

# GREAT BARRIER REEF MARINE MONITORING PROGRAM

## Inshore seagrass monitoring Annual Report 2024–25



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

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

TropWATER- Centre for Tropical Water and Aquatic Ecosystem Research

James Cook University

Townsville, Qld 4811

[Tropwater@jcu.edu.au](mailto:Tropwater@jcu.edu.au)

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## Acronyms, abbreviations and units

Authority	Great Barrier Reef Marine Park Authority
BoM	Bureau of Meteorology
CV	coefficient of variation
GAMM	generalised additive mixed effect model
GBRWHA	Great Barrier Reef World Heritage Area
GNSS	Global Navigation Satellite System
ha	hectare
JCU	James Cook University
km	kilometre
m	metre
MMP	Great Barrier Reef Marine Monitoring Program
MSL	mean sea level
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
RJFMP	Reef Joint Field Management Program
PRM	Pesticide Risk Metric
SE	Standard Error
Seagrass Index	seagrass condition index
SW	Seagrass-Watch
TropWATER	Centre for Tropical Water & Aquatic Ecosystem Research
WT	Reef Water Type 1–4 (e.g., WT1)
WQ water year	1 October to 30 September (e.g., 2010–11)

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## 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 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 declined in overall condition in 2024–25, with a grade change in the Seagrass Index from moderate to **‘poor’** (Figure 1). Seagrass condition in the far northern region (Cape York) declined to **‘poor’**, whereas condition in the 2 southern most regions (Fitzroy and Burnett–Mary) improved from ‘poor’ to **‘moderate’** and **‘good’**, respectively. Condition in the northern (Wet Tropics and Burdekin) and central (Mackay–Whitsunday) regions remained largely unchanged (but with declines in score) in **‘poor’** and **‘moderate’** states, respectively.

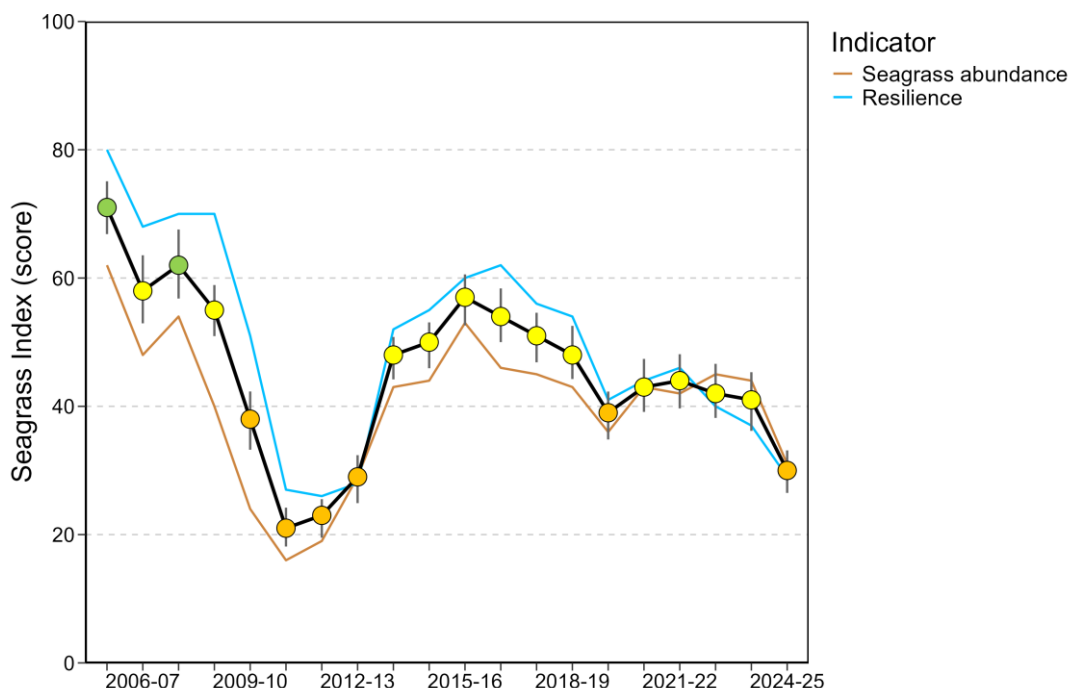


Figure 1. Overall inshore Reef Seagrass Index (±SE) with contributing indicator scores over the life of the MMP (Great Barrier Reef Marine Monitoring Program). The Index is weighted on inshore seagrass area and derived from the aggregate of metric scores for indicators of seagrass condition: abundance 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 declined in 2024–25, resulting in a grade change from ‘moderate’ to ‘poor’, following only a marginal decline in score in the previous period. Reef-wide, around 57 per cent of the long-term monitoring sites saw deterioration in abundances in 2024–25. Abundances generally decreased in the northern regions, including Cape York, Wet Tropics, Burdekin, and Mackay–Whitsunday. Specifically, Cape York declined to the lowest levels since monitoring commenced, driven by deterioration at all habitats, particularly in the south of the region. Conversely, abundances in the Fitzroy and Burnett–Mary regions in the very southern Reef, improved to some of the highest levels since the MMP commenced. The increase can be attributed to improved conditions in estuarine and coastal seagrass habitats, while the status of reef seagrass habitats has

remained unchanged. Long-term monitoring data on Reef-wide seagrass abundance indicate periods of loss and recovery within specific regions. Over the last decade, the decrease in seagrass abundance from 2015 to 2019 was largely attributed to losses in the Mackay–Whitsunday and Burdekin regions, while smaller declines were simultaneously observed in Cape York and the Wet Tropics. From 2019 to 2023, the losses in the northernmost regions subsided, with the Burdekin showing significant improvement as it recovered from the effects of heavy rainfall and above-average river discharge in early 2019. Despite these positive trends, seagrass abundance continued to decline in all 3 southern regions from 2020. Although the declines abated in Mackay-Whitsunday, they persisted in the Fitzroy and Burnett-Mary regions, significantly impacted by extreme flooding in early 2022. Since 2023, as the southern regions have experienced significant recovery with more favourable conditions for seagrass growth, cyclones and associated flooding in the north have resulted in considerable deterioration in the northern regions, including the current period.

The Resilience score continued to decline in 2024–25, mirroring the overall Index and reaching its lowest level in more than a decade. Long-term trends in resilience closely follow those of abundance at the Reef scale, with major declines during 2009–12 associated with extreme weather, recovery to a good state in 2016–17, and a predominantly declining trajectory since. In 2024–25, resilience declined to poor in Cape York, the Wet Tropics and the Burdekin, remained stable in the Mackay–Whitsunday region, and improved in the Fitzroy and Burnett–Mary regions to ‘moderate’ and ‘good’, respectively. Most declines were driven by coastal habitats where low cover and reduced reproductive effort weakened recovery potential, although estuarine intertidal habitats (assessed only in the southern 3 regions) showed substantial improvements. As in previous years, reef intertidal habitat resilience was constrained by low or absent reproductive structures and was mostly limited to colonising species. Regional declines reflected abundances falling below resistance thresholds in Cape York, the northern and southern Wet Tropics, and the Burdekin, alongside a persistently high proportion of colonising species in the Burdekin. Reproductive effort of foundational species remained low or absent across Cape York, the northern and southern Wet Tropics and the Burdekin—even at typically reproductive coastal sites.

### **Influencing pressures**

Pressures affecting inshore Reef seagrass habitats varied across different regions and habitats during 2024–25. The 2024–25 wet season was characterised by an absence of significant cyclone activity across the inshore and broader Reef.

Rainfall during the 2024–25 wet season exceeded the 30-year long-term average in the Wet Tropics, Burdekin, Mackay–Whitsunday, and Burnett–Mary regions, while remaining near average in Cape York and Fitzroy. River discharge to the inshore Reef was 1.5 to 2 times above the long-term median in 2024–25, the wettest year since 2010–11, following 3 consecutive wet years. Discharges were exceptionally high in the Burdekin, Mackay–Whitsunday, and Burnett–Mary regions (6.8, 2.8, and 3.4 times the long-term medians), with most other basins also recording flows well above historical averages.

Turbid sediment laden and algae enriched waters dominated inshore conditions across the Reef during the 2024–25 wet season, extending further offshore than average in the Wet Tropics, Burdekin, and Mackay–Whitsunday regions. Exposure of seagrass monitoring sites to turbid waters during the wet season was around average. At the Reef scale, the total area of seagrass exposed to water quality risk categories remained close to the long-term average. Regionally, there were small shifts toward higher-risk categories in the Burdekin and Mackay–Whitsunday regions, while the Fitzroy region recorded a modest increase in exposure associated with lower-risk categories.

Daily average benthic light availability in 2024–25 was lower than the long-term average (2008–2024) across most Reef inshore seagrass meadows. Substantial reductions in light were observed in regions affected by elevated river discharge during 2024–25, as well as in the southernmost region, where elevated discharge events in preceding years have likely

contributed to ongoing light limitation. Most monitoring sites recorded below-average daily light, with the largest negative anomalies occurring at Dunk Island, Magnetic Island, Midge Point, Burrum Heads and Urangan. Light conditions in the Burdekin region did not fall below average; however, this reflects that measurements in 2024–25 were collected only from reef intertidal habitats where light was lower than average for the site, but reef habitat light levels are higher than coastal habitats. In the Fitzroy region, data coverage was insufficient to calculate a reliable regional average. Overall, the 2024–25 season was characterised by reduced benthic light availability across much of the inshore Reef system, coinciding with widespread freshwater discharge and persistent turbidity that constrained light reaching seagrass canopies during key growth periods.

Within-canopy water temperatures represent a critical environmental pressures impacting inshore Reef seagrass meadows. During the 2024–25 period, seawater temperatures within the seagrass canopy were similar to the previous period, yet they remained above average for the fourth consecutive year, marking the sixth warmest on record. All regions, with the exception of the Burdekin, recorded annual average temperatures surpassing the long-term average. Notably, only the Wet Tropics and Fitzroy regions experienced at least one day of extreme heat (>40 °C). Interestingly, the Mackay–Whitsunday region, which typically sees the highest number of extreme temperature days, did not experience any for the first time in a decade. As a result, it is unlikely that high temperatures led to burning or mortality during this period.

Since the inception of the MMP, inshore seagrass meadows of the Reef have been subject to sustained and cumulative environmental pressures. Nearly every year, one or more regions have experienced impacts from cyclones, flooding, thermal stress events, or extended periods of very low light availability. The most severe and widespread pressures occurred between 2009 and 2012, when above-average river discharge combined with localised cyclone damage resulted in very poor Seagrass Index scores across multiple regions.

Subsequent years have also seen regionally significant disturbances. These include Cyclone Debbie in 2016–17, which affected the Mackay–Whitsunday region; major flooding in the Burdekin region in 2018–19; and further flood impacts in the Burnett–Mary region in 2021–22. The legacy of these events continues to shape current seagrass condition, with many meadows still in stages of recovery and not yet able to achieve higher Seagrass Index values. The above-average river discharge recorded in 2024–25 across much of the Reef—especially in the central region—is expected to exert additional pressure on seagrass communities. However, the full ecological consequences of this most recent disturbance will only become evident in subsequent reporting years, as seagrass responses typically unfold over multiple seasons.

