Australian Government

Great Barrier Reef
Marine Park Authority

Great Barrier Reef
MARINE MONITORING PROGRAM

Annual Report
INSHORE SEAGRASS MONITORING

2020-21

JAMES COOK UNIVERSITY AUSTRALIA

TropWATER
Centre for Tropical Water and Aquatic Ecosystem Research

Published by the Great Barrier Reef Marine Park Authority

ISSN: 2208-4037

A catalogue record for this publication is available from the National Library of Australia

This publication should be cited as:

McKenzie, L.J., Collier, C.J, Langlois, L.A., Yoshida, R.L. and Waycott, M. 2022, *Marine Monitoring Program: Annual Report for Inshore Seagrass Monitoring 2020–21. Report for the Great Barrier Reef Marine Park Authority*, Great Barrier Reef Marine Park Authority, Townsville, 177pp.

Front cover image: Over-under image showing shallow water seagrass meadow (mainly *Cymodocea* with *Thalassia* and *Syringodium*), taken on the reef flat at Green Island © Len McKenzie.

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.

Comments and questions regarding this document are welcome and should be addressed to:

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This project is supported by the Great Barrier Reef Marine Park Authority through funding from the Great Barrier Reef Marine Monitoring Program and James Cook University.

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Acronyms, abbreviations and units

Authority	Great Barrier Reef Marine Park Authority
BoM	Bureau of Meteorology
CV	coefficient of variation
DES	Department of Environment and Science, Queensland
GAM	generalised additive model
JCU	James Cook University
km	kilometre
m	metre
MMP	Great Barrier Reef Marine Monitoring Program
NRM	Natural Resource Management
Paddock to Reef program	Paddock to Reef Integrated Monitoring, Modelling and Reporting Program
PAR	Photosynthetically available radiation
QA/QC	quality assurance/quality control
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
RJFMP	Reef Joint Field Management Program
SE	Standard Error
SW	Seagrass-Watch
The Reef	Great Barrier Reef
TropWATER	Centre for Tropical Water & Aquatic Ecosystem Research

Acknowledgements

We thank Abby Fatland, Jasmina Uusitalo, and Naomi Smith, who assisted with field monitoring. We thank Sascha Taylor and the QPWS rangers who conducted the subtidal drop camera field assessments in Cape York, southern Wet Tropics and Mackay–Whitsunday. We also thank Abby Fatland for assisting with the processing of laboratory samples and the many Seagrass-Watch volunteers who assisted and shared their data with us from Shelley Beach, Pioneer Bay, Hydeaway Bay, St Helens Beach and Clairview. We thank the water quality team at TropWATER (Moran *et al.* 2022) for climate data including rainfall, river discharge and turbid water exposure maps included in this report. We also thank the Traditional Owners of the Sea Countries we visited to conduct our monitoring. In particular, we would like to thank the Wuthathi, Kuku Yau, Yuku Baja Muliku and Girringun groups for assisting in the field.

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 Mission Beach.

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.

We acknowledge the Australian Government funding through the Great Barrier Reef Marine Park Authority (the Authority) for financial and technical support of this program. We thank Bronwyn Houlden and Carol Honchin from the Authority for their overall project management and program guidance.

We sincerely thank our reviewers, including several anonymous reviewers, for the time they spent on the careful reading of our report and their many insightful comments and suggestions which improved earlier versions of this and previous reports.

Executive summary

This document reports on the long-term health of inshore seagrass meadows in the Great Barrier Reef (the Reef). Results are presented in the context of the pressures faced by the ecosystem. Long-term health of inshore seagrass meadows is measured through seagrass abundance and resilience, which are summarised as the seagrass condition index, and supported by information on the proportion of colonising species, reproductive status, meadow extent, epiphytes on seagrass leaves and macroalgal presence.

Trends in key inshore seagrass indicators

Inshore seagrass meadows across the Reef improved slightly in overall condition in 2020–21, with an uptick in the condition grade to **moderate** (Figure 1). The three northern most regions (Cape York, Wet Tropics and Burdekin), all had an overall seagrass condition grade that improved and was moderate. In contrast, the three southern most regions (Mackay–Whitsunday, Fitzroy and Burnett–Mary) had an overall seagrass condition grade that declined to poor.

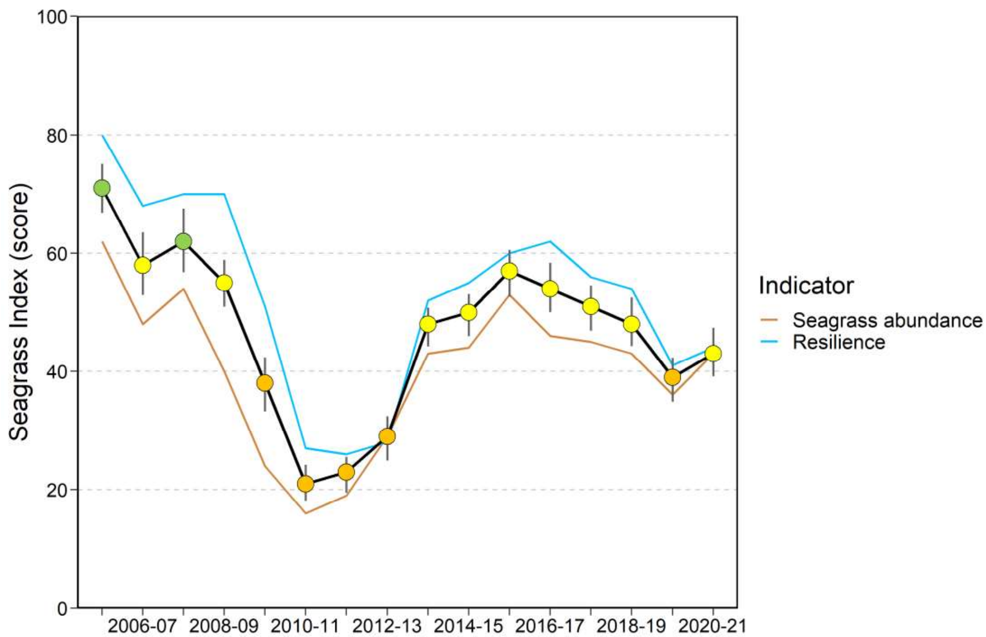


Figure 1. Overall inshore Reef seagrass condition index (±SE) with contributing indicator scores over the life of the MMP. The index is derived from the aggregate of metric scores for indicators of seagrass condition: abundance and resilience. Index scores scaled from 0–100 and graded: ● = very good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

Reef-wide inshore seagrass abundance had been increasing on average since 2010–11, but declined in the previous four reporting years until improving in 2020–21. Abundances at two thirds of the 69 monitoring sites either improved or remained stable in 2020–21. The post 2015–16 decline was driven mostly by losses in the Mackay–Whitsunday and Burdekin regions, with smaller declines simultaneously occurring in Cape York and the Wet Tropics. These losses in the northern most regions have abated, with the greatest improvement in the Burdekin as it recovered from the effects of heavy rainfall and above-average discharge from rivers in early 2019. However, there were declining abundances in all three southern regions during 2020–21, negating improvements from the previous period.

Resilience improved slightly in 2020–21 and was moderate for the inshore Reef overall. Resilience includes both resistance and recovery potential following disturbance events. At the majority of sites where resilience is measured (32 out of 48), seagrass condition indicated adequate levels of resistance (based on abundance threshold and composition).

Less than a quarter of those sites had reproductive structures in 2020–21, but a further 11 sites had a recent history (<3 years) of reproduction. Because of these features, nearly half of these sites had a resilience score of 50 or more. The remaining half of the sites had no reproductive history or were in poor condition and/or had a resilience score of 50 or less.

There are further signs of recovery based on additional indicators, including:

- decreasing or stable proportion of colonising species,which are the first to establish after a disturbance. The decreasinf trend indicates recovery towards species that are foundational to the meadows.
- increasing or stable meadow extent at almost three quarters of sites, culminating in the greatest meadow extents in the last three years. However,seagrass within estuarine habitats in the Burnett–Mary region, reef habitat in the Fitzroy region and subtidal reef habitat in the Mackay–Whitsunday regions remain vulnerable to large disturbances because meadow seascapes remain highly fragmented.
- increasing seed banks at coastal habitats, but decrease or absence of seed banks across other habitats.
- decreasing and low epiphyte cover on seagrass leaves across all habitats, accompanied by continued low macroalgae abundance.

Influencing pressures

Pressures affecting inshore Reef seagrass habitats were low to negligible, but variable among regions and habitats in 2020–21. There was limited cyclone impact on the Reef, and rainfall and river discharge were close to the long-term median. Inshore seagrass sites were exposed to primary and secondary waters during many weeks of the wet season (December–April), but at a lower frequency than in recent years.

Benthic light availability was around the long-term average for inshore Reef seagrass meadows but lower than the long-term average (by more than 0.5 mol m^{-2} d^{-1}) at 12 of the 27 monitoring locations in all regions except the Fitzroy, and around or higher than the long-term average at the remainder of locations.

Within–canopy water temperature of inshore Reef seagrass meadows was around the long-term average, and excessive temperatures (>38°C) were the lowest in six consecutive years.

To summarise by region for this reporting year, wet season rainfall and discharge were above average in the three northern regions, yet benthic light and temperature were moderate and around the long-term average. In these regions, seagrass condition improved. Wet season rainfall and river discharge were well below average in the three southern regions, while temperature and benthic light were also around average for the regions. Despite this, seagrass condition declined in the three southern regions. This is likely attributed to a legacy of recent (3–4 years) extreme events (e.g. cyclone) or local processes such as sediment movement at some locations.

There is a history of cumulative pressures facing Reef inshore seagrass meadows since the MMP inception and in most years, some or all regions have been affected by cyclones, floods, thermal anomalies or periods of very low light availability. Particularly severe and widespread pressures occurred in the period from 2009–10 to 2011–12, when there was above-average river discharge and localised cyclone damage leading to the very poor seagrass condition index. Other regionally-significant impacts were caused by cyclone Debbie in 2016–17 affecting the Mackay–Whitsunday region, and floods in the Burdekin region in 2018–19. Legacy effects of these past pressures are evident in current seagrass condition and the ongoing need for recovery to reach a higher seagrass index.

Conclusions

Reef-wide inshore seagrass condition improved slightly in 2020–21, with the condition grade increasing to moderate. Inshore seagrass condition improved to a moderate grade in the northern Natural Resource Management (NRM) regions (Cape York, Wet Tropics and Burdekin), while condition deteriorated in the southern regions (Mackay–Whitsunday, Fitzroy and Burnett–Mary), with the grade declining to poor.

Of concern is the inshore seagrass condition in the southern regions, in particular Mackay-Whitsunday and Fitzroy. In these regions, seagrass abundance has decreased over the long-term, meadow extents remain low and highly fragmented, a considerable portion of meadows are dominated by colonising rather than foundational seagrass species, reproductive effort and seed banks are low, and overall resilience is poor. These declines in seagrass condition in the southern regions appear either a legacy of recent (3–4 years) extreme events (e.g. cyclone) or localised disturbances. Findings from the current monitoring period suggest seagrass ecosystems in the Mackay–Whitsunday and Fitzroy regions may be more vulnerable to adverse or severe disturbances in the near future.

Climate change is the most significant threat to the Reef's long-term outlook, and the 2021–22 wet season is expected to include intensifying pressures (rainfall, river discharge and tropical storms) as a consequence of a La Niña climatic phenomena. Securing a future for Reef seagrass ecosystems will require an increased need to maintain and build meadow resilience. Water quality improvements to catchment run-off are expected to provide some relief from these impacts and improve meadow condition and resilience, but further options for improving resilience need to be explored.

1 Introduction

Approximately 3,464 km^2 of inshore seagrass meadows have been mapped in the Great Barrier Reef World Heritage Area (the World Heritage Area) in waters shallower than 15 m (McKenzie *et al.* 2014b; Saunders *et al.* 2015; Carter *et al.* 2016; McKenzie *et al.* 2016; Howley, Unpublished data). The remaining modelled extent (90 per cent 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). These deepwater meadowsare 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 nearly 48 per cent of the total recorded area of seagrass in Australia and between 13 per cent and 22 per cent globally (McKenzie *et al.* 2020), 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 species of seagrass in the Reef (Waycott *et al.* 2007) and a high diversity of seagrass habitat and community types is provided by extensive bays, estuaries, rivers and the 2,300 km length of the Reef with its inshore lagoon and reef platforms. Seagrasses can be found on sand or muddy beaches, on reef platforms and in reef lagoons, and on sandy and muddy bottoms down to 70 m or more below Mean Sea Level (MSL) (Carter *et al.* 2021b).

Seagrasses in the Reef can be separated into four major habitat types: estuary/inlet, coastal, reef and deepwater (Carruthers *et al.* 2002). Environmental variables that influence seagrass species composition within these habitats include depth, tidal exposure, latitude, current speed, benthic light, proportion of mud, water type, water temperature, salinity, and wind speed (Carter *et al.* 2021a) (Figure 2). 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.

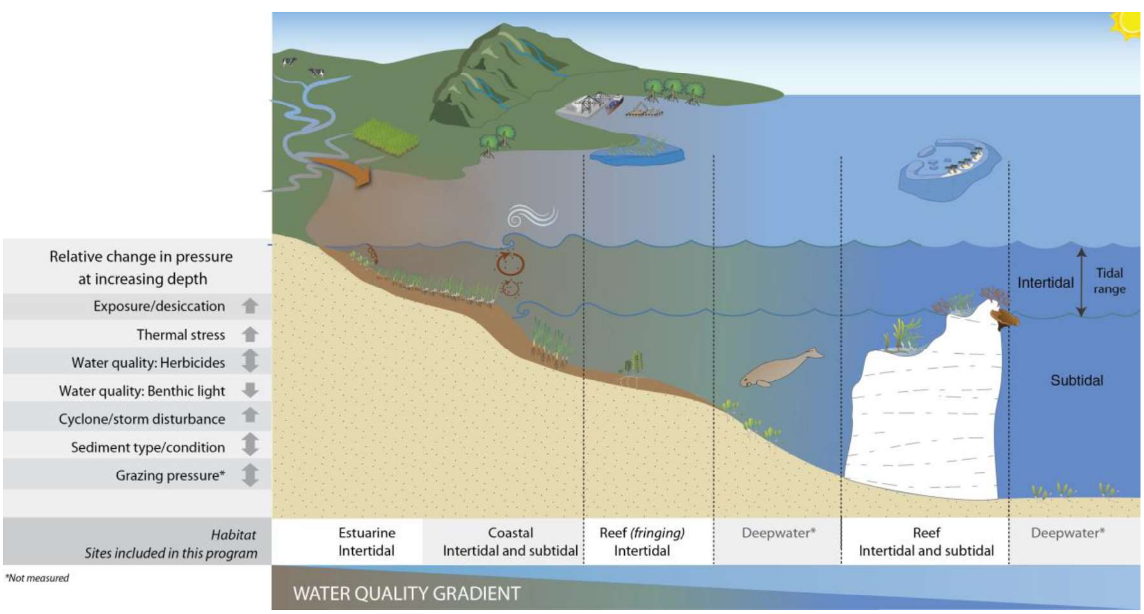


Figure 2. General conceptual model of seagrass habitats in north east Australia and the water quality impacts affecting the habitat (adapted from Carruthers *et al.* 2002, and Collier *et al.* 2014). Grey arrows indicate increase, decease or variable response with increasing depth.

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* spp), 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 areas in 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 strategic priority for the Great Barrier Reef Marine Park Authority (the Authority) is to sustain the Reef's outstanding universal value, build resilience and improve ecosystem health over each successive decade (GBRMPA 2014). Improving water quality is a key objective, because good water quality aids the resilience of coastal and inshore ecosystems of the Reef (GBRMPA, 2014a, b).

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 Great Barrier Reef 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.

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 program has been delivered by James Cook University (JCU) and the Authority since 2005. The seagrass sub-program is also supported by contributions from the Seagrass-Watch program (Burdekin and Mackay–Whitsunday) and Queensland Parks and Wildlife Service (QPWS) through the Reef Joint Field Management Program (RJFMP).

Further information on the program objectives, and details on each sub-program are available on-line (GBRMPA 2021; 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 that can be used to assess how they respond to environmental pressures, and these measures are referred to herein as indicators (Figure 3).

These indicators respond at different temporal scales, with sub-lethal indicators able to respond from seconds to months, while the meadow-scale effects 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 (e.g. abundance, reproductive effort), to months and even years (e.g.

composition and meadow extent). Furthermore, indicators are conceptually linked to each other and to environmental drivers of concern, in particular, water quality.

Report Card category	Indicator category	Minutes-Days / Weeks / Months / Years	Seagrass report	Report card
Water quality	Climate	Cyclones	Y	
		Rainfall & river discharge^	Y	
		Wind (resuspension of sediments, scouring of sediments, currents)	Y	
		Extreme water temperature (hours/days > threshold)	Y	
		Chronic temperature rise (weekly anomalies)	Y	
	Water quality	Total suspended solids, turbidity, Secchi depth^		Y
		Chlorophyll a^		Y
		Nutrients (dissolved and particle forms of N, P & C)^		
		Temperature and salinity^		
		Water colour (weekly colour classes)^	Y	
		Benthic light (at seagrass canopy)	Y	
Seagrass	Habitat features	Sediment composition	Y	
		Epiphytes and macroalgae	Y	
	Seagrass condition	Abundance (per cent cover)	Y	Y
		Spatial extent	Y	
	Seagrass resilience	Reproductive structures	Y	
		Species composition	Y	Y
		Abundance threshold	Y	
		Seed bank	Y	

^Water quality monitoring program (TropWATER James Cook University, Australian Institute of Marine Science, Howley consulting)

*Coral monitoring program (Australian Institute of Marine Science)

Figure 3. Climate, environmental, seagrass condition and seagrass resilience indicators reported as part of inshore seagrass monitoring (see Table 2 for details on data source). Indicators that are used to calculate the Seagrass Index and Water Quality Index (indicating potential water quality pressures on Reef habitats) for the Reef Report Card are also indicated. All indicators are shown against their response time. Indicators colour grouped by category.

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 temperatures)
- 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 and macroalgae)
- substrate (e.g. grain size composition).

Indicators that 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 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 results in flooding, which exacerbates light limitation and transports pollutants (nutrients and pesticides), resulting in further seagrass loss (Preen *et al.* 1995).

- Daily 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 light. Therefore, it may be more informative to look at change relative to the sites.
- Elevated 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.* 2012).
- 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).

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 extent tends to adjust over longer time-scales (months to years). Seagrass extent 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 mega herbivores, such as dugongs and turtles. Therefore, extent and abundance 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).

Seagrasses expand and produce new shoots through clonal growth, but seagrasses are also angiosperms (flowering plants). Sexual reproductive structures (flowers, fruits, and seeds) are an important feature of a healthy seagrass meadow (Kenworthy 2000; Jarvis and Moore 2010; Rasheed *et al.* 2014). Sexual reproduction is necessary to form seed banks, which facilitate meadow recovery following periods of decline, and seed germination increases clonal diversity of the meadow (richness). The level of reproductive effort (reproductive structures per unit area) by a meadow in each season provides the basis of new propagules for recruitment in the following year (Lawrence and Gladish 2018; McKenzie *et al.* 2021a).

Seagrasses possess the ability to resist disturbances through physiological processes and modifications to morphology (i.e. growth form), and recover following loss by regeneration from seed and through clonal growth (sexual and asexual reproduction, respectively). Seagrass species vary in their dependence on resistance and recovery strategies. Broadly, we categorise species as having either persistent or colonising traits based on their ability to resist or recover, and species with a mixture of those traits are categorised as opportunistic (Kilminster *et al.* 2015) (Figure 4). The contributions of species, with different life history strategies, differs between seagrass habitats, and varies through time based on pressures acting on the habitats. Meadows dominated by colonising species have lower ability to resist pressures, but higher capacity to recover from disturbances. Therefore, changes in the species composition of a meadow can indicate meadow state and infer disturbance levels. For example, 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 4).

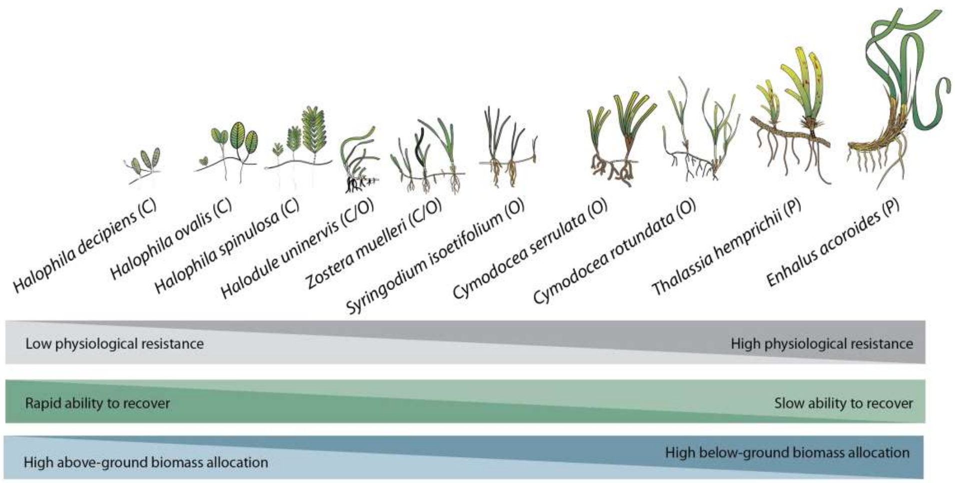


Figure 4. Dominant traits among the Reef seagrass species, with emphasis on their ability to either resist disturbances, or to recover following loss: colonising (C), opportunistic (O), or persistent (P). Adapted from Collier *et al*. (2021b) and Kilminster *et al*. (2015).

1.3 Structure of the Report

This report presents data from the fifteenth period of monitoring inshore seagrass ecosystems of the Reef under the MMP (undertaken from June 2020 to May 2021; hereafter called 2020–21). The inshore seagrass monitoring sub-program of the MMP reports on:

- abundance and species composition of seagrass (including seascape mapping) in the late dry season of 2020 and the late wet season of 2021 at inshore intertidal and subtidal locations
- resilience, including reproductive status of the seagrass species present at inshore intertidal and subtidal locations
- spatial and temporal patterns in light, turbidity, and temperature at sites where autonomous loggers are deployed
- trends in seagrass condition, measured as abundance (per cent cover) and resilience
- seagrass species composition 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 3 describes the drivers and pressures on the Reef during 2020–21, in the Driver-Pressure-State-Impact-Response (DPSIR) framework, followed by Section 4, which describes the condition and trend of inshore seagrass in the context of environmental factors.

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 Moran *et al*. (2022) are replicated to support the interpretation of the inshore seagrass results.

2 Methods summary

In the following, an overview is given of the data 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.* (2021c).

2.1 Climate and environmental pressures

Climate and environmental pressures affect seagrass condition and resilience (Figure 2). The pressures of greatest concern are:

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

The measures are either climate variables, which are generally not collected at a site-specific level, and within-canopy measure recorded at each site. The data source and sampling frequency is summarised in Table 2.

2.1.1. Climate

Cyclone tracks and total daily rainfall were accessed from the Australian Bureau of Meteorology from meteorological stations which were proximal to monitoring locations and provided by the MMP water quality sub-program (Moran *et al.* 2022).

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.

Information on exposure to different optical water types is generated by the MMP water quality sub-program (Moran *et al.* 2022). The inshore water quality sub-program includes a remote sensing component, which describes water quality characteristics for 22 weeks of the wet season (December–April). Water quality is described as water types of turbid, brown primary water, green secondary water, and tertiary waters. Colours are based on the Florel-Ule scale and are derived from daily Sentinel-3 OLCI Level 2 colour satellite images (Petus *et al.* 2019). Methods are detailed in Moran *et al*. (2022). Water colour has been confirmed as a predictor of changes in seagrass abundance (Petus *et al.* 2016). Primary and secondary water types have the greatest effect on seagrass habitats because light is attenuated by the high levels of suspended particulate matter, phytoplankton (chlorophyll-*a*) and dissolved matter. Exposure maps are therefore based on frequency of exposure to primary and secondary water types, while tertiary water exposure is also presented in summary tables for each site.

Table 1 Optical water types used to assess exposure of seagrass to water quality pressures (from Moran *et al.* 2022).

Optical water type	Description	Colour of water to the eye
Primary	Waters with high phytoplankton levels and increasing sediment and dissolved organic matter	Brownish-green
Secondary	Waters with colour still dominated by algae, but increased dissolved organic matter and some sediment may be present	Greenish water
Tertiary	Slightly below ambient water quality, but with high light penetration	Greenish-blue
Ambient/marine	High light penetration	Blue

Tidal height observations were used to determine if the tidal exposure regime may be increasing stress on seagrass and hence drive seagrass decline. Tidal observations were accessed from Maritime Safety Queensland and duration of annual air-exposure (hours) was determined for each meadow (i.e. monitoring site), based on the meadows height relative to the lowest astronomical tide (Appendix 2, Table 20).

2.1.2. Environment within or at the seagrass canopy

Autonomous iBTag™ submersible temperature loggers (iBCod™22L) were deployed at all sites identified in Appendix 3, Table 19. The loggers recorded temperature (accuracy 0.0625°C) within the seagrass canopy every 30–90 minutes (Table 2). 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 19). The light sensor is positioned upright at the seagrass canopy. 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}), hereinafter daily light. 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.* 2020). 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 (see 2.2.2) 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 2. 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	1968–2021	remote sensing and observations at nearest weather station	yearly	No. yr^{-1}	Bureau of Meteorology
Rainfall	1889–2021*	rain gauges at nearest weather station	daily	mm mo^{-1} mm yr^{-1}	Bureau of Meteorology
Riverine discharge	1970–2021	water gauging stations at river mouth		L d^{-1} L yr^{-1}	DES#, compiled by (from Moran *et al.* 2022)
Plume exposure	2006–2021 wet season (Nov–Apr)	remote sensing and field validation	weekly	frequency of water type (1–6) at the site	MMP inshore water quality program (from Moran *et al.* 2022)
Tidal exposure	1999–2021	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	2002–2021	iBTag	30–90 min	°C, temperature anomalies, exceedance of thresholds	MMP Inshore Seagrass monitoring
Light	2008–2021	Odyssey 2Pi PAR light loggers with wiper unit	15 min	daily light (mol m^{-2} d^{-1}) frequency of threshold exceedance (per cent of days)	MMP Inshore Seagrass monitoring
Sediment grain size	1999–2021	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 & site selection

Monitoring of inshore seagrass meadows occurred in the six natural resource management (NRM) regions with catchments draining into the Reef: Cape York, Wet Tropics, Burdekin, Mackay–Whitsunday, Fitzroy and Burnett–Mary (Table 3, Figure 5). Sixty-nine sites at 31 locations were assessed during the 2020–21 monitoring period (Table 3, Appendix 3, Table 19). This covered fifteen coastal, four estuarine, and twelve reef locations.

Table 3. Inshore seagrass monitoring locations and annual sampling. SW= Seagrass-Watch, RJFMP = Reef Joint Field Management Program, ● indicates late dry and late wet, ◐ indicates late dry only , and ◑ indicates late wet only. Shading indicates location not established. Blank cells indicate location not assessed. * indicates MMP assessments ceased in 2018.

NRM Region	Location	Program	2005–06	2006–07	2007–08	2008–09	2009–10	2010–11	2011–12	2012–13	2013–14	2014–15	2015–16	2016–17	2017–18	2018–19	2019–20	2020–21
Cape York	Shelburne Bay	MMP								●	●	◐	◐		◐	◐	◐	◐
	Piper Reef	MMP								●	●	◐	◐	◐	◐	◐	◐	◐
	Flinders Group	MMP, RJFMP								●	●	●	●	◐	◐	◐	◐	◐
	Bathurst Bay	MMP, RJFMP								●	●	◐	●	◐	◐	◐	◐	◐
	Weymouth Bay	SW							◑	◑		◐						
	Lloyd Bay	RJFMP											◐	◐	◐		◐	◐
	Archer Point	MMP*, SW	●	●	●	●	●	●	●	●	●	◐	◐	◐	◐			
Wet Tropics	Low Isles	MMP				●	●	●	●	●	●	●	●	●	●	●	●	●
	Yule Point	MMP	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●
	Green Island	MMP	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●
	Mission Beach	MMP	●	●	●	●	●	●	●	●	●	●	●	●	●	●	◐	●
	Dunk Island	MMP			●	●	●	●	●	●	●	●	●	●	●	●	●	●
	Rockingham Bay	SW				◐	◐	◐	◐	◐			◐	◐				
	Missionary Bay	RJFMP											◐	◐	◐	◐	◐	◐
Burdekin	Magnetic Island	MMP	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●
	Townsville	MMP, SW	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●
	Bowling Green Bay	MMP								●	●	●	●	●	●	●	●	●
	Bowen	SW		◑	●	●	●	◑									●	●
Mackay–Whitsunday	Shoal Bay	SW	●	●	●	●	●	●	●	◐	◐	◐	◐	●	●	●	●	●
	Pioneer Bay	MMP, SW	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●
	Whitsunday Island	RJFMP											◐	◐	◐	◐	◐	◐
	Hamilton Island	MMP			●	●	●	●	●	●	●	●	●	●	●	●	◐	●
	Lindeman Island	MMP													●	●	◐	●
	Repulse Bay	MMP	●	●	◑	◑	◑	◑	●	●	●	●	●	●	●	●	●	●
	St Helens Bay	SW													◐	◐	◐	◐
	Newry Islands	RJFMP											◐	◐	◐	◐	◐	◐
	Sarina Inlet	MMP	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●
	Clairview	SW													◐	◐	◐	◐
Fitzroy	Shoalwater Bay	MMP	●	●	●	●	●	●	●	●	●	◐	◐	◐	●	◐	●	◐
	Keppel Islands	MMP			●	●	●	●	●	●	●	◐	◐	◐	●	●	●	●
	Gladstone Harbour	MMP	●	●	●	●	●	●	●	●	●	◐	◐	●	●	●	●	●
Burnett–Mary	Rodds Bay	MMP			●	●	●	●	●	●	●	●	◐	●	●	●	●	●
	Burrum Heads	MMP, SW	●	●	◐	●	◐	●	●	●	●	◐	●	●	●	●	●	●
	Hervey Bay	MMP	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●

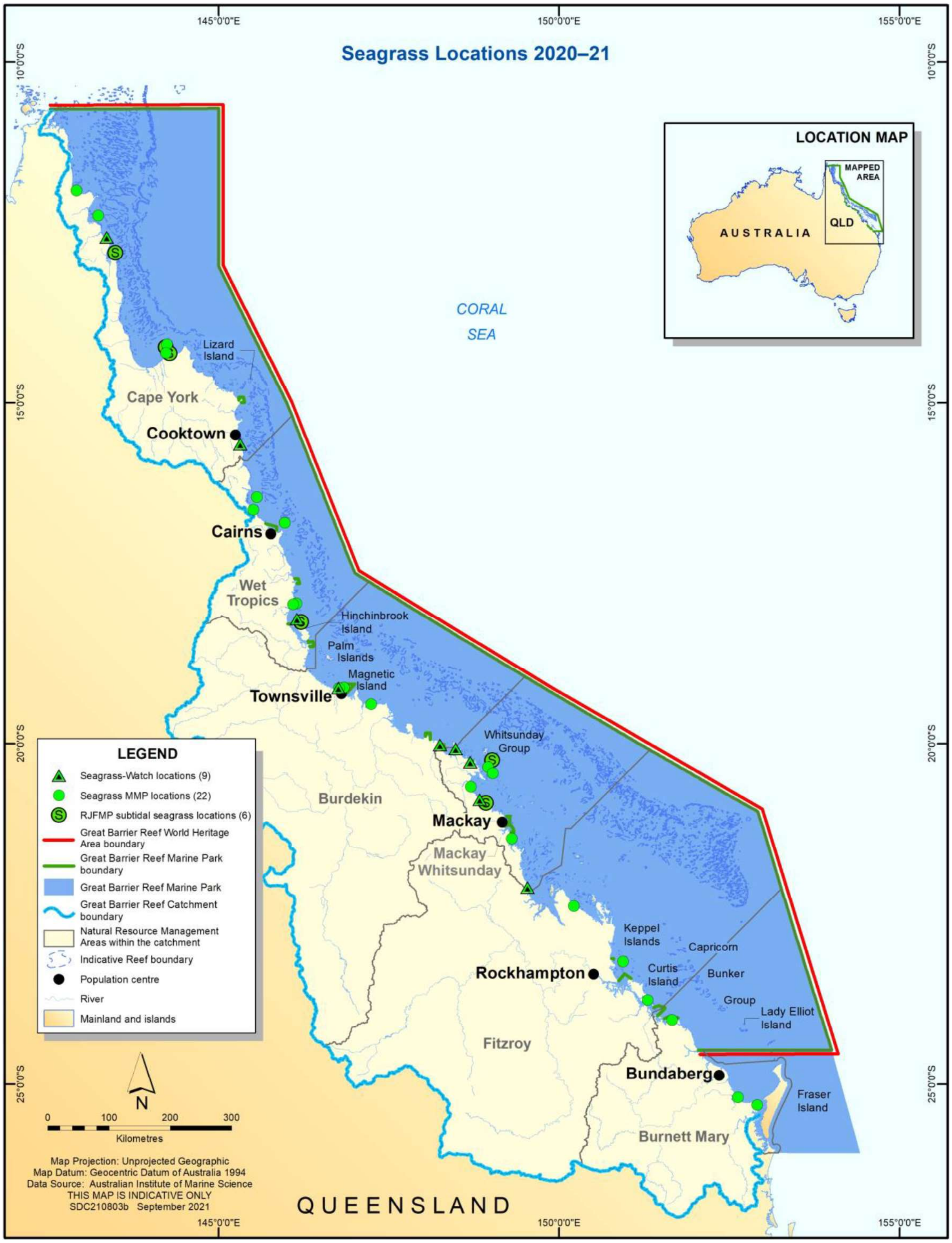


Figure 5. Inshore seagrass survey locations that exist as of 2020-21. However, not all locations were surveyed in 2020-21 (see Table 2).

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)
- meadows that span a range in exposure to riverine discharge with those in estuarine and coastal habitats generally having the highest degree of exposure, and reef meadows
- 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)
- meadows that are not extremely variable in per cent cover throughout the survey area i.e. a Minimum Detectable Difference (MDD) below 20 per cent (at the 5 per cent level of significance with 80 per cent power) (Bros and Cowell 1987).

Sentinel monitoring 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 per cent) it was based on historic (>5 yr) information.

Representative meadows were those which (1) covered the greater extent within the inshore region, (2) were generally the dominant seagrass community type and (3) those meadows 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. reproductive effort, seed banks).

The long-term median annual daylight exposure (the time intertidal meadows are exposed to air during daylight hours) was 1.7 per cent (all meadows pooled) (Table 20). This limited the time monitoring could be conducted to the very low spring tides within small tidal windows (mostly 1–4 hrs per day for 3–6 days per month for 6–9 months of the year).

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 nearshore to clearer offshore waters
- 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 depth) 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 per cent level of significance with 80 per cent power (McKenzie *et al.* 1998).

The meadows chosen for monitoring were in fact lower littoral (rarely exposed to air), although classified intertidal within the MMP.. 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, such as reef locations in Cape York, Wet Tropics, Burdekin and Mackay–Whitsunday (Table 4).

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 opportunistic 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 that 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, Syringodium, Thalassia* and *Zostera* (Table 4).

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. Changes in indicators at sites sampled in the late dry only (e.g. Cape York) are most likely to be in response to wet season conditions in the previous reporting period.

Apart from the 47 MMP long-term monitoring sites, data included 10 sites from Seagrass-Watch and 12 sites from QPWS to improve the spatial resolution and representation of subtidal habitats (Table 5).

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 19). The seagrass species (including foundation) present at each monitoring site is listed in Table 4 and Table 5.

2.2.2 Seagrass abundance, composition and extent

Seagrass abundance, species composition, and meadow spatial extent were assessed from samples collected in the late dry 2020 and late wet 2021 at locations identified in Table 4. 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. Sites were defined as a 5.5 hectare area intertidally and 3.1 hectares subtidally, within a relatively homogenous section of a representative seagrass community/meadow (McKenzie *et al.* 2003).

Monitoring at sites in the late dry (September-November 2020) and late wet (March-May 2021) 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). Transects are placed in the same position (±3 m) each assessment. The sampling strategy for subtidal sites was modified in 2020–21, as a result of the discontinuation of SCUBA divingdriven by budgetary constraints, logistic and occupational health and safety issues relating to diving in poor visibility coastal waters. At each site, a GoPro® drop–camera assembly (incl. frame with 0.25 m^2 quadrat in field of view), was used to visually assess the seabed and the photoquadrat footage captured for post-field analysis. Along three 50 m transects within a 50 m radius of a central point, between 10 and 33 photoquadrats were assessed for seagrass percentage cover, species composition and macroalgae abundance. Subtidal assessments were conducted using a real time drop-camera slaved to a surface tablet, to ensure photoquadrats were sufficiently spaced apart and the vision captured was suitable for post-field analysis. A van Veen grab was used to validate seagrass species observed on the tablet screen and to assess sediment composition.

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.* 2021c).

Mapping of the meadow extent and meadow–scape (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 at intertidal sites and drop-camera at subtidal sites. Seagrass meadow–scape that tended to grade from dense continuous cover to no cover(i.e. over a continuum that included small patches and shoots of decreasing density) had the meadow edge delineated where there was a non-vegetated space with the distance of more than 3 metres (i.e. accuracy of the GPS). Each entire site (5.5 ha intertidal and 3.1 ha subtidal) was mapped (seagrass and no seagrass). It should be noted that within a site, areas that are not suitable for seagrass can occur, e.g. consolidated sediments, coral reef or dry sandy beach. The relative spatial extent was calculated by dividing the mapped seagrass area by the total habitable area for seagrass within the entire site.

Table 4. Inshore sentinel seagrass long-term monitoring site details including presence of foundation (■) and other (□) seagrass species in the current or previous reporting periods. * = 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		Longitude	Latitude	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	142.914	-11.887			■		□		■		■	
				SR2*	Shelburne Bay	142.916	-11.888										
			Piper Reef *reef*	FR1*	Farmer Is.	143.234	-12.256	■				□				■	
				FR2*	Farmer Is.	143.236	-12.257										
		Normanby / Jeannie	Flinders Group *reef*	ST1*	Stanley Island	144.245	-14.143	■		■		□		■	■	■	
				ST2*	Stanley Island	144.243	-14.142										
			Bathurst Bay *coastal*	BY1*	Bathurst Bay	144.233	-14.268	■				□		■	■	■	■
				BY2*	Bathurst Bay	144.232	-14.268										
Northern	Wet Tropics (*Terrain NRM*)	Daintree	Low Isles *reef*	LI1*	Low Isles	145.565	-16.385					□		■		■	
				LI2^	Low Isles	145.564	-16.383					□		■			
		Mossman / Barron / Mulgrave-Russell / Johnstone	Yule Point *coastal*	YP1*	Yule Point	145.512	-16.569					□		■			■
				YP2*	Yule Point	145.509	-16.564										
			Green Island *reef*	GI1*	Green Island	145.973	-16.762	■	■			□		■		■	
				GI2*	Green Island	145.976	-16.761										
				GI3^	Green Island	145.973	-16.755	■	■			□		■	■	■	
		Tully / Murray / Herbert	Mission Beach *coastal*	LB1*	Lugger Bay	146.093	-17.961					□		■			
				LB2*	Lugger Bay	146.094	-17.961										
			Dunk Island *reef*	DI1*	Pallon Beach	146.141	-17.944	■	■			□		■		■	
				DI2*	Pallon Beach	146.141	-17.946										
				DI3^	Brammo Bay	146.140	-17.932		■		□	□		■			
Central	Burdekin (*NQ Dry Tropics*)	Ross / Burdekin	Magnetic island *reef*	MI1*	Picnic Bay	146.841	-19.179					□		■			■
				MI2*	Cockle Bay	146.829	-19.177	■	■			□		■	■	■	
				MI3^	Picnic Bay	146.841	-19.179		■		□	□	□	■			
			Townsville *coastal*	SB1*	Shelley Beach	146.771	-19.186		■			□		■			■
				BB1*	Bushland Beach	146.683	-19.184										
			Bowling Green Bay *coastal*	JR1*	Jerona (Barratta CK)	147.241	-19.423					□		■			■
				JR2*	Jerona (Barratta CK)	147.240	-19.421										
	Mackay–Whitsunday (*Reef Catchments*)	Proserpine / O'Connell	Lindeman Island *reef*	LN1^	Lindeman Is.	149.028	-20.438					□		■			
				LN2^	Lindeman Is.	149.032	-20.434										
			Repulse Bay *coastal*	MP2*	Midge Point	148.702	-20.635					□		■			■
				MP3*	Midge Point	148.705	-20.635										
			Hamilton Island *reef*	HM1*	Catseye Bay - west	148.957	-20.344					□		■	■		■
				HM2*	Catseye Bay - east	148.971	-20.347										
		Plane	Sarina Inlet *estuarine*	SI1*	Point Salisbury	149.304	-21.396					□		■			■
				SI2*	Point Salisbury	149.305	-21.395										
Southern	Fitzroy (*Fitzroy Basin Association*)	Shoalwater / Fitzroy	Shoalwater Bay *coastal*	RC1*	Ross Creek	150.213	-22.382					□		■			■
				WH1*	Wheelans Hut	150.275	-22.397										
			Keppel Islands *reef*	GK1*	Great Keppel Is.	150.939	-23.196					□	□	■			■
				GK2*	Great Keppel Is.	150.940	-23.194										
		Calliope / Boyne	Gladstone Harbour *estuarine*	GH1*	Pelican Banks	151.301	-23.767					□		■			■
				GH2*	Pelican Banks	151.304	-23.765										
	Burnett–Mary (*Burnett–Mary Regional Group*)	Baffle	Rodds Bay *estuarine*	RD1*	Cay Bank	151.655	-24.058					□		■			■
				RD3*	Turkey Beach	151.589	-24.038										
		Burrum	Burrum Heads *coastal*	BH1*	Burrum Heads	152.626	-25.188					□		■			■
				BH3*	Burrum Heads	152.639	-25.210										
		Mary	Hervey Bay *estuarine*	UG1*	Urangan	152.907	-25.301					□		■			■
				UG2*	Urangan	152.906	-25.303										

Table 5. Additional inshore sentinel seagrass long-term monitoring sites integrated from the Seagrass-Watch (intertidal sites)* and RJFMP drop-camera (subtidal sites)^ programs, including presence of foundation (■) and other (□) seagrass species in the current or previous reporting periods. NRM region from www.nrm.gov.au. * = intertidal, ^ =subtidal, ~ =not assessed in 2020–21.

Region	NRM region (*Board*)	Basin	Monitoring location	Site		Longitude	Latitude	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	143.36059	-12.571	■	■	■		□		■		■	
			Lloyd Bay *coastal*	LR1^	Lloyd Bay	143.485	-12.797					□	□	■			
				LR2^	Lloyd Bay	143.475	-12.825										
		Normanby / Jeannie	Flinders Group *reef*	FG1^	Flinders Island	144.225	-14.182					□	□	■			
				FG2^	Flinders Island	144.225	-14.182										
			Bathurst Bay *coastal*	BY3^	Bathurst Bay	144.285	-14.276					□		■			
				BY4^	Bathurst Bay	144.300	-14.275										
		Endeavour	Archer Point *reef*	AP1*~	Archer Point	145.31894	-15.60832	■	■	■		□		■		■	□
				AP2*~	Archer Point	145.31847	-15.60875										
Northern	Wet Tropics	Tully / Murray / Herbert	Rockingham Bay *reef*	GO1*~	Goold Island	146.15327	-18.17395	■	■			□		■			
			Missionary Bay *coastal*	MS1^	Cape Richards	146.213	-18.216					□		■			
				MS2^	Macushla	146.217	-18.205										
Central	Burdekin (*NQ Dry Tropics*)	Ross / Burdekin	Townsville *coastal*	SB2*	Shelley Beach	146.763	-19.182		■			□		■			■
		Don	Bowen *coastal*	BW1*	Port Dennison	148.250	-20.017					□		■			■
				BW2*	Port Dennison	148.252	-20.017										
	Mackay–Whitsunday (*Reef Catchments*)	Proserpine	Shoal Bay *reef*	HB1*	Hydeaway Bay	148.482	-20.075	■				□		■		■	
				HB2*	Hydeaway Bay	148.481	-20.072										
			Pioneer Bay *coastal*	PI2*	Pigeon Island	148.693	-20.269					□	□	■			■
				PI3*	Pigeon Island	148.698	-20.271										
		Proserpine / O'Connell	Whitsunday Island *reef*	TO1^	Tongue Bay	149.016	-20.240					□		■		■	
				TO2^	Tongue Bay	149.012	-20.242										
		O'Connell / Pioneer	St Helens Bay *coastal*	SH1*	St Helens Bch	148.835	-20.822					□		■			■
			Newry Islands *coastal*	NB1^	Newry Bay	148.926	-20.868		■			□	□	■	■		
				NB2^	Newry Bay	148.924	-20.872										
		Plane	Clairview *coastal*	CV1*	Clairview	149.533	-22.104					□		■			■
				CV2*	Clairview	149.535	-22.108										

2.2.3 Seagrass reproductive status

Seagrass reproductive state was assessed from samples collected in the late dry 2020 and late wet 2021 at locations identified in Table 4. Samples were processed according to standard methodologies (McKenzie *et al.* 2021c).

In the field, 15 haphazardly placed cores (100 mm diameter x 100 mm depth) of seagrass were collected 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 (2 mm mesh) 30 cores (50 mm diameter, 100 mm 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 Epiphytes and macroalgae

Epiphyte and macroalgae cover were measured in the late dry and late wet seasons 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 Calculating Report Card scores

2.3.1 Seagrass abundance

Seagrass abundance state in the MMP 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 18).

Abundance 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 per cent 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 6) 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 6. Scoring threshold table to determine seagrass abundance grade. low = 10th or 20th percentile guideline. NB: scores are unitless.

Grade	Percentile category	Score
very good	75–100	100
good	50–75	75
moderate	low–50	50
poor	<low	25
very poor	<low by >20 per cent	0

2.3.2 Seagrass resilience

Resilience can be described as the capacity of an ecosystem to cope with disturbance (Connolly *et al.* 2018), and to adapt to change without switching to an alternative state (Holling 1973; Unsworth *et al.* 2015). For monitoring and reporting, '*a set of measurable biological characteristics that exemplify seagrass meadows' resistance to pressures and essential mechanisms for recovery*' are required to assess resilience (Udy *et al.* 2018). The resilience indicator takes a subset of measurable characteristics for which long-term data is available to develop a score.

The seagrass resilience indicator is based on the premise that resilience includes a resistance and recovery element. Seagrass species vary in their dependence on these traits. 'Colonising' species generally have low levels of resistance traits and 'persistent' species have high levels of these traits. Resistance is incorporated into the metric through meadow condition, and whether abundance and species composition exceed critical thresholds (<20th percentile or >50 per cent, respectively). It is also influenced by the proportion of persistent species. Sites that are dominated by colonising species therefore have low levels of resistance, making them highly vulnerable to events such as periods of elevated turbidity caused by flood plumes. Sites that are in impacted state and have low abundance relative to the average for that site are also vulnerable.

Reproductive effort indicates potential for recovery from seeds and likelihood of high clonal diversity. By contrast, traits that enable the species to recover following an impact are the highest in 'colonising' species and lowest in 'persistent' species. These traits include forming a seed bank from flowers and rapid growth rates. 'Opportunistic' species have traits of both resistance and recovery.

The resilience score is calculated using a decision tree. It includes resistance potential and likelihood of recovery based on reproductive effort (as a proxy for seed/propagules) graded according to the species in the habitat.

Sites are scored from 0 to 100 in each year using a decision tree (Collier *et al.* 2021a). The three main categories within the tree are:

- low resistance sites
- high resistance sites but non-reproductive (low recovery potential)
- high resistance and reproductive (increased recovery potential).

The conceptual basis for the resilience indicator and the statistical analysis supporting the decisions in the tree are detailed in Appendix 2, Figure 91.

The resilience scores are graded as: very poor (<20), poor ($20 \leq 40$), moderate ($40 \leq 60$), good ($60 \leq 80$), very good ($80 \leq 100$).

Table 7. Scoring thresholds and decisions for the resilience metric. *Foundational = opportunistic and persistent species. NB: scores are unitless.

Description	Species composition / abundance	Reproductive effort	Score calculation	Score	Category
1 Low resistance	Per cent colonising species >50 per cent AND/OR total per cent cover <20th percentile of site	Reproduction not present	Proportion of colonising species	0–15	1.1
		Reproduction present (any species)	Proportion of foundational species and reproductive presence/absence	5–30	1.2
2.1 High resistance but low recovery potential	Per cent foundational species > 50 per cent AND total cover >20th percentile of site	Reproduction (foundational) not present last 3 years	Proportion of persistent species present (min <10th percentile, max 95th percentile)	30–50	2.1.1
		Not reproductive this year, but reproductive (foundational) in last 3 years (seed bank is likely to be present)		50–70	2.1.2
2.2 High resistance and high recovery potential	Per cent foundational species >50 per cent AND total cover >20th percentile of sites AND persistent species present	Reproduction (foundational) present	Reproductive structure count (min <10th percentile, max 95th percentile)	70–100	2.2.1
				85–100	2.2.2

2.3.3 Seagrass condition index

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

- seagrass abundance (per cent cover)
- seagrass resilience.

Each indicator is equally weighted, in accordance with the Paddock to Reef Integration Team's original recommendations. 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 8). *Please note: Cape York omitted from the score in reporting prior to 2012 due to poor representation of inshore monitoring sites*.

Table 8. Area of seagrass shallower than 15 m in each region within the World Heritage Area boundaries.(from McKenzie *et al.* 2014a; McKenzie *et al.* 2014b; 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**

2.4 Data analyses

All analysis was run in the software R 4.1.0 (R Core Team 2021).

2.4.1 Score propagation of error

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 Reef), 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 two seagrass indicators and overall index.

2.4.2 Abundance (per cent cover) generalised additive models (GAM)

Due to the high proportion of zeros and the unbalance of the per cent cover data through time (different sites monitored at each seasonal sampling period), we used a two-step approach to show the temporal trend.

1) Modelling the per cent cover average and confidence intervals for each sampling event.

The first step of the analysis was to accurately estimate the mean and 95 per cent CI for each season sampling period across various level (e.g. Reef wide, per NRM region, per habitat types). Because the data we want to analysed is a percentage with a high proportion of 0, we need to use a zero-inflated beta distribution (ZABE) (Zuur, Beginner's Guide to Zero-Inflated Models with R ,2016). The package gamlss (Rigby and Stasinopoulos 2005) was used for the analysis with the family BEZI (https://search.r-project.org/CRAN/refmans/gamlss.dist/html/BEZI.html).

The zero-inflated beta distribution is given as :

1) if (y=0) – Binomial model

f(y) = nu

2) if y=(0,1) – Beta model

f(y|mu,sigma)=(1-nu)*(Gamma(sigma)/Gamma(mu*sigma)*Gamma((1-mu)*sigma))*y^(mu*sigma-1)*(1-y)^(((1-mu)*sigma)-1)

The parameters satisfy 0<mu<1, sigma>0 and 0<nu<1.

The expected values (E) and variance (VAR) are:

E(y)=(1-nu)*mu

Var(y)=(1-nu)*(mu*(1-mu))/(sigma+1) + nu*(1-nu)*mu^2

Per cent cover at the quadrat level for each seasonal date was analysed separately to be able to include the random effect of Site as they vary through time and cannot be accurately estimated over the whole time series. The intercept model fitted was as followed:

Formula : Percent_cover ~ 1 + re (random(~1|Site)

The random effect of site was included in the three parameters estimated (mu, sigma and nu) but was dropped for sigma and nu if a parametrization error was encountered. In the extreme case of a zero-inflation superior to 95 per cent all random effects were dropped due to very limited number of quadrats with seagrass present.

We used a common bootstrapping method where a random distribution of 10000 was produced for mu and nu based on their parameter estimates and standard error outputted by the gamlss package to calculate the mean and 95 per cent CI of the resulting model.. This gave 10000 expected values where the mean, 2.5 quantile and 97.5 quantile were calculated.

In the case where only a few sites were included (<5) and one of the sites only had 0 per cent cover for all quadrats, the algorithm was having difficulties estimating the zero-inflation parameter (nu) with the inclusion of site as a random effect. This resulted in the bootstrapped expected values to not be normally distributed (2 separate peaks of values centred on 0 and on the mean of the sites with seagrass present) which would not lead to an appropriate estimate of the overall mean. In these very rare scenarios, the same zero-inflated beta model was run but with site as a fixed effect which led to a distribution of bootstrapped expected values for each site. The overall mean was obtained as the arithmetic mean of the site bootstrapped mean and the 2.5 quantile and 97.5 quantile were respectively the minimum and maximum of the 2.5 quantile and 97.5 quantile of the site bootstrapped CI.

This process was repeated of each seasonal date at various scales. As part of our regular validation process the residuals of all models were checked for violations of the generalised model assumptions.

2) Trends in per cent cover

Generalised additive models (GAMs) with the beta (logit link) family were fitted to resulting mean and 95 per cent CI from the first process to identify the presence and consistency of trends through time, using the mgcv (Wood 2020) package. The GAMs were used in a multilevel approach to show trends at the Reef, NRM region, habitat, location and site levels. The details and summary outputs of all the GAMs shown in the figures can be found in the Appendix (Table 23 Table 24, Table 25). There was no significant autocorrelation observed for consecutive years of order 1 to 3. However, the GAMs were weighted based on how many sites were included in the mean calculated to ensure the seasonality and unbalanced nature of our sampling was not affecting the long-term trend.

The final results presented were:

- the prediction for the GAM fitted through the mean points
- lower CI as the predictions – 1.96*SE of the GAM fitted through the lower 95 per cent CI points
- upper CI as the predictions + 1.96*SE of the GAM fitted through the upper 95 per cent CI points

2.4.3 Abundance (per cent cover) long-term trends

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. 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).

2.4.4 Resilience

Analysis of trends in the resilience scores was conducted using Generalised Linear Models (GLMs) with a gaussian distribution instead of GAMs, as this metric relies on samples collected once a year. Due to the low frequency of sampling the use of a smoother (GAM) is not recommended.

2.5 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, 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 (GAM 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 National Mapping Division of Geosciences Australia geodata topographic basemap (1:100 000 scale digital data).

3 Drivers and pressures influencing seagrass meadows in 2020–21

The following section provides detail on the overall climate and environmental pressures during the 2020–21 monitoring period, at a relatively broad level as context for understanding trends in seagrass condition. It includes:

- Climate (cyclones and rainfall), river discharge and turbid water exposure
- daily light (within-canopy)
- within-canopy temperature and threshold exceedance
- seagrass meadow sediment characteristics.

Supporting data is detailed within Appendix 2 and 3:

3.1 Summary

Long-term trends in the Water Quality Index (without interannual variability) indicate recent improvement in water quality in the Wet Tropics (after declining from good to moderate in 2008–2018), while it has declined over the last decade in the Burdekin and Mackay–Whitsunday regions, but stabilised in recent years (Moran *et al.* 2022). The annual condition index (sensitive to year-to-year variability) in 2020–21 was moderate in the Wet Tropics, Burdekin, Mackay-Whitsunday and Cape York.

Environmental stressors in 2020–21 were around average for rainfall and river discharge, following a dry year in 2019–20 (Table 9). River discharge was only slightly higher than the long-term median for the Reef catchment area, but this depended on the region: discharge was elevated in the northern three regions (Cape York, Wet Tropics and Burdekin) and below the median in the three southern regions (Mackay–Whitsunday, Fitzroy, and Burnett–Mary).

The frequency with which the monitoring sites were exposed to turbid primary and secondary waters was slightly below the long-term average across the Reef, but in the southern NRM regions there was considerably lower prevalence of primary water and more secondary water exposure (Figure 9). The presence of this turbid water is affected by resuspension-driven events as well as discharge and the relative attribution to these processes is discussed in further detail by Moran *et al*. (2022).

Table 9. Summary of environmental conditions at monitoring sites across the Reef in 2020–21 compared to previous monitoring period and the long-term average (range indicated for each data set). *intertidal only.

Environmental pressure	Long-term average	2019–20	2020–21
Climate			
Cyclones, number of events (1968–2020)	4	0	2
Wet season daily rainfall, mm d^{-1} (1960–1991)	4.0	3.0	4.0
Riverine discharge, ML yr^{-1} (1986–2016)	51,812,207	30,911,889	56,547,662
Wet season turbid water exposure, per cent (2003–2018)	89	92	81
Within seagrass canopy			
Temperature, °C (±) (max) (2003–2020)*	25.7 ±0.1 (46.6)	25.9 ±0.2 (41.1)	25.7 ±0.1 (41.9)
Daily light, mol m^{-2} d^{-1} (2008–2020) annual average (min site–max site)	12.4 (3.3–20.8)	13.1 (4.2–22.2)	12.5 (2.7–17.9)
Proportion mud, per cent			
estuarine intertidal (1999–2020)	45.1 ±2.1	42.0 ±2.9	39.2 ±2. 6
coastal intertidal (1999–2020)	28.0 ±2.1	22.3 ±1.7	20.6 ±1.6
coastal subtidal (2015–2020)	52.8 ±2.3	48.2 ±2.4	55.2 ±3.9
reef intertidal (2001–2020)	4.3 ±1.2	4.0 ±0	4.1 ±0.4
reef subtidal (2008–2020)	12.5 ±0.9	12.8 ±2.5	38.5 ±0.5

Daily light levels were around the long-term Reef average in 2020–21. Light was around the long-term average in four regions, but below-average in the Mackay–Whitsunday and Fitzroy regions. It was lower than average at more than half of the light monitoring locations. Light levels were higher than estimated annual light requirements for optimal growth (10 mol m^{-2} d^{-1}) at all but eight locations.

Within canopy temperatures in 2020–21 were slightly lower than the 2019–20 period, similar to the long-term average, and the coolest in seven years (Figure 8). The number of extreme heat days (days >40°C) were the lowest in five years, and restricted to the Mackay–Whitsunday and Fitzroy regions (Figure 12).

Two tropical cyclones entered the Reef waters in the 2020–21 wet season, but neither crossed the coast (Moran *et al.* 2022). The first was cyclone Kimi in mid-January (16 to 19 January 2021), which briefly formed over the central Reef but did not make landfall (BOM 2021). The second was severe tropical cyclone Niran which formed in the central section of the Reef in late February (27 February to 5 March 2021) and caused elevated wind/wave conditions and rainfall along the Wet Tropics coast, although it then moved further offshore and did not make landfall (BOM 2021).

Additionally, two other significant storm events influenced the Reef during the period, both being rain depressions. The first was the remnants of cyclone Imogen in early January (1 to 6 January 2021) that had formed in the Gulf of Carpentaria before weakening and causing considerable rainfall along the central catchments of the Reef. Similarly, cyclone Lucas formed in the Gulf of Carpentaria and moved eastwards across the Queensland mainland over the northern sections of the Reef catchments (BOM 2021).

3.2 Rainfall

Rainfall was below the long-term average in basins from the Don River south including all basins in the Mackay–Whitsunday, Fitzroy and Burnett–Mary regions (Figure 6, Figure 7). Rainfall was above the long-term average in most of the other central and northern catchments except the Daintree and Endeavour Rivers. The largest deviations from the long-term averages occurred in southern Wet Tropics (Tully, Murray and Herbert Rivers) and the Stewart River draining into the Reef just north of Princess Charlotte Bay in Cape York.

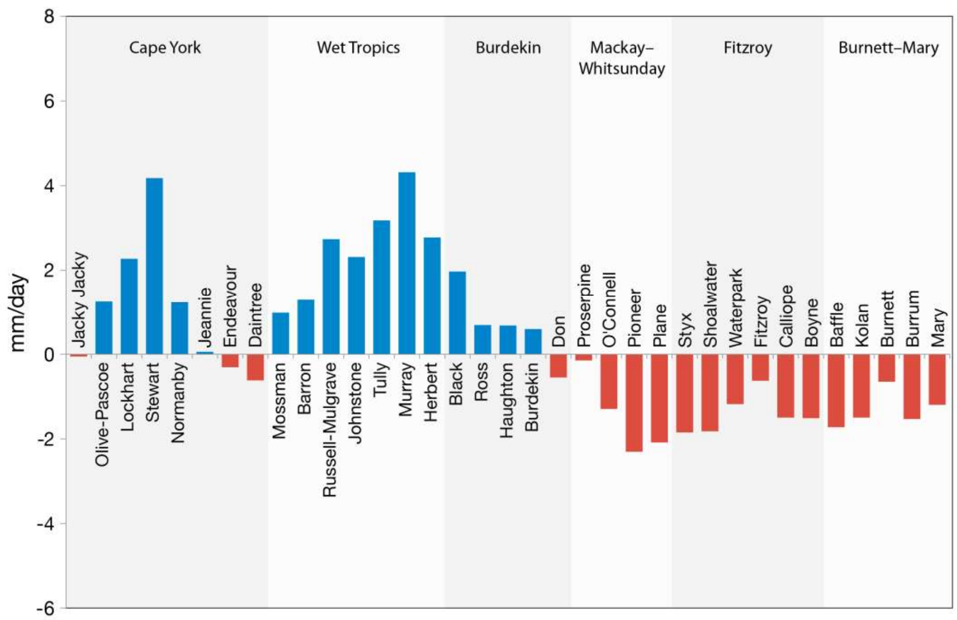


Figure 6. Per basin difference between annual average daily wet season rainfall (December 2020–April 2021) and the long-term average (1961–1990). Red and blue bars denote basins with rainfall below and above the long-term average, respectively. Note that the basins are ordered from north to south (left to right). Basins have been grouped into NRM regions as indicated by shaded panels. Compiled by Moran *et al.* (2022).

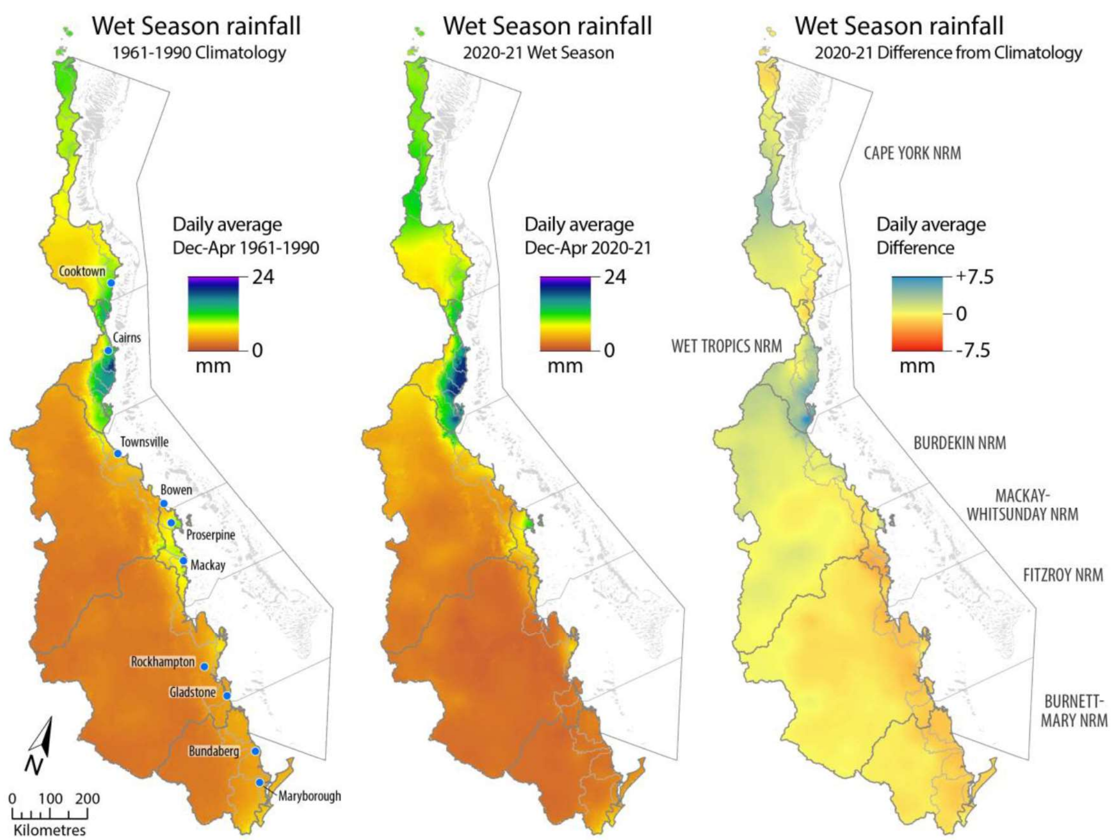


Figure 7. Average daily rainfall (mm day^{-1}) in the Reef catchment: (left) long-term annual average (1961–1990; time period produced by BOM), (centre) 2020–21 and (right) the difference between the long-term annual average and 2020–21 rainfall patterns where negative values indicate less rain. From by Moran *et al.* (2022).

