis has been conducted at this stage.

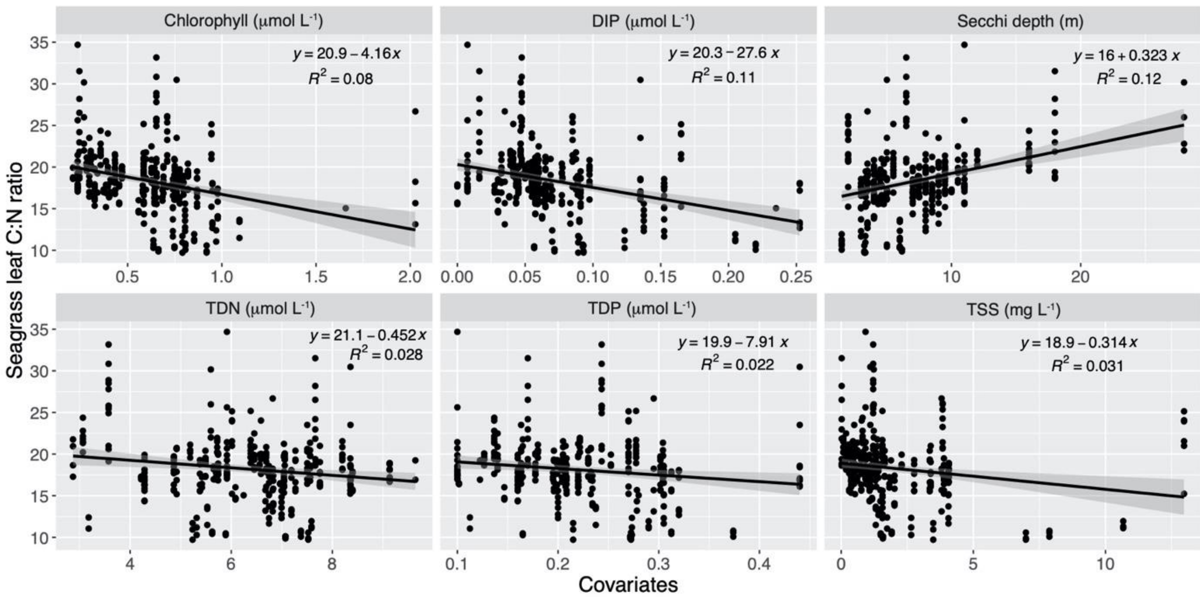

Figure 8. Exploratory analysis of seagrass leaf C:N in relation to *in situ* water quality data from Gruber *et al.* 2019a. The solid line represents the fitted linear regression with shaded areas indicating the 95% prediction confidence intervals (warning: most of these fitted linear models violate the homogeneity of variance assumption).

4. C:N as a predictor of changes in seagrass abundance

To test whether seagrass C:N can forewarn of changes in seagrass abundance, the C:N (all species, in year x) was plotted against the change in seagrass abundance for that site from year x to year x+1. There were no observable patterns in the combined seagrass data (Figure 9), but this was further interrogated at the region and habitat level. Again, there was no observable pattern with the exception of the coastal habitat in the Burdekin region, in which higher C:N was associated with loss in the following year (Figure 10). No further analysis has been undertaken at this stage.

This indicates that the assumption that C:N may be a suitable early-warning indicator of future changes in seagrass abundance is unsubstantiated, which is not surprising when considering the complexity of species and habitat types present in the Reef, the sampling resolution (3 samples per site and species per year), and the time-frames of measuring and reporting (annual).

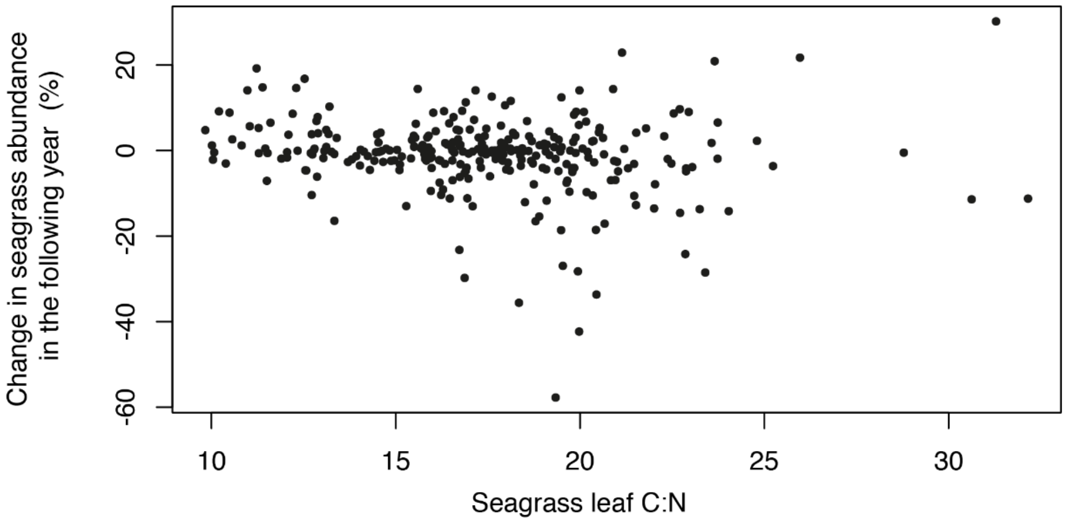

Figure 9. *Seagrass leaf C:N (year x) in relation to change in seagrass cover in the following year for all data combined.*

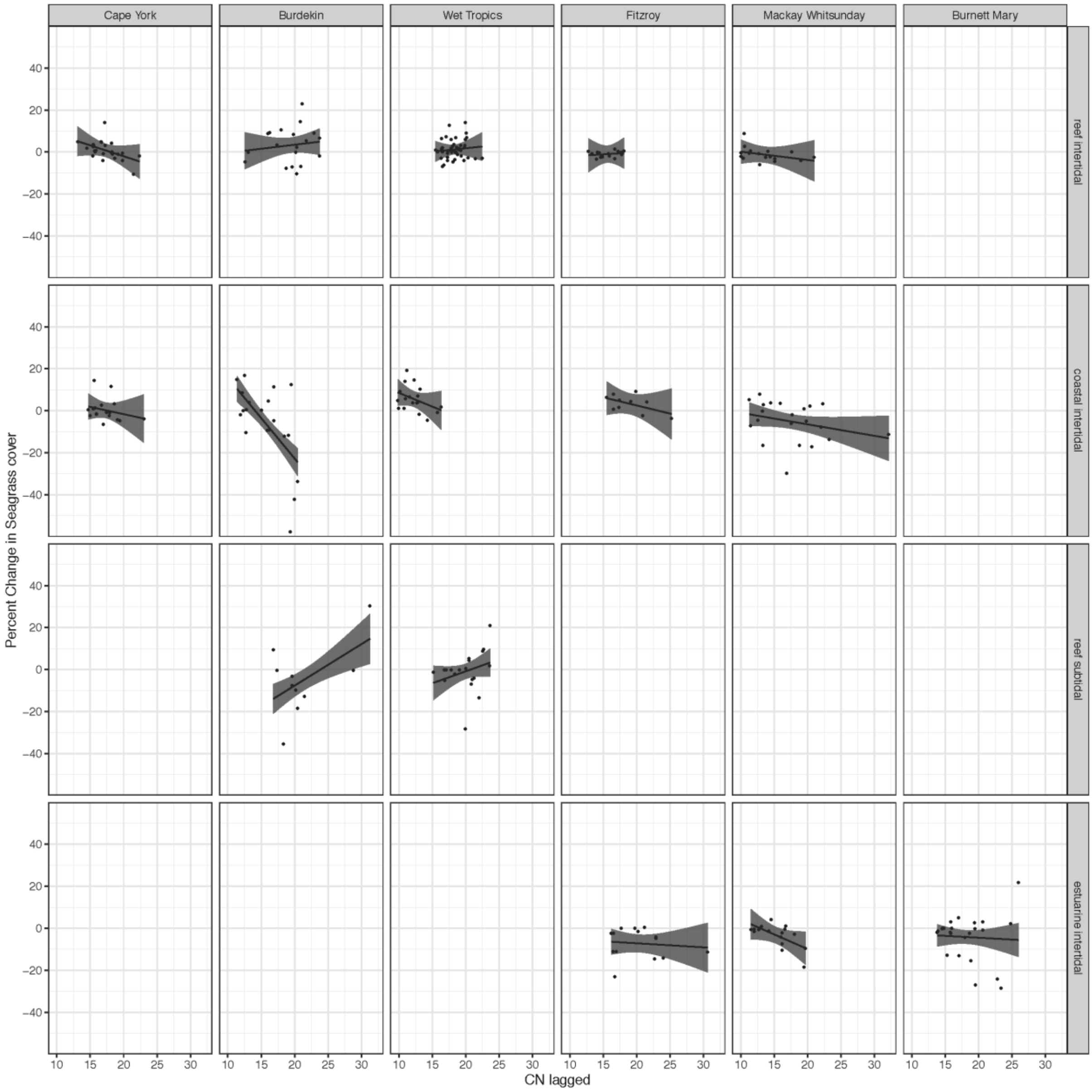

Figure 10. *Percent change in seagrass cover in relation to the C:N in the previous dry season for each region and habitat. Shaded areas indicate the 95% prediction confidence intervals from the model.*

5. C:N ratio and the Reef health index

Seagrass leaf C:N ratio is included in the Reef health index as one of 3 metrics, together with reproductive effort and seagrass abundance. This analysis calls into question the suitability of retaining such a strong weighting of C:N. We have therefore tested the effect of the C:N metric on the reef health Index by comparing the index as it is currently reported, to a 50% weighting of C:N towards the Index, and removing C:N from the health Index (Figure 11).

In general, down-weighting or removing C:N from the health index reduces the overall Index, and occasionally causes it to drop to a lower category. However, most of the time the re-calculated score remains within the confidence intervals for the score and is therefore not a ‘significant’ change (Figure 12). The exceptions are in the Fitzroy and Burnett-Mary regions, where the health Index is currently buoyed by a high C:N score due in part to the dominance of *Z. muelleri* in those regions, which has a higher C:N (and therefore receives a higher rating for that metric) than *H. uninervis* which dominates in other regions. It is important to note that the reproduction metric (reproductive effort) is also under consideration for review. Any potential changes to the calculation of the Index will consider changes to the other metrics at the same time.

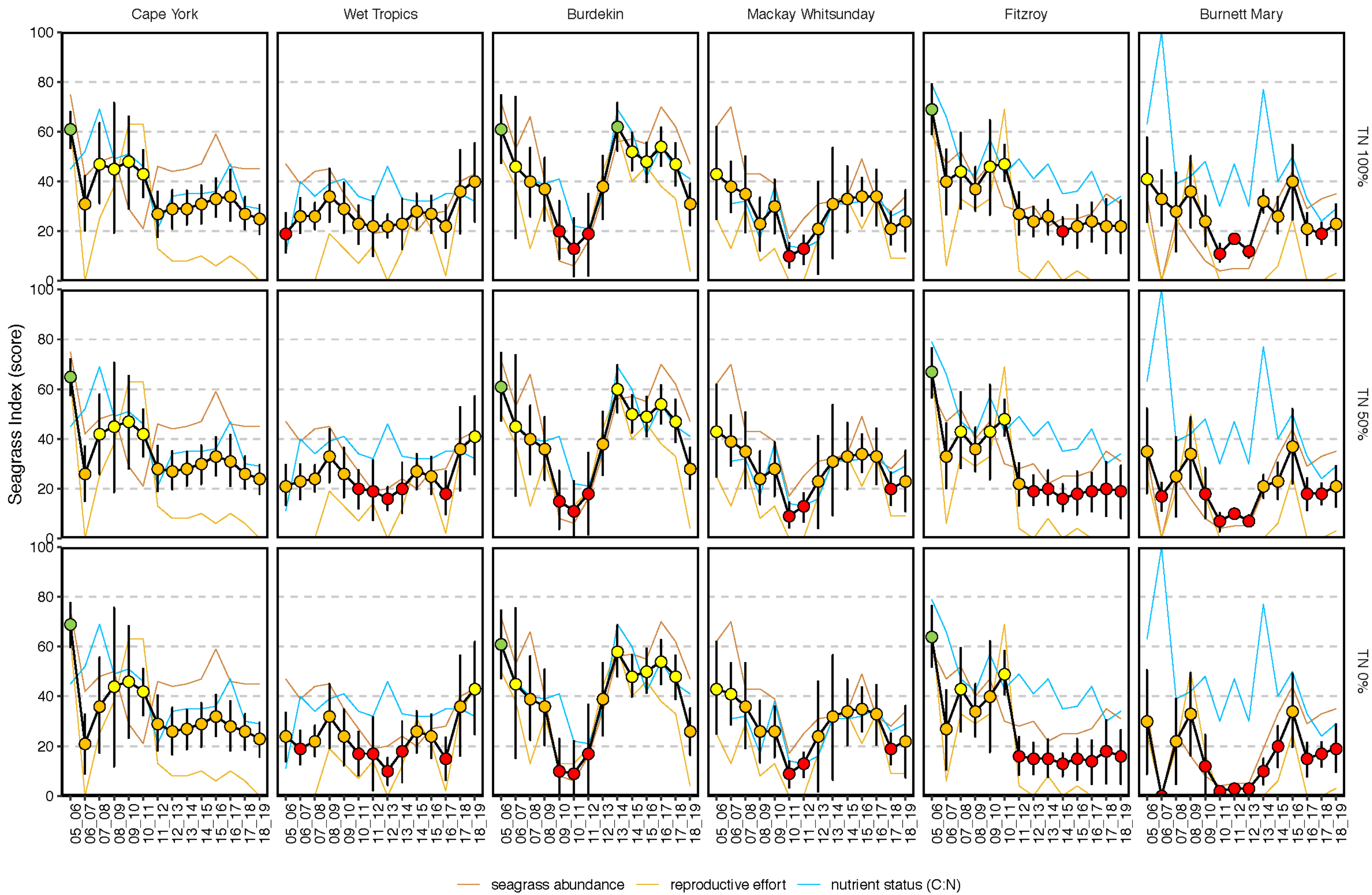

Figure 11. *The seagrass health index from the reporting period 2005-06 through to 2018-19 with C:N weighted equally (100%) to abundance and reproductive effort (top), and with a 50% (centre) and 0% weighting (bottom), in each NRM region.*

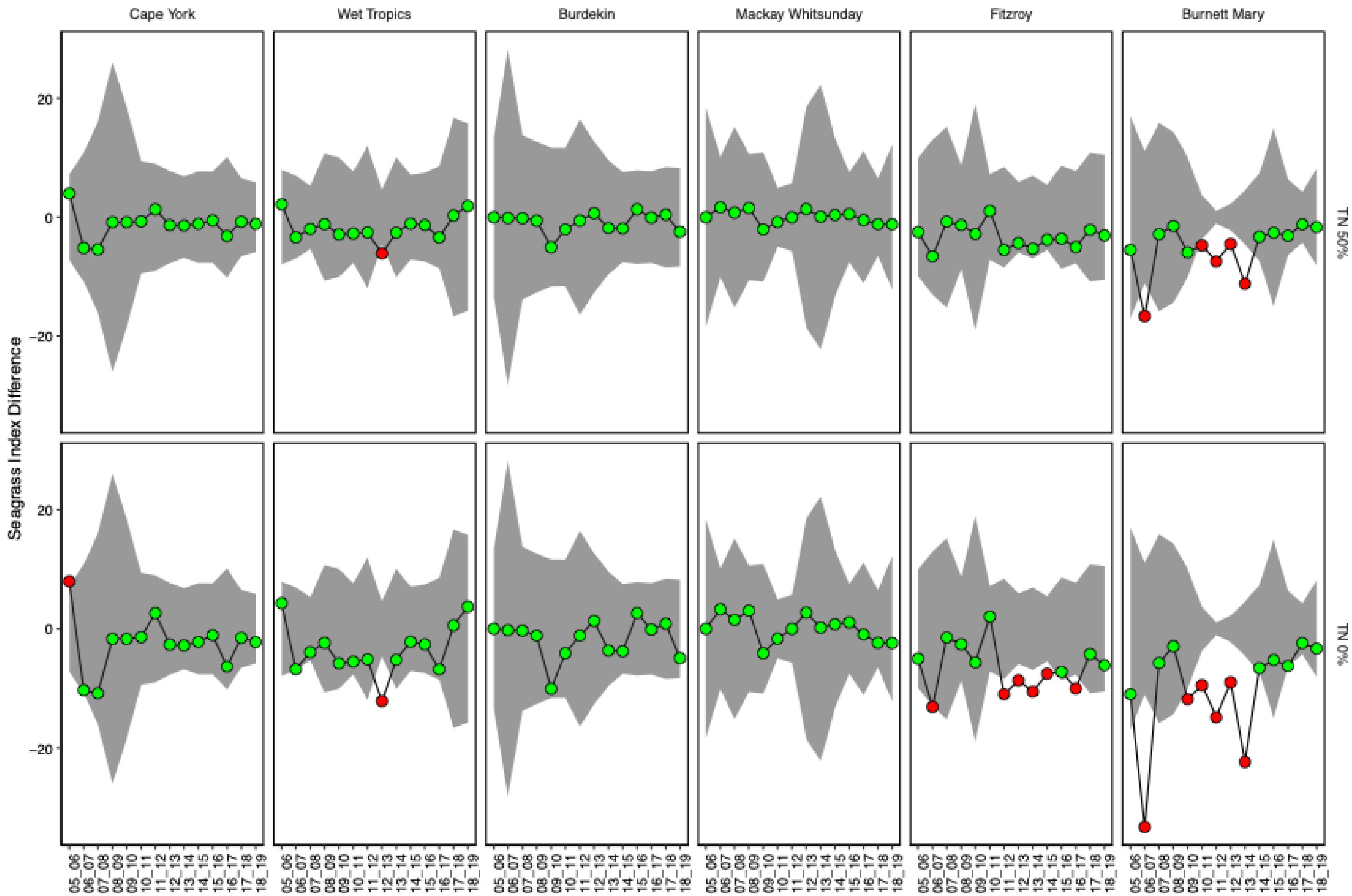

Figure 12. *Difference in the seagrass health index when C:N is down-weighted (50%), or removed (0%). Grey shaded areas are the confidence intervals (error bars) from the original index in Figure 11. Green is for when the index remains within the intervals and red for when the calculation falls outside of the intervals.*

Conclusions

Seagrass leaf C:N at inshore seagrass monitoring sites within the MMP is generally highly variable and influenced by daily light and predicted relative DIN loads in different ways depending on species, habitat and region. It is an expected finding that habitat and region influences how C:N responds to water quality. However, the high complexity of the interactions, and the counter-intuitive or non-significant relationship between C:N and DIN and light in some habitats and regions makes it difficult to confidently interpret C:N as an indicator of water quality. For example, in coastal habitats, there were clear and predictable effects of light and DIN loads on C:N within the Wet Tropics, Burdekin and Mackay-Whitsunday, but this trend was not observed for other habitats, or when exploring results on species separately.