## Conclusions

Reef-wide inshore seagrass condition was assessed as **‘poor’** in 2024–25, declining from the ‘moderate’ condition reported over the previous 4 monitoring periods. Inshore seagrass condition declined across all northern and central regions, with Cape York, Wet Tropics and Burdekin classified as ‘poor’, and the Mackay–Whitsunday region assessed as borderline ‘moderate’. In contrast, condition improved in the southern regions, increasing to ‘moderate’ in the Fitzroy and to ‘good’ in the Burnett–Mary.

These regional patterns reflect a system under sustained pressure from multiple environmental drivers, including elevated river discharge, marine heatwaves, cyclones and local-scale disturbances. The observed spatial variability in condition and indicator scores is strongly influenced by the legacy of past extreme events, such as Cyclone Debbie in 2016–17 in the Mackay–Whitsunday region, widespread above-average riverine discharge across

the southern and central Reef, and the 2018–19 marine heatwave that affected northern and central regions. In 2024–25, extreme and widespread elevated discharge affected the inshore Reef—especially the central region—and contributed to below-average light availability, a key limiting factor for seagrass productivity. Daily within-canopy seawater temperatures were comparable to the previous reporting year but reflected the fourth consecutive year of above-average temperatures, and the sixth-highest recorded since the MMP began (with 2016–17 the highest).

Climate change remains the most significant long-term threat to Reef seagrass condition. While the 2025–26 wet season is currently forecast to be near average—given a neutral El Niño–Southern Oscillation (ENSO) state with weak indications of La Niña—ongoing high local sea surface temperatures are expected to influence early-season rainfall patterns and cyclone likelihood. An average wet season may offer short-term opportunities for stabilisation or early recovery in some regions, particularly in the north; however, the cumulative nature of recent pressures means that resilience-building is now critical.

Addressing these challenges will require strengthening the MMP to provide a more integrated, predictive, and scalable approach to understanding the resilience and long-term health of inshore seagrass habitats, given the escalating pressures they face. Key improvements include integrating cumulative pressure assessments to better diagnose combined stressors and guide management priorities, refining thermal risk models and light indicators, developing fragmentation metrics, incorporating local quantitative indicators, and upscaling monitoring to improve seagrass assessment, connectivity insights, and regional management.

In parallel, improving water quality through reductions in catchment run-off remains central to reducing stress on seagrass meadows. Complementary actions will also be required to support recovery and enhance resilience, including developing recovery models to guide intervention timing, advancing targeted restoration research for large-scale application, and applying environmental engineering to improve habitat conditions by addressing limiting factors. Additional strategies, such as enhancing heat resilience through the use of pre-adapted plants, thermal priming of seedlings, and selecting heat-tolerant species and genotypes for restoration, will further strengthen recovery capacity. Together, these measures will be essential for securing the long-term health and functioning of inshore Reef seagrass ecosystems in an increasingly challenging climate.

# 1 Introduction

Approximately 3,464 km<sup>2</sup> of inshore seagrass meadows have been mapped with moderate to high confidence 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 seagrass extent (90 per cent or 32,215 km<sup>2</sup>) in the World Heritage Area is located in waters deeper than 15 m and mapped with low to moderate confidence (McKenzie *et al.* 2022). Comparatively, these deepwater meadows are relatively sparse, structurally smaller, highly dynamic, composed of colonising species, and less productive for fisheries resources than inshore seagrass meadows (McKenzie *et al.* 2010b; Derbyshire *et al.* 1995). Overall, the total estimated area of seagrass (35,679 km<sup>2</sup>) within the World Heritage Area represents nearly half 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.

The tropical seagrass ecosystems of the Reef represent a complex mosaic of diverse habitat types, consisting of multiple seagrass species (Carruthers *et al.* 2002). The Reef is home to 15 species of seagrass (Waycott *et al.* 2007), and it boasts a rich variety of seagrass habitats and community types. This diversity is supported by extensive bays, estuaries, and rivers that stretch along the 2,300 km length of the Reef, alongside its inshore lagoon and reef platforms. Seagrasses can be found on intertidal banks, on reef platforms and in shallow reef lagoons, colonising sand and mud substrates down to 70 m or more below mean sea level (Carter *et al.* 2021b).

Seagrasses in the Reef can be separated into 4 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 temperature, salinity, and shelter (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.

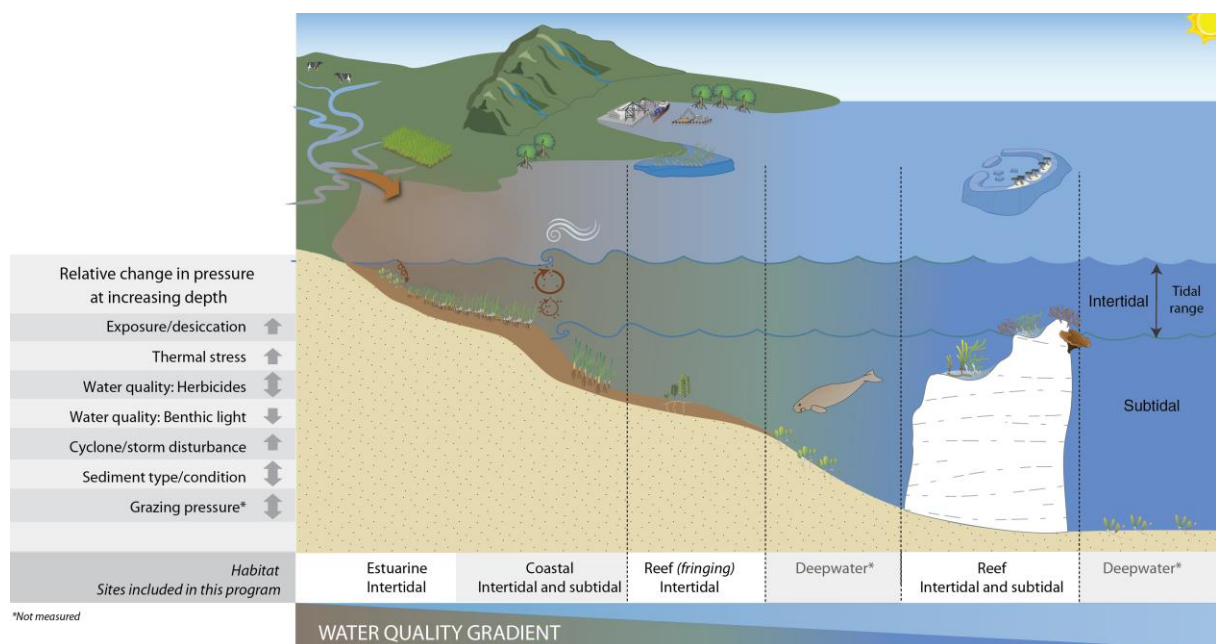


Figure 2. General conceptual model of seagrass habitats in north east Australia and the water quality impacts affecting the habitat (adapted from Carruthers *et al.* 2002, and Collier *et al.* 2014). Grey arrows indicate increase, decrease or variable response with increasing depth.

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, e.g., *Thalassia hemprichii*, 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 (Great Barrier Reef Marine Park Authority 2014). The most urgent initiatives for the Reef are those that will halt and reverse climate change and those that will effectively improve water quality at the regional scale (Great Barrier Reef Marine Park Authority 2024). Good water quality supports the health and resilience of coastal and inshore ecosystems of the Reef (Waterhouse *et al.* 2024a).

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 and Queensland Governments 2018), and integrated as a major component of Reef 2050 Long-Term Sustainability Plan (Reef 2050 Plan) (Commonwealth of Australia 2021), 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 and Queensland Governments 2018). The Great Barrier Reef Marine Monitoring Program (MMP) forms an integral part of the Paddock to Reef program. The MMP has 3 components: inshore water quality, coral and seagrass.

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

The inshore water quality monitoring component has been delivered by James Cook University (JCU) and the Authority since 2005. The seagrass component is also supported by contributions from the Seagrass-Watch program 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 component are available on-line (Great Barrier Reef Marine Park Authority 2025).

## 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 (Table 1).

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.

Table 1. 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.

Report Card category	Indicator category	Minutes-Days	Weeks	Months	Years	Seagrass report	Report card
Water quality	Climate	Cyclones				Y	
		Rainfall & river discharge <sup>^</sup>				Y	
		Wind (resuspension of sediments, scouring of sediments, currents)				Y	
		Extreme water temperature (hours/days > threshold)				Y	
	Water quality	Chronic temperature rise (weekly anomalies)				Y	
		Total suspended solids, turbidity, Secchi depth <sup>^</sup>					Y
		Chlorophyll a <sup>^</sup>					Y
		Nutrients (dissolved and particle forms of N, P & C) <sup>^</sup>					
		Temperature and salinity <sup>^</sup>					
		Water colour (weekly colour classes) <sup>^</sup>				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	

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

### Measures of Environmental stressors

Climate and environment stressors are aspects of the environment, either physiochemical or biological, that affect seagrass meadow condition. Some environmental stressors change rapidly (minutes/days/weeks/months) but can also undergo chronic shifts (years) (Table 1).

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 impending 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 (plant-like organisms that grow on the surface of seagrass leaf blades) 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<sup>-2</sup> d<sup>-1</sup> are unlikely to support long-term growth of seagrass, and periods below 6 mol m<sup>-2</sup> d<sup>-1</sup> for more than 4 weeks can cause loss

(Collier *et al.* 2016). 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 overlying 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 (Table 1). 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 and 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.

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 3). 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 3).

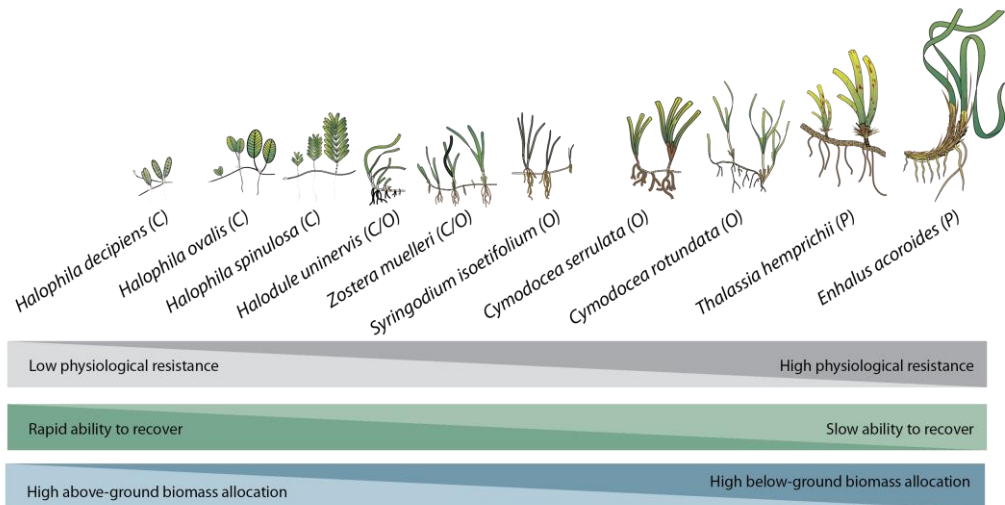


Figure 3. Dominant traits among the Reef seagrass species, with emphasis on their ability to either resist disturbances, or to recover following loss: colonising (C), opportunistic (O), or persistent (P). Adapted from Collier *et al.* (2021b) and Kilminster *et al.* (2015).