3.3 River discharge

Annual river discharge for the Reef was around the long-term average in 2020–21 following a dry year in 2019–20, and a wet year in 2018–19 (Table 10). Discharges from basins entering the central and southern Reef were below average in most except some of the small basins in the Fitzroy region, the Black River in the Burdekin region, and the Burdekin River which was more than 1.5 times larger than the long-term median due to rainfall events in the upper catchment. In the Wet Tropics and Cape York, river discharge was above average in all but one of the basins. Substantial discharge (>1.5 times the long-term median) occurred in the southern Wet Tropics (Herbert and Murray Rivers) and northern Cape York.

Table 10. Annual water year discharge (ML) of the main Reef rivers (1 October 2020 to 30 September 2021, 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 Moran *et al.* (2022).

Region	Basin	LT median	2017–18	2018–19	2019–20	2020–21
Cape York	Jacky Jacky Creek	2,047,129	2,689,450	3,124,009	1,920,007	3,324,787
	Olive Pascoe River	2,580,727	3,424,596	6,992,798	3,189,195	5,361,951
	Lockhart River	1,634,460	2,168,911	4,428,772	2,019,824	3,395,902
	Stewart River	674,618	826,499	3,109,052	584,988	1,470,654
	Normanby River	4,159,062	4,333,023	12,102,053	2,792,858	5,928,821
	Jeannie River	1,263,328	1,721,175	3,350,682	932,300	1,782,930
	Endeavour River	1,393,744	1,796,913	3,847,478	773,315	1,552,254
Wet Tropics	Daintree River	1,512,054	1,439,220	4,752,327	901,248	1,490,754
	Mossman River	858,320	1,069,336	1,885,921	555,280	910,701
	Barron River	574,567	946,635	1,535,892	320,056	615,937
	Mulgrave-Russell River	2,600,465	3,359,834	3,550,093	1,694,470	3,025,022
	Johnstone River	3,953,262	4,950,329	4,774,747	2,743,805	4,485,038
	Tully River	3,241,383	3,883,954	4,020,452	2,200,744	4,123,338
	Murray River	380,472	521,465	519,739	199,630	592,702
	Herbert River	3,556,376	6,385,655	5,707,209	1,472,338	6,271,988
Burdekin	Black River	208,308	386,030	965,544	102,296	304,652
	Ross River	377,011	83,113	2,371,556	133,165	72,975
	Haughton River	419,051	598,668	2,363,209	251,321	446,782
	Burdekin River	4,406,780	5,542,306	17,451,417	2,203,056	8,560,072
	Don River	508,117	321,875	1,356,004	398,312	441,329
Mackay-Whitsunday	Proserpine River	284,542	174,183	837,962	205,680	148,928
	O'Connell River	478,097	260,937	1,223,297	279,585	253,873
	Pioneer River	692,342	249,530	1,158,768	383,506	235,359
	Plane Creek	309,931	75,052	351,879	299,502	125,665
Fitzroy	Styx River	155,384	218,115	109,376	225,782	280,934
	Shoalwater Creek	129,487	181,763	91,147	188,152	234,112
	Water Park Creek	97,115	136,322	68,360	141,114	175,584
	Fitzroy River	2,852,307	954,533	1,339,964	2,533,631	397,027
	Calliope River	152,965	141,438	2,682	80,255	25,097
	Boyne River	38,691	35,775	678	20,300	6,348
Burnett-Mary	Baffle Creek	215,446	1,081,646	930	47,143	12,271
	Kolan River	52,455	325,578	4,958	5,304	114
	Burnett River	230,755	849,051	202,436	332,366	118,241
	Burrum River	79,112	715,449	63,972	70,928	14,743
	Mary River	981,183	1,630,741	658,014	472,580	360,779
	Sum of basins	51,812,207	53,479,101	94,323,378	30,674,035	56,547,662

3.4 Turbid water exposure and flood plume extent

The frequency of exposure to wet season water types, extent of the water types, and the within-canopy environmental pressures daily light and water temperature are summarised in Figure 8.

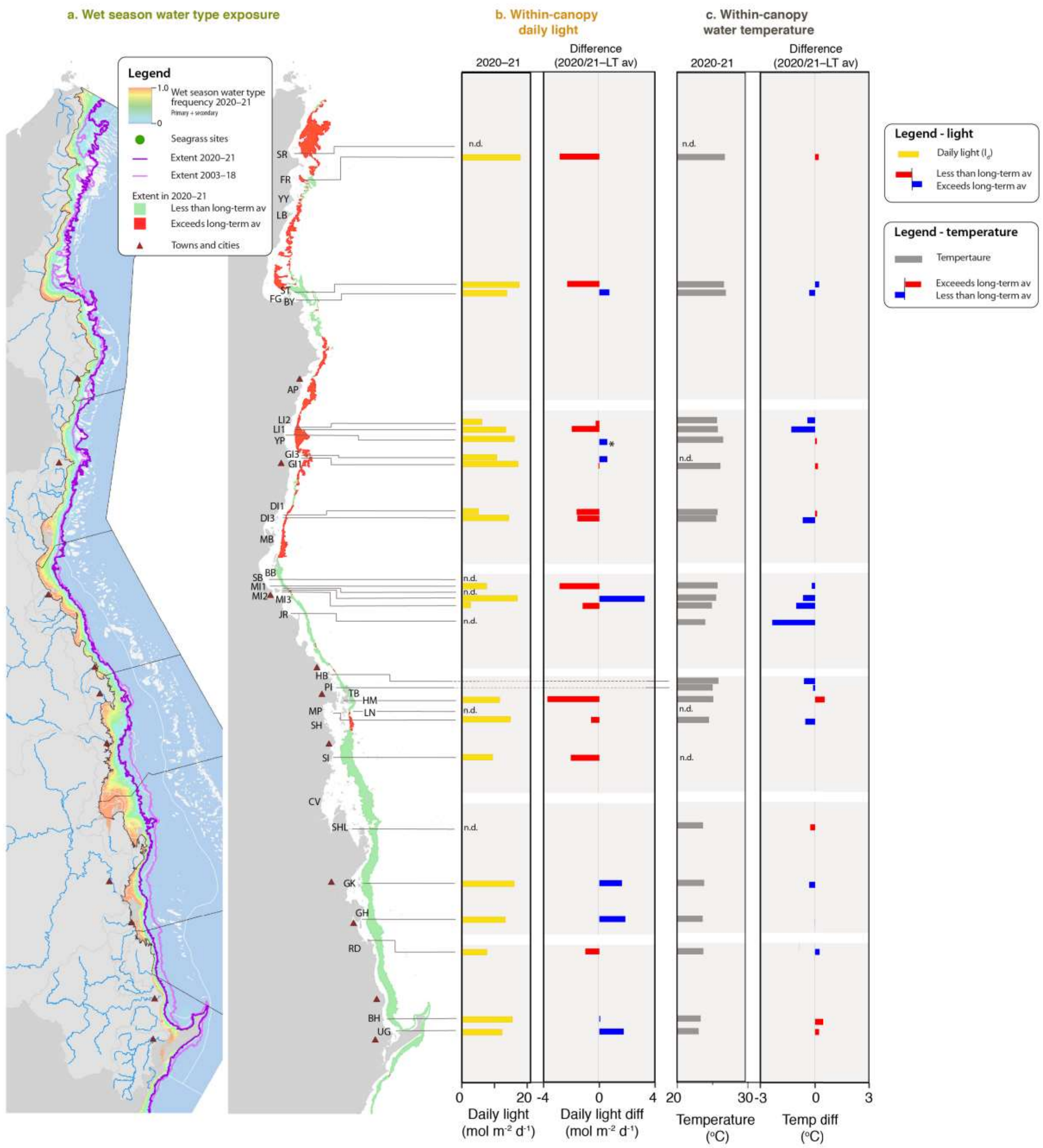


Figure 8. Environmental pressures in the Reef during 2020–21 and relative to long-term: a. Frequency of turbid water (primary and secondary water) exposure shown in the left-hand panel in the Reef from December 2020 to April 2021 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed), and right-hand panel the distribution of primary and secondary waters (10 per cent boundary) in 2020–21 relative to the long-term average, with red showing that that these water types extended further in 2020–21 and green showing they did not extend as far; b. within canopy daily light (shown as I_d) 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. Panels a and b from Moran *et al*. (2022).

Turbid coloured water (primary or secondary) reached all seagrass locations in 2020–21 as is characteristic of inshore conditions over the long-term (2003–2019, Figure 8). Secondary water extended considerably further than average in throughout the southern Reef, but less than average in the northern Reef (Figure 8, panel 2).

The frequency of exposure to primary water types during the wet season weeks (December 2019–April 2020) is typically very high in the inshore regions of the Reef. It was below multiannual conditions in all regions, with the largest differences occurring in the Fitzroy and Burnett-Mary regions (Figure 9). This indicates a lower level of exposure to water with high levels of plankton and fine-sediment. The sites exposed to higher frequency of primary water in the region were all coastal or estuarine. The frequency of exposure to both primary and secondary water, shows that all regions were slightly below the multiannual level of exposure, with the largest change occurring in Cape York (Figure 8). The optical water type classification changed to the Sentinel Forel-ule colour scale in 2020–21, as detailed in Moran *et al*. (2022) and Petus *et al*. (2019).

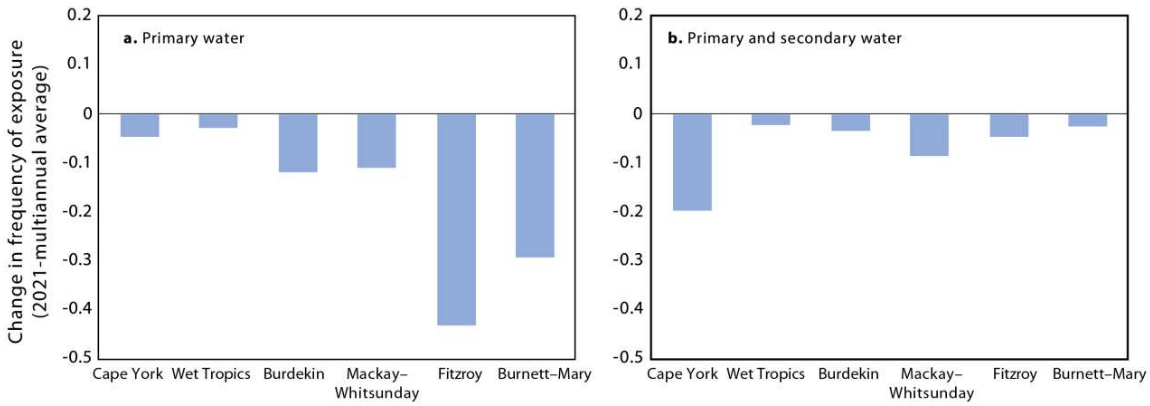


Figure 9. Difference in the frequency of exposure to primary (left) and primary and secondary optical water types (right) at seagrass monitoring sites during the wet season (December 2020–April 2021) compared to the long-term multiannual exposure (2003–2018).

3.5 Daily 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 daily light among seagrass meadows reported here are largely a reflection of site-specific differences in water quality, except 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 2020–21 was 12.5 mol m^{-2} d^{-1} when averaged for all sites (Table 9), compared to a long-term average of 12.4 mol m^{-2} d^{-1}. At almost half of the locations where light is monitored, daily light was lower than the long-term average, and these were in each region except the Fitzroy (Figure 8). There are regional, habitat and location levels differences.

Daily light in the regions in 2020–21 from north to south were (↓ = lower than, ↑ = greater than the long-term, ↕ = similar to long-term i.e. <0.5 mol m^{-2} d^{-1} difference):

- Cape York (16.5 mol m^{-2} d^{-1}) ↕
- northern Wet Tropics (12.7 mol m^{-2} d^{-1}) ↕
- southern Wet Tropics (10.6 mol m^{-2} d^{-1}) ↕
- Burdekin (9.9 mol m^{-2} d^{-1}) ↕
- Mackay–Whitsunday (10.8 mol m^{-2} d^{-1}) ↓
- Fitzroy (16.0 mol m^{-2} d^{-1}) ↓
- Burnett–Mary (12.0 mol m^{-2} d^{-1}) ↕

Daily light in the habitats in 2020–21 from highest to lowest were (↓ = lower than, ↑ = greater than the long-term):

- reef intertidal, n = 9 (15.0 mol m^{-2} d^{-1})↓
- coastal intertidal, n = 10 (14.0 mol m^{-2} d^{-1})↑
- estuarine, n = 3 (10.6 mol m^{-2} d^{-1})↓
- reef subtidal, n = 5 (5.6 mol m^{-2} d^{-1})↓.

Daily light for each of the sites is presented in Figure 8. There were ten locations in which the annual daily light level was lower than 10 mol m^{-2} d^{-1}, the light threshold that is likely to support optimal long-term growth requirements of the species in these habitats (Collier *et al.* 2016a). Three of these were subtidal sites (except Green Island). The other locations below 10 mol m^{-2} d^{-1} were estuarine or coastal intertidal locations at Bushland Beach and Shelley Beach in the Burdekin, Lindeman Island and Sarina Inlet in Mackay-Whitsunday and Rodds Bay in the Burnett–Mary.

Long-term trends show a peak in within canopy daily light that occurs from September to December, as incident solar irradiation reaches its maximum and prior to wet season conditions (Figure 10). This also coincides with the peak seagrass growth season, and the predominant sampling period in this program. The lowest light levels typically occur in the wet season, particularly in January to July. In 2019–20, daily light steadily increased from post-wet season minima to a peak in late November and early December and declined sharply in December thereafter. This followed an extended period of low light that was below the wet season average.

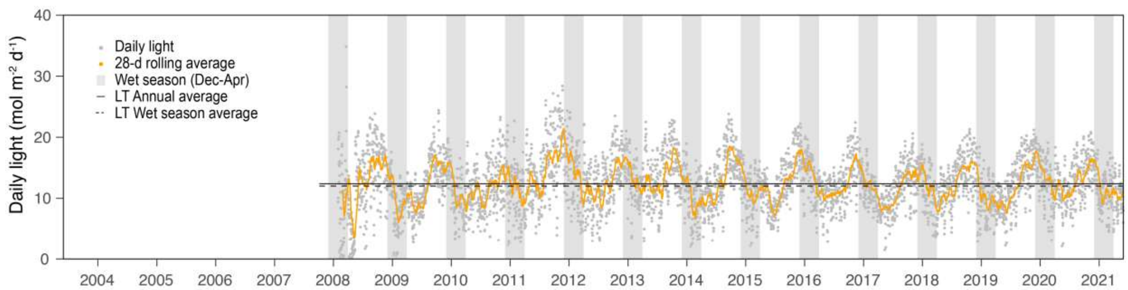


Figure 10. Daily light for all sites combined from 2008 to 2021. In 2008–2009, light data is from the Burdekin and Wet Tropics regions only. Other regions were included from 2009–2010, with Cape York added post 2012–2013 reporting period. Shaded vertical bars indicate the wet season months (December to April) used for analysis of wet season optical water types Moran *et al.* (2022). The solid horizontal line indicates the long-term Reef average, and the dashed line indicates the wet season long-term Reef average.

3.6 Within-canopy seawater temperature

Daily within-canopy seawater temperature across the inshore Reef in 2020–21 was similar to the previous reporting period (Figure 11). Since 2013, the frequency of weekly warm water deviations appears to have increased, relative to cooler occurrences (Figure 11). The 2020–21 temperature was on average (25.9 ±0.2°C) similar to the long-term (2003–20, 25.7°C) (Table 9). However, there were regional and habitat differences relative to the long-term (Figure 8).

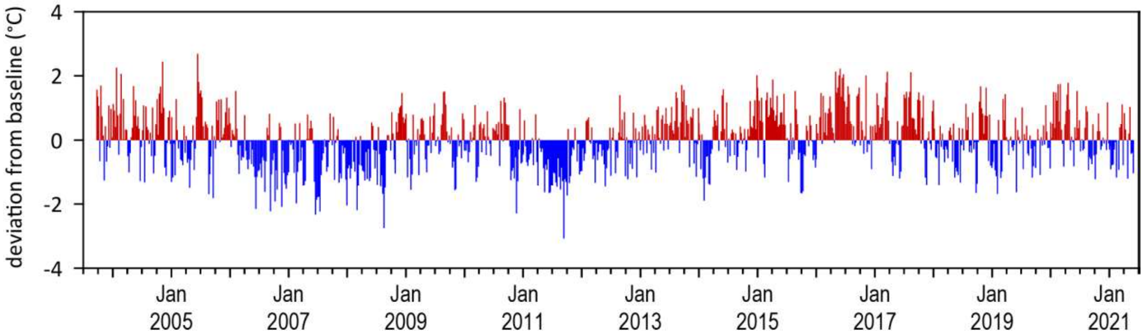


Figure 11. Inshore intertidal sea temperature deviations from baseline for Reef seagrass habitats from 2003 to 2021. Data presented are deviations from 16-year mean weekly temperature records (based on records from September 2003 to June 2020). Weeks above the long-term average are represented as red 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.

Daily within-canopy seawater temperatures in the regions in 2020–21 (including number of days above 35°C and 40°C) from north to south as difference (greater than 0.5°C) relative to the long-term average (↑ = above, ↓ = below, ↕ = similar to long-term) were:

- Cape York (avg = 27.7°C, max = 38.1°C, $days_{>35°C}$ = 40)↑
- northern Wet Tropics (avg = 27.1°C, max = 39.6°C, $days_{>35°C}$ = 63)↕
- southern Wet Tropics (avg = 26.3°C, max = 34.0°C, $days_{>35°C}$ =0)↓
- Burdekin (avg = 25.6°C, max = 38.2°C, $days_{>35°C}$ =13)↓
- Mackay–Whitsunday (avg = 25.5°C, max = 40.3°C, $days_{>35≤40°C}$ =40, $days_{>40°C}$ =1)↕
- Fitzroy (avg = 24.2°C, max = 41.9°C, $days_{>35≤40°C}$ =42, $days_{>40°C}$ =2)↕
- Burnett–Mary (avg = 23.7°C, max = 39.0°C, $days_{>35°C}$ =3)↕

Daily within-canopy seawater temperatures in each habitat in 2020–21 relative to respective long-term average (↑ = above, ↓ = below, ↕ = similar to long-term, difference = greater than 0.2°C) were:

- estuarine habitat (avg = 24.0°C, max = 36.9°C)↕
- coastal intertidal habitat (avg = 25.8°C, max = 41.2°C)↓
- reef intertidal habitat (avg = 26.3°C, max = 41.9°C)↕

The hottest seawater temperature recorded at inshore seagrass sites along the Reef during 2020–21 was 41.9°C in the Fitzroy region, and only the southern regions (Mackay–Whitsunday, and Fitzroy), had at least one day above 40°C (Figure 12). Extreme temperature days (>40°C) can cause photoinhibition but when occurring at such low frequency, they were unlikely to cause burning or mortality.

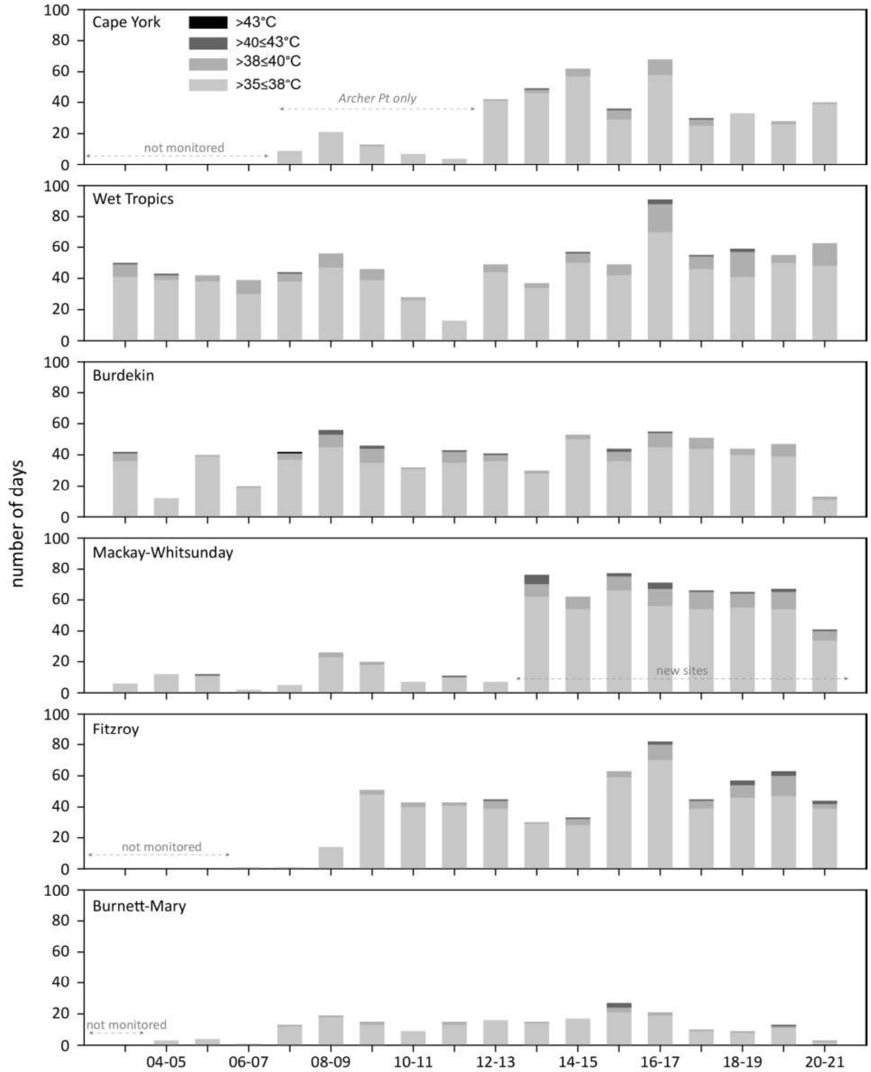


Figure 12. Number of days when inshore intertidal sea temperature exceeded 35°C, 38°C, 40°C and 43°C in each monitoring period in each NRM region.Thresholds adapted from Campbell *et al*. 2006; Collier *et al*. 2012a.

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 11). Sediments at intertidal coastal habitats were predominately medium and fine sands, while reef habitats (intertidal and subtidal) were dominated by medium sands (Table 11).

Table 11. Long-term average (±SE) sediment composition for each seagrass habitat (pooled across regions and time) monitoring within the Reef (1999–2020). *only 5 years of data.

Habitat	Mud	Fine sand	Sand	Coarse sand	Gravel
estuarine intertidal	45.1 ±2.1	22.5 ±2.1	30.3 ±1.8	0.1 ±0.4	2.0 ±0.9
coastal intertidal	28.0 ±2.1	30.6 ±2.4	37.0 ±2.5	0.4 ±0.6	4.0 ±1.2
coastal subtidal*	52.8 ±2.3	9.6 ±0.4	18.7 ±2.3	6.7 ±1.0	12.3 ±1.1
reef intertidal	4.3 ±1.2	6.8 ±1.8	52.6 ±2.8	15.0 ±1.9	21.3 ±2.4
reef subtidal	12.5 ±0.9	16.5 ±1.1	57.8 ±5.7	1.3 ±0.5	11.9 ±5.7

During the 2020–21 monitoring period there were small fluctuations within intertidal habitats, in the contribution of mud sediments to sediment type relative to the previous year (Figure 13). In subtidal habitats, the contribution of mud sediments increased above the long-term average (Figure 13). Historically, the composition of sediments has fluctuated at all habitats, with the proportion of mud declining below the long-term average at estuarine and coastal habitats immediately following periods of physical disturbance from storms when seagrass cover greatly declines (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).

Finer-textured sediments (i.e. mud) tend to have higher nutrient concentrations 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, providing sufficient light for photosynthesis is available.

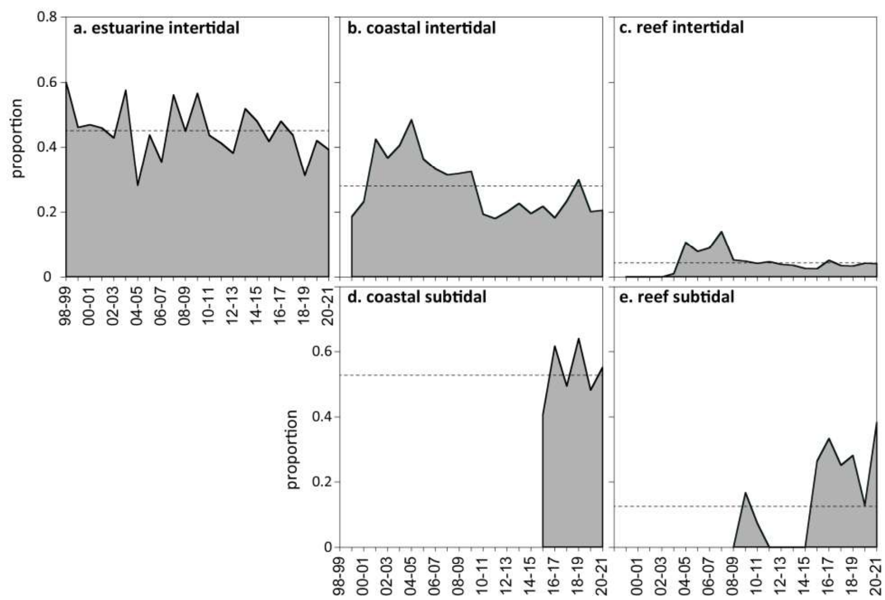


Figure 13. Proportion of sediment composed of mud (grain size <63µm) at inshore Reef seagrass monitoring habitats from 1999–2021. Dashed line illustrates the Reef long-term average for each habitat type.

4 Seagrass condition and trend

The following results section provides detail on the overall seagrass responses for the 2020–21 monitoring period, in context of longer-term trends. It is structured as an overall inshore Reef summary with condition and trend for each habitat type presented separately, including:

- a summary of the key findings from the overall section including a summary of the report card score
- seagrass abundance (per cent cover) and spatial extent
- seagrass species composition based on life history traits
- seagrass reproductive effort and seed banks
- 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 Appendix 3 and 4.

4.1 Overall inshore Reef seagrass condition and trend

Inshore seagrass meadows across the Reef improved in overall condition in 2020–21, with the condition grade changing from poor to **moderate** (Figure 14).

In summary, the improvement was due to small increases in both seagrass abundance and resilience indicators:

- The seagrass abundance indicator increased in 2020–21 after reaching a seven-year low in 2019–20. Seagrass abundance at meadows monitored in the MMP declined from 2005–2006 until 2011–2012, caused by multiple years of above-average rainfall, and resultant discharges of poor quality water, followed by extreme weather events, after which abundance increased (Figure 14, Figure 16b). Seagrass abundance subsequently increased until 2015–16, after which it declined until 2020–21 when it increased again. Based on the average score against the seagrass guidelines (determined at the site level), the abundance of inshore seagrass in the Reef over the 2020–21 were in a moderate condition (Figure 14).

- The resilience indicator was introduced in 2020–21 and back-dated to the start of the program. Resilience increased to moderate in 2020–21 after reaching a seven-year low in 2019–20 (Figure 14). Although the slight uptick in 2020–21 may suggest seagrass habitats are possibly on a recovering trajectory following a seven-year low in 2019–20, seagrass in some regions remain vulnerable to further disturbances.

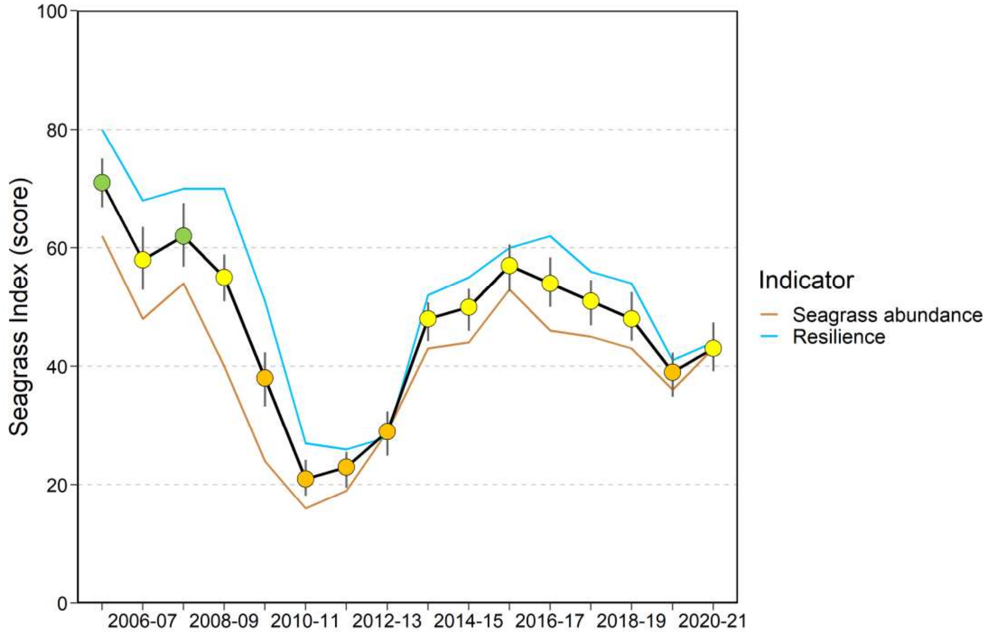


Figure 14. Overall inshore Reef seagrass condition index (± SE) with contributing indicator scores over the life of the MMP. The index is derived from the aggregate of metric scores for indicators of seagrass condition: abundance and resilience. Index scores scaled from 0–100 and graded: ● = very good (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 overall inshore Reef score for seagrass is derived from the average of seagrass indicator scores in each of the six NRM regions, weighted by inshore seagrass area. In 2020–21 the score improved in the northern regions (Cape York, Wet Tropics and Burdekin), but decreased in the southern regions (Mackay-Whitsunday, Fitzroy and Burnett-Mary) (Figure 15). Over the long term, the indicators tend to diverge during periods of elevated disturbance and loss, but converge and follow a similar trend during periods of low disturbance. These patterns and trends in the indicators are more apparent at the regional scale, with the variation among the six regions:

- The seagrass abundance score was moderate in all northern regions, but poor in all southern regions (Figure 15). The score increased in the 2020–21 monitoring period in the northern regions compared to the previous monitoring period, but remained relatively unchanged in the southern regions. The largest changes to the abundance score have occurred in the Burdekin region, which reached a good rating in 2015–16, but declined to poor in 2019–20, before improving back to moderate in 2020–21. The Fitzroy region has not achieved a rating greater than poor since 2010–11.
- The seagrass resilience scores were moderate in all regions except Mackay–Whitsunday and Fitzroy, where the scores were poor (Figure 15). In 2020–21, resilience declined in the Mackay–Whitsunday and Fitzroy regions and increased in Cape York, compared to the previous monitoring period (Figure 15). The resilience grade was unchanged in the other regions, however the score declined in the Burnett-Mary region.

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 2016–17 (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 in 2018–19 in the Burdekin region (above-average riverine discharge).

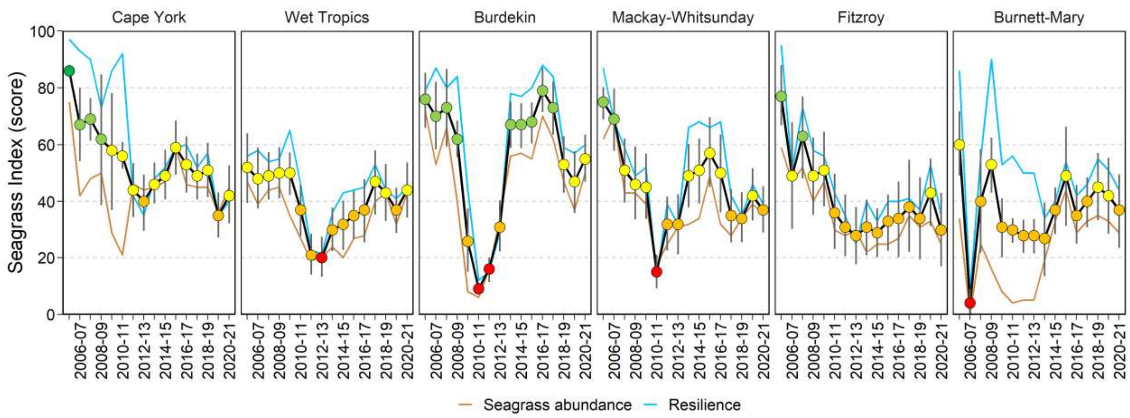


Figure 15. Seagrass condition index (± SE) with contributing indicator scores for each NRM region over the life of the MMP. The index is derived from the aggregate of metric scores for indicators of seagrass condition: abundance and resilience. Values are indexed scores scaled from 0–100 and graded: ● = very good (81–100), ● = good (61–80), ● = moderate (41–60), ● = poor (21–40), ● = very poor (0–20). NB: Scores are unitless.

The long-term trends for each of the contributing indicators used to calculated the Seagrass Index are shown in Figure 16. Results from the generalised additive models are presented for per cent cover to show long-term trends. Seagrass abundance has varied over decadal time-scales, declining in the 2009–10 through 2011–12 monitoring periods, then recovering to some extent depending on region, and subsequently declining over recent years. The resilience indicator score has similarly declined to its lowest levels in the 2010–11 through 2012–13 monitoring periods. The resilience score increased in 2020–21.

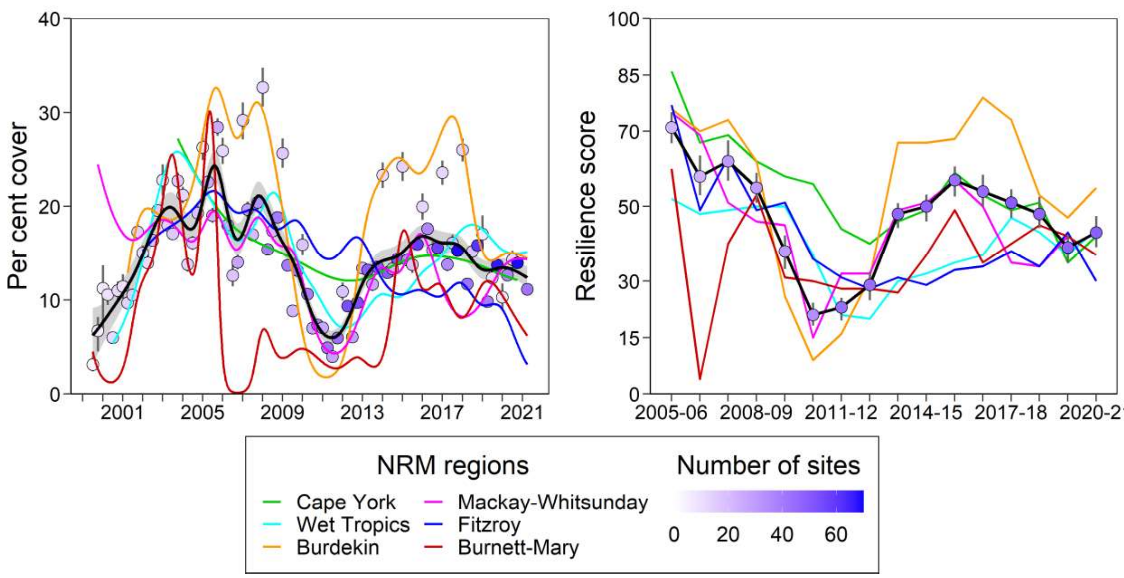


Figure 16. Trends in the seagrass indicators used to calculate the condition index including trends in Reef seagrass abundance (per cent cover, ± SE) represented by a GAM plot (black line with shaded areas defining 95 per cent confidence interval), and coloured lines representing NRM trends (left), and trends in Reef resilience score (black line and circles, ± SE) and coloured lines represent trends in NRM resilience scores (right). Circle colour relates to number of sites assessed. Please note: Reef resilience scores are weighted.

4.3 Trends in seagrass condition indicators by habitat type

4.3.1 Seagrass abundance, composition and extent

Seagrass abundance has fluctuated since monitoring was established. An examination of long-term abundances at Reef sites indicates no significant trend overall, with:

- no significant trends at 70 per cent of long-term monitoring sites assessed, although 10 per cent of sites significantly increased in abundance and 21 per cent decreased (Appendix 3, Table 21)
- the rate of change in abundance was higher at sites increasing (0.6 ±0.3 per cent, sampling event^{-1}) than decreasing (-0.2 ±0.1 per cent sampling event^{-1}) (Appendix 3, Table 21)
- the most variable seagrass habitat in abundance (since 2005) was estuarine intertidal (CV=109.4 per cent), followed by reef habitats (intertidal CV=80.6 per cent and subtidal CV=106.7 per cent), and lastly, coastal habitats (intertidal CV=67.8 per cent and subtidal CV=48.7 per cent).

Since 1999, the median percentage cover values for the Reef were mostly below 25 per cent cover, and depending on habitat, the 75th percentile occasionally extended beyond 50 per cent cover (Figure 17). 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 per cent (Lee Long *et al.* 1993). The findings highlight the need to use locally-relevant reference sites and score thresholds.

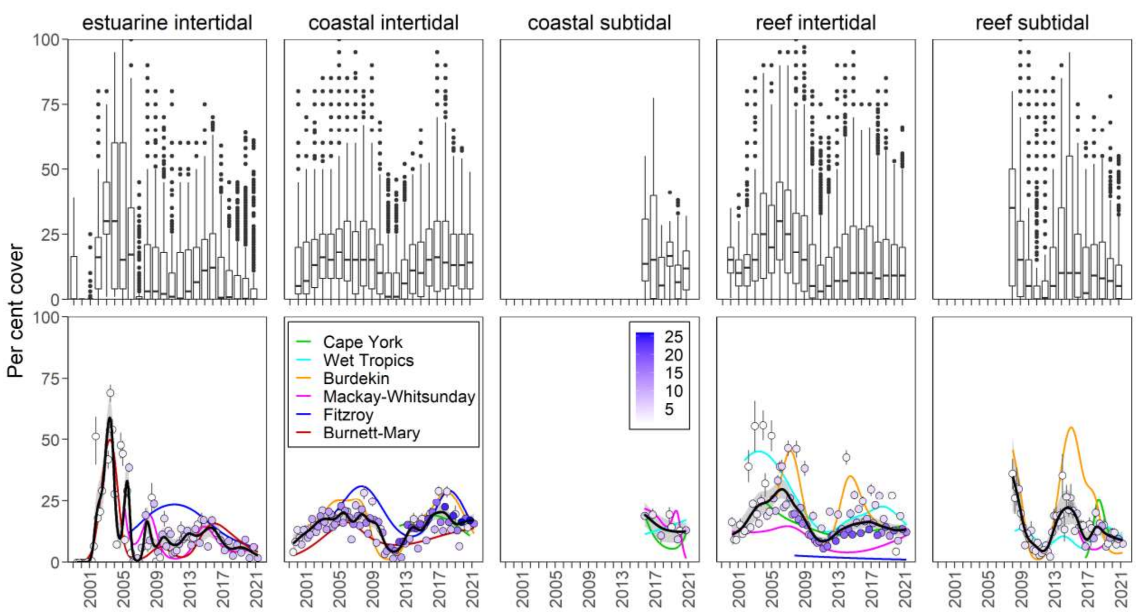


Figure 17. Seagrass per cent cover measures per quadrat from habitats monitored from June 1999 to May 2021 (sites pooled). In the whisker plots (top), the box represents the interquartile range of values, where the boundary of the box closest to zero indicates the 25th percentile, a line within the box 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), showing trends for each NRM (coloured lines) and combined as dark lines with shaded areas defining 95th confidence intervals of those trends. Colour of circles represents the number of sites assessed to calculate the average, and vertical error bars represent standard error.

In 2020–21, coastal sites had the highest average abundance of the habitat types, and estuarine sites had the lowest (Figure 17). Over the past decade, the patterns of seagrass abundance in each habitat have been similar between intertidal sites in coastal and reef habitats; gradually increasing from 2001 to 2008 (with a mild depression in coastal habitats 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 NRM regions (Mackay–Whitsunday, Fitzroy and Burnett–Mary), reached record per cent cover in 2002 to 2003, but have remained low since 2005–06. Trends have fluctuated at a location level in estuarine habitats, most often at smaller localised scales where there have been some acute event related changes, e.g. sediment deposition and/or reduced light availability due to discharge events, or sediment movement due to climatic pressures.

Following 2011, seagrass abundance has progressively improved, although most still remained below the 2005 levels on average in each year since, with the exception of coastal meadows, which have recovered (Figure 17).

In 2020–21, the overall inshore Reef relative meadow spatial extent was similar to the previous year, however these remain lower than the baseline (2005), 2014–15 and 2015–16 (Figure 18).

Since the MMP was established in 2005, meadow extent across inshore monitoring sites declined in early 2011, recovering within 3–4 years (Figure 18). 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 four years appear as a consequence of severe weather events (e.g. cyclones) and location specific climate (frequency of strong wind days).

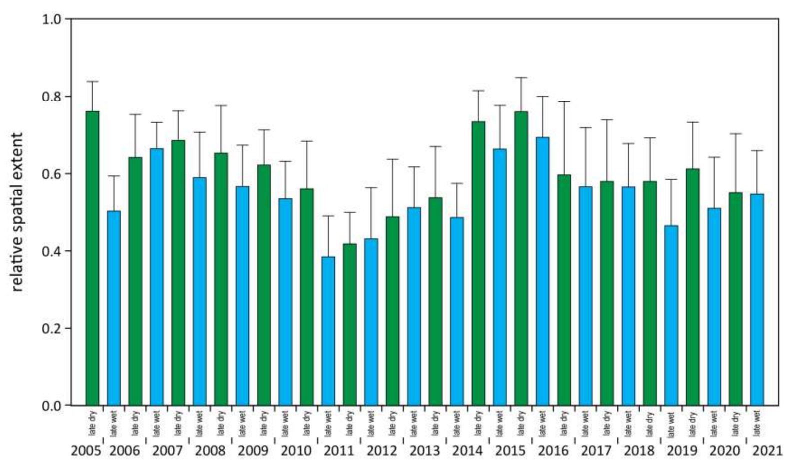


Figure 18. Average relative spatial extent of seagrass distribution at monitoring sites across inshore Reef (locations, habitats and NRM regions pooled, + SE).

After the extreme weather events in 2009 to 2011 that caused widespread declines in seagrass extent (Figure 18) and abundance, there was increasing proliferation of species displaying colonising traits, such as *H. ovalis,* at coastal and reef sites (Figure 19). Over the 2020–21 monitoring period, the proportion of species displaying colonising traits remained around or lower than the overall inshore Reef 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. At reef subtidal habitats, the proportion of colonising species was the second highest since 2012–13.

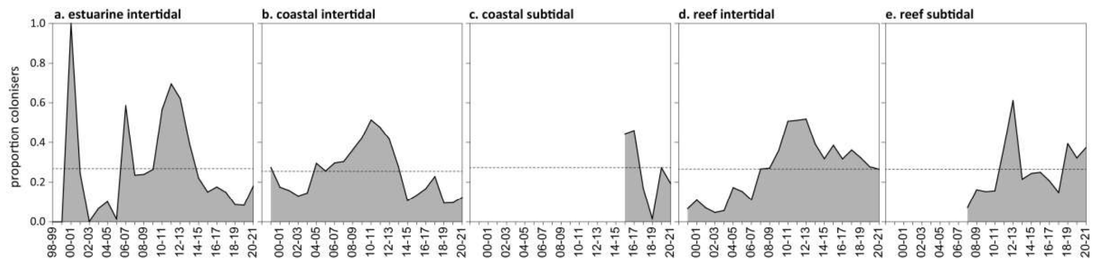


Figure 19. Proportion of total seagrass abundance composed of species displaying colonising traits (e.g. *Halophila ovalis*) in: a) estuarine intertidal, b) coastal intertidal, c) coastal subtidal, d) reef intertidal and e) reef subtidal habitats (sites pooled) of the Reef (regions pooled) for each monitoring 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 in 2020–21, with the lowest levels since 2013–14 (Figure 20). Reproductive effort also continued to decline in estuarine habitats for the second year in a row, whereas coastal habitats were the only habitats with more reproductive structures than the previous period.

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. Reproductive effort across the inshore Reef meadows are typically higher in the late dry season, while seed density fluctuates less seasonally (Figure 20, Figure 21).

Reproductive effort had gradually been increasing at estuarine and coastal habitats since 2011, with large rises from 2013–14.However, it decreased significantly in estuaries in 2018–19 and continued to remain low in 2020–21 (Figure 20). This trend was observed in all three southern regions where estuaries are monitored and reflects trends in abundance in estuarine habitats. Seed banks remained largely unchanged over the previous 9 years in estuaries (Figure 21).

In coastal habitats, reproductive effort and seed density varies inter-annually, more than in other habitats. The historically high reproductive effort in coastal habitats is due to a record number of reproductive structures in the northern Wet Tropics (Yule Point) and Burdekin (Bushland Beach and Jerona). Overall inshore Reef reproductive effort improved slightly in 2020–21 with increases occurring across a quarter of sites, with largest improvements in the northern Wet Tropics and Burdekin regions (Figure 20). Seed densities in seed banks also improved in coastal habitats (Figure 21).

Reef habitats have had the lowest reproductive effort of all habitats (Figure 20), while seed density in seed banks have typically been the lowest in reef intertidal habitats. In 2020–21, reproductive effort remained low across reef habitats, but there were some minor increases in the Burdekin and Fitzroy intertidal meadows and a subtidal meadow in the northern Wet Tropics (Green Island). No seeds have ever been observed at over half of the reef sites (intertidal or subtidal), including sites in Cape York, northern Wet Tropics, Mackay–Whitsunday and Fitzroy regions. During 2020–21, persistent seed banks declined at 29 per cent of sites, and the only site where the seed bank increased was in the intertidal meadow at Dunk Island (southern Wet Tropics).

Reductions in seed density could have been caused by reduced reproductive success (failure to form seeds) or loss of seed bank (germination or grazing). This indicates vulnerability of these habitats to future disturbances, as recovery may be hampered although the actual count of seeds needed to initiate or optimise recovery is unknown.

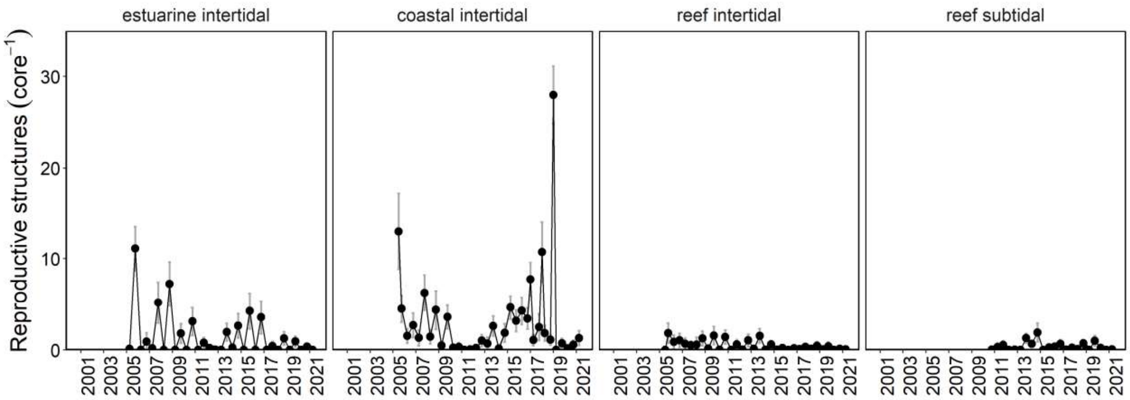


Figure 20. Seagrass reproductive effort (number of reproductive structures produced by all seagrass species, ± SE) in Reef seagrass habitats for a) estuarine intertidal; b) coastal intertidal; c) reef intertidal; d) reef subtidal.

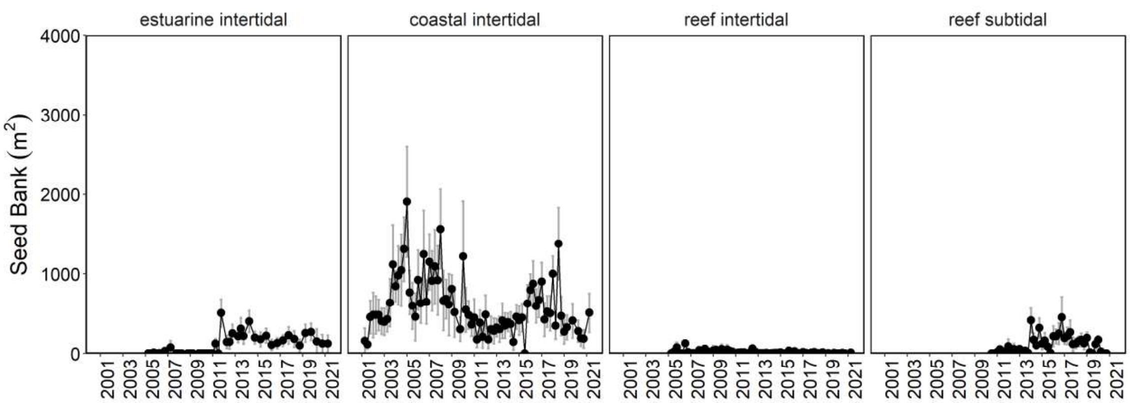


Figure 21. Average seeds banks (seeds per square metre of sediment surface, all sites and species pooled, ± SE) in Reef seagrass habitats: a) estuarine intertidal; b) coastal intertidal; c) reef intertidal; d) reef subtidal.

4.3.2.1 Resilience

Resilience declined and was the lowest among habitats at estuarine sites (Figure 22, Table 22), where most sites were in poor condition indicating low levels of resistance to disturbance. The resilience score in 2020–21 in estuarine habitats was the lowest in the history of the program. Only one estuarine site had reproductive structures in 2020–21 but a further two had recent history (≤3years) of reproductive effort. Coastal intertidal habitats had the highest and improving levels of resilience in 2020–21 (Figure 22, Table 22). The majority of sites were in good condition exceeding thresholds indicative of low resistance and had reproductive structures present in 2020–21 or in recent years. There were a few coastal sites in Cape York and one in the Wet Tropics that were in poor condition (high proportion of colonising species and/or very low abundance) and reducing the resilience score for coastal habitats.

The resilience score was stable in reef intertidal habitats (Figure 22, Table 22). Most reef intertidal sites exceeded condition thresholds and therefore were not in the low resistance category (category 1). Only one site had reproductive structures in 2020–21 but a further six sites had recent history of reproduction and were scored with a 50 or more. The resilience score declined slightly at reef subtidal sites and they had the second lowest resilience grade overall among habitat types (Figure 22, Table 22). The majority of sites exceeded condition thresholds and therefore were not in the low resistance category. Nevertheless, only one site had reproductive structures in 2020–21 (Green Island, Wet Tropics) and one had recent history of reproduction (Lindeman Island, Mackay–Whitsunday).

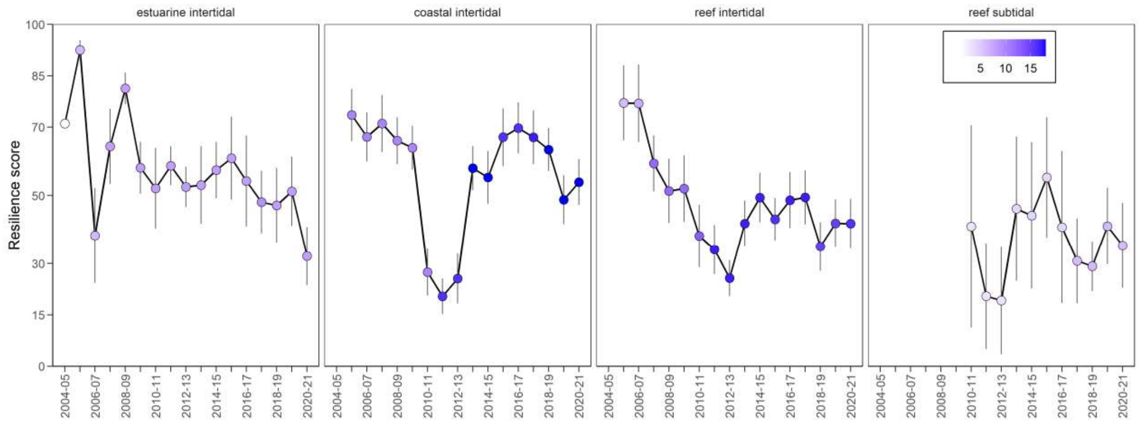


Figure 22. Resilience score summarised for each habitat type of the Reef. Blue shading of points indicates the number of sites contributing to the score. Vertical error bars represent standard error.

Resilience was the highest in the Burdekin region where meadow condition exceeded critical thresholds indicating resistant meadows. Reproductive structures were also present at most sites, indicating recovery potential, but they were present in low numbers compared to historical records. Overall resilience was similar in other regions but this varied with habitat type.

4.3.3 Epiphytes and macroalgae

Epiphyte cover on seagrass leaves during 2020–21 was below the overall inshore Reef long-term average at estuarine intertidal, coastal intertidal and reef subtidal habitats, and seasonally variable in reef intertidal habitats (Figure 23). Epiphytes historically varied the most in estuarine habitats (by 50 per cent).Over the previous 10 years, epiphytes have mostly varied by a small amount (<20 per cent) around the long-term average in both coasts and reef.

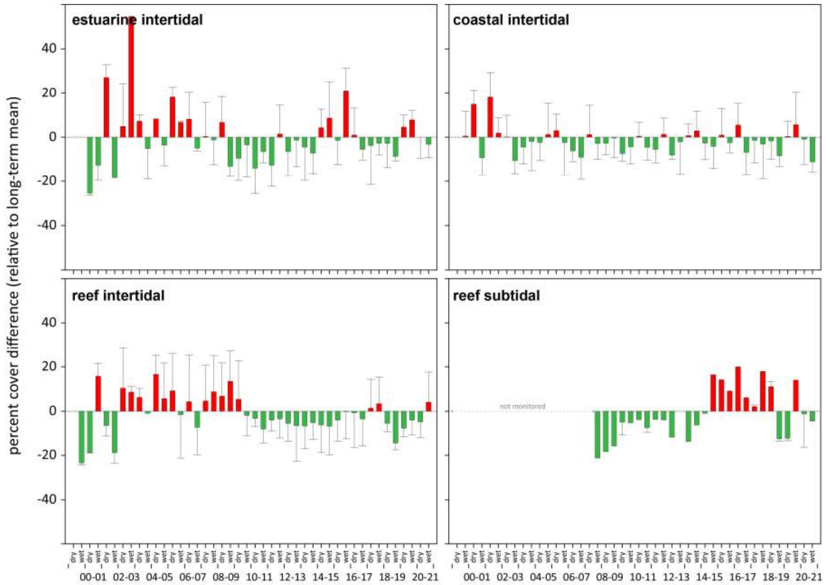


Figure 23. Epiphyte abundance (per cent cover) relative to the long-term average (the zero axis) for each Reef seagrass habitat (sites pooled, ± SE). Reef long-term average (2005 to 2020); estuarine = 25.3 ±5.7 per cent, coastal intertidal = 17.5 ±3.6 per cent, reef intertidal = 22.1 ±4.1 per cent, reef subtidal = 20.0 ±3.0 per cent.

Macroalgae abundance in 2020–21 followed the general trends of the previous 10 years in estuarine and coastal habitats, remaining below the overall inshore Reef long-term average for each of the habitats (Figure 24). Macroalgae abundance remained above the long-term average at reef intertidal sites, in particular at Magnetic Island (MI2), Low Isles (LI1) and Hydeaway Bay (HB1). In contrast, macroalgal abundance at reef subtidal sites slightly increased, negating a declining trend which had occurred over the last few years, and ending the period marginally above the long-term average.

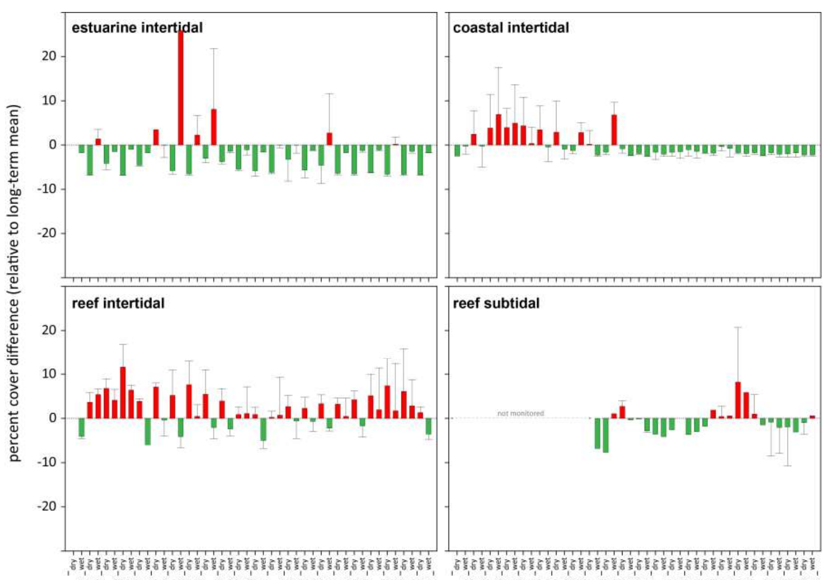


Figure 24. Macroalgae abundance (per cent cover) relative to the long-term average for each inshore Reef seagrass habitat. (sites pooled, ± SE). Reef long-term average; estuarine = 2.0 ±1.0 per cent, coastal intertidal = 2.3 ±1.2 per cent, reef intertidal = 7.0 ±1.9 per cent, reef subtidal = 6.7 ±2.0 per cent.

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 at each monitoring location
- sediment grain size composition at each monitoring site
- tables detailing statistical analysis.

5.1 Cape York

5.1.1 2020–21 Summary

The region experienced above average annual rainfall and river discharge yet below average turbid water exposure and average daily light levels. There were above average within-canopy water temperatures for the ninth consecutive year.

Seagrass condition is assessed only in the late dry in Cape York, before the wet season when the elevated rainfall and river discharge occurs. Seagrass meadow condition across the Cape York NRM region in 2020–21 increased to moderate, from the poor grade in 2019–20. The increase was due to higher scores in both the abundance and resilience indicators. For the indicators:

- abundance score was moderate
- resilience score was moderate.

On average, seagrass abundance (per cent cover) increased relative to the previous period. Seagrass abundance increased at half of the Cape York sites, in all but intertidal reef meadows, where the greatest losses occurred.

The resilience score was moderate overall. Low scores occurred at Shellburne Bay and at one site at Piper Reef where abundances were below thresholds and indicative of low resilience, but were moderate to high at other sites. Reproductive structures continue to be rarely observed in Cape York in 2020–21 for the second consecutive year, which may hinder replenishment of the declining seed banks and weaken capacity to recover from seeds in the near future.

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. Per cent cover progressively decreased at intertidal reef habitats across Cape York from 2003 to 2012, with signs of improvement since, particularly at Stanley Island. Coastal intertidal and subtidal habitats monitored since 2012 and 2015 respectively, improved in 2020–21, but minor declines were observed in the north of Cape York. Meadow relative extent across the region continues to remain relatively stable across the region.

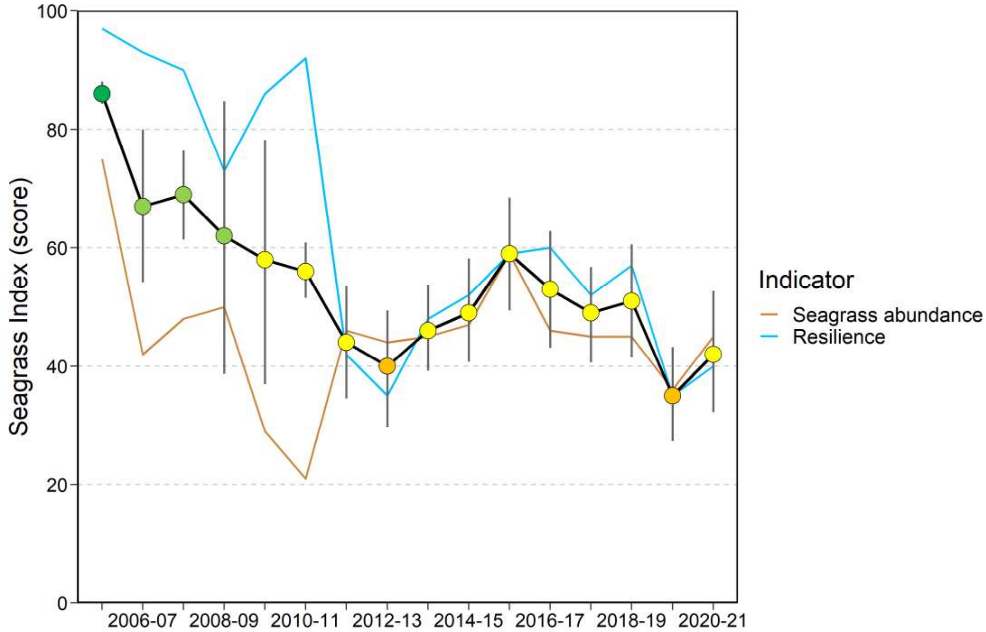


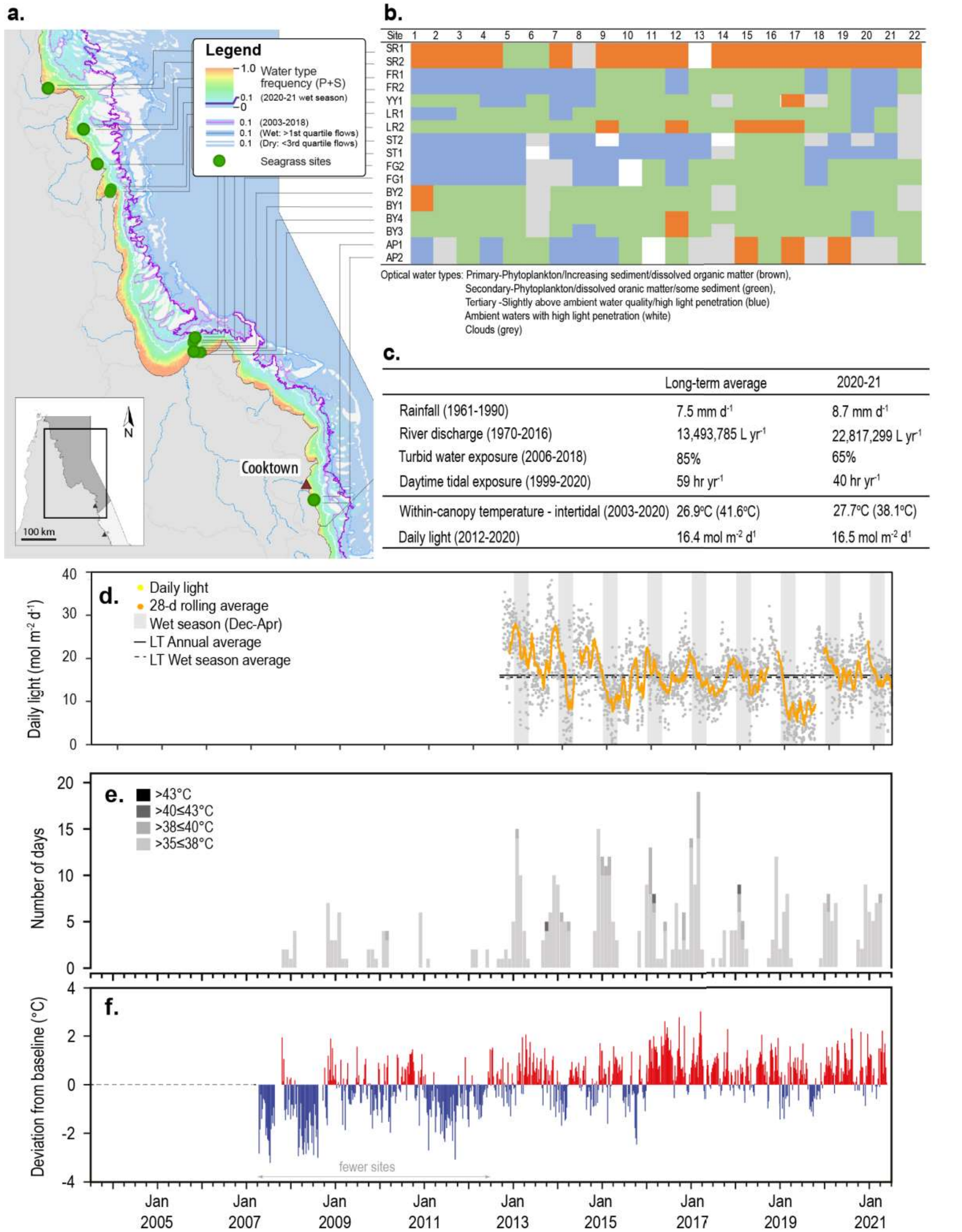
Figure 25. Temporal trend in seagrass condition index (± SE) with contributing indicator scores for the Cape York NRM region (averaged across habitats and sites). Index scores scaled from 0–100 and graded: ● = very good (81–100), ● = good (61–80), ● = moderate (41–60), ● = poor (21–40), ● = very poor (0–20). NB: Scores are unitless.