Furthermore, C:N does not appear to relate to future changes in seagrass abundance. In all habitats and regions, there was no correlation between C:N and the change in seagrass cover over the following year. The exception was for the coastal sites in the Burdekin region, in which higher C:N was strongly associated with greater seagrass loss, which is counter to the findings from previous experimental studies (McMahon et al 2013). Therefore, C:N is not an ‘early-warning’ indicator of imminent seagrass decline, at least not when applied over the time-scales used here.

Tissue nutrients, including but not limited to C:N is used as an indicator of ecosystem health elsewhere in the world including in the USA and in Europe (Duarte 1990a; Fourqurean et al. 2003; Roca et al. 2016) and is sensitive to variable conditions in both *in-situ* and experimental studies (Grice et al. 1996b; Collier et al. 2009a; Cabaço et al. 2013). Therefore, it was selected by the integration team as one of three indicators for scoring seagrass health for the Reef report card, which initially focused on water quality improvement in the Reef, based on management activities within Reef catchments. In the Reef, the seagrass diversity as well as the complexity of environmental and biological drivers of seagrass condition and of C:N appears to overshadow the effect of the water quality variables used in these models. Furthermore, C:N is measured once per year in the growth season while the response time of the indicator is weeks to months (McMahon et al 2013, Roca et al. 2016). This annual reporting time-frame may not be suitable for this indicator. However, over 15 years of annual tissue nutrient sampling, a substantial ‘base-line’ C:N has been developed for comparison to global averages (Figure 2 and 3) and which could be used for tracking long-term changes in nutrient availability over time.

The inconsistent and unpredictable response of C:N to water quality may be related to the lack of variation in the predictor variables used (relative DIN loads and benthic light), and the way they were summarised for analysis. Aside from benthic light levels, there is no water quality data that is available at a proximity and scale that is ideal for the seagrass sites in the MMP, and so we have relied on coarse-scale DIN exposure and *in situ* water quality from nearby sites. DIN loads did not vary much in several of the habitats and regions, in particular in the Wet Tropics, Fitzroy and Burnett-Mary regions. Loads from these rivers do vary over time (Griber et al 2020), but the predicted load reaching seagrass sites is affected by proximity to the river (some are very close, while others are very distant from the rivers) as well as assumptions in the model about dispersion (Gruber et al 2020). The loading models are in the process of being updated, including updates to the dispersion.

Another reason for the low range may be due to the method of summarising the variables over annual time frames (annual loads, annual average benthic light), which was necessary to match the C:N data measured in the late dry (September – November) to load data from the previous wet season. Even benthic light, while measured within the seagrass habitat, has been averaged to a coarse level (annual average benthic light levels) in an attempt to span the range of other data (DIN loads in previous wet season, December to April), and the C:N which is collected in the following dry season (either September or October). Therefore, the complexity of responses in C:N relative to water quality parameters may also be affected by the suitability of the predictor variables used in the models here.

In some cases we have undertaken only preliminary exploratory analysis i.e. with the *in situ* water quality data. These analyses could be explored in more detail, and could include analysis of C:P and N:P, and any other available nutrient data (e.g. historical sediment nutrient data, eReefs predicted loads, updated loading model results).

This analysis has made us reconsider the relevance and reliability of retaining the C:N ratio as a metric with equal weight into the reef health Index calculation

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Case study Supplementary Material – Statistical outputs

All species pooled

Table A1. *Model 1 (M1) – summary of the GLM used to test for the effects of DIN loading, mean daily light, region, and habitat on the C:N ratio of all seagrasses species pooled. Estimates are on the log scale. Bold P-values indicates significant*

Parameter	Estimate	Std Error	t-value	P-value
Intercept (Burdekin and Coastal intertidal)	2.47E+00	5.75E-02	43.025	**<2.00E-16**
DIN	-5.16E-03	3.81E-03	-1.355	0.17583
Habitat – Estuarine intertidal (EI)	1.56E+00	5.13E-01	3.041	**0.00246**
Habitat – Reef intertidal (RI)	2.08E-02	1.61E-01	0.129	0.89707
Mean_DL	4.33E-02	5.31E-03	8.150	**1.85E-15**
NRM_subregion – Mackay–Whitsunday (MW)	-1.02E-01	3.40E-02	-2.992	**0.00288**
NRM_subregion – Northern Wet Tropics (NWT)	-7.75E-01	7.77E-02	-9.975	**<2.00E-16**
NRM_subregion - Southern Wet Tropics (SWT)	-8.58E-02	8.86E-02	-0.969	0.33306
DIN:EI	-9.07E-02	3.39E-02	-2.679	**0.00757**
DIN:RI	2.53E-02	8.80E-03	2.873	**0.0042**
DIN:Mean_DL	2.49E-05	2.72E-04	0.091	0.92732
DIN:MW	2.64E-04	2.55E-03	0.104	0.91755
DIN:NWT	1.05E-02	3.16E-03	3.308	**0.00099**
DIN:SWT	5.10E-04	4.21E-03	0.121	0.90369
EI:Mean_DL	-1.41E-01	4.58E-02	-3.089	**0.0021**
RI:NRM_subregionMackay	-3.68E-03	1.18E-02	-0.313	0.75413
RI:MW	-3.86E-01	4.58E-02	-8.422	**2.33E-16**
RI:SWT	4.78E-01	5.35E-02	8.941	**<2.00E-16**
DIN:EI:Mean_DL	7.51E-03	2.94E-03	2.555	**0.01083**
DIN:RI:Mean_DL	-1.75E-03	5.98E-04	-2.921	**0.00361**

Table A2. *Model 1 (M1) – ANOVA of the effects of DIN loading, mean daily light, region, and habitat on the C:N ratio of all seagrasses species pooled. Bold P-values indicates significant.*

Parameters	Df	Sum Sq	Mean Sq	F value	P-value
DIN	1	6	6.1	0.754	0.38557
Habitat	2	952	475.8	59.209	**<2.00E-16**
Mean_DL	1	167	166.6	20.731	**6.30E-06**
NRM_subregion	3	762	254.1	31.620	**<2.00E-16**
DIN:Habitat	2	370	185.0	23.016	**2.19E-10**
DIN:Mean_DL	1	29	28.6	3.560	**0.05963**
DIN:NRM_subregion	3	279	93.0	11.576	**2.12E-07**
Habitat:Mean_DL	2	364	181.9	22.633	**3.13E-10**
Habitat:NRM_subregion	2	1962	980.8	122.048	**<2.00E-16**
DIN:Habitat:Mean_DL	2	105	52.6	6.539	**0.00154**
Residuals	655	5264	8.0		

Halodule uninervis

Table A3. *Model 1 (M_Hu) – summary of the GLM used to test for the effects of DIN loading, mean daily light, region, and habitat on the C:N ratio of H. uninervis. Estimates are on the log scale. Bold font indicates significant*

	Estimate	Std Error	t value	P-value
Intercept (Burdekin and Coastal intertidal and f_DL1)	2.72E+00	3.90E-02	69.856	**<2.00E-16**
DIN	-1.55E-02	3.39E-03	-4.563	**6.49E-06**
f_DL2	1.33E-01	4.51E-02	2.939	**0.003455**
Habitat – Reef intertidal (RI)	2.71E-01	5.76E-02	4.707	**3.34E-06**
Habitat – Reef subtidal (RS)	2.47E-01	8.15E-02	3.023	**0.002642**
NRM_subregion - Mackay–Whitsunday (MW)	-1.13E-01	5.13E-02	-2.199	**0.028341**
NRM_subregion – Northern Wet Tropics (NWT)	-3.91E-01	1.12E-01	-3.490	**0.000530**
NRM_subregion – Southern Wet Tropics (SWT)	2.26E-02	1.23E-01	0.183	0.854589
DIN:f_DL2	3.53E-03	2.73E-03	1.294	0.196374
DIN:RI	9.63E-03	4.26E-03	2.261	**0.024215**
DIN:RS	9.46E-03	4.91E-03	1.926	0.054692
DIN:MW	9.39E-04	4.78E-03	0.197	0.844260
DIN:NWT	1.90E-02	5.53E-03	3.443	**0.000628**
DIN:SWT	4.24E-05	5.83E-03	0.007	0.994202
f_DL2:RI	2.30E-02	6.55E-02	0.351	0.725965
f_DL2:RS	1.73E-01	9.38E-02	1.846	0.065602
RI:MW	-5.32E-01	7.31E-02	-7.270	**1.56E-12**
RI:NWT	1.76E-01	1.44E-01	1.220	0.222983
RS:NWT	1.81E-01	1.56E-01	1.158	0.247463
RI:SWT	-6.37E-02	1.53E-01	-0.416	0.677439
DIN:f_DL2:RI	-7.97E-03	3.69E-03	-2.158	**0.031408**
DIN:f_DL2:RS	-1.53E-02	4.69E-03	-3.256	**0.001215**
DIN:RI:MW	5.75E-03	6.23E-03	0.923	0.356258
DIN:RI:NWT	-7.79E-03	7.05E-03	-1.105	0.269671
DIN:RS:NWT	-3.66E-03	7.37E-03	-0.496	0.620017
DIN:RI:SWT	2.27E-03	7.33E-03	0.309	0.757181

Table A4. *Model 1 (M_Hu) – ANOVA of the effects of DIN loading, mean daily light, region, and habitat on the C:N ratio of H. uninervis.*

	Df	Sum Sq	Mean Sq	F value	P-value
DIN	1	20.9	20.9	3.217	0.07354
f_DL	1	334.5	334.5	51.602	**2.76E-12**
Habitat	2	2468	1234	190.370	**<2.00E-16**
NRM_subregion	3	1108.8	369.6	57.020	**<2.00E-16**
DIN:f_DL	1	68.6	68.6	10.587	**0.00122**
DIN:Habitat	2	180.5	90.2	13.921	**1.35E-06**
DIN:NRM_subregion	3	151.4	50.5	7.784	**4.44E-05**
f_DL:Habitat	2	25.5	12.7	1.965	0.14129
Habitat:NRM_subregion	4	599.1	149.8	23.107	**<2.00E-16**
DIN:f_DL:Habitat	2	61.0	30.5	4.702	**0.00952**
DIN:Habitat:NRM_subregion	4	12.0	3.0	0.462	0.76335

Zostera muelleri

Table A5. *Model 1 (M_Zm) – summary of the GLM used to test for the effects of DIN loading, mean daily light, region, and habitat on the C:N ratio of Z. muelleri. Estimates are on the log scale.*

Parameters	Estimate	Std.Error	t value	Pr(>|t|)
Intercept (Burdekin and Coastal intertidal and f_DL1)	3.247998	0.180446	18.000	**<2.00E-16**
DIN	-0.013697	0.012920	-1.060	0.289919
f_DL2	-0.364545	0.215927	-1.688	0.092361
Habitat – Estuarine intertidal (EI)	0.180338	0.079781	2.260	**0.024488**
Habitat – Reef intertidal (RI)	-0.044212	0.071794	-0.616	0.538465
NRM_subregion – Burnett–Mary (BM)	-0.655728	0.209507	-3.130	**0.001915**
NRM_subregion – Fitzroy (F)	-0.861091	0.267786	-3.216	**0.001439**
NRM_subregion – Mackay–Whitsunday (MW)	-0.478684	0.185727	-2.577	**0.010416**
DIN:f_DL2	0.042922	0.038632	1.111	0.267403
DIN:EI	-0.01885	0.006537	-2.884	**0.004204**
DIN:RI	0.001711	0.005387	0.318	0.751016
DIN:BM	0.093021	0.021728	4.281	**2.48E-05**
DIN:F	0.260597	0.081380	3.202	**0.001505**
DIN:MW	0.010012	0.013486	0.742	0.458404
f_DL2:EI	-0.421424	0.107156	-3.933	**0.000104**
f_DL2:RI	0.112340	0.132489	0.848	0.397133
f_DL2:BM	0.754068	0.253329	2.977	**0.003143**
f_DL2:F	0.876388	0.298419	2.937	**0.003564**
f_DL2:MW	0.631384	0.223604	2.824	**0.005054**
EI:F	0.284178	0.224101	1.268	0.205718
RI:F	0.476852	0.251480	1.896	0.058863
EI:MW	NA	NA	NA	NA
RI:MW	NA	NA	NA	NA
DIN:f_DL2:EI	0.014667	0.007928	1.850	0.065243
DIN:f_DL2:RI	-0.033107	0.011805	-2.804	**0.005357**
DIN:f_DL2:BM	-0.103669	0.043571	-2.379	**0.017948**
DIN:f_DL2:F	-0.283624	0.090880	-3.121	**0.001972**
DIN:f_DL2:MW	-0.036486	0.038869	-0.939	0.348630
DIN:EI:F	-0.213508	0.080761	-2.644	**0.008616**
DIN:RI:F	-0.257602	0.081934	-3.144	**0.001827**
f_DL2:EI:F	0.071625	0.251743	0.285	0.776203
f_DL2:RI:F	-0.822970	0.294658	-2.793	**0.005546**
DIN:f_DL2:EI:F	0.201547	0.084951	2.373	**0.018276**
DIN:f_DL2:RI:F	0.299109	0.085025	3.518	**0.000500**

Table A6. *Model 1 (M_Zm) – ANOVA of the effects of DIN loading, mean daily light, region, and habitat on the C:N ratio of Z. muelleri.*

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
DIN	1	13.9	13.90	1.789	0.182055
f_DL	1	0.4	0.39	0.051	0.822194
Habitat	2	436.2	218.09	28.065	**6.23E-12**
NRM_subregion	3	340.1	113.37	14.589	**6.50E-09**
DIN:f_DL	1	145.5	145.53	18.728	**2.03E-05**
DIN:Habitat	2	28.3	14.13	1.819	0.163971
DIN:NRM_subregion	3	412.0	137.32	17.671	**1.29E-10**
f_DL:Habitat	2	215.5	107.73	13.864	**1.71E-06**
f_DL:NRM_subregion	3	358.6	119.53	15.382	**2.35E-09**
Habitat:NRM_subregion	2	182.4	91.20	11.736	**1.22E-05**
DIN:f_DL:Habitat	2	63.5	31.75	4.086	**0.017714**
DIN:f_DL:NRM_subregion	3	76.8	25.60	3.294	**0.020868**
DIN:Habitat:NRM_subregion	2	3.0	1.50	0.193	0.824878
f_DL:Habitat:NRM_subregion	2	120.5	60.23	7.750	**0.000519**
DIN:f_DL:Habitat:NRM_subregion	2	151.8	75.89	9.766	**7.70E-05**

Appendix 2 Seagrass condition indicator guidelines

A2.1 Seagrass abundance

The status of seagrass abundance (per cent cover) was determined using the seagrass abundance guidelines developed by McKenzie (2009). The seagrass abundance measure in the MMP is the average per cent cover of seagrass per monitoring site. Individual site and subregional (habitat type within each NRM region) seagrass abundance guidelines were developed based on per cent cover data collected from individual sites and/or reference sites (McKenzie 2009). Guidelines for individual sites were only applied if the conditions of the site aligned with reference site conditions.

A reference site is a site whose condition is considered to be a suitable baseline or benchmark for assessment and management of sites in similar habitats. Ideally, seagrass meadows in near pristine condition with a long-term abundance database would have priority as reference sites. However, as near-pristine meadows are not available, sites which have received less intense impacts can justifiably be used. In such situations, reference sites are those where the condition of the site has been subject to minimal/limited disturbance for 3-5 years. The duration of 3-5 years is based on recovery from impact times (Campbell and McKenzie 2004).

There is no set/established protocol for the selection of reference sites and the process is ultimately iterative. The criteria for defining a minimally/least disturbed seagrass reference site is based on Monitoring River Health Initiative (1994) and includes some or all of the following:

- beyond 10 km of a major river: as most suspended solids and particulate nutrients are deposited within a few kilometres of river mouths (McCulloch *et al.* 2003; Webster and Ford 2010; Bainbridge *et al.* 2012; Brodie *et al.* 2012)
- no major urban area/development (>5000 population) within 10 km upstream (prevailing current)
- no significant point source wastewater discharge within the estuary
- has not been impacted by an event (anthropogenic or extreme climate) in the last 3-5 years
- where the species composition is dominated by the foundation species expected for the habitats (Carruthers *et al.* 2002)
- does not suggest the meadow is in recovery (i.e. dominated by early colonising).

The 80th, 50th and 20th percentiles were used to define the guideline values as these are recommended for water quality guidelines (Department of Environment and Resource Management 2009), and there is no evidence that this approach would not be appropriate for seagrass meadows in the Reef. At the request of the Paddock to Reef Integration Team, the 80th percentile was changed to 75th to align with other Paddock to Reef report card components. By plotting the percentile estimates with increasing sample size, the reduction in error becomes apparent as it moves towards the true value (e.g. Figure 89).