### 1.3 Structure of the Report

This report presents data from the twentieth monitoring period for the inshore seagrass ecosystems of the Reef under the MMP (undertaken from June 2024 to May 2025; hereafter called 2024–25). The inshore seagrass monitoring component of the MMP reports on:

- abundance and species composition of seagrass (including meadowscape mapping) in the late dry season (September to November) of 2024 and the late wet season (March to May) of 2025 at inshore intertidal and subtidal locations
- resilience, including sexual 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 2024–25, 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 Gruber *et al.* (2026) 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 and Collier (2025).

### 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.

Climatic pressure measurements are typically gathered at various scales, including location, sub-region, or regional levels. In contrast, environmental pressure measures are collected at a site-specific level, focusing on the area within the meadow canopy. The data source and sampling frequency is summarised in Table 3.

#### 2.1.1. Climate and inshore environment

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 inshore water quality component (Gruber *et al.* 2026).

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 use 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 component (Gruber *et al.* 2026). The inshore water quality component 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 (Table 2). 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 Gruber *et al.* (2026).

Table 2 Reef optical water types used to assess exposure of seagrass to water quality pressures (from Gruber *et al.* 2026).

Reef water type	Description	Colour of water to the eye
WT1 (Primary)	Waters with high phytoplankton levels and increasing sediment and dissolved organic matter	Brownish-green
WT2 (Secondary)	Waters with colour still dominated by algae, but increased dissolved organic matter and some sediment may be present	Greenish water
WT3 (Tertiary)	Slightly below ambient water quality, but with high light penetration	Greenish-blue
WT4 (Marine)	Ambient marine water with high light penetration	Blue

Water colour has been confirmed as a predictor of changes in seagrass abundance (Petus *et al.* 2016). Primary and secondary water types (WT1 and WT2) 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 (WT3) exposure is also presented in summary tables for each site. It is important to note that Reef water types, do not always correspond to direct catchment discharge influence, and can be due to marine processes (especially the Reef WT3) and to resuspension in shallow areas (especially the Reef WT1).

Light attenuation is just one of the stressors from wet season water types that impact inshore seagrass. Land-based runoff introduces several additional stressors that can be detrimental to seagrass. Pesticides, particularly PSII herbicides, that enter coastal waters can harm seagrass at particular concentrations, resulting in negative physiological or ecological effects. Seagrasses are not only affected by acute herbicide concentrations, but are also sensitive to chronic exposure to low concentrations (Negri *et al.* 2015). Prior to 2015, sediment herbicides within the seagrass rhizosphere were assessed in the late wet season across 30 MMP long-term sentinel sites (McKenzie *et al.* 2015). Of the thirteen herbicides assessed, the PSII herbicide Diuron was the most prevalent and widespread. High concentrations of herbicides have been reported in the water column over seagrass meadows immediately following flow events (McMahon *et al.* 2005). Although, concentrations leading to measurable effects are rarely detected in the Reef (Brodie *et al.* 2013), cumulative impacts (e.g. reduced photosynthetic C uptake caused by herbicides and low light) may occur (Ostrowski *et al.* 2022).

In 2024–25, pesticides were assessed as part of the MMP inshore water quality component at sites in the Wet Tropics, Burdekin and Mackay–Whitsunday regions (Gruber *et al.* 2026). Sampling included both Passive samplers and Grab samples (Gruber *et al.* 2026). Passive sampler deployment in close proximity ( $\leq 1$  km) of seagrass monitoring sites included: Low Isles (250 m from LI2, 480 m from LI1), and Dunk Island (700 m from DI1, 1.0 km from DI3) in the Wet Tropics (Gruber *et al.* 2026). Flood event Grab samples in the vicinity (1–5 km) of seagrass sites included: Green Island (closest sample was 2.5 km from GI1), Dunk Island (850 m from DI3 and 1 km from DI1) Lugga Bay (2.8 km from LB2) and Missionary Bay (9.8 km from MS1) in the Wet Tropics; Magnetic Island (closest was 3.8 km from MI3) and Jerona (1.2 km from JR2) in the Burdekin; and Repulse Bay (6 km from MP2) and Lindeman Island (2.3 km from LN1) in the Mackay–Whitsunday region. Results are presented as concentrations relative to ANZECC and ARMCANZ guidelines (e.g., 270 ng L<sup>-1</sup> for the PSII herbicide Diuron), and Pesticide Risk Metric threshold of 1% of species affected (Gruber *et al.* 2026).

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 21). If the tide gauge was too far or unavailable, official BOM tidal height predictions at nearest port were used.

### 2.1.2. Environment within the seagrass habitats

Autonomous HOBO<sup>®</sup> submersible temperature loggers (HOBO<sup>®</sup> MX2201) were deployed at all sites identified in Appendix 2, Table 20. The HOBO<sup>®</sup> MX2201 loggers recorded temperature (resolution 0.04 °C, accuracy  $\pm 0.5$  °C) every 10 minutes (Table 3). 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 23 intertidal 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 2, Table 20). The light sensor is positioned upright at the seagrass canopy. Detailed methodology for the light monitoring can be found in McKenzie *et al.* (2021b). Automatic wiper brushes clean the optical surface of the sensor every 15 minutes to prevent marine organisms fouling. Measurements were recorded by the logger every 15 minutes and are reported as total daily light ( $\text{mol m}^{-2} \text{d}^{-1}$ ), hereinafter daily light. Light data presented for NRM and GBR-wide plots uses only site data where there is more than 50 per cent of annual data available.

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 33 quadrats at each site in conjunction with seagrass abundance measures (see 2.2.3) 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). The average proportion of mud at each site during the monitoring period is compared against the Reef-wide long-term average guideline for each habitat. This serves as an indicator of change and helps identify trends over time.

Table 3. 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
<i>Climate</i>					
Cyclones	1968–2025	remote sensing and observations at nearest weather station	yearly	No. yr <sup>-1</sup>	Bureau of Meteorology
Rainfall	1889–2025*	rain gauges at nearest weather station	daily	mm mo <sup>-1</sup> mm yr <sup>-1</sup>	Bureau of Meteorology
Riverine discharge	1970–2025	water gauging stations at river mouth		L d <sup>-1</sup> L yr <sup>-1</sup>	DES#, compiled by (from Gruber <i>et al.</i> 2026)
Plume exposure	2006–2025 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 Gruber <i>et al.</i> 2026)
Pesticides in water column	2005–2025*	<i>In situ</i> Passive samplers and opportunistic flood event Grab samples	November to May	concentration (ng L <sup>-1</sup> ) and Pesticide Risk Metric (PRM) (per cent of species affected)	MMP inshore water quality program (from Gruber <i>et al.</i> 2026)
Tidal exposure	2005–2025	water level readings recorded at location closest to monitoring site using an automatic tide gauge. If the tide gauge was too far or unavailable official BOM tidal height prediction at nearest port was used	10 min	hours exposed during daylight	Maritime Safety Queensland, BOM tidal unit, water depth over sites measured by MMP Inshore Seagrass monitoring
<i>Environment within seagrass habitat</i>					
Water temperature	2002–2025	iBTag or HOBO® MX2201	15–90 min	°C, temperature anomalies, exceedance of thresholds	MMP Inshore Seagrass monitoring
Light	2008–2025	Odyssey 2Pi PAR light loggers with wiper unit	15 min	daily light (mol m <sup>-2</sup> d <sup>-1</sup> ) frequency of threshold exceedance (per cent of days)	MMP Inshore Seagrass monitoring
Sediment grain size	1999–2025	visual / tactile description of sediment grain size composition	biannual	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 6 natural resource management (NRM) regions with catchments draining into the Reef: Cape York, Wet Tropics, Burdekin, Mackay–Whitsunday, Fitzroy and Burnett–Mary (Table 4, Figure 4). Within inshore seagrass meadows, seventy-five sites across 34 locations were assessed during the 2024–25 monitoring period (Table 4, Appendix 2, Table 20). This covered 4 estuarine, seventeen coastal, and thirteen reef locations.

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 inshore meadows was based upon several key factors:

- *Representation*: the meadows needed to be 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),
- *Exposure to Riverine Discharge*: meadows in habitats that spanned a range in exposure to riverine discharge, with estuarine and coastal habitats generally experiencing the highest degree of exposure, and reef meadows having the least,
- *Legacy Sites*: where possible meadows which included previously monitored 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),
- *Low variability*: meadows that exhibited minimal variability in per cent cover, 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).

Inshore meadows were mapped across the regions prior to selection and establishment of monitoring sites. 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 2 sites were selected at each location. If meadow overall extent was larger than ~15 hectares (0.15 km<sup>2</sup>), 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.

Table 4. Inshore seagrass monitoring locations and annual sampling. MMP = Marine Monitoring Program, SW= Seagrass-Watch, RJFMP = Reef Joint Field Management Program, ● indicates late dry (September to December) and late wet (March to May), ○ 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	2021-22	2022-23	2023-24	2024-25
Cape York	Shelburne Bay	MMP								●	●	○	○		○	○	○	○	○	○	○	○
	Margaret Bay	RJFMP																	○	○	○	○
	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	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●
	Cid Harbour	RJFMP																	○	○	○	○
	Tongue Bay	RJFMP												○	○	○	○	○	○	○	○	○
	Whitehaven Beach	RJFMP																	○	○	○	○
	Hamilton Island	MMP			●	●	●	●	●	●	●	●	●	●	●	●	●	○	●	●	●	●
	Lindeman Island	MMP													●	●	○	●	●	●	●	
	Repulse Bay	MMP	●	●	○	○	○	○	●	●	●	●	●	●	●	●	●	●	●	●	●	●
	St Helens Bay	SW													○	○	○	○	○	○	○	
	Newry Islands	RJFMP												○	○	○	○	○	○	○	○	
	Sarina Inlet	MMP	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●
	Llewellyn Bay	SW																	○	○	○	
	Clairview	SW													○	○	○	○	○	○	○	
Fitzroy	Shoalwater Bay	MMP	●	●	●	●	●	●	●	●	●	○	○	○	●	○	○	○	○	○	○	
	Keppel Islands	MMP			●	●	●	●	●	●	●	○	○	○	●	●	●	●	○	○	○	
	Gladstone Harbour	MMP	●	●	●	●	●	●	●	●	●	○	○	●	●	●	●	●	○	○	○	
Burnett-Mary	Rodds Bay	MMP			●	●	●	●	●	●	●	●	○	○	●	●	●	●	○	○	○	
	Burrum Heads	MMP, SW	●	●	○	○	○	○	●	●	●	○	○	○	○	○	○	○	○	○	○	
	Hervey Bay	MMP	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	●	