5.1.2 Climate and environmental pressures

There were no tropical cyclones that directly affected the Cape York region in the 2020–21 wet season. Tropical cyclone Kimi formed in the Coral Sea outside of Cape York on the 16th of January 2021, and crossed into the Reef in the Wet Tropics region. A tropical low passed through Cape York just north of Princess Charlotte Bay on January 29th 2021, and formed into tropical cyclone Lucas when well offshore (Moran *et al.* 2022). Rainfall was slightly above the long-term average in Cape York in 2020–21, while river discharge exceeded the long-term average by more than 1.5 times for the region as a whole. Discharge from the Olive–Pascoe, Lockhart and Stewart Rivers in central Cape York, which likely influence Piper Reef and Shelburne Bay, were more than twice the long-term average (Table 10).

Exposure to primary and secondary water types was below the long-term average in Cape York. The frequency of exposure ranged from 15 per cent to 100 per cent of wet season weeks at seagrass monitoring sites (Figure 26a) (Figure 8 and Figure 9). The inshore waters of Cape York had predominantly secondary water type, and some tertiary influence over the wet season in December-April (Figure 26b). Shelburne Bay sites (SR1 and SR2) had the highest exposure to turbid primary water, consistent with previous years. Reef habitats (Piper Reef FR, Stanley Island ST and Flinders Group, FG) had the lowest level of exposure to primary or seconddary water amongst the inshore seagrass monitoring sites.

Daily light (mol m^{-2} d^{-1}) reaching the top of the seagrass canopy is generally very high at all Cape York sites, largely because it is measured only at intertidal sites (long-term average = 16.4 mol m^{-2} d^{-1}) (Figure 100). In 2020–21, daily light (16.5 mol m^{-2} d^{-1}) was around the long-term average (Figure 26d). Cape York sites are surveyed only once per year, and the instruments are not able to function for a full year due to battery life, and inevitable fouling.



	Long-term average	2020-21
Rainfall (1961-1990)	7.5 mm d^{-1}	8.7 mm d^{-1}
River discharge (1970-2016)	13,493,785 L yr^{-1}	22,817,299 L yr^{-1}
Turbid water exposure (2006-2018)	85%	65%
Daytime tidal exposure (1999-2020)	59 hr yr^{-1}	40 hr yr^{-1}
Within-canopy temperature - intertidal (2003-2020)	26.9°C (41.6°C)	27.7°C (38.1°C)
Daily light (2012-2020)	16.4 mol m^{-2} d^{-1}	16.5 mol m^{-2} d^{-1}

Figure 26. Environmental pressures in the Cape York region including: a. frequency of exposure to primary and seconday water from December 2020 to April 2021 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed) (white = no data), also showing the long-term average (2003–2018) exposure boundary (purple line), and the first (blue line) and third quartile (white line) of the long-term average (from Moran *et al.* 2022), b. wet season water type at each site; c. average conditions and max temperature over the long-term and in 2020–21; 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 at intertidal sites.

Notably, 2020–21 was the second warmest year of intertidal within-canopy temperatures since monitoring was established in the region; the warmest year was 2016–17 (Figure 26c). Maximum within-canopy temperatures exceeded 35°C for a total of 40 days (in total among all sites where temperature is monitored) during 2020–21 (Figure 26e), with the highest temperature recorded at 38.1°C (ST1, 2pm 06Apr21). Daytime tidal exposure (hours water has drained from the meadow) was below the long-term median for Cape York (Figure 26c, Figure 92), 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 fine sand (Appendix 3, Figure 107, Figure 108).

5.1.3 Inshore seagrass and habitat condition

There are 17 seagrass monitoring sites in Cape York from 9 locations (Table 12). Four seagrass habitat types were assessed across the Cape York region in 2020–21, with data from 14 of the 17 long-term monitoring sites (Table 12, Table 19).

Table 12. 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 4 and Table 5. Open square indicates not measured in 2020–21, blank cells indicate data not usually collected/measured at site. † drop camera sampling (RJFMP), *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 2020–21 monitoring period, the seagrass condition index score for the Cape York region improved slightly since the previous monitoring period, with the overall grade being **moderate** (Figure 27).

There were improvements in both abundance and resilience (Figure 27). The greatest score improvement occurred in abundance, which improved from poor in 2019–20, to moderate in

2020–21. Prior to 2019–20, abundance across the region was graded as moderate for six consecutive years.

Although the resilience score improved in 2020–21, the overall grade remained poor for the second consecutive year. This was partly a consequence of low reproductive effort and declining seed banks in coastal habitats, e.g. Shelburne Bay (Figure 27).

Overall, the Cape York seagrass condition index remains well below the 2005–06 baseline and in 2020–21 was the third lowest over the last decade.

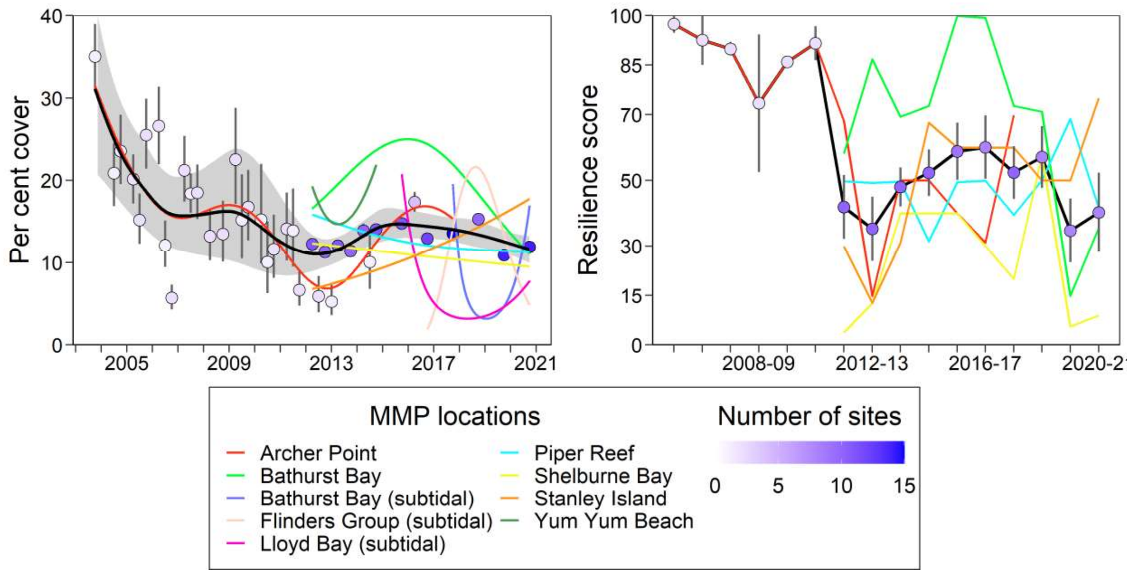


Figure 27. Temporal trends in the Cape York seagrass indicators used to calculate the seagrass condition index: a. average (circles, ±SE) seasonal abundance (per cent cover) and GAM plots of seagrass abundance trends for each location (coloured lines) and the region (black line with grey shaded area defining 95 per cent confidence intervals); b. average annual resilience score (±SE) and trends for each location (coloured lines). Colour of circles represents the number of sites assessed to calculate the average.

An examination of the long-term trends in abundance 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 compared to the highest levels recorded for the region (Figure 27). Archer Point, which was the only location monitored prior to 2012–13, has not been included in the resilience score since October 2017, when monitoring continued only as part of Seagrass-Watch due to logistical difficulties.

5.1.3.2 Seagrass abundance, composition and extent

The improvement in seagrass abundance in 2020–21 is a consequence of increases in per cent cover at coastal intertidal and subtidal sites at Bathurst and Lloyd Bays, and one of the reef subtidal sites in the Flinders Group (Figure 28). The majority of these are adjacent to the Normanby River mouth, where the discharge for the last two wet seasons was below or near its annual median volume. Seagrass abundance was either unchanged or slightly decreased in the more northern regions of Cape York where rivers discharged volumes 1.5 to 3 times above the long-term median in the 2020–21 wet season, which was after the assessment of abundance.

An examination of the long-term trend in seagrass abundance shows seagrass per cent cover progressively decreased at reef intertidal habitats across Cape York from 2003 to 2012, after which there was a gradual improvement, particularly at Stanley Island, but abundances at the reef intertidal sites remain low (Figure 28, Table 21). Coastal intertidal and subtidal habitats which have only been monitored since 2012 and 2015 respectively, showed no long-term trend (Figure 28, Table 21).

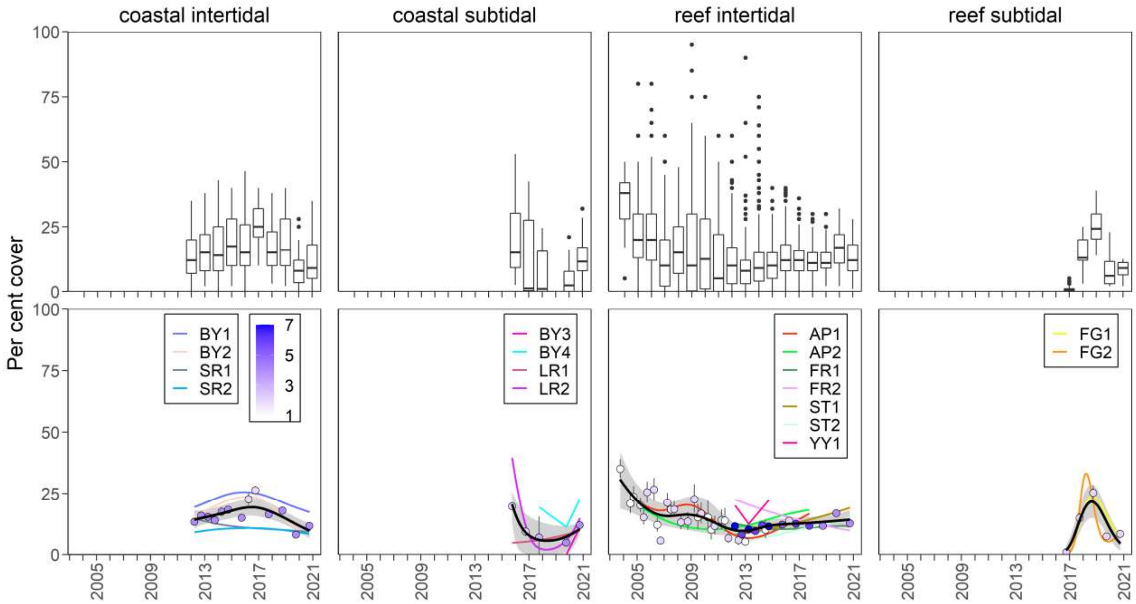


Figure 28. Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends for each habitat monitored in the Cape York region from June 2005 to May 2021. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates 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. GAM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.

In 2020–21, the proportion of species displaying colonising species traits (largely *Halophila ovalis*) were slightly lower than the previous reporting year in all habitats in the Cape York region. With the exception of reef habitats, the proportions of colonising species were above the Reef long-term averages for all other habitats in 2020–21. Reef subtidal habitats were exclusively colonising species (Figure 29).

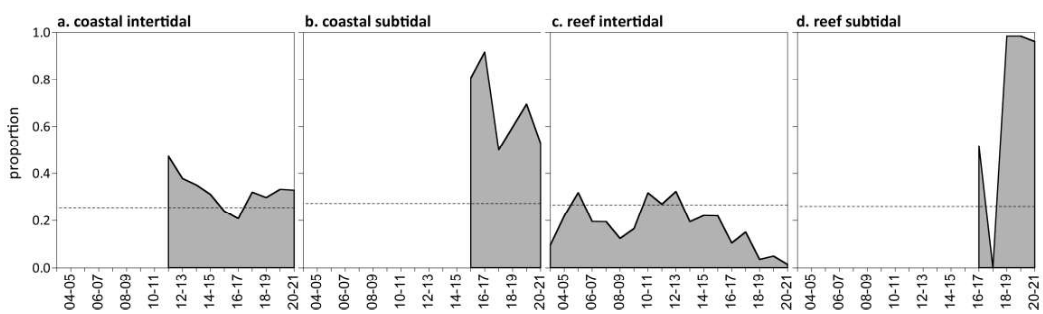


Figure 29. Proportion of seagrass abundance composed of species displaying colonising traits in each inshore habitat in the Cape York region.The dashed line represents Reef long-term average for each habitat type.

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 30), 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, relative meadow extent has been reasonably stable since 2012 (Figure 30), though meadow extent has declined in coastal intertidal habitats, due primarily to changes in drainage channels.

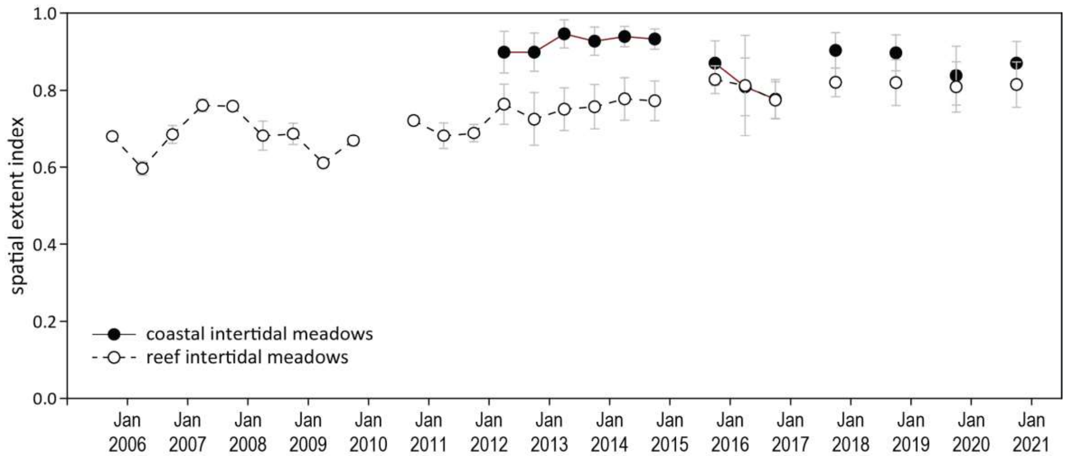


Figure 30. Change in relative spatial extent (± SE) of seagrass meadows within monitoring sites for each intertidal coastal and reef habitat and monitoring period across the eastern Cape York NRM region.

5.1.3.3 Seagrass reproductive status

Total reproductive effort is only monitored at intertidal meadows in Cape York. Reproductive structures were only reported at two of the eight sites examined in 2020–21; one site in each habitat. Historically, from 2006 to 2012, reproductive effort in reef intertidal habitats was recorded only at Archer Point, which has not been assessed since 2017, and is now based on sites introduced in 2012, which have consistently low numbers of reproductive structures. Reproductive effort remained low at coastal habitats across the region, after declining in 2019–20 (Figure 31).

Seed banks are also only measured at intertidal sites across Cape York and are dominated by *H. uninervis*. Seeds are typically low in density in reef intertidal habitats, and remained absent in 2020–21. Seed density in seed banks also declined at coastal habitats across the region in 2020–21, but seed banks persist at all but one of the sites. The low reproductive effort for the second year in a row will hinder replenishment of the declining seed banks, rendering most meadows vulnerable to further disturbances because of their limited capacity to recover from seed (i.e. low resilience).

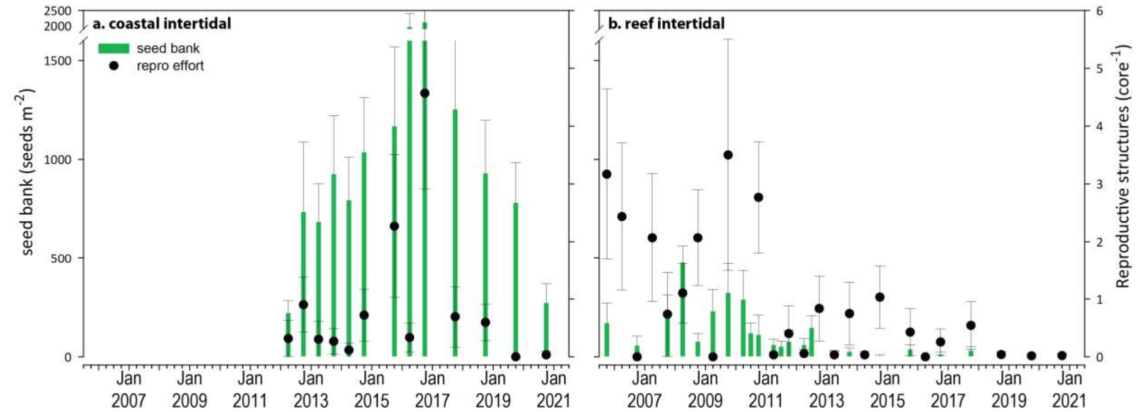


Figure 31. Seed banks and reproductive effort at inshore intertidal coastal (a) and reef (b) habitats in the Cape York region, for late dry season, 2005–21(species and sites pooled). Seed banks (green bars, ± SE) presented as the total number of seeds per m[2] sediment surface. Reproductive effort (dots, ± SE) presented as the average number of reproductive structures per core. NB. Reproductive effort was also assessed in the late wet season from 2008 to 2016.

5.1.3.4 Resilience

The resilience score is calculated for locations where reproductive effort is assessed. In Cape York, this is at intertidal coastal and reef habitats. In 2020–21, the resilience score was moderate overall.

At coastal sites, the score was low but increased slightly compared to the previous year. At Bathurst Bay, abundance was low at BY2 and there were no reproductive structures of opportunistic or persistent species at either site. However, there had been reproductive structures at BY1 in previous years raising the score for this site. At Shelburne Bay, per cent cover was low and below the 20th percentile for the site — a sign of vulnerability — and there were no reproductive structures present in 2020–21.

Resilience was higher at reef intertidal sites where meadow abundance and composition was more stable, and above resilience thresholds. Reproductive structures are not recorded in every year and were only recorded at one reef site at Stanley Island (ST1) in 2020–21, but recent history of reproductive effort indicates likelihood of a seedbank and clonal diversity, which are important for resilience.

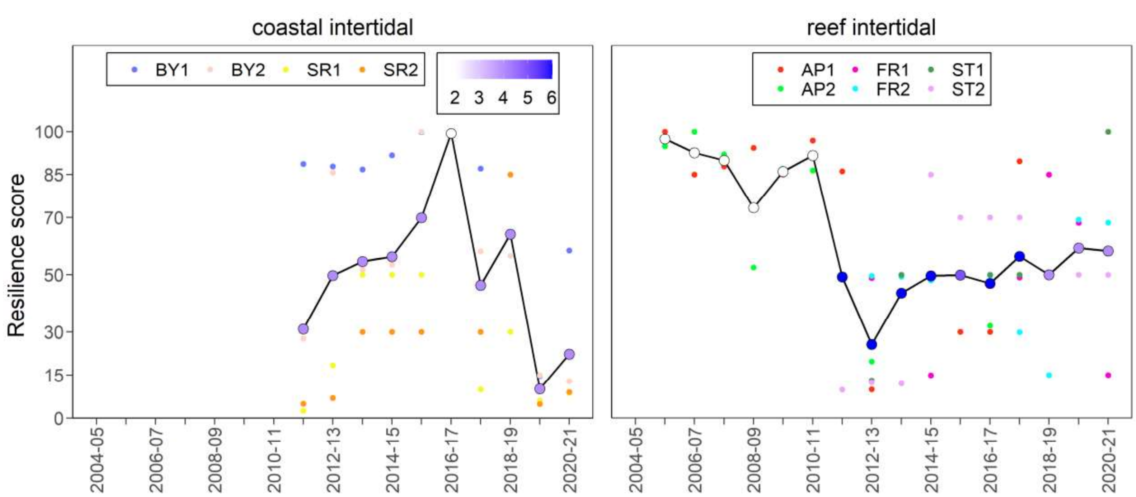


Figure 32. Temporal trend in the resilience score for each habitat monitored in the Cape York NRM region from 2005–2021. Coloured small points represent different sites. Shades of blue for the larger points indicate the number of sites that contribute to the score.

5.1.3.5 Epiphytes and macroalgae

Epiphyte cover on seagrass leaf blades at intertidal meadows remained below the long-term average at both coastal and reef habitats (Figure 33).

Per cent cover of macroalgae was variable between locations. Macroalgae cover at coastal sites varied little and in 2020—21 remained below the overall inshore Reef long-term average (Figure 33). At intertidal reef habitats, macroalgae cover remained above the Reef long-term average in the central and north of the region for the seventh consecutive year (Figure 33e), with macroalgae growing attached to coral rubble in the meadow, and not considered to be at levels sufficient to impact seagrass. Macroalgae at reef subtidal sites continued to remain below the overall inshore Reef long-term average.

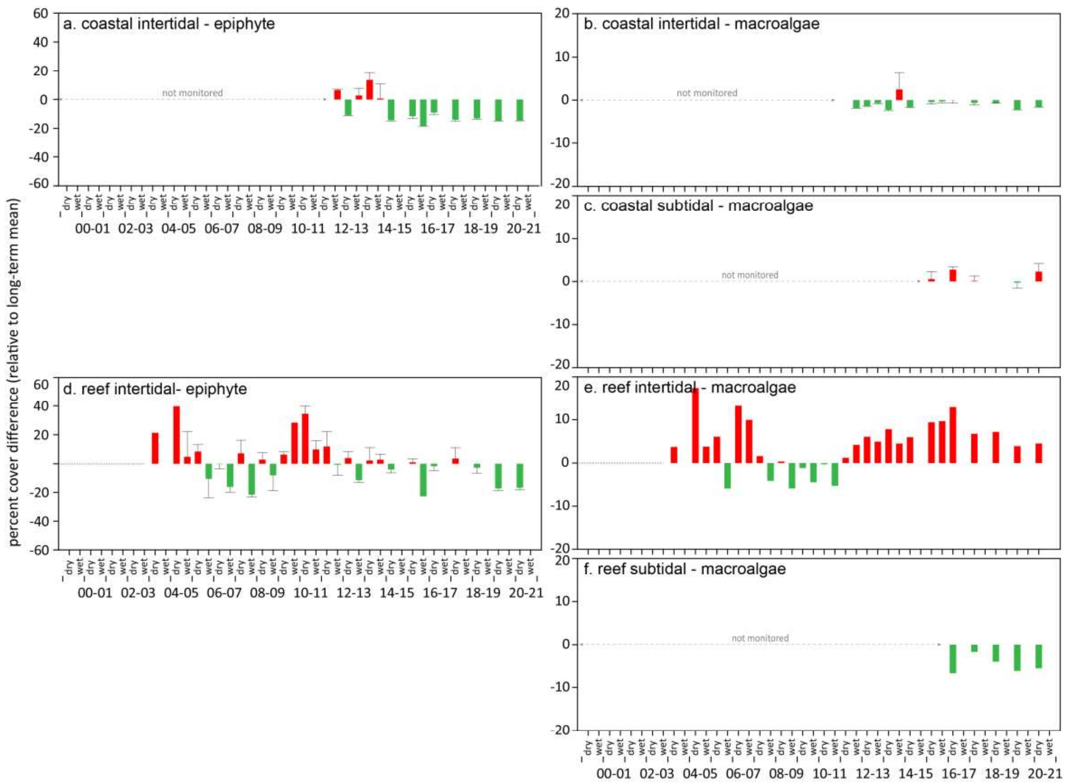


Figure 33. Deviations in mean epiphyte and macroalgae abundance (per cent cover) at monitoring habitats in the Cape York region, relative to the Reef long-term average (sites pooled, ± SE).

5.2 Wet Tropics

5.2.1 2020–21 Summary

Environmental conditions were relatively benign in 2020–21 in the northern Wet Tropics while rainfall and river discharge were higher than the long-term median in the southern Wet Tropics and there was lower than average daily light levels, but water temperature was below average.

Seagrass meadows within the Wet Tropics showed an overall improvement in the seagrass condition index in 2020–21, but remain in a vulnerable state in the southern Wet Tropics region. Seagrass condition in the northern Wet Tropics NRM region increased and was moderate (Figure 34). Seagrass condition improved but remained poor in the southern Wet Tropics (Figure 34). The combined regional condition was **moderate** (Figure 34).

Contributing indicators in the north were:

- abundance was moderate
- resilience was moderate.

Contributing indicators in the south were:

- abundance was poor
- resilience was poor.

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 northern Wet Tropics sites, seagrass abundance improved across the region in 2020–21 relative to the previous period because of increasing trends at intertidal reef and coastal sites, and mild climatic conditions across the sub-region.

In the southern Wet Tropics, seagrass abundance remained on an increasing trajectory since 2012–13, with the overall abundance in 2020–21 the highest since 2009: primarily driven by coastal subtidal and reef intertidal habitats. Overall abundance was low compared to the northern sub-region, and abundances significantly declined over the long-term at coastal intertidal sites. The declines were a legacy of losses that occurred from 2009 to 2011, the result of multiple years of severe weather, above-average rainfall and elevated discharge. Recovery of seagrass meadows post 2011 was challenged, particularly in the south, by unstable substrates, chronic poor water quality compared to the north (high turbidity, light limitation) and limited recruitment capacity.

Resilience was moderate overall in the northern Wet Tropics, but varied among habitat and site. The largest contributing factor to low scores at sites in the north was the dominance of colonising species at the reef intertidal and subtidal sites at Low Isles, and in the south it was low abundances at Lugger Bay. Coastal habitats in the north maintained a healthy seed bank, and in 2020–21 seed density was the fourth highest on record. Reproductive effort improved at coastal sites, but was greatly depressed at reef intertidal sites signalling a potential future decline in seeds, but slightly higher at one of the reef subtidal sites. In the south, reproductive effort was similarly depressed and declined at reef intertidal and subtidal habitats; sexual reproduction remained absent in coastal habitat. A depauperate seed bank persisted at only one site in the south, where seeds continued to be absent across all other sites. The absence of sexual propagules, indicating low resilience, is likely a contributor to slow recovery in the sub-region.

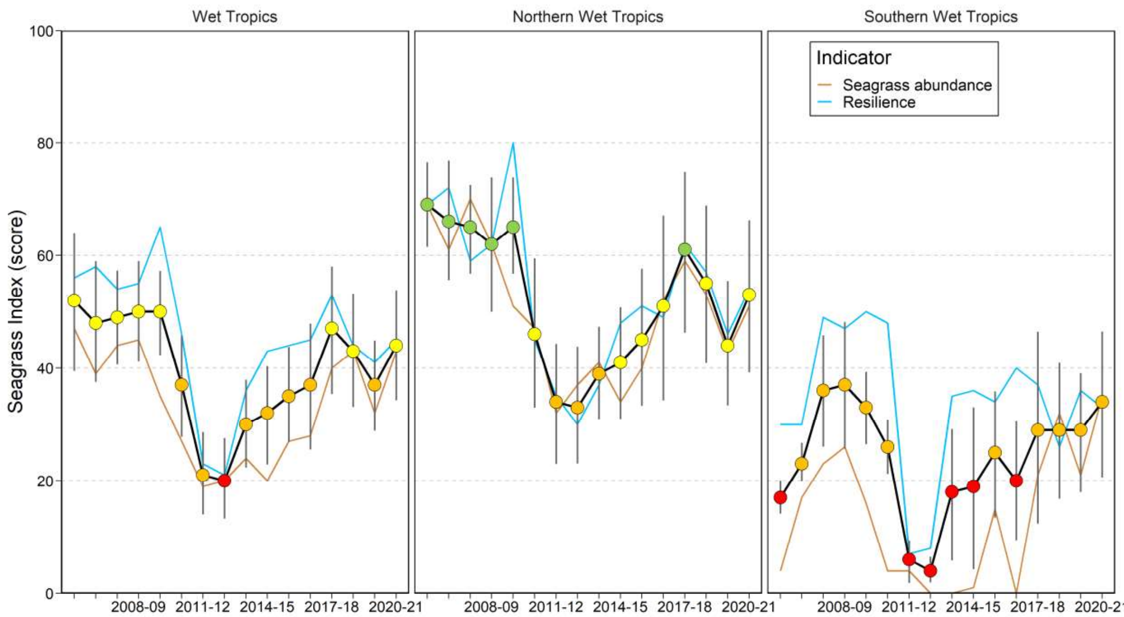


Figure 34. Report card of seagrass index and indicators for the Wet Tropics NRM region, including northern and southern sections (average across habitats and sites). Values are indexed scores scaled 0–100 (± SE) and graded: ● = very good (81–100), ● = good (61–80), ● = moderate (41–60), ● = poor (21–40), ● = very poor (0–20). NB: Scores are unitless.

5.2.2 Climate and environmental pressures

There were two tropical cyclones to affect the Wet Tropics region in 2020–21. Tropical cyclone Kimi moved south through the outer Wet Tropics between the 16th and 19th January, not making landfall due to a sudden and unexpected weakening as it approached the coast. Severe Tropical Cyclone Niran was a category 5 cyclone that moved through the Coral Sea to New Caledonia between the 25th and the 5th March, only briefly affecting the Wet Tropics before it intensified as it moved away from Australia. Annual rainfall and river discharge were slightly higher than average in the northern Wet Tropics in 2020–21 across the region.

Exposure to primary or secondary turbid water was similar to the long-term average across the northern Wet Tropics during 2020–21 (Figure 35a, b). Sites were primarily exposed to secondary water at reef sites and primary water at the coastal sites at Yule Point (Moran *et al.* 2022). Daily light levels (12.7 mol m^{-2} d^{-1} in 2020–21) were around the long-term average in the northern Wet Tropics (Figure 35c, d).

Intertidal within-canopy temperatures in the northern Wet Tropics were above the long-term average in intertidal habitats for the seventh consecutive year in 2020–21 (Figure 35e). Maximum intertidal within-canopy temperatures exceeded 35°C for a total of 63 days during 2020–21, with the highest temperature recorded at 39.6°C (YP1, 2:00pm 23Mar21).

Daytime tidal exposure in the north was below the long-term median (Figure 35c, Figure 93, Figure 94), which may have provided some respite from the elevated temperatures, particularly in coastal habitats.

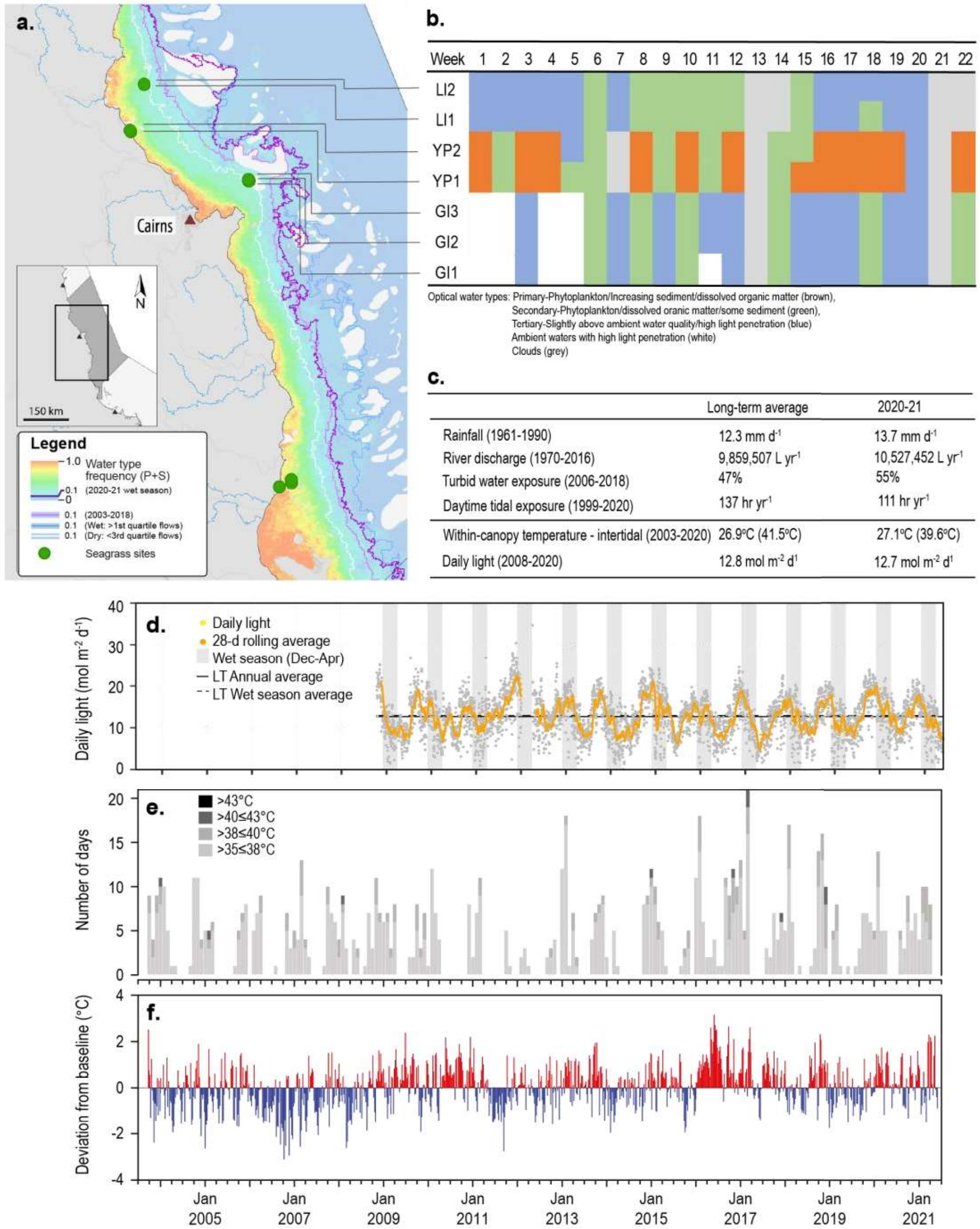


Figure 35. Environmental pressures in the northern Wet Tropics region including: a. frequency of exposure to primary and seconday water from December 2020 to April 2021 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed) (white = no data), also showing the long-term average (2003–2018) exposure boundary (purple line), and the first (blue line) and third quartile (white line) of the long-term average (from Moran *et al.* 2022); b. wet season water type at each site; c. average conditions and max temperature over the long-term and in 2020–21; 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; and f. deviations from 13-year mean weekly temperature records at intertidal sites.

Annual rainfall and river discharge were higher than average across the southern Wet Tropics during 2020–21. The largest deviations were in the most southern parts including the Murray and Herbert River catchments, where discharge was more than 1.5 times the long-term median (Figure 6). Exposure to primary or secondary turbid water occurred 89 per cent of weeks during the wet season, which was a lower level of exposure than average (99 per cent) (Figure 36a, c). There was less frequent exposure to primary water and more exposure to secondary water at coastal sites including Lugger Bay (LB1 and LB2) and Missionary Bay (MS1 and MS2) compared to previous years (Figure 36b).

Light was measured at Dunk Island in the southern Wet Tropics. At the subtidal site, the annual average (5.1 mol m^{-2} d^{-1}) was lower than the long-term average (6.8 mol m^{-2} d^{-1}) and 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 36d, Figure 102). There were periods where no data was recorded in the early part of the reporting year (Figure 102). At the intertidal site, the annual average (14.4 mol m^{-2} d^{-1}) was also lower than the long-term average (16.0 mol m^{-2} d^{-1}) and at this site there were no data gaps.

In the southern Wet Tropics, within-canopy temperatures in 2020–21 were below the long-term average (Figure 36b). Maximum intertidal within-canopy temperatures during 2020–21 did not exceed 35°C for the first time in a decade, with the highest temperature recorded at 34°C (DI2, 1pm 26Sep20) (Figure 36e, f). Daytime tidal exposure was slightly below the long-term average (Figure 36b, Figure 93, Figure 94).

Overall, the inshore seagrass habitats throughout the southern Wet Tropics experienced similar levels of environmental pressures in 2020–21 as those in the northern Wet Tropics, remaining around average based on most indicators except rainfall and river discharge.

In 2020–21, sediments appeared similar to the long-term average and the proportion of fine sediments (i.e. mud) was well below the overall inshore Reef long-term average across all habitats. Nevertheless, a slight increase in mud was noted at one of the coastal sites (YP2) in the north (Figure 109, Figure 110). Across the Wet Tropics region, coastal sediments were composed primarily of fine sand, while reef habitats were composed of sand and coarser sediments (Figure 109, Figure 110).

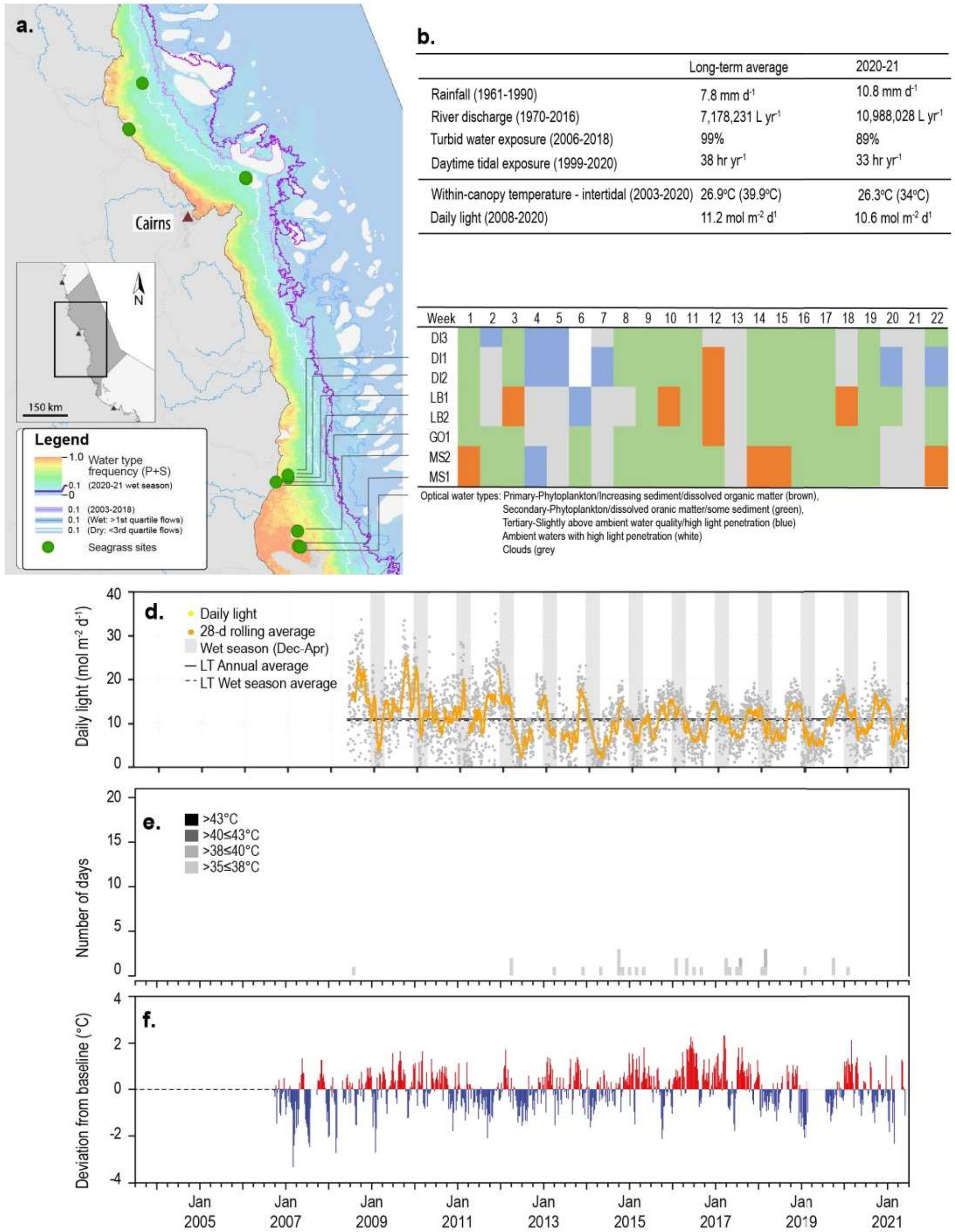


Figure 36. Environmental pressures in the southern Wet Tropics region including: a. frequency of exposure to primary and seconday water from December 2020 to April 2021 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed) (white = no data), also showing the long-term average (2003–2018) exposure boundary (purple line), and the first (blue line) and third quartile (white line) of the long-term average (from Moran *et al.* 2022); b. average conditions and max temperature over the long-term and in 2020–21; c. wet season water type at each site; 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; and f. deviations from 13-year mean weekly temperature records at intertidal sites.

5.2.3 Inshore seagrass and habitat condition

Three seagrass habitat types were assessed across the Wet Tropics region with data from 14 sites (Table 13).

Table 13. List of data sources of seagrass and environmental condition indicators for each seagrass habitat type in the Wet Tropics NRM region.Open square indicates not measured in 2020–21, blank cell indicates data not usually collected/measured at site. † drop camera sampling (RJFMP), *Seagrass-Watch. For site details see Table 4 and Table 5.

Sub region	Habitat	Site		abundance	composition	extent	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 2020–21 monitoring period, the seagrass condition index for the overall Wet Tropics region improved from poor in 2019–20 to moderate (Figure 15). Although both indicators increased, the overall improvement was primarilyy due to seagrass abundance, which increased from poor to moderate, while resilience remained poor. Examination of the sub-regional scores highlights the differences between seagrass condition in the north and south of the Wet Tropics (Figure 34).

In the northern Wet Tropics, the seagrass condition index increased in 2020–21, but remained moderate and below the 2018–19 peak (Figure 37). Similar to the overall NRM regional grade, the improvement was primarily due to increasing abundances across most locations and improved reproductive effort at coastal habitats. The long-term trend in seagrass per cent cover is variable between monitoring locations (Table 21), but closely reflects the sub-regional scores with improved cover from 2014–15.

Seagrass resilience has fluctuated over the life of the MMP, peaking in 2009–10, after which it declined for the next three consecutive years and has generally been on an improving trajectory since (Figure 37).

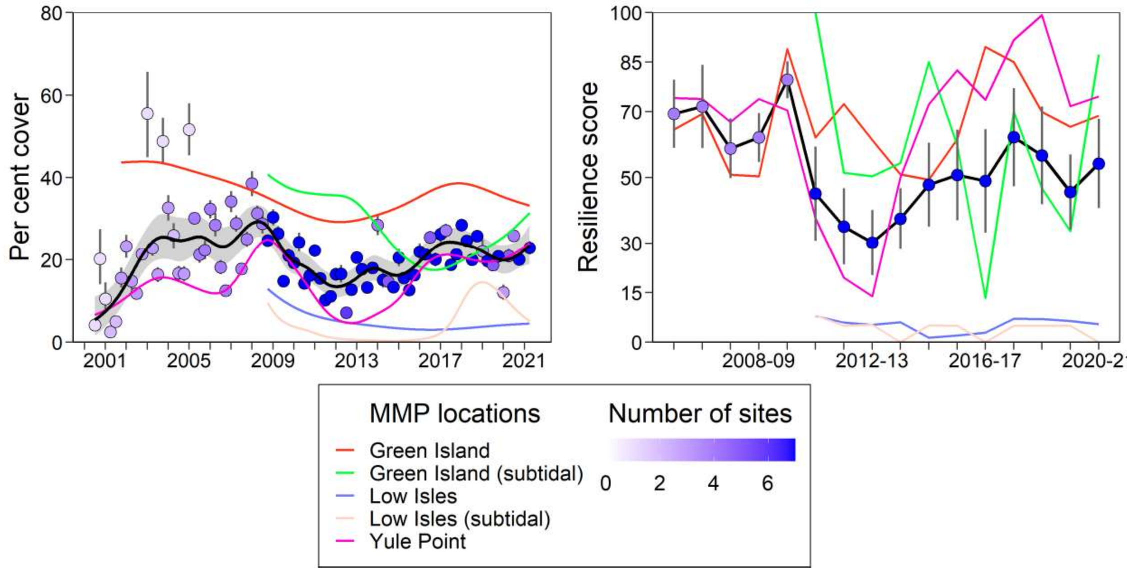


Figure 37. Temporal trends in the northern Wet Tropics seagrass indicators used to calculate the seagrass condition index: a. average (circles, ±SE) seasonal abundance (per cent cover) and GAM plots of seagrass abundance trends for each location (coloured lines) and the region (black line with grey shaded area defining 95 per cent confidence intervals); b. average annual resilience score (±SE) and trends for each location (coloured lines). Colour of circles represents the number of sites assessed to calculate the average.

In the southern Wet Tropics, the seagrass condition index improved for the first time in three years in 2020–21; a consequence of improved abundances (Figure 38). Both the abundance and resilience indicators have been highly variable since 2012–13, often with what appears as an annual lag from abundance to resilience (Figure 34). In 2020–21, resilience decreased relative to the previous period (Figure 34). The index remained poor in 2020–21.

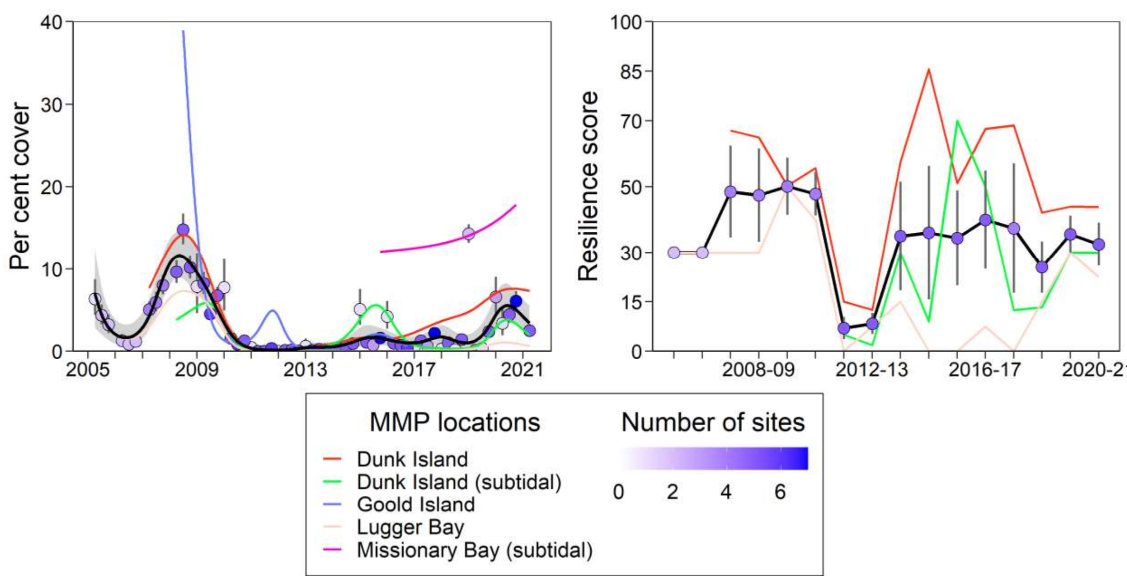


Figure 38. Temporal trends in the southern Wet Tropics seagrass indicators used to calculate the seagrass condition index: a. average (circles, ±SE) seasonal abundance (per cent cover) and GAM plots of seagrass abundance trends for each location (coloured lines) and the region (black line with grey shaded area defining 95 per cent confidence intervals); b. average annual resilience score (±SE) and trends for each location (coloured lines). Colour of circles represents the number of sites assessed to calculate the average.

5.2.3.2 Seagrass abundance, community and extent

Seagrass meadows are more abundant (higher per cent cover) across all habitats in the northern than the southern Wet Tropics (Figure 39, Figure 40). In the northern Wet Tropics, seagrass abundance over the long-term is higher at intertidal reef (28.3 ±2.1 per cent) than subtidal reef (17.1 ±2.4 per cent) or coastal habitats (14.8 ±1.6 per cent). In 2020–21, seagrass abundances improved overall in the northern Wet Tropics. Despite abundances remaining steady at 3 of the 7 sites assessed, the declines observed at Low Isles (both intertidal and subtidal), were offset by increases at an intertidal reef and an intertidal coastal site (Figure 39).

Seagrass losses have occurred at the local level (e.g. individual sites) for some period over the duration of the monitoring, but 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 per cent in the dry and 19.8 ±2.1 per cent in the late dry-monsoon) and years (from 9.5 ±1.9 per cent to 31.3 ±2.1 per cent annual average).

In the southern Wet Tropics, although long-term seagrass abundance is higher at intertidal reef (4.5 ±1.0 per cent) than at subtidal reef (1.9 ±0.8 per cent) or coastal habitats (1.8 ±0.6 per cent), the abundances were only a tenth of those observed in 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 was sustained for 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 which have changed little in the following two years. Similarly, abundances improved at the reef habitats, with both intertidal and subtidal abundances having recovered to levels similar to the onset of monitoring in 2006. In the south, overall seagrass abundance remains on an increasing trajectory since 2012–13, with abundances in 2020–21 being the highest since 2009: primarily driven by coastal subtidal and reef intertidal habitats.

An examination of temporal trends in seagrass abundance across the Wet Tropics NRM region showed no significant trend over the long-term i.e. from the first year of monitoring to 2021 (Table 21). In the northern Wet Tropics, changes in seagrass abundance were variable among habitats, with 3 of the 7 of sites significantly declining over the long-term, while only one of the remaining sites showing an increasing trend. The declines in the north were all in reef habitats, at both an intertidal and a subtidal site. In the southern sub-region, two of the eight sites significantly declined over the long-term, but these only occurred at the coastal intertidal sites (Lugger Bay). No long-term trend was apparent in the reef habitats of the southern sub-region.

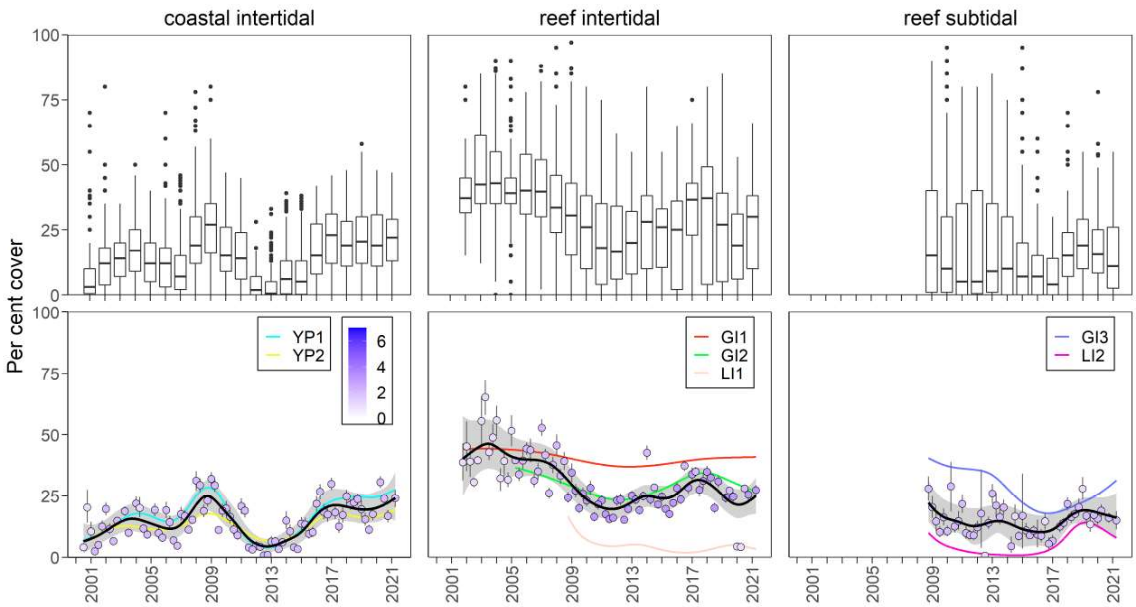


Figure 39. Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the northern Wet Tropics NRM region from 2001 to 2021. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero 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. GAM plots (bottom), show trends for each habitat and coloured lines represent individual site trends

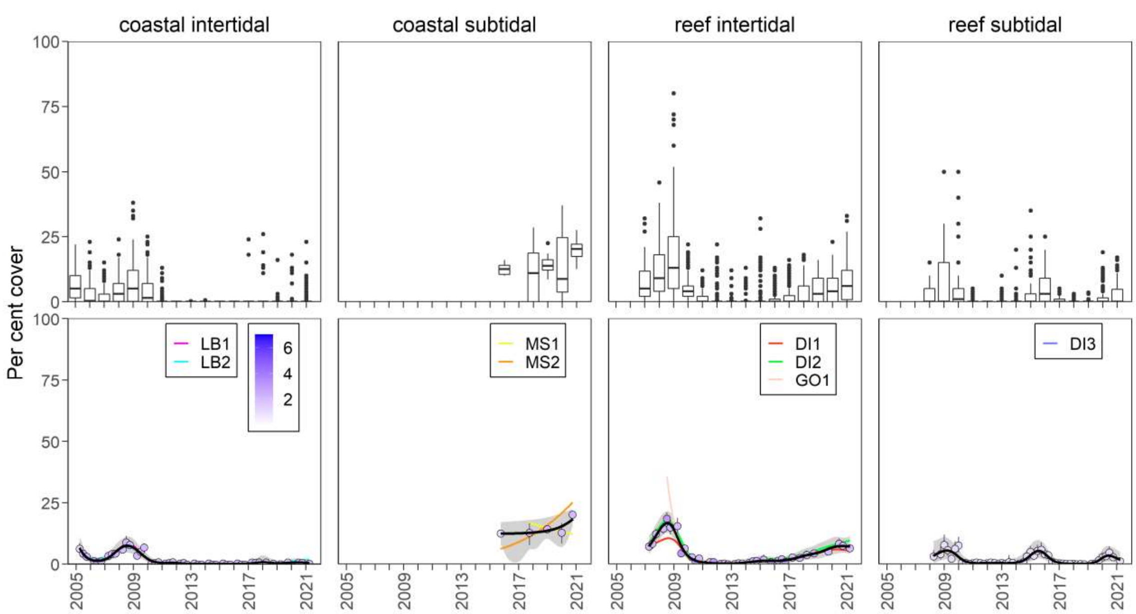


Figure 40. Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the southern Wet Tropics NRM region from 2001 to 2021. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero 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. GAM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.

The proportion of seagrass species displaying colonising traits in the northern Wet Tropics was above the long-term average for each habitat type in 2020–21 (Figure 41). At coastal intertidal habitats (Yule Point), the proportion increased slightly compared to the previous period, suggesting minor levels of physical disturbance in 2020–21. On reefs, colonising species decreased in intertidal habitats, but were unchanged in subtidal habitats.

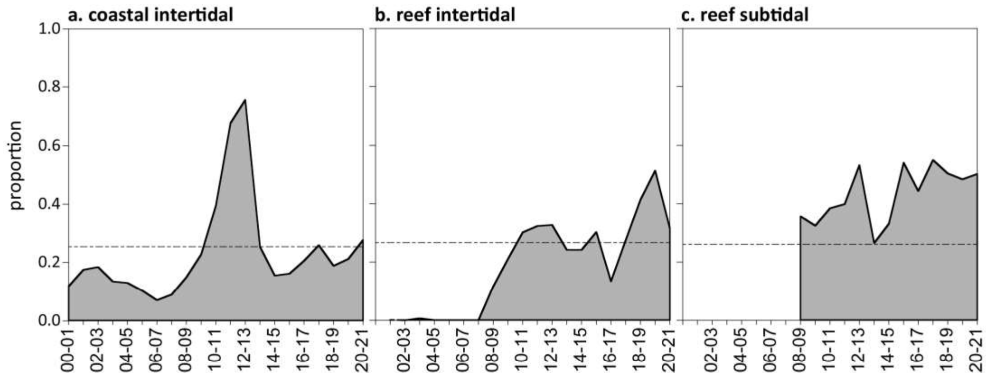


Figure 41. Proportion of seagrass abundance composed of colonising species at inshore habitats in the northern Wet Tropics region, from the 2000–2001 to the 2020–21 reporting periods. The dashed line represents the overall inshore Reef average for each habitat type.

In the southern Wet Tropics, the proportion of seagrass species displaying colonising traits varied across habitats (Figure 42). In the coastal intertidal habitat there have been cycles of changing species composition since the substrate at Lugger Bay was eroded in 2011 (caused by Tropical cyclone Yasi). Opportunistic species appear unable to establish enduring meadows, potentially due to light limitation associated with deepening of the habitat. Colonising species become dominant following periodic decline of other species in what appears to be recalcitrant degradation. In 2019–20, the proportion of seagrass species displaying colonising traits decreased to zero at coastal intertidal habitats and remained at that level throughout 2020–21. Colonising species remained in low proportions in reef habitats, however they increased at coastal subtidal habitats.

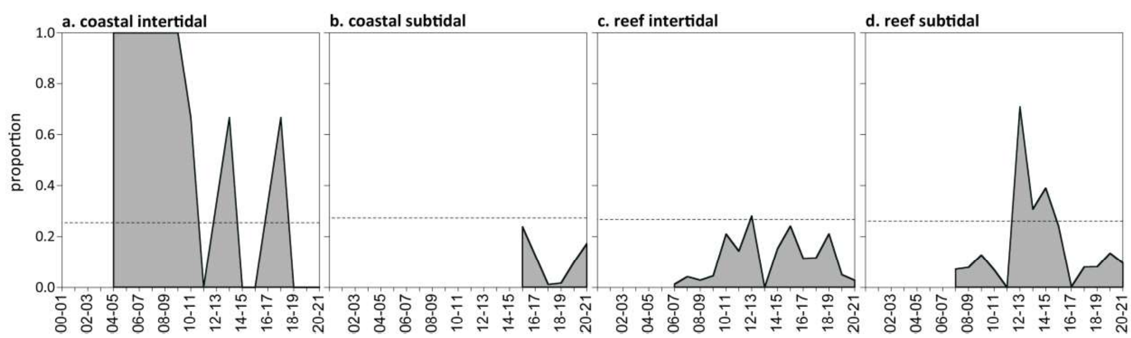


Figure 42. Proportion of seagrass abundance composed of colonising species at inshore habitats in the southern Wet Tropics region, from the 2000–2001 to the 2020–21 reporting periods. The dashed line represents the Overall inshore Reef average for each habitat type.

Seagrass meadow spatial extent within all monitoring sites has fluctuated within and between years (Figure 43). At intertidal coastal habitats in the northern Wet Tropics, meadow extent has gradually improved since 2011 and was only slightly lower than the previous highest extent. Subtidal reef meadows in the north are quite variable over seasonal and inter-annual time-scales but had peaked in extent in 2015 than earlier years. There has been little change in seagrass extent in 2020–21 compared to the previous period.

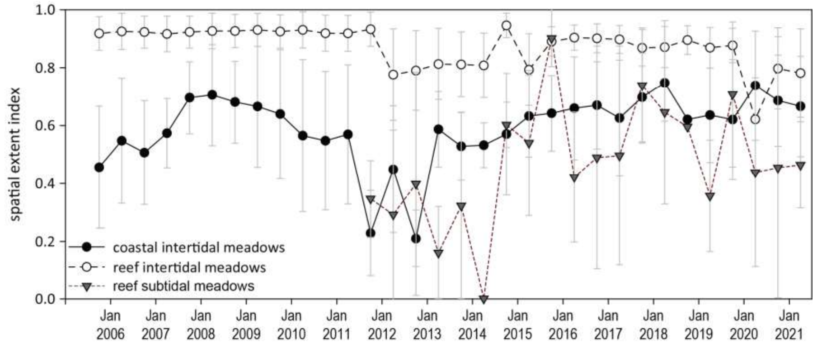


Figure 43. Change in relative spatial extent (±SE) of seagrass meadows within monitoring sites for each habitat and monitoring period across the northern Wet Tropics NRM region.

In the southern Wet Tropics, all seagrass meadows with long-term monitoring sites were lost in early 2011 as a consequence of Tropical cyclone Yasi (Figure 44). Since then, intertidal reef meadows have progressively improved, with the greatest extent since 2011 measured in 2020–21. At intertidal coastal habitats, the meadows have slowly been improving, with the isolated patches which colonised in mid-2018 continuing to expand and coalesce. The greatest improvement in extent has occurred in subtidal reef meadows.

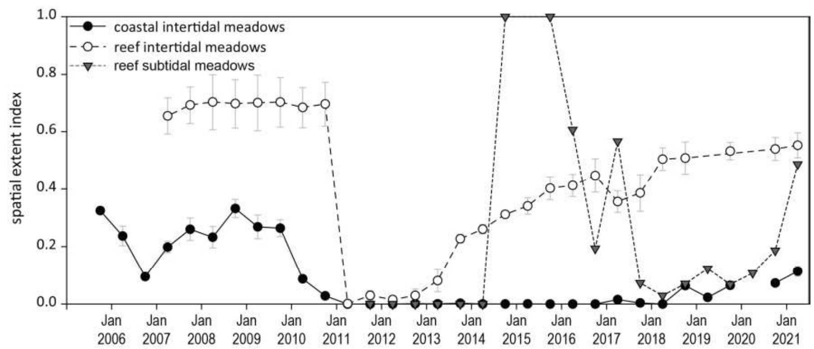


Figure 44. Change in relative spatial extent (±SE) of seagrass meadows within monitoring sites for each habitat and monitoring period across the southern Wet Tropics NRM region.

5.2.3.3 Seagrass reproductive status

Reproductive effort varies across habitats in the Wet Tropics, and is generally higher in the northern sub-region than the south. In general, reproductive effort and seed density have been buoyed in the Wet Tropics in recent years, though with some variability among habitats and regions. In the northern Wet Tropics, reproductive effort increased during 2020–21 in coastal intertidal habitats (Yule Point) (Figure 45). However, reproductive effort was greatly depressed in reef habitats in 2020–21, with reproductive structures absent from all sites, with the exception of a subtidal site (GI3) where reproductive effort increased relative to the previous period.