Across the majority of reference sites, variance for the 50th and 20th percentiles levelled off at around 15–20 samples (i.e. sampling events), suggesting this number of samples was sufficient to provide a reasonable estimate of the true percentile value. This sample size is reasonably close to the ANZECC (2000) Guidelines recommendation of 24 data values. If the variance had not plateud, the percentile values at 24 sampling events was selected to best represent the variance as being captured. This conforms with Kiliminster *et al.* (2015) definition where a enduring meadow is present for 5 years.

Nonlinear regressions (exponential rise to maximum, two parameter) were then fitted to per cent cover percentile values at each number of sampling events using the following model:

$$y = a\left(1 - e^{-bx}\right)$$

where *y* is the seagrass cover percentile at each number of sampling events (*x*), *a* is the asymptotic average of the seagrass cover percentile, and *b* is the rate coefficient that determines how quickly (or slowly) the maximum is attained (i.e. the slope). The asymptotic average was then used as the guideline value for each percentile (Table 17).

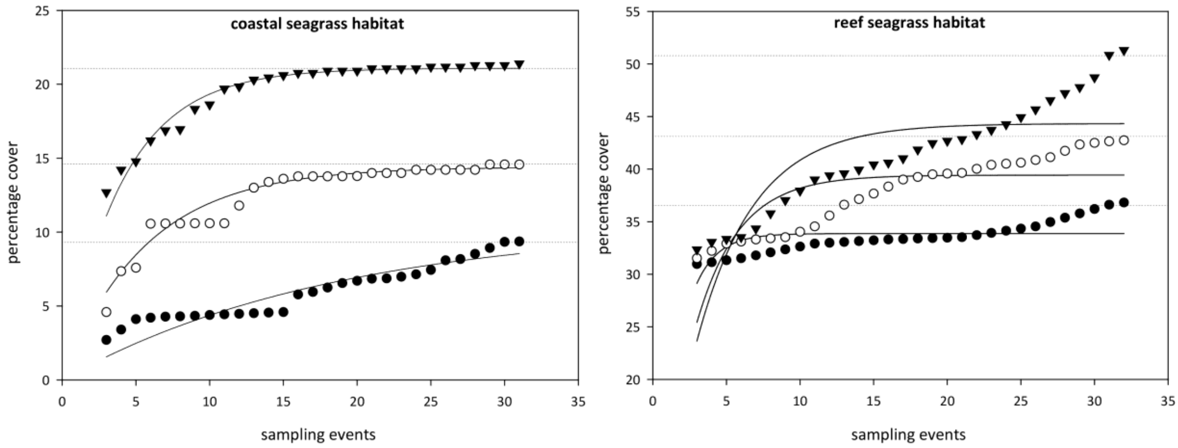

Figure 89. *Relationship between sample size and the error in estimation of percentile values for seagrass abundance (per cent cover) in coastal and reef seagrass habitats in the Wet Tropics NRM. ▼= 75th percentile, ○ = 50th percentile,● = 20th percentile. Horizontal lines are asymptotic averages for each percentile plot.*

As sampling events occur every 3-6 months depending on the site, this is equivalent to 3–10 years of monitoring to establish percentile values. Based on the analyses, it was recommended that estimates of the 20th percentile at a reference site should be based on a minimum of 18 samples collected over at least three years. For the 50th percentile a smaller minimum number of samples (approximately 10–12) would be adequate but in most situations it would be necessary to collect sufficient data for the 20th percentile anyway. For seagrass habitats with low variability, a more appropriate guideline was the 10th percentile primarily the result of seasonal fluctuations (as nearly every seasonal low would fall below the 20th percentile). Percentile variability was further reduced within a habitat type of each region by pooling at least two (preferably more) reference sites to derive guidelines. The subregional guideline is calculated from the mean of all reference sites within a habitat type within a region.

Using the seagrass guidelines, seagrass state can be determined for each monitoring event at each site and allocated as:

- good (median abundance at or above 50th percentile)
- moderate (median abundance below 50th percentile and at or above 20th percentile)
- poor (median abundance below 20th or 10th percentile).

For example, when the median seagrass abundance for Yule Point is plotted against the 20th and 50th percentiles for coastal habitats in the Wet Tropics (Figure 90), it indicates that the meadows were in a poor condition in mid-2000, mid-2001 and mid-2006 (based on abundance).

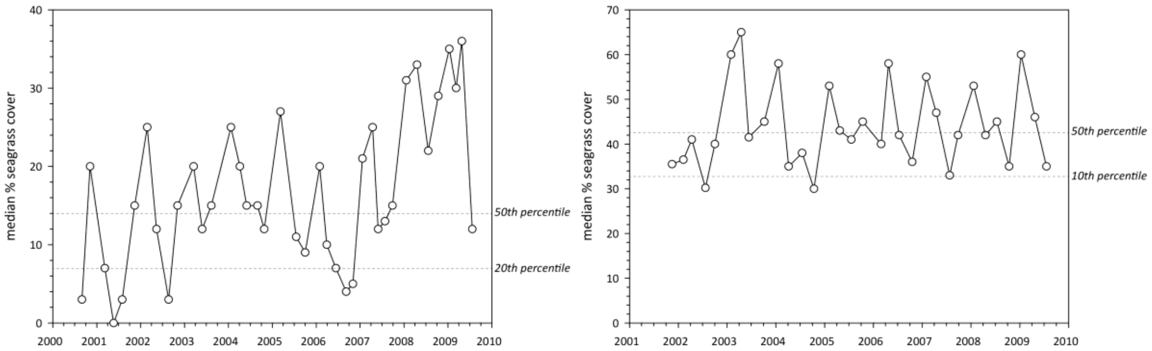

Figure 90. *Median seagrass abundance (per cent cover) at Yule Point (left) and Green Island (right) plotted against the 50th and 20th percentiles for coastal and intertidal reef seagrass habitat in the Wet Tropics.*

Similarly, when the median seagrass abundance for Green Island is plotted against the 20th and 50th percentiles for intertidal reef habitats in the Wet Tropics, it indicates that the meadows were in a poor condition in the middle of most years (based on abundance). However, the poor rating is most likely a consequence of seasonal lows in abundance. Therefore, in this instance, it was more appropriate to set the guideline at the 10th rather than the 20th percentile.

Using this approach, subregional seagrass abundance guidelines (hereafter known as "the seagrass guidelines") were developed for each seagrass habitat type where possible (Table 17). If an individual site had 18 or more sampling events and no identified impacts (e.g. major loss from cyclone), an abundance guideline was determined at the site or location level rather than using the subregional guideline from the reference sites (i.e. as more guidelines are developed at the site level, they contribute to the subregional guideline).

After discussions with GBRMPA scientists and the Paddock to Reef integration team, the seagrass guidelines were further refined by allocating the additional categories of:

- very good (median abundance at or above 75th percentile)
- very poor (median abundance below 20th or 10th percentile and declined by >20% since previous sampling event).

Seagrass state was then rescaled to a five point scale from 0 to 100 to allow integration with other components of the Paddock to Reef report card (Department of the Premier and Cabinet 2014). Please note that the scale from 0 to 100 is unitless and should not be interpreted as a proportion or ratio.

Table 17. *Seagrass percentage cover guidelines ("the seagrass guidelines") for each site/location and the subregional guidelines (bold) for each NRM habitat. Values in light grey not used. ^ denotes regional reference site, * from nearest adjacent region. For site details, see Tables 3 & 4.*

NRM region	site/ location	Habitat	percentile guideline 10th	20th	50th	75th
Cape York						
	AP1^	reef intertidal	11	16.8	18.9	23.7
	AP2	reef intertidal	11		18.9	23.7
	FR	reef intertidal		16.8	18.9	23.7
	ST	reef intertidal		16.8	18.9	23.7
	YY	reef intertidal		16.8	18.9	23.7
	NRM	***reef intertidal***	***11***	***16.8***	***18.9***	***23.7***
	SR*	coastal intertidal		6.6	12.9	14.8
	BY*	coastal intertidal		6.6	12.9	14.8
	NRM	***coastal intertidal****	***5***	***6.6***	***12.9***	***14.8***
	LR	coastal subtidal		6.6	12.9	14.8
	NRM	***coastal subtidal***		***6.6***	***12.9***	***14.8***

Wet Tropics						
	LB	coastal intertidal		6.6	12.9	14.8
	YP1^	coastal intertidal	4.3	7	14	15.4
	YP2^	coastal intertidal	5.7	6.2	11.8	14.2
	NRM	***coastal intertidal***	***5***	***6.6***	***12.9***	***14.8***
	MS	coastal subtidal		6.6	12.9	14.8
	NRM	***coastal subtidal***		***6.6***	***12.9***	***14.8***
	DI	reef intertidal	27.5		37.7	41
	GI1^	reef intertidal	32.5	38.2	42.7	45.5
	GI2^	reef intertidal	22.5	25.6	32.7	36.7
	LI1	reef intertidal	27.5		37.7	41
	GO1	reef intertidal	27.5		37.7	41
	NRM	***reef intertidal***	***27.5***	***31.9***	***37.7***	***41***
	DI3	reef subtidal	22	26	33	39.2
	GI3^	reef subtidal	22	26	33	39.2
	LI2	reef subtidal	22	26	33	39.2
	NRM	***reef subtidal***	***22***	***26***	***33***	***39.2***
Burdekin	BB1^	coastal intertidal	16.3	21.4	25.4	35.2
	SB1^	coastal intertidal	7.5	10	16.8	22
	SB2	coastal intertidal		10	16.8	22
	JR	coastal intertidal		*15.7*	*21.1*	*28.6*
	NRM	***coastal intertidal***	***11.9***	***15.7***	***21.1***	***28.6***
	MI1^	reef intertidal	23	26	33.4	37
	MI2^	reef intertidal	21.3	26.5	35.6	41
	NRM	***reef intertidal***	***22.2***	***26.3***	***34.5***	***39***
	MI3^	reef subtidal	18	22.5	32.7	36.7
	NRM	***reef subtidal***	***18***	***22.5***	***32.7***	***36.7***
Mackay–Whitsunday						
	SI	estuarine intertidal		18	34.1	54
	NRM	***estuarine intertidal***	***10.8****	***18****	***34.1****	***54****
	PI2^	coastal intertidal	18.1	18.7	25.1	27.6
	PI3^	coastal intertidal	6.1	7.6	13.1	16.8
	MP2	coastal intertidal		18.9	22.8	25.4
	MP3	coastal intertidal		17.9	20	22.3
	NRM	***coastal intertidal***	***12.1***	***13.2***	***19.1***	***22.2***
	NB	coastal subtidal		13.2	19.1	22.2
	NRM	***coastal subtidal***	***12.1***	***13.2***	***19.1***	***22.2***
	HB1^	reef intertidal		10.53	12.9	14.2
	HB2^	reef intertidal		7.95	11.59	13.4
	HM	reef intertidal		9.2	12.2	13.8
	NRM	***reef intertidal***		***9.2***	***12.2***	***13.8***
	TO	reef subtidal		22.5	32.7	36.7
	NRM	***reef subtidal****	***18****	***22.5****	***32.7****	***36.7****
Fitzroy						
	GH	estuarine intertidal		18	34.1	54
	NRM	***estuarine intertidal***	***10.8****	***18****	***34.1****	***54****
	RC1^	coastal intertidal	18.6	20.6	24.4	34.5
	WH1^	coastal intertidal	13.1	14.4	18.8	22.3
	NRM	***coastal intertidal***	***15.85***	***17.5***	***21.6***	***28.4***
	GK	reef intertidal		9.2	12.2	13.8
	NRM	***reef intertidal***		***9.2****	***12.2****	***13.8****
Burnett–Mary						
	RD	estuarine intertidal		18	34.1	54
	UG1^	estuarine intertidal	10.8	18	34.1	54
	UG2	estuarine intertidal		18	34.1	54
	NRM	***estuarine intertidal***	***10.8***	***18***	***34.1***	***54***
	BH1^	coastal intertidal		7.8	11.9	21.6
	BH3	coastal intertidal		7.8	11.9	21.6
	NRM	***coastal intertidal***		***7.8***	***11.9***	***21.6***

A2.2 Seagrass reproductive effort

The reproductive effort is the number of reproductive structures (inflorescence, fruit, spathe, seed) per core. Given the high diversity of seagrass species that occur in the Reef coastal zone (Waycott *et al.* 2007), and their variability in production of reproductive structures (e.g. Orth *et al.* 2006), a metric that incorporates all available information on the production of flowers and fruits per unit area is used.

The production of seeds also reflects a simple measure of the capacity of a seagrass meadow to recover following large scale impacts (Collier and Waycott 2009). As it is well recognized that coastal seagrasses are prone to small scale disturbances that cause local losses (Collier and Waycott 2009) and then recover in relatively short periods of time, the need for a local seed source is considerable. In the Reef, the production of seeds comes in numerous forms and seed banks examined at MMP sites are limited to foundational seagrass species (seeds >0.5mm diameter). At this time, seed banks have not been included in the metric for reproductive effort, but methods for future incorporation are being explored.

Using the annual mean of all species pooled in the late dry and comparing with the long-term (2005–2010) average for Reef habitat (coastal intertidal = 8.22±0.71, estuarine intertidal = 5.07±0.41, reef intertidal = 1.32±0.14), the reproductive effort is scored as the number of reproductive structures per core and the overall status determined as the ratio of the average number observed divided by the long term average.

A2.3 Seagrass nutrient status.

The molar ratios of seagrass tissue carbon relative to nitrogen (C:N) were chosen as the indicator for seagrass nutrient status, as an atomic C:N ratio of <20 may suggest either reduced light availability or nitrogen enrichment. Both of these deviations may indicate reduced water quality.

As changing leaf C:N ratios have been found in a number of experiments and field surveys to be related to available nutrient and light levels (Abal *et al.* 1994; Grice *et al.* 1996; Cabaço and Santos 2007; Collier *et al.* 2009) they can be used as an indicator of the light that the plant is receiving relative to nitrogen availability or N surplus to light. With light limitation, seagrass plants are unable to build structure, hence the proportion of carbon in the leaves decreases relative to nitrogen. Experiments on seagrasses in Queensland have reported that at an atomic C:N ratio of <20, may suggest reduced light availability relative to nitrogen availability (Abal *et al.* 1994; AM Grice*, et al.*, 1996;). The light availability to seagrass is not necessarily an indicator of light in the water column, but an indicator of the light that the plant is receiving as available light can be highly impacted by epiphytic growth or sediment smothering photosynthetic leaf tissue. However, C:N must be interpreted with caution as the level of N can also influence the ratio in oligotrophic environments (Atkinson and Smith 1983; Fourqurean *et al.* 1992). Support for choosing the elemental C:N ratio as the indicator also comes from preliminary analysis of MMP data in 2009 which found that the C:N ratio was the only nutrient ratio that showed a significant relationship (positive) with seagrass cover at coastal and estuarine sites; seagrass tissue C:N ratios explained 58% of the variance of the inter-site seagrass cover data (McKenzie and Unsworth 2009). Using the guideline ratio of 20:1 for the foundation seagrass species, C:N ratios were categorised on their departure from the guideline and transformed to a 0 to 100 score using:

Equation 2 $$\overline{R} = (\mathrm{C:N} \times 5) - 50$$

NB: C:N ratios >35 scored as 100, C:N ratios <10 scored as 0

The score was then used to represent the status to allow integration with other components of the report card.