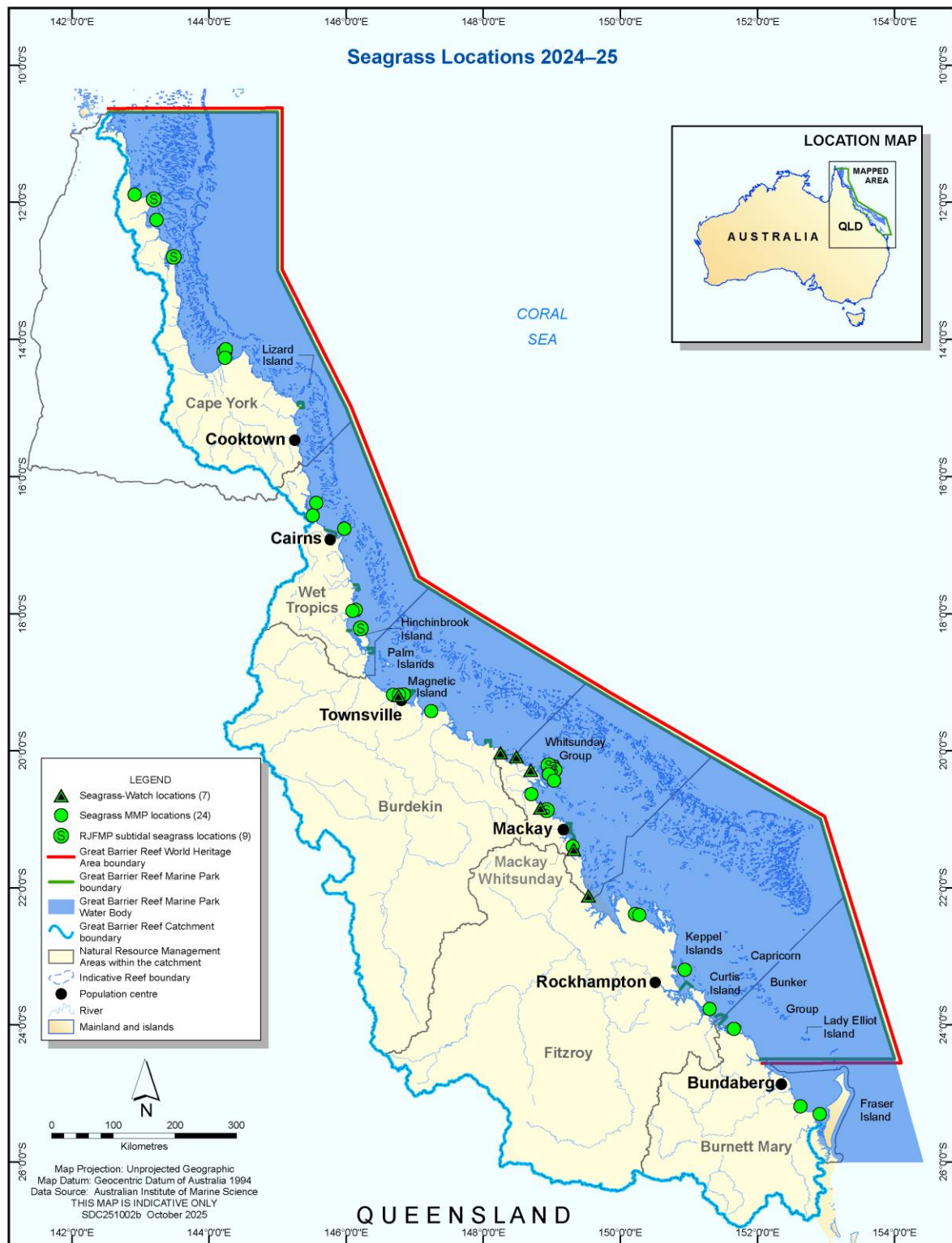


Figure 4. Inshore seagrass survey locations that exist as of 2024-25. However, not all locations were surveyed in 2024-25 (see Table 2).

Some of the restrictions for working in hazardous waters are overcome by using drop cameras and grab samplers.

The long-term median annual daylight exposure (the time intertidal meadows are exposed to air during daylight hours) was 3.6 per cent (all meadows pooled) (Table 21). 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).

The meadows chosen for monitoring were lower littoral (rarely exposed to air), although classified intertidal within the MMP. Predominantly 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 5).

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 genera: *Cymodocea*, *Enhalus*, *Halodule*, *Zostera*, *Syringodium* and *Thalassia* (Table 5).

The monitoring approach implemented for MMP did not include depth range monitoring as this has had limited success in subtropical/tropical seagrass meadows 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 3.42 m (Cairns) to 10.4 m (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 meadowscapes
- near-absence of shallow subtidal meadows south of Mackay–Whitsunday due to the large tides which scour the seabed.

Deepwater (>15 m depth) meadows occur 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 were excluded from

monitoring as 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).

### 2.2.2 Sampling timing and frequency

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 season (September–November) and late wet season (March–May) to capture the condition of seagrass pre– and post–wet. Changes in indicators at sites sampled in the late dry only (Table 4) (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 assessed by JCU, an additional 15 sites from Seagrass-Watch and 18 sites from QPWS were established to improve the spatial resolution and representation of intertidal and subtidal habitats, respectively (Table 6). In 2024–25, however, only 10 sites were assessed by Seagrass-Watch and 18 by QPWS (Table 20).

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 20). The seagrass species (including foundation) present at each monitoring site is listed in Table 5 and Table 6.

### 2.2.3 Seagrass abundance, composition and extent

Seagrass abundance, species composition, and meadow spatial extent were assessed from samples collected in the late dry 2024 and late wet 2025 at locations identified in Table 5. Field survey methodology followed globally standardised protocols (detailed in McKenzie *et al.* 2003). At each location, with the exception of subtidal sites, sampling included 2 sites nested within 500 m of each other. Subtidal sites were not always replicated within locations. Sites were defined as a 5.5 ha area intertidally and 3.1 ha 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 2024) and late wet (March–May 2025) 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 3 50 m transects, located 25 m apart). Transects are placed in the same position ( $\pm 3$  m) each assessment. These monitoring sites represented the seagrass population within a meadow and as the unit of measure (quadrat) was not anchored to the substrate between sampling events, the probability of repeatedly sampling the same shoot or ramet was unlikely. Therefore, the sampling design and spread of quadrats are treated as random in data analysis, given the clonal nature of seagrasses, the fast growth of tropical species, errors in positioning of fixed transects and quadrats, and physical losses at scales of shoot, plant, and meadow.

The sampling strategy for subtidal sites was modified in 2021–22, as a result of the discontinuation of SCUBA diving; driven by budgetary constraints and logistic and occupational health and safety issues relating to diving in poor visibility coastal waters. At each site, a drop–camera assembly (incl. frame with 0.25 m<sup>2</sup> quadrat in field of view), was used to visually assess the seabed and the photoquadrat footage captured for post-field analysis. Along 3 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.

Table 5. Inshore sentinel seagrass long-term monitoring site details including presence of foundation (■) and other (□) seagrass species in the current or previous reporting periods.  
 ^ =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*, ZC = *Zostera muelleri*.

Region	NRM region (Board)	Basin	Monitoring location	Site	Longitude	Latitude	CR	CS	EA	HD	HO	HS	HU	SI	TH	ZC		
Far Northern	Cape York (Cape York Natural Resource Management)	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	■				□				■		
		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	■				□			■	■	■	■
Northern	Wet Tropics (Terrain NRM)	Daintree	Low Isles reef	LI1	Low Isles	145.565	-16.385				□			■		■		
				LI2^	Low Isles	145.564	-16.383				□			■				
			Yule Point coastal	YP1	Yule Point	145.512	-16.569					□			■			■
		Mulgrave-Russell / Johnstone	Green Island reef	GI1	Green Island	145.973	-16.762	■	■			□			■		■	
				GI2	Green Island	145.976	-16.761											
				GI3^	Green Island	145.973	-16.755	■	■			□			■	■	■	
		Mission Beach coastal		LB1	Lugger Bay	146.093	-17.961					□			■			
		Tully / Murray / Herbert	Lugger Bay coastal	LB2	Lugger Bay	146.094	-17.961											
				Pallon Beach coastal	D11	Pallon Beach	146.141	-17.944	■	■			□			■		
			Dunk Island reef	D12	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					□		■				
				LN3	Lindeman Is.	149.033	-20.438					□		■				
			Repulse Bay coastal	MP2	Midge Point	148.702	-20.635					□		■			■	
		Hamilton Island reef	MP3	Midge Point	148.705	-20.635												
			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 6. Additional inshore sentinel seagrass long-term monitoring sites integrated from the Seagrass-Watch (intertidal sites) and RJFMP drop-camera (subtidal sites)<sup>^</sup> programs, including presence of foundation (■) and other (□) seagrass species in the current or previous reporting periods. NRM region from www.nrm.gov.au. <sup>^</sup>=subtidal, ~ =not assessed in 2024–25.

Region	NRM region (Board)	Basin	Monitoring location	Site	Longitude	Latitude	CR	CS	EA	HD	HO	HS	HU	SI	TH	ZC	
Far Northern	Cape York (Cape York Nat Res Manage)	Jacky	Margaret Bay coastal	MA1 <sup>^</sup>	Margaret Bay	143.19358	-11.9574					□	□	■			
				MA2 <sup>^</sup>	Margaret Bay	143.20338	-11.9559										
		Lockhart	Weymouth Bay reef	YY1~	Yum Beach	143.36059	-12.571	■	■	■		□		■		■	
				LR1 <sup>^</sup>	Lloyd Bay	143.485	-12.797					□	□	■			
		Normanby / Jeannie	Flinders Group reef	FG1 <sup>^</sup>	Flinders Island	144.225	-14.182					□	□	■			
				FG2 <sup>^</sup>	Flinders Island	144.225	-14.182										
			Bathurst Bay coastal	BY3 <sup>^</sup>	Bathurst Bay	144.285	-14.276					□		■			
				BY4 <sup>^</sup>	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	■	■			□	■				
				MS1 <sup>^</sup>	Cape Richards	146.213	-18.216					□		■			
		Missionary Bay coastal	MS2 <sup>^</sup>	Macushla	146.217	-18.205											
Ross / Burdekin	Townsville coastal		SB2	Shelley Beach	146.763	-19.182		■			□		■			■	
		Don	Bowen coastal	BW2	Port Dennison	148.25196	-20.01731					□		■			■
Proserpine	Shoal Bay reef			HB1	Hydeaway Bay	148.482	-20.075	■				□		■		■	
		Pioneer Bay coastal	PI2	Pigeon Island	148.693	-20.269					□	□	■				■
Proserpine / O'Connell	Cid Harbour reef		CH4 <sup>^</sup>	Cid Harbour	148.9506	-20.213		■			□	□	■	■			
		CH5 <sup>^</sup>	Cid Harbour	148.9451	-20.222												
O'Connell / Pioneer	Tongue Bay reef	TO1 <sup>^</sup>	Tongue Bay	149.016	-20.240					□		■		■			
		TO2 <sup>^</sup>	Tongue Bay	149.012	-20.242												
Whitehaven Beach reef	WB1 <sup>^</sup>	Whitehaven Bch	149.0386	-20.2808		■			□	□	■	■					
	WB3 <sup>^</sup>	Whitehaven Bch	149.0534	-20.2944													
St Helens Bay coastal	SH1~	St Helens Bch	148.835	-20.822						□		■			■		
	Newry Islands coastal	NB1 <sup>^</sup>	Newry Bay	148.926	-20.868		■				□	□	■	■			
NB2 <sup>^</sup>		Newry Bay	148.924	-20.872													
	Llewellyn Bay coastal	LL1	Deception Inlet	149.318	-21.424						□					■	
Clairview coastal		CV1	Clairview	149.533	-22.104						□		■			■	
	CV2	Clairview	149.535	-22.108													

Seagrass species were identified as per Waycott *et al.* (2004). The recent proposal to reclassify *Zostera muelleri* to *Nanozostera muelleri* (Sullivan and Short 2023), has not been adopted to date as it has yet to be supported by the Council of Heads of Australasian Herbaria (CHAH) (Australian Plant Name Index 2025). Species were further categorised according to their life history traits and strategies and classified into colonising, opportunistic or persistent as broadly defined by Kilminster *et al.* (2015) (for detailed methods, see McKenzie and Collier 2025).