Seed density was the fourth highest on record at coastal intertidal habitats, likely a consequence of high reproductive effort in the previous and current year. To date, seed banks have remained very low across the region in reef habitats (Figure 45). The absence of seeds in the reef meadows examined in 2020–21, is likely the result of the greatly depressed reproductive effort over the last two years. Other 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).

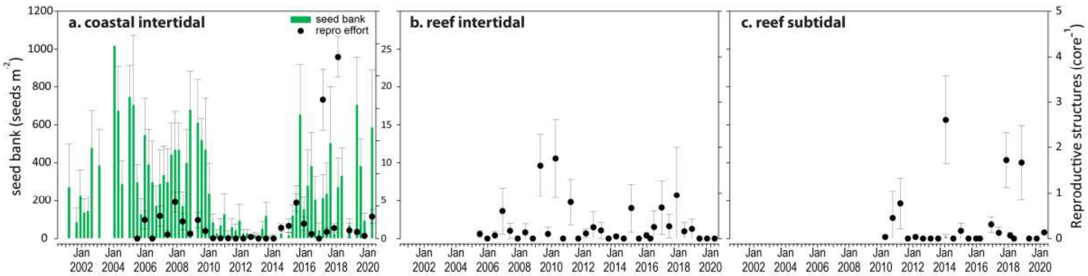


Figure 45. Seed bank and reproductive effort at inshore coastal intertidal and reef intertidal and subtidal habitats in the northern Wet Tropics region, 2001–21.Seed banks presented as the total number of seeds per m^2 of sediment surface (green bars ±SE). Reproductive effort presented as the average number of reproductive structures per core (species and sites pooled) (dots ±SE). Y-axis labels are different in panel a to those in panels b and c.

In the southern Wet Tropics, sexually reproductive structures and seed banks were absent from coastal intertidal meadows and declined in reef intertidal and subtidal habitats (Figure 46). A seed bank persists at only one site (DI1) in the southern Wet Tropics. 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, three years of high to above average reproductive effort recorded in reef intertidal habitats occurred in conjunction with small increases in abundance and extent and, together, indicate recovering habitats (Figure 46).

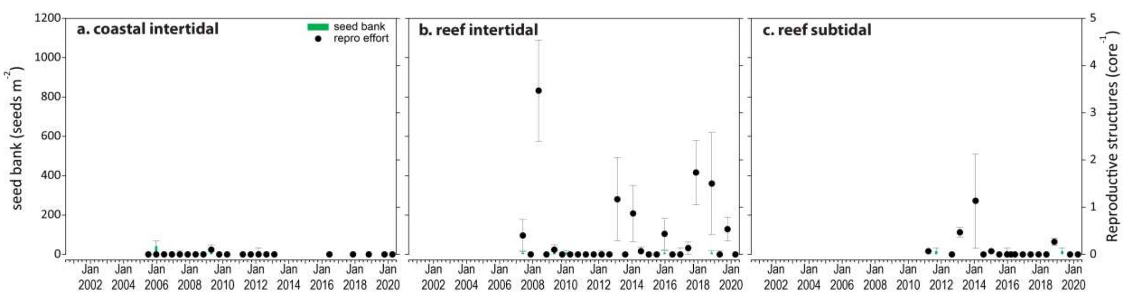


Figure 46. Seeds banks and reproductive effort for inshore coastal intertidal and reef intertidal and subtidal habitats in the southern Wet Tropics region for the late dry and late wet season, 2001–21.Seed banks presented as the total number of seeds per m^2 sediment surface (green bars ±SE). Reproductive effort presented as the average number of reproductive structures per core (species and sites pooled) (dots ±SE).

5.2.3.4 Resilience

Resilience was moderate overall in the northern Wet Tropics, but varied among habitat and site (Figure 47). At Yule Point coastal sites, meadow condition was above critical thresholds for abundance and composition, and although reproductive structures were present there were fewer than in recent years.

At reef intertidal sites at Green Island, meadow condition was above critical thresholds for abundance and composition, but reproductive structures were absent in 2020–21 for the first time in three years. At Low Isles, colonising species continue to dominate the species composition, making the meadow vulnerable to short-term disturbances.

At reef subtidal sites, the Green Island meadow condition was above critical thresholds for abundance and composition. There were no reproductive structures observed in 2020–21 or the previous three years. However, the composition of persistent species (*T. hemprichii*) was very high for the site, which increased the level of resistance within the meadow. At Low Isles, the meadow was comprised of only colonising species and there were no reproductive structures, rendering the meadow highly vulnerable to even short-term disturbances such as elevated discharge.

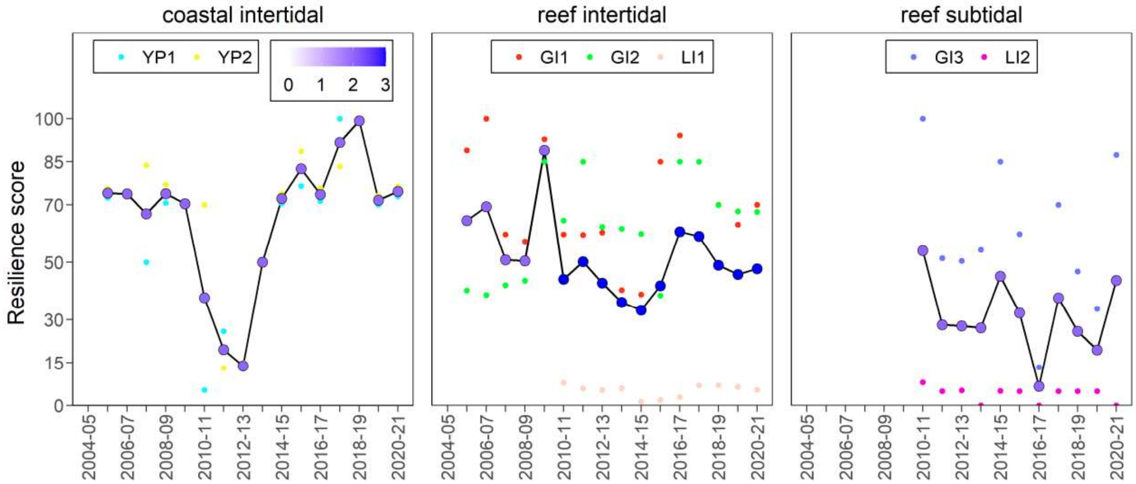


Figure 47. Resilience score for each habitat type in the northern Wet Tropics. Coloured small points represent different sites. Shades of blue for the larger points indicate the number of sites that contribute to the score.

In the southern Wet Tropics, resilience was poor overall (Figure 48). At the coastal intertidal sites at Lugger Bay, the meadow was below critical per cent cover thresholds at LB1 but the meadow was comprised of only opportunistic species that were not flowering. LB2 was above the critical thresholds for composition and per cent cover, but there were no reproductive structures present and none have been observed in the past three years.

At reef intertidal sites, meadow condition was above critical thresholds for species composition and per cent cover. There were no reproductive structures in 2020–21 but there was a history of reproductive effort at DI2. At the reef subtidal site meadow condition was above critical thresholds for species composition and per cent cover but there were no reproductive structures observed in 2020–21 or in the previous three years.

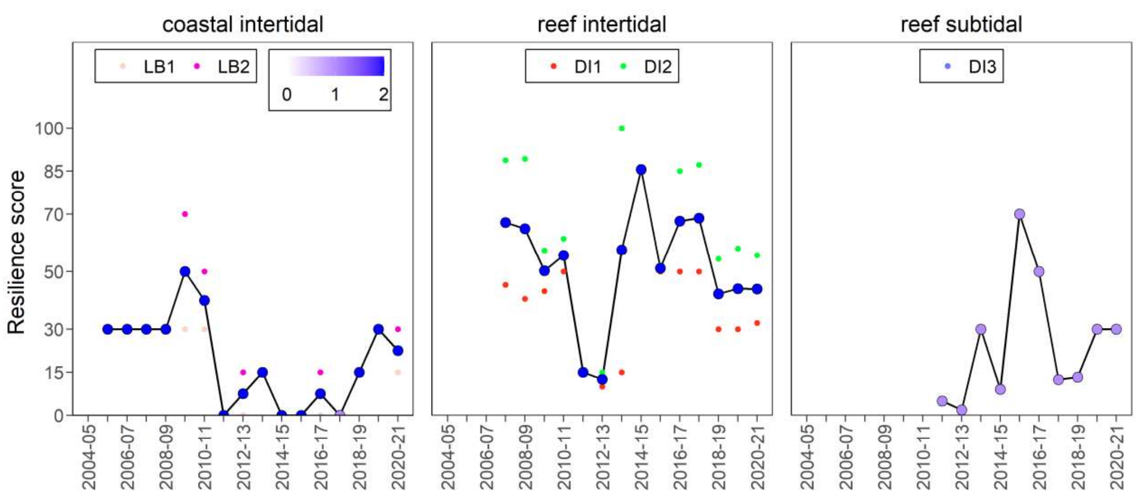


Figure 48 Resilience score for each habitat type in the southern Wet Tropics. Coloured small points represent different sites. Shades of blue for the larger points indicate the number of sites that contribute to the score.

5.2.3.5 Epiphytes and macroalgae

Epiphyte cover on seagrass leaves remained above the overall inshore Reef long-term average in coastal habitats in the northern Wet Tropics in 2020–21 (Figure 49), but below average in reef habitats.

Macroalgae cover was lower than the Reef long-term average in coastal habitat and reef subtidal habitats in both the wet and dry season (Figure 49). Macroalgae cover was slightly lower than the previous period but remained higher than the Reef long-term average in reef intertidal habitats, as is typical for the habitat because it attaches to coral rubble.

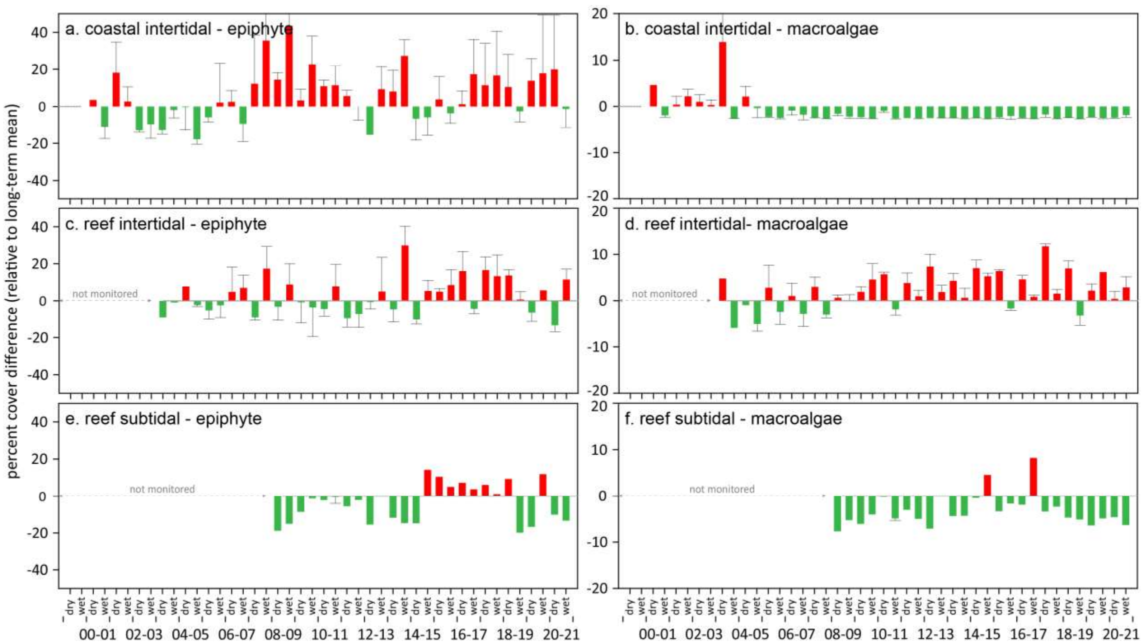


Figure 49. Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average for each inshore seagrass habitat in the northern Wet Tropics region, 2001–2021 (sites pooled, ±SE).

In the southern Wet Tropics, epiphyte cover in intertidal reef habitats remained above the Reef long-term average and increased in 2020–21 relative to the previous period (Figure 50).

Macroalgae cover was below the Reef long-term average in all habitats except reef subtidal in the southern Wet Tropics (Figure 50). Macroalgae cover at the reef subtidal site was the highest observed since monitoring commenced, occurring during the late wet season.

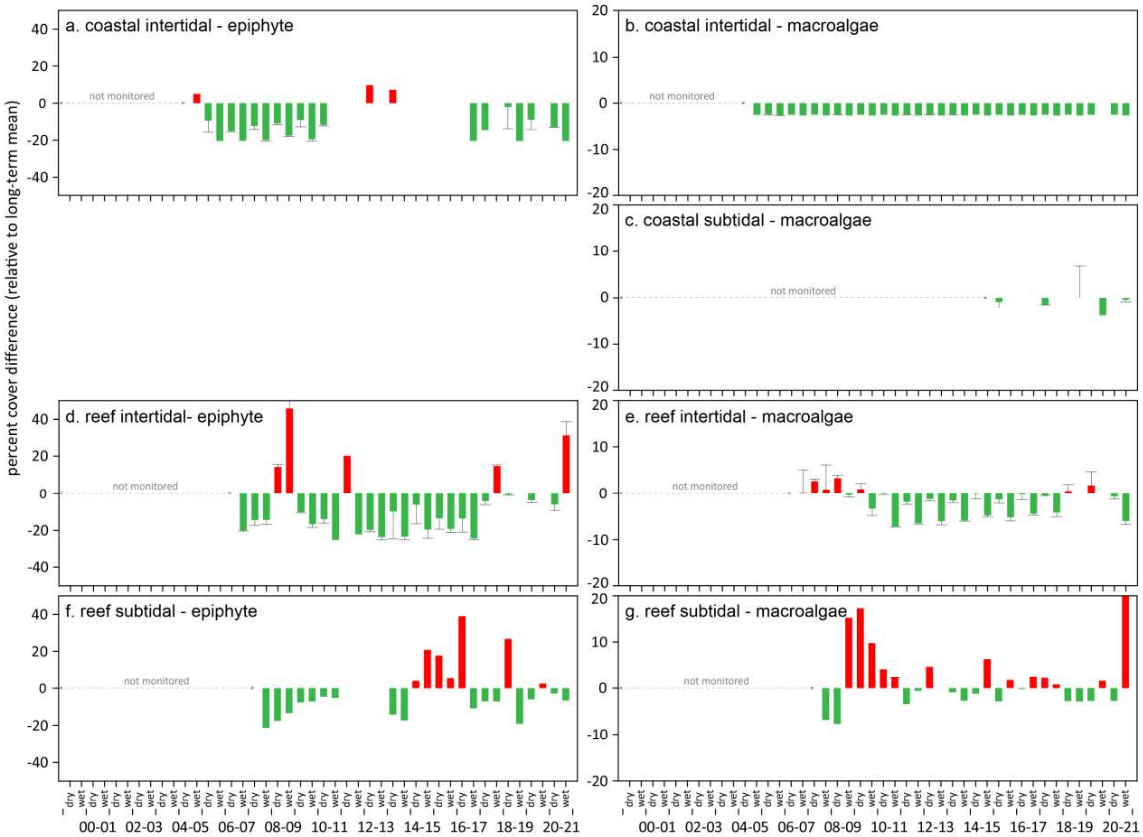


Figure 50. Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average for each inshore seagrass habitat in the southern Wet Tropics region, 2001–2021 (sites pooled, ±SE).

5.3 Burdekin

5.3.1 2020–21 Summary

In 2020–21, rainfall and river discharge were above the long-term median for all of the basins in the Burdekin region (Figure 52, Table 10).

Seagrass meadows across the Burdekin NRM region increased slightly in overall condition in 2020–21 but remained **moderate** (Figure 51). Condition indicators contributing to this were:

- abundance score was moderate
- resilience score was moderate.

Seagrass abundance remains low, but marginally increased relative to the previous period, elevating the score from poor to moderate. The low abundances at some sites were likely the legacy from the 2019 wet season when losses occurred due to river discharge from the Burdekin River in concert with unusually large discharges from the smaller creeks and rivers entering Cleveland Bay. Sediment loads in the discharge and wind-driven resuspension elevated turbidity and reduced daily light during the wet season, but light levels quickly returned to seasonally-expected levels.

Seagrass resilience increased marginally in 2020–21 compared to the previous reporting period, however the score remained poor. Patterns were inconsistent among habitat types. In coastal intertidal habitat reproductive effort increased in 2020–21 to the highest level since 2018, and similarly seed banks increased to the highest level in two years. Reproductive effort and seed banks remained very low in reef intertidal and subtidal habitats. In all habitats in 2020–21, seed density was higher in the late wet season, indicating a possible late flowering and seed set.

Since monitoring was established, seagrass meadows of the Burdekin region have demonstrated high resilience particularly through their capacity for recovery. This may reflect a conditioning to disturbance (large 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.

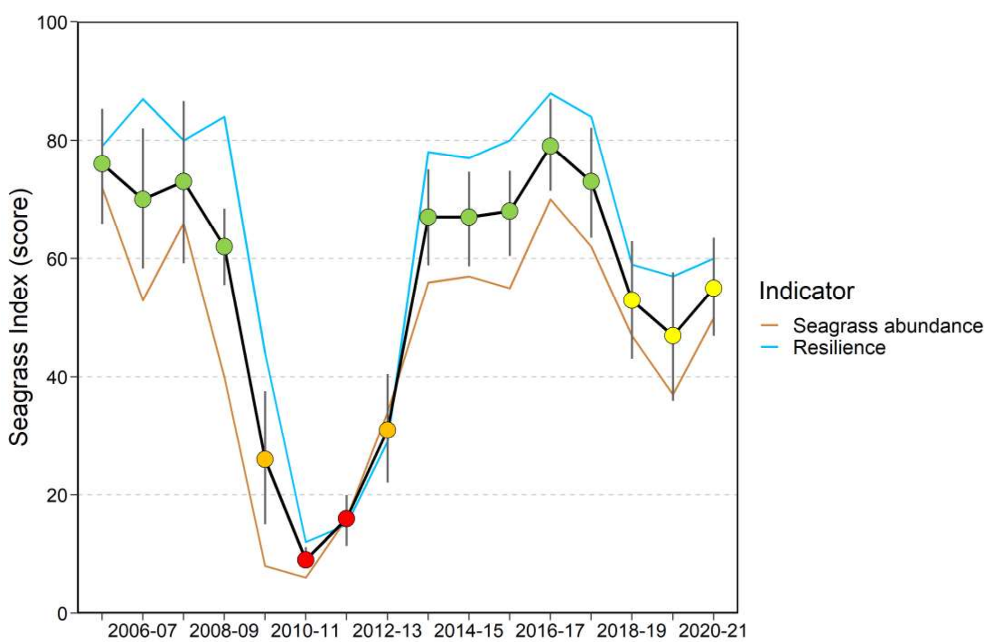


Figure 51. Report card of seagrass status indicators and index for the Burdekin NRM region (averages across habitats and sites). Values are indexed scores scaled from 0–100 (± SE) and graded: ● = very good (81–100), ● = good (61–80), ● = moderate (41–60), ● = poor (21–40), ● = very poor (0–20). NB: Scores are unitless.

5.3.2 Climate and environmental pressures

Tropical cyclone Kimi, which travelled down the Reef from 16-19th January 2021, briefly entered the Burdekin region before abating. Rainfall and river discharge were slightly above the long-term average for the region due to a large rainfall event in early January in the upper Burdekin catchment (Moran *et al.* 2022). Inshore seagrass sites in the region have a very high frequency of exposure to turbid waters during the wet season and they are the highest among all regions. In 2020–21, exposure to turbid water was around the long-term average with most sites exposed to ‘primary’ or ‘secondary’ turbid water for the entire wet season. Coastal sites (BB, SB and JR) experienced the highest exposure to ‘primary’ turbid, sediment laden, waters while reef sites at Magnetic Island were exposed predominately to secondary, phytoplankton rich waters for most of the wet season (Figure 52a, b).

Daily light levels in the Burdekin region were 9.9 mol m^{-2} d^{-1} on average in 2020–21, and therefore around the long-term average for sites in the region (Figure 52c, d). However, the trend in 2020–21 depended on the site and habitat. Annual average daily light at the reef subtidal (MI3) and the nearby Townsville coastal sites (Bushland Beach, BB1 and Shelley Beach SB1) were below average. Annual average daily light at the subtidal site was less than half (2.7 mol m^{-2} d^{-1}) the long-term average (5.6 mol m^{-2} d^{-1}) and the lowest annual daily light recorded since 2008 due to low light levels in both the dry and wet seasons (Figure 103). Daily light levels at the reef intertidal sites and the Jerona coastal intertidal site were higher than the long-term average. This combination of results suggests high incident light (due to low levels of cloud cover and/or low tides), but higher than average light attenuation/turbid water around Magnetic Island and the northern beaches. In 2020–21, the regional trend in light followed what is typically observed in other regions. Daily light levels are high throughout the winter months and late dry season, and sharply decline in the wet season months (Figure 52d).

This year intertidal within-canopy temperatures were lower than the previous period and below the long-term average (Figure 52c). Maximum intertidal within-canopy temperatures exceeded 35°C for a total of 13 days during 2020–21, with the highest temperature recorded at 38.2°C (MI2, 4pm 20Aug20); the lowest extreme temperature in 15 years (Figure 52e, f). Daytime tidal exposure was below the long-term median at all sites (Figure 52c, Figure 95, Figure 96), 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 112). Post 2011, Townsville coastal meadows have been dominated by fine sediments, although the proportion of mud has fluctuated at Bushland Beach over the last five years (Figure 112). Conversely, reef habitats remain dominated by sand sediments, although the composition of fine sediments and mud has persisted at Cockle Bay (MI2) in the last few years (Figure 113, Figure 114).

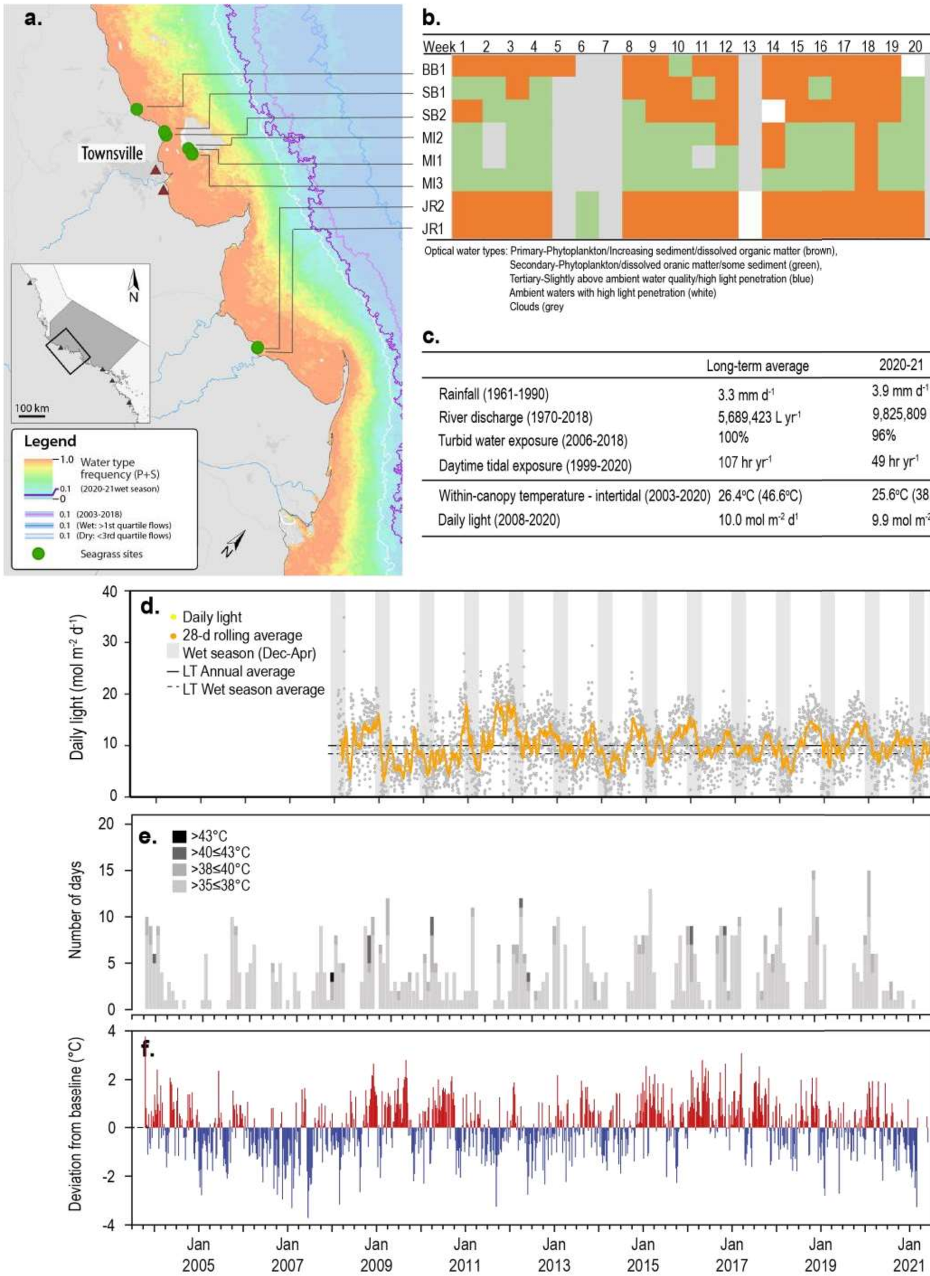


Figure 52. Environmental pressures in the Burdekin region including: a. frequency of exposure to primary and seconday water from December 2020 to April 2021 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed) (white = no data), also showing the long-term average (2003–2018) exposure boundary (purple line), and the first (blue line) and third quartile (white line) of the long-term average (from Moran *et al.* 2022); b. wet season water type at each site; c. average conditions and max temperature over the long-term and in 2020–21; 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 at intertidal sites.

5.3.3 Inshore seagrass and habitat condition

Three seagrass habitat types were assessed across the Burdekin region in 2020–21, with data from 10 sites (Table 14, Table 19).

Table 14. List of data sources of seagrass and environmental condition indicators for each seagrass habitat type in the Burdekin NRM region. Blank cell indicates data not usually collected/measured at site *Seagrass-Watch. For site details see Table 4 and Table 5.

Habitat	Site code and location		abundance	composition	distribution	reproductive effort	seed banks	leaf tissue nutrients	meadow sediments	epiphytes & macroalgae
coastal intertidal	BB1	Bushland Beach (Townsville)	■	■	■	■	■	■	■	■
	BW1*	Front Beach (Bowen)	■	■			■		■	■
	BW2*	Front Beach (Bowen)	■	■			■		■	■
	JR1	Jerona (Barratta CK, Bowling Green Bay)	■	■	■	■	■	■	■	■
	JR2	Jerona (Barratta CK, Bowling Green Bay)	■	■	■	■	■	■	■	■
	SB1	Shelley Beach (Townsville)	■	■	■	■	■	■	■	■
	SB2*	Shelley Beach (Townsville)	■	■			■		■	■
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 2020–21 monitoring period, the seagrass condition index for the Burdekin region increased slightly to the highest level in two years, but remained **moderate** (Figure 51). The grade continued to appear a legacy of the previous monitoring periods, which were influenced by region-wide above average wet season rainfall and river discharge in early 2019, and have carried into the 2020–21 reporting period.

Both indicators contributing to seagrass condition improved in 2020–21. Examination of the 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 habitats in 2020–21 would appear to be improving, with the exception of the reef subtidal at Magnetic Island (Figure 53).

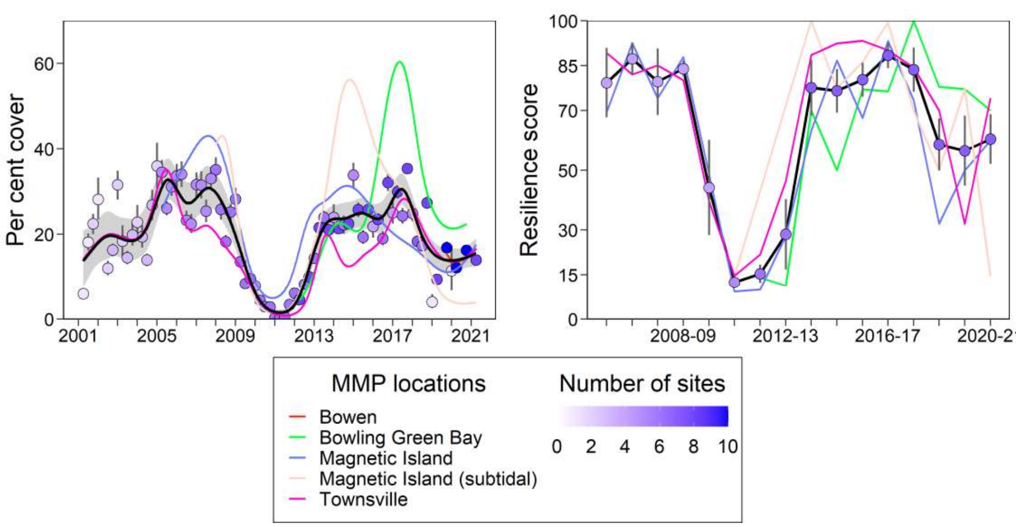


Figure 53. Temporal trends in the Burdekin seagrass indicators used to calculate the seagrass condition index: a. average (circles, ±SE) seasonal abundance (per cent cover) and GAM plots of seagrass abundance trends for each location (coloured lines) and the region (black line with grey shaded area defining 95 per cent confidence intervals); b. average

annual resilience score (±SE) and trends for each location (coloured lines). Colour of circles represents the number of sites assessed to calculate the average.

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. Between 2008–09 and 2010–11, losses occurred as a result of multiple consecutive years of above-average rainfall (river discharge) and severe weather (cyclone Yasi). From 2011, seagrass rapidly recovered. However, since 2014, recovery has abated, with seagrass abundance progressively declining at reef (intertidal and subtidal) habitats since 2015. In 2017–18, coastal habitats increased to their highest abundance since 2001, immediately followed by large declines in 2018–19. Declines in abundances occurred across the region in 2018–19, with the largest losses in reef subtidal and coastal intertidal habitats. Declining abundances continued into 2019–20, either stabilising or improving slightly in 2020–21. Reef intertidal habitats showed the only improvement in 2020–21.

An examination of the long-term abundances across the Burdekin region indicates no significant regional trend (from first measure to 2020–21), although significant trends were detected at two of the seven coastal sites. One site (SB2), which has been monitored for two decades (since 2001), showed a decreasing trend (Table 21). The other site (JR2), near Jerona (Barratta Ck, Bowling Green Bay), has been monitored for a decade, and predictably showed a significant increasing trend in abundance, as this coincides with the main recovery period after the 2010–11 regional losses. A significant long-term decline occurred at Cockle Bay, Magnetic Island (reef intertidal, MI2) since monitoring began in 2005 (Table 21).

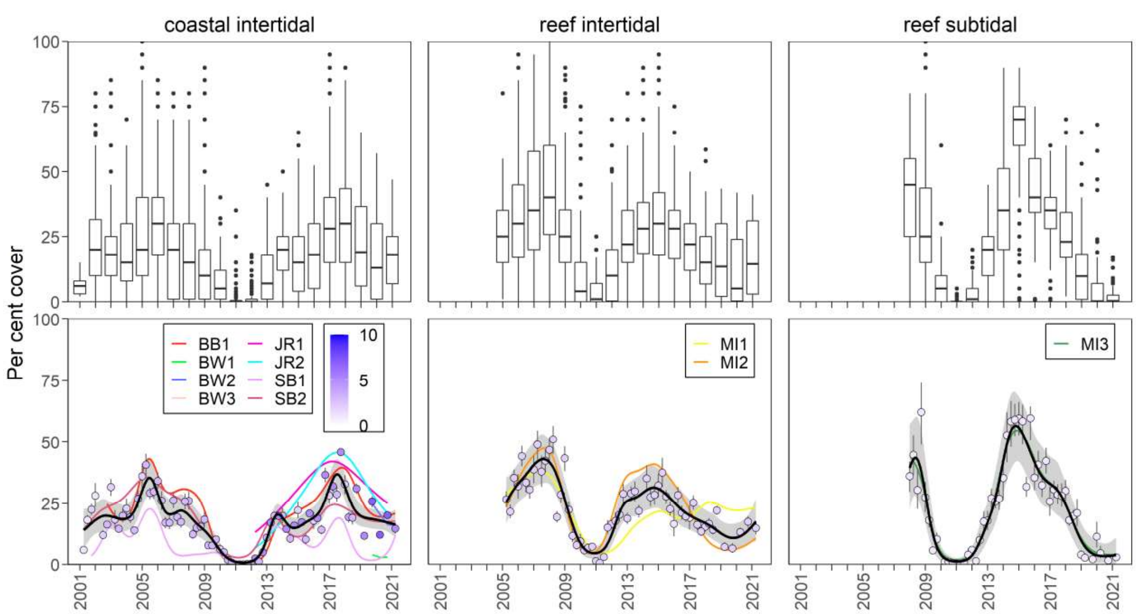


Figure 54. Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the Burdekin NRM region from 2001 to 2021. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates 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. GAM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.

This year was the first year since 2014, that the proportion of species displaying colonising traits (e.g. *H. ovalis*) increased above the Reef long-term average at reef intertidal habitats in the region (Figure 55). Coastal and reef subtidal habitats remained dominated by opportunistic species (*H. uninervis, Z. muelleri, C. serrulata*). Opportunistic foundation species have a capacity to resist stress (survive, through reallocation of resources) caused by acute disturbances (Collier *et al.* 2012b), and therefore, current species composition in coastal and reef subtidal habitats provides greater overall resilience in Burdekin meadows. The increased presence of colonising species is not surprising given the declines in seagrass abundance observed over the past few years. Colonising species are important for recovery following loss (Kilminster *et al.* 2015).

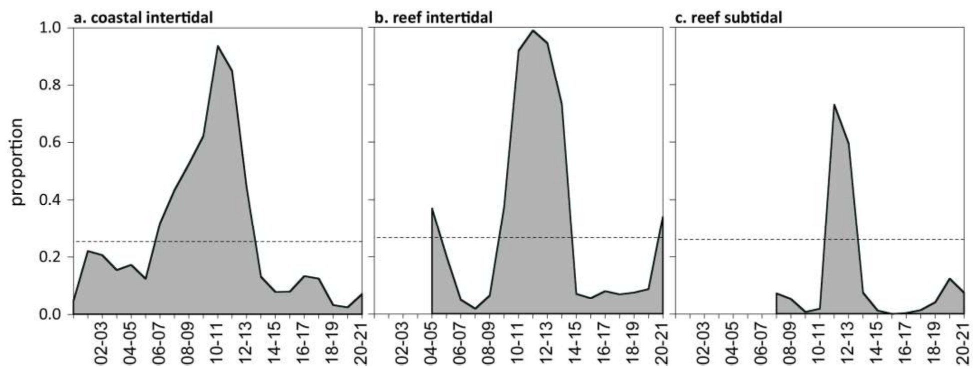


Figure 55. Proportion of seagrass abundance composed of colonising species at inshore habitats in the Burdekin region, 2001–2021. Dashed line represents Reef long-term average proportion of colonising species for each habitat type.

Meadow spatial extent improved in 2020-21 from the lowest level recorded in reef subtidal habitats in early 2020, back to extents similar to 2018, prior the flood events in early 2019 (Figure 56).

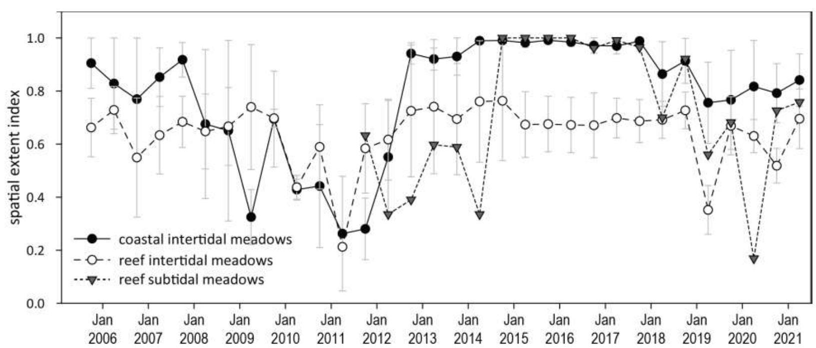


Figure 56. Change in spatial extent (± SE) of seagrass meadows within monitoring sites for each inshore intertidal habitat and monitoring period across the Burdekin region, 2005–2021.

5.3.3.3 Seagrass reproductive status

Reproductive effort was highly variable across Burdekin region habitats, particularly in coastal habitats where very high and anomalous levels of reproductive effort can occur, usually at times when abundance is also very high (Figure 57). In 2020–21, overall reproductive effort improved relative to the previous period, but was mixed across coastal and reef intertidal habitats, with both increases and losses observed. These variable responses appear species related. For example, increases occurred in coastal meadows dominated by *H. uninervis* (BB1, SB1), but losses were observed in *Zostera* dominated (JR1, JR2). Sexual reproductive structures were also depressed in reef habitats, being absent from subtidal meadows. Seed banks persisted in all habitats in 2020–21, however seed densities declined across the region. Low reproductive effort will hinder replenishment of reduced seed banks, and seed banks are therefore likely to remain low in coming years. This may limit the capacity of meadows to recover from seed should reproductive effort and seeds banks continue to decline.

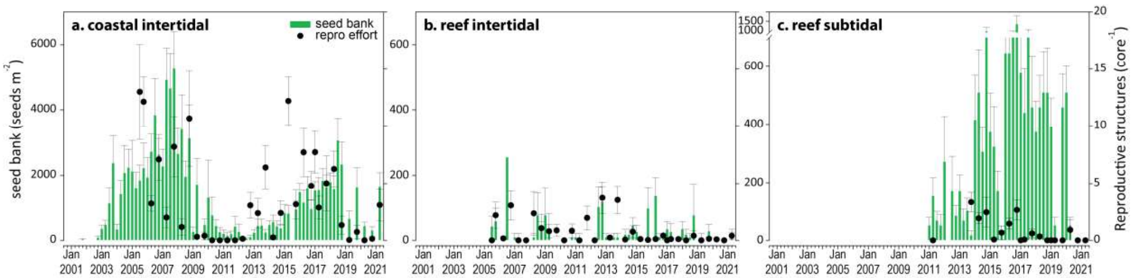


Figure 57. Seedbank and reproductive effort at inshore coastal intertidal and reef subtidal and intertidal habitats in the Burdekin region. Seed bank presented as the total number of seeds per m^2 sediment surface (green bars ±SE). Reproductive effort for the late dry season and late wet season presented 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 Resilience

The overall resilience score for the Burdekin was moderate, with large variability between habitats (Figure 58). At intertidal sites, seagrass condition exceeded abundance and composition thresholds. At coastal sites there were reproductive structures present, but at low levels compared to historical flowering densities. At reef intertidal sites there were no reproductive structures present, but there has been in recent years, and there are some persistent species present.

At the reef subtidal site the resilience score was very low (Figure 58). Abundance was below the per cent cover threshold indicating substantial loss and low levels of resistance. Furthermore, there were no reproductive structures present.

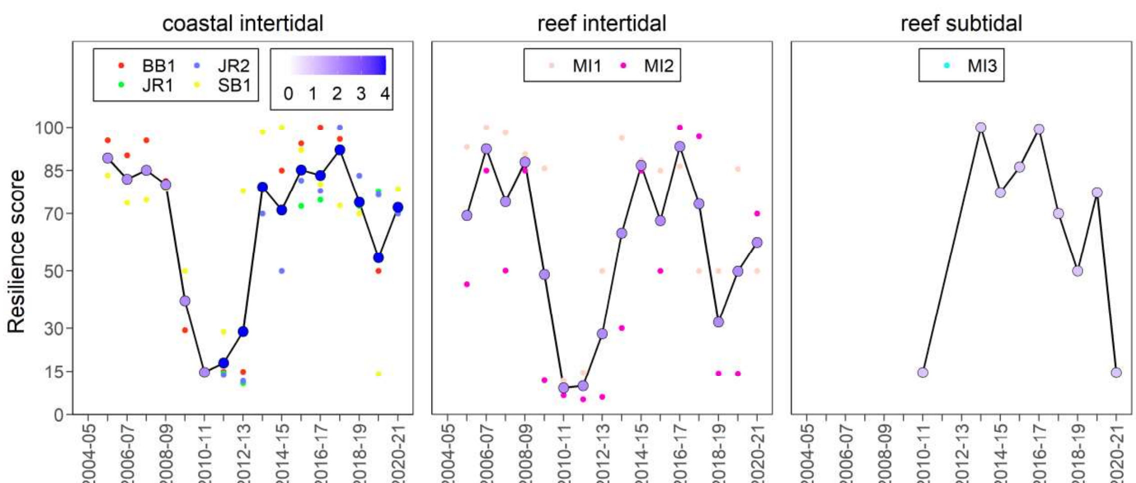


Figure 58. Resilience score in each habitat in the Burdekin, 2006–2021. Coloured small points represent different sites. Shades of blue for the larger points indicate the number of sites that contribute to the score.

5.3.3.5 Epiphytes and macroalgae

Epiphyte cover on seagrass leaf blades was slightly lower at coastal meadows in 2020–21. Unlike the previous period where cover remained above the inshore Reef average throughout, this year it declined in the wet season in coastal habitats (Figure 59). Conversely, at reef intertidal habitats, epiphyte cover was higher in the wet than the dry. Epiphyte cover on reef subtidal seagrasses were either at or below the Reef average.

Macroalgae abundance remained low and below the long-term average across the region in 2020–21 at all seagrass habitats (Figure 59). Overall, epiphyte and macroalgae cover this year has declined relative to the previous period and appears below levels which would be expected to impact light availability for seagrass growth.

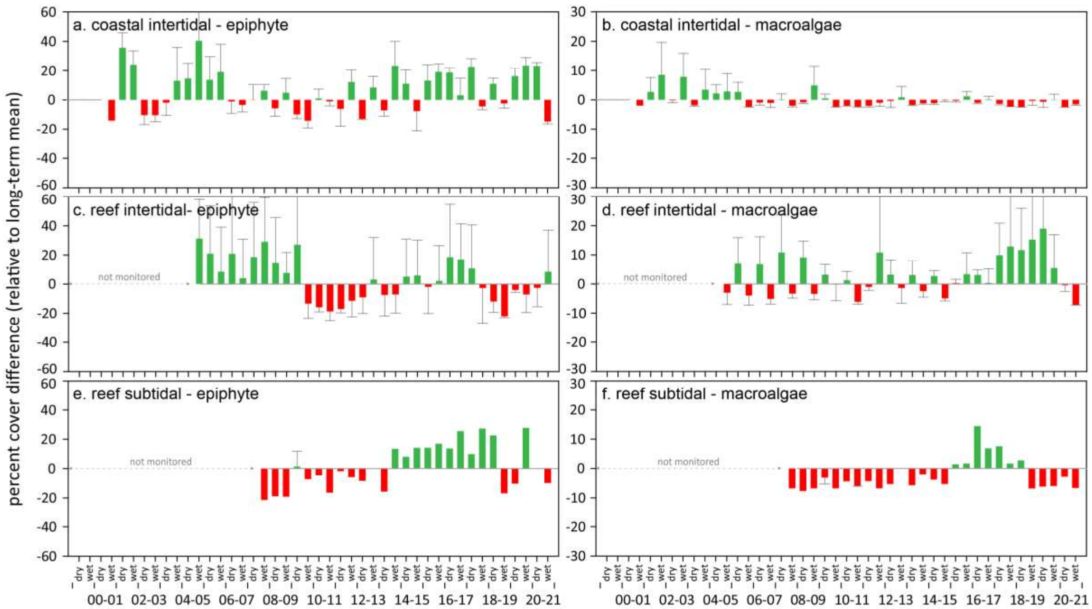


Figure 59. Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term Reef average for each inshore seagrass habitat in the Burdekin region (sites pooled, ±SE).

5.4 Mackay–Whitsunday

5.4.1 2020–21 Summary

The 2020–21 monitoring period in the Mackay–Whitsunday region was relatively benign with environmental pressures around or below the long-term averages. It was characterised by rainfall and discharge that was below the long-term average and temperatures that were around the long-term average, while daily light levels were lower than average (Figure 7, Table 10, Figure 52).

Inshore seagrass meadows across the Mackay–Whitsunday NRM region reduced in overall condition in 2020–21, and the condition grade declined to **poor** (Figure 60). There was a small decline in both indicators. Indicators for the overall condition score were:

- abundance score was poor
- resilience was poor.

Seagrass condition in the Mackay–Whitsundays is highly variable, due to a range of environmental pressures.

Seagrass abundance decreased slightly in 2020–21, with losses at 40 per cent of sites relative to the previous period. The greatest losses occurred in the coastal subtidal and reef intertidal habitats. Overall, the long-term trend indicates a declining trajectory, however improvements over the last three years indicate a region verging on recovering from the losses experienced in early 2017, but possibly hindered by localised and chronic pressures.

The overall resilience score for the Mackay–Whitsunday region was poor, and the third lowest level since records began. This was due to poor meadow condition and low or absent reproductive effort at most reef intertidal and estuarine intertidal sites. However, resilience was high at coastal sites and there were some improvements in reproductive effort at coastal sites. Reproductive effort at the estuarine site is highly variable both inter-annually and seasonally, and although no reproductive structures were observed this year, seed banks increased in the late wet season, suggesting successful sexual reproduction in the intervals between field assessments. Seeds are persisting within the seed bank of all habitats, which provides some capacity to recover from future impacts.

Up until 2016–17, the Mackay–Whitsunday regional seagrass condition had been improving from 2010–2011, when it reached its lowest level since monitoring commenced. After this time, the recovery trend abated and dropped to poor, as a consequence of cyclone Debbie in March 2017. In 2019–20, the score returned to moderate, but in in 2020–21 it once again declined to poor, with both declining abundances and resilience. Future improvement and return to a moderate or good state will depend on favourable conditions and alleviated pressures in future.

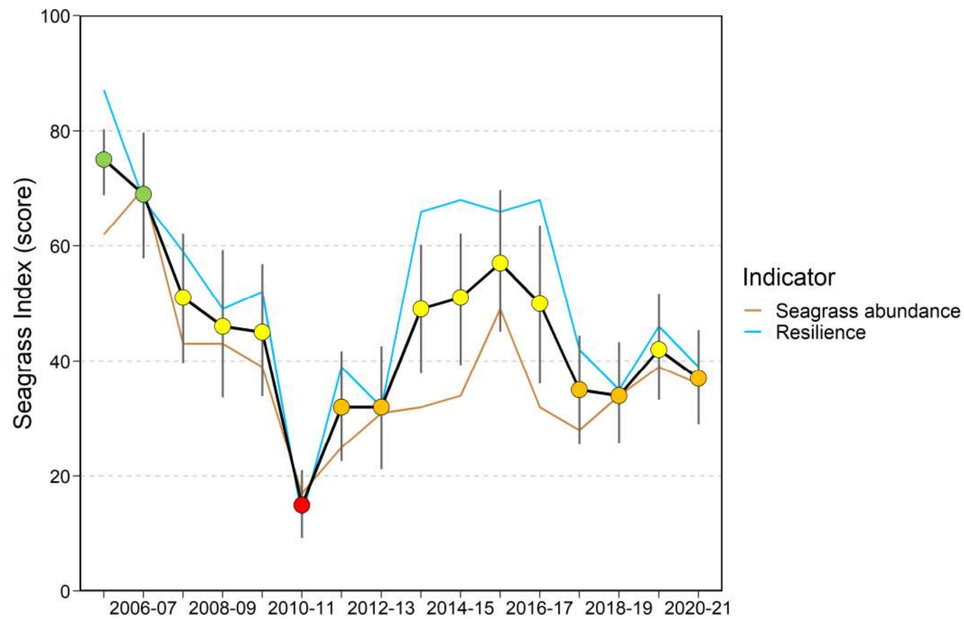


Figure 60. Report card of seagrass status indicators and index for the Mackay–Whitsunday NRM region (averages across habitats and sites). Values are indexed scores scaled from 0–100 (± SE) and graded: ● = very good (81–100), ● = good (61–80), ● = moderate (41–60), ● = poor (21–40), ● = very poor (0–20).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

5.4.2 Climate and environmental pressures

Environmental conditions were relatively favourable for seagrasses in the Mackay–Whitsunday region in 2020–21. There were no cyclones to affect the region and rainfall and river discharge were also well below the long-term average.

Exposure of inshore seagrass to turbid waters during the wet season were below the long-term average (Figure 61a, c). Exposure to either primary or secondary turbid water was also variable among seagrass habitats (Figure 61b). Estuarine and coastal sites were not only exposed to turbid waters for the entire wet season, but were the only habitats exposed to primary waters. Reef habitats fringing the mainland (HB1 and HB2) and located on offshore islands (HM1 and HM2, LN1 and LN2) were not exposed to any primary water (Figure 9, Figure 61b).

Daily light was slightly lower than the long-term average combined within the region (Figure 9, Figure 61c, Figure 104). At the site level, daily light was considerably lower in 2020–21 (11.7 mol m^{-2} d^{-1}) than average (15.4 mol m^{-2} d^{-1}) at Hamilton Island and slightly lower at Midge Point (14.9 mol m^{-2} d^{-1}) than the long-term average (15.5 mol m^{-2} d^{-1}). At Sarina Inlet, daily light throughout much of the wet season was below the long-term average for the wet season average (Figure 104).

The 2020–21 reporting period was the eighth consecutive year when intertidal within-canopy temperatures were above the long-term average, but the difference was marginal (Figure 61c). Maximum intertidal within-canopy temperatures exceeded 35°C for a total of 41 days during 2020–21, with the highest temperature recorded at 40.3°C (MP2, 2pm 24Mar21) (Figure 61e, f). Daytime tidal exposure was below the long-term average in 2020–21 at all habitats except estuarine, where sites were above average for the third consecutive year (Figure 61c, Figure 97), which may have exacerbated the stresses at these sites.

The proportion of fine grain sizes decreased in the sediments of the seagrass monitoring sites with distance from the coast in the Mackay–Whitsunday region. The proportion of mud in estuarine sediments varied in 2020–21 relative to the previous period, either increasing or remaining below the overall inshore Reef long-term average (Figure 115). Coastal habitat meadows generally had less mud than estuarine habitats over the long term, but fluctuate within and between both meadows and years. In 2020–21 most sites contained a higher proportion of mud than the Reef long-term average (Figure 116). Reef habitats were composed predominately of fine to medium sand, with little change in 2020–21 relative to the previous period (Figure 117).

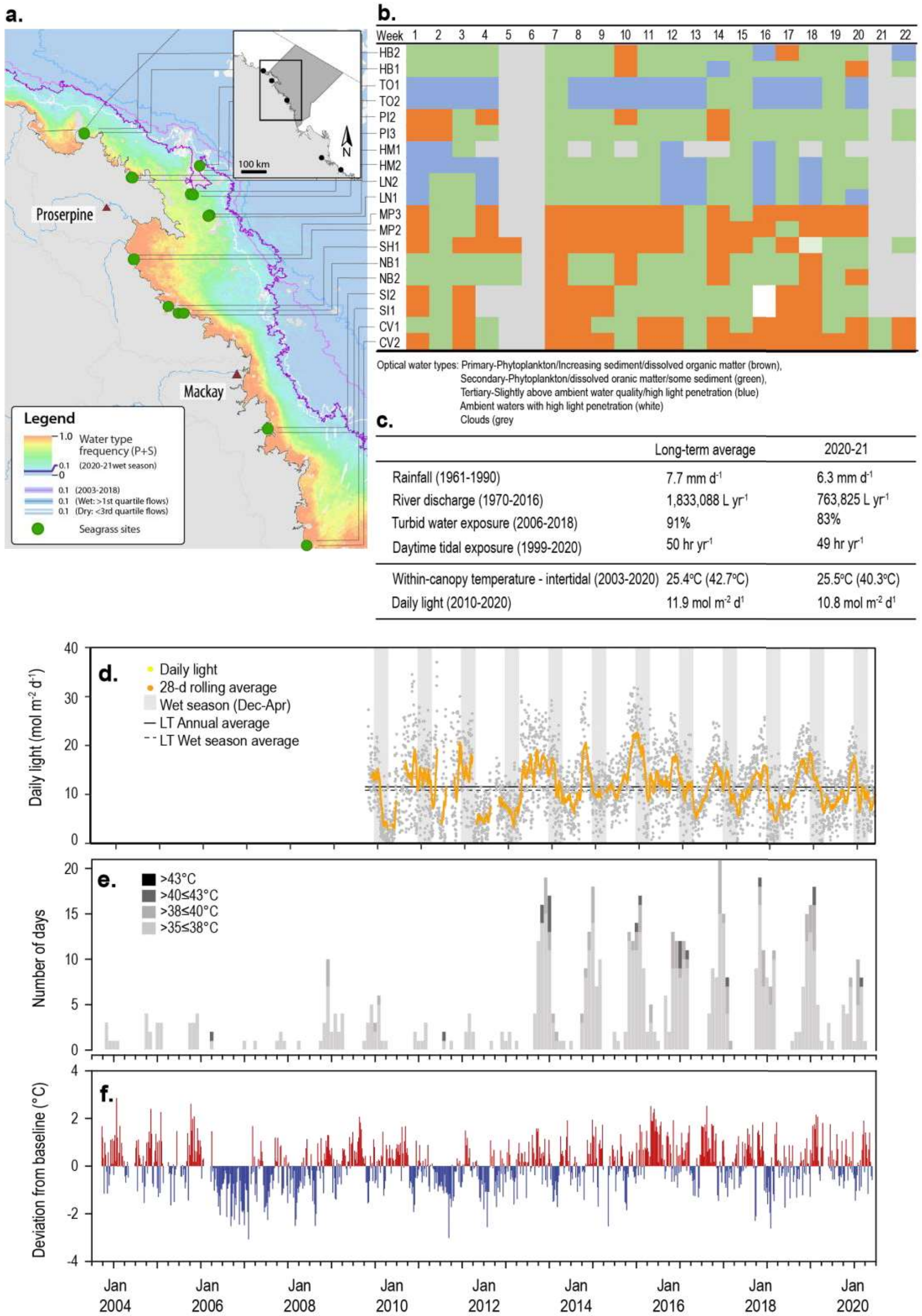


Figure 61. Environmental pressures in the Mackay–Whitsunday NRM region including: a. frequency of exposure to primary and seconday water from December 2020 to April 2021 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed) (white = no data), also showing the long-term average (2003–2018) exposure boundary (purple line), and the first (blue line) and third quartile (white line) of the long-term average (from Moran *et al.* 2022); b. wet season water type at each site; c. average conditions and max temperature over the long-term and in 2020–21; 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 at intertidal sites.

5.4.3 Inshore seagrass and habitat condition

Five seagrass habitat types were assessed across the Mackay–Whitsunday region this year, with data from 19 sites (Table 15, Table 19).

Table 15. List of data sources of seagrass and environmental condition indicators for each seagrass habitat type in the Mackay–Whitsunday NRM region. Blank cells indicate data not usually collected/measured at site. † drop camera sampling (QPWS), *Seagrass-Watch. For site details see Table 4 and Table 5.

Habitat	Site		abundance	composition	distribution	reproductive effort	seed banks	leaf tissue nutrients	meadow sediments	epiphytes	macroalgae
estuarine 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 2020–21 monitoring period, the Mackay–Whitsunday region seagrass condition index decreased from the previous year, falling back to a **poor** grading (Figure 62).

In 2019–20, the score returned from poor to moderate, but in 2020–21 it once again declined to poor, with both declining abundances and resilience. Future improvement and return to a moderate or good state will depend on favourable conditions and alleviated pressures in future.

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 Tropical cyclone Debbie (Figure 62). The following year both abundance and resilience declined, and in 2018–19 reached its lowest level since 2012–13, driven by declining resilience. During the last monitoring period, both abundance and resilience improved, raising the grade to moderate. However, in 2020–21, the overall score declined and the grade fell back to poor, due to both declining abundances and resilience.

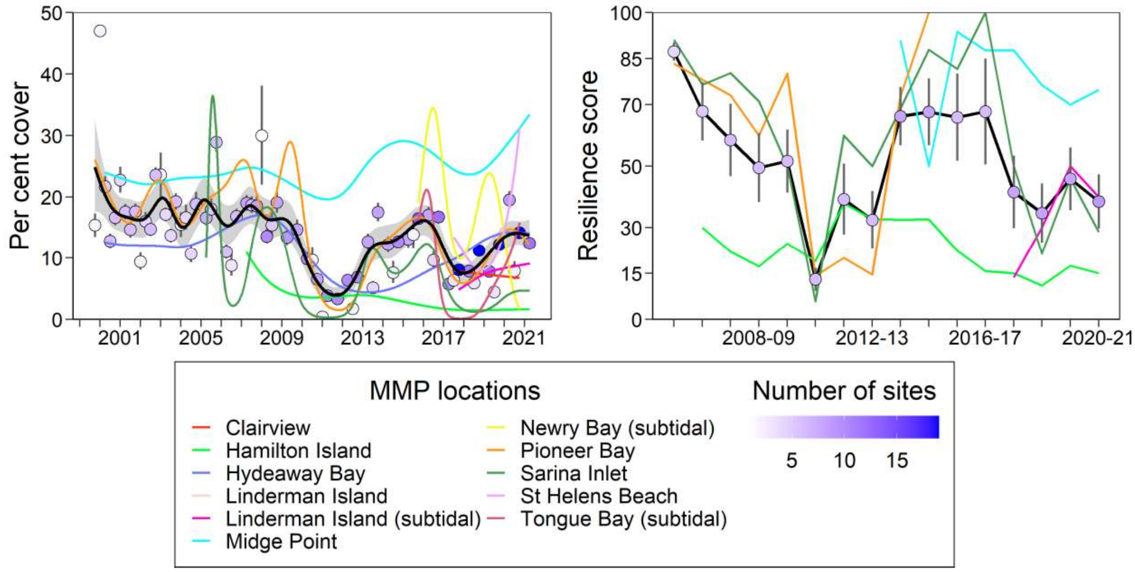


Figure 62. Temporal trends in the Mackay–Whitsunday seagrass indicators used to calculate the seagrass condition index: a. average (circles, ±SE) seasonal abundance (per cent cover) and GAM plots of seagrass abundance trends for each location (coloured lines) and the region (black line with grey shaded area defining 95 per cent confidence intervals); b. average annual resilience score (±SE) and trends for each location (coloured lines). Colour of circles represents the number of sites assessed to calculate the average.

5.4.3.2 Seagrass abundance, community and extent

Overall seagrass abundance decreased in the Mackay–Whitsunday region in 2020–21, with losses at 40 per cent of sites across the region, relative to the previous period; negating some of the improvements over the previous period (Figure 63). Conversely, gains were observed at only 25 per cent of sites, with the remained of sites unchanged. The largest losses were observed in coastal subtidal habitats, followed by reef intertidal. The largest gains were in estuarine habitats, which have struggled to recover since the catastrophic losses in early 2011, further enduring extreme climatic events such as cyclone Debbie in early 2017, which negated most of the gains made over the prior six years.

Seagrass abundance (per cent cover) in the Mackay–Whitsunday region in 2020–21 was higher in coastal habitats (intertidal = 18.6 ±1.8 per cent, subtidal = 18.7 ±2.1 per cent) than reef (intertidal = 9.2 ±1.5 per cent, subtidal = 8.1 ±0.8 per cent) or estuarine habitats (5.8 ±1.6 per cent), respectively. Seagrass per cent cover differed seasonally in estuarine meadows over 2020–21, being higher in the late dry than late monsoon (7.0 ±1.7 per cent, and 3.6 ±1.0 per cent, respectively). Little or no change was detected between seasons in all other habitats within 2020–21 (Figure 63).

Seagrass abundance at estuarine and coastal 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 63). The regional long-term trend indicates a declining trajectory (Table 21), with a region verging on recovering from losses in the years leading up to 2010–11 and in early 2017.

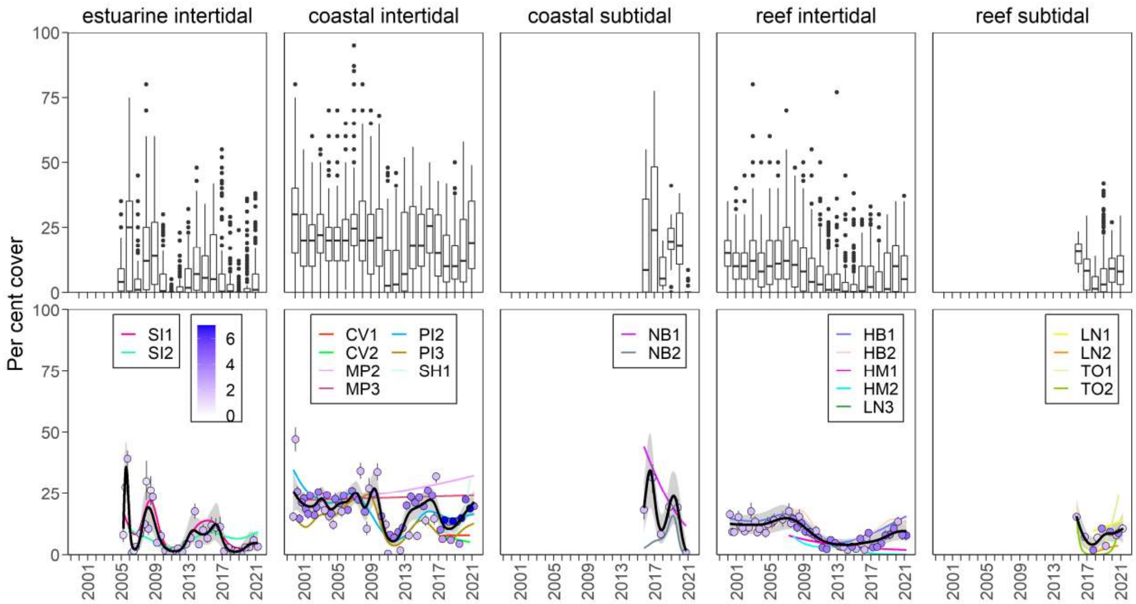


Figure 63. Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the Mackay–Whitsunday NRM region from 1999 to 2021. Whisker plots (top) show thee box representing the interquartile range of values, where the boundary of the box closest to zero 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. GAM 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 *H. uninervis* and *Z. muelleri*, mixed with the colonising species *H. ovalis*. Colonising species dominated intertidal meadows across the Mackay–Whitsunday region in the first few years following the extreme weather in 2011. In the last three years, there has been a reduction in colonising species in coastal and reef habitats. In all habitats, opportunistic foundational species (*H. uninervis* and *Z. muelleri*) now dominate (Figure 64), suggesting meadows may have an improved ecosystem resistance to tolerate disturbances (Figure 64). In contrast, colonising species in intertidal estuarine habitats (Sarina Inlet), increased above the Reef long-term average in 2020–21 (Figure 64).