Appendix 3 Detailed data

Table 18. *Samples collected at each MMP inshore monitoring site per parameter for each season. Activities include: SG = seagrass cover & composition, SM=seed monitoring, TN=tissue nutrients, EM=edge mapping, RH=reproductive health, TL=temperature loggers, LL=light loggers, SH=sediment herbicides. ^=subtidal.*

GBR region	NRM region	Basin	Monitoring location		late dry Season (2018)							late wet Season (2019)					
					SG	**SB**	**TN**	**EM**	**RH**	**TL**	**LL**	**SG**	**SB**	**EM**	**RH**	**TL**	**LL**
Far Northern	Cape York	Jacky Jacky / Olive Pascoe	Shelburne Bay	**SR1**	33	30	3	✓	15	✓							
				SR2	33	30	3	✓	15	✓	✓						
			Piper Reef	**FR1**	33	30	3	✓	15	✓							
				FR2	33	30	3	✓	15	✓	✓						
		Lockhart	Weymouth Bay	**YY1**													
			Lloyd Bay	**LR1^**													
				LR2^													
		Normanby / Jeanie	Flinders Group	**ST1**	33	30	3	✓	15	✓	✓						
				ST2	33	30	3	✓	15	✓							
				FG1^	10												
				FG2^	10												
			Bathurst Bay	**BY1**	33	30	3	✓	15	✓							
				BY2	33	30	3	✓	15	✓	✓						
				BY3^													
				BY4^													
		Endeavour	Archer Point	**AP1**													
				AP2													
Northern	Wet Tropics	Daintree	Low Isles	**LI1**	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
				LI2^	33	30			15	✓	✓	33	30	✓	15	✓	✓
		Mossman / Barron / Mulgrave - Russell / Johnstone	Yule Point	**YP1**	33	30	3	✓	15	✓		33	30	✓	15	✓	
				YP2	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
			Green Island	**GI1**	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
				GI2	33	30	3	✓	15	✓		33	30	✓	15	✓	
				GI3^	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
		Tully / Murray / Herbert	Mission Beach	**LB1**	33	30						33	30				
				LB2	33	30						33	30				
			Dunk Island	**DI1**	33	30	3	✓	15	✓						✓	
				DI2	33	30	3	✓	15	✓	✓					✓	✓
				DI3^	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
			Rockingham Bay	**GO1**													
			Missionary Bay	**MS1^**	9												
				MS2^	9												
Central	Burdekin	Ross / Burdekin	Magnetic Island	**MI1**	33	30	3	✓	15	✓		33	30	✓	15	✓	
				MI2	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
				MI3^	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
			Townsville	**SB1**	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
				SB2	33	30				✓		33	30	✓	15	✓	
				BB1	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
			Bowling Green Bay	**JR1**	33	30	3	✓	15	✓	✓	33		✓		✓	✓
				JR2	33	30	3	✓	15	✓						✓	
	Mackay Whitsunday	Don	Shoal Bay	**HB1**	33	30				✓						✓	
				HB2	33	30				✓						✓	

GBR region	NRM region	Basin	Monitoring location		late dry Season (2018)							late wet Season (2019)					
					SG	SB	TN	EM	RH	TL	LL	SG	SB	EM	RH	TL	LL
		Proserpine	Pioneer Bay	**PI2**	33	30				✓						✓	
				PI3	33	30				✓						✓	
		Proserpine / O'Connell	Repulse Bay	**MP2**	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
				MP3	33	30	3	✓	15	✓		33	30	✓	15	✓	
			Hamilton Is.	**HM1**	33	30	3	✓	15	✓		33	30	✓	15	✓	
				HM2	30	30	3	✓	15	✓	✓	30	30	✓	15	✓	✓
			Whitsunday Island	**TO1^**	9												
				TO2^	10												
			Lindeman Island	**LN1^**	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
				LN2^	33	30	3	✓	15	✓							
		O'Connell	Newry Islands	**NB1^**	10												
				NB2^	10												
		Plane	Sarina Inlet	**SI1**	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
				SI2	33	30	3	✓	15	✓		33	30	✓	15	✓	
Southern	Fitzroy	Fitzroy	Shoalwater Bay	**RC1**	33	30	3	✓	15	✓						✓	
				WH1	33	30	3	✓	15	✓	✓					✓	✓
			Great Keppel Island	**GK1**	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
				GK2	33	30	3	✓	15	✓		33	30	✓	15	✓	
		Boyne	Gladstone Harbour	**GH1**	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
				GH2	33	30	3	✓	15	✓		33	30	✓	15	✓	
	Burnett–Mary	Burnett	Rodds Bay	**RD1**	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
				RD3	33	30	3	✓	15	✓		33	30	✓	15	✓	
		Burrum	Burrum Heads	**BH1**	33	30	3	✓	15	✓		33	30	✓	15	✓	
				BH3	33	30	3	✓	15	✓		33	30	✓	15	✓	
		Mary	Hervey Bay	**UG1**	33	30	3	✓	15	✓		33	30	✓	15	✓	
				UG2	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓

A3.1 Environmental pressures

A3.1.1 Tidal exposure

Table 19. *Height of intertidal monitoring meadows/sites above lowest astronomical tide (LAT) and annual daytime tidal exposure (total hours) when meadows become exposed at a low tide. Year is June–May. Observed tidal heights courtesy Maritime Safety Queensland, 2019. NB: Meadow heights have not yet been determined in the far northern Cape York.*

NRM	Site	Meadow height (above LAT)	Site depth (bMSL)	Meadow height (above LAT) relative to Standard Port	Annual median hours exposed during daylight (long-term)	Per cent of annual daylight hours meadow exposed (long-term)	Annual daytime exposure 2018–19 (hrs)	Per cent of annual daylight hours meadow exposed (2018–19)
Cape Y	AP1	0.46	1.02	0.46	64.17	1.46	43.50	0.99
	AP2	0.46	1.02	0.46	64.17	1.46	43.50	0.99
Wet Tropics	LI1	0.65	0.90	0.65	176.67	4.03	132.50	3.03
	YP1	0.64	0.94	0.64	169.67	3.87	127.00	2.90
	YP2	0.52	1.06	0.52	96.00	2.19	72.00	1.64
	GI1	0.51	1.03	0.61	118.25	2.70	107.00	2.44
	GI2	0.57	0.97	0.67	154.58	3.53	143.50	3.28
	DI1	0.65	1.14	0.54	73.67	1.68	78.500	1.79
	DI2	0.55	1.24	0.44	42.17	0.96	50.00	1.14
	LB1	0.42	1.37	0.31	17.75	0.40	32.50	0.74
	LB2	0.46	1.33	0.35	19.25	0.44	29.50	0.67
Burdekin	BB1	0.58	1.30	0.58	84.5	1.93	57.50	1.31
	SB1	0.57	1.31	0.57	67.08	1.53	56.00	1.28
	MI1	0.65	1.19	0.67	183.00	4.18	84.00	1.92
	MI2	0.54	1.30	0.56	170.00	3.88	54.00	1.23
	JR1	0.47	1.32	0.47	63.33	1.44	57.00	1.30
	JR2	0.47	1.32	0.47	63.33	1.44	57.00	1.30
Mackay–Whitsunday	PI2	0.28	1.47	0.44	80.17	1.83	103.00	2.35
	PI3	0.17	1.58	0.33	40.00	0.91	52.00	1.19
	HM1	0.68	1.52	0.38	55.107	1.26	69.00	1.58
	HM2	0.68	1.52	0.38	55.107	1.26	69.00	1.58
	SI1	0.60	2.80	0.54	24.75	0.56	50.00	1.14
	SI2	0.60	2.80	0.54	24.75	0.56	50.00	1.14
Fitzroy	RC1	2.03	1.30	1.06	163.67	3.73	248.50	5.67
	WH1	2.16	1.17	1.19	236.17	5.39	332.50	7.59
	GK1	0.52	1.93	0.43	33.25	0.76	36.00	0.82
	GK2	0.58	1.87	0.49	49.83	1.14	51.00	1.16
	GH1	0.80	1.57	0.69	97.33	2.22	111.00	2.53
	GH2	0.80	1.57	0.69	91.58	2.09	111.00	2.53
Burnett–Mary	RD1	0.56	1.48	0.56	66.58	1.52	77.00	1.76
	RD2	0.63	1.41	0.63	93.17	2.13	114.00	2.60
	UG1	0.70	1.41	0.70	144.00	3.29	118.50	2.71
	UG2	0.64	1.47	0.64	105.83	2.41	51.50	1.18

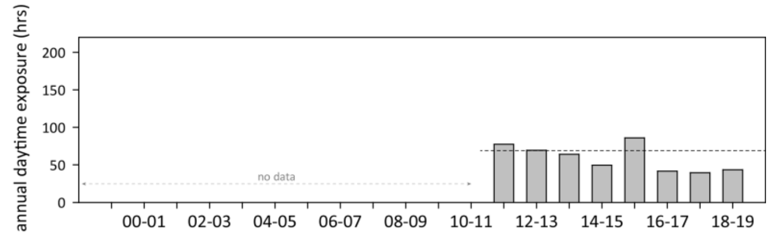

Figure 91. *Annual daytime tidal exposure (total hours) and long-term median (dashed line) of intertidal reef seagrass meadows at Archer Point, Cape York NRM region; 2011–2019. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 19. Observed tidal heights courtesy Maritime Safety Queensland, 2019. NB: Meadow heights have not yet been determined in the far northern Cape York sites.*

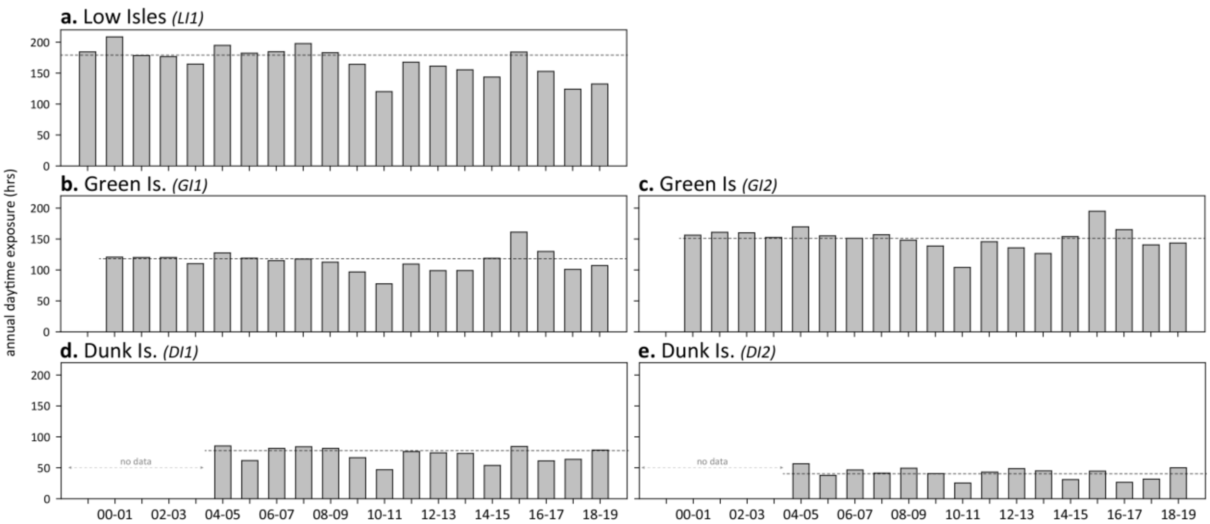

Figure 92. *Annual daytime tidal exposure (total hours) and long-term median (dashed line) of intertidal reef seagrass meadows in the Wet Tropics NRM region; 1999–2019. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 19. Observed tidal heights courtesy Maritime Safety Queensland, 2019.*

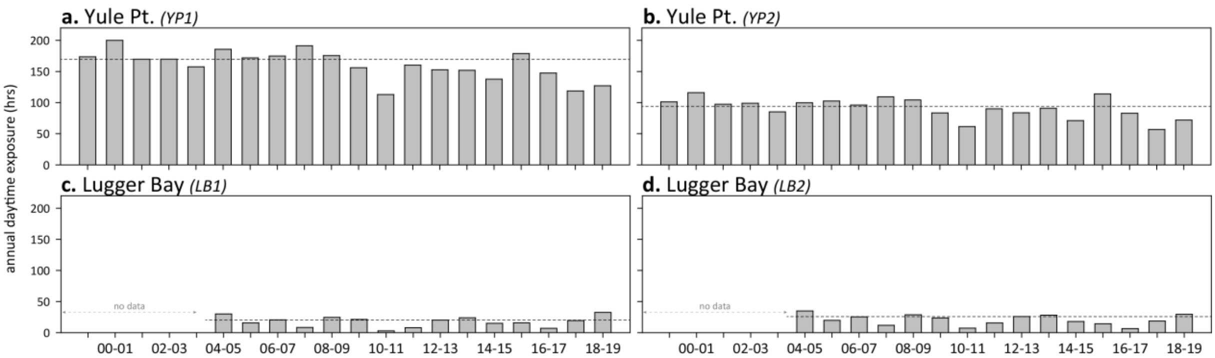

Figure 93. *Annual daytime tidal exposure (total hours) and long-term median (dashed line) of intertidal coastal seagrass meadows in Wet Tropics NRM region; 1999–2019. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 19. Observed tidal heights courtesy Maritime Safety Queensland, 2019.*

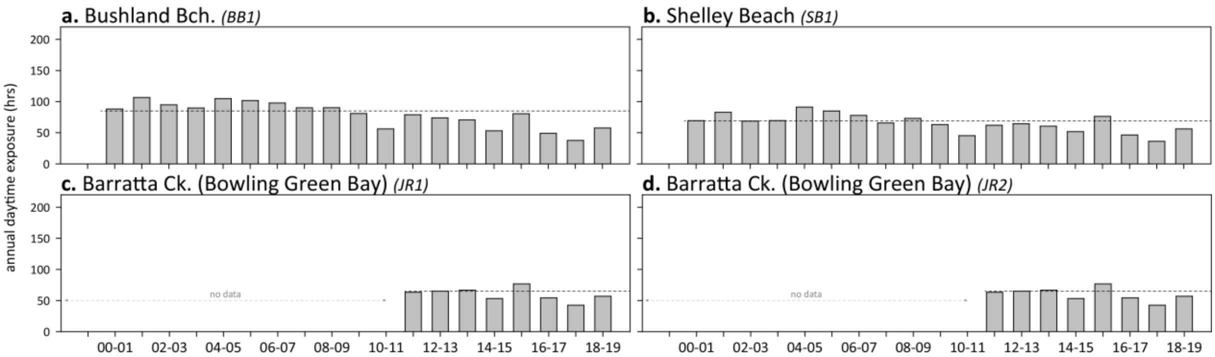

Figure 94. *Annual daytime tidal exposure (total hours) and long-term median (dashed line) of intertidal coastal seagrass meadows in Burdekin NRM region; 2000–2019. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 19. Observed tidal heights courtesy Maritime Safety Queensland, 2019.*

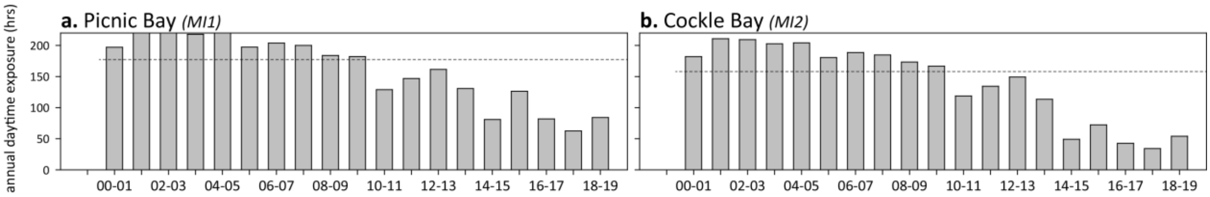

Figure 95. *Annual daytime tidal exposure (total hours) and long-term median (dashed line) of intertidal reef seagrass meadows in Burdekin NRM region; 2000–2019. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 19. Observed tidal heights courtesy Maritime Safety Queensland, 2019.*

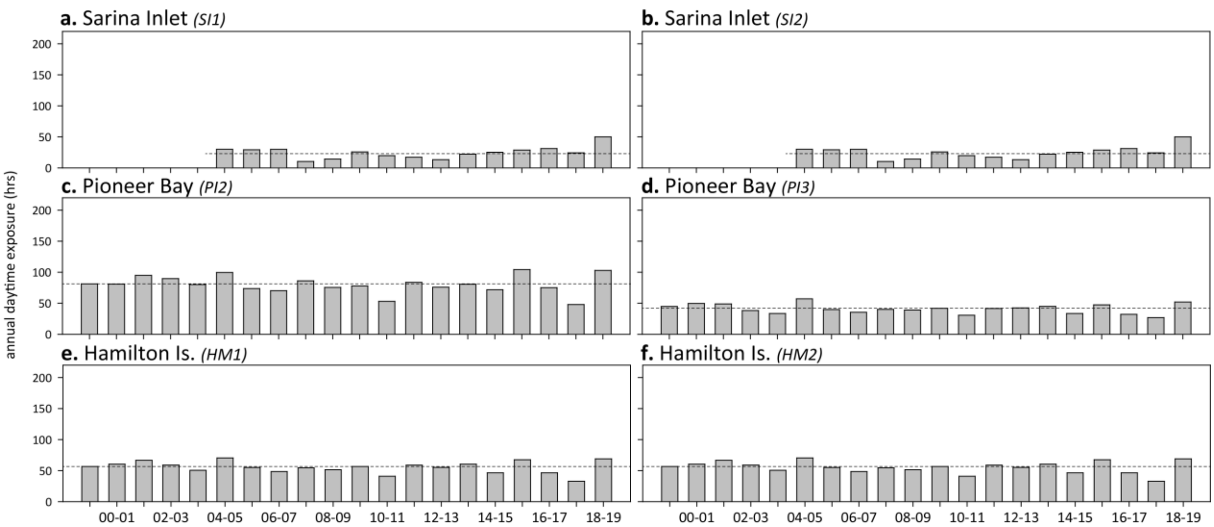

Figure 96. *Annual daytime tidal exposure (total hours) and long-term median (dashed line) of intertidal estuarine (a, b) coastal (c, d) and reef (e, f) seagrass meadows in Mackay–Whitsunday NRM region; 1999–2019. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 19. Observed tidal heights courtesy Maritime Safety Queensland, 2019.*

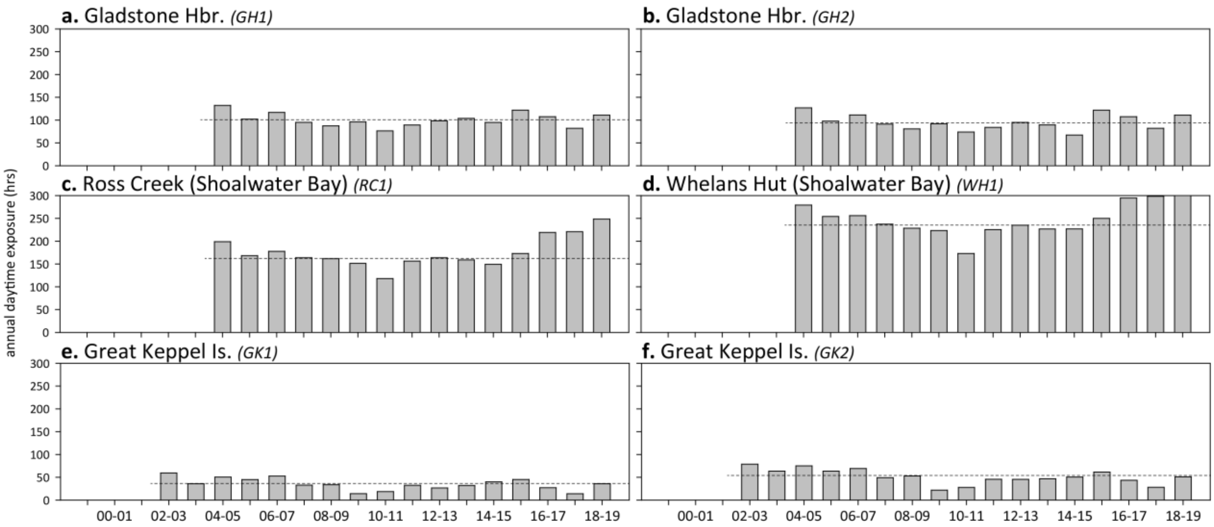

Figure 97. *Annual daytime tidal exposure (total hours) and long-term median (dashed line) of intertidal estuarine (a, b) coastal (c, d) and reef (e, f) seagrass meadows in the Fitzroy NRM region; 1999–2019. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 19. Observed tidal heights courtesy Maritime Safety Queensland, 2019.*