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 GNSS receiver 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 greater than 2 m (i.e. accuracy of the GNSS). Each entire site (5.5 ha intertidal and 3.1 ha subtidal) was mapped with high confidence (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.

Where permitted and when conditions were suitable (e.g., intertidal seagrass exposed with winds <20 kts and no rain), the meadows were mapped using a UAV (DJI Mavic 3 Enterprise) captured orthomosaic (McKenzie *et al.* 2022). The flight mission and image data capture was executed using the DroneDeploy app, with an altitude between 35 m and 60 m and 80 % front and side overlap. The orthomosaic was produced using Pix4D or Agisoft Metashape. The resulting seagrass maps were mainly digitized in QGIS from visual interpretation of the orthomosaic. For the more complex or fragmented sites, only small representative parts were manually digitized to train a deep learning segmentation model (Unet) (McKenzie *et al.* 2022). The final maps were then obtained from the prediction of seagrass presence from the model (probability>0.25).

#### 2.2.4 Seagrass reproductive status

Seagrass reproductive state was assessed from samples collected in the late dry 2024 and late wet 2025 at the 47 sites identified in Table 20. Samples were processed according to standard methodologies (McKenzie *et al.* 2021b).

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.

Seed banks and abundance of germinated seeds were sampled according to standard methods (McKenzie *et al.* 2021b) by sieving (1 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. Seeds are no longer collected at subtidal sites as sampling on SCUBA was discontinued as a result of budgetary constraints, logistic and occupational health and safety issues.

#### 2.2.5 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 1, Table 19).

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

- *very good*, median per cent cover at or above 75<sup>th</sup> percentile
- *good*, median per cent cover at or above 50<sup>th</sup> percentile
- *moderate*, median per cent cover below 50<sup>th</sup> 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 20<sup>th</sup> or 10<sup>th</sup> percentile was used for the low guideline depended on the within-site variability; generally, the 20<sup>th</sup> percentile is used, unless within-site variability was low (e.g. CV<0.6), whereby the 10<sup>th</sup> 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 20<sup>th</sup> percentile). Details on the per cent cover guidelines can be found in Appendix 1.

A grade score from 0 to 100 (Table 7) 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 7. Scoring threshold table to determine seagrass abundance grade. low = 10<sup>th</sup> or 20<sup>th</sup> percentile guideline. NB: scores are unitless.

Grade	Percentile category	Score
<i>very good</i>	75–100	100
<i>good</i>	50–75	75
<i>moderate</i>	low–50	50
<i>poor</i>	<low	25
<i>very poor</i>	<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 multiple elements of resistance and recovery.

Resistance is incorporated into the metric through meadow condition, and whether abundance and species composition exceed critical thresholds (<20<sup>th</sup> percentile or >50<sup>th</sup> percentile, 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.

Recovery is reflected in the metric through reproductive effort, which 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 3 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 1, Figure 90.

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

Table 8. 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 <20 <sup>th</sup> 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 >20 <sup>th</sup> percentile of site	Reproduction (foundational) not present last 3 years	Proportion of persistent species present (min <10 <sup>th</sup> percentile, max 95 <sup>th</sup> 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 >20 <sup>th</sup> percentile of sites AND persistent species present	Reproduction (foundational) present	Reproductive structure count (min <10 <sup>th</sup> percentile, max 95 <sup>th</sup> percentile)	70–100	2.2.1
			Reproductive structure count (min <10 <sup>th</sup> percentile, max 95 <sup>th</sup> percentile)	85–100	2.2.2

### 2.3.3 Seagrass Index

The seagrass condition index (Seagrass Index) is an average score (0–100) of the 2 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 9). *Please note: Cape York omitted from the score in reporting prior to 2012 due to poor representation of inshore monitoring sites.*

Table 9. Area of seagrass shallower than 15 m depth 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 <sup>2</sup> )	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
<b>World Heritage Area</b>	<b>3,464</b>	<b>1.00</b>

## 2.4 Data analyses

All analysis was run in the software R-4.5.2 (R Core Team 2025). The R code is available on request from L.J.M. and L.A.L.

### 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 2 seagrass indicators and overall Seagrass 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  
 $f(y|\mu,\sigma)=(1-\nu)*(\text{Gamma}(\sigma)/\text{Gamma}(\mu*\sigma))*\text{Gamma}((1-\mu)*\sigma)*y^{(\mu*\sigma-1)}*(1-y)^{((1-\mu)*\sigma)-1}$   
 The parameters satisfy  $0<\mu<1$ ,  $\sigma>0$  and  $0<\nu<1$ .

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

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

$$\text{Var}(y)=(1-\nu)*(\mu*(1-\mu))/(\sigma+1) + \nu*(1-\nu)*\mu^2$$

In our models Site was included as a random effect. Because some sites had very drastic changes in their abundance through time with sometimes complete seagrass loss, random effect cannot be accurately estimated over the whole time series. Therefore, per cent cover at the quadrat level for each seasonal date was analysed separately. The inclusion of random effect in the model is important to account for site-specific variance which results in more accurate estimations of confidence intervals around the mean across the various levels. The intercept model fitted was as followed:

$$\text{Formula : Percent\_cover} \sim 1 + re (\text{random}(\sim 1|\text{Site}))$$

The random effect of site was included in the 3 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 24 Table 25, Table 26). 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- $\tau$ ), 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 2024–25

The following section provides detail on the overall climate and environmental pressures during the 2024–25 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 Appendix 3:

#### 3.1 Summary

Environmental stressors for the Reef in 2024–25 were well above the long-term average/median for rainfall and river discharge, especially for the central NRM regions (Table 10) although there was significant variability across different regions. The 2024–25 wet season was not influenced by any significant cyclonic activity across the inshore or broader Reef. Rainfall in 2024–25 was above the 30-year long-term average across the Wet Tropics, Burdekin, Mackay–Whitsunday, and Burnett–Mary regions, whereas totals in Cape York and Fitzroy were close to the long-term mean. In 2024–25, river discharge to the inshore Reef was well above the long-term median, marking the wettest year since 2010–11 after 3 successive wet years. Discharge was particularly elevated in the Burdekin, Mackay–Whitsunday, and Burnett–Mary regions—6.8, 2.8, and 3.4 times higher than their respective long-term medians—with most other basins also experiencing above-average flows.

During the 2024–25 wet season, turbid primary (WT1) and secondary (WT2) waters dominated inshore Reef conditions, extending further offshore than usual in the Wet Tropics, Burdekin, and Mackay–Whitsunday regions. Exposure of seagrass monitoring sites to turbid waters was generally consistent with long-term averages. Across the Reef, the total area of seagrass exposed to water quality risk categories remained close to average, with minor regional shifts toward higher-risk categories in the Burdekin and Mackay–Whitsunday regions, and a modest increase in lower-risk exposure in the Fitzroy region.

Table 10. Summary of environmental conditions at monitoring sites across the Reef in 2024–25 compared to previous monitoring period and the long-term average (range indicated for each data set). \*intertidal only.

Environmental pressure	Long-term average	2023–24	2024–25
<i>Climate</i>			
Cyclones, number of events (1968–2024)	4	2	0
Wet season daily rainfall, mm d <sup>-1</sup> (1961–1990)	4.0	4.8	5.4
Riverine discharge, ML yr <sup>-1</sup> (1986–2016)	51,812,207	104,086,500	116,663,938
Wet season turbid water exposure, per cent (2003–2022)	86	84	84
<i>Within seagrass canopy</i>			
Temperature, °C (±) (max) (2003–2024)*	25.7 ±0.5 (50.8)	25.9 ±0.3 (50.8)	25.9 ±0.3 (41.2)
Daily light, mol m <sup>-2</sup> d <sup>-1</sup> (2008–2024) annual average (min site–max site)	13.6 (5.9–20.2)	12.0 (5.3–17.4)	12.4 (6.3–20.2)
Proportion mud, per cent			
<i>estuarine intertidal</i> (1999–2024)	44.6 ±2.2	52.7 ±2.3	50.1 ±2.3
<i>coastal intertidal</i> (1999–2024)	27.4 ±2	29.9 ±1.7	29 ±1.7
<i>coastal subtidal</i> (2015–2024)	53.8 ±5	60.7 ±0.4	41.3 ±0.4
<i>reef intertidal</i> (2001–2024)	4.2 ±1.2	5.4 ±0.4	3.3 ±0.4
<i>reef subtidal</i> (2008–2024)	16 ±1.0	25 ±0.6	19.8 ±0.6

Long-term trends in the Water Quality Index indicate that water quality has either improved or remained stable across most of the Reef regions examined in 2024–25 (Gruber *et al.* 2026). Two consecutive years of above-median river discharge across several Reef regions has resulted in deteriorating water quality conditions in the Wet Tropics and Mackay-Whitsunday regions, while maintaining stable water quality in the Burdekin and Fitzroy regions (Gruber *et al.* 2026). On a positive note, this year saw the Cape York region recover to 'good' condition after suffering severe impacts from Cyclone Jasper in 2023–24 (Gruber *et al.* 2026). Nevertheless, some indicators still exceed water quality guideline values across many focus regions and need further improvement. Even so, the general trend is promising given sustained high flows in the central Reef.

Daily light availability in 2024–25 remained below the long-term Reef average, and was lower than the previous period. This is based on 15 of the 23 sites where light is monitored, as the other 8 sites had insufficient data for inclusion in the averages. Daily light was lower than average in all regions except the Burdekin, where in 2024–25 light data was only available from reef sites which tend to have higher light than coastal sites.

The 2024–25 period was the fourth consecutive year of above-average within canopy seawater temperatures and sixth warmest on record. All regions, except for the Burdekin, recorded annual average temperatures exceeding the long-term average. Notably, only the Wet Tropics and Fitzroy regions experienced at least one day of extreme heat (over 40 °C). It is important to mention that the temperature data for the Fitzroy region is incomplete, which is particularly noteworthy since this area had the highest number of extreme temperature days in the previous period and recorded the hottest seawater temperature of 50.8 °C since the establishment of the MMP. Of interest, the Mackay–Whitsunday region, which has the highest number of extreme temperature days on average, did not experience any for the first time in a decade.