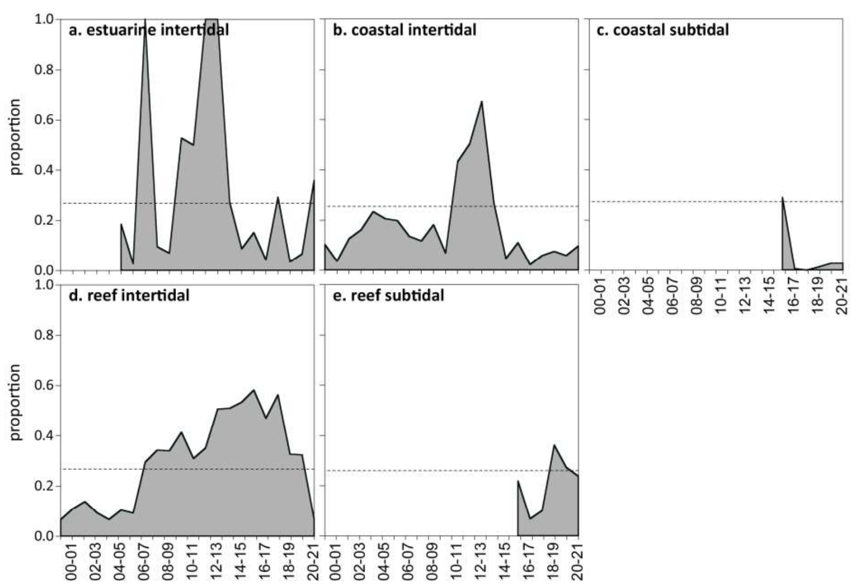


Figure 64. Proportion of seagrass abundance composed of colonising species at inshore intertidal habitats in the Mackay–Whitsunday region, 1999–2021. Dashed line represents Reef long-term average proportion of colonising species for each habitat type.

Seagrass meadow landscape mapping was conducted within all sentinel monitoring sites in October 2020 and the majority of sites in April 2021 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 at reef intertidal meadows following the declines experienced in 2016–2017 as a consequence of the destructive effects of cyclone Debbie. At estuarine and coastal meadows, extent remained steady, with only slight increases relative to the previous period (Figure 65).

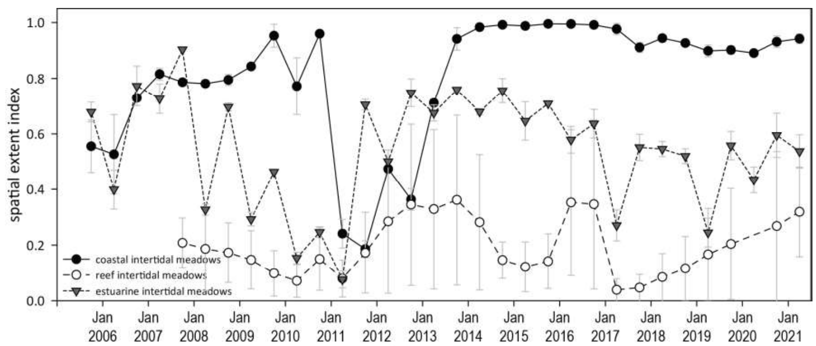


Figure 65. Change in spatial extent (± SE) of seagrass meadows within monitoring sites for each inshore intertidal habitat and monitoring period across the Mackay–Whitsunday NRM region.

5.4.3.3 Seagrass reproductive status

Reproductive effort was highly seasonal and highly variable between years and seagrass habitats in the Mackay–Whitsunday region, but declined slightly overall in 2020–21 (Figure 66). Reproductive effort and seed banks improved slightly in coastal habitats, relative to the previous period. At the estuarine meadow (Sarina Inlet), sexual reproductive structures were not observed during 2020 or in early 2021, however seed banks increased in 2020–21, suggesting the occurrence of flowers, fruits or spathes in the intervals between field assessments. In contrast, reproductive effort and the seeds density continued to remain very low at reef sites in 2020–21, which appears typical for reef habitat meadows (Figure 66).

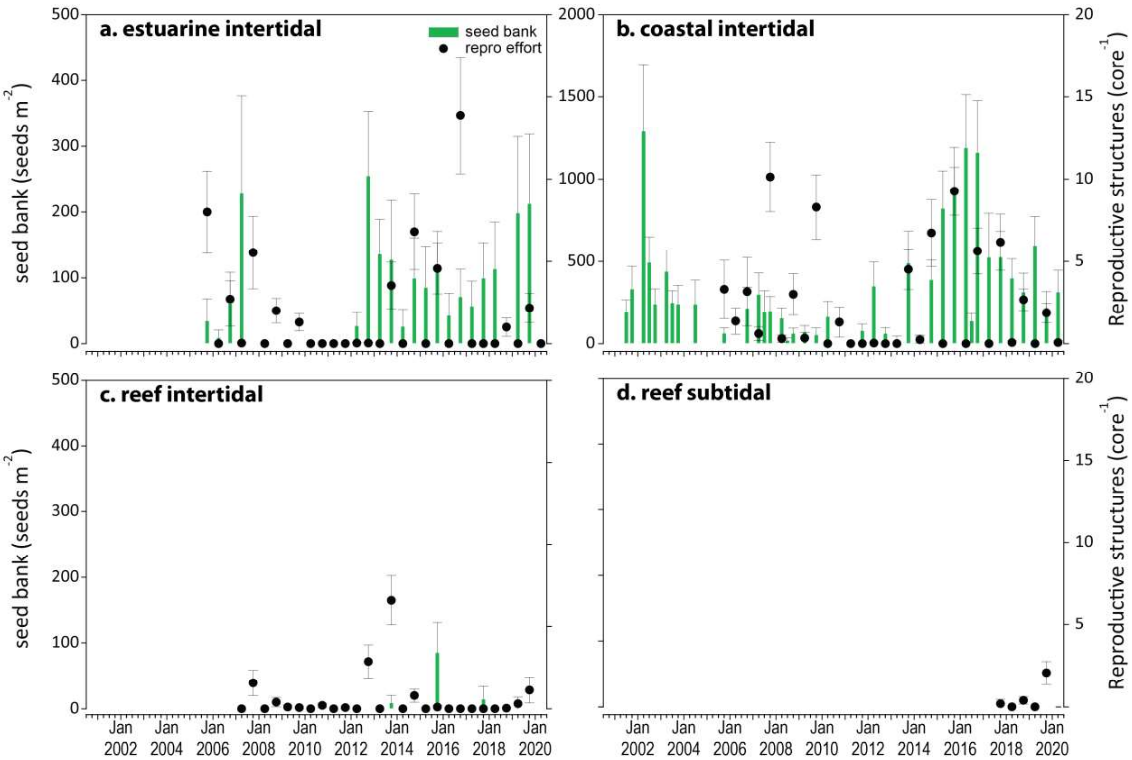


Figure 66. Seed bank and reproductive effort at inshore estuarine intertidal, coastal intertidal and reef intertidal and subtidal habitats in the Mackay–Whitsunday region, 2001–2021. Seed bank presented as the total number of seeds per m^2 sediment surface (green bars ±SE), and late dry season reproductive 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.

5.4.3.4 Resilience

The overall resilience score for the Mackay–Whitsunday region was poor, and the third lowest level since 2005–06 (Figure 67). However, resilience was high at coastal sites, and low at estuarine and reef sites. At coastal intertidal sites, meadow condition was good, indicating that the meadows will have high resistance to disturbances, and reproductive structures were present, but at low numbers compared to historical levels. Resilience at Pioneer Bay (PI2 and PI3) was variable, but is no longer assessed. Since resilience has been measured at Midge Point (MP2, MP3) in 2012–13, resilience has been stable.

At estuarine sites at Sarina Inlet, meadow condition at SI1 was below critical thresholds indicating low levels of resistance and no reproductive structure were present. Condition was better at SI2, and there was recent history of reproductive effort.

At reef intertidal sites, at HM2 there was no seagrass, while at HM1 and LN3, meadow condition was above critical thresholds but there were no reproductive structures in 2020–21, or in the previous three years.

At the reef subtidal sites, meadow condition had inadequate levels of resistance based on abundance threshold and composition. There were no reproductive structures observed in 2020–21, but there had been some observed at LN1 in the past three years.

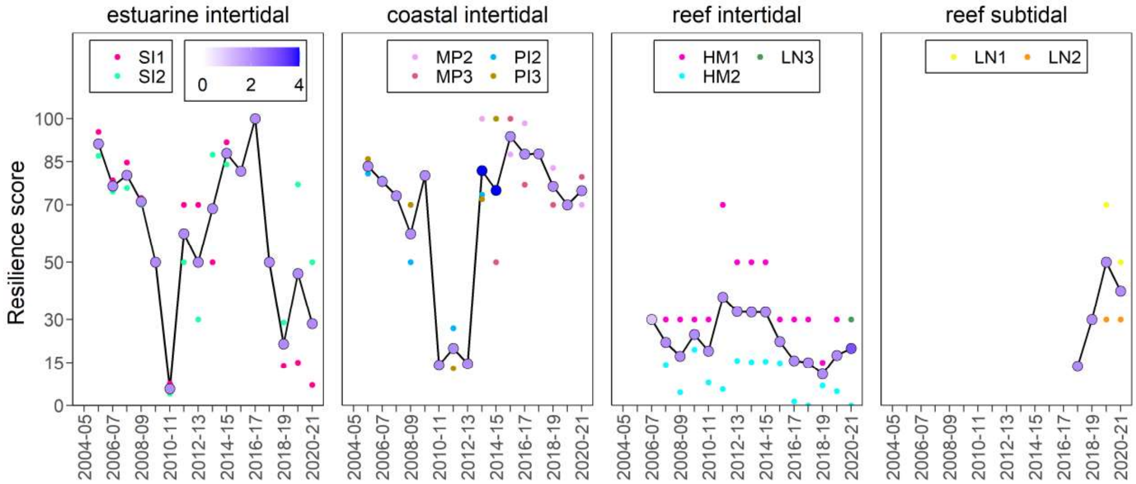


Figure 67. Resilience for each habitat type in the Mackay–Whitsunday region, 2006–2021. Coloured small points represent different sites. Shades of blue for the larger points indicate the number of sites that contribute to the score.

5.4.3.5 Epiphytes and macroalgae

Epiphyte cover on seagrass leaf blades in 2020–21 has remained below the overall inshore Reef long-term average at coastal and reef intertidal habitats, and increased above at reef subtidal habitats. At the estuarine meadow in Sarina Inlet, epiphyte cover increased above the Reef long-term average during the late dry, but returned to below average within six months (Figure 68).

Percentage cover of macroalgae remained unchanged, at or below the overall inshore Reef long-term average for estuarine and coastal intertidal habitats throughout 2020–21 (Figure 68). At coastal subtidal habitats, macroalgae cover remained above the Reef long-term average and increased slightly, while at reef intertidal and subtidal meadows, macroalgae cover remained above for much of 2020–21 (Figure 68).

Figure 68. 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, 1999–2021 (sites pooled, ±SE).

5.5 Fitzroy

5.5.1 2020–21 Summary

Environmental conditions were relatively benign in 2020–21, with conditions generally better than the long-term average levels for the region. Rainfall and river discharge were below average, and daily light levels were slightly higher than average. Average annual water temperature was around the average, but there were a number of high temperature days, including two days when temperature exceeded 40°C, a threshold likely to impart stress on all species, and in particular on *Z. muelleri*.

Overall, the seagrass condition score for the Fitzroy NRM region reduced from moderate to **poor** in 2020–21 (Figure 69). Both indicators declined:

- abundance score was poor
- resilience was poor.

Seagrass abundance declined at half of the sites across the Fitzroy region in 2020–21, with the remaining sites marginally increasing relative to the previous period. The largest declines were at the estuarine sites in Gladstone Harbour. 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. In Shoalwater Bay, the coastal sites varied with increases at one site offset by decreases at the other. The long-term trend in the seagrass abundance score across the region is largely unchanged over the past few years.

Overall resilience in the Fitzroy region was poor but varied among habitats. Reproductive effort remains well below historical peaks for all habitats in the region. However, the consistent presence of some reproductive structures, albeit low, 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 and no seed bank is present, limiting the meadows capacity to recover.

Inshore seagrass meadows across the region continue to 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 estuarine habitats had been improving until this year, while other habitats demonstrate a legacy of reduced resilience.

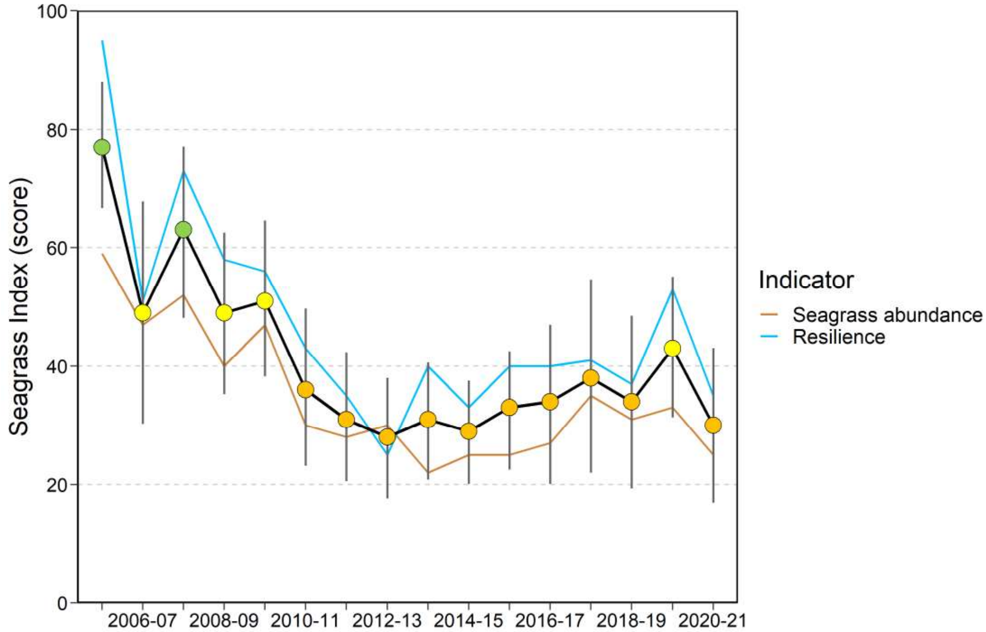


Figure 69. Report card of seagrass status index and indicators for the Fitzroy NRM region (averages across habitats and sites). Values are indexed scores scaled from 0–100 (± SE) and graded: ● = very good (81–100), ● = good (61–80), ● = moderate (41–60), ● = poor (21–40), ● = very poor (0–20). NB: Scores are unitless.

5.5.2 Climate and environmental pressures

Rainfall in 2020–21 was below the long-term average for the Fitzroy region, and river discharge was less than half the annual median for the region (Figure 70c). Exposure of inshore seagrass to turbid waters during the wet season was also lower than the long-term average in 2020–21 (Figure 70c). Of the turbid waters, there was relatively more frequent secondary waters and relatively less exposure to primary waters that are richer in fine suspended sediments (Figure 70a, b).

Annual daily light availability was also higher in 2021–21 than the long-term average for the region (Figure 9, Figure 70c, d). This was due to improvement in daily light at all locations (Figure 105). Daytime tidal exposure was above the long-term average for the region, which increases the risk of desiccation stress, but in the turbid shallow waters can provide windows of light for photosynthesis (Figure 98).

2020–21 within-canopy temperatures were similar to the previous period and the long-term average (Figure 70c). Maximum intertidal within-canopy temperatures exceeded 35°C for a total of 44 days during 2020–21, with the highest ever temperature recorded in the region at 41.9°C (GK2, 3pm 10Apr21) (Figure 70e). Daytime tidal exposure in 2020–21 was below the long-term average at coastal and reef habitats, but above at coastal habitats for the sixth consecutive year (Figure 70c, Figure 97), which may have exacerbated stresses experienced at these intertidal sites.

The proportion of fine sediment grains in meadows generally decreases with distance from the coast/river mouths. Estuarine sediments were composed primarily of finer sediments, with the mud portion fluctuating around the overall inshore Reef long-term average (Figure 119). Coastal and reef habitat sediments are dominated by fine sand/sand, with the proportion of mud in coastal habitats marginally increasing in 2020–21 (Figure 120, Figure 121).

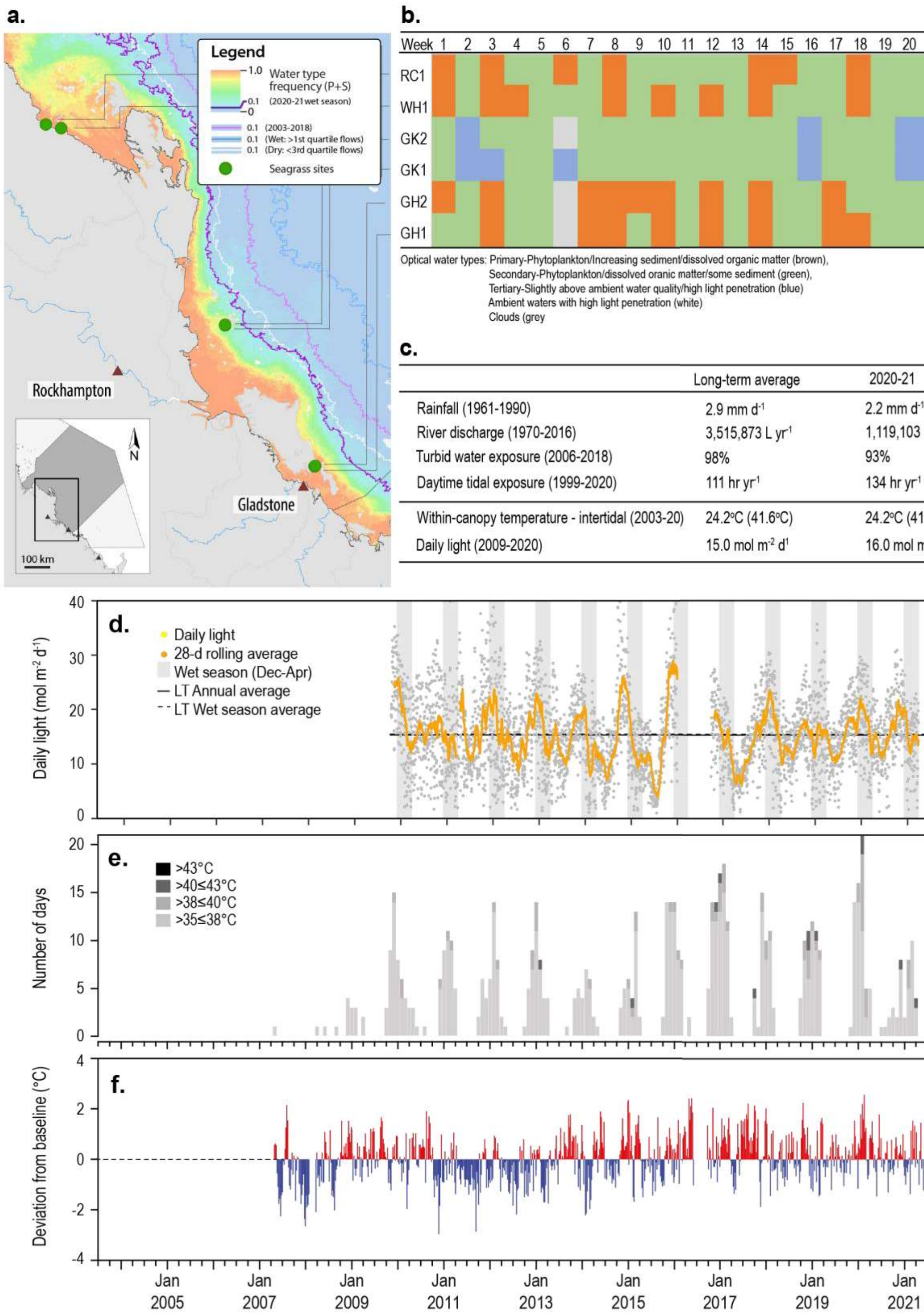


Figure 70. Environmental pressures in the Fitzroy region including: a. frequency of exposure to primary and seconday water from December 2020 to April 2021 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed) (white = no data), also showing the long-term average (2003–2018) exposure boundary (purple line), and the first (blue line) and third quartile (white line) of the long-term average (from Moran *et al.* 2022); b. wet season water type at each site; c. average conditions and max temperature over the long-term and in 2020–21; 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 at intertidal sites.

5.5.3 Inshore seagrass and habitat condition

Three seagrass habitat types were assessed across the Fitzroy region in 2020–21, with data from 6 sites (Table 16).

Table 16. 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 4 and Table 5.

Habitat	Site		abundance	composition	distribution	reproductive effort	seed banks	leaf tissue nutrients	meadow sediments	epiphytes	macroalgae
estuarine intertidal	GH1	Gladstone Hbr	■	■	■	■	■	■	■	■	■
	GH2	Gladstone Hbr	■	■	■	■	■	■	■	■	■
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 2020–21 monitoring period, the seagrass condition index declined from a moderate to a **poor** grading; reversing the improving trend since 2014–15 (Figure 71)

The abundance score decreased to the lowest level in five years, but remained poor (Figure 71). In 2020–21, the resilience score had the largest annual decrease (18 points) since 2007, declining from the highest score since 2009–10, to the fourth lowest since monitoring commenced (Figure 71). This was primarily driven by declining abundances and resilience at the Gladstone Harbour meadow on Pelican Banks.

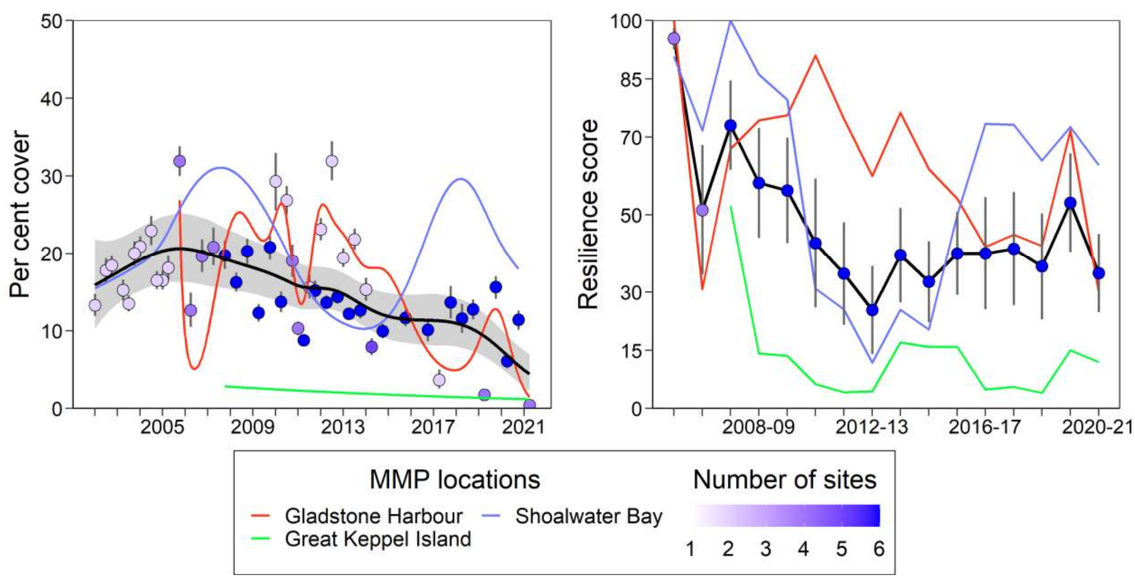


Figure 71. Temporal trends in the Fitzroy seagrass indicators used to calculate the seagrass condition index: a. average (circles, ±SE) seasonal abundance (per cent cover) and GAM plots of seagrass abundance trends for each location (coloured lines) and the region (black line with grey shaded area defining 95 per cent confidence intervals); b. average annual resilience score (±SE) and trends for each location (coloured lines). Colour of circles represents the number of sites assessed to calculate the average.

5.5.3.2 Seagrass abundance, composition and extent

In 2020–21, seagrass abundance across the Fitzroy region declined to the lowest level since monitoring was established. Seagrass abundance at estuarine and coastal intertidal habitats have fluctuated greatly between years over the life of the monitoring program, with some sites experiencing total or near total loss followed by recovery (Figure 72). In 2020–21, seagrass abundance in estuarine and coastal habitats declined from the previous period, however, reef habitats, which have been below 3 per cent cover since the onset of monitoring, marginally increased. Seagrass abundance (per cent cover) in the Fitzroy region in 2020–21 was significantly higher in coastal (22.1 ±0.9 per cent) habitats than estuarine (3.9 ±1.5 per cent), and reef habitats (0.9 ±0.4 per cent) (Figure 72). Seagrass abundances across all habitats were higher in the late dry than the late wet season (e.g. estuarine meadow in Gladstone Harbour, 7.0 ±2.1 per cent and 0.7 ±0.4 per cent, respectively).

Examination of the long-term trend in seagrass abundance (per cent cover) across the region reveals a significant decrease (Figure 71, Table 21). These decreases have primarily occurred in the estuarine and coastal habitats, although two thirds of all monitoring sites in the region (including coastal) show no significant trend (Table 21).

Seagrass abundance in the estuarine habitat was increasing in 2017–18 and 2018–19, as meadow integrity (e.g. reduced scarring) improved due to reduced sediment movement and bioturbation. However, the cause of the recent decline is unclear and may require further investigation.

In the north of the region, coastal sites receive low river discharge, however, the meadows were still exposed to primary sediment laden waters for much of the year. These 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 30 days in Shoalwater Bay during 2020–21, with a highest temperature of 41.2°C. The high temperatures are particularly stressful for *Zostera muelleri* communities which dominate the coastal meadows 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). Water temperatures at Pelican banks in Gladstone Harbour exceeded 35°C (max 36.9) for only 6 days in 2020–21, which was much lower than the previous period and less likely to have placed a substantial stress on these *Z. muelleri* dominated seagrass meadows.

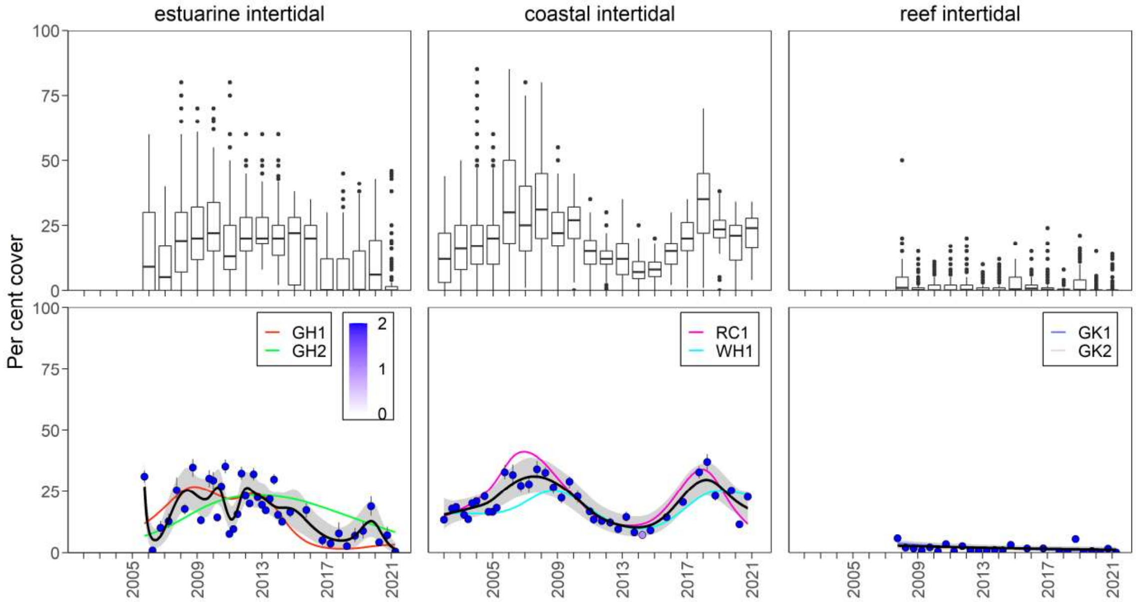


Figure 72. Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the Fitzroy NRM region from 2002 to 2021. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates 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. GAM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.

The seagrass species in the coastal meadows in Shoalwater Bay (Ross Creek and Wheelans Hut) have returned to compositions dominated by the opportunistic species *Z. muelleri* and *H. uninervis*, with the lowest proportion of colonising species (*H. ovalis*) since 2005. The proportion of colonising species (*H. ovalis*) peaked after the extreme climatic events of 2011, and has gradually been declining since (Figure 73). In 2020–21, the proportion of these opportunistic species similarly remained very low at estuarine sites (Figure 73) which continued to be dominated by *Z. muelleri*. However, colonising species continued to dominate the reef habitat sites (well above the overall inshore Reef long-term average), which appears a direct relationship with decreased abundances over the last few years (Figure 73).

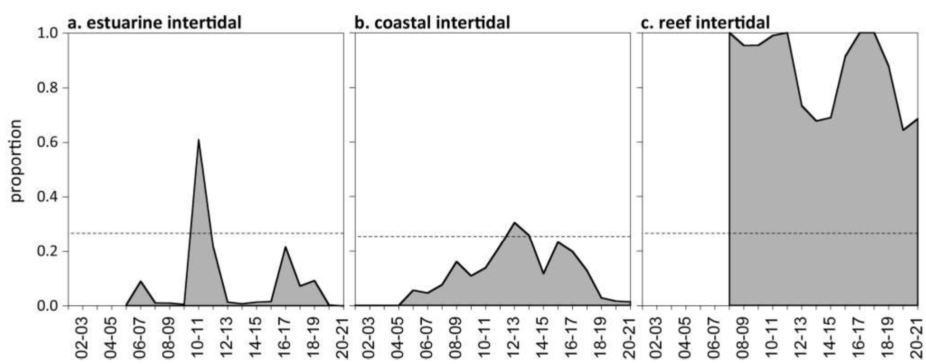


Figure 73. Proportion of seagrass abundance composed of colonising species in inshore intertidal habitats of the Fitzroy region, 2001–2021. Dashed line represents Reef long-term average proportion of colonising species for each habitat type.

The extent of the coastal meadows within monitoring sites in Shoalwater Bay changed little since monitoring commenced in 2005. Conversely, the extent of the estuarine meadows at Pelican Banks in Gladstone Harbour fluctuated from 2015–16 when there was a large reduction in one of the sites due to extensive scarring and sediment deposition. In 2019–20, the sediment deposition abated and the meadow was showing signs of recovering with shoot extension and improved meadow cohesion. However, in 2020–21 the entire meadow seascape deteriorated (Figure 74), with increased erosion along drainage channels and increased scarring. Meadows on the reef flat at Great Keppel Island remained highly fragmented after the 2015–16 losses and continued to show little sign of recovery in 2020–21.

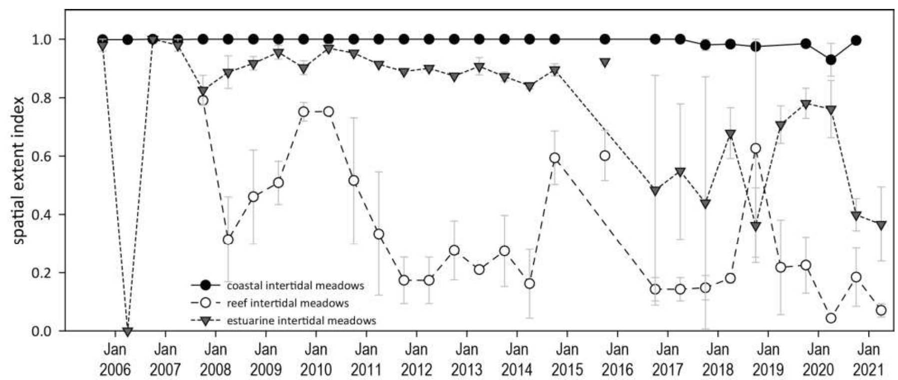


Figure 74. Change in spatial extent (± SE) of seagrass meadows within monitoring sites for each inshore intertidal habitat across the Fitzroy NRM region, 2005–20.

5.5.3.3 Seagrass reproductive status

Reproductive status varied seasonally and inconsistently between years and across habitats in the Fitzroy region over the life of the MMP (Figure 75). Reproductive effort was higher in the late dry season and although remained steady at coastal and estuarine sites since 2017, the number of sexually reproductive structures remained below pre-2011 levels (Figure 75). A seed bank also persisted at coastal and estuarine sites since 2012, although densities were near the lowest levels in 2020–21. Reproductive effort remained very low at reef sites in 2020–21, together with an absent seed bank (Figure 75). No seeds have ever been observed in the reef meadows at Great Keppel Island. This limits the meadow capacity to recover making them highly vulnerable to future disturbances. The absence of seeds in the reef meadows was likely the result of the chronic and greatly depressed reproductive effort. Other possible explanations for the low seed bank include failure to set seed, or rapid loss of seeds from germination or grazing (Heck and Orth 2006).

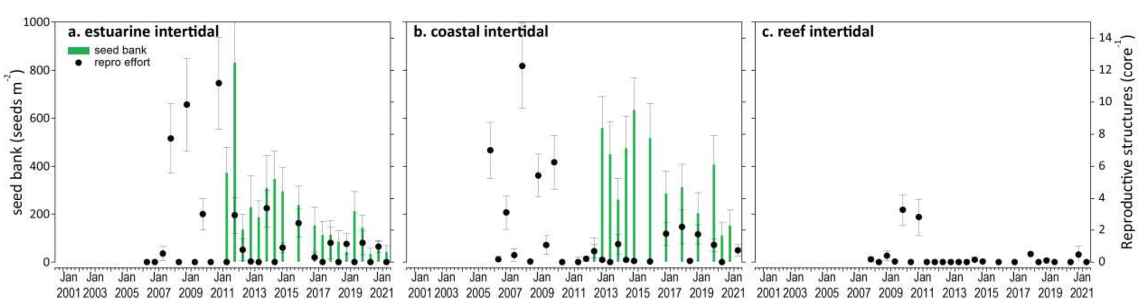


Figure 75. Seedbank and reproductive effort at inshore intertidal coastal, estuarine and reef habitats in the Fitzroy region, 2005–2021. Seed bank presented as the total number of seeds per m^2 of sediment surface (green bars ±SE). Reproductive effort for the late dry season presented as the average number of reproductive structures per core (species and sites pooled) (dots ±SE).

5.5.3.4 Resilience

Overall resilience in the Fitzroy region was poor but varied among habitats (Figure 76).

At estuarine intertidal habitats meadow condition was below critical thresholds for resistance due to very low overall abundance (<20th percentile) and so were in category 1.1. However, the species composition consisted of opportunistic species (no colonisers present), and reproductive structures were present.

At coastal intertidal sites, overall condition for species composition and abundance exceeded thresholds indicating meadows were resistant to disturbances. There were reproductive structures present in low numbers at WH1 but none at RC1, although there had been in the previous three years.

At reef intertidal sites resilience was low. Both sites were dominated by colonising species and had low abundances, indicating meadows with low levels of resistance to disturbances. There were reproductive structures of opportunistic species present at GK2, but not at GK1.

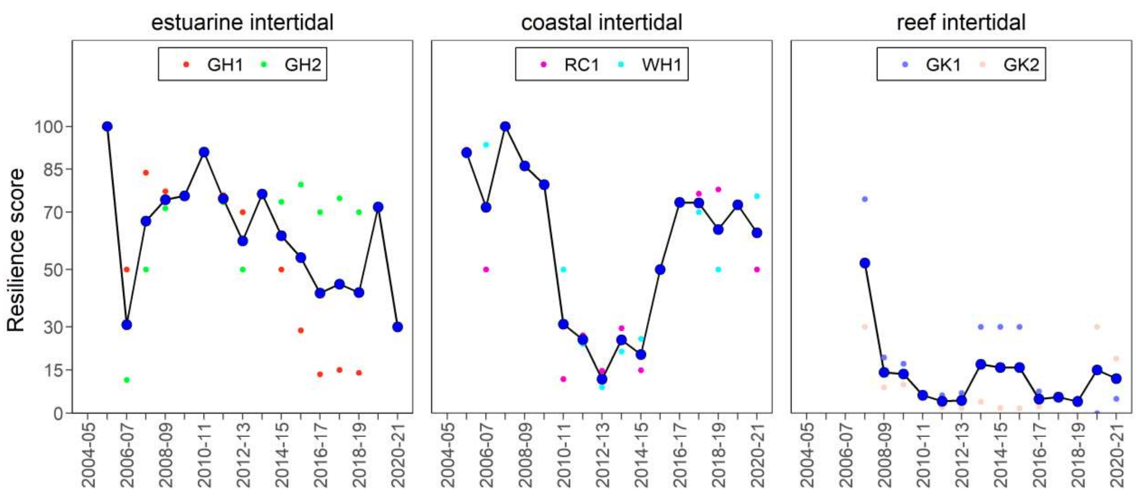


Figure 76. Resilience in each habitat in the Fitzroy region 2006–2021. Coloured small points represent different sites. Shades of blue for the larger points indicate the number of sites that contribute to the score.

5.5.3.5 Epiphytes and Macroalgae

Epiphyte cover on seagrass leaves generally decreased across the region in 2020–1, with covers below the overall inshore Reef long-term average for most habitats (Figure 77). The only significant increase in epiphyte cover was during the late dry season in the reef meadows at Great Keppel Island. This was the first time in seven years that epiphyte abundance was above the long-term average.

Macroalgae cover remained very low and below the overall inshore Reef long-term average at all habitats in the Fitzroy region, for the second consecutive year (Figure 77).

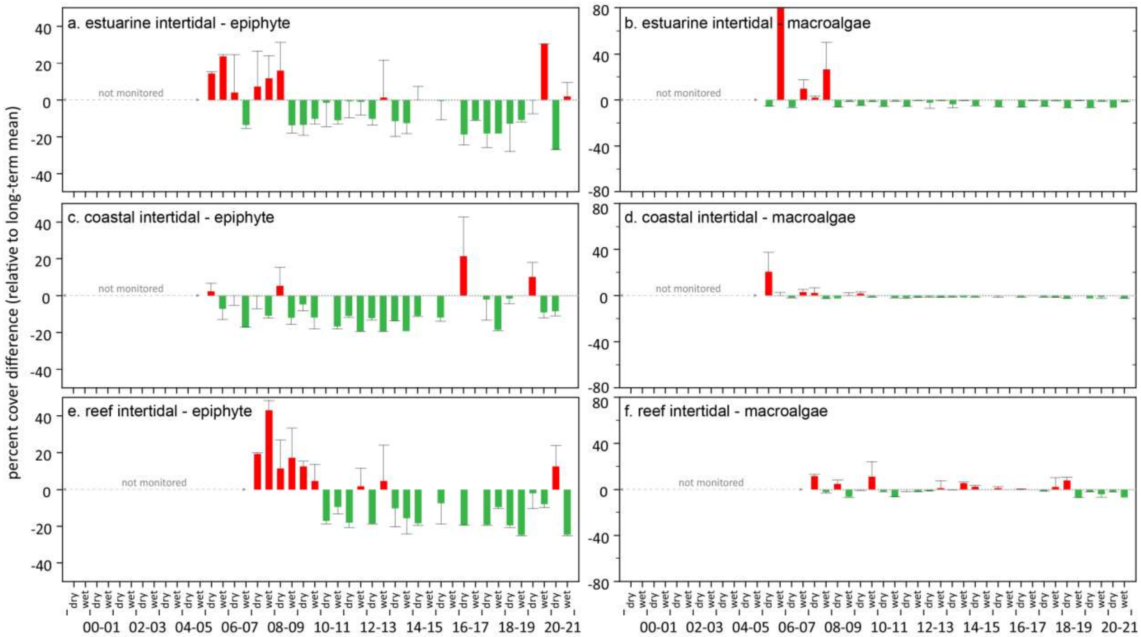


Figure 77. Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average (2005-2018) for each inshore intertidal seagrass habitat in the Fitzroy region, 2005–2021 (sites pooled, ±SE).

5.6 Burnett–Mary

5.6.1 2020–21 Summary

Environmental conditions were generally moderate in 2020–21, with rainfall and river discharge well below average, and yet all sites continued to be exposed to high frequencies of optically turbid water during the wet season. Daily light was around average for the region as a whole. Within-canopy temperature in 2020–21 was around the long-term average for the region, but there were a few high water temperature days.

Inshore seagrass meadows across the Burnett–Mary NRM region declined slightly in overall condition in 2020–21, with the index score declining to a **poor** grade (Figure 78). Contributing indicators to the overall score were:

- abundance score was poor
- resilience score was moderate.

Seagrass abundance continued to decline marginally overall for the second consecutive year in 2020–21, but there are location-specific variations in the trends in the region. While coastal meadow spatial extents remain unchanged, abundances were mixed, with losses at one of the meadows. Meadow extents in estuarine habitats continued to decline across the region, coupled with declining abundances at sites in the south (at Urangan), but little change in abundances at northern sites (at Rodds Bay).

Resilience was moderate overall in the Burnett–Mary NRM region, but resilience varied among locations and sites within locations. Resilience was the lowest at estuarine sites at Urangan due to low overall abundance, and species composition dominated by colonisers at one site. The persistent seed banks coupled with stable abundances in meadows in the estuarine habitats may indicate some level of resilience. However, reproductive effort continues to remain very low, possibly limiting replenishment of the seed bank.

The decrease in the Burnett–Mary region seagrass condition index in the 2020–21 continues from losses in 2019–20. Both the seagrass abundance and resilience indicators have declined over the last two years, driven primarily by losses in the estuarine meadows.

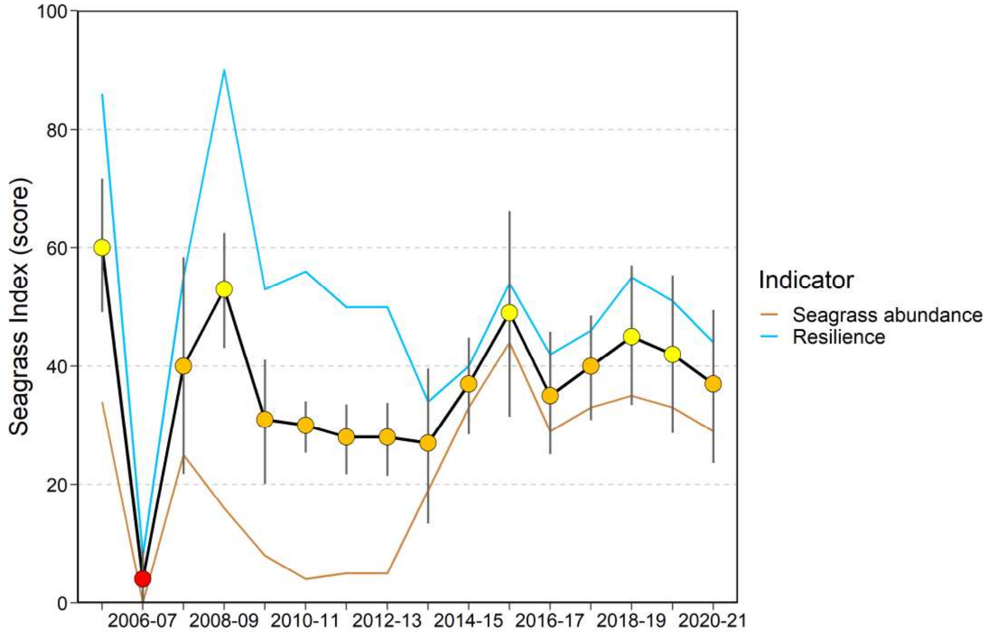


Figure 78. Report card of seagrass index and indicators for the Burnett–Mary region (averages across habitats and sites). Values are indexed scores scaled from 0–100 (± SE) and graded: ● = very good (81–100), ● = good (61–80), ● = moderate (41–60), ● = poor (21–40), ● = very poor (0–20). NB: Scores are unitless.

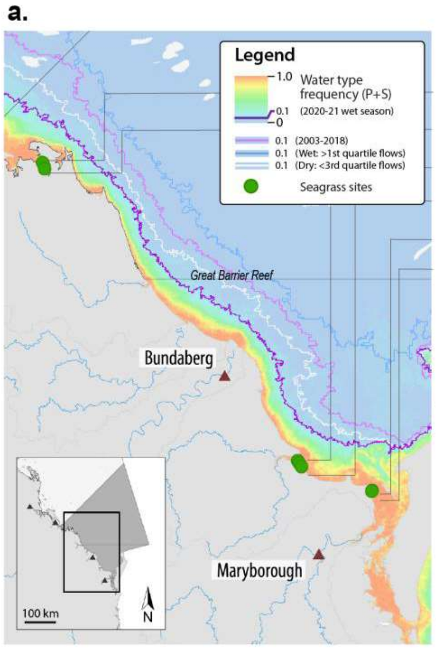
5.6.2 Climate and environmental pressures

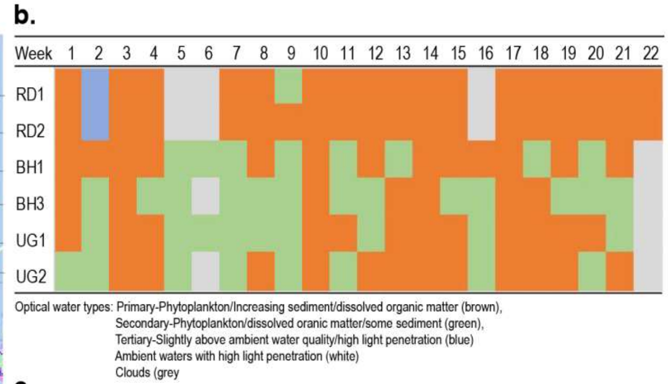
During 2020–21, rainfall and river discharge in the Burnett–Mary region were below average (Figure 79c, Table 9). In the Burnett-Mary region there are only estuarine and coastal monitoring locations, and these are generally exposed to high frequencies of primary water, but in 2020–21 there were periods of exposure to secondary water, which is atypical for these monitoring locations. Optically 'green', secondary waters have higher light penetration than primary waters (Figure 79a, b). But despite this, daily light levels were around the long-term average for the region, but the trends varied among locations (Figure 79c, d). At Rodds Bay, wet season light levels were well below the wet season average for the site with the 28-day average reaching as low as 1 mol m^{-2} d^{-1} (Figure 106); a level and exposure time that drive declines in the abundance of the species at Rodds Bay (Collier *et al.* 2016a). Daily light levels at the other sites were around or higher than the long-term average (Figure 106).

Within-canopy temperatures in 2020–21 were slightly cooler than the previous year and similar to the long-term average (Figure 79c). Maximum intertidal within-canopy temperatures exceeded 35°C for a total of 3 days during 2020–21, with the highest temperature recorded at 39°C (BH3, 3pm 11Apr21) (Figure 79e).

Although daytime tidal exposure was below or at the long-term average for the region (Figure 79c), levels of exposure differed with meadows in the north exposed for longer than those in the south (Figure 99). The less than long-term average exposure may have reduced the risk of temperature and desiccation stress in the south, but may also have increased the risk of light limitation in the turbid water areas.

Sediments in the estuarine seagrass habitats of the Burnett–Mary region are generally dominated by mud. In 2020–21, the proportion of mud continued to increase in the meadows in the south of the region, after experiencing a period of increased sands in 2018–19. Meadows in the north varied, with a noticeable increase in mud content at one site (RD1) (Figure 122). Coastal meadows in 2020–21 continued to be dominated by fine sand with little change from the previous year (Figure 123).





c.

	Long-term average	2020-21
Rainfall (1961-1990)	3.3 mm d^{-1}	2.4 mm d^{-1}
River discharge (1970-2016)	1,596,593 L yr^{-1}	506,148 L yr^{-1}
Turbid water exposure (2006-2018)	100%	97%
Daytime tidal exposure (1999-2020)	111 hr yr^{-1}	95 hr yr^{-1}
Within-canopy temperature - intertidal (2003-20)	23.5°C (40.9°C)	23.7°C (39.0°C)
Daily light (2009-2020)	12.2 mol m^{-2} d^{1}	12.0 mol m^{-2} d^{1}

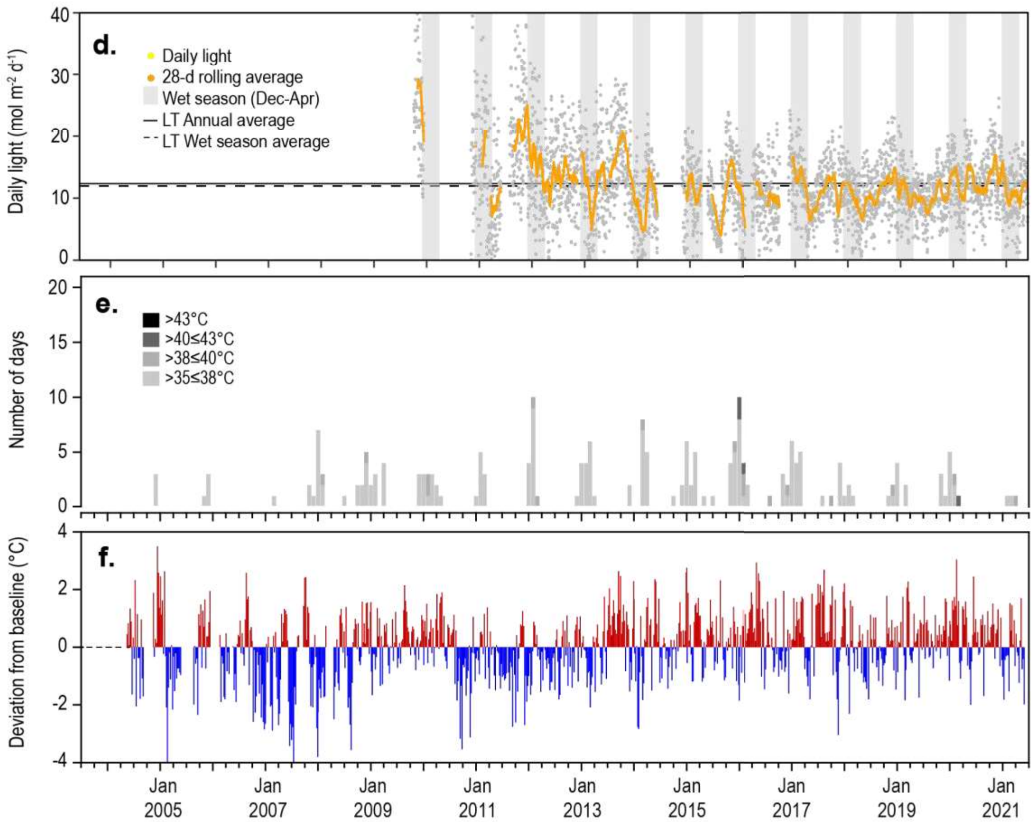


Figure 79. Environmental pressures in the Burnett–Mary region including: a. frequency of exposure to primary and seconday water from December 2020 to April 2021 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed) (white = no data), also showing the long-term average (2003–2018) exposure boundary (purple line), and the first (blue line) and third quartile (white line) of the long-term average (from Moran *et al.* 2022); b. wet season water type at each site; c. average conditions and max temperature over the long-term and in 2020–21; 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 at intertidal sites.

5.6.3 Inshore seagrass and habitat condition

Only estuarine and coastal habitats were assessed across the Burnett–Mary region in 2020–21, with data from 6 sites (Table 17).

Table 17. 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 4 and Table 5.

Habitat	Site		abundance	composition	distribution	reproductive effort	seed banks	meadow sediments	epiphytes & macroalgae
estuarine intertidal	RD1	Rodds Bay	■	■	■	■	■	■	■
	RD3	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 2020–21 monitoring period, the Burnett–Mary region seagrass condition index declined slightly overall and rated as a poor grade (Figure 78). The index remained below the 2015–2016 level (which was the third highest on record) due to declines in both of the indicators (Figure 80).

Over the long-term the regional average of seagrass abundance has fluctuated greatly (e.g. periods of loss and subsequent recovery). Increases between 2012 and 2016 were largely due to large increases at Urangan, which then declined, while recent trends in abundances at other locations followed different patterns. The long-term trend suggests that where losses have been observed, they are not part of a declining trend (Table 21). Seagrass abundance continued to decline marginally overall for the second consecutive year in 2020–21. While average coastal meadow abundance remained relatively unchanged, average abundance at estuarine meadows in Rodds Bay and Urangan, either declined or remained very low and unchanged, respectively.

Seagrass resilience declined in 2020–21, but remained moderate for the sixth consecutive year. This was primarily driven by the higher proportion of colonising species coupled with low abundances in the estuarine meadows at Urangan (Figure 80).

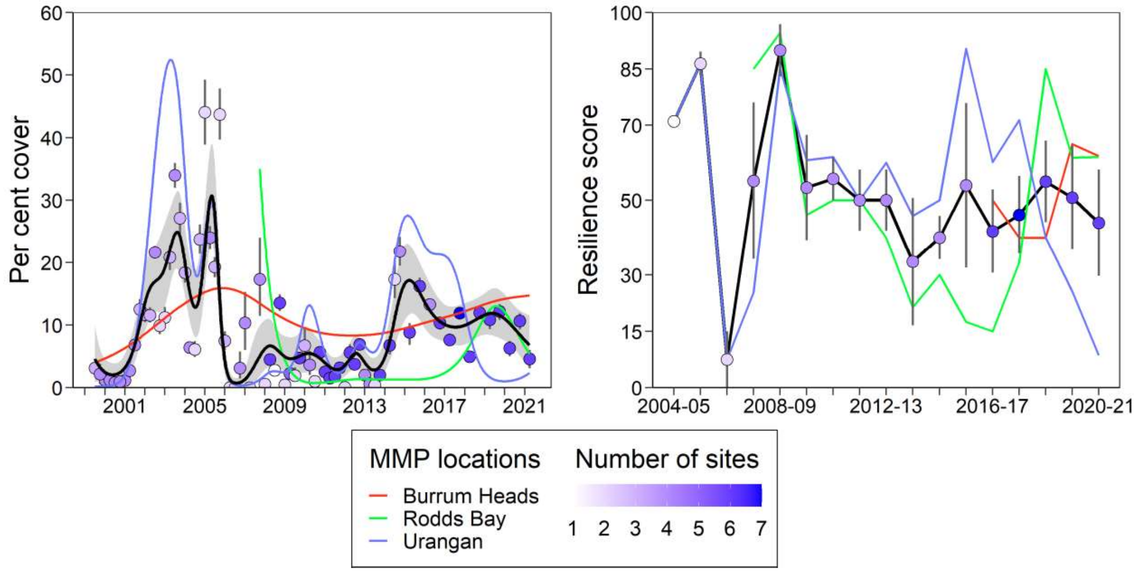


Figure 80. Temporal trends in the Burnett–Mary seagrass indicators used to calculate the seagrass condition index: a. average (circles, ±SE) seasonal abundance (per cent cover) and GAM plots of seagrass abundance trends for each location (coloured lines) and the region (black line with grey shaded area defining 95 per cent confidence intervals); b. average annual resilience score (±SE) and trends for each location (coloured lines). Colour of circles represents the number of sites assessed to calculate the average.

5.6.3.2 Seagrass abundance, composition and extent

Seagrass abundances (per cent cover) across the Burnett–Mary region in 2020–21 were greater in coastal than estuarine habitats (11.73 ±0.6 per cent and 5.8 ±1.6 per cent, respectively), however average estuarine abundance was higher in the late dry than the late wet season (9.2 ±2.1 per cent and 2.4 ±0.8 per cent, respectively). Although abundances remained very low across the region, abundance at a third of the monitoring sites continued to decrease marginally in 2020–21 relative to the previous period, while it improved at only 17 per cent (Figure 81). Half of all sites from each habitat type were unchanged in abundance in 2020–21 relative to the previous period. Overall, seagrass abundance declined in 2020–21 for the second consecutive year.

Since monitoring was established, the estuarine meadows have come and gone on an irregular basis, with no apparent long-term trend (Table 21).

The estuarine and coastal seagrass habitats have remained dominated by *Z. muelleri* with varying components of *H. ovalis*. In 2020–21, the proportion of colonising species increased at coastal meadows compared to the previous monitoring year, but conversely continued to decline well below the Reef long-term average in estuarine meadows (Figure 82). An increase in the proportion of colonising species in the meadows suggests some level of physical disturbance which may reduce ability to resist major disturbances in future.

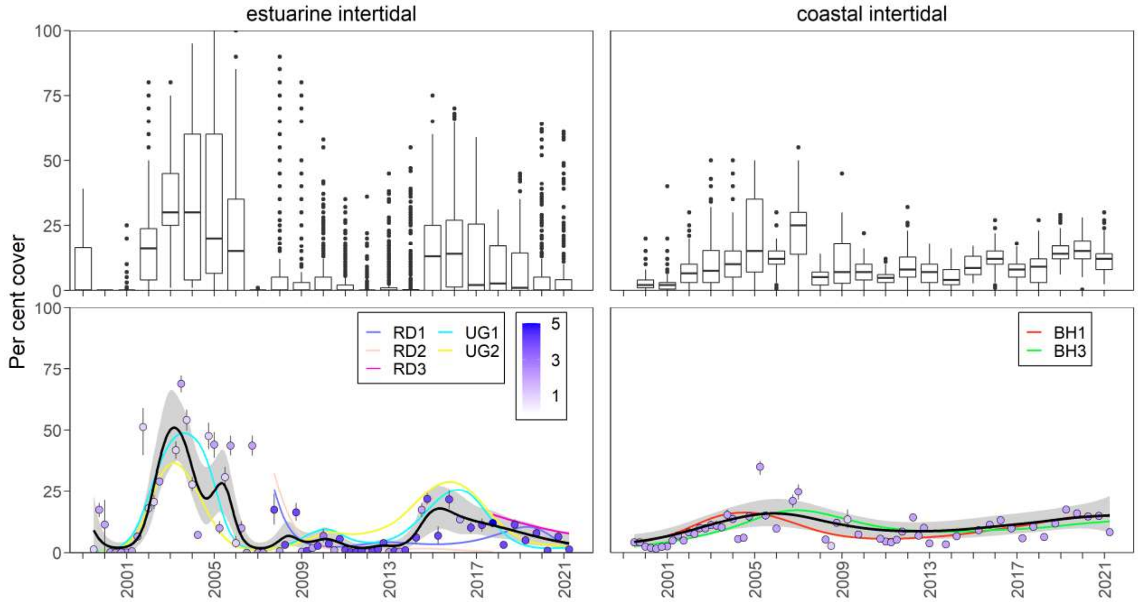


Figure 81. Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the Burnett–Mary NRM region from 1999 to 2021. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates 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. GAM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.

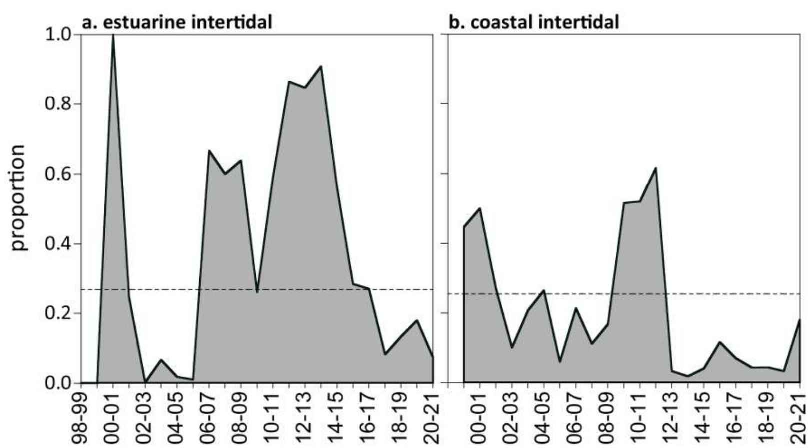


Figure 82. Proportion of seagrass abundance composed of colonising species at: a. estuarine and b. coastal habitats in the Burnett–Mary region, 1998–2021. Dashed line represents Reef long-term average proportion of colonising species for each habitat type.

Meadow spatial extent has remained stable at coastal meadows relative to the previous year (Figure 83). Estuarine meadowscontinued to decline slightly in extent in the late wet season. 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.

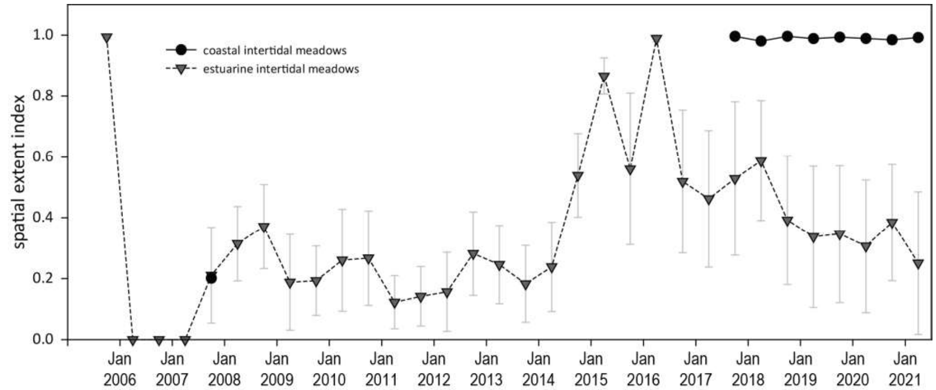


Figure 83. Change in spatial extent (± SE) of estuarine seagrass meadows within monitoring sites for each habitat and monitoring period across the Burnett–Mary NRM region.

5.6.3.3 Seagrass reproductive status

Seagrass reproductive effort in the dry season was similar to the previous period at coastal habitats, and higher than at estuarine habitats which were lower than the previous monitoring period (Figure 84). Seed banks persist at all but one of the meadows monitored even though seed banks declined at nearly all sites across the region in 2020–21 compared to the previous period.,. The biggest declines in seed banks occurred at estuarine sites (Figure 84). The smaller seed banks may be a consequence of increased germination in mid–2020, resulting in the seasonal (late dry) increase in abundance after the late wet decline 6 months earlier. However, the lower reproductive effort in the estuarine meadows may 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).

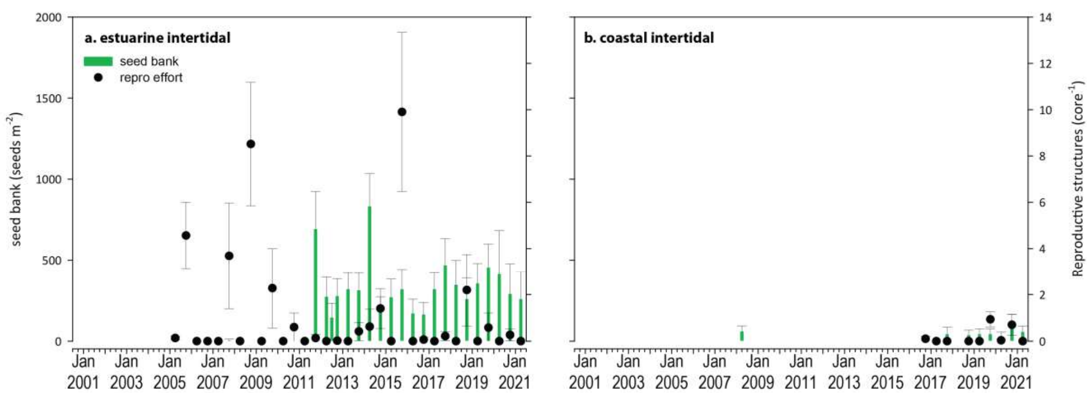


Figure 84. Seedbank and reproductive effort at inshore coastal and estuarine intertidal habitats in the Burnett–Mary region. Seed bank presented as the total number of seeds per m^2 sediment surface (green bars ±SE). Reproductive effort for late dry season presented as the average number of reproductive structures per core (species and sites pooled) (dots ±SE).

5.6.3.4 Resilience

Resilience was moderate overall in the Burnett–Mary NRM region.

At estuarine intertidal sites, resilience varied among locations (Figure 85). Per cent cover was above critical thresholds at Rodds Bay for meadow condition (per cent cover and composition), albeit only just at RD3. There were reproductive structures at RD1 in 2020–21, but none at RD3 although there had been in recent years. At Urangan, both sites were below condition thresholds due to low per cent cover or high composition of colonising species indicating vulnerability to disturbances. There was a higher proportion of colonising species at UG1 resulting in a lower score. No reproductive structures were present.

At coastal intertidal sites at Burrum Heads, both sites were in a good condition indicative of high resistance capacity to disturbances. There were reproductive structures at BH1 but there have been no reproductive structures observed at BH3 in 2020–21 or in the previous three years.