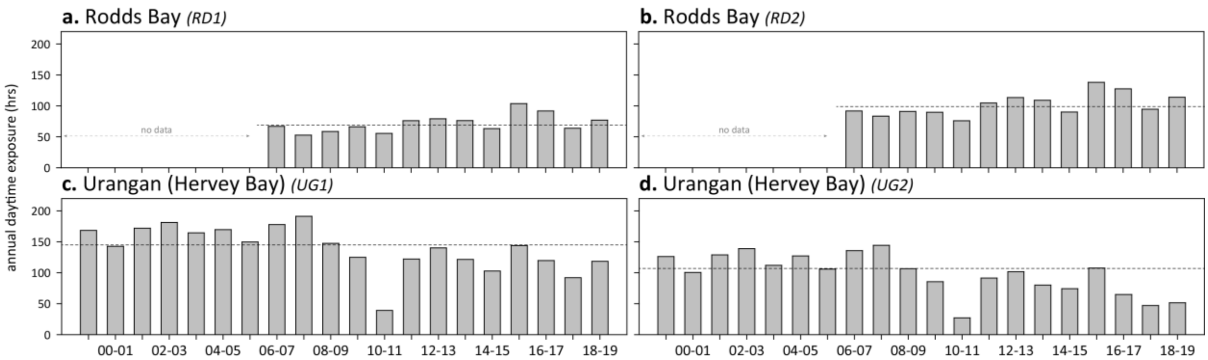

Figure 98. *Annual daytime tidal exposure (total hours) and long-term median (dashed line) of intertidal estuarine seagrass meadows in the Burnett–Mary NRM region; 1999–2019. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 19. Observed tidal heights courtesy Maritime Safety Queensland, 2019.*

A3.1.2 Light at seagrass canopy

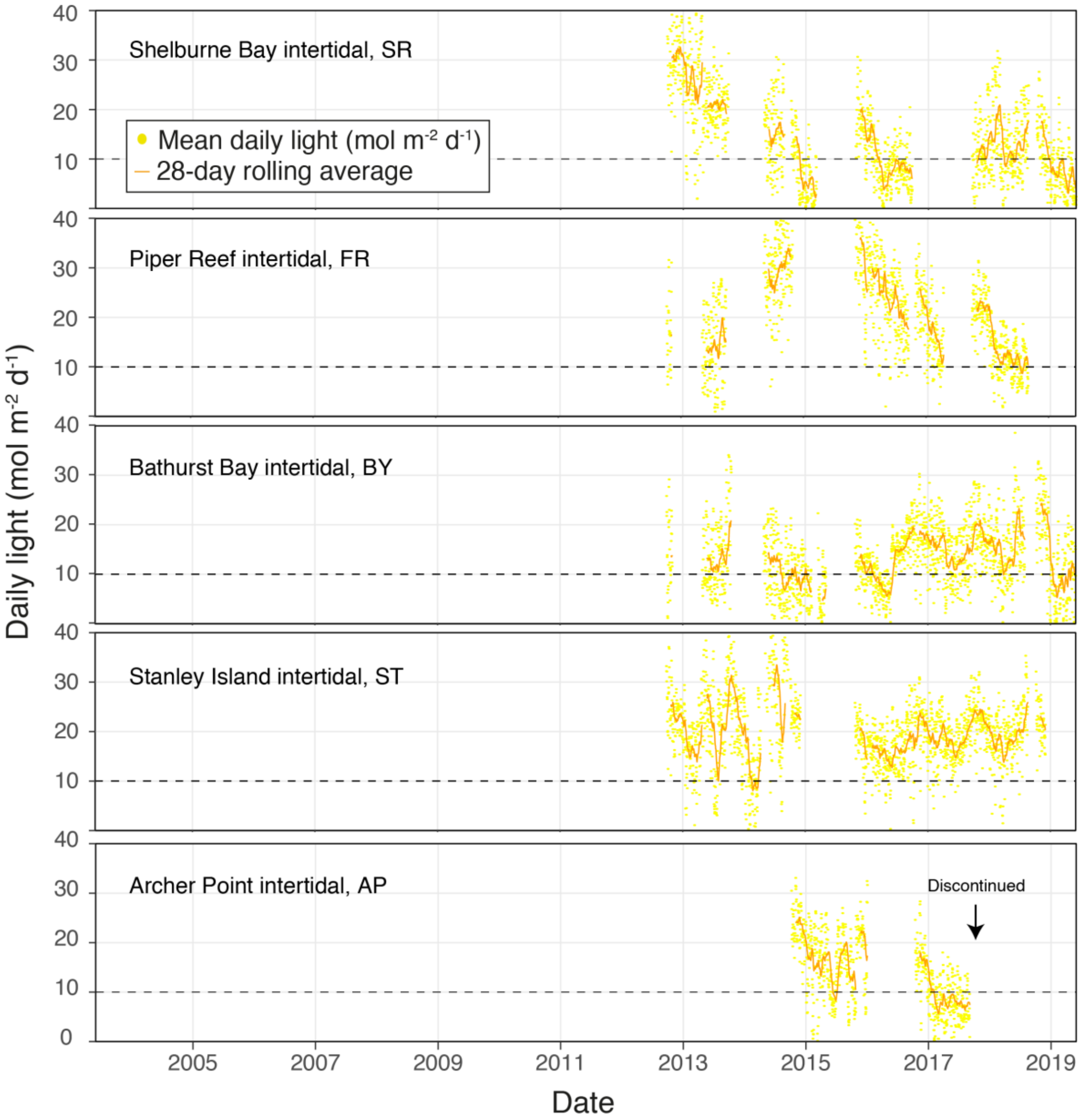

Figure 99. *Daily light and 28-day rolling average at Cape York locations.*

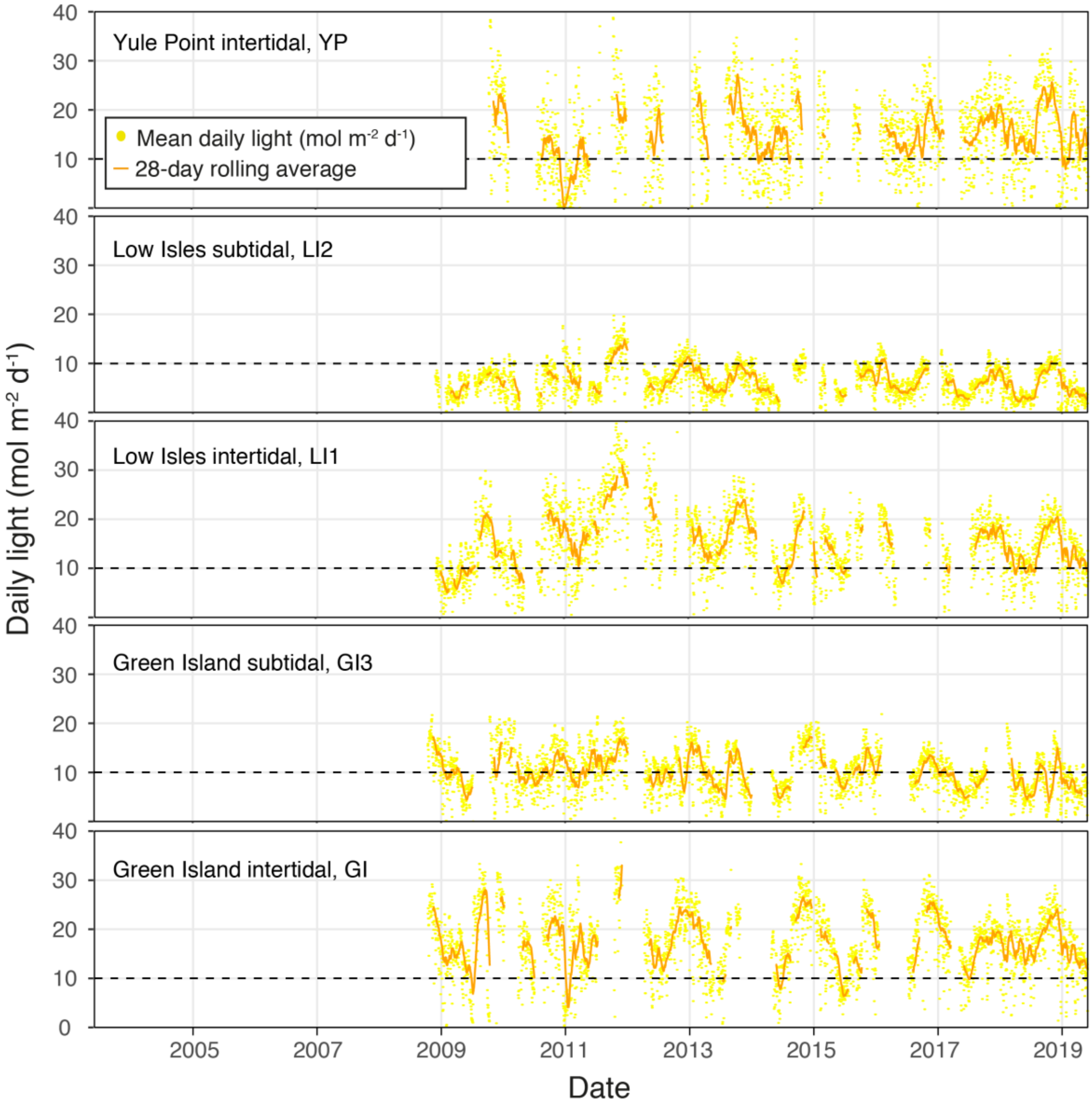

Figure 100. *Daily light (yellow line) and 28-day rolling average (orange, bold line) for locations in the northern Wet Tropics.*

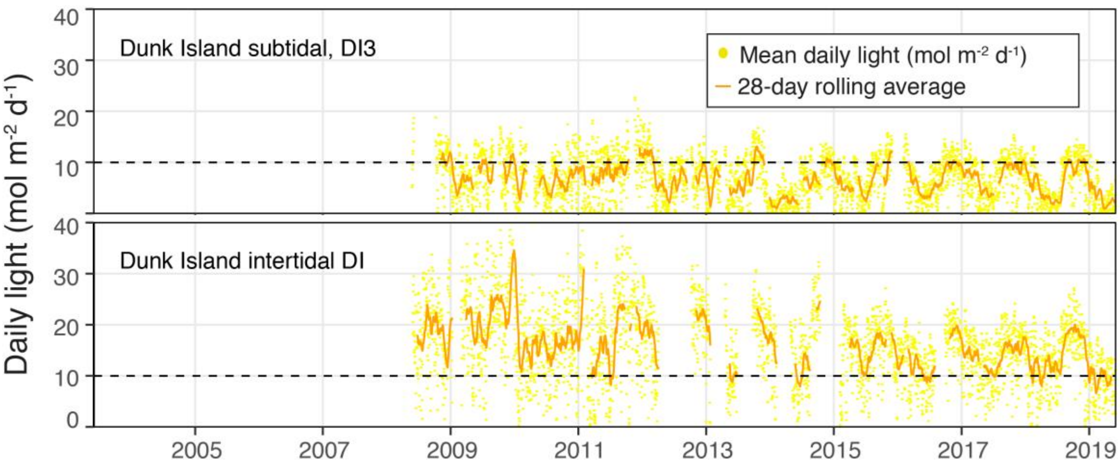

Figure 101. *Daily light (yellow line) and 28-day rolling average (orange, bold line) for locations in the southern Wet Tropics.*

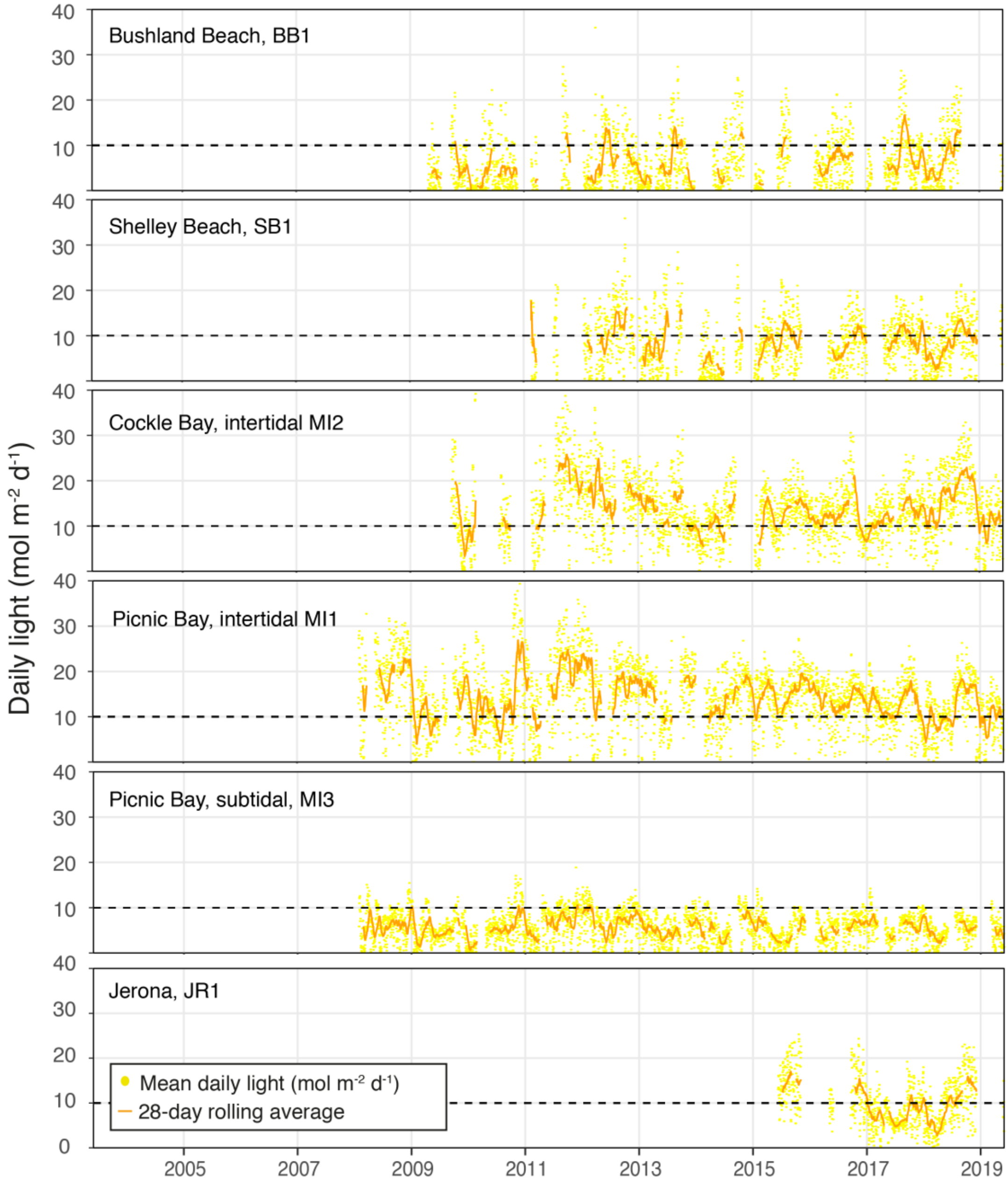

Figure 102. *Daily light (yellow line) and 28-day rolling average (orange, bold line) at locations in the Burdekin region.*

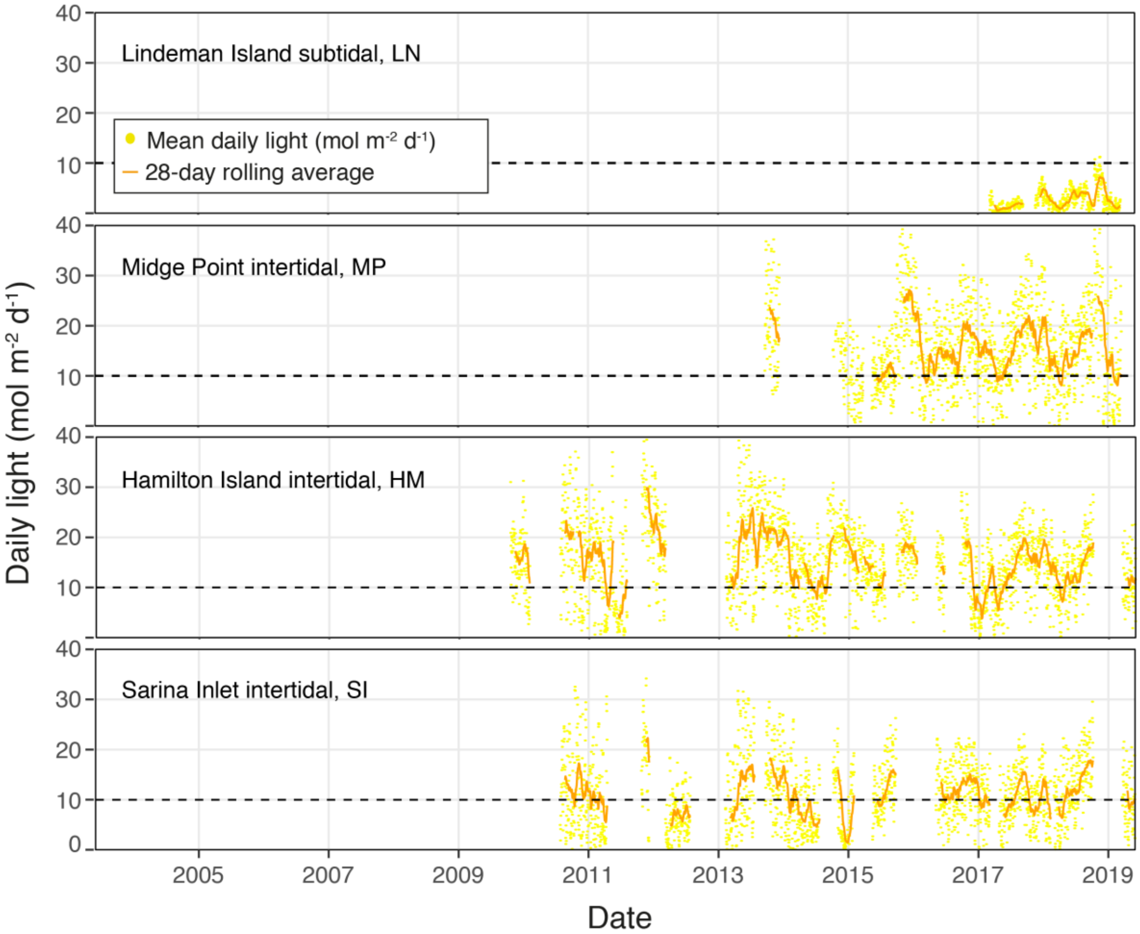

Figure 103. *Daily light (yellow line) and 28-day rolling average (orange, bold line) at Mackay–Whitsunday habitats.*