### 3.2 Cyclones

Over the 2024–25 reporting period, no significant cyclonic activity influenced the inshore or broader Reef (Gruber *et al.* 2026, Table 10). Tropical Cyclone Alfred affected south-east Queensland, but remained well offshore and had no impact on the southern Reef. Since 2005, 4 intense systems have resulted in widespread physical damage and flooding, adversely affecting inshore seagrass habitats. Between 2005 and 2020, cyclones Larry (2006) and Yasi (2011) impacted the Wet Tropics and Burdekin regions, and Cyclone Debbie (2017) impacted the Mackay–Whitsunday region. Over the last 5 years, the most intense system was Cyclone Jasper (2023), which impacted the Wet Tropics, southern Cape York and to a lesser degree the Burdekin region (Figure 5). A number of other systems have crossed the inshore Reef over the last 5 years, in particular Cyclone Tiffany (2022) in Cape York and more recently Cyclone Kirrily in the previous period in the Burdekin region.

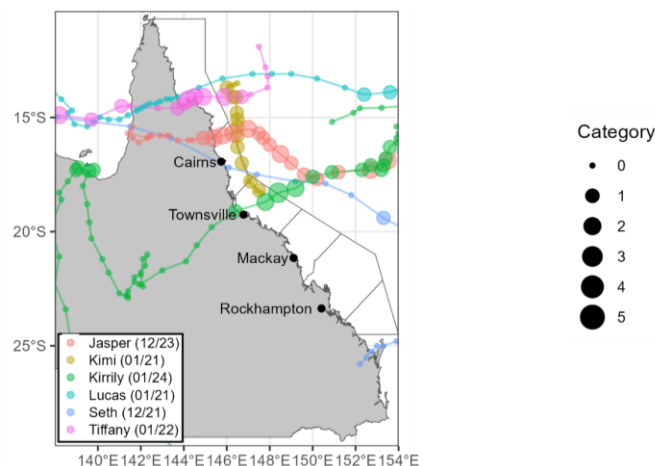


Figure 5. Cyclone tracks for systems crossing the inshore Reef between 2020 and 2025. Figure courtesy Thompson *et al.* 2026. Tracks sourced from the BoM.

### 3.3 Rainfall

Rainfall across the Reef regions in the 2024–25 wet season was above the 30-year long-term average (1961 to 1990) in the Wet Tropics, Burdekin, Mackay–Whitsunday and Burnett–Mary regions. Rainfall in Cape York and the Fitzroy regions was similar to the long-term average (Figure 6, Figure 7) (Gruber *et al.* 2026).

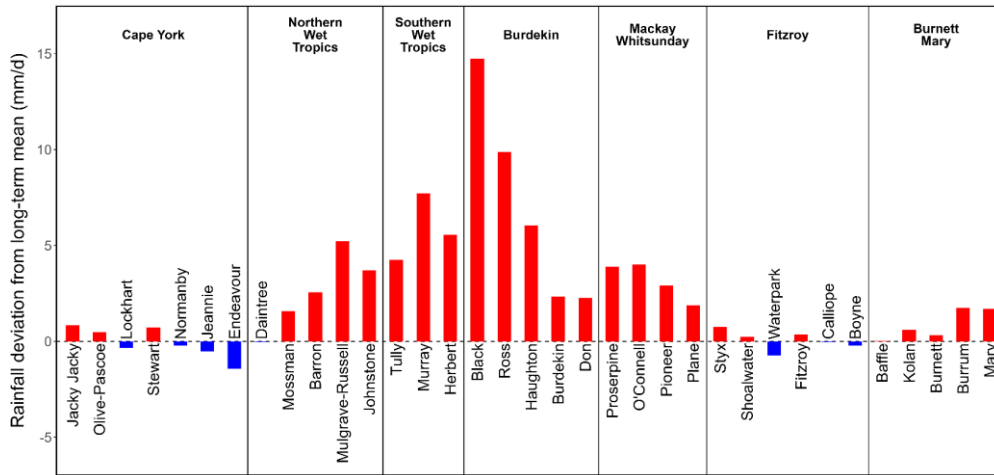


Figure 6. Per basin difference between annual average daily wet season rainfall (December 2024–April 2025) and the long-term average (1961–1990). Red and blue bars denote basins with rainfall above and below the long-term average, respectively. Note that the basins are ordered from north to south (left to right). Basins have been grouped into NRM regions as indicated by shaded panels. Compiled by Gruber *et al.* 2026.

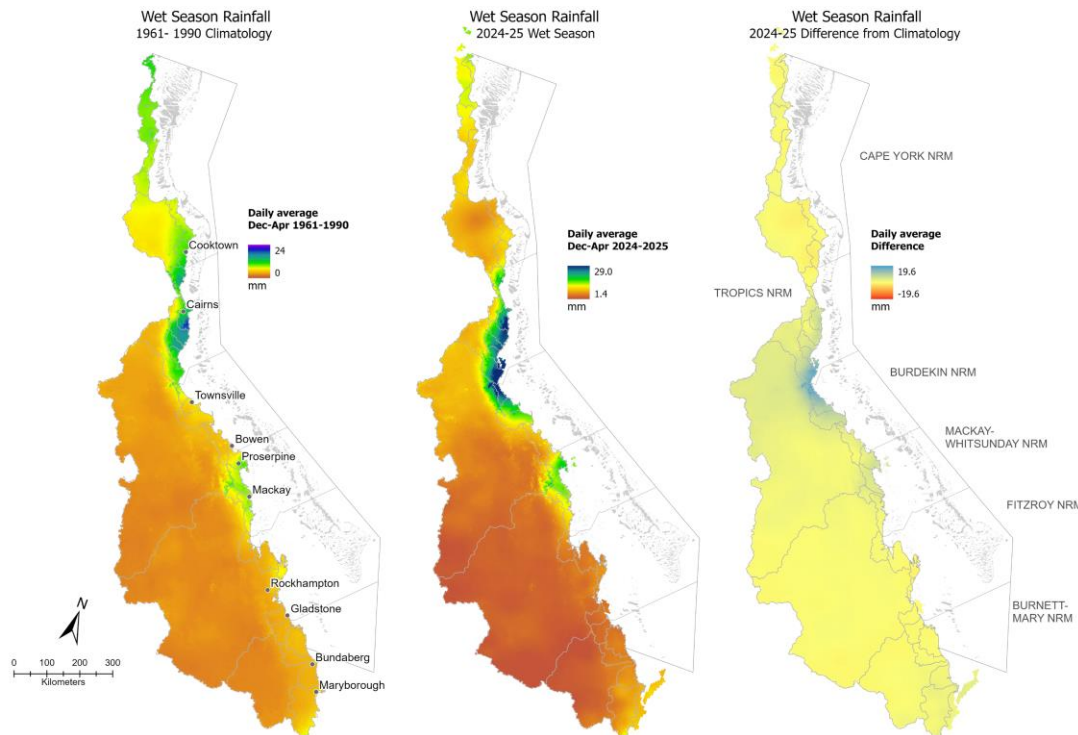


Figure 7. Average daily rainfall (mm day<sup>-1</sup>) in the Reef catchment: (left) long-term annual average (1961–1990; time period produced by BOM), (centre) 2024–25 and (right) the difference between the long-term annual average and 2024–25 rainfall patterns where negative values indicate less rain. From Gruber *et al.* 2026.

### 3.4 River discharge

The discharge of rivers into the inshore Reef is closely linked to rainfall during the wet season, significantly impacting the quality of coastal waters (Gruber *et al.* 2026). Annual river discharge for the Reef was above the long-term median in 2024–25. This increase followed 3 consecutive wet years and marked the wettest period since 2010–11 (Table 11). Discharges from basins in the Burdekin, Mackay–Whitsunday and Burnett–Mary regions were well above their respective long-term medians (6.8, 2.8, 3.4 times). Although not as high, discharges were above the long-term medians in the Cape York, Wet Tropics and Fitzroy regions 1.4, 1.6 and 1.2 times).

All 5 basins within the Burdekin NRM region had discharges exceeding 3 times their respective long-term medians (Table 11). The Burnett, Burrum and Mary basins in the also had discharge exceeding 3 times their respective long-term medians. All 4 basins of the Mackay–Whitsunday NRM region as well as the Barron and Herbert basins in the Wet Tropics had discharges between 2 and 3 times their respective long-term medians. The Jacky Jacky, Jeannie, Endeavour, Mossman and Murray basins all recorded discharge between 1.5 and 2.0 times their long-term medians (Table 11).

Table 11. Annual water year discharge (ML) of the main Reef rivers (1 October 2024 to 30 September 2025, inclusive) compared to the previous 3 wet seasons and long-term (LT) median discharge (1990–91 to 2019–20). Colours indicate levels above the long-term median: yellow = 1.5 to 2 times, orange = 2 to 3 times, and red = greater than 3 times. Compiled by Gruber *et al.* 2026.

Region	Basin	LT median	2021 - 2022	2022 - 2023	2023 - 2024	2024 - 2025
Cape York	Jacky Jacky Creek	2,471,267	2,365,731	4,611,721	3,487,440	4,035,439
	Olive Pascoe River	3,180,267	4,879,388	6,053,581	6,050,915	3,789,564
	Lockhart River	1,538,839	2,360,994	2,929,152	2,927,862	1,833,660
	Stewart River	758,172	569,738	1,366,633	1,100,668	37,954
	Normanby River	3,864,344	3,562,637	11,791,399	16,300,347	5,278,693
	Jeannie River	1,428,920	1,566,621	2,093,623	4,440,165	2,733,093
	Endeavour River	1,583,881	1,734,492	2,310,900	4,877,431	3,010,320
Wet Tropics	Daintree River	1,918,174	2,519,318	4,685,640	9,176,968	2,753,396
	Mossman River	604,711	800,754	815,267	1,745,893	972,623
	Barron River	622,447	692,908	1,217,590	3,603,793	1,355,584
	Mulgrave-Russell River	4,222,711	4,091,750	4,291,804	6,786,526	5,087,551
	Johnstone River	4,797,163	4,712,174	5,385,426	8,157,637	6,552,984
	Tully River	3,393,025	3,175,489	3,660,701	5,563,920	4,584,882
	Murray River	1,484,246	1,269,280	1,526,232	2,595,878	2,283,300
Burdekin	Herbert River	3,879,683	3,283,590	4,919,143	8,516,360	10,739,104
	Black River	293,525	273,677	353,756	526,432	2,335,237
	Ross River	279,376	202,811	209,681	285,424	2,403,245
	Haughton River	558,735	735,754	1,219,825	583,152	3,576,080
	Burdekin River	4,406,780	5,442,976	9,702,259	5,745,479	30,577,586
Mackay-Whitsunday	Don River	496,485	383,927	999,723	372,511	1,538,521
	Proserpine River	859,348	446,839	1,869,821	618,392	2,482,665
	O'Connell River	835,478	434,427	1,817,882	601,214	2,413,702
	Pioneer River	616,216	277,610	761,905	589,249	1,340,373
Fitzroy	Plane Creek	1,058,985	489,222	1,440,350	632,961	2,136,628
	Styx River	629,037	1,080,829	849,506	1,030,316	652,123
	Shoalwater Creek	727,306	1,250,433	982,586	1,191,945	754,037
	Water Park Creek	392,614	820,627	601,479	772,773	414,485
	Fitzroy River	2,875,792	4,505,289	3,078,896	2,100,507	3,858,636
	Calliope River	257,050	250,551	135,396	172,394	227,810
Burnett-Mary	Boyne River	179,108	171,925	44,649	85,541	146,790
	Baffle Creek	347,271	1,000,587	170,693	424,436	276,134
	Kolan River	115,841	818,716	83,734	139,893	126,425
	Burnett River	264,307	3,894,616	358,852	598,898	1,248,609
	Burrum River	130,835	1,612,683	270,059	476,512	926,346
Mary River	908,873	10,139,380	673,298	1,806,668	4,180,359	
<b>Sum of basins</b>		<b>60,746,947</b>	<b>71,817,742</b>	<b>83,283,163</b>	<b>104,086,501</b>	<b>116,663,938</b>