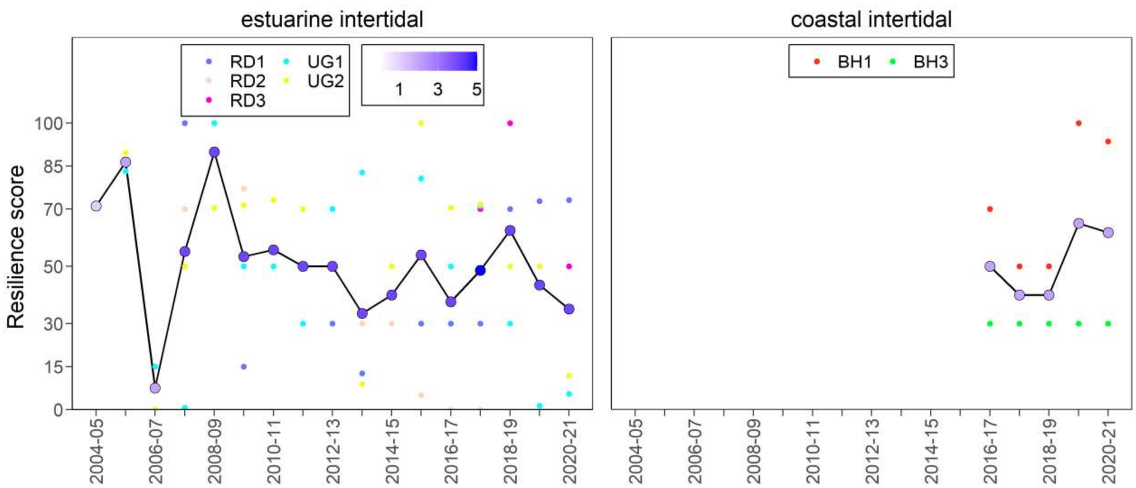


Figure 85. Resilience score in each habitat in the Burnett–Mary region from 2006–2021. Coloured small points represent different sites. Shades of blue for the larger points indicate the number of sites that contribute to the score.

5.6.3.5 Epiphytes and macroalgae

Epiphyte cover on seagrass leaf blades in 2020–21 generally remained higher than the long-term average for the seventh consecutive year at estuarine habitats (Figure 86). However, at coastal habitats, epiphyte abundance remained below the long-term average for the fifth consecutive year (Figure 86).

Per cent cover of macroalgae remained low and below the long-term average at across the habitats monitored (Figure 86).

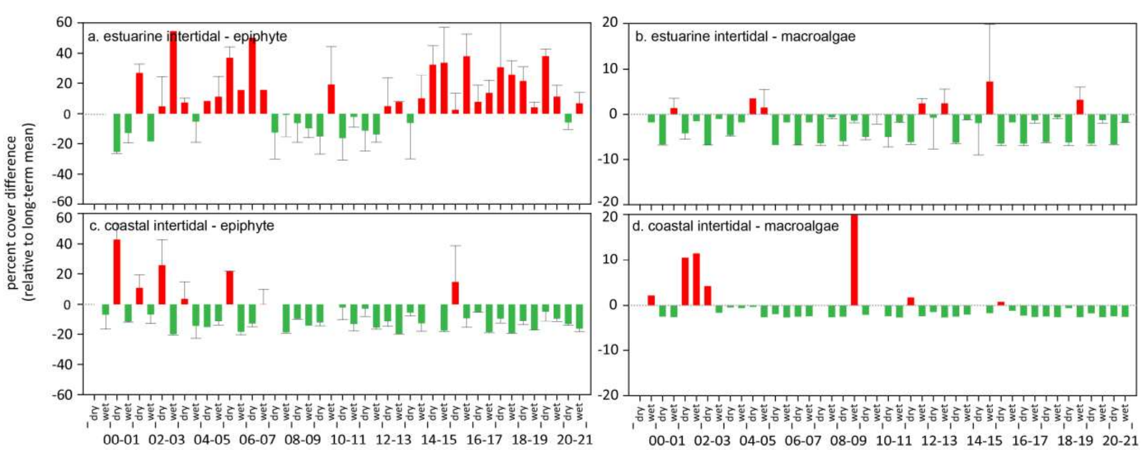


Figure 86. Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average for each seagrass habitat in the Burnett–Mary NRM region (sites pooled, ±SE).

6 Discussion

Inshore seagrass condition improved overall in 2020–21, however, this was driven by improved conditions in the northern regions, as southern regions declined.

Despite 2020–21 being the second consecutive year where environmental pressures were relatively benign, some seagrass habitats of the Reef are failing to recover to abundance levels observed during the first few years of the MMP (2005–2008).

Natural recovery requires environmental conditions that enable expansion following loss, and subsequent sexual reproduction and seed bank formation. Our monitoring reveals that it can take more than five years for foundational seagrass species of the Reef to recover following loss. However, multiple, cumulative and consecutive pressures over the past 15 years have likely hampered recovery.

Chronic declines in inshore water quality of 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). This has been caused in part by intensive use of the catchments for agriculture and grazing, which have led to an increase in the anthropogenic sediment, organic matter and nutrient load to the Reef (Lewis *et al.* 2021). Flood waters deliver these terrestrially sourced pollutants dispersing them over the sensitive inshore ecosystems, including seagrass meadows (summarised in Schaffelke *et al.* 2013). These in turn reduce water clarity and the amount of light able to penetrate to benthic habitats (Bainbridge *et al.* 2018).

Concerns over the health of inshore water quality underpin the Reef 2050 Water Quality Improvement Plan, and the Paddock to Reef Integrated Monitoring, Modelling and Reporting Program, of which the MMP and inshore seagrass monitoring is a component. But multiple pressures are the cause of ecological decline, including cyclone damage and coastal development for urban centres and commercial ports (Schaffelke *et al.* 2017; De'ath *et al.* 2012), while climate change and rising temperature has left the Reef less resilient, and more challenging to manage (GBRMPA, 2019).

Cumulative pressures appear to have slowed and abated inshore seagrass recovery across the Reef, which in turn may reduce capacity of the seagrass to produce viable seed banks in some locations (van Katwijk *et al.* 2010). There were frequent and repeated disturbances over the past decade and a half, and some of these pressures are summarised in Figure 87.

Cyclones de-stabilise sediments and physically remove seagrass plants and seed banks. Though these impacts tend to be localised, they can be very severe and recovery can be difficult if the substrate is altered and propagules (including plants and seeds) are lost.

Cyclones are more common in the northern region of the Reef (Figure 87). While Cape York is generally less affected by anthropogenic activities than the southern regions, frequent cyclone disturbances occur. Both Cape York and the Wet Tropics have been affected by cyclones in 5 of the past 15 years. Cyclones are one of the principal causes of loss and low recovery in the southern Wet Tropics which was affected by severe cyclones Larry in 2006 and Yasi in 2011. The Mackay–Whitsunday region has also been affected by cyclones in five of the previous 15 years with lasting impacts in some locations, e.g. Whitsunday Islands.

The more widespread impacts of cyclones arise from heavy rainfall and elevated river discharge. Large discharges can be caused by rainfall associated with the cyclone itself, or by generally unstable wet season conditions and rainfall associated with the monsoon trough, when cyclones are also more likely to occur. There were consecutive years of above average discharge before and after 2011, particularly in the central and southern regions.

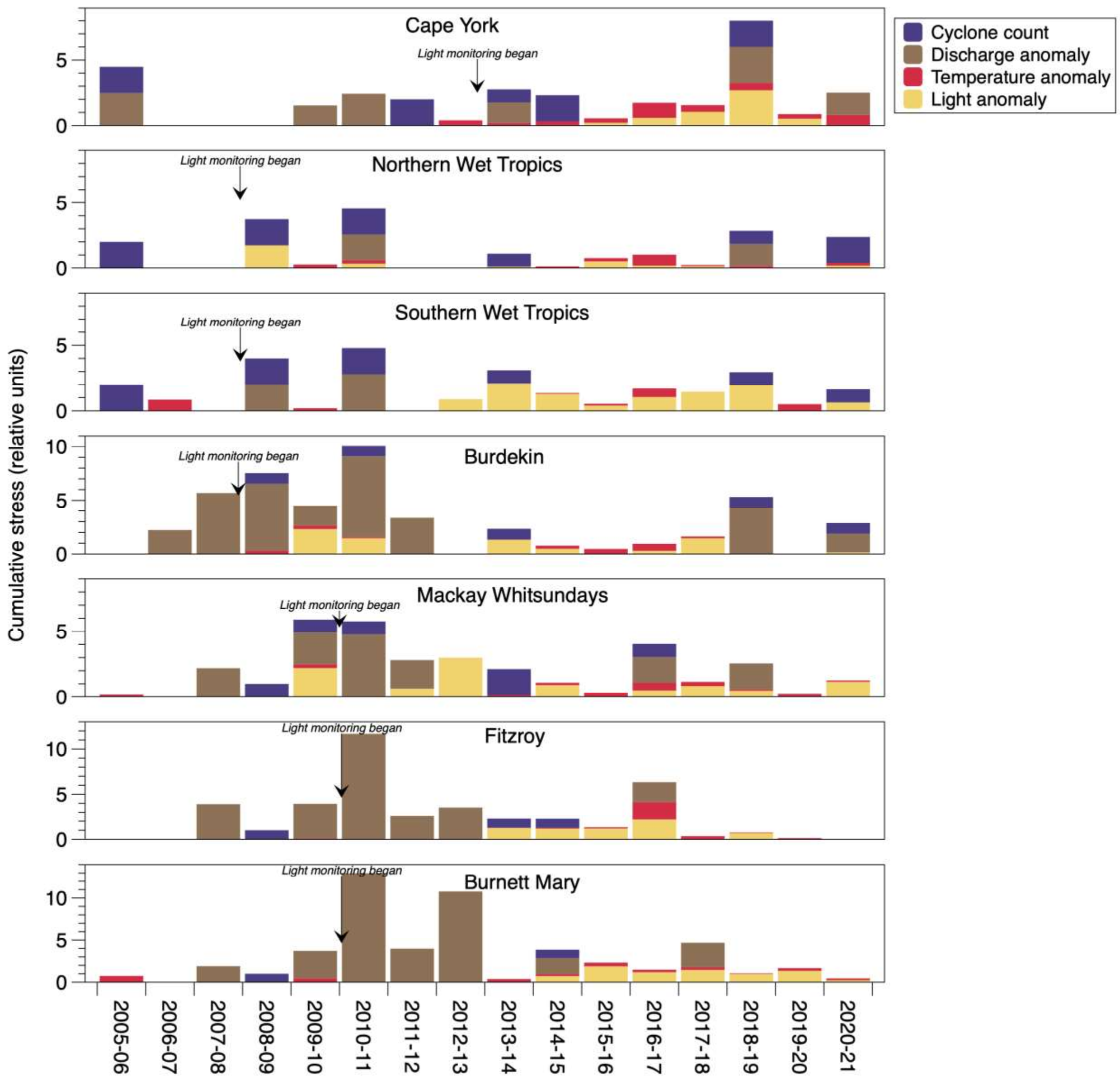


Figure 87. Cumulative pressures on seagrass habitats of the inshore regions of Reef, by NRM region from 2005 to 2021. This includes count of cyclones to affect each region, discharge anomaly as the magnitude of discharge volume greater than 1.5 times the median value, annual average within-canopy temperature above the long-term annual average, and the annual average above-canopy daily light less than the long-term average. Initiation of light monitoring is also indicated.

One of the principal pathways through which discharge affects seagrass ecosystems is the reduction in daily light associated with high concentrations of suspended sediments, nutrients and organic matter of discharges (Bainbridge *et al.* 2018; Lewis *et al.* 2021). Resuspension of this material prolongs the impact of discharge for months or even longer in inshore regions (Fabricius *et al.* 2016). Indeed seagrass monitoring sites are exposed to a very high frequency of coloured or turbid water even in low discharge years (Figure 26, Figure 35, Figure 36, Figure 52, Figure 61, Figure 70, Figure 79).

Daily light levels were also below average for a number of years in all regions since light monitoring began, even when discharge levels were lower than average (Figure 10). There were low and variable light levels across the Reef habitats from 2014–15 to 2018–19 in most regions, but this trend appears to have reversed in 2019–20 and 2020–21 (Figure 8, Figure 87). Additionally, the effects of low light can take some time to manifest, as seagrasses are able to tolerate low light by drawing on carbohydrate reserves. As these deplete, morphological change and shoot loss occurs (Collier *et al.* 2012b; Collier *et al.* 2016a; O'Brien *et al.* 2018). As an example, declines in abundance in the Burdekin region, which are a legacy of floods and low light conditions in 2019, are the main contributor to low overall abundance in 2019–20. This is of high significance in a region which contains the second highest area of inshore seagrass in the Reef and where declining seagrass condition can severely impact “downstream” species of conservation concern which are dependent on seagrass e.g. dugongs and turtles (Wooldridge 2017).

These periods of low light have generally coincided with years of elevated water temperature. Climate change is the most significant threat to the Reef’s long-term outlook (GBRMPA, 2019), and thermal anomalies are emerging in seagrass habitats as well. It has become more common for within-canopy water temperature in any week to be above average than below average since 2013 (Figure 11).

Annual temperature was above average in most years in most regions since 2013 (Figure 87). Extreme temperatures that cause photoinhibition and ‘burning’ (>40ºC) occur when heatwaves coincide with low tides are still relatively rare, but increasing in some regions such as the Fitzroy (Figure 70). The chronic effect of rising water temperature may be taking a physiological toll by increasing respiration rates and seagrass light requirements (Collier *et al.* 2012a; Collier *et al.* 2016a). These high temperatures have been occurring in years when light levels were also low, and have likely been acting in concert to hamper recovery rates.

There are numerous other potential stressors including changes to herbivory, habitat fragmentation, acidification, competition with macroalgae, disease and increased desiccation.

Except for extreme events (very large discharge and cyclones), it is difficult to ascribe cause to any one pressure when there are many occurring successively or concurrently. However, through targeted research, cumulative pressures can be quantified and cumulative indices of pressure developed (Uthicke *et al.* 2016; Lawrence 2019; Uthicke *et al.* 2020).

Securing a future for seagrasses on the Great Barrier Reef

This year (2020–21) was the first year the new seagrass condition index was reported (including back-dating to the start of the program, see Appendix 1). The abundance indicator has been retained without any changes, however the reproductive effort and tissue nutrient indicators have been removed and after extensive review, have been replaced with a single resilience indicator. The resilience score is calculated using a decision tree. It includes resistance potential and likelihood of recovery based on reproductive effort (as a proxy for seed/propagules) graded according to the species in the habitat.

We believe the new seagrass condition index better represents the state of the inshore seagrass meadows of the Reef, and provides management with an enhanced evidence base to help focus management efforts and build ecosystem resilience to future disturbances to secure the future for seagrasses on the Great Barrier Reef.

Resilience-based management responses place a strong emphasis on the use of forecasting tools to inform planning and actions, together with monitoring and diagnostic tools to adjust actions. These actions need to be designed to maximise recovery and limit disturbances or impacts.

Practicable conservation opportunities exist, 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 to Reef Program), are expected to benefit inshore seagrass by improving resilience to other stressors. Minimising localised pressures from coastal and urban runoff, and the direct effects of coastal development (e.g. dredging) will also reduce cumulative stress.

In addition to direct action, improving the accuracy of indicators, and refining thresholds and indices of pressures, including cumulative stress, will improve our understanding of the processes of resilience to guide management actions and adaptation responses.

Some of these management options were outlined in previous reports (McKenzie *et al.* 2021a, McKenzie *et al.* 2021b), and are summarized and updated here:

1. Accurate models of seagrass recovery to identify when recovery is on track or when intervention actions may be required.
2. Risk assessments updated to ensure that the most relevant pressures are being measured (in the most relevant manner), and methods for assessing cumulative impacts developed.
3. Localised (site-level) monitoring undertaken in this program scaled to broader-levels (e.g. RIMReP) to fully capture the extent of habitat decline and recovery so that the potential ecological consequences can be more accurately inferred. For example, continuous improvements in earth observing (airborne and spaceborne) image capture of the Reef using Unoccupied Aerial Vehicles (UAV) and Autonomous Underwater Vehicles (AUV), along with advances in machine- and deep-learning to process images, offer opportunities for broad-scale assessment of seagrass condition and health in some habitat types that were not available in the past.
4. Continuous review and revision of indicators. Although we have now included a resilience indicator to replace the previous reproductive metric, resilience is complex and the new indicator includes quantitative measures of only a few elements of resilience (Udy *et al.* 2018). Further exploration of practicable ways to assess resilience that inform current status and future risk would be informative.
5. Improving our understanding of poor and variable reproductive effort through focussed research, as reproduction underpins the capacity for meadows to recover naturally, and seeding offers a potential restoration strategy.
6. Active seagrass restoration or enhancement of resilience may be of benefit, but significant research is required before techniques can be operationalised (see also Tan *et al.* 2020). This may include active environmental engineering in localised areas to improve habitat suitability, by mitigating limiting factors (e.g. wave energy, erosion) or creating new habitat.
7. Enhancing the use of existing tools and new approaches and technologies to build resilience.

7 Conclusion

In 2020–21 inshore seagrass meadows across the Reef improved in overall condition, with the seagrass Index increasing to **moderate**. Both seagrass condition indicators improved in 2020–21 after reaching the lowest score in 2019–20 that had been observed in seven years. The abundance score improved from poor to moderate, and the resilience score similarly increased, but remained moderate.

Environmental conditions were relatively benign across the Reef for the second consecutive year, but there were legacy effects of pressures from previous years.

In 2020–21, the inshore seagrass of the Reef was in a moderate condition in all northern NRM regions, but poor in all southern regions. The score increased in the northern regions compared to the previous monitoring period, but declined in southern regions. Improvements overall were driven mostly by increases in the abundance indicator.

Seagrass meadows of the Reef are dynamic, with large changes in abundance being seemingly typical in some regions (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 these changes (i.e. declines and subsequent recovery) are complex.

In late 2008, locations in the northern Wet Tropics and Burdekin regions were in a good 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 and moderate state, respectively, with low abundance, reduced reproductive effort and small or absent seed banks (Figure 88).

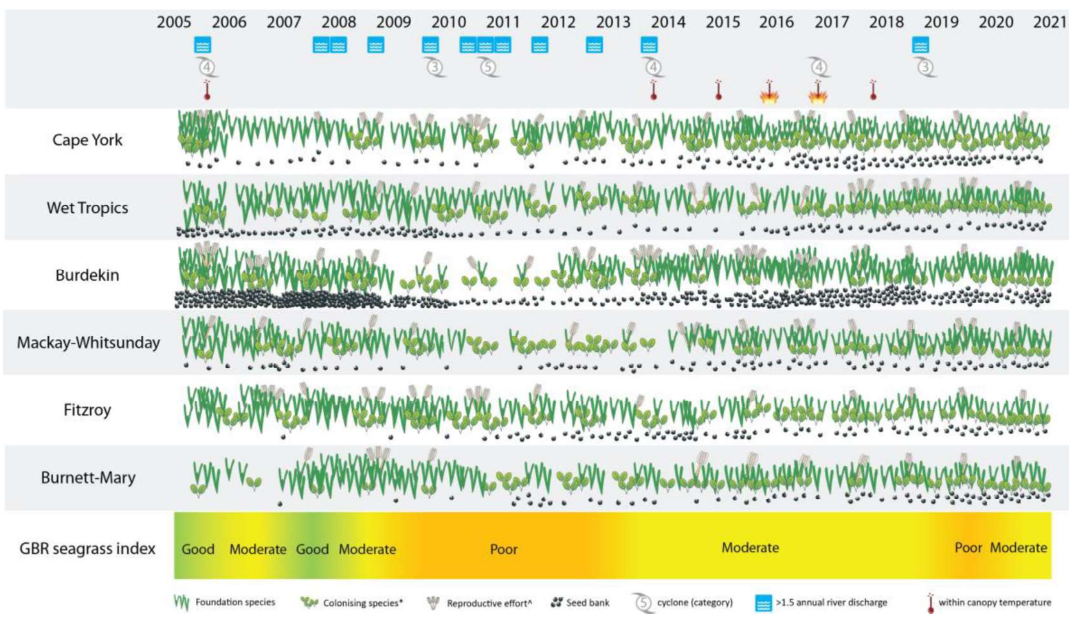


Figure 88. Summary of inshore seagrass state illustrating pressures, abundance of foundation / colonising species, seed bank and reproductive effort in each NRM from 2005 to 2021. * colonising species are represented by the genus *Halophila*, however, *Zostera* and *Halodule* can be both colonising and 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 where 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 2016–17, recovery had slowed or stalled across most of the regions, and seagrass condition had been gradually declining. It appears cumulative pressures continue to undermine the resilience of inshore seagrass meadows of the Reef. Frequent and repeated disturbances seem to be maintaining lower seagrass abundance at some locations, perpetuated by feedbacks, which in turn may be reducing capacity of the plants to expand and produce viable seed banks.

The Wet Tropics and Fitzroy regions have shown the slowest recovery rates since 2012, although there have been recent declines in all regions except the Mackay–Whitsundays as well. The causes 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.

In the southern Wet Tropics, severe impacts to the substrate from scouring and subsequent deposition of fine sediments in 2011, significantly delayed the onset of recovery. From 2018, the substrate appeared to be stabilising and was more conducive for seagrass growth (increasing and less mobile fine sands). However, expansion of the meadows has not occurred as fast as previously experienced (e.g. following cyclone Larry in 2006). It is likely the low seagrass cover is continuing sediment resuspension, i.e. feedbacks are maintaining a disturbed state under average conditions. In such a state, seagrass may require lower environmental thresholds, such as below average temperatures and higher light availability, before recovery rates improve.

For the Reef's inshore seagrass meadows to continue improving will require extended periods of conducive conditions for seagrass growth and reduced environmental pressures. While climatic conditions cannot be controlled, the scale of effect they have on seagrasses can be lessened through initiatives such as the Paddock to Reef Program. It is imperative that resilience, including ability to recover following loss, remains at the forefront of research and management priorities.

To secure the future of the Reef's seagrass ecosystems, improved ecosystem science on resilience and recovery would be valuable. In conjunction with over-arching research, it is critical to maintain adaptive resilience-based management by placing a strong emphasis on the use of forecasting tools to inform planning and actions, together with monitoring and diagnostic tools to adjust and implement actions to enhance resilience, maximise recovery and limit disturbances or impacts.

8 References

Angelini, C., Altieri, A.H., Silliman, B.R., Bertness, M.D. 2011, Interactions among Foundation Species and their Consequences for Community Organization, Biodiversity, and Conservation. *BioScience,* 61(10): 782-789.

Anthony, K.R.N., Ridd, P.V., Orpin, A.R., Larcombe, P., Lough, J. 2004, Temporal variation in light availability in coastal benthic habitats: Effects of clouds, turbidity, and tides. *Limnology and Oceanography,* 49(6): 2201-2211.

ANZECC. 2000, *Australian and New Zealand guidelines for fresh and marine water quality*. Canberra: Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand.

Australian Government and Queensland Government. 2018a, *Reef 2050 Long-Term Sustainability Plan—July 2018*. Canberra : Brisbane: Australian Government ; Queensland Government. 124.

Australian Government and Queensland Government. 2018b, *Reef 2050 Water Quality Improvement Plan: 2017 - 2022*. Brisbane: State of Queensland. 55.

Bainbridge, Z., Lewis, S., Bartley, R., Fabricius, K., Collier, C., Waterhouse, J., Garzon-Garcia, A., Robson, B., Burton, J., Wenger, A., Brodie, J. 2018, Fine sediment and particulate organic matter: A review and case study on ridge-to-reef transport, transformations, fates, and impacts on marine ecosystems. *Marine Pollution Bulletin,* 135: 1205-1220. doi: https://doi.org/10.1016/j.marpolbul.2018.08.002

Bainbridge, Z., Wolanski, E., Lewis, S., Brodie, J. 2012, Fine sediment and nutrient dynamics related to particle size and floc formation in a Burdekin River flood plume, Australia. *Marine Pollution Bulletin,* 65: 236-248.

Baird, M.E., Adams, M.P., Babcock, R.C., Oubelkheir, K., Mongin, M., Wild-Allen, K.A., Skerratt, J., Robson, B.J., Petrou, K., Ralph, P.J., O'Brien, K.R., Carter, A.B., Jarvis, J.C., Rasheed, M.A. 2016, A biophysical representation of seagrass growth for application in a complex shallow-water biogeochemical model. *Ecological Modelling,* 325: 13-27. doi: http://dx.doi.org/10.1016/j.ecolmodel.2015.12.011

Baird, M.E., Wild-Allen, K.A., Parslow, J., Mongin, M., Robson, B., Skerratt, J., Rizwi, F., Soja-Woznaik, M., Jones, E., Herzfeld, M., Margvelashvili, N., Andrewartha, J., Langlais, C., Adams, M.P., Cherukuru, N., Gustafsson, M., Hadley, S., Ralph, P.J., Rosebrock, U., Schroeder, T., Laiolo, L., Harrison, D., Steven, A.D.L. 2019, CSIRO Environmental Modelling Suite (EMS): Scientific description of the optical and biogeochemical models (vB3p0). *Geosci. Model Dev. Discuss.,* 2019: 1-107. doi: 10.5194/gmd-2019-115

Birch, W. and Birch, M. 1984, Succession and pattern of tropical intertidal seagrasses in Cockle Bay, Queensland, Australia: a decade of observations. *Aquatic Botany,* 19: 343-367.

BOM. 2021, Australian Federal Bureau of Meteorology. *http://www.bom.gov.au*.

Brodie, J.E., Kroon, F.J., Schaffelke, B., Wolanski, E.C., Lewis, S.E., Devlin, M.J., Bohnet, I.C., Bainbridge, Z.T., Waterhouse, J., Davis, A.M. 2012, Terrestrial pollutant runoff to the Great Barrier Reef: An update of issues, priorities and management responses. *Marine Pollution Bulletin,* 65(4–9): 81-100.

Bros, W.E. and Cowell, B.C. 1987, A technique for optimising sample size (replication). *Journal of Experimental Marine Biology and Ecology,* 114: 63-71.

Campbell, S.J. and McKenzie, L.J. 2001, Seagrass and algal abundance in the Whitsundays region. Status Report.

Campbell, S.J. and McKenzie, L.J. 2004, Flood related loss and recovery of intertidal seagrass meadows in southern Queensland, Australia. *Estuarine, Coastal and Shelf Science,* 60(3): 477-490.

Campbell, S.J., McKenzie, L.J., Kerville, S.P. 2006, Photosynthetic responses of seven tropical seagrasses to elevated seawater temperature. *Journal of Experimental Marine Biology and Ecology,* 330: 455-468.

Campbell, S.J., Roder, C.A., McKenzie, L.J., Lee Long, W.J. 2002, Seagrass resources in the Whitsunday region 1999 and 2000. *DPI Information Series QI02043*, 50.

Carruthers, T., Dennison, W., Longstaff, B., Waycott, M., Abal, E.G., McKenzie, L.J., Lee Long, W. 2002, Seagrass habitats of north east Australia: models of key processes and controls. *Bulletin of Marine Science,* 71(3): 1153-1169.

Carter, A.B., Collier, C., Lawrence, E., Rasheed, M.A., Robson, B.J., Coles, R. 2021a, A spatial analysis of seagrass habitat and community diversity in the Great Barrier Reef World Heritage Area. *Scientific Reports,* 11(1): 22344. doi: 10.1038/s41598-021-01471-4

Carter, A.B., McKenna, S.A., Rasheed, M.A., Collier, C., McKenzie, L., Pitcher, R., Coles, R. 2021b, Synthesizing 35 years of seagrass spatial data from the Great Barrier Reef World Heritage Area, Queensland, Australia. *Limnology and Oceanography Letters,* 6(4): 216-226. doi: https://doi.org/10.1002/lol2.10193

Carter, A.B., McKenna, S.A., Rasheed, M.A., McKenzie, L.J., Coles, R.G. 2016, Seagrass mapping synthesis: A resource for coastal management in the Great Barrier Reef World Heritage Area. Report to the National Environmental Science Programme., 22.

Chartrand, K.M., Szabó, M., Sinutok, S., Rasheed, M.A., Ralph, P.J. 2018, Living at the margins – The response of deep-water seagrasses to light and temperature renders them susceptible to acute impacts. *Marine Environmental Research,* 136: 126-138. doi: https://doi.org/10.1016/j.marenvres.2018.02.006

Coles, R., McKenzie, L.J., De'ath, G., Roelofs, A., Lee Long, W.J. 2009, Spatial distribution of deepwater seagrass in the inter-reef lagoon of the Great Barrier Reef World Heritage Area. *Marine Ecology Progress Series,* 392: 57-68. doi: 10.3354/meps08197

Coles, R.G., McKenzie, L.J., Mellors, J.E., Yoshida, R.L. 2001a, Validation and GIS of seagrass surveys between Cairns and Bowen – October/November 1987. CD Rom.

Coles, R.G., McKenzie, L.J., Yoshida, R.L. 2001b, Validation and GIS of seagrass surveys between Bowen and Water Park Point– March/April 1987. CD Rom.

Coles, R.G., McKenzie, L.J., Yoshida, R.L. 2001c, Validation and GIS of seagrass surveys between Cape York and Cairns - November 1984. CD Rom.

Coles, R.G., McKenzie, L.J., Yoshida, R.L. 2001d, Validation and GIS of seagrass surveys between Water Park Point and Hervey Bay – October/November 1988. CD Rom.

Collier, C., Devlin, M., Langlois, L., Petus, C., McKenzie, L.J., Texeira da Silva, E., McMahon, K., Adams, M., O'Brien, K., Statton, J., Waycott, M. 2014, *Thresholds and indicators of declining water quality as tools for tropical seagrass management. Report to the National Environmental Research Program*. Cairns: Reef and Rainforest Research Centre Limited. 93.

Collier, C. and Waycott, M. 2009, Drivers of change to seagrass distributions and communities on the Great Barrier Reef: Literature review and gaps analysis. *Report to the Marine and Tropical Sciences Research Facility*.

Collier, C.J., Adams, M.P., Langlois, L., Waycott, M., O'Brien, K.R., Maxwell, P.S., McKenzie, L. 2016a, Thresholds for morphological response to light reduction for four tropical seagrass species. *Ecological Indicators,* 67: 358-366. doi: http://dx.doi.org/10.1016/j.ecolind.2016.02.050

Collier, C.J., Carter, A.B., Rasheed, M., McKenzie, L.J., Udy, J., Coles, R., Brodie, J., Waycott, M., O'Brien, K.R., Saunders, M., Adams, M., Martin, K., Honchin, C., Petus, C., Lawrence, E. 2020, An evidence-based approach for setting desired state in a complex Great Barrier Reef seagrass ecosystem: A case study from Cleveland Bay. *Environmental and Sustainability Indicators,* 7: 100042. doi: https://doi.org/10.1016/j.indic.2020.100042

Collier, C.J., Chartrand, K., Honchin, C., Fletcher, A., Rasheed, M. 2016b, *Light thresholds for seagrasses of the GBR: a synthesis and guiding document. Including knowledge gaps and future priorities. Report to the National Environmental Science Programme*. Cairns Reef and Rainforest Research Centre Limited. 41.

Collier, C.J., Langlois, L., Waycott, M., McKenzie, L.J. 2021a, *Resilience in practice: Development of a seagrass resilience metric for the Great Barrier Reef Marine Monitoring Program*. Townsville: Great Barrier Reef Marine Park Authority. 61.

Collier, C.J., Langlois, L.M., McMahon, K.M., Udy, J., Rasheed, M., Lawrence, E., Carter, A.B., Fraser, M.W., McKenzie, L.J. 2021b, What lies beneath: Predicting seagrass below-ground biomass from above-ground biomass, environmental conditions and seagrass community composition. *Ecological Indicators,* 121: 107156. doi: https://doi.org/10.1016/j.ecolind.2020.107156

Collier, C.J., Lawerence, E., Waycott, M., Langlois, L.A., McKenzie, L.J. 2019, Reproductive effort as a predictor of future seagrass cover: Model assessment and implications for report card metrics and the development of a seagrass resilience indicator. In LJ McKenzie, CJ Collier, LA Langlois, RL Yoshida, J Uusitalo, N Smith & M Waycott (Eds.), *Marine Monitoring Program: Annual Report for Inshore Seagrass Monitoring 2017–18. Report for the Great Barrier Reef Marine Park Authority* (pp. 135-142). Townsville: Great Barrier Reef Marine Park Authority.

Collier, C.J., Ow, Y.X., Langlois, L., Uthicke, S., Johansson, C.L., O'Brien, K.R., Hrebien, V., Adams, M.P. 2017, Temperatures for net primary productivity of three tropical seagrass species. *Frontiers in Plant Science,* 8. doi: 10.3389/fpls.2017.01446

Collier, C.J. and Waycott, M. 2014, Temperature extremes reduce seagrass growth and induce mortality. *Marine Pollution Bulletin,* 83(2): 483-490. doi: http://dx.doi.org/10.1016/j.marpolbul.2014.03.050

Collier, C.J., Waycott, M., McKenzie, L.J. 2012a, Light thresholds derived from seagrass loss in the coastal zone of the northern Great Barrier Reef, Australia. *Ecological Indicators,* 23(0): 211-219. doi: 10.1016/j.ecolind.2012.04.005

Collier, C.J., Waycott, M., Ospina, A.G. 2012b, Responses of four Indo-West Pacific seagrass species to shading. *Marine Pollution Bulletin,* 65(4-9): 342-354. doi: 10.1016/j.marpolbul.2011.06.017

Connolly, R.M., Jackson, E.L., Macreadie, P.I., Maxwell, P.S., O'Brien, K.R. 2018, Seagrass dynamics and resilience. Chapter 7. In AWD Larkum, GA Kendrick & PJ Ralph (Eds.), *Seagrasses of Australia.*: Springer.

De'ath, G. and Fabricius, K. 2010, Water quality as a regional driver of coral biodiversity and macroalgae on the Great Barrier Reef. *Ecological Applications,* 20(3): 840-850. doi: 10.1890/08-2023.1

De'ath, G., Fabricius, K.E., Sweatman, H., Puotinen, M. 2012, The 27–year decline of coral cover on the Great Barrier Reef and its causes. *Proceedings of the National Academy of Sciences,* 109(44): 17995-17999. doi: 10.1073/pnas.1208909109

Dennison, W.C., Longstaff, B.J., O'Donohue, M.J. 1997, Seagrasses as Bio-indicators. In S Hillman & S Raaymakers (Eds.), *Karumba dredging 1996 - Environmental Monitoring Report. EcoPorts Monograph Series No.6* (pp. 255). Brisbane: Ports Corporation of Queensland.

Dennison, W.C., O'Donnohue, M.K., Abal, E.G. 1995, *An assessment of nutrient bioindicators using marine plants in the region of Pioneer Bay, Airlie Beach, Queensland. Dry season sampling, September 1995*. Brisbane: Marine Botany Section, Department of Botany, University of Queensland.

Department of Environment and Resource Management. 2009, *Queensland Water Quality Guidelines, Version 3. ISBN 978-0-9806986-0-2*. Brisbane: State of Queensland (Department of Environment and Resource Management). 121.

Department of the Premier and Cabinet. 2014. Great Barrier Reef report card 2012 and 2013, Reef Water Quality Protection Plan, Scoring system. Retrieved 14/11/2014, from http://www.reefplan.qld.gov.au/measuring-success/methods/scoring-system.aspx

Derbyshire, K.J., Willoughby, S.R., McColl, A.L., Hocroft, D.M. 1995, *Small prawn habitat and recruitment study : final report to the Fisheries Research and Development Corporation and the Queensland Fisheries Management Authority*. Cairns: Department of Primary Industries. 43.

Ellison, A.M. 2019, Foundation Species, Non-trophic Interactions, and the Value of Being Common. *iScience,* 13: 254-268. doi: https://doi.org/10.1016/j.isci.2019.02.020

Fabricius, K., De'Ath, G., Humphrey, C., Zagorskis, I., Schaffelke, B. 2012, Intr-annual variation in turbidity in response to terrestrial runoff on near-shore coral reefs of the Great Barrier Reef. *Estuarine, Coastal and Shelf Science,* 3: 458-470.

Fabricius, K.E., Logan, M., Weeks, S.J., Lewis, S.E., Brodie, J. 2016, Changes in water clarity in response to river discharges on the Great Barrier Reef continental shelf: 2002–2013. *Estuarine, Coastal and Shelf Science,* 173: A1-A15. doi: http://dx.doi.org/10.1016/j.ecss.2016.03.001

GBRMPA. 2014, *Great Barrier Reef Region Strategic Assessment: Program report*. Townsville: GBRMPA. 100.

GBRMPA. 2019, *Great Barrier Reef Outlook Report 2019*. Townsville: Great Barrier Reef Marine Park Authority 354.

GBRMPA. 2021. Marine Monitoring Program Retrieved 09 December, 2021, from http://www.gbrmpa.gov.au/our-work/our-programs-and-projects/reef-2050-marine-monitoring-program

Goldsworthy, P.M. 1994, Seagrasses. In L Benson, P Goldsworthy, R Butler & J Oliver (Eds.), *Townsville Port Authority Capital Dredging Works 1993: Environmental Monitoring Program* (pp. 89-115). Townsville: Townsville Port Authority.

Gruber, R., Waterhouse, J., Logan, M., Petus, C., Howley, C., Lewis, S., Tracey, D., Langlois, L., Tonin, H., Skuza, M., Costello, P., Davidson, J., Gunn, K., Lefevre, C., Shanahan, M., Wright, M., Zagorskis, I., Kroon, F., Neilen, A. 2019, Marine Monitoring Program: Annual Report for Inshore Water Quality Monitoring 2017-18. Report for the Great Barrier Reef Marine Park Authority.

Hamed, K.H. and Rao, A.R. 1998, A modified Mann-Kendall trend test for autocorrelated data. *Journal of Hydrology,* 204(1): 182-196. doi: https://doi.org/10.1016/S0022-1694(97)00125-X

Haynes, D., Müller, J., Carter, S. 2000, Pesticide and Herbicide Residues in Sediments and Seagrasses from the Great Barrier Reef World Heritage Area and Queensland Coast. *Marine Pollution Bulletin,* 41(7-12): 279-287.

Heap, A.D., Murray, E., Ryan, D.A., Gallagher, J., Tobin, G., Creasey, J., Dyall, A. 2015, Queensland Coastal Waterways Geomorphic Habitat Mapping, Version 2 (1:100 000 scale digital data). [northlimit=-10.6; southlimit=-28.2; westlimit=137.8; eastLimit=153.2; projection=WGS84].

Heck, K.L. and Orth, R.J. 2006, Predation in seagrass beds. In WD Larkum, RJ Orth & CM Duarte (Eds.), *Seagrasses: Biology, Ecology and Conservation*. Dordrecht: Springer.

Holling, C.S. 1973, Resilience and stability of ecological systems. *Annual Review of Ecology and Systematics,* 4: 1-23.

Inglis, G.J. 1999, Variation in the recruitment behaviour of seagrass seeds: implications for population dynamics and resource management. *Pacific Conservation Biology,* 5: 251-259.

Jarvis, J.C. and Moore, K.A. 2010, The role of seedlings and seed bank viability in the recovery of Chesapeake Bay, USA, *Zostera marina* populations following a large-scale decline. *Hydrobiologia,* 649(1): 55-68. doi: 10.1007/s10750-010-0258-z

Kenworthy, W.J. 2000, The role of sexual reproduction in maintaining populations of *Halophila decipiens*: implications for the biodiversity and conservation of tropical seagrass ecosystems. *Pacific Conservation Biology,* 5: 260-268.

Kilminster, K., McMahon, K., Waycott, M., Kendrick, G.A., Scanes, P., McKenzie, L., O'Brien, K.R., Lyons, M., Ferguson, A., Maxwell, P., Glasby, T., Udy, J. 2015, Unravelling complexity in seagrass systems for management: Australia as a microcosm. *Science of The Total Environment,* 534: 97-109. doi: http://dx.doi.org/10.1016/j.scitotenv.2015.04.061

Koch, E.M. 2001, Beyond light: Physical, geological, and geochemical parameters as possible submersed aquatic vegetation habitat requirements. *Estuaries,* 24(1): 1-17.

Ku, H.H. 1966, Notes on the use of propagation of error formulas. *Journal of Research of the National Bureau of Standards. Section C: Engineering and Instrumentation,* 70C(4): 263-273. doi: citeulike-article-id:11657425

Kuhnert, P.M., Liu, Y., Henderson, B.L., Dambacher, J.M., Lawrence, E., Kroon, F.J. 2015, Review of the Marine Monitoring Program (MMP), Final Report for the Great Barrier Reef Marine Park Authority (GBRMPA.*)*.

Langlois, L.A., Collier, C.J., Lewis, S., Tracey, D., Gruber, R., McKenzie, L.J. 2021, Leaf tissue nutrient C:N ratio in relation to water quality: Model assessment and implications for report card metrics. In LJ McKenzie, CJ Collier, LA Langlois, RL Yoshida, J Uusitalo & M Waycott (Eds.), *Marine Monitoring Program: Annual Report for Inshore Seagrass Monitoring 2018–19. Report for the Great Barrier Reef Marine Park Authority* (pp. 141-171). Townsville: Great Barrier Reef Marine Park Authority.

Lawrence, E. 2019, *Modelling the environmental drivers and abundance of seagrass communities in Cleveland Bay*. Townsville: Great Barrier Reef Marine Park Authority. 56.

Lawrence, E. and Gladish, D. 2018, Analysis of seagrass and pressures data across the Great Barrier Reef. *A report to the Great Barrier Reef Marine Park Authority*.

Lee Long, W.J., Coles, R.G., McKenzie, L.J. 1999, Issues for Seagrass conservation management in Queensland. *Pacific Conservation Biology,* 5(4): 321-328.

Lee Long, W.J., McKenzie, L.J., Coles, R.G. 1997, Seagrass Communities in the Shoalwater Bay Region, Queensland; Spring (September) 1995 & Autumn (April) 1996. [Information Series]. *Queensland Department of Primary Industries Information Series QI96042*, 38.

Lee Long, W.J., McKenzie, L.J., Roelofs, A.J., Makey, L.J., Coles, R.G., C.A., R. 1998, Baseline Survey of Hinchinbrook Region Seagrasses - October (Spring) 1996. Research publication No. 51. 26.

Lee Long, W.J., Mellors, J.E., Coles, R.G. 1993, Seagrasses between Cape York and Hervey Bay, Queensland, Australia. *Australian Journal of Marine and Freshwater Research,* 44: 19-32.

Lewis, S.E., Bartley, R., Wilkinson, S.N., Bainbridge, Z.T., Henderson, A.E., James, C.S., Irvine, S.A., Brodie, J.E. 2021, Land use change in the river basins of the Great Barrier Reef, 1860 to 2019: A foundation for understanding environmental history across the catchment to reef continuum. *Marine Pollution Bulletin,* 166: 112193. doi: https://doi.org/10.1016/j.marpolbul.2021.112193

Limpus, C.J., Limpus, D.J., Arthur, K.E., Parmenter, C.J. 2005, *Monitoring Green Turtle Population Dynamics in Shoalwater Bay:2000 - 2004. GBRMPA Research Publication No 83* Townsville: Great Barrier Reef Marine Park Authority.

Lobb, K.F. (2006). *Broad scale coastal nutrient assessment of the inshore Great Barrier Reef using carbon and nitrogen in marine plants and sediments.* Bachelor of Science with Honours, The University of Queensland, Brisbane.

Magno-Canto, M., Robson, B.J., McKinna, L., Fabricius, K. 2019, Model for deriving benthic irradiance in the Great Barrier Reef from MODIS satellite imagery. *Optics Express,* 27(20). doi: https://doi.org/10.1364/OE.27.0A1350

Maxwell, W.G.H. 1968, *Atlas of the Great Barrier Reef*. Amsterdam: Elsevier Publishing Company. 258.

McCulloch, M., Pailles, C., Moody, P., Martin, C.E. 2003, Tracing the source of sediment and phosphorus into the Great Barrier Reef lagoon. *Earth and Planetary Science Letters,* 210: 249-258.

McKenzie, L., Mellors, J., Waycott, M. 2007, Great Barrier Reef Water Quality Protection Plan - Marine Monitoring Program Intertidal Seagrass. FINAL REPORT for the sampling period 1st September 2006 - 31st May 2007. *Project 1.1.3 Condition trend and risk in coastal habitats: Seagrass Indicators, distribution and thresholds of potential concern*.

McKenzie, L.J. 2007, *Relationships between seagrass communities and sediment properties along the Queensland coast. Progress report to the Marine and Tropical Sciences Research Facility*. Cairns Reef and Rainforest Research Centre Ltd. 25.

McKenzie, L.J. 2009, Observing change in seagrass habitats of the GBR– Seagrass-Watch monitoring: Deriving seagrass abundance indicators for regional habitat guidelines,. In LJ McKenzie & M Waycott (Eds.), *Marine and Tropical Sciences Research Facility Milestone and Progress Report #3, 2008-2009 (ARP 3) Project 1.1.3 Report 3, 11th June 2000* (pp. 7-11). Cairns: RRRC.

McKenzie, L.J., Campbell, S.J., Roder, C.A. 2003, *Seagrass-Watch: Manual for Mapping & Monitoring Seagrass Resources* (2nd ed.). Cairns: QFS, NFC.

McKenzie, L.J., Collier, C.J., Langlois, L.A., Yoshida, R.L., Uusitalo, J., Waycott, M. 2021a, *Marine Monitoring Program: Annual Report for Inshore Seagrass Monitoring 2018–19. Report for the Great Barrier Reef Marine Park Authority*. Townsville: Great Barrier Reef Marine Park Authority. 206.

McKenzie, L.J., Collier, C.J., Langlois, L.A., Yoshida, R.L., Uusitalo, J., Waycott, M. 2021b, *Marine Monitoring Program: Annual Report for Inshore Seagrass Monitoring 2019–20. Report for the Great Barrier Reef Marine Park Authority*. Townsville: Great Barrier Reef Marine Park Authority. 168.

McKenzie, L.J., Collier, C.J., Waycott, M. 2012, Reef Rescue Marine Monitoring Program: Inshore seagrass, annual report for the sampling period 1st September 2010-31st May 2011. 230pp.

McKenzie, L.J., Finkbeiner, M.A., Kirkman, H. 2001, Methods for mapping seagrass distribution. In FT Short & RG Coles (Eds.), *Global Seagrass Research Methods* (pp. 101-121). Amsterdam: Elsevier Science B.V.

McKenzie, L.J., Lee Long, W.J., Roelofs, A., Roder, C.A., Coles, R. 1998, *Port of Mourilyan seagrass monitoring - first four years. EcoPorts Monograph Series No. 15.* Brisbane: Ports Corporation of Queensland. 30.

McKenzie, L.J., Nordlund, L.M., Jones, B.L., Cullen-Unsworth, L.C., Roelfsema, C., Unsworth, R.K. 2020, The global distribution of seagrass meadows. *Environmental Research Letters,* 15(7): 074041. doi: https://doi.org/10.1088/1748-9326/ab7d06

McKenzie, L.J., Roder, C.A., Roelofs, A.J., Lee Long, W.J. 2000, Post-flood monitoring of seagrasses in Hervey Bay and the Great Sandy Strait, 1999: Implications for dugong, turtle and fisheries management. [DPI Information Series]. *Department of Primary Industries Information Series QI00059*, 46.

McKenzie, L.J., Roder, C.A., Yoshida, R.L. 2016, Seagrass and associated benthic community data derived from field surveys at Low Isles, Great Barrier Reef, conducted July-August, 1997. *PANGAEA*: doi:10.1594/PANGAEA.858945. doi: 10.1594/PANGAEA.858945

McKenzie, L.J., Unsworth, R.K.F., Waycott, M. 2010a, Reef Rescue Marine Monitoring Program: Intertidal Seagrass, Annual Report for the sampling period 1st September 2009 – 31st May 2010. 136.

McKenzie, L.J., Waycott, M., Unsworth Richard, K.F., Collier, C. 2018, Inshore seagrass monitoring *Marine Monitoring Program quality assurance and quality control manual 2016/17* (pp. 78-95). Townsville: Great Barrier Reef Marine Park Authority

McKenzie, L.J., Waycott, M., Unsworth Richard, K.F., Collier, C. 2019, Inshore seagrass monitoring. In Great Barrier Reef Marine Park Authority (Ed.), *Marine Monitoring Program quality assurance and quality control manual 2017–2018* (pp. 73-98). Townsville: Great Barrier Reef Marine Park Authority

McKenzie, L.J., Waycott, M., Unsworth Richard, K.F., Collier, C. 2021c, Inshore seagrass monitoring. In Great Barrier Reef Marine Park Authority (Ed.), *Marine Monitoring Program quality assurance and quality control manual 2019-20* (pp. 67-89). Townsville: Great Barrier Reef Marine Park Authority

McKenzie, L.J., Yoshida, R.L., Grech, A., Coles, R. 2010b, *Queensland seagrasses. Status 2010 - Torres Strait and East Coast*. Cairns: Fisheries Queensland (DEEDI). 6.

McKenzie, L.J., Yoshida, R.L., Grech, A., Coles, R. 2014a, Composite of coastal seagrass meadows in Queensland, Australia - November 1984 to June 2010. . *PANGAEA*: http://doi.pangaea.de/10.1594/PANGAEA.826368.

McKenzie, L.J., Yoshida, R.L., Unsworth, R.K.F. 2014b, Disturbance influences the invasion of a seagrass into an existing meadow. *Marine Pollution Bulletin,* 86(1–2): 186-196. doi: http://dx.doi.org/10.1016/j.marpolbul.2014.07.019

McMahon, K.M., Bengston-Nash, S., Eaglesham, G.K., Mueller, J., Duke, N.C., Winderlich, S. 2005, Herbicide contamination and the potential impact to seagrass meadows in Hervey Bay, Queensland, Australia. *Marine Pollution Bulletin,* 51: 325-334.

Mellors, J., Waycott, M., Marsh, H. 2005, Variation in biogeochemical parameters across intertidal seagrass meadows in the central Great Barrier Reef region. *Marine Pollution Bulletin,* 51(1-4): 335-342. doi: 10.1016/j.marpolbul.2004.10.046

Mellors, J.E. (2003). *Sediment and nutrient dynamics in coastal intertidal seagrass of north eastern tropical Australia. PhD Thesis.*, James Cook University, Townsville, School of Tropical Environment Studies and Geography.

Monitoring River Health Initiative. 1994, *River Bioassessment Manual, National River Processes and Management Program, Tasmania.* http://www.environment.gov.au/water/publications/environmental/rivers/nrhp/bioassess.html, *accessed 10 February 2014.* 44.

Moran, D., Robson, B., Gruber, R., Waterhouse, J., Logan, M., Petus, C., Howley, C., Lewis, S., Tracey, D., James, C., Mellors, J., Bove, U., Davidson, J., Glasson, K., Jaworski, S., Lefevre, C., Macadam, A., Shanahan, M., Vasile, R., Zagorskis, I., Shellberg, J. 2022, Marine Monitoring Program: Annual Report for Inshore Water Quality Monitoring 2020–21. Report for the Great Barrier Reef Marine Park Authority. 318.

O'Brien, K.R., Waycott, M., Maxwell, P., Kendrick, G.A., Udy, J.W., Ferguson, A.J.P., Kilminster, K., Scanes, P., McKenzie, L.J., McMahon, K., Adams, M.P., Samper-Villarreal, J., Collier, C., Lyons, M., Mumby, P.J., Radke, L., Christianen, M.J.A., Dennison, W.C. 2018, Seagrass ecosystem trajectory depends on the relative timescales of resistance, recovery and disturbance. *Marine Pollution Bulletin,* 134: 166-176. doi: https://doi.org/10.1016/j.marpolbul.2017.09.006

Petus, C., Devlin, M., Thompson, A., McKenzie, L., Teixeira da Silva, E., Collier, C., Tracey, D., Martin, K. 2016, Estimating the Exposure of Coral Reefs and Seagrass Meadows to Land-Sourced Contaminants in River Flood Plumes of the Great Barrier Reef: Validating a Simple Satellite Risk Framework with Environmental Data. *Remote Sensing,* 8(3): 210.

Petus, C., Waterhouse, J., Lewis, S., Vacher, M., Tracey, D., Devlin, M. 2019, A flood of information: Using Sentinel-3 water colour products to assure continuity in the monitoring of water quality trends in the Great Barrier Reef (Australia). *Journal of Environmental Management,* 248: 109255. doi: https://doi.org/10.1016/j.jenvman.2019.07.026

Preen, A.R., Lee Long, W.J., Coles, R.G. 1995, Flood and cyclone related loss, and partial recovery, of more than 1,000 km^2 of seagrass in Hervey Bay, Queensland, Australia. *Aquatic Botany,* 52: 3-17.

Rasheed, M.A., McKenna, S.A., Carter, A.B., Coles, R.G. 2014, Contrasting recovery of shallow and deep water seagrass communities following climate associated losses in tropical north Queensland, Australia. *Marine Pollution Bulletin,* 83(2): 491-499. doi: http://dx.doi.org/10.1016/j.marpolbul.2014.02.013

Rasheed, M.A., Thomas, R., Roelofs, A.J., Neil, K.M., Kerville, S.P. 2003, *Port Curtis and Rodds Bay seagrass and benthic macro-invertebrate community baseline survey, November/December 2002. DPI Information Series QI03058*. Cairns: DPI. 47

Rasheed, M.A. and Unsworth, R.K.F. 2011, Long-term climate-associated dynamics of a tropical seagrass meadow: implications for the future. *Marine Ecology Progress Series,* 422: 93-103.

Rigby, R.A. and Stasinopoulos, D.M. 2005, Generalized additive models for location, scale and shape. *Appl.Statist,* 54(3): 507-554.

Robson, B., Canto, M., Collier, C., di Perna, S., Logan, M., Menendez, P., McKinna, L., Noonan, S., Fabricius, K. 2019, Benthic light as an ecologically-validated GBR-wide indicator for water quality. *Report to the National Environmental Science Program*, 40.

Roff, G., Clark, T.R., Reymond, C.E., Zhao, J.-x., Feng, Y., McCook, L.J., Done, T.J., Pandolfi, J.M. 2013, Palaeoecological evidence of a historical collapse of corals at Pelorus Island, inshore Great Barrier Reef, following European settlement. *Proceedings of the Royal Society of London B: Biological Sciences,* 280(1750). doi: 10.1098/rspb.2012.2100

Saunders, M.I., Bayraktarov, E., Roelfsema, C.M., Leona, J.X., Samper-Villarreal, J., Phinn, S.R., Lovelock, C.E., Mumby, P.J. 2015, Spatial and temporal variability of seagrass at Lizard Island, Great Barrier Reef. *Botanica Marina,* 58(1): 35–49.

Schaffelke, B., Anthony, K., Blake, J., Brodie, J., Collier, C., Devlin, M., Fabricius, K., Martin, K., McKenzie, L.J., Negri, A., Ronan, M., Thompson, A., Warne, M. 2013, Marine and coastal ecosystem impacts *Synthesis of evidence to support the Reef Water Quality Scientific Consensus Statement 2013* (pp. 47). Brisbane: Department of the Premier and Cabinet, Queensland Government.

Schaffelke, B., Collier, C., Kroon, F., Lough, J., McKenzie, L.J., Ronan, M., Uthicke, S., Brodie, J. 2017, The condition of coastal and marine ecosystems of the Great Barrier Reef and their responses to water quality and disturbances *Scientific Consensus Statement 2017: A synthesis of the science of land-based water quality impacts on the Great Barrier Reef* (pp. 83). Brisbane: Department of the Premier and Cabinet, Queensland Government.

Shelton, A.O. 2008, Skewed sex ratios, pollen limitation, and reproductive failure in the dioecious seagrass *Phyllospadix*. *Ecology,* 89: 3020-3029.

Tan, Y.M., Dalby, O., Kendrick, G.A., Statton, J., Sinclair, E.A., Fraser, M.W., Macreadie, P.I., Gillies, C.L., Coleman, R.A., Waycott, M., van Dijk, K.-j., Vergés, A., Ross, J.D., Campbell, M.L., Matheson, F.E., Jackson, E.L., Irving, A.D., Govers, L.L., Connolly, R.M., McLeod, I.M., Rasheed, M.A., Kirkman, H., Flindt, M.R., Lange, T., Miller, A.D., Sherman, C.D.H. 2020, Seagrass Restoration Is Possible: Insights and Lessons From Australia and New Zealand. [Review]. *Frontiers in Marine Science,* 7(617). doi: 10.3389/fmars.2020.00617

Thorogood, J. and Boggon, T. 1999, Pioneer Bay Environmental Monitoring Program: Fourth monitoring event, November 1999. Undertaken on behalf of Whitsunday Shire Council. (PRC Ref: 98.04.16iii).

Timpane-Padgham, B.L., Beechie, T., Klinger, T. 2017, A systematic review of ecological attributes that confer resilience to climate change in environmental restoration. *PLoS ONE,* 12(3): e0173812. doi: 10.1371/journal.pone.0173812

Udy, J., Waycott, M., Collier, C., Kilminster, K., McMahon, K., Rasheed, M., MCKENZIE, L.J., Carter, A., Lawrence, E., Maxwell, P., Dwane, G., Martin, K., Honchin, C. 2018, *Monitoring seagrass within the Reef 2050 Integrated Monitoring and Reporting Program*. Townsville: Great Barrier Reef Marine Park Authority. 94.

Udy, J.W., Dennison, W.C., Lee Long, W.J., McKenzie, L.J. 1999, Responses of seagrass to nutrients in the Great Barrier Reef, Australia. *Marine Ecology Progress Series,* 185: 257-271.

Unsworth, R.K.F., Collier, C.J., Waycott, M., McKenzie, L.J., Cullen-Unsworth, L.C. 2015, A framework for the resilience of seagrass ecosystems. *Marine Pollution Bulletin,* 100(1): 34-46. doi: http://dx.doi.org/10.1016/j.marpolbul.2015.08.016

Unsworth, R.K.F., Rasheed, M.A., Chartrand, K.M., Roelofs, A.J. 2012, Solar radiation and tidal exposure as environmental drivers of *Enhalus acoroides* dominated seagrass meadows. *Plos One,* 7(3). doi: 10.1371/journal.pone.0034133

Uthicke, S., Castro-Sanguino, C., Ferrari, R., Fabricius, K., Lawrey, E., Flores, F., Patel, F., Brunner, C., Negri, A. 2020, *From Exposure to Risk: Novel Experimental Approaches to Analyse Cumulative Impacts and Determine Thresholds in the Great Barrier Reef World Heritage Area (GBRWHA). Report to the National Environmental Science Program*.

Uthicke, S., Fabricius, K., De'ath, G., Negri, A., Smith, R., Warne, M., Noonan, S., Johansson, C., Gorsuch, H., Anthony, K. 2016, *Multiple and cumulative impacts on the GBR: assessment of current status and development of improved approaches for management Final Report. Report to the National Environmental Science Programme.*

van Katwijk, M.M., Bos, A.R., Hermus, D.C.R., Suykerbuyk, W. 2010, Sediment modification by seagrass beds: Muddification and sandification induced by plant cover and environmental conditions. *Estuarine Coastal And Shelf Science,* 89(2): 175-181. doi: 10.1016/j.ecss.2010.06.008

Waterhouse, J., Brodie, J., Coppo, C., Tracey, D., da Silva, E., Howley, C., Petus, C., McKenzie, L., Lewis, S., McCloskey, G., Higham, W. 2016, Assessment of the relative risk of water quality to ecosystems of the eastern Cape York NRM Region, Great Barrier Reef. A report to South Cape York Catchments. TropWATER Report 16/24, . 105.

Waycott, M., Collier, C., McMahon, K., Ralph, P.J., McKenzie, L.J., Udy, J.W., Grech, A. 2007, Vulnerability of seagrasses in the Great Barrier Reef to climate change - Chapter 8: . In JE Johnson & PA Marshall (Eds.), *Climate Change and the Great Barrier Reef: A Vulnerability Assessment, Part II: Species and species groups* (pp. 193-236). Townsville: Great Barrier Reef Marine Park Authority

Waycott, M., McMahon, K.M., Mellors, J.E., Calladine, A., Kleine, D. 2004, *A guide to tropical seagrasses of the Indo-West Pacific*. Townsville: James Cook University. 72.

Webster, I. and Ford, P. 2010, Delivery, deposition and redistribution of fine sediments within macrotidal Fitzroy Estuary/Keppel Bay: southern Great Barrier Reef, Australia. *Continental Shelf Research,* 30: 793–805.

Wood, S.N. 2020, mgcv: mixed GAM computation vehicle with automatic smoothness estimation. R-package version 1.8–33. https://CRAN.R-project.org/package=mgcv.

Wooldridge, S.A. 2017, Preventable fine sediment export from the Burdekin River catchment reduces coastal seagrass abundance and increases dugong mortality within the Townsville region of the Great Barrier Reef, Australia. *Marine Pollution Bulletin,* 114(2): 671-678. doi: http://dx.doi.org/10.1016/j.marpolbul.2016.10.053

York, P.H., Carter, A.B., Chartrand, K., Sankey, T., Wells, L., Rasheed, M.A. 2015, Dynamics of a deep-water seagrass population on the Great Barrier Reef: annual occurrence and response to a major dredging program. *Scientific Reports,* 5: 13167. doi: 10.1038/srep13167 http://www.nature.com/articles/srep13167#supplementary-information

Appendix 1 Case study

Towards an improved Seagrass Index: tissue nutrients and seagrass reproductive effort metric changes

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Executive summary

This document summarises the proposed changes to the seagrass metrics and scoring for the Seagrass Index of the inshore seagrass component of the Marine Monitoring Program (MMP). These proposed changes include replacement of the reproductive effort metric with a resilience metric, and removal of the seagrass nutrient status metric. There are three supporting case studies (Collier *et al.* 2019; Collier *et al.* 2021a; Langlois *et al.* 2021), which build on a history of investigations and assessments that have been undertaken (as described below) to reach this stage at which a change to the Index is proposed below. The MMP Inshore Seagrass monitoring team have undertaken this work as part of a commitment to program improvement, and to provide the transparency and evidence needed to support changes in the Index. These will be the first changes to the seagrass Index of the MMP since its implementation in 2009.

Summary of metrics

Reproductive effort and resilience

Sexual reproduction is important for seagrass resilience as it is needed to form seed banks, which facilitate meadow recovery following periods of decline, and seed germination increases clonal diversity of the meadow (richness). It is therefore a good indicator of seagrass health (Kenworthy 2000; Jarvis and Moore 2010; Rasheed *et al.* 2014) and partially explains inshore Great Barrier Reef (Reef) seagrass abundance (per cent cover) in the subsequent sampling year of the MMP (Lawrence and Gladish 2018; McKenzie *et al.* 2021a). It is also simple to measure. For these reasons, reproductive effort has been measured in the MMP Inshore seagrass monitoring since its inception and is one of three seagrass metrics reported in the Reef health index, until 2020–21.

Based on over 15 years of data, the measure and metric have been re-evaluated in a series of studies (Kuhnert *et al.* 2015; Lawrence and Gladish 2018; Collier *et al.* 2019; Collier *et al.* 2021a). These studies identified that there is low power in the reproductive effort data because of a large count of zeros and high variation among samples (standard deviation). There were concerns that the category thresholds led to “jumpy” inter-annual variability in scores, which could be affected by the timing of sampling, and scoring was not broadly applicable to different habitat types, and species. For example, healthy intertidal reef habitat sites such as Green Island (GI1 and GI2), which maintain high abundance scores (mostly 50 or higher) but have very low reproduction scores (mostly 0 or 25), because those meadows have a different resilience ‘strategy’ — they resist disturbance and persist over time. This highlights that reproductive effort is just one aspect of resilience.