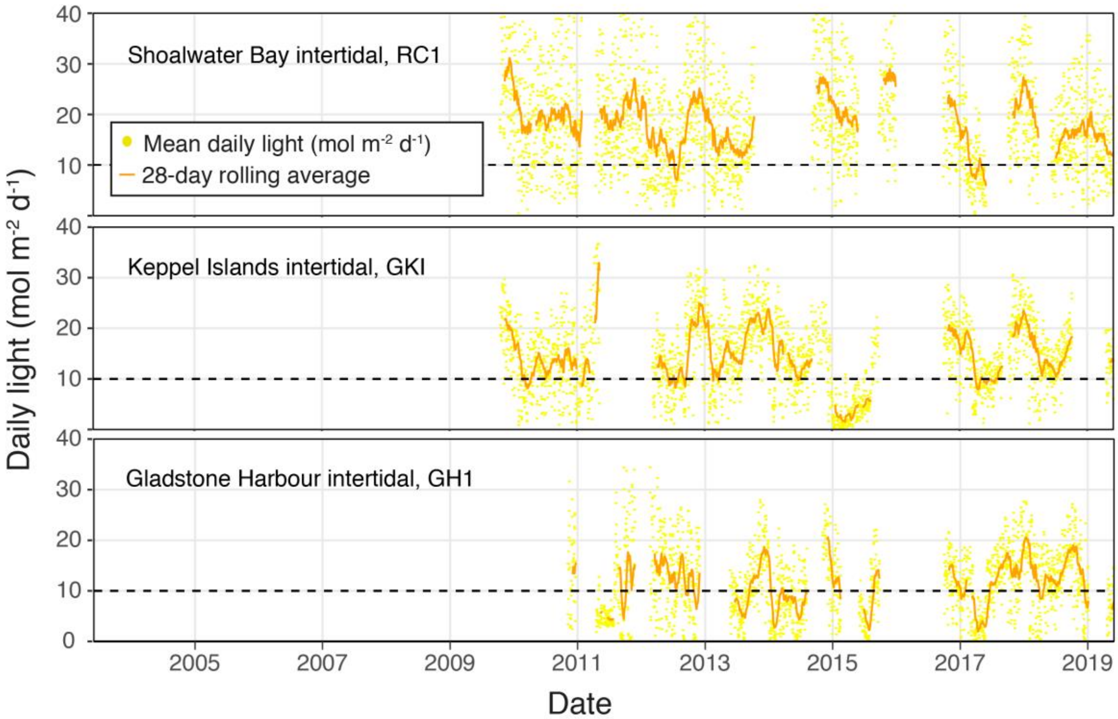

Figure 104. *Daily light (yellow line) and 28-day rolling average (orange, bold line) at monitoring locations in the Fitzroy NRM region.*

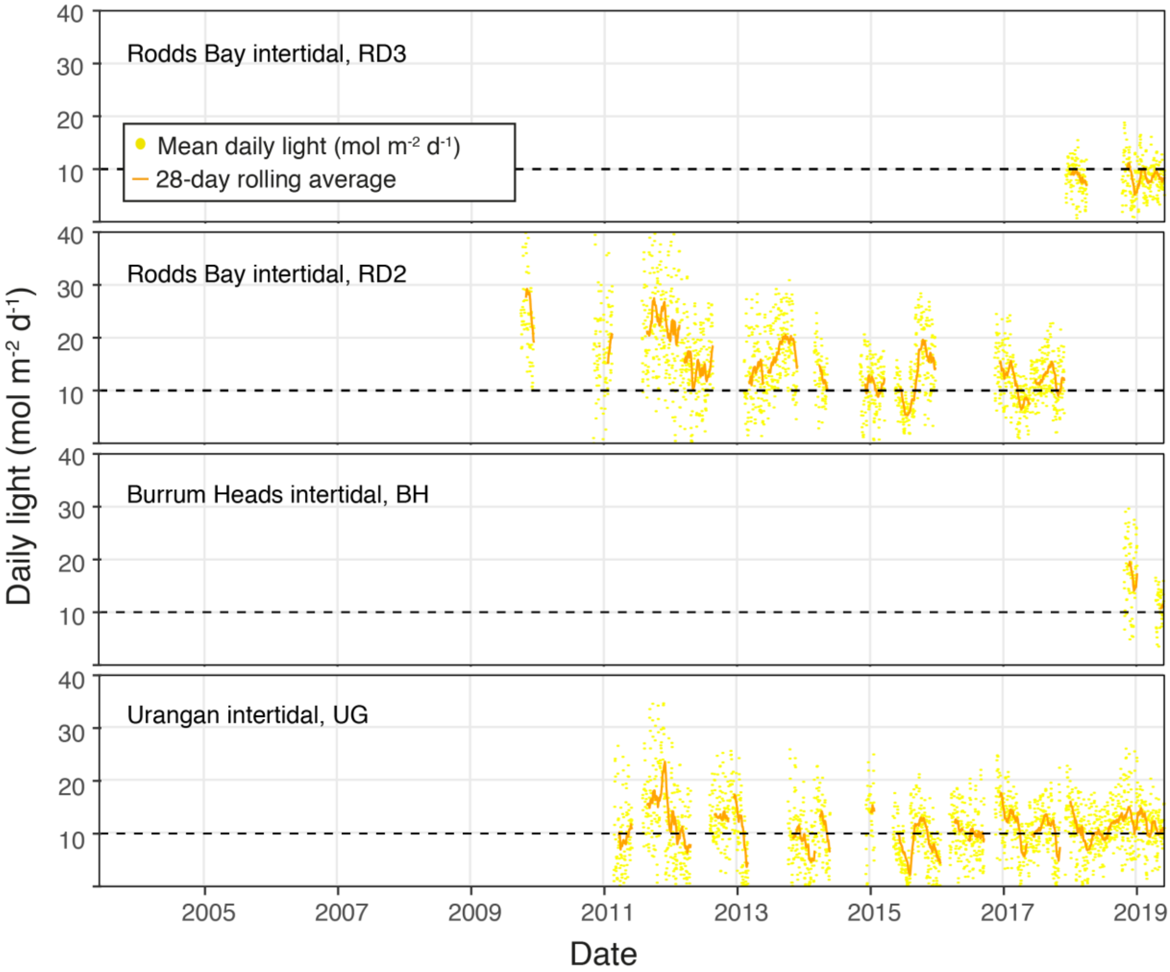

Figure 105. *Daily light (yellow line) and 28-day rolling average (orange, bold line) at monitoring locations in the Burnett–Mary NRM region.*

A3.2 Seagrass habitat condition: Sediments composition

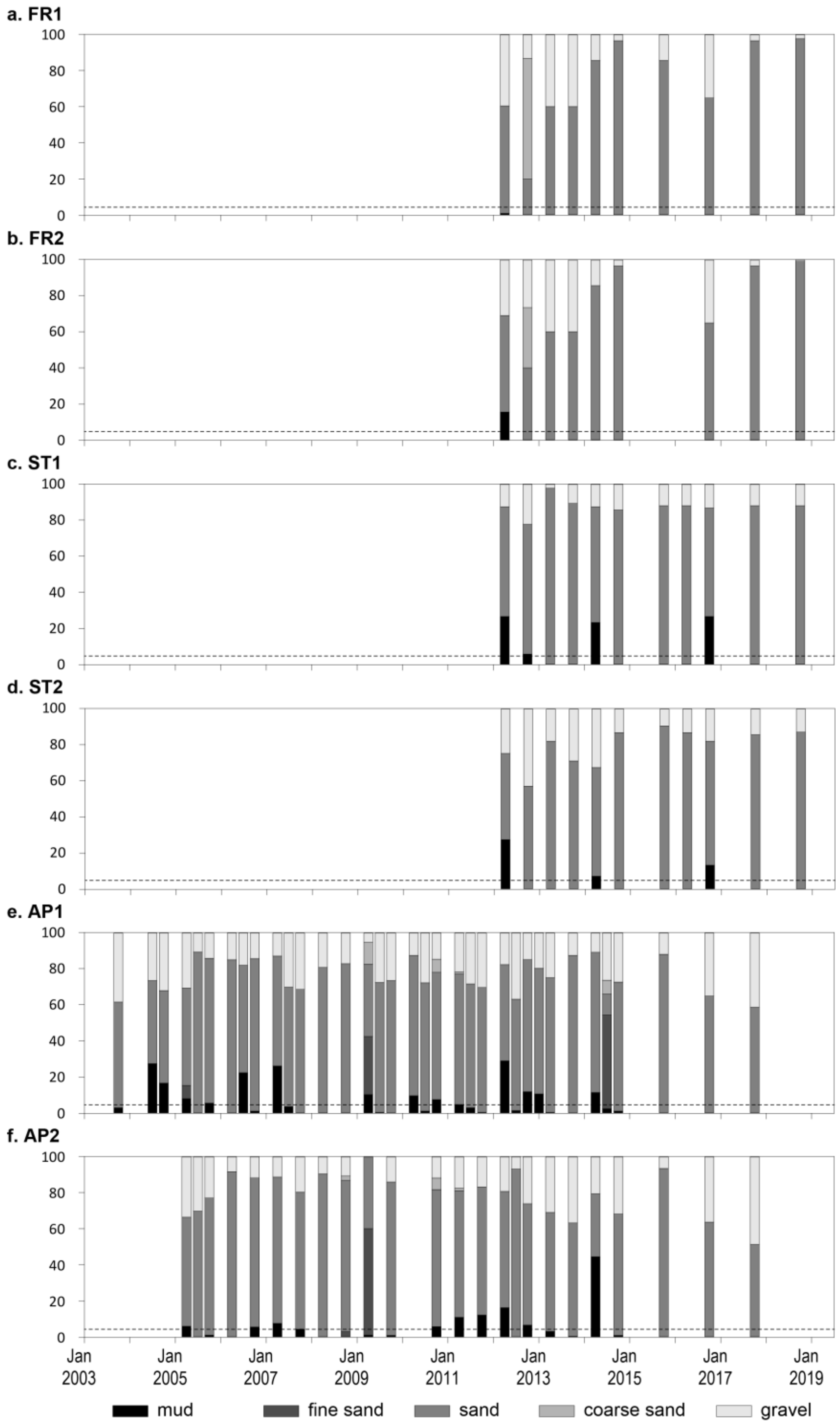

Figure 106. *Sediment grain size composition at reef habitat monitoring sites in the Cape York region, 2003–2019. Dashed line is the Reef long-term average proportion of mud.*

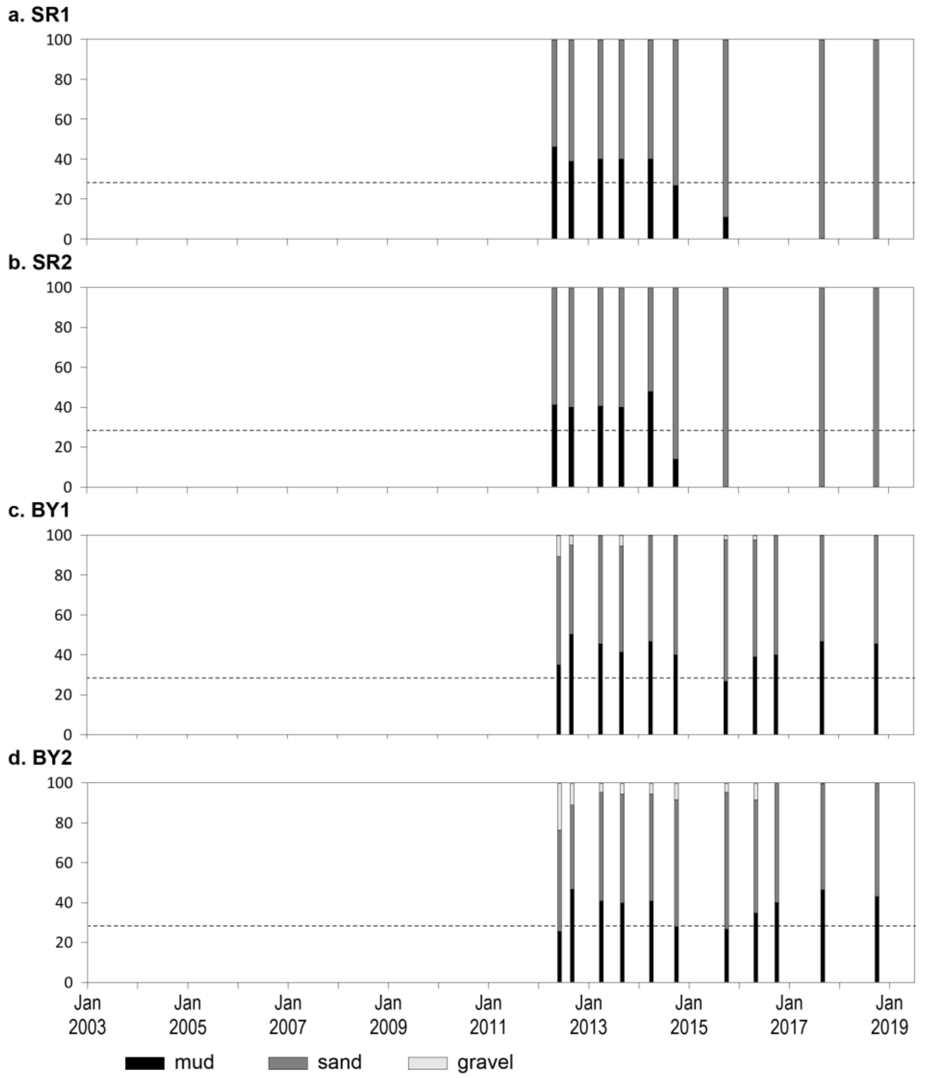

Figure 107. *Sediment grain size composition at coastal habitat monitoring sites in the Cape York region, 2010–-2019. Dashed line is the Reef long-term average proportion of mud.*

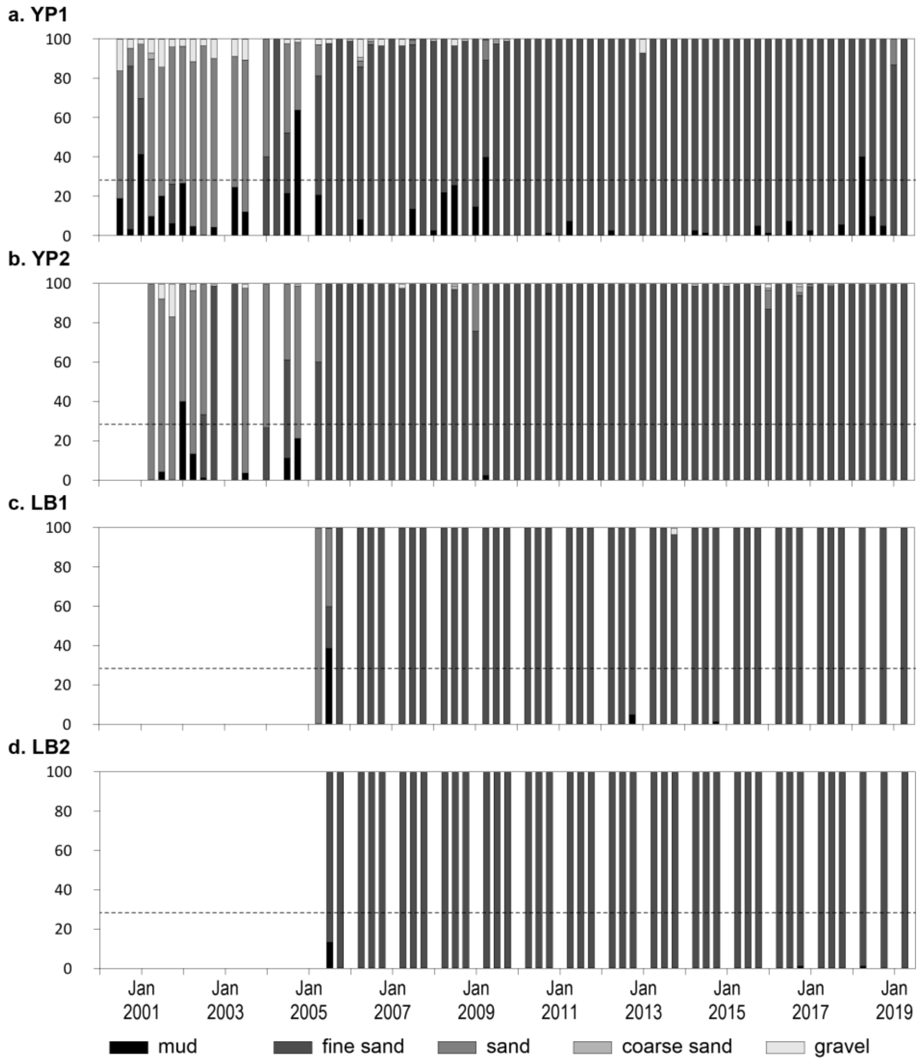

Figure 108. *Sediment grain size composition at intertidal coastal habitat monitoring sites in the Wet Tropics region, 2001–2019. Dashed line is the Reef long-term average proportion of mud.*