### 3.5 Water quality index

The MMP's water quality component assesses the annual and long-term condition of inshore water quality across the Reef, based on 20 years of monitoring data (Gruber *et al.* 2026). Inshore water quality is evaluated using an index derived from 5 key indicators: water clarity, nitrate/nitrite, particulate nitrogen, particulate phosphorus, and chlorophyll *a*. This data is reported annually for the Cape York, Wet Tropics, Burdekin, Mackay-Whitsunday, and Fitzroy regions (Gruber *et al.* 2026). Satellite imagery and remote sensing are integrated with on-site monitoring data to estimate inshore areas' exposure to river end-of-catchment loads across all Reef catchment regions.

The annual Water Quality condition Index for 2024–25 (1 October 2024 to 30 September 2025) as reported by Gruber *et al.* (2026) was:

- **'good'** in the Cape York region, representing improvement compared with the previous year's 'moderate' score following Cyclone Jasper;
- **'moderate'** in the Wet Tropics region, following 6 years of 'good' scores;
- **'good'** in the Burdekin region, similar to 2023–24;
- **'moderate'** in the Mackay-Whitsunday region following seven years of 'moderate' scores; and
- **'good'** in the Fitzroy region, similar to the previous 4 years.

### 3.6 Turbid water exposure and flood plume extent

The frequency and extent of exposure to wet season water types, along with their deviation from long-term averages, are summarised in Figure 7. Turbid, coloured waters (WT1 and WT2) dominated the inshore Reef during the 2024–25 wet season (December 2024 – April 2025), consistent with long-term inshore conditions (2003–2019, Figure 8a). WT1 and WT2 extended further offshore than the long-term average in the Wet Tropics, Burdekin, and Mackay–Whitsunday regions (Figure 8a), while their extent in Cape York, Fitzroy, and Burnett–Mary regions was close to average, varying locally within each region. Exposure of seagrass monitoring sites to WT1 was slightly above average in Cape York and the Wet Tropics (Figure 9a), but combined exposure to WT1 + WT2 was near multiannual averages (Figure 9b), reflecting the typically inshore location of monitoring sites frequently exposed to turbid conditions. Cloud cover in 2024–25 limited satellite observations, likely underestimating true exposure frequency.

In 2024–25, pesticide concentrations from Passive samplers near Wet Tropics seagrass sites were lowest at Low Isles. In contrast, Dunk Island recorded the highest total  $\Sigma$ pesticide concentration (81 ng L<sup>-1</sup>, January 2025), as well as the highest Diuron concentration among all Reef sites assessed (47 ng L<sup>-1</sup>) (Gruber *et al.* 2026). Grab samples collected from plumes within 1–5 km of seagrass sites detected Diuron concentrations well below the 270 ng L<sup>-1</sup> ANZECC and ARMCANZ guidelines. Overall, the Pesticide Risk Metric (PRM) for PSII herbicides was low, with all samples well below the 1% of species affected threshold (Gruber *et al.* 2026). The highest PSII PRMs were 0.4% and 0.14% of species affected, detected in waters in the vicinity of Repulse Bay and Dunk Island seagrass monitoring sites, respectively, associated with discharges in February 2025 (Gruber *et al.* 2026).

At the Reef scale, the area of seagrass exposed to water quality risk categories II–IV was near the long-term average (Gruber *et al.* 2026). Regionally, exposure patterns varied: in Cape York, exposure matched long-term trends; in the Burdekin, the total area remained similar but shifted toward higher-risk categories (more category III, less category II). The Mackay–Whitsunday region showed a slight reduction in total exposure area but a comparable shift toward higher-risk categories. Conversely, in the Fitzroy region, total exposure increased but shifted toward lower-risk categories (Gruber *et al.* 2026).

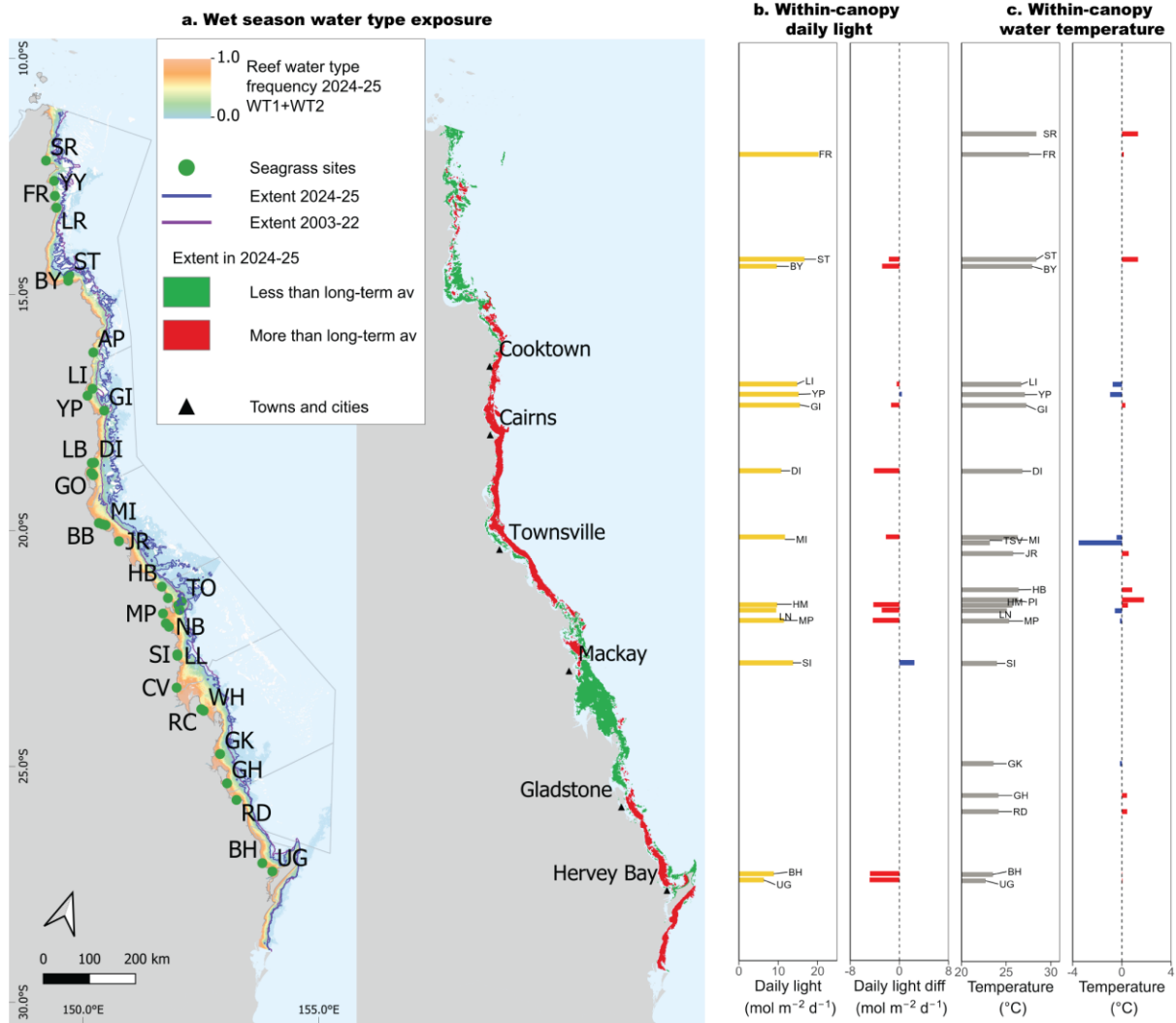


Figure 8. Environmental pressures in the Reef during 2024–25 and relative to long-term: a. Frequency of turbid water (WT1 and WT2) exposure extent shown in the left-hand panel in the Reef from December 2024 to April 2025 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed), and right-hand panel the distribution of WT1 and WT2 (10 per cent boundary) in 2024–25 relative to the long-term average, with red showing that that these water types extended further in 2024–25 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. average within canopy water temperature, and deviation from the long-term average. Panels a and b present data sourced from Gruber *et al.* 2026.

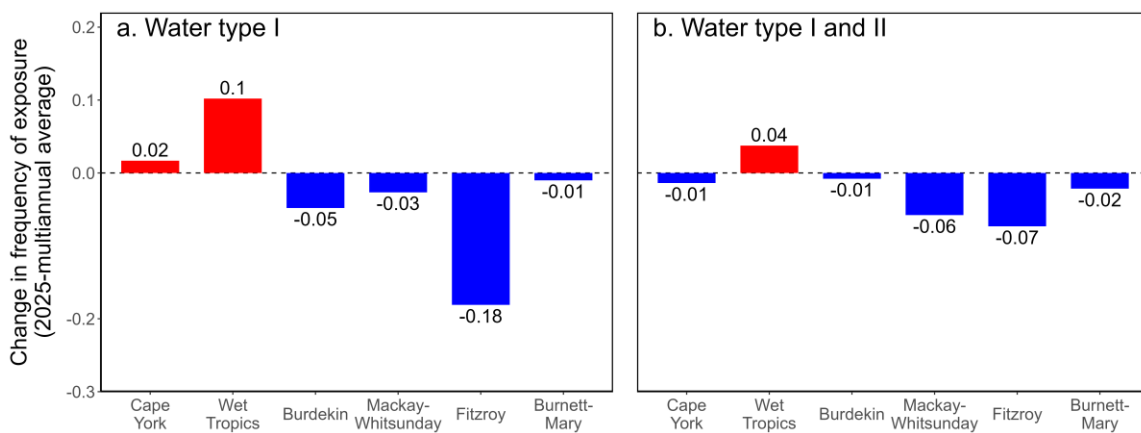


Figure 9. Difference in the frequency of exposure to primary (WT1, left) and primary and secondary optical water types (WT1 and WT2, right) at seagrass monitoring sites during the wet season (December 2023–April 2024) compared to the long-term multiannual exposure (2003–2022). Data sourced from Gruber *et al.* 2026.