A multivariate composite resilience metric was recommended and developed (Collier *et al.* 2021a), which aligns with Reef 2050 objectives which include *“….maintaining and restoring the connectivity, resilience and condition….*”. To accommodate the different data sources that input to the resilience metric, this new metric was designed as a decision tree with groupings and scores ranging within that grouping. The decisions are based on low resistance thresholds including species composition and abundance, reproductive structures and previous reproductive history (probability of having formed a seed bank). It is designed around differences in resistance and recovery strategies cognizant of different seagrass species and site history. The metric is scored on a continuous scale, which is subject to less inter-annual variability compared to the categorical scoring of the reproductive effort metric.

Nutrient status (C:N)

Seagrass leaf tissue nutrients were measured as an indicator of changes in water quality, including changes in nutrient availability, and to plant growth requirements. Luxury nitrogen (N) uptake can occur if there is an increase in N availability, or if demand is low (due to low growth rates, low carbon (C) fixation, and often attributed to low light) leading to a decline in C:N (McKenzie *et al.* 2021a). The seagrass leaf tissue carbon to nitrogen (C:N) ratio has been reported as a metric in the seagrass Index since its inception. There were no other indicators of water quality available at the seagrass monitoring sites at the time of the original metrics inception, and so C:N was a surrogate indicator, integrating water quality and seagrass responses to it over time (i.e. a measure on any one day reflects previous weeks or even months of uptake and growth).

After 15 years of measuring seagrass leaf tissue nutrients, there was sufficient C:N data for analysis with suitable data from the water quality sub-program (including predicted annual nitrogen loads) and daily light levels measured at seagrass sites (Langlois *et al.* 2021). C:N responded to the water quality variables most consistently in coastal habitat where there is a wide range in their values. For other habitats, and when investigating species separately, the response of C:N to the water quality variables was mostly inconsistent and unpredictable. It is important to note that the water quality variables used were summarised at coarse scales (annual) to accommodate the influence of wet season loads on a seagrass measure taken in the late-dry season, and that finer temporal data may have shown clearer responses between water quality indicators and C:N. There was also no indication that C:N was acting as an 'early-warning indicator' of imminent seagrass decline or recovery in following years, possibly due the annual time-scale over which it is measured and reported.

For four over-arching reasons, it was proposed that the seagrass leaf tissue nutrient metric should no longer be included in the seagrass Index, because of:

- the findings of analysis of the historical seagrass leaf tissue C:N data (Langlois *et al.* 2021),
- a slight change in the focus of the MMP to monitoring condition, trend and resilience of the inshore Reef to pressures in general, as opposed to focussing principally on nutrient pollutants and river inputs e.g. McKenzie *et al.* 2007,
- an increase in the availability of data on changes to water quality and other pressures through remote sensing (Gruber *et al.* 2019; Magno-Canto *et al.* 2019; Petus *et al.* 2019; Robson *et al.* 2019) and modelling (Baird *et al.* 2016; Baird *et al.* 2019 that are suitable for regional or Reef-level reporting (but are not as suitable for site-level reporting),
- a need to ensure cost-effectiveness of the program while meeting the program objectives; processing and analysing seagrass tissue nutrients is costly.

The previous 15 years of C:N data have established a baseline of sorts, that can be used to track long-term changes in seagrass tissue nutrients over time. There is justification for ongoing or sporadic (e.g. every 3 years) collection of the tissue nutrient samples to identify chronic changes in nutrient pools, processes and primary sources in inshore seagrass meadows. The recommendation given here is for removal of C:N from the Index, rather than for complete removal of it as a measure in the Reef.

New vs old Scores comparisons

When the reproductive effort metric is replaced with the resilience metric, the overall effect raises the Index (Figure 1), particularly in 'Good' years. The reasons for this are described in Collier *et al.* (2021a). Thus, the trend over time is retained and even enhanced by the resilience metric. At a Reef-scale, the abundance and resilience metric track together over time.

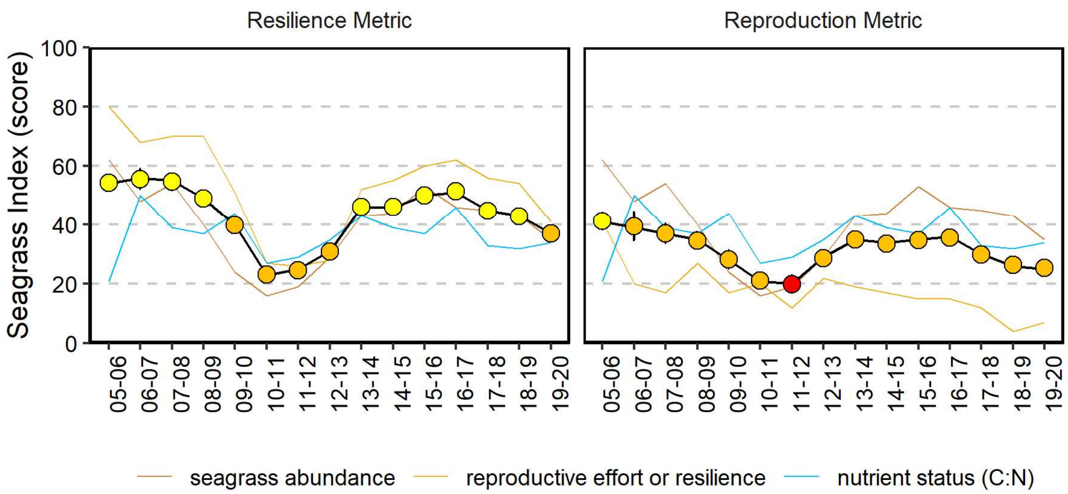

Figure 1. Comparison of Seagrass Index scores of the Reef between the new proposed resilience metric and the old reproduction metric

Similar overall effects were observed in the regions i.e. overall lift in the Index, especially in good years (Figure 2). In most NRMs the resilience metric varies over time in a different manner to the reproductive metric, especially in Cape York, Fitzroy and Burnett-Mary regions.

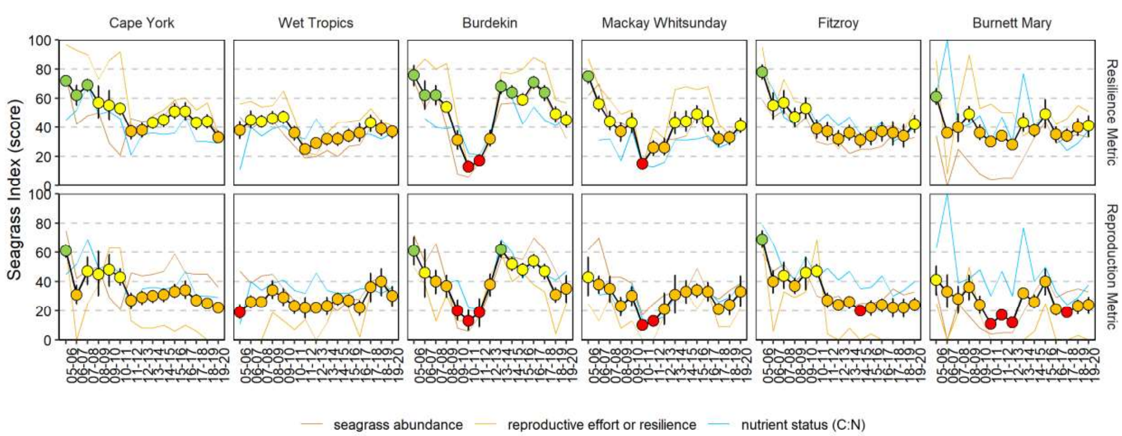

Figure 2. Comparison of Seagrass Index scores by NRM Region between the old reproduction metric and the new proposed resilience metric

The next step was to examine the influence of the nutrient status score on the Seagrass Index. The removal of C:N further lifts the Seagrass index, but it stays within the same broad categories except in early years (Figure 3)

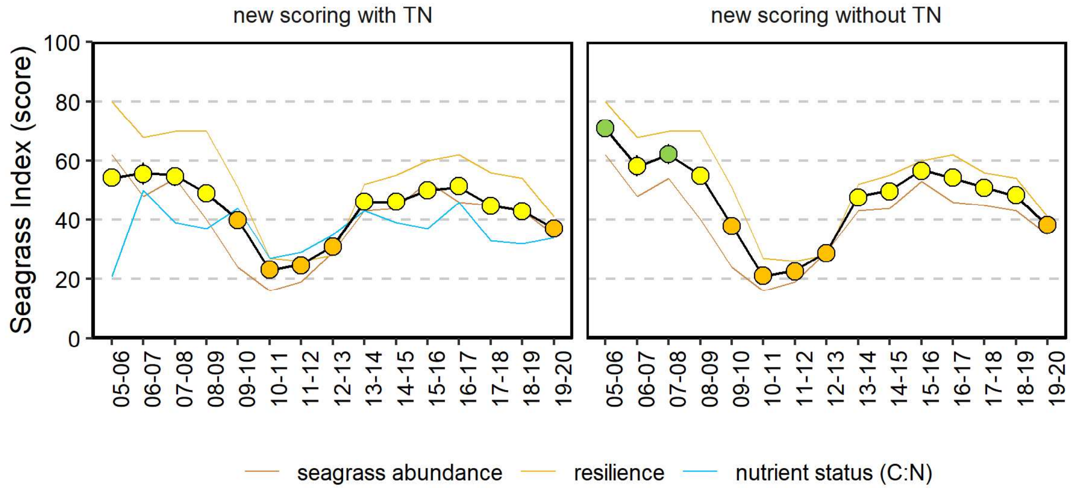

Figure 3. Comparison of Seagrass Index scores of the Reef with new proposed resilience metric and with or without the tissue nutrient indicator C:N (labelled as TN).

The nutrient status scores have much more influence when looking at smaller scales including NRMs and sites (not shown here). Removal of C:N raises the high and reduces the low scores (in most cases), because nutrient status acted like a ‘stabiliser’, as it did not vary much between years (Figure 4). The largest effects were in the Wet tropics and Burnett-Mary with the other NRMs having only minor changes. The single largest change was in the Burnett-Mary in 2006-07 when abundance and reproductive effort were very poor, but nutrient status was very good, so its removal had a large effect on the score in that year.

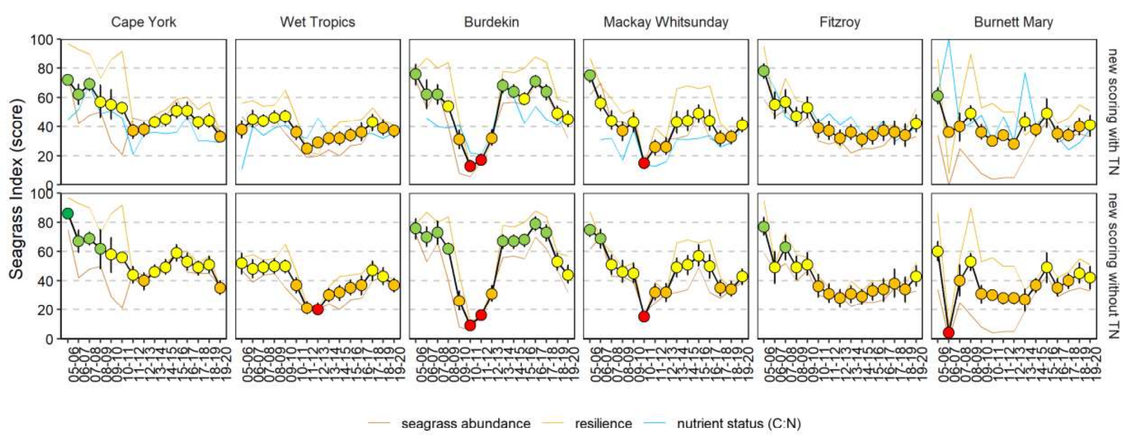

Figure 4. Comparison of Seagrass Index scores by NRM Region with new proposed resilience metric and with or without TN.

The proposed new Index with abundance and resilience metrics lead to a more defined trend over time (Figure 5). Furthermore, a moderate score becomes more common, whereas previously a poor score was the most common. The Index was good or moderate in the early years, then there was a sharp decline over the 2009-10 and 2010-11 period due to extreme disturbances (cyclones Hamish and Yasi), and finally recovery and stabilisation occurred over the most recent years. In both cases (old and new score), the Index in 2019-20 is one grade lower than it was at project inception (2005-06), so in that sense the over-all trend is retained. Then in 2019-20, the new Index declined to poor, and is now two grades lower than in 2005-06 while the old scoring stayed at poor, highlighting the more dynamic nature and greater range of the new Index. There is greater scope to track change over time (particularly declines) with the Index more commonly at moderate, instead of poor.

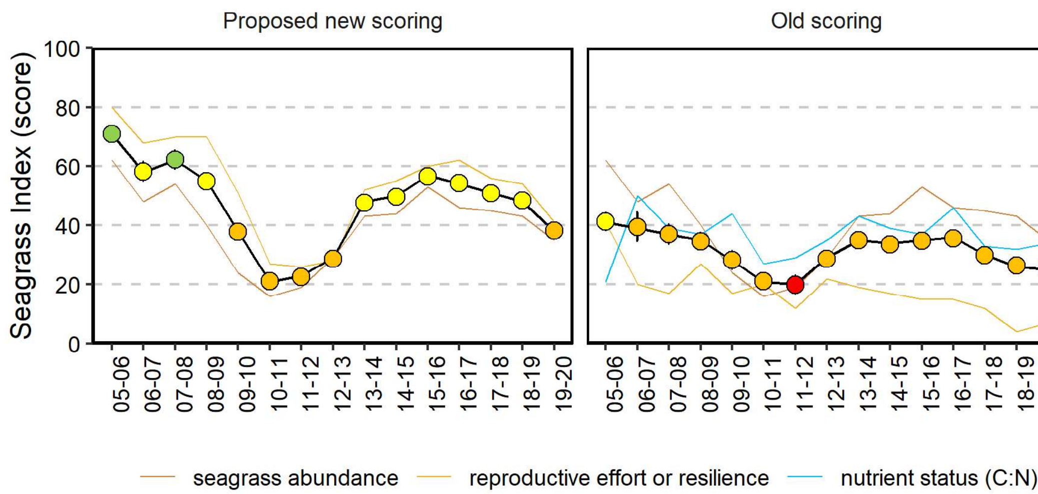

Figure 5. Reef seagrass index of new proposed scoring (abundance and resilience scores only) and of the old scoring (abundance, reproduction and TN).

Similar to the Reef Seagrass Index, the Index per NRM is more pronounced with the combined changes (Figure 6).

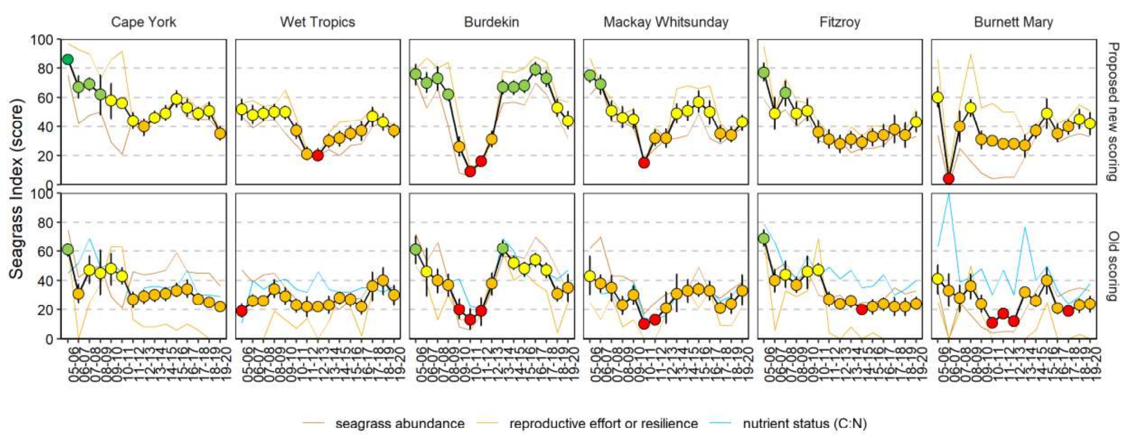

Figure 6. Per NRM region seagrass index of new proposed scoring (abundance and resilience scores only) and of the old scoring (abundance, reproduction and TN).

Conclusions

The new proposed Index and metric scores were able to be developed as a result of the long-term data that is available now, which was not available at Report Card inception in 2009, and through evolving scientific understanding of seagrass health and resilience. The proposed new Index has been developed with the current Reef 2050 Plan, WQIP and MMP program objectives in mind. We have undertaken two detailed supporting case studies in conjunction with additional supporting external reviews and analyses to provide the quantitative evidence for the proposed changes. These provide transparency and communicate the need for the changes. The result is an Index that represents seagrass condition and resilience using existing MMP measures that enable a long-term trend to be reported. The new proposed Index varies more over time than the old Index and is higher on average, providing more capacity to detect decline in future years. We re-iterate previous recommendations to further investigate indicators that can be used to report against the Reef 2050 LTSP objectives including connectivity and a process-based understanding of resilience (Udy *et al.* 2018), and adapt the metrics and Index in the future accordingly.

References

Baird, M.E., Adams, M.P., Babcock, R.C., Oubelkheir, K., Mongin, M., Wild-Allen, K.A., Skerratt, J., Robson, B.J., Petrou, K., Ralph, P.J., O'Brien, K.R., Carter, A.B., Jarvis, J.C., Rasheed, M.A. 2016, A biophysical representation of seagrass growth for application in a complex shallow-water biogeochemical model. *Ecological Modelling*, 325: 13-27. doi: 10.1016/j.ecolmodel.2015.12.011

Baird, M.E., Wild-Allen, K.A., Parslow, J., Mongin, M., Robson, B., Skerratt, J., Rizwi, F., Soja-Woznaik, M., Jones, E., Herzfeld, M., Margvelashvili, N., Andrewartha, J., Langlais, C., Adams, M.P., Cherukuru, N., Gustafsson, M., Hadley, S., Ralph, P.J., Rosebrock, U., Schroeder, T., Laiolo, L., Harrison, D., Steven, A.D.L. 2019, CSIRO Environmental Modelling Suite (EMS): Scientific description of the optical and biogeochemical models (vB3p0). *Geosci. Model Dev. Discuss.*, 2019: 1-107. doi: 10.5194/gmd-2019-115

Collier, C.J., Langlois, L., Waycott, M., McKenzie, L.J. 2021, Resilience in practice : development of a seagrass resilience metric for the GBR inshore seagrass Marine Monitoring Program. Case Study for the Great Barrier Reef Marine Park Authority. 61.

Collier, C.J., Lawerence, E., Waycott, M., Langlois, L.A., McKenzie, L.J. 2019a, Reproductive effort as a predictor of future seagrass cover: Model assessment and implications for report card metrics and the development of a seagrass resilience indicator Marine Monitoring Program: Annual Report for Inshore Seagrass Monitoring 2017–18. Report for the Great Barrier Reef Marine Park Authority (pp. 135-142). Townsville: Great Barrier Reef Marine Park Authority.

Collier, C.J., Lawrence, E., Waycott, M., Langlois, L.A., McKenzie, L.J. 2019b, Case Study #2: Reproductive effort as a predictor of future seagrass cover: Model assessment and implications for report card metrics and the development of a seagrass resilience indicator. Annual report for the inshore seagrass monitoring 2017-2018.

Gruber, R., Waterhouse, J., Logan, M., Petus, C., Howley, C., Lewis, S., Tracey, D., Langlois, L., Tonin, H., Skuza, M., Costello, P., Davidson, J., Gunn, K., Lefevre, C., Shanahan, M., Wright, M., Zagorskis, I., Kroon, F., Neilen, A. 2019, Marine Monitoring Program: Annual Report for Inshore Water Quality Monitoring 2017-18. Report for the Great Barrier Reef Marine Park Authority.

Jarvis, J. and Moore, K. 2010, The role of seedlings and seed bank viability in the recovery of Chesapeake Bay, USA, *Zostera marina* populations following a large-scale decline. *Hydrobiologia*, 649(1): 55-68. doi: 10.1007/s10750-010-0258-z

Kenworthy, J.W. 1999, The role of sexual reproduction in maintaining populations of *Halophila decipiens*: implications for the biodiversity and conservation of tropical seagrass ecosystems. Pacific Conservation Biology, 5(4): 260-268.

Kuhnert, P.M., Liu, Y., Henderson, B.L., Dambacher, J.M., Lawrence, E., Kroon, F.J. 2015, Review of the Marine Monitoring Program (MMP), Final Report for the Great Barrier Reef Marine Park Authority (GBRMPA.).

Langlois, L.A., Collier, C.J., Lewis, S., Tracey, D., Gruber, R., McKenzie, L.J. 2021, Leaf tissue nutrient C:N ratio in relation to water quality: Model assessment and implications for report card metrics. In LJ McKenzie, CJ Collier, LA Langlois, RL Yoshida, J Uusitalo & M Waycott (Eds.), Marine Monitoring Program: Annual Report for Inshore Seagrass Monitoring 2018–19. Report for the Great Barrier Reef Marine Park Authority (pp. 141-171). Townsville: Great Barrier Reef Marine Park Authority.

Lawrence, E. and Gladish, D. 2018, Analysis of seagrass and pressures data across the Great Barrier Reef. A report to the Great Barrier Reef Marine Park Authority.

Magno-Canto, M., Robson, B.J., McKinna, L., Fabricius, K. 2019, Model for deriving benthic irradiance in the Great Barrier Reef from MODIS satellite imagery. *Optics Express*, 27(20). doi: 10.1364/OE.27.0A1350

McKenzie, L., Mellors, J., Waycott, M. 2007, Great Barrier Reef Water Quality Protection Plan - Marine Monitoring Program Intertidal Seagrass. FINAL REPORT for the sampling period 1st September 2006 - 31st May 2007. Project 1.1.3 Condition trend and risk in coastal habitats: Seagrass Indicators, distribution and thresholds of potential concern.

McKenzie, L.J., Collier, C.J., Langlois, L.A., Yoshida, R.L., Uusitalo, J., Smith, N., Waycott, M. 2020, Marine Monitoring Program: Annual Report for inshore seagrass monitoring 2018-2019. Report for the Great Barrier Reef Marine Park Authority.

Petus, C., Waterhouse, J., Lewis, S., Vacher, M., Tracey, D., Devlin, M. 2019, A flood of information: Using Sentinel-3 water colour products to assure continuity in the monitoring of water quality trends in the Great Barrier Reef (Australia). *Journal of Environmental Management*, 248: 109255. doi: 10.1016/j.jenvman.2019.07.026

Rasheed, M.A., McKenna, S.A., Carter, A.B., Coles, R.G. (2014). Contrasting recovery of shallow and deep water seagrass communities following climate associated losses in tropical north Queensland, Australia. *Marine Pollution Bulletin*, 83(2), 491-499. doi: 10.1016/j.marpolbul.2014.02.013

Robson, B., Canto, M., Collier, C., di Perna, S., Logan, M., Menendez, P., McKinna, L., Noonan, S., Fabricius, K. 2019, Benthic light as an ecologically-validated GBR-wide indicator for water quality. Report to the National Environmental Science Program, 40.

Udy, J., Waycott, M., Collier, C.J., Kilminster, K., McMahon, K., Rasheed, M., McKenzie, L., Carter, A.B., Lawrence, E., Maxwell, P., Dwane, G., Martin, K., Honchin, C. 2018, Monitoring seagrass within the Reef 2050 Integrated Monitoring and Reporting Program: Final report of the seagrass working group.

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 plateaud, 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 an 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 18).

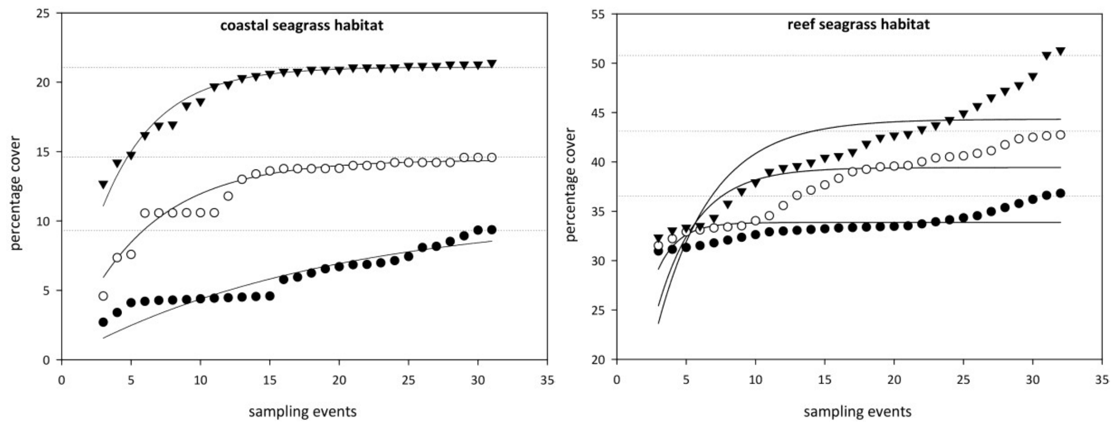

Figure 89. Relationship between sample size and the error in estimation of percentile values for seagrass abundance (per cent cover) in coastal and reef seagrass habitats in the Wet Tropics NRM. ▼ = 75th percentile, ○ = 50th percentile, ● = 20th percentile. Horizontal lines are asymptotic averages for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

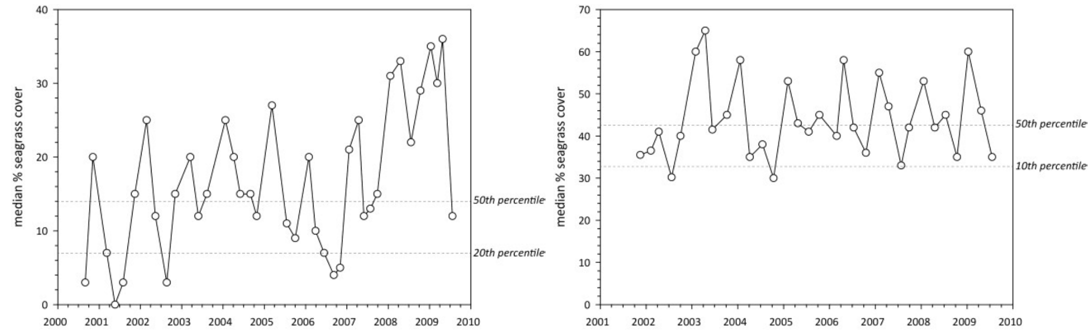

Figure 90. Median seagrass abundance (per cent cover) at Yule Point (left) and Green Island (right) plotted against the 50th and 20th percentiles for coastal and intertidal reef seagrass habitat in the Wet Tropics.

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 18). 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 per cent 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 18. 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***
	FG	reef subtidal	22	26	33	39.2
	NRM	***reef subtidal****	***22***	***26***	***33***	***39.2***
	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
	BY*	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
	BW	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
	CV	coastal intertidal		13.2	19.1	22.2
	SH1	coastal intertidal		13.2	19.1	22.2
	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
	LN	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 resilience

The status of seagrass resilience was determined using a multi-faceted resilience metric informed by existing metrics, historical data, and a conceptual understanding of resilience. Resilience can be considered as having two main elements (e.g. Timpane-Padgham *et al.* 2017; Connolly *et al.* 2018): an ability to resist disturbance, and an ability to recover from disturbances. We used a decision tree approach, which includes thresholds defining the splits, and methods for calculating scores (Figure 91). The main splits in the tree are based around:

- a ‘resistance’ component that assesses the seagrass meadow capacity to cope with disturbance based on their seagrass abundance and species composition. A low resistance site is one that has very low abundance based on the history of that site and/or has a high proportion of colonising species. These meadows are considered to be highly vulnerable to disturbances and, therefore, to have very low resilience.
- a ‘reproduction' component that is based around likelihood of producing seed banks given the presence and count of reproductive structures. These are scored based on the levels of expected reproductive effort given the life history strategy of the species present. For example, some ‘persistent’ species such as *Thalassia* are not expected to have a high number of reproductive structures, and nor does it depend on them quite as much for long-term survival compared to ‘colonising’ species.

Those two components work both individually and in collaboration, thus giving the best estimate of resilience using the existing data and indicators. The metric is scored linearly from 0 to 100. The 0–100 scale was split into thirds (rounded to the nearest ten score). This resulted in the following:

- Low resistance sites = 0–30
- Non-reproductive high resistance site = 30–70
- Reproductive high resistance site = 70–100

The methods used to arrive at each step are outlined in detail in Collier *et al*. (2021a).

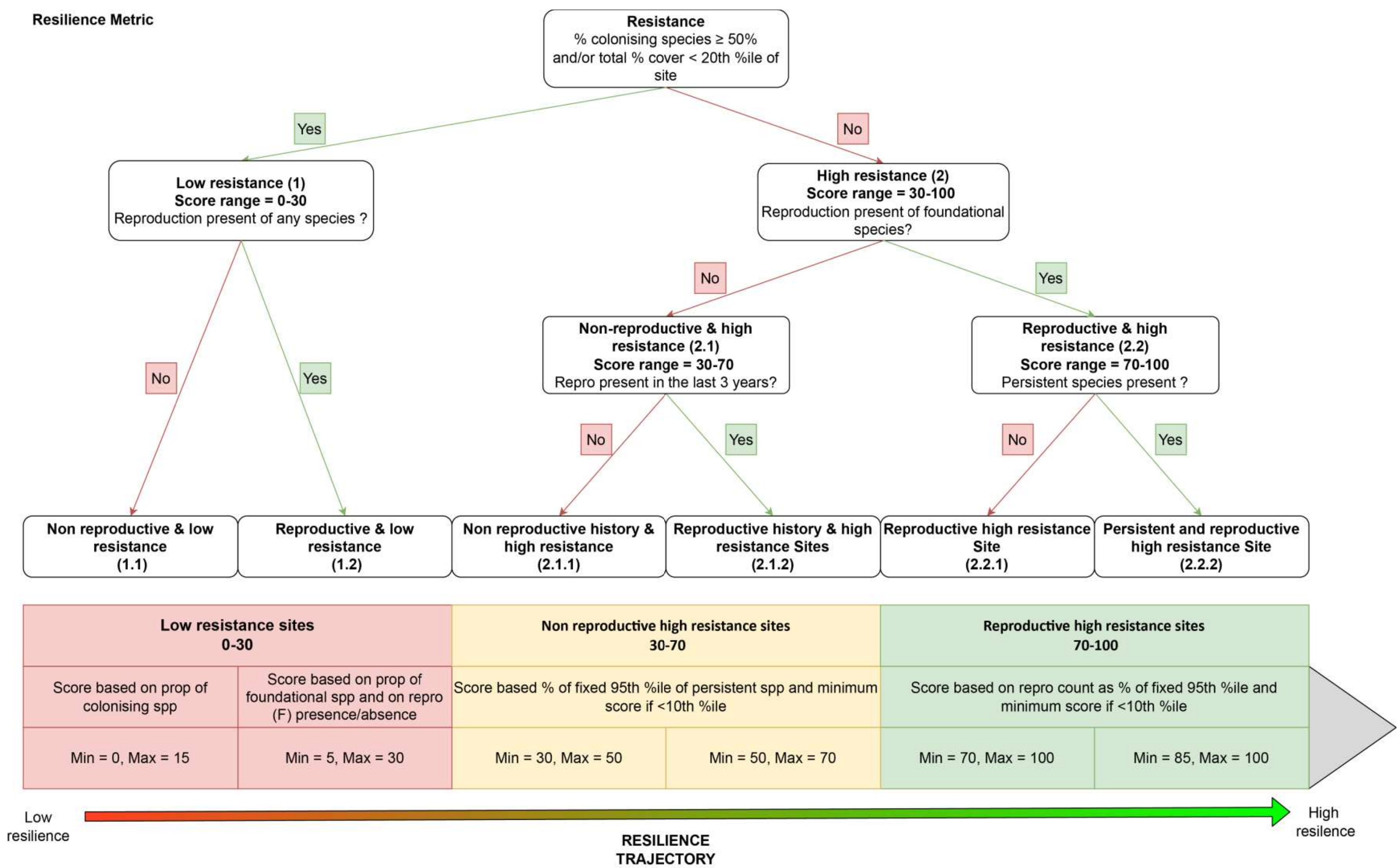

Figure 91. Overall structure of the proposed MMP resilience metric. The score ranges from 0 to 100. Splits in the tree are used to place a site in a grouping (red, yellow, or green), with grading within each grouping based on species composition and reproductive effort. Reproduction refers to sexual reproduction. From Collier *et al*. (2021a).

Appendix 3 Detailed data

Table 19. Samples collected at each inshore monitoring site per parameter for each season. Activities include: SG = seagrass cover & composition, SB=seed bank monitoring, TN=tissue samples for future nutrient analysis, EM=edge mapping, RH=reproductive effort, TL=temperature loggers, LL=light loggers. ^=subtidal.

Reef region	NRM region	Basin	Monitoring location		late dry Season (2019)							late wet Season (2020)					
					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^**	10												
				LR2^	10												
		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^	10												
				BY4^	11												
		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	3	✓	15			33	30	✓	15		
				LB2	33	30	3	✓	15			33	30	✓	15		
			Dunk Island	**DI1**	33	30	3	✓	15	✓		33	30	✓	15	✓	
				DI2	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
				DI3^	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
			Rockingham Bay	**GO1**													
			Missionary Bay	**MS1^**	10												
				MS2^	10												
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	30	✓	15	✓	✓
				JR2	33	30	3	✓	15	✓		33	30	✓	15	✓	
		Don	Bowen	**BW1**	33	30						33	30				
				BW3	33	30						33	30				
	Mackay–Whitsunday	Don	Shoal Bay	**HB1**	33	30				✓		33	30			✓	
				HB2	33	30				✓		33	30			✓	

Reef region	NRM region	Basin	Monitoring location		late dry Season (2019)							late wet Season (2020)					
					SG	SB	TN	EM	RH	TL	LL	SG	SB	EM	RH	TL	LL
		Proserpine	Pioneer Bay	**PI2**	33	30				✓		33	30			✓	
				PI3	33	30				✓		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^**	10												
				TO2^	10												
			Lindeman Island	**LN1^**	33	30	3	✓	15	✓	✓	33	30	✓	15	✓	✓
				LN2^	33	30	3	✓	15	✓							
		O'Connell	St Helens Bay	**SH1**	33	30						33	30				
			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	✓	
			Clairview	**CV1**	33							33					
				CV2	33							33					
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 20. 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, 2021. * are predicted. 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 2020–21 (hrs)	Per cent of annual daylight hours meadow exposed (2020–21)
Cape York	AP1	0.46	1.02	0.46	58.3	1.34	39.5	0.90
	AP2	0.46	1.02	0.46	58.3	1.34	39.5	0.90
Wet Tropics	LI1	0.65	0.90	0.65	141.00	3.53	128.83	2.94
	YP1	0.64	0.94	0.64	135.50	3.42	122.00	2.79
	YP2	0.52	1.06	0.52	72.00	1.90	62.83	1.43
	GI1	0.51	1.03	0.61	118.83	2.79	102.17	2.33
	GI2	0.57	0.97	0.67	153.50	3.61	139.17	3.18
	DI1	0.65	1.14	0.54	75.08	1.69	69.33	1.58
	DI2	0.55	1.24	0.44	42.17	0.95	35.33	0.81
	LB1	0.42	1.37	0.31	18.33	0.40	14.83	0.34
	LB2	0.46	1.33	0.35	19.25	0.46	13.67	0.31
Burdekin	BB1	0.58	1.30	0.58	53.17	1.24	46.67	1.07
	SB1	0.57	1.31	0.57	51.67	1.13	42.67	0.97
	MI1	0.65	1.19	0.67	81.83	2.18	73.83	1.69
	MI2	0.54	1.30	0.56	49.17	1.51	40.67	0.93
	JR1	0.47	1.32	0.47	57	1.31	40.83	0.93
	JR2	0.47	1.32	0.47	57	1.31	40.83	0.93
Mackay–Whitsunday	PI2*	0.28	1.47	0.44	80.67	1.88	76.50	1.75
	PI3*	0.17	1.58	0.33	41.50	0.96	30.83	0.70
	HM1*	0.68	1.52	0.38	56.67	1.30	46.83	1.07
	HM2*	0.68	1.52	0.38	56.67	1.30	46.83	1.07
	SI1	0.60	2.80	0.54	25.50	0.59	47.33	1.08
	SI2	0.60	2.80	0.54	25.50	0.59	47.33	1.08
Fitzroy	RC1	2.03	1.30	1.06	165.92	4.10	238.33	5.44
	WH1	2.16	1.17	1.19	243.67	5.84	314.67	7.18
	GK1	0.52	1.93	0.43	33.25	0.80	33.33	0.76
	GK2	0.58	1.87	0.49	49.83	1.19	48.67	1.11
	GH1	0.80	1.57	0.69	97.33	2.27	84.67	1.93
	GH2	0.80	1.57	0.69	91.58	2.15	84.67	1.93
Burnett–Mary	RD1	0.56	1.48	0.56	66.58	1.62	69.17	1.58
	RD2	0.63	1.41	0.63	93.17	2.31	96.50	2.20
	UG1	0.70	1.41	0.70	142.83	3.20	122.83	2.80
	UG2	0.64	1.47	0.64	101.83	2.23	90.83	2.07

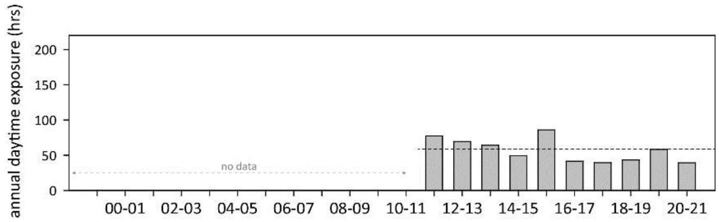

Figure 92. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of reef intertidal seagrass meadows at Archer Point, Cape York NRM region; 2011–2021. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 20. Observed tidal heights courtesy Maritime Safety Queensland, 2021. NB: Meadow heights have not yet been determined in the far northern Cape York sites.

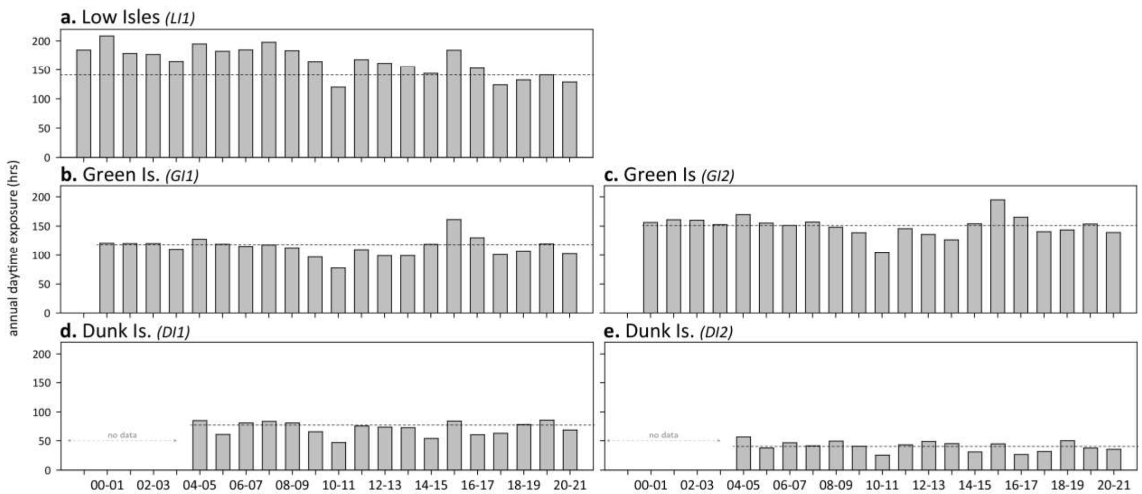

Figure 93. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of reef intertidal seagrass meadows in the Wet Tropics NRM region; 1999–2021. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 20. Observed tidal heights courtesy Maritime Safety Queensland, 2021.

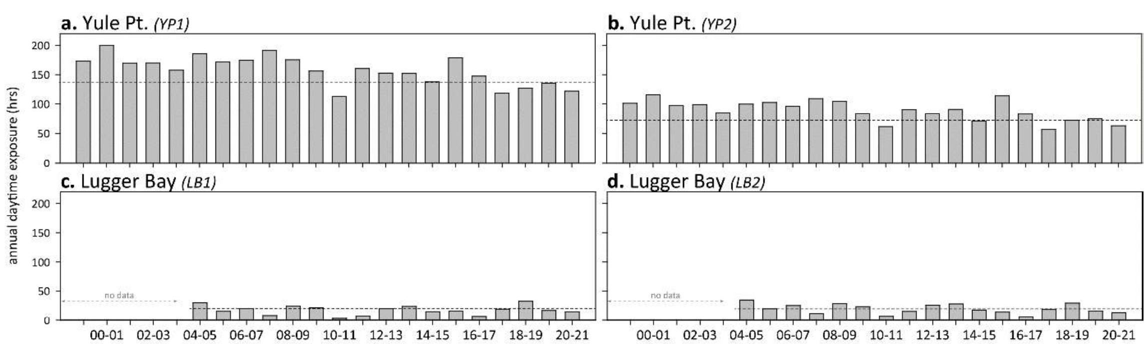

Figure 94. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of coastal intertidal seagrass meadows in Wet Tropics NRM region; 1999–2021. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 20. Observed tidal heights courtesy Maritime Safety Queensland, 2021.

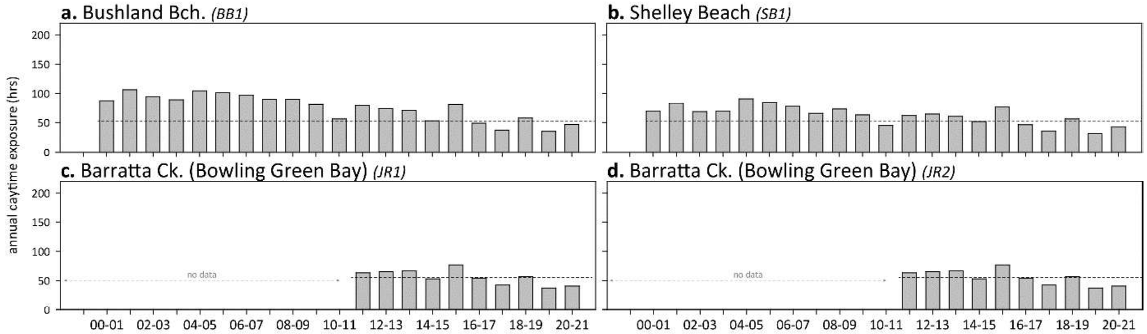

Figure 95. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of coastal intertidal seagrass meadows in Burdekin NRM region; 2000–2021. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 20. Observed tidal heights courtesy Maritime Safety Queensland, 2021.

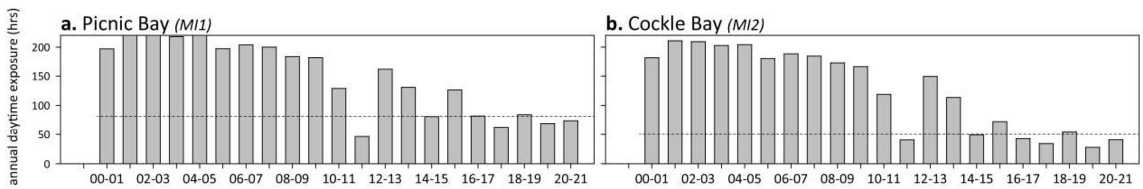

Figure 96. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of reef intertidal seagrass meadows in Burdekin NRM region; 2000–2021. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 20. Observed tidal heights courtesy Maritime Safety Queensland, 2021.

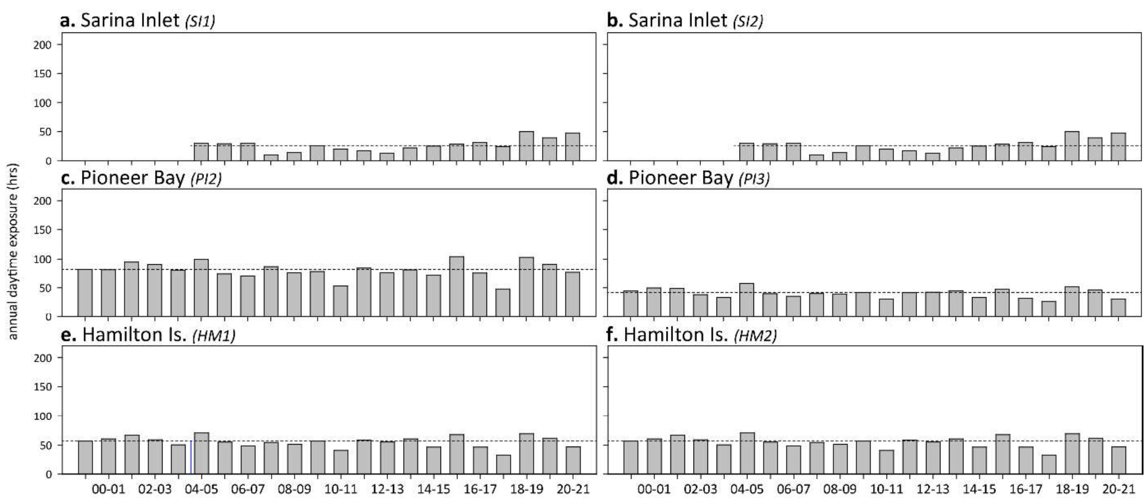

Figure 97. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of estuarine intertidal (a, b) coastal intertidal (c, d) and reef intertidal (e, f) seagrass meadows in Mackay–Whitsunday NRM region; 1999–2021. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 20. Observed tidal heights courtesy Maritime Safety Queensland, 2021.

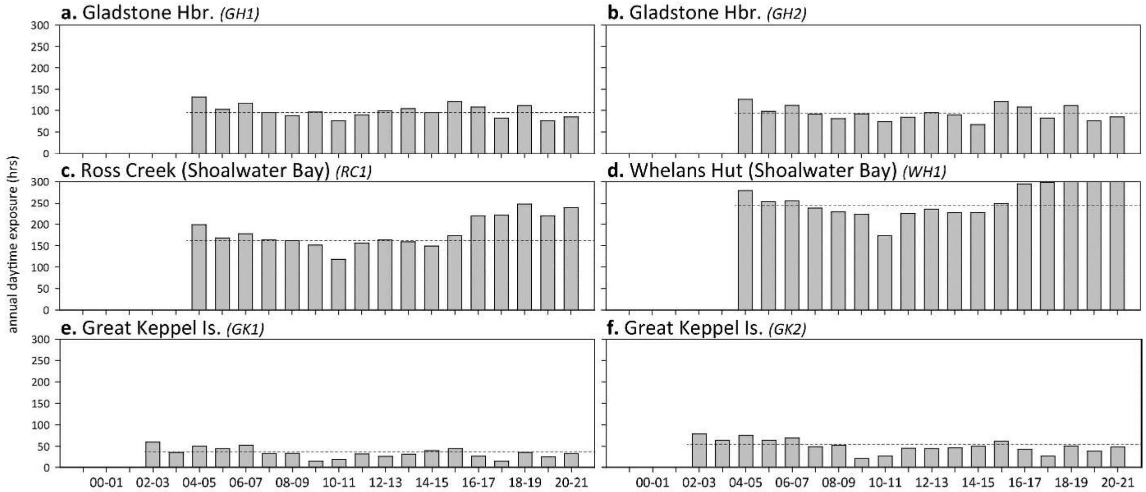

Figure 98. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of estuarine intertidal (a, b) coastal intertidal (c, d) and reef intertidal (e, f) seagrass meadows in the Fitzroy NRM region; 1999–2021. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 20. Observed tidal heights courtesy Maritime Safety Queensland, 2021.

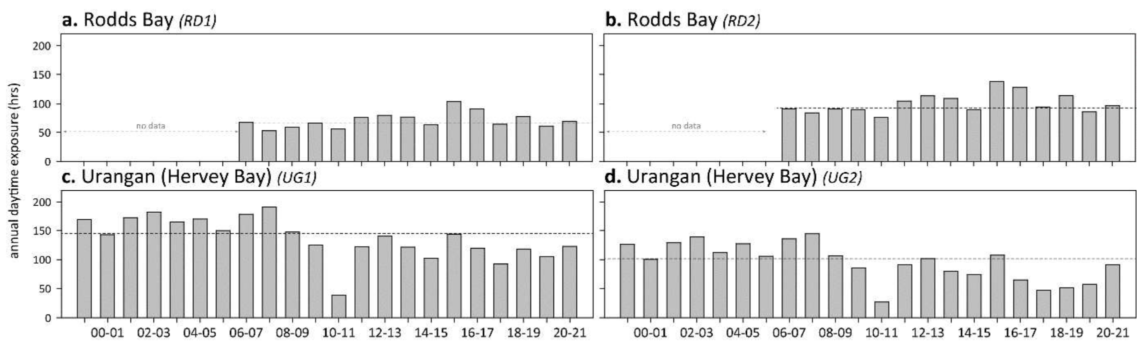

Figure 99. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of estuarine intertidal seagrass meadows in the Burnett–Mary NRM region; 1999–2021. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 20. Observed tidal heights courtesy Maritime Safety Queensland, 2021.

A3.1.2 Light at seagrass canopy

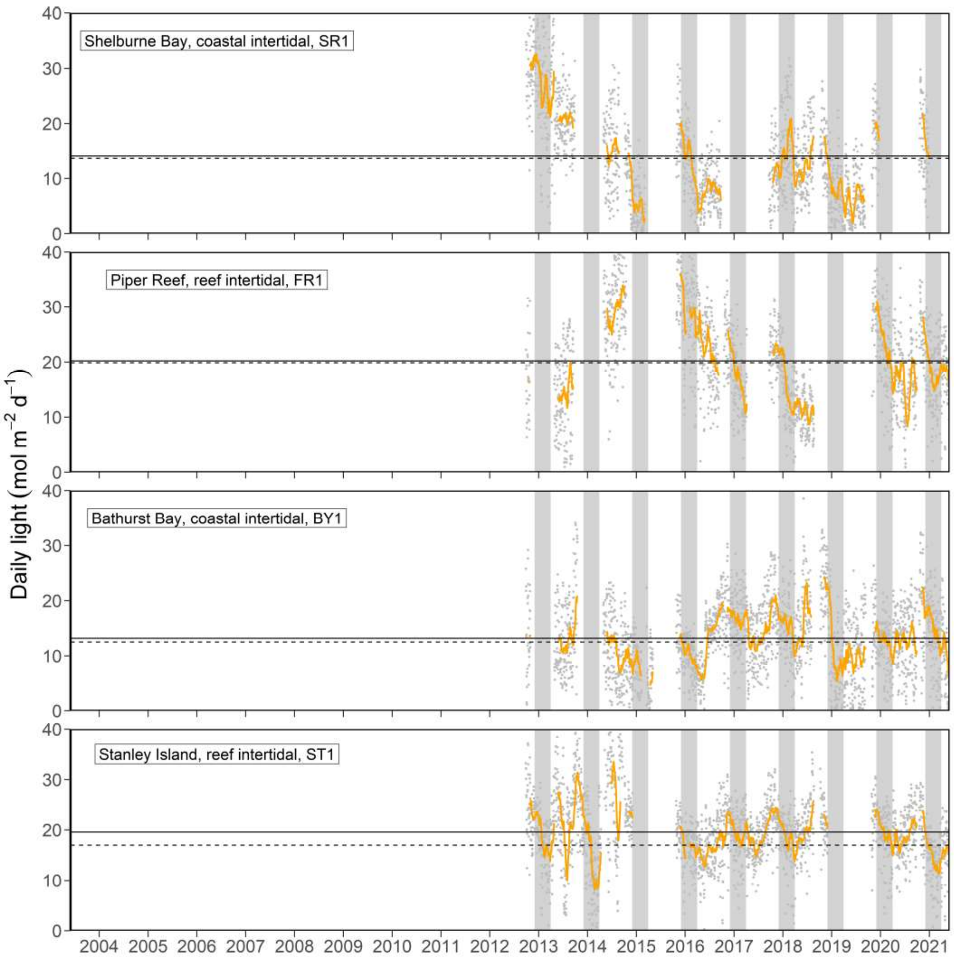

Figure 100. Daily light (yellow points) and 28-day rolling average (orange, bold line) at monitoring locations in the Cape York NRM region.

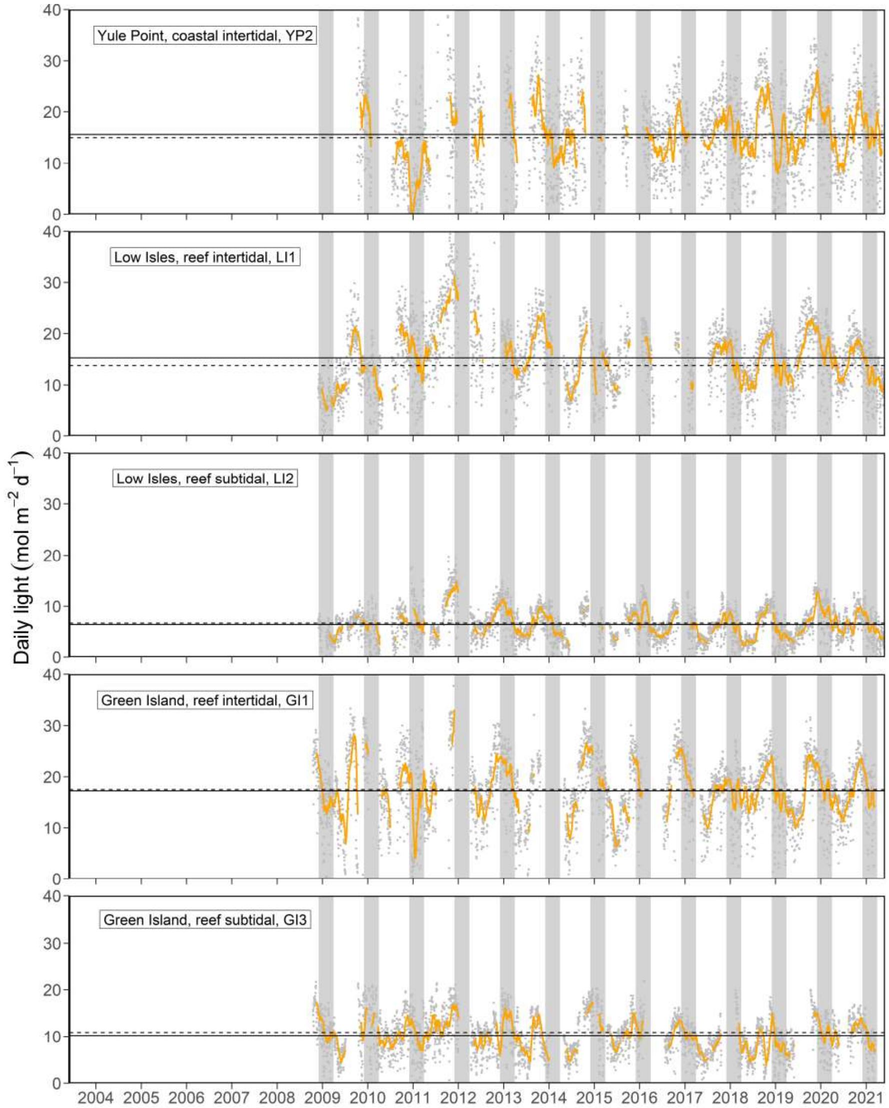

Figure 101. Daily light (yellow line) and 28-day rolling average (orange, bold line) at monitoring locations in the northern Wet Tropics.

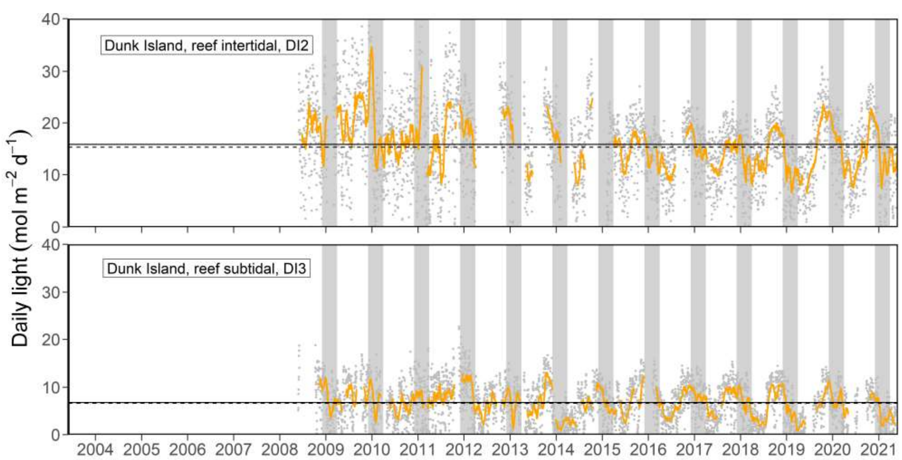

Figure 102. Daily light (yellow line) and 28-day rolling average (orange, bold line) at monitoring locations in the southern Wet Tropics.

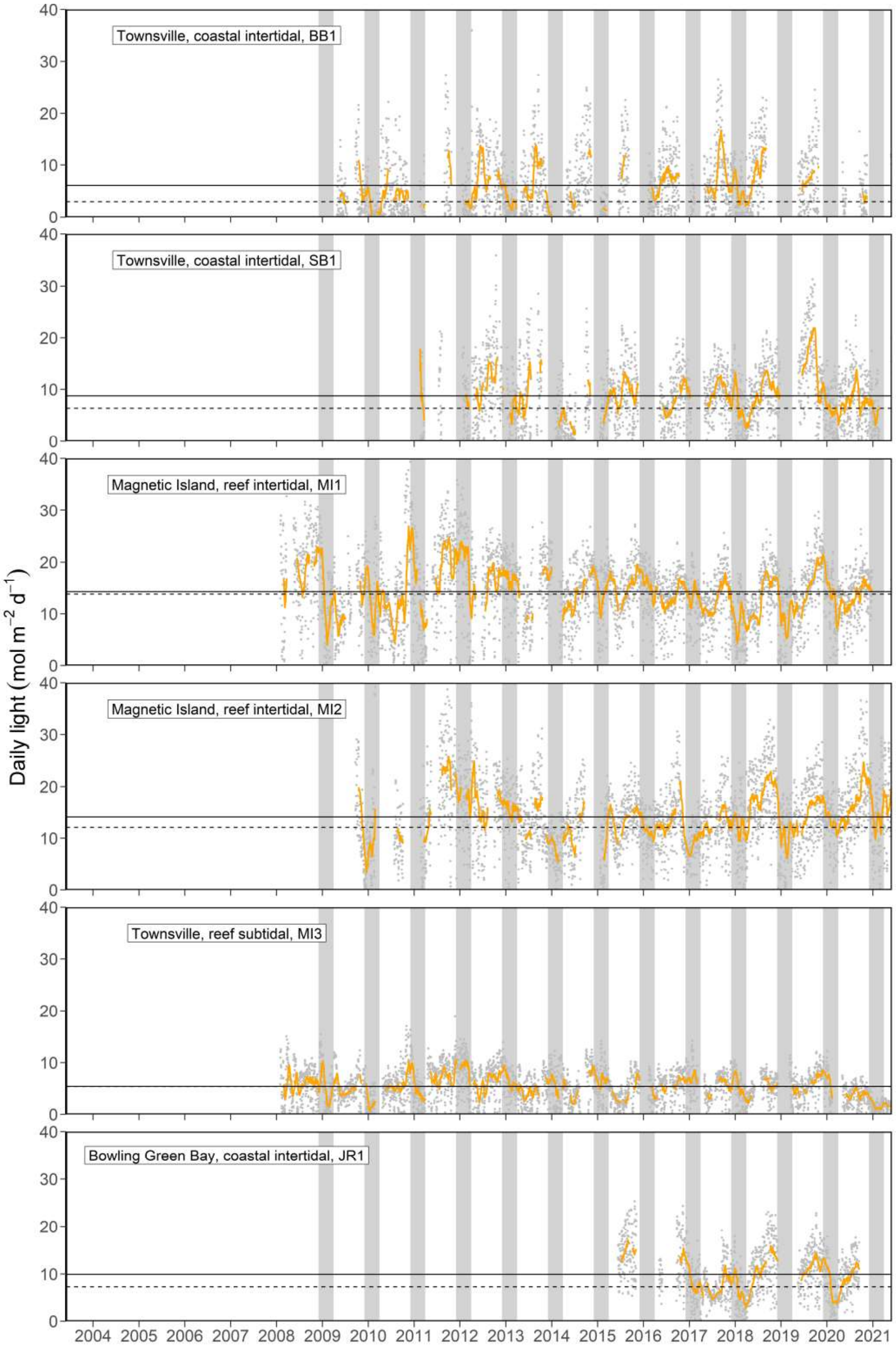

Figure 103. Daily light (yellow line) and 28-day rolling average (orange, bold line) at monitoring locations in the Burdekin region.

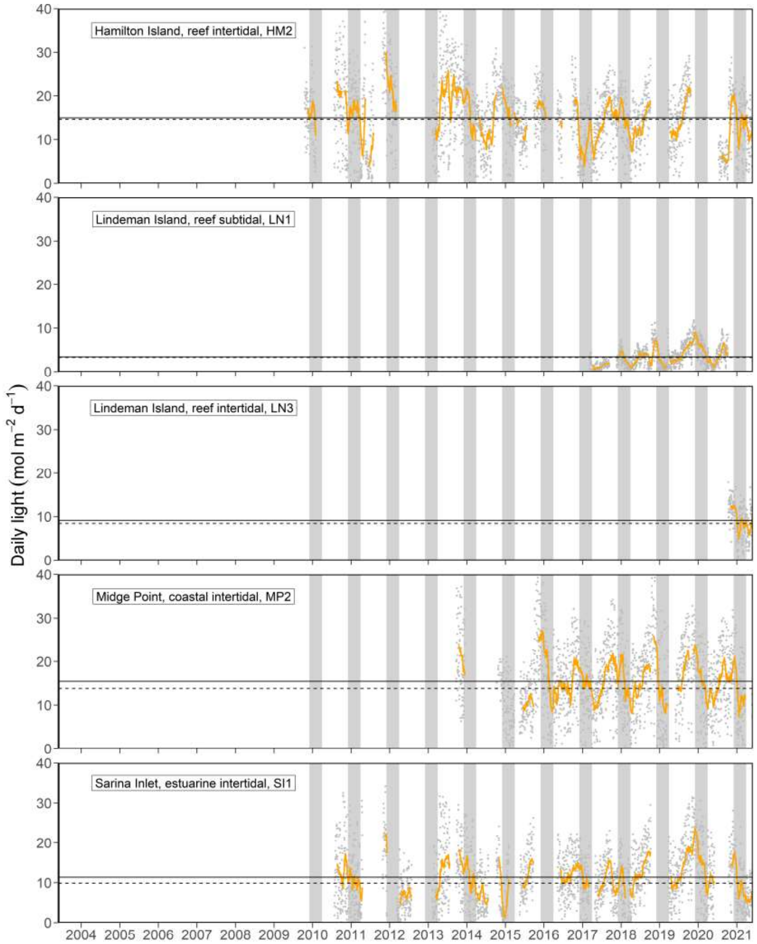

Figure 104. Daily light (yellow line) and 28-day rolling average (orange, bold line) at monitoring locations in the Mackay–Whitsunday NRM region.