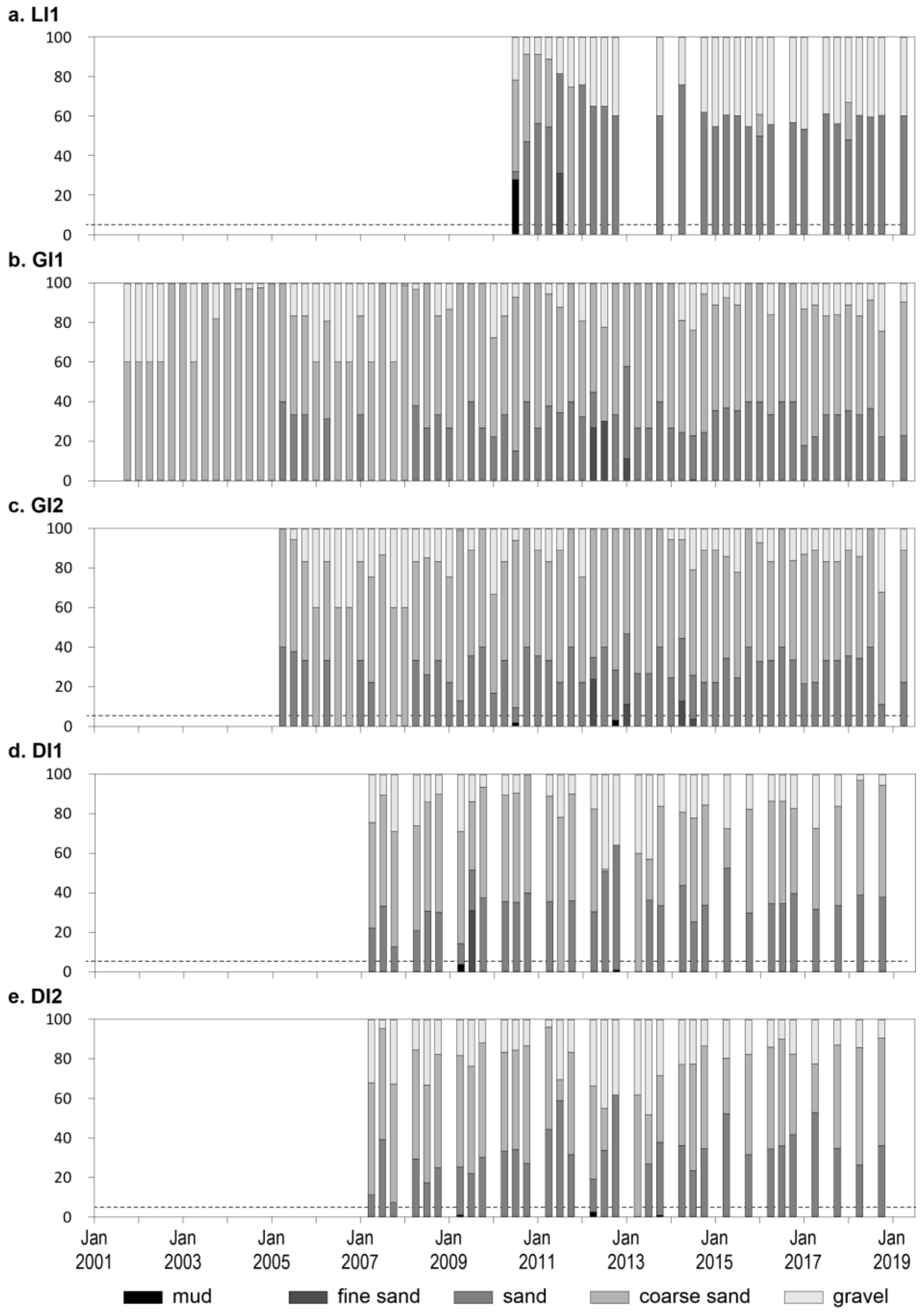

Figure 109. *Sediment grain size composition at intertidal reef habitat monitoring sites in the Wet Tropics region, 2001–2019. Dashed line is the Reef long-term average proportion of mud.*

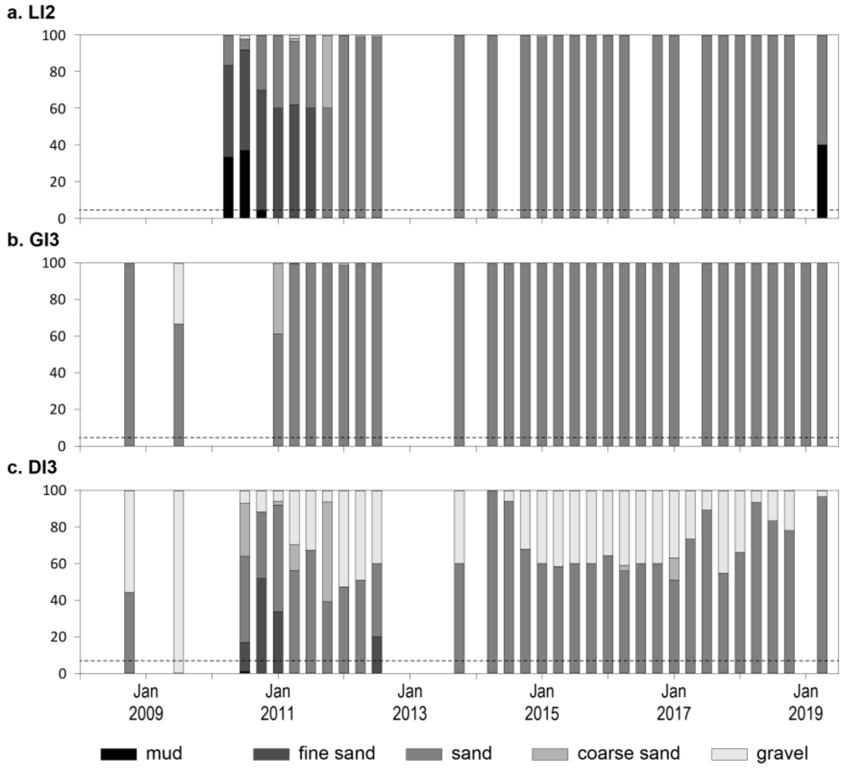

Figure 110. *Sediment grain size composition at subtidal reef habitat monitoring sites in the Wet Tropics region, 2008–2019. Dashed line is the Reef long-term average proportion of mud.*

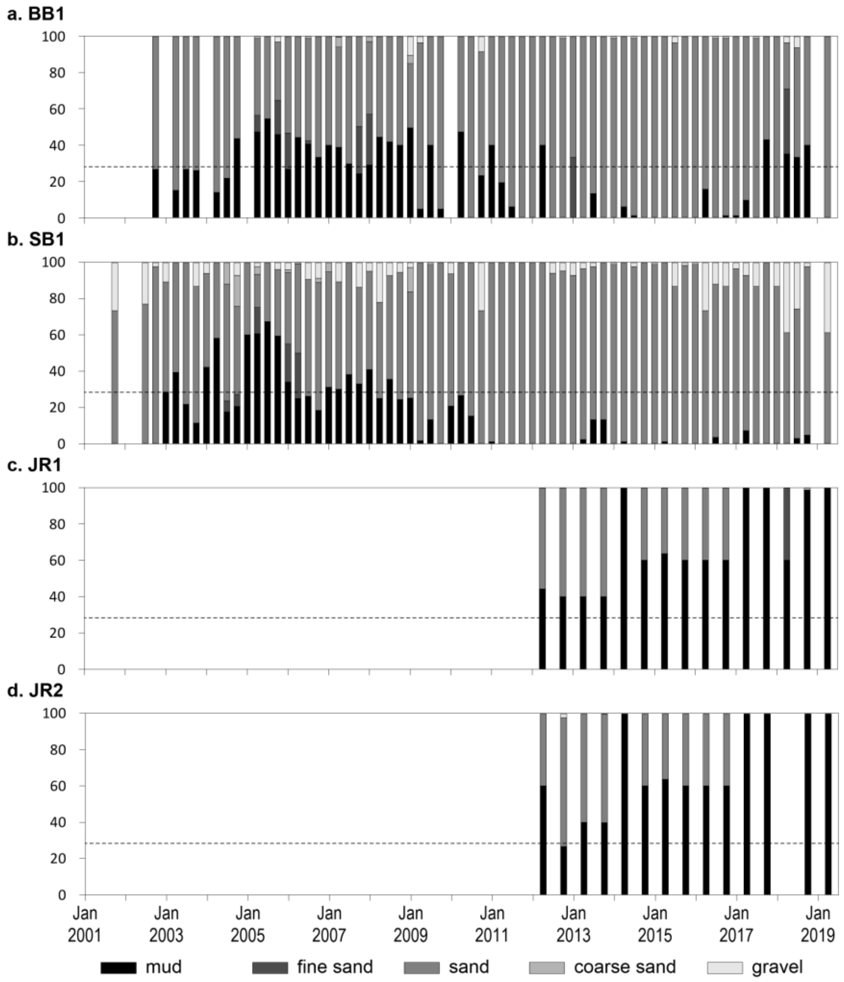

Figure 111. *Sediment grain size composition at intertidal coastal habitat monitoring sites in the Burdekin region, 2001–2019. Dashed line is the Reef long-term average proportion of mud.*

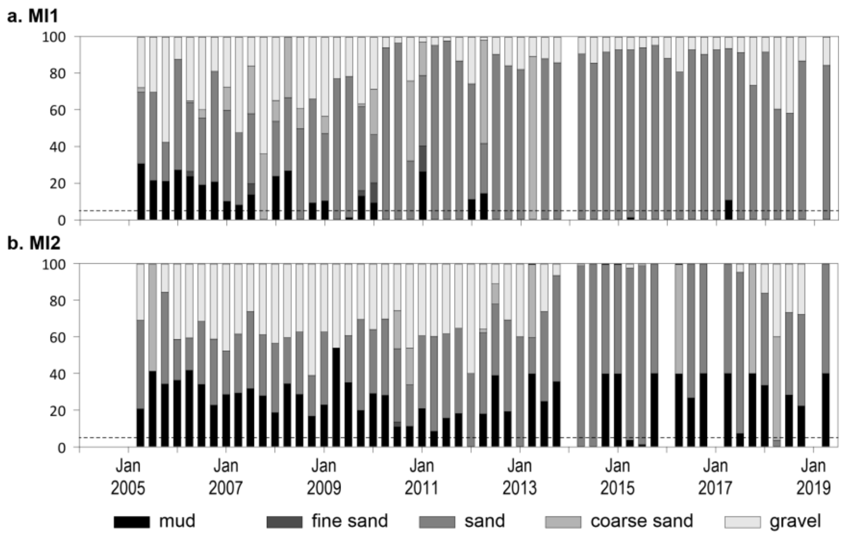

Figure 112. *Sediment grain size composition at intertidal reef habitat monitoring sites in the Burdekin region, 2004–2019. Dashed line is the Reef long-term average proportion of mud.*

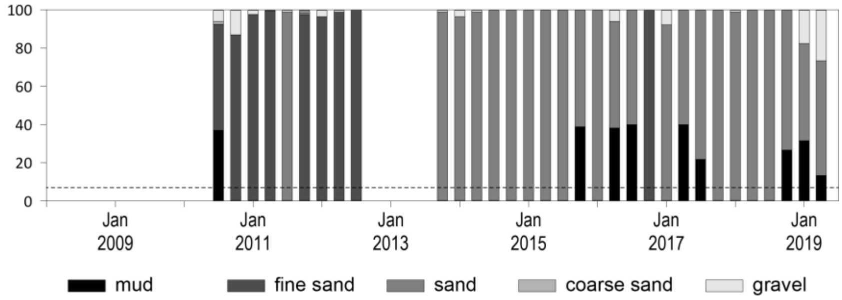

Figure 113. *Sediment grain size composition at subtidal reef habitat monitoring sites in the Burdekin region, 2010–2019. Dashed line is the Reef long-term average proportion of mud.*

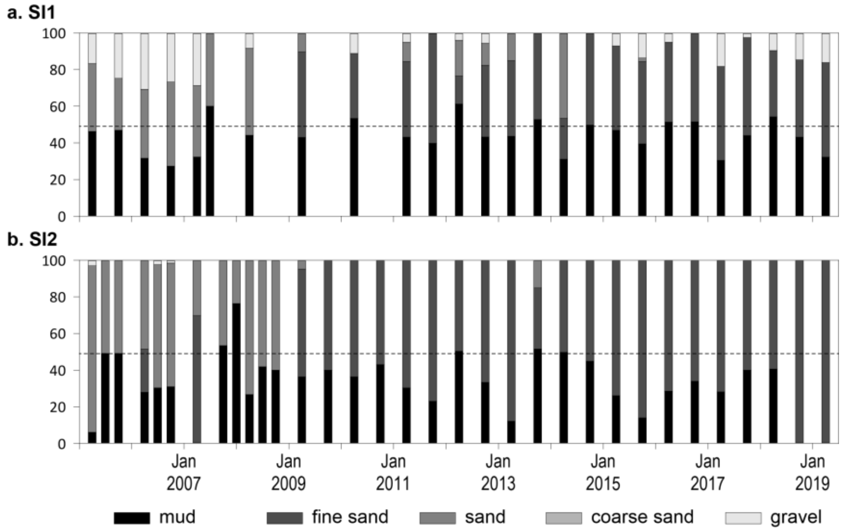

Figure 114. *Sediment grain size composition at intertidal estuary habitat monitoring sites in the Mackay–Whitsunday region, 2005–2019. Dashed line is the Reef long-term average proportion of mud.*

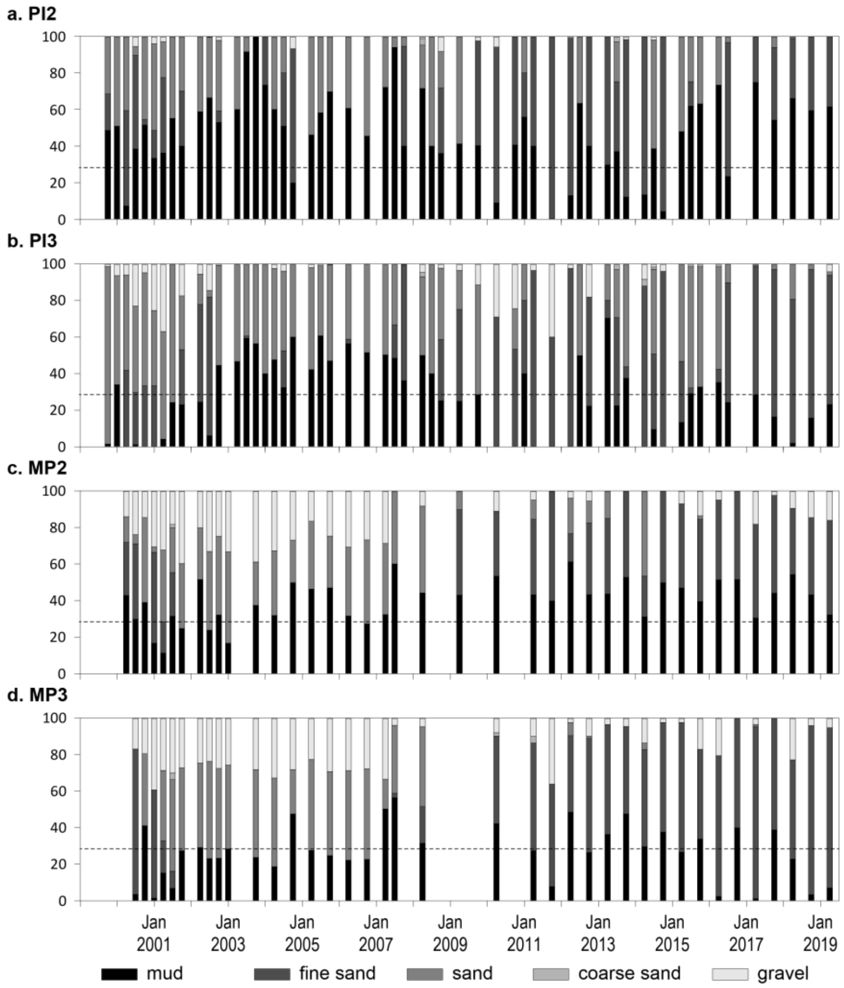

Figure 115. *Sediment grain size composition at intertidal coastal habitat monitoring sites in the Mackay–Whitsunday region, 1999–2019. Dashed line is the Reef long-term average proportion of mud.*

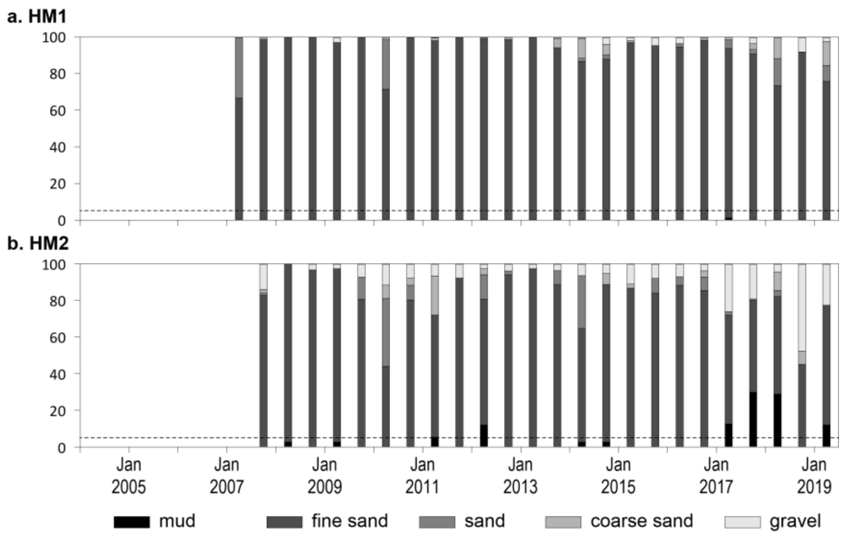

Figure 116. *Sediment grain size composition at intertidal reef habitat monitoring sites in the Mackay–Whitsunday region, 2007–2019. Dashed line is the Reef long-term average proportion of mud.*