### 3.7 Daily light

Daily light reaching the top of the seagrass canopy across the Reef in 2024–25 was  $12.4 \text{ mol m}^{-2} \text{ d}^{-1}$  when averaged for all sites (Table 10), compared to a long-term average of  $13.6 \text{ mol m}^{-2} \text{ d}^{-1}$ . Daily light availability in seagrass habitats is influenced by site depth, cloud cover, and water quality, which are affected by river discharge and sediment resuspension driven by wind and currents (Anthony *et al.* 2004; Fabricius *et al.* 2012). Tidal fluctuations also influence the duration and frequency of full sunlight exposure, particularly at low tide in intertidal sites—the primary focus of this monitoring program. Consequently, variations in daily light among seagrass sites mainly reflect site-specific differences in water quality and cloud cover; however, differences in the level and frequency of intertidal exposure among sites may also contribute to observed variability.

Regional averages in daily light are also affected by data availability, as substantial logger failures occurred at many monitoring sites in 2024–25 (9 of 23 locations with <50% data available for the period), resulting in their exclusion from analysis. For instance, in the Burdekin region, only reef habitat sites (MI1) provided sufficient data, while the typically more turbid coastal sites did not, leading to an apparent regional increase in light levels. Detailed data for each site are provided in the appendices (Figures 98–104).

Daily light in the regions in 2024–25 from north to south were ( $\downarrow$  = lower than,  $\uparrow$  = greater than the long-term,  $\ddagger$  = similar to long-term i.e.  $<0.5 \text{ mol m}^{-2} \text{ d}^{-1}$  difference):

- Cape York (15.5 mol m<sup>-2</sup> d<sup>-1</sup>)  $\downarrow$
- northern Wet Tropics (15.2 mol m<sup>-2</sup> d<sup>-1</sup>)  $\downarrow$
- southern Wet Tropics (10.8 mol m<sup>-2</sup> d<sup>-1</sup>)  $\downarrow$
- Burdekin (11.7 mol m<sup>-2</sup> d<sup>-1</sup>)  $\uparrow$  \*
- Mackay–Whitsunday (11.1 mol m<sup>-2</sup> d<sup>-1</sup>)  $\downarrow$
- Fitzroy (*insufficient data*)
- Burnett–Mary (7.6 mol m<sup>-2</sup> d<sup>-1</sup>)  $\downarrow$

\*please note, Burdekin data only for reef intertidal habitat

Daily light in the habitats in 2024–25 from highest to lowest were ( $\downarrow$  = lower than,  $\uparrow$  = greater than,  $\ddagger$  = similar to long-term i.e.  $<0.5 \text{ mol m}^{-2} \text{ d}^{-1}$  difference):

- reef intertidal, n = 9 (13.4 mol m<sup>-2</sup> d<sup>-1</sup>)  $\downarrow$
- coastal intertidal, n = 4 (11.3 mol m<sup>-2</sup> d<sup>-1</sup>)  $\downarrow$
- estuarine, n = 3 (10.0 mol m<sup>-2</sup> d<sup>-1</sup>)  $\downarrow$

Average daily light levels follow a gradient increasing from inshore to offshore: reef intertidal sites have the highest daily light levels followed by coastal intertidal and estuarine intertidal sites. Daily light for each of the sites is presented in Figure 8b. The annual daily light level was lower than the long-term average at most sites (Figure 8), however, there was a lot of information not recorded in 2024–25 due to instrument failure or loss. Low light occurred throughout the Reef in all NRM regions but the anomalies in Cape York were largely attributed to missing data. There was insufficient data in the Fitzroy region to include in these anomalies. There were very low light levels in the regions affected by elevated discharge in 2024–25, but also in the most southern region, where there had been elevated discharge in recent years. The largest anomalies were at:

- Dunk Island, DI1 (southern Wet Tropics) where light was low in the dry and wet season, and light has been low for 3 consecutive years
- Magnetic Island, MI1 (Burdekin), where light was low during the wet season and MI2 where light was very low during the 2024 dry season (wet season data are missing)
- Midge Point, MP2 (Mackay–Whitsundays) where light was very low in the wet season
- Burrum Heads, BH1 and Urangan, UG2 in the Burnett–Mary where light was very low in the dry and wet season.

Long-term patterns indicate that within-canopy daily light typically peaks between September and December, coinciding with maximum incident solar radiation prior to the onset of wet-season conditions (Figure 10). This period also aligns with the peak seagrass growth season and, consequently, the predominant sampling window for this monitoring program. Over the duration of the dataset, the magnitude of this annual light peak appears to have progressively declined, with the 2024–25 dry season representing one of the lowest peaks recorded.

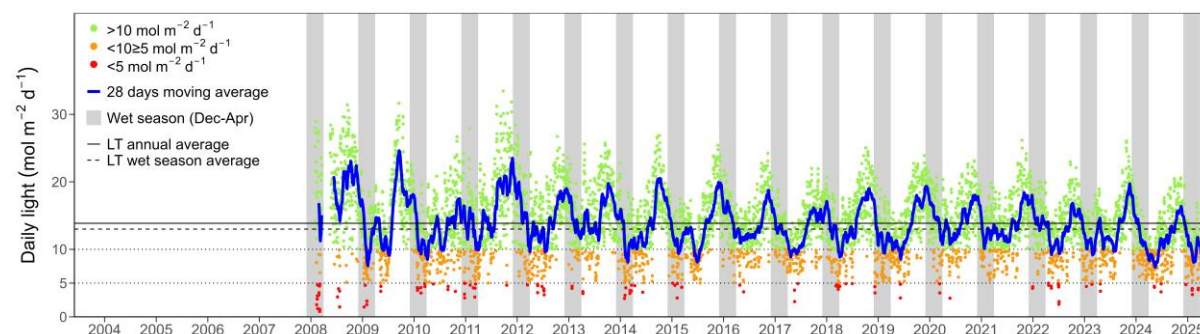


Figure 10. Daily light (coloured points) and 28-days moving average (blue, bold line) for all sites combined from 2008 to 2024. In 2008–2009, light data is from the Burdekin and Wet Tropics regions only. Other regions were included from 2009–2010, with Cape York added post 2012–2013 reporting period. Shaded vertical bars indicate the wet season months (December to April) used for analysis of wet season optical water types in Gruber *et al.* (2026). The solid horizontal line indicates the long-term Reef average, and the dashed line indicates the wet season long-term Reef average. Dotted lines are for visual reference and indicate an approximate short-term light threshold ( $5 \text{ mol m}^{-2} \text{ d}^{-1}$ , NB  $6 \text{ mol m}^{-2} \text{ d}^{-1}$  may also be used as a management threshold) with red points being values below the line and long-term light threshold ( $10 \text{ mol m}^{-2} \text{ d}^{-1}$ ) with orange points showing values below it (Collier *et al.* 2016).

During the wet season, tidal cycles rarely expose seagrass meadows to low-tides (shallower waters) where high-irradiance conditions occur during daylight hours, contributing to the seasonal decline in available light (Figure 10). In 2024–25, daily light levels decreased rapidly with the onset of the wet season and remained below the long-term wet-season average. Light availability was also lower than average during the late wet season and immediate post-wet season period.

### 3.8 Within-canopy seawater temperature

Daily within-canopy seawater temperature across the inshore Reef in 2024–25 was similar to the previous reporting period (Figure 11). Since 2013, the frequency of weekly warm water deviations appears to have generally increased, relative to cooler occurrences (Figure 11). The 2024–25 average temperature ( $25.9 \pm 0.3 \text{ }^\circ\text{C}$ ) was the fourth consecutive year of above average temperatures and the sixth highest since the MMP was established (2016-17 was the highest) (Table 10). However, there were regional and habitat differences relative to the long-term (Figure 8).

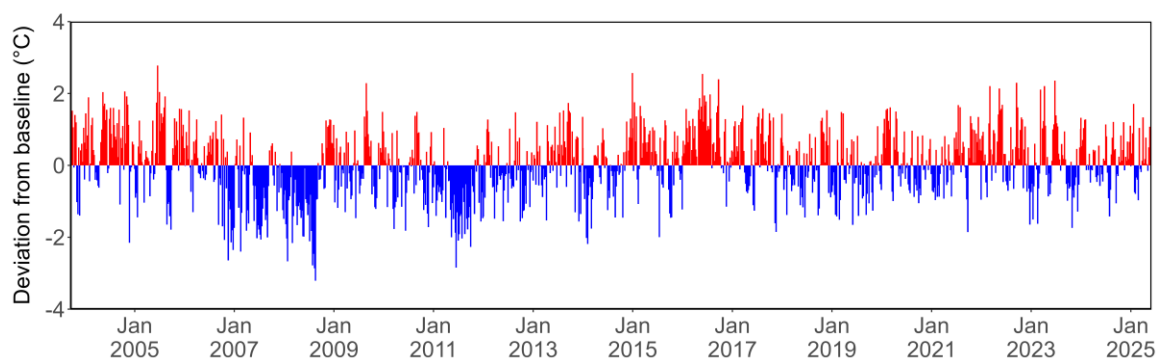


Figure 11. Inshore intertidal sea temperature deviations from baseline for Reef seagrass habitats from 2003 to 2025. Data presented are deviations from 19-year mean weekly temperature records (based on records from September 2003 to June 2023). Weeks above the long-term average are represented as red 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 2024–25 (including number of days above 35°C and 40°C) from north to south as difference relative to the long-term average (↑ = above, ↓ = below, † = similar to long-term, difference = greater than 0.3 °C) were:

- Cape York (avg = 27.5°C, max = 38.3°C, days<sub>>35≤40°C</sub> = 37) †
- northern Wet Tropics (avg = 27.3°C, max = 41.2°C, days<sub>>35≤40°C</sub> = 53, days<sub>>40°C</sub> = 1) †
- southern Wet Tropics (avg = 26.8°C, max = 35.5°C, days<sub>>35≤40°C</sub> = 1) †
- Burdekin (avg = 26.3°C, max = 37.7°C, days<sub>>35≤40°C</sub> = 26) †
- Mackay–Whitsunday (avg = 25.8°C, max = 42.4°C, days<sub>>35≤40°C</sub> = 55) †
- Fitzroy (avg = 24.3°C, max = 40.0°C, days<sub>>35≤40°C</sub> = 19, days<sub>>40°C</sub> = 1) † \*
- Burnett–Mary (avg = 23.8°C, max = 38.7°C, days<sub>>35°C</sub> = 3) †

\*please note, Shoalwater Bay annual deployment not retrieved at time of publication

Daily within-canopy seawater temperatures in each habitat in 2024–25 relative to respective long-term average (↑ = above, ↓ = below, † = similar to long-term, difference = greater than 0.3°C) were:

- estuarine habitat (avg = 24.2°C, max = 38.7°C) †
- coastal intertidal habitat (avg = 26.1°C, max = 40.0°C) †
- reef intertidal habitat (avg = 26.4°C, max = 41.2°C) †

The highest seawater temperature recorded at inshore seagrass sites along the Reef during 2024–25 reached 41.2°C at Low Isles (28 April 2025 at 4:00pm) located in the northern Wet Tropics region. The Fitzroy region experienced the most days exceeding 35°C, closely followed by the northern Wet Tropics (Figure 