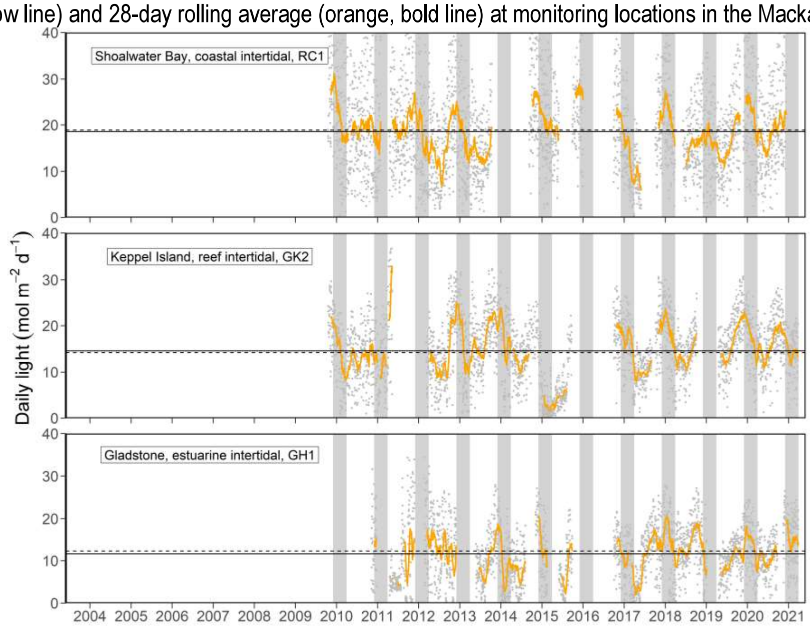

Figure 105. Daily light (yellow line) and 28-day rolling average (orange, bold line) at monitoring locations in the Fitzroy NRM region.

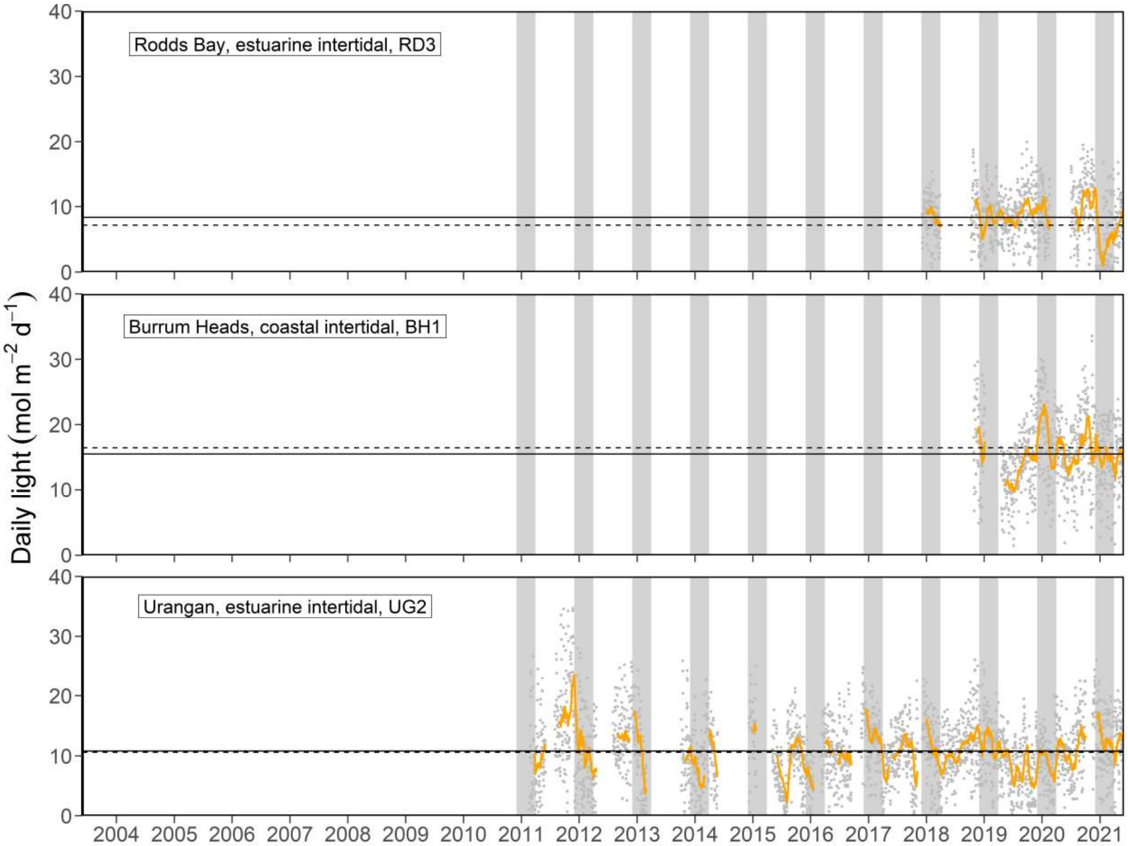

Figure 106. Daily light (yellow line) and 28-day rolling average (orange, bold line) at monitoring locations in the Burnett–Mary NRM region.

A3.2 Seagrass habitat condition: Sediments composition

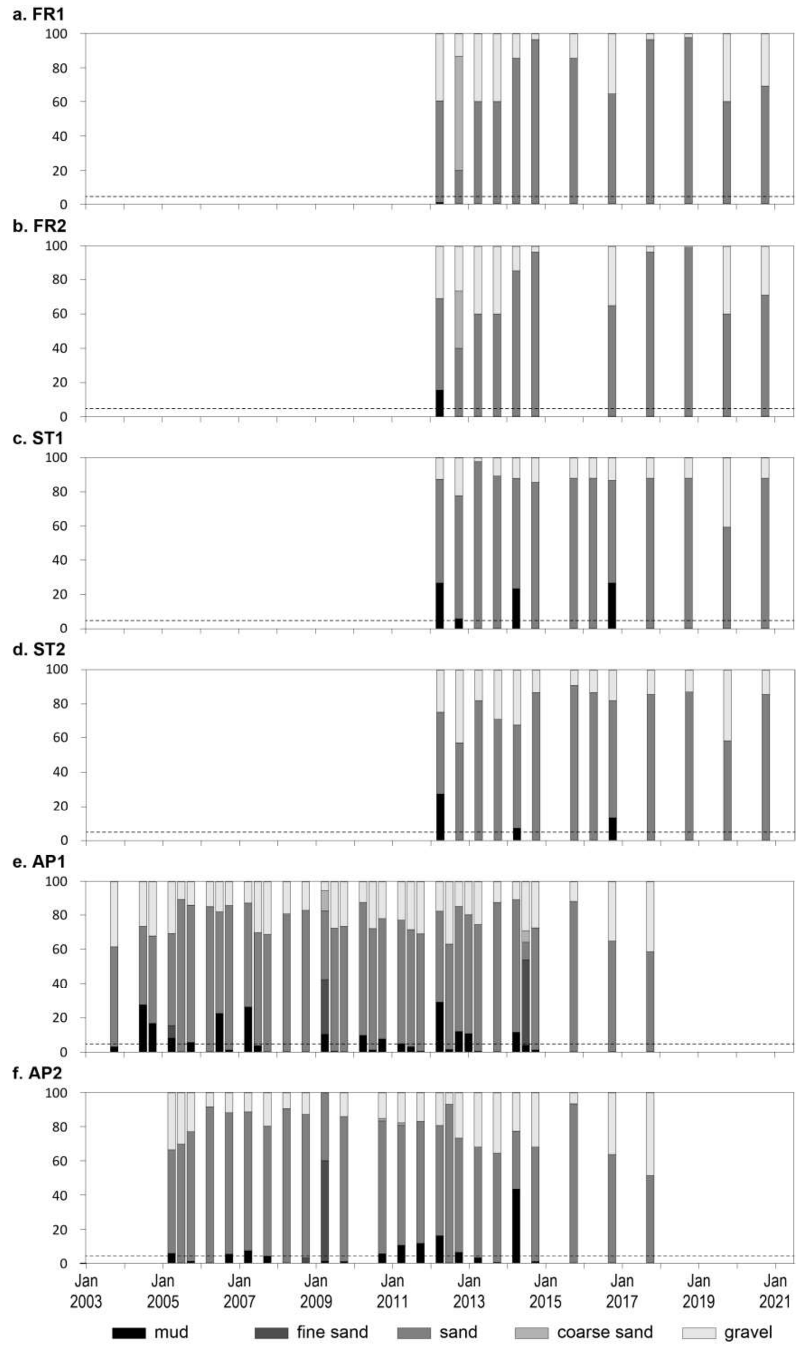

Figure 107. Sediment grain size composition at reef habitat monitoring sites in the Cape York region, 2003–2021. Dashed line is the Reef long-term average proportion of mud.

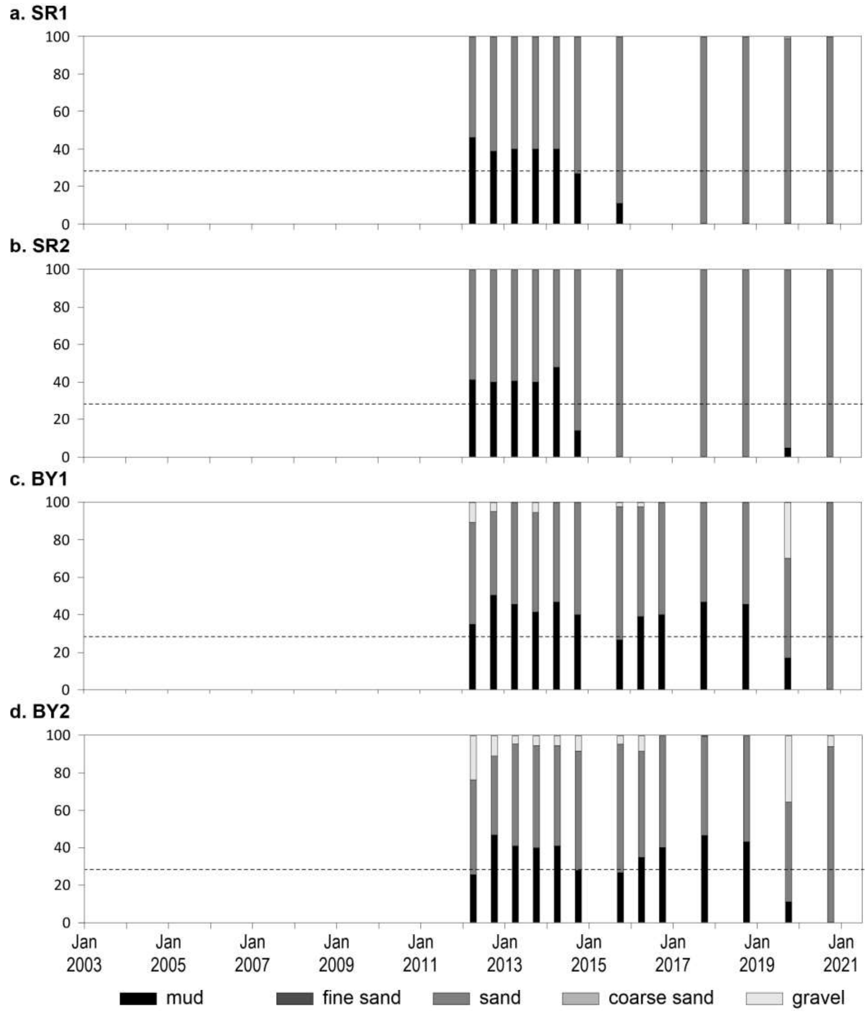

Figure 108. Sediment grain size composition at coastal habitat monitoring sites in the Cape York region, 2012—2021. Dashed line is the Reef long-term average proportion of mud.

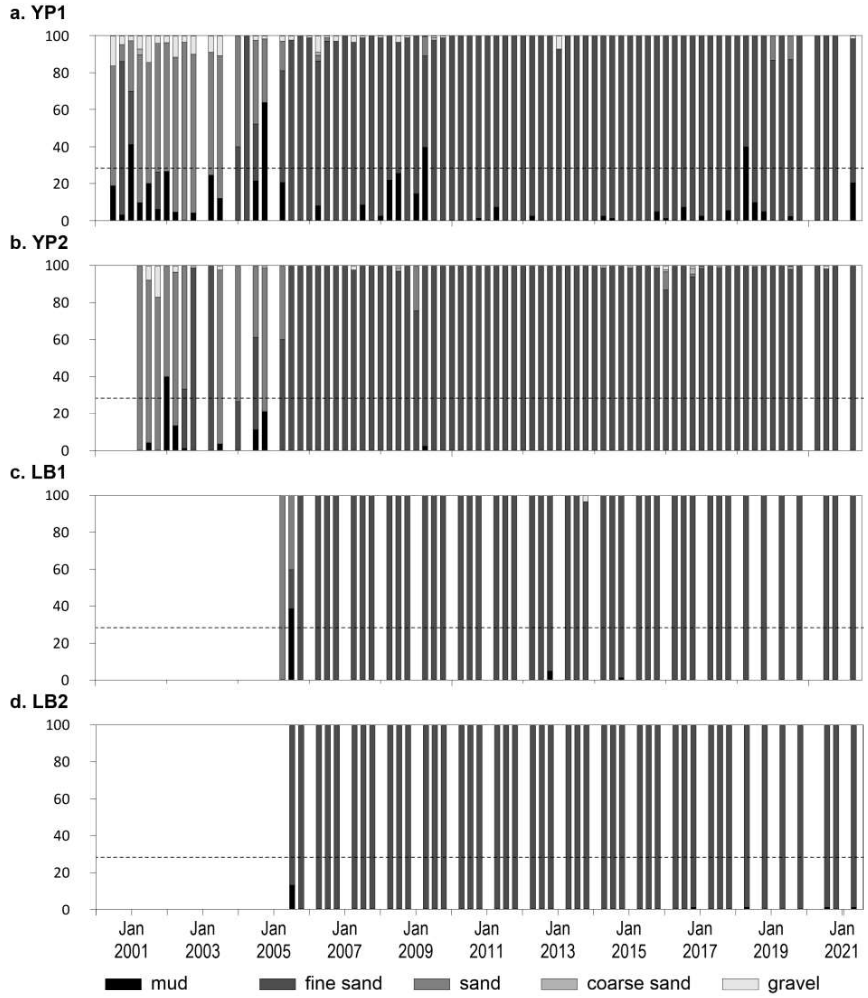

Figure 109. Sediment grain size composition at coastal intertidal habitat monitoring sites in the Wet Tropics region, 2001–2021. Dashed line is the Reef long-term average proportion of mud.

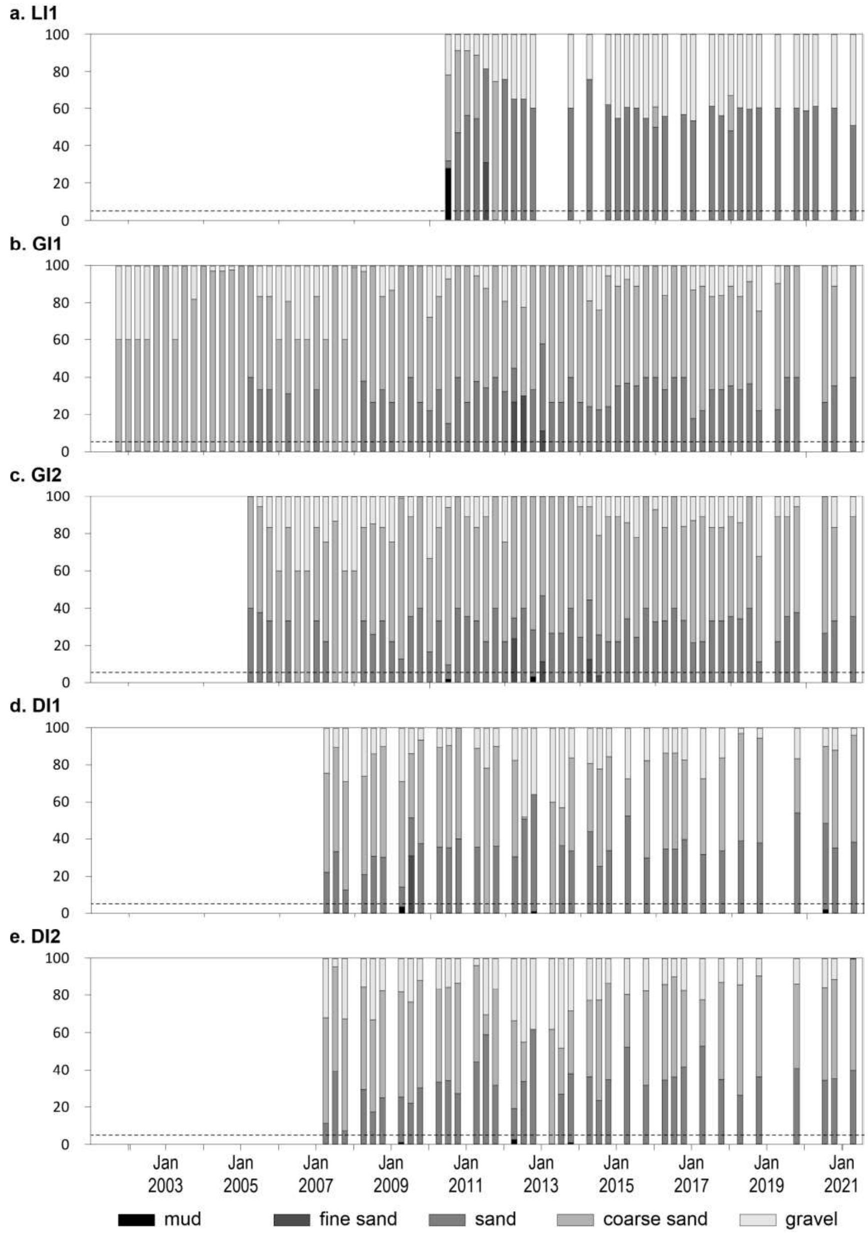

Figure 110. Sediment grain size composition at intertidal reef habitat monitoring sites in the Wet Tropics region, 2001–2021. Dashed line is the Reef long-term average proportion of mud.

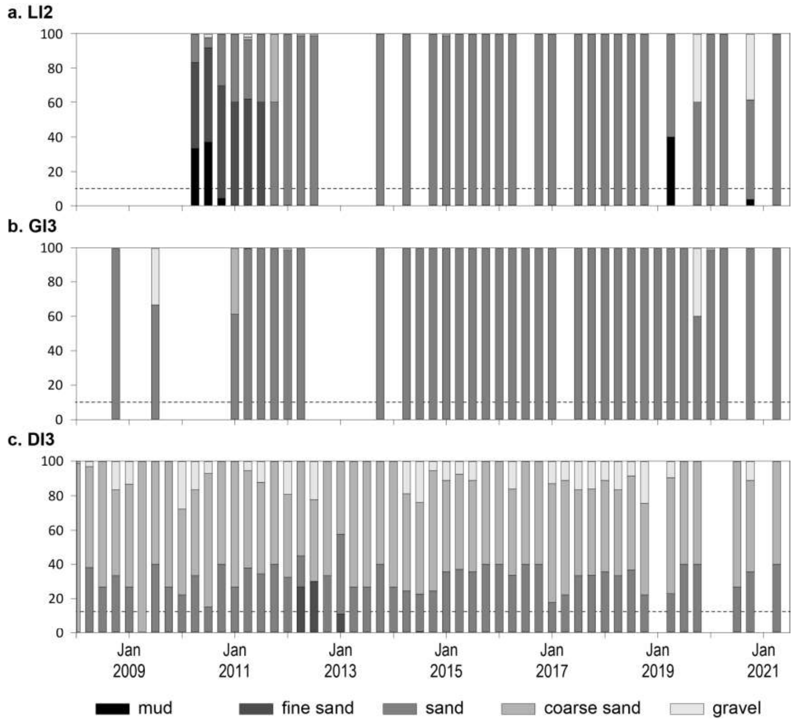

Figure 111. Sediment grain size composition at subtidal reef habitat monitoring sites in the Wet Tropics region, 2008–2021. Dashed line is the Reef long-term average proportion of mud.

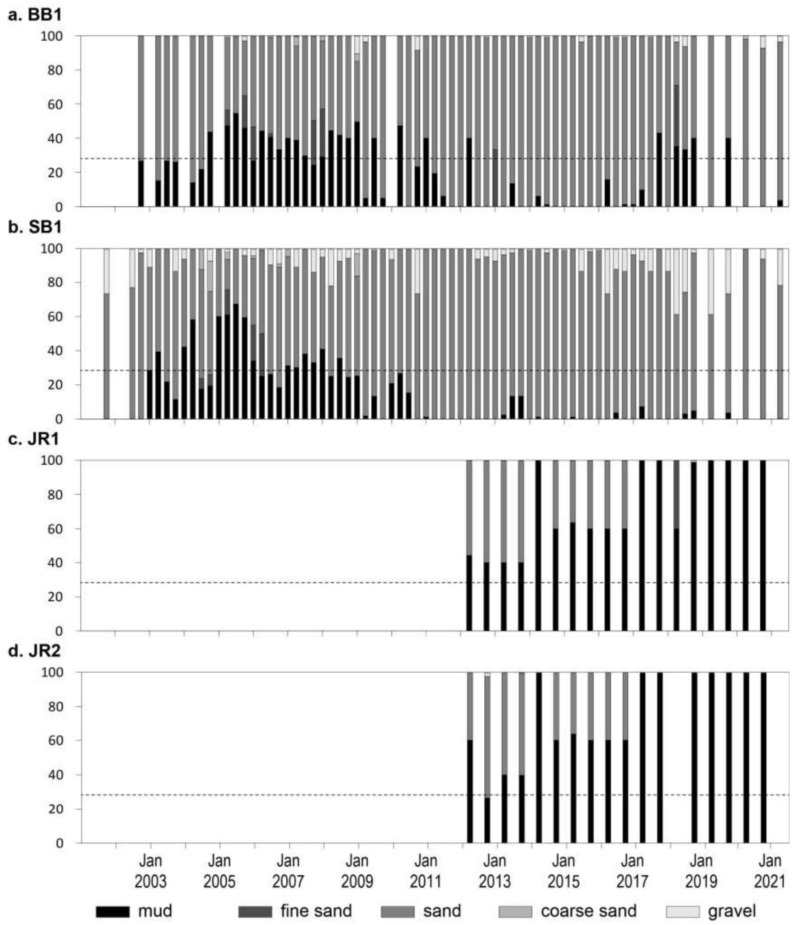

Figure 112. Sediment grain size composition at coastal intertidal habitat monitoring sites in the Burdekin region, 2001–2021. Dashed line is the Reef long-term average proportion of mud.

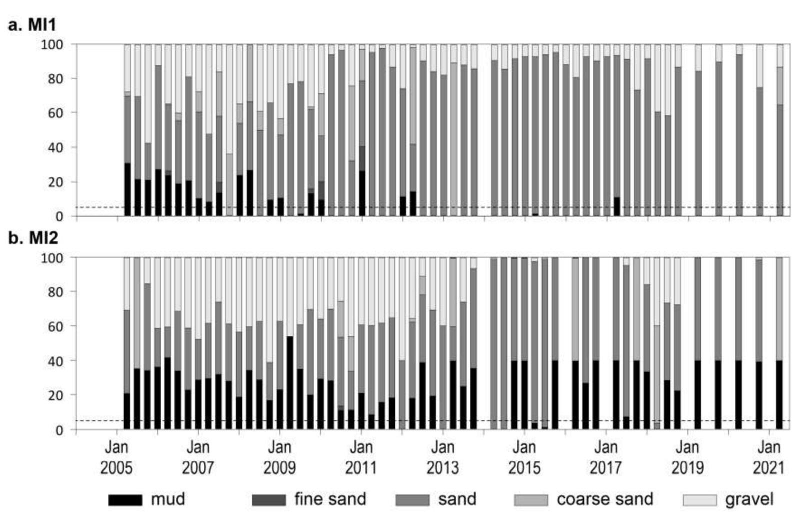

Figure 113. Sediment grain size composition at intertidal reef habitat monitoring sites in the Burdekin region, 2004–2021. Dashed line is the Reef long-term average proportion of mud.

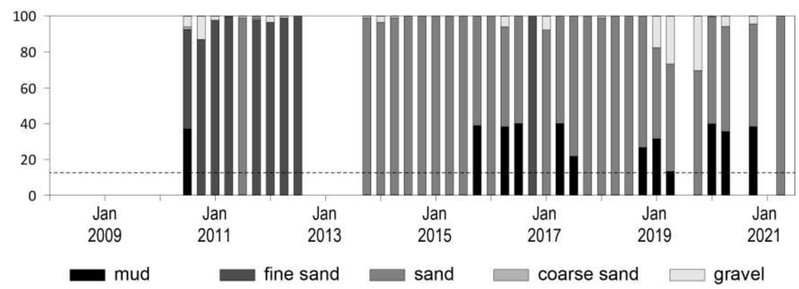

Figure 114. Sediment grain size composition at subtidal reef habitat monitoring sites in the Burdekin region, 2010–2021. Dashed line is the Reef long-term average proportion of mud.

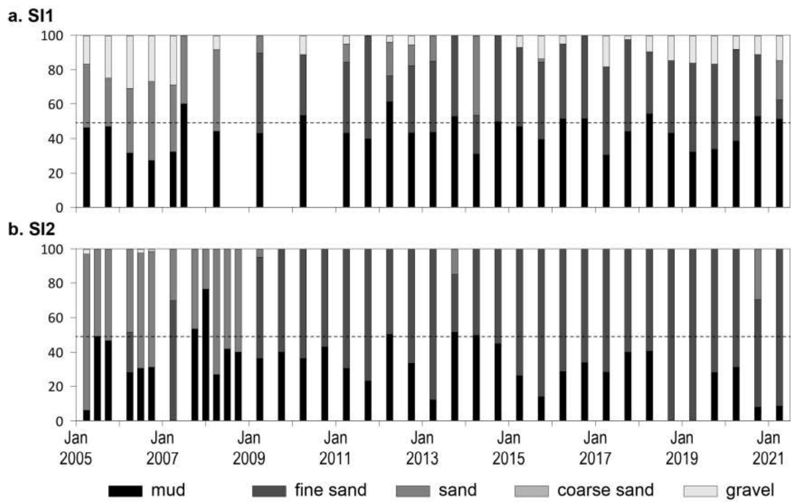

Figure 115. Sediment grain size composition at intertidal estuarine habitat monitoring sites in the Mackay–Whitsunday region, 2005–2021. Dashed line is the Reef long-term average proportion of mud.

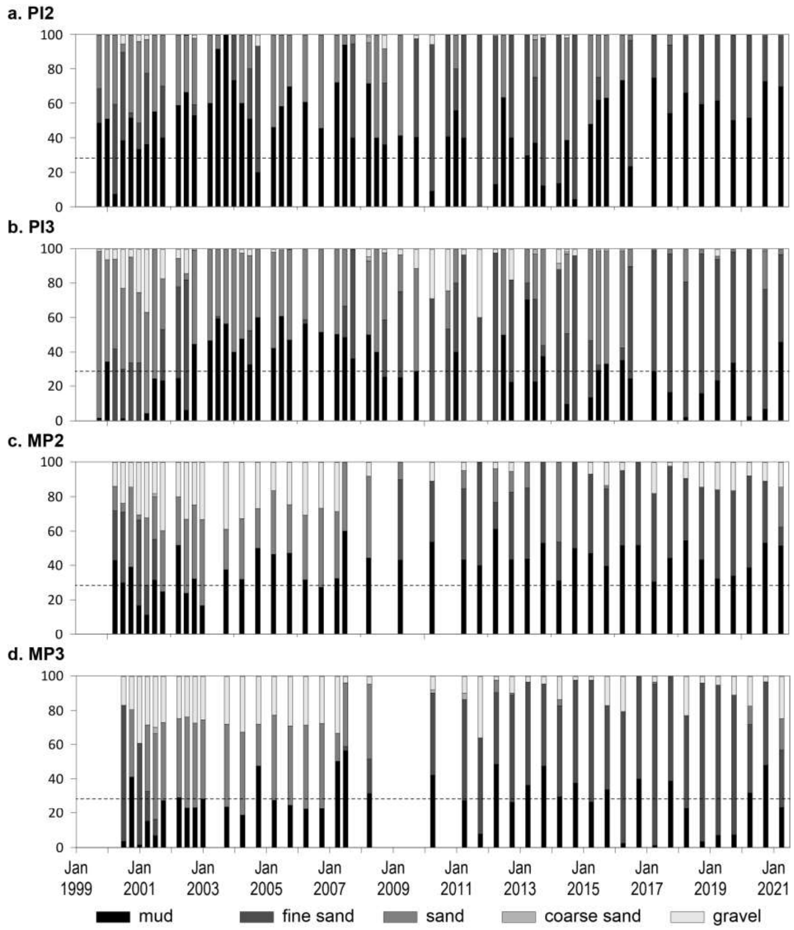

Figure 116. Sediment grain size composition at coastal intertidal habitat monitoring sites in the Mackay–Whitsunday region, 1999–2021. Dashed line is the Reef long-term average proportion of mud.

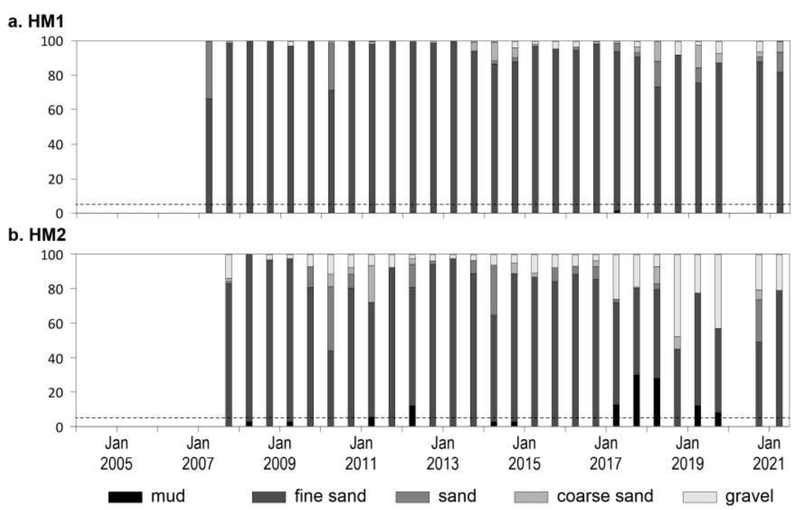

Figure 117. Sediment grain size composition at reef intertidal habitat monitoring sites in the Mackay–Whitsunday region, 2007–2021. Dashed line is the Reef long-term average proportion of mud.

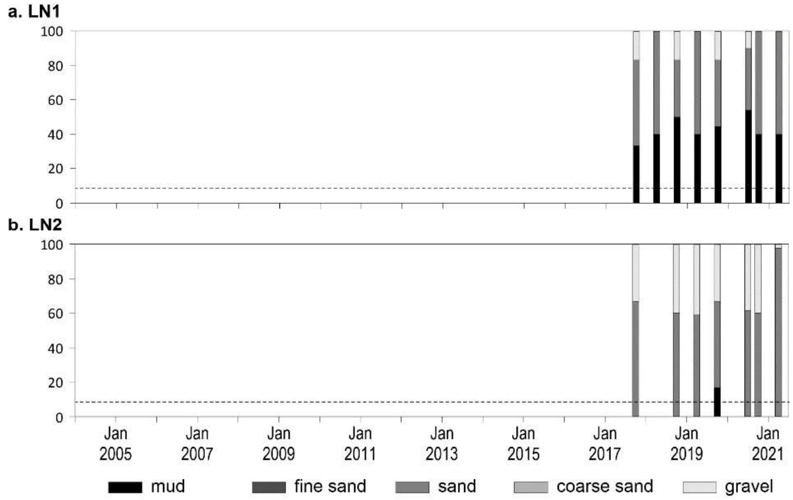

Figure 118. Sediment grain size composition at reef subtidal habitat monitoring sites in the Mackay–Whitsunday region, 2017–2021. Dashed line is the Reef long-term average proportion of mud.

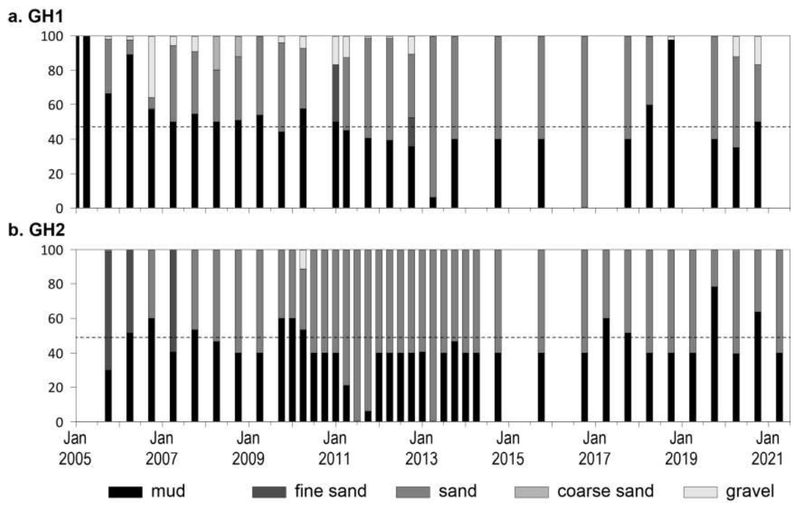

Figure 119. Sediment grain size composition at estuarine intertidal habitat monitoring sites in the Fitzroy region, 2005–2021. Dashed line is the Reef long-term average proportion of mud.

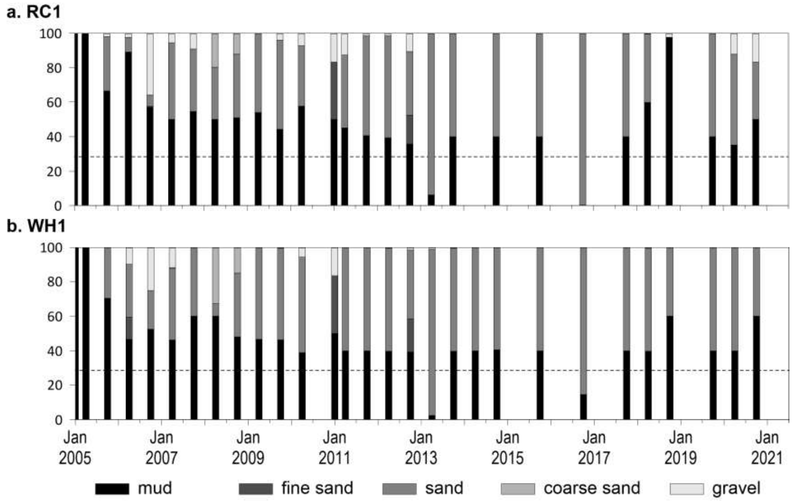

Figure 120. Sediment grain size composition at coastal intertidal habitat monitoring sites in the Fitzroy region, 2005–2021. Dashed line is the Reef long-term average proportion of mud.

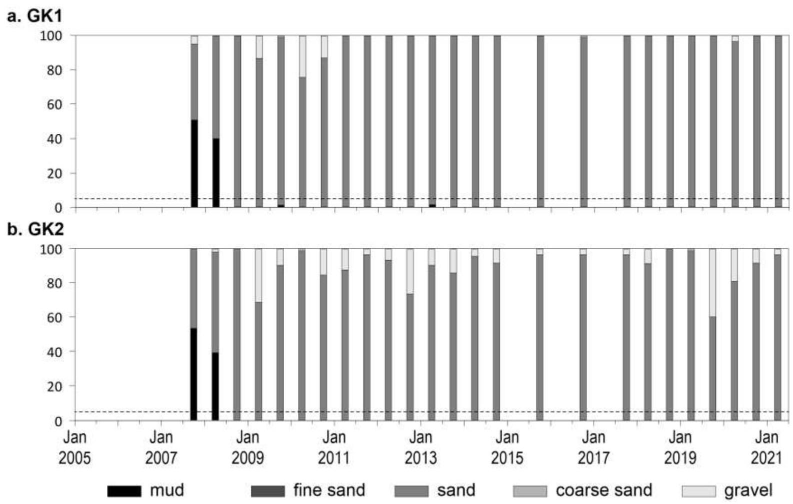

Figure 121. Sediment grain size composition at reef intertidal habitat monitoring sites in the Fitzroy region, 2007–2021. Dashed line is the Reef long-term average proportion of mud.

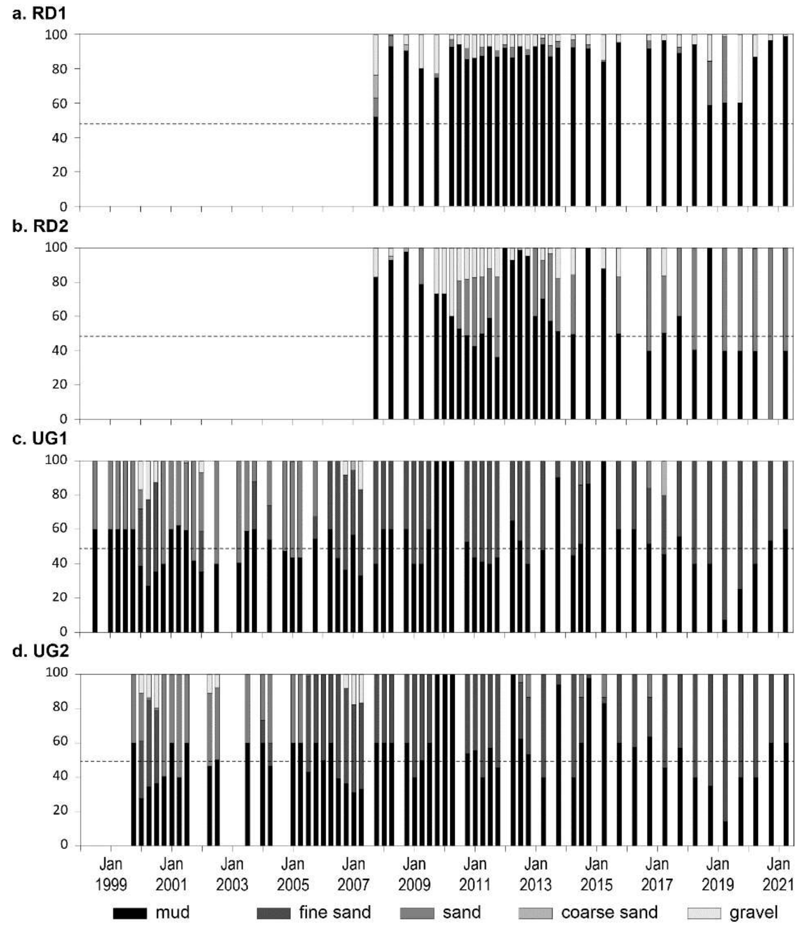

Figure 122. Sediment grain size composition at estuarine intertidal habitat monitoring sites in the Burnett–Mary region, 1999–2021. Dashed line is the Reef long-term average proportion of mud.

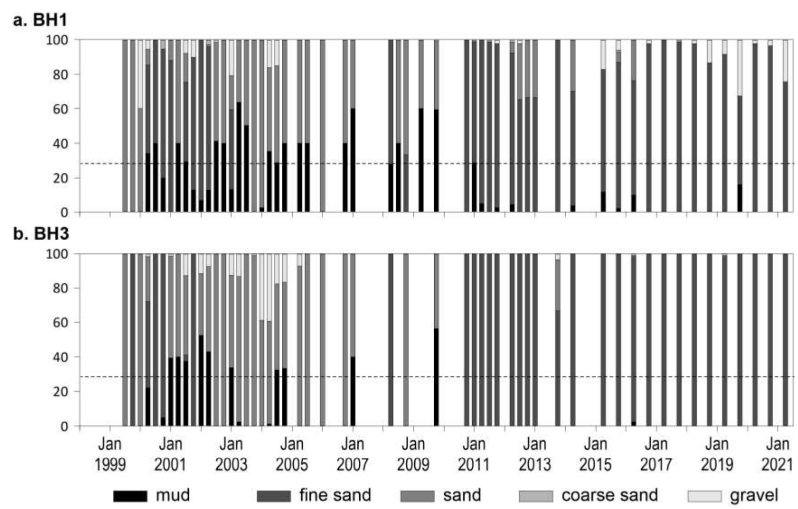

Figure 123. Sediment grain size composition at coastal intertidal habitat monitoring sites in the Burnett–Mary region, 1999–2021. Dashed line is the Reef long-term average proportion of mud.

Appendix 4 Results of statistical analysis

Table 21. Results of Mann-Kendall analysis to assess for 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), two-sided p-value (significant at α = 0.05 in bold), the Sen's slope (showing sign and strength of trend –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	2020	13	0	1	0.082	no trend
		BY2	2012	2020	13	0.051	0.85	0.132	no trend
		SR1	2012	2020	11	-0.455	0.061	-0.697	no trend
		SR2	2012	2020	11	-0.127	0.640	-0.152	no trend
	coastal subtidal	BY4	2017	2020	3	0.333	1	1.641	no trend
		LR1	2015	2020	5	0.2	0.806	2.196	no trend
		LR2	2015	2020	5	-0.4	0.462	-7.078	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	2020	12	-0.107	0.68	-0.08	no trend
		FR2	2012	2020	11	-0.418	0.087	-1.246	no trend
		ST1	2012	2020	13	**0.692**	**0.001**	**0.814** (0.408 to 1.212)	**increase**
		ST2	2012	2020	13	**0.761**	**<0.001**	**0.814** (0.515 to 1.144)	**increase**
		YY1	2012	2014	3	0.333	1.0000	1.045	no trend
	reef subtidal	FG1	2016	2020	5	0.2	0.806	1.165	no trend
		FG2	2016	2020	5	0.2	0.806	1.793	no trend
	pooled		2003	2020	39	**-0.374**	**0.006**	**-0.265** (-0.401 to -0.088)	**decrease**
Wet Tropics	coastal intertidal	**LB1**	2005	2021	46	**-0.451**	**<0.001**	**-0.0297** (-0.0897 to -0.0005)	**decrease**
		LB2	2005	2021	45	**-0.259**	**0.002**	**-0.0255** (-0.068 to 0)	**decrease**
		YP1	2000	2021	78	**0.173**	**0.024**	**0.119** (0.017 to 0.220)	**increase**
		YP2	2001	2021	74	0.133	0.091	0.06	no trend
	coastal subtidal	MS1	2015	2020	3	0	1	-2.466	no trend
		MS2	2015	2020	4	0.6	0.22	3.690	no trend
	reef intertidal	DI1	2007	2021	37	-0.042	0.723	-0.022	no trend
		DI2	2007	2021	34	-0.114	0.350	-0.092	no trend

NRM region	Habitat	Site	First Year	Last Year	*n*	*Kendall-τ*	*p* (2-sided)	Sen's slope (confidence interval)	trend
		GI1	2001	2021	74	-0.114	0.149	-0.061	no trend
		GI2	2005	2021	60	-0.055	0.533	-0.045	no trend
		GO1	2008	2016	7	-0.429	0.2296	-1.682	no trend
		LI1	2008	2021	41	**-0.327**	**0.002**	**-0.116** (-0.206 to -0.050)	**decrease**
	reef subtidal	DI3	2008	2021	48	-0.035	0.730	-0.004	no trend
		GI3	2008	2021	44	**-0.343**	**<0.001**	**-0.452** (-0.698 to -0.215)	**decrease**
		LI2	2008	2021	41	0.184	0.084	0.108	no trend
	pooled		2000	2021	88	-0.140	0.052	-0.06	no trend
Burdekin	coastal intertidal	BB1	2002	2021	66	0.031	0.715	0.025	no trend
		SB1	2001	2021	72	-0.072	0.371	-0.042	no trend
		SB2	2001	2021	71	**-0.178**	**0.029**	**-0.149** (-0.300 to -0.016)	**decrease**
		JR1	2012	2020	18	0.203	0.256	1.167	no trend
		JR2	2012	2020	17	**0.368**	**0.044**	**2.006** (0.003 to 3.486)	**increase**
		BW1	2019	2020	3	-1	0.96	-0.445	no trend
		BW2	2019	2021	4	-0.667	0.308	-3.439	no trend
	reef intertidal	MI1	2005	2021	59	-0.080	0.374	-0.1	no trend
		MI2	2005	2021	57	**-0.237**	**0.009**	**-0.426** (-0.727 to -0.106)	**decrease**
	reef subtidal	MI3	2008	2021	49	-0.011	0.920	-0.492	no trend
	pooled		2001	2021	79	-0.058	0.45	-0.048	no trend
Mackay Whitsunday	estuarine intertidal	**SI1**	2005	2021	37	**-0.309**	**0.007**	**-0.287** (-0.600 to -0.066)	**decrease**
		SI2	2005	2021	32	0.052	0.685	0.031	no trend
	coastal intertidal	**MP2**	2000	2021	44	**0.304**	**0.004**	**0.240** (0.085 to 0.378)	**increase**
		MP3	2000	2021	42	0.129	0.233	0.069	no trend
		PI2	1999	2021	60	**-0.301**	**0.001**	**-0.265** (-0.421 to -0.124)	**decrease**
		PI3	1999	2021	60	-0.164	0.064	-0.113	no trend
		CV1	2017	2020	7	-0.048	1	-0.004	no trend
		CV2	2017	2020	7	-0.143	0.764	-0.076	no trend

NRM region	Habitat	Site	First Year	Last Year	*n*	*Kendall-τ*	*p* (2-sided)	Sen's slope (confidence interval)	trend
		SH1	2017	2020	8	0.429	0.174	2.355	no trend
	coastal subtidal	NB1	2015	2020	6	-0.467	0.260	-5.858	no trend
		NB2	2015	2020	6	0.067	1	0.893	no trend
	reef intertidal	**HB1**	2000	2021	46	**-0.206**	**0.045**	**-0.122** (-0.238 to -0.003)	**decrease**
		HB2	2000	2021	45	-0.006	0.961	-0.006	no trend
		HM1	2007	2021	28	**-0.418**	**0.002**	**-0.204** (-0.370 to -0.065)	**decrease**
		HM2	2007	2021	27	**-0.448**	**0.001**	**-0.141** (-0.282 to -0.054)	**decrease**
	Reef subtidal	TO1	2015	2020	6	-0.067	1	0.750	no trend
		TO2	2015	2020	6	-0.067	1	0.013	no trend
		LN1	2017	2021	8	0.214	0.536	0.776	no trend
		LN2	2017	2020	6	0.333	0.452	0.313	no trend
	pooled		1999	2021	70	**-0.382**	**<0.001**	**-0.161** (-0.221 to -0.099)	**decrease**
Fitzroy	estuarine intertidal	**GH1**	2005	2021	39	**-0.441**	**<0.001**	**-0.697** (-1.031 to -0.376)	**decrease**
		GH2	2005	2021	39	-0.100	0.377	-0.182	no trend
	coastal intertidal	RC1	2002	2020	37	-0.068	0.555	-0.102	no trend
		WH1	2002	2020	38	0.082	0.468	0.074	no trend
	reef intertidal	**GK1**	2007	2021	25	**-0.491**	**<0.001**	**-0.102** (-1.66 to -0.049)	**decrease**
		GK2	2007	2021	25	-0.077	0.607	-0.016	no trend
	pooled		2002	2021	51	**-0.369**	**<0.001**	**-0.215** (-0.321 to -0.115)	**decrease**
Burnett Mary	estuarine intertidal	RD1	2007	2021	34	0.103	0.406	0.006	no trend
		RD2	2007	2017	28	**-0.409**	**0.003**	**-0.009** (-0.096 to -0.001)	**decrease**
		RD3	2017	2021	8	-0.429	0.174	-0.740	no trend
		UG1	1998	2021	65	0.083	0.335	0.002	no trend
		UG2	1999	2021	59	**0.191**	**0.031**	**0.029** (0.0001 to 0.137)	**increase**
	coastal intertidal	BH1	1999	2021	56	0.136	0.140	0.069	no trend
		BH3	1999	2021	54	**0.391**	**<0.001**	**0.166** (0.101 to 0.229)	**increase**
	pooled		1998	2021	78	0.024	0.76	0.008	no trend

Table 22. Resilience score and resilience score category for each site in 2020–21.

Region	Site	Habitat	Score	Score category
Cape York	BY1	coastal intertidal	58	2.1.2
	BY2	coastal intertidal	13	1.1
	FR1	reef intertidal	15	1.1
	FR2	reef intertidal	68	2.1.2
	SR1	coastal intertidal	9	1.1
	SR2	coastal intertidal	9	1.1
	ST1	reef intertidal	100	2.2.2
	ST2	reef intertidal	50	2.1.1
Wet Tropics	GI1	reef intertidal	70	2.1.2
	GI2	reef intertidal	68	2.1.2
	GI3	reef subtidal	87	2.2.2
	LI1	reef intertidal	5	1.1
	LI2	reef subtidal	0	1.1
	YP1	coastal intertidal	73	2.2.1
	YP2	coastal intertidal	76	2.2.1
	DI1	reef intertidal	32	2.1.1
	DI2	reef intertidal	56	2.1.2
	DI3	reef subtidal	30	2.1.1
	LB1	coastal intertidal	15	1.1
	LB2	coastal intertidal	30	2.1.1
Burdekin	BB1	coastal intertidal	70	2.2.1
	JR1	coastal intertidal	70	2.2.1
	JR2	coastal intertidal	70	2.2.1
	MI1	reef intertidal	50	2.1.2
	MI2	reef intertidal	70	2.1.2
	MI3	reef subtidal	15	1.1
	SB1	coastal intertidal	78	2.2.1
Mackay–Whitsunday	HM1	reef intertidal	30	2.1.1
	HM2	reef intertidal	0	1.1
	LN1	reef subtidal	50	2.1.2
	LN2	reef subtidal	30	2.1.1
	LN3	reef intertidal	30	2.1.1
	MP2	coastal intertidal	70	2.2.1
	MP3	coastal intertidal	80	2.2.1
	SI1	estuarine intertidal	7	1.1
	SI2	estuarine intertidal	50	2.1.2
Fitzroy	GH1	estuarine intertidal	30	1.2
	GH2	estuarine intertidal	30	1.2
	GK1	reef intertidal	5	1.2
	GK2	reef intertidal	19	1.2
	RC1	coastal intertidal	50	2.1.2
	WH1	coastal intertidal	76	2.2.1
Burnett–Mary	BH1	coastal intertidal	94	2.2.1
	BH3	coastal intertidal	30	2.1.1
	RD1	estuarine intertidal	73	2.2.1
	RD3	estuarine intertidal	50	2.1.2
	UG1	estuarine intertidal	5	1.1
	UG2	estuarine intertidal	12	1.1

Table 23 Results of Generalised additive models (GAMs) fitted to Reef-level abundance with habitat and NRM region as a fixed effect.

MODELS - REEF	N	EDF	CHI-SQ	*P*-VALUE	R-SQ (ADJ)	DEVIANCE EXPLAINED
% cover = s(date)	87	20.39	5173	<2e-16	0.581	0.741
% cover = s(date) + Habitat	300				0.503	0.805
Coastal intertidal		18.451	922.96	<2e-16		
Coastal subtidal		1.949	13.36	0.0289		
Estuarine intertidal		20.647	1329.78	<2e-16		
Reef intertidal		13.405	924	<2e-16		
Reef subtidal		11.179	380.93	<2e-16		
% cover = s(date) + NRM	392				0.59	0.792
Cape York		5.066	51.52	<2e-16		
Wet Tropics		15.217	673.63	<2e-16		
Burdekin		17.625	1148.36	<2e-16		
Mackay Whitsunday		17.624	494.96	<2e-16		
Fitzroy		12.912	190.00	<2e-16		
Burnett Mary		20.611	1120.17	<2e-16		

Table 24 Results of Generalised additive models (GAMs) fitted to NRM region-level abundance with habitat, location or site as a fixed effect.

MODELS PER NRM REGIONS	N	EDF	CHI-SQ	*P*-VALUE	R-SQ (ADJ)	DEVIANCE EXPLAINED
Cape York						
% cover = s(date)	39	8.131	182	<2e-16	0.421	0.489
% cover = s(date) + Habitat	62				0.57	0.713
Coastal intertidal		2.805	33.428	9.65e-07		
Coastal subtidal		1.971	6.839	0.0193		
Reef intertidal		6.420	150.635	<2e-16		
Reef subtidal		1.984	3.265	0.1991		
% cover = s(date) + Location	100				0.645	0.775
Reef intertidal [AP]		6.730	108.134	<2e-16		
Coastal intertidal [BY]		2.646	25.445	9.96e-06		
Coastal subtidal [BY]		1.948	2.178	0.328		
Reef subtidal [FG]		1.978	3.138	0.213		
Coastal subtidal [LR]		1.964	12.317	0.00123		
Reef intertidal [FR]		1.479	2.609	0.147		
Coastal intertidal [SR]		1	1.840	0.175		
Reef intertidal [ST]		1	27.504	<2e-16		
Reef intertidal [YY]		1.652	0.869	0.6833		
% cover = s(date) + Site						
AP1	35	5.157	46.977	<2e-16	0.603	0.687
AP2	24	2.647	8.544	0.042	0.269	0.340
BY1	13	2.115	3.307	0.242	0.190	0.326
BY2	13	2.676	11.415	0.013	0.502	0.638
BY3	NA	NA	NA	NA	NA	NA
BY4	NA	NA	NA	NA	NA	NA
FG1	5	2.729	27.872	0.000	0.785	0.963
FG2	5	2.963	72.157	<2e-16	0.938	0.990
FR1	12	2.121	1.440	0.469	0.139	0.298
FR2	11	1.099	14.787	0.000	0.592	0.625
LR1	5	1.114	1.282	0.360	-0.218	0.246
LR2	5	2.470	7.552	0.055	-0.186	0.867
SR1	11	1.517	5.382	0.086	0.336	0.392
SR2	11	2.101	3.110	0.324	0.167	0.366
ST1	13	1.000	32.754	<2e-16	0.713	0.747
ST2	13	1.565	52.396	<2e-16	0.795	0.839
YY1	NA	NA	NA	NA	NA	NA
Northern Wet Tropics						
% cover = s(date)	83	14.85	349	<2e-16	0.343	0.504
% cover = s(date) + Habitat	203				0.706	0.757
Coastal intertidal		12.063	205.27	<2e-16		
Reef intertidal		10.489	221.23	<2e-16		
Reef subtidal		6.468	32.63	6.83e-05		
% cover = s(date) + Location	286				0.824	0.911
Reef intertidal [LI1]		2.854	26.30	3.7e-5		
Reef subtidal [LI2]		6.555	138.66	<2e-16		
Coastal intertidal [YP]		11.624	182.10	<2e-16		
Reef intertidal [GI]		5.545	48.22	<2e-16		
Reef subtidal [GI3]		4.680	55.51	<2e-16		
% cover = s(date) + Site						
GI1	75	3.262	11.225	0.023	0.127	0.168
GI2	61	4.612	23.769	0.0005	0.287	0.341
GI3	46	4.387	49.400	<2e-16	0.520	0.584
LI1	43	5.618	54.514	<2e-16	0.557	0.611
LI2	43	4.946	60.321	<2e-16	0.391	0.643
YP1	79	9.817	95.850	<2e-16	0.551	0.701
YP2	75	7.980	42.392	<0.0001	0.323	0.465
Southern Wet Tropics						
% cover = s(date)	60	13.82	1271	<2e-16	0.725	0.914
% cover = s(date) + Habitat	137				0.926	0.958

MODELS PER NRM REGIONS	N	EDF	CHI-SQ	*P*-VALUE	R-SQ (ADJ)	DEVIANCE EXPLAINED
Coastal intertidal		11.715	613.66	<2e-16		
Coastal subtidal		2.093	10.64	0.0307		
Reef intertidal		10.179	846.63	<2e-16		
Reef subtidal		10.731	206.43	<2e-16		
% cover = s(date) + Location	144				0.93	0.988
Coastal intertidal [LB]		11.745	1099.31	<2e-16		
Reef intertidal [DI]		11.034	518.43	<2e-16		
Reef subtidal [DI3]		11.11	277.60	<2e-16		
Reef intertidal [GO]		5.431	178.42	<2e-16		
Coastal subtidal [MS]		1.641	5.33	0.0643		
% cover = s(date) + Site						
DI1	37	9.285	267.602	<2e-16	0.929	0.967
DI2	37	8.718	226.614	<2e-16	0.830	0.960
DI3	49	10.232	249.566	<2e-16	0.733	0.961
GO1	7	2.943	42.166	<2e-16	0.923	0.905
LB1	46	9.874	549.743	<2e-16	0.905	0.989
LB2	45	8.091	245.259	<2e-16	0.771	0.952
MS1	4	1.000	0.209	0.6477	-0.478	0.071
MS2	5	1.014	2.665	0.1088	0.278	0.367
Burdekin						
% cover = s(date)	77	17.73	1596	<2e-16	0.777	0.908
% cover = s(date) + Habitat	184				0.776	0.908
Coastal intertidal		17.24	703.5	<2e-16		
Reef intertidal		12.41	396.8	<2e-16		
Reef subtidal		10.6	396	<2e-16		
% cover = s(date) + Location	206				0.743	0.894
Coastal intertidal [JR]		6.689	159.4	<2e-16		
Coastal intertidal [TSV]		16.709	485.112	<2e-16		
Coastal intertidal [BW]		1.384	0.252	0.886		
Reef intertidal [MI]		11.847	318.997	<2e-16		
Reef subtidal [MI3]		10.127	327.349	<2e-16		
% cover = s(date) + Site						
BB1	66	13.048	222.090	<2e-16	0.736	0.945
BW1	NA	NA	NA	NA	NA	NA
BW2	NA	NA	NA	NA	NA	NA
BW3	NA	NA	NA	NA	NA	NA
JR1	18	2.349	6.894	0.078	0.245	0.391
JR2	17	2.813	15.632	0.002	0.452	0.640
MI1	59	9.928	188.531	<2e-16	0.776	0.869
MI2	57	10.058	143.714	<2e-16	0.728	0.845
MI3	50	8.926	240.598	<2e-16	0.846	0.928
SB1	72	15.237	205.256	<2e-16	0.716	0.916
Mackay Whitsunday						
% cover = s(date)	70	18.2	777.1	<2e-16	0.496	0.70
% cover = s(date) + Habitat	169				0.678	0.822
Coastal intertidal		17.058	278.03	<2e-16		
Coastal subtidal		4.665	42.51	6.81e-07		
Estuarine intertidal		14.671	244.80	<2e-16		
Reef intertidal		7.224	159.61	<2e-16		
Reef subtidal		3.709	16.61	0.00176		
% cover = s(date) + Location	251				0.763	0.918
Coastal intertidal [CV]		1.001	0.218	0.639		
Coastal intertidal [MP]		7.814	33.428	0.000235		
Coastal intertidal [PI]		17.3	285.076	<2e-16		
Coastal subtidal [NB]		4.681	44.671	0.2990		
Reef subtidal [LN]		1.581	2.044	0.288		
Reef intertidal [LN3]		0	0	1		
Reef intertidal [HM]		4.411	48.661	<2e-16		
Estuarine intertidal [SI]		14.70	303.129	<2e-16		
Reef intertidal [HB]		7.964	77.009	<2e-16		

MODELS PER NRM REGIONS	N	EDF	CHI-SQ	*P*-VALUE	R-SQ (ADJ)	DEVIANCE EXPLAINED
Reef subtidal [TO]		4.619	61.387	8.63e-06		
Coastal intertidal [SH1]		2.444	15.376	0.0007		
% cover = s(date) + Site						
CV1	7	1.000	0.001	0.9805	-0.200	0.002
CV2	7	1.000	0.896	0.3438	0.006	0.149
HB1	46	6.094	49.903	<2e-16	0.502	0.655
HB2	45	8.863	89.899	<2e-16	0.688	0.776
HM1	28	1.364	15.284	0.0012	0.328	0.342
HM2	27	4.515	56.588	<2e-16	0.413	0.838
LN1	8	1.000	0.490	0.4842	-0.168	0.084
LN2	6	1.283	2.403	0.2825	-0.050	0.422
LN3	NA	NA	NA	NA	NA	NA
MP2	44	1.645	9.579	0.0090	0.189	0.197
MP3	42	1.000	0.557	0.4556	-0.008	0.014
NB1	6	1.000	4.483	0.0342	0.314	0.575
NB2	6	3.457	19.577	0.0005	0.670	0.983
PI2	60	7.069	43.615	0.0000	0.348	0.584
PI3	60	10.747	67.028	<2e-16	0.485	0.696
SH1	8	1.863	11.138	0.0026	0.703	0.697
SI1	37	8.445	50.754	0.0000	0.411	0.765
SI2	32	4.374	9.147	0.1555	0.051	0.379
TO1	6	2.519	9.287	0.0278	0.000	0.823
TO2	6	3.163	70.530	<2e-16	0.998	0.991
Fitzroy						
% cover = s(date)	50	6.876	145.9	<2e-16	0.307	0.526
% cover = s(date) + Habitat	102				0.783	0.916
Coastal intertidal		8.316	111.409	<2e-16		
Estuarine intertidal		14.034	186.633	<2e-16		
Reef intertidal		1	6.356	0.0117		
% cover = s(date) + Location	102				0.783	0.916
Coastal intertidal [SWB]		8.316	111.371	<2e-16		
Reef intertidal [GK]		1	6.375	0.0116		
Estuarine intertidal [GH]		14.033	186.562	<2e-16		
% cover = s(date) + Site						
GH1	39	5.686	70.684	<2e-16	0.536	0.836
GH2	39	3.062	17.033	0.0020	0.128	0.411
GK1	25	1.000	16.702	0.0000	0.145	0.479
GK2	25	1.001	0.316	0.5749	-0.023	0.012
RC1	37	7.648	73.812	<2e-16	0.684	0.753
WH1	38	7.644	88.127	<2e-16	0.707	0.780
Burnett Mary						
% cover = s(date)	73	19.73	584.3	<2e-16	0.475	0.735
% cover = s(date) + Habitat	125				0.429	0.684
Coastal intertidal		4.851	31.23	1.71e-05		
Estuarine intertidal		16.981	406.38	<2e-16		
% cover = s(date) + Location	156				0.578	0.891
Estuarine intertidal [RD]		7.068	193.93	<2e-16		
Estuarine intertidal [UG]		18.3	621.09	<2e-16		
Coastal intertidal [BH]		5.118	37.45	<2e-16		
% cover = s(date) + Site						
BH1	56	5.719	41.663	0.0000	0.420	0.509
BH3	54	4.943	39.438	0.0000	0.381	0.521
RD1	34	5.181	16.757	0.0133	0.336	0.435
RD2	28	3.794	52.461	<2e-16	0.550	0.755
RD3	8	1.000	1.826	0.1766	0.051	0.232
UG1	61	11.084	154.081	<2e-16	0.535	0.883
UG2	59	9.934	119.678	<2e-16	0.534	0.845

Table 25. Results of Generalised additive models (GAMs) fitted to habitat-level abundance with NRM region as a fixed effect

MODELS PER HABITAT	N	EDF	CHI-SQ	*P*-VALUE	R-SQ (ADJ)	DEVIANCE EXPLAINED
Estuarine Intertidal						
% cover = s(date) + NRM	145				0.421	0.792
Burnett Mary		8.697	387.71	<2e-16		
Fitzroy		3.175	37.53	5.91e-07		
Mackay Whitsunday		6.488	51.40	<2e-16		
Coastal Intertidal						
% cover = s(date) + NRM	326				0.577	0.765
Burdekin		8.583	485.50	<2e-16		
Burnett Mary		5.740	72.72	<2e-16		
Cape York		2.402	10.27	0.0412		
Fitzroy		6.578	77.47	<2e-16		
Mackay Whitsunday		8.496	153.76	<2e-16		
Wet Tropics		8.499	276.50	<2e-16		
Reef Intertidal						
% cover = s(date) + NRM	251				0.758	0.848
Burdekin		7.268	433.365	<2e-16		
Cape York		3.666	55.483	<2e-16		
Fitzroy		1.001	6.026	0.0141		
Mackay Whitsunday		5.993	132.706	<2e-16		
Wet Tropics		7.293	544.769	<2e-16		
Reef Subtidal						
% cover = s(date) + NRM	115				0.795	0.806
Burdekin		8.285	318.337	<2e-16		
Cape York		2.908	9.120	0.0163		
Mackay Whitsunday		2.856	4.962	0.11		
Wet Tropics		7.179	53.175	<2e-16		
Coastal Subtidal						
% cover = s(date) + NRM	16				0.223	0.831
Cape York		2.651	21.888	6.87e-05		
Mackay Whitsunday		3.796	28.487	1.01e-05		
Wet Tropics		1	1.741	0.187		