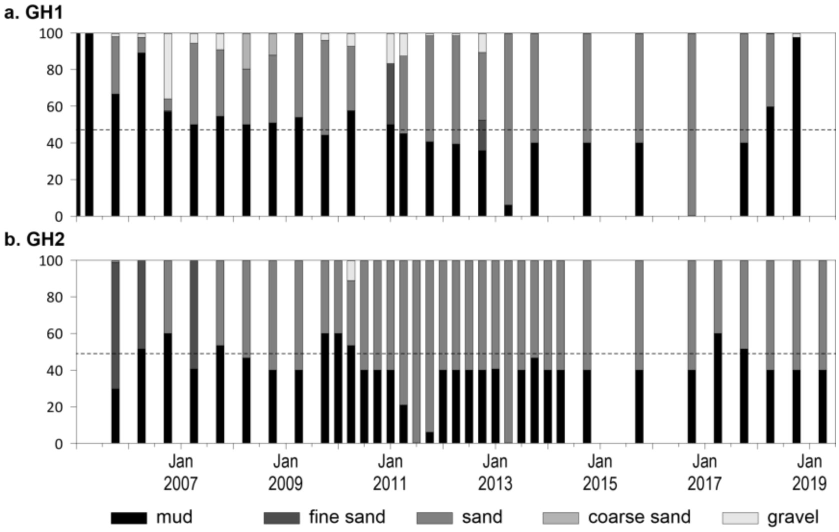

Figure 117. *Sediment grain size composition at intertidal estuary habitat monitoring sites in the Fitzroy region, 2005–2019. Dashed line is the Reef long-term average proportion of mud.*

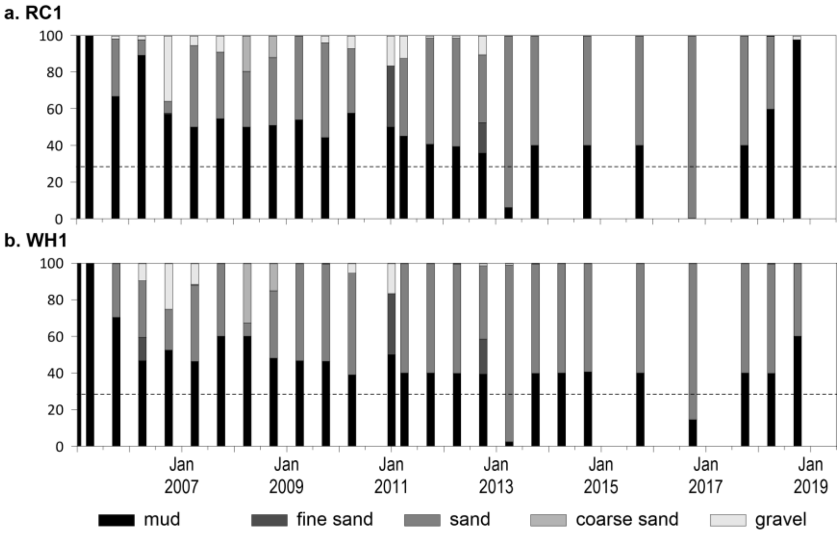

Figure 118. *Sediment grain size composition at intertidal coastal habitat monitoring sites in the Fitzroy region, 2005–2019. Dashed line is the Reef long-term average proportion of mud.*

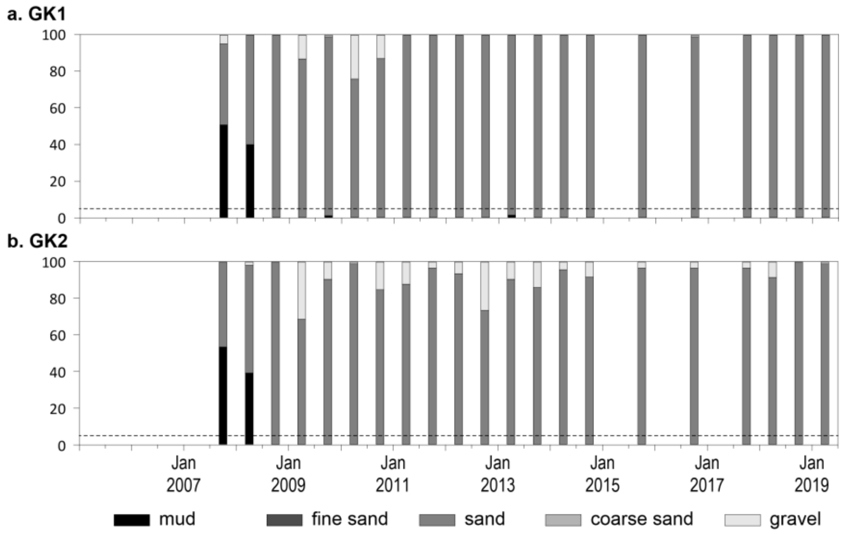

Figure 119. *Sediment grain size composition at intertidal reef habitat monitoring sites in the Fitzroy region, 2007–2019. Dashed line is the Reef long-term average proportion of mud.*

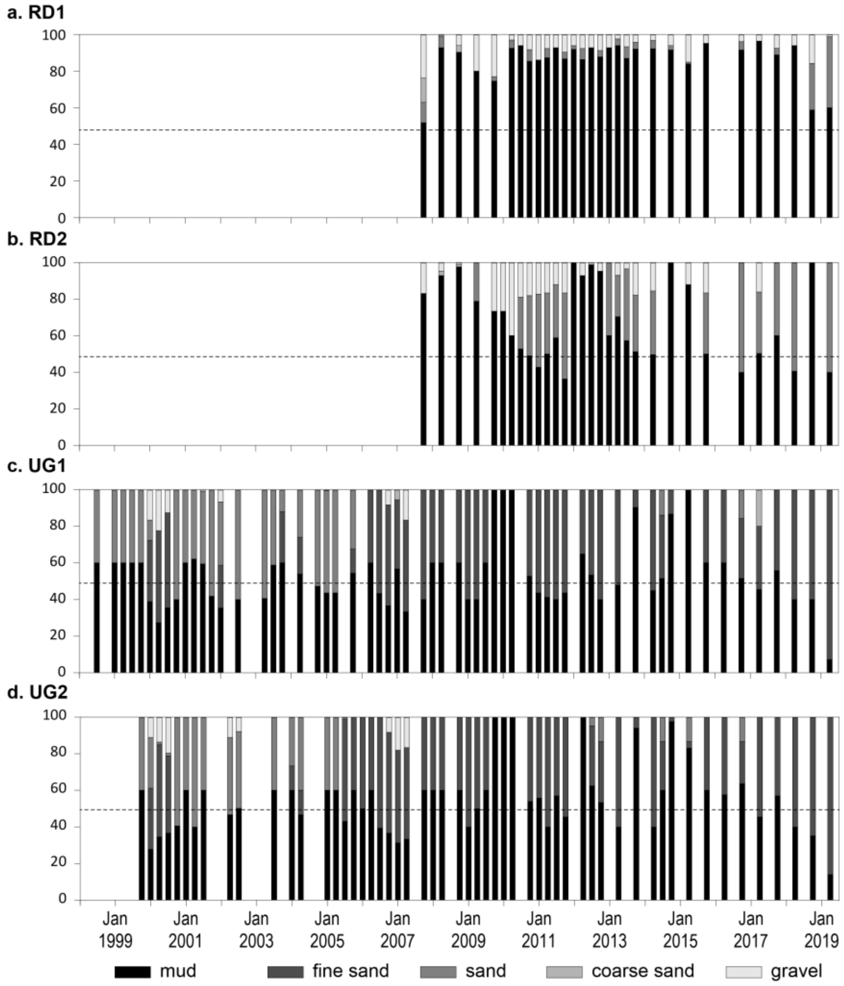

Figure 120. *Sediment grain size composition at intertidal estuary habitat monitoring sites in the Burnett–Mary region, 1999–2019. Dashed line is the Reef long-term average proportion of mud.*

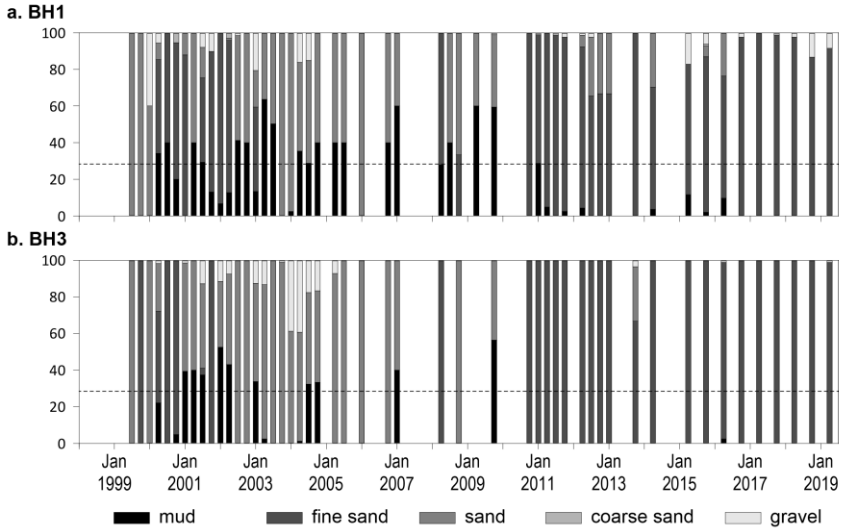

Figure 121. *Sediment grain size composition at intertidal coastal habitat monitoring sites in the Burnett–Mary region, 1999–2019. Dashed line is the Reef long-term average proportion of mud.*

Appendix 4 Results of statistical analysis

Table 20. *Results of Mann-Kendall analysis to assess if there was a significant trend (decline or increase) over time in seagrass abundance (per cent cover).The reported output of the tests performed are Kendall's tau coefficient (Kendall-τ), the two-sided p-value (significant at α = 0.05 in bold), the Sen's slope (showing the sign and strength of the trend – including confidence intervals if significant) and the long-term trend.*

NRM region	Habitat	Site	First Year	Last Year	*n*	*Kendall-τ*	*p* (2-sided)	Sen's slope (confidence interval)	trend
Cape York	coastal intertidal	BY1	2012	2018	11	0.273	0.2757	0.793	no trend
		BY2	2012	2018	11	0.445	0.062	0.898	no trend
		SR1	2012	2018	9	-0.333	0.251	-0.697	no trend
		SR2	2012	2018	9	0.222	0.466	0.261	no trend
	coastal subtidal	LR1	2015	2017	3	-0.333	1.0000	-3.376	no trend
		LR2	2015	2017	3	-1.000	0.2963	-16.635	no trend
	reef intertidal	**AP1**	2003	2017	35	**-0.459**	**0.0001**	**-0.533** (-0.763 to -0.283)	**decrease**
		AP2	2005	2017	24	-0.022	0.9013	-0.030	no trend
		FR1	2012	2018	10	-0.225	0.419	-0.212	no trend
		FR2	2012	2018	9	-0.444	0.118	-1.621	no trend
		ST1	2012	2018	11	**0.600**	**0.013**	**0.666 (**0.121 to 1.137)	**Increase**
		ST2	2012	2018	11	**0.697**	**0.004**	**0.709** (0.465 to 1.065)	**increase**
		YY1	2012	2014	3	0.333	1.0000	1.045	no trend
	Reef subtidal	FG1	2016	2018	3	1	0.296	13.020	no trend
		FG2	2016	2018	3	1	0.296	11.260	no trend
	pooled		2003	2018	37	**-0.351**	**0.002**	**-0.289** (-0.428 to -0.083)	**decrease**
Wet Tropics	coastal intertidal	**LB1**	2005	2019	42	**-0.541**	**<0.001**	**-0.040** (-0.118 to -0.005)	**decrease**
		LB2	2005	2019	41	**-0.395**	**0.001**	**-0.042** (-0.098 to 0)	**decrease**
		YP1	2000	2019	73	0.119	0.139	0.090	no trend
		YP2	2001	2019	69	0.083	0.315	0.042	no trend
	coastal subtidal	MS1	2015	2019	3	0.333	1	6.222	no trend
		MS2	2015	2019	3	0.333	1	1.889	no trend
	reef intertidal	DI1	2007	2018	33	-0.182	0.141	-0.130	no trend
		DI2	2007	2018	33	-0.150	0.227	-0.128	no trend

NRM region	**Habitat**	**Site**	**First Year**	**Last Year**	***n***	***Kendall-τ***	***p*** (2-sided)	**Sen's slope** (confidence interval)	**trend**
		GI1	2001	2019	70	-0.094	0.250	-0.058	no trend
		GI2	2005	2019	56	-0.015	0.876	-0.016	no trend
		GO1	2008	2016	7	-0.429	0.2296	-1.682	no trend
		LI1	2008	2019	38	**-0.431**	**<0.001**	**-0.164** (-0.269 to -0.077)	**decrease**
		DI3	2008	2019	42	-0.171	0.115	-0.019	no trend
	reef subtidal	**GI3**	2008	2019	41	**-0.398**	**<0.001**	**-0.589** (-0.846 to -0.332)	**decrease**
		LI2	2008	2019	38	0.103	0.372	0.062	no trend
	pooled		2000	2019	80	-0.141	0.0072	-0.082	no trend
		BB1	2002	2019	62	0.052	0.552	0.056	no trend
		SB1	2001	2019	68	-0.039	0.641	-0.030	no trend
	coastal intertidal	SB2	2001	2019	67	**-0.189**	**0.024**	**-0.176** (-0.342 to -0.022)	**decrease**
		JR1	2012	2019	15	0.276	0.166	1.939	no trend
Burdekin		**JR2**	2012	2019	14	**0.538**	**0.009**	**3.006** (1.288 to 5.233)	**increase**
	reef intertidal	MI1	2005	2019	55	-0.110	0.240	-0.152	no trend
		MI2	2005	2019	53	-0.177	0.062	-0.328	no trend
	reef subtidal	MI3	2008	2019	45	0.123	0.237	0.267	no trend
	pooled		2001	2019	74	0.000	1	0.000	no trend
	estuarine intertidal	SI1	2005	2019	33	-0.250	0.042	-0.285	no trend
		SI2	2005	2019	28	-0.037	0.797	-0.037	no trend
		MP2	2000	2019	40	0.194	0.081	0.166	no trend
		MP3	2000	2019	38	0.021	0.860	0.015	no trend
		PI2	1999	2019	56	**-0.331**	**<0.001**	**-0.302** (-0.485 to -0.153)	**decrease**
Mackay–Whitsunday	coastal intertidal	PI3	1999	2019	56	-0.173	0.060	-0.135	no trend
		CV1	2017	2018	4	0.667	0.308	1.934	no trend
		CV2	2017	2018	4	0	1	0.360	no trend
	coastal subtidal	NB1	2015	2018	4	-0.333	0.734	-7.082	no trend
		NB2	2015	2018	4	0.667	0.308	3.887	no trend

NRM region	Habitat	Site	First Year	Last Year	*n*	*Kendall-τ*	*p* (2-sided)	Sen's slope (confidence interval)	trend
	reef intertidal	**HB1**	2000	2019	42	**-0.329**	**0.002**	**-0.212** (-0.333 to -0.086)	**decrease**
		HB2	2000	2019	41	-0.066	0.552	-0.044	no trend
		HM1	2007	2019	25	**-0.540**	**<0.001**	**-0.266** (-0.424 to -0.140)	**decrease**
		HM2	2007	2019	24	**-0.339**	**0.022**	**-0.133** (-0.323 to -0.023)	**decrease**
	Reef subtidal	TO1	2015	2018	4	-0.667	0.308	-5.524	no trend
		TO2	2015	2018	4	-0.667	0.308	-2.969	no trend
		LN1	2017	2019	4	0.333	0.734	1.221	no trend
		LN2	2017	2019	3	0.333	1	0.276	no trend
	pooled		1999	2018	64	**-0.409**	**<0.001**	**-0.190** (-0.261 to -0.121)	**decrease**
Fitzroy	estuarine intertidal	**GH1**	2005	2019	35	**-0.388**	**0.001**	**-0.733** (-1.127 to -0.267)	**decrease**
		GH2	2005	2019	35	-0.008	0.955	0.016	no trend
	coastal intertidal	RC1	2002	2018	35	-0.013	0.921	-0.022	no trend
		WH1	2002	2018	36	0.006	0.967	0.003	no trend
	reef intertidal	**GK1**	2007	2019	21	**-0.377**	**0.018**	**-0.108** (-0.219 to -0.031)	**decrease**
		GK2	2007	2019	21	-0.038	0.833	-0.009	no trend
	pooled		2002	2018	47	**-0.300**	**0.003**	**-0.183** (-0.306 to -0.078)	**decrease**
Burnett–Mary	estuarine intertidal	RD1	2007	2019	30	0.049	0.721	0.002	no trend
		RD2	2007	2017	28	**-0.409**	**0.003**	**-0.009** (-0.096 to -0.001)	**decrease**
		RD3	2017	2019	4	-0.333	0.734	-0.936	no trend
		UG1	1998	2019	61	0.153	0.086	0.013	no trend
		UG2	1999	2019	57	**0.283**	**0.002**	**0.088** (0.015 to 0.265)	**increase**
	coastal intertidal	BH1	1999	2019	52	0.078	0.416	0.038	no trend
		BH3	1999	2019	50	**0.381**	**<0.001**	**0.176** (0.099 to 0.253)	**increase**
	pooled		1998	2019	80	0.033	0.683	0.011	no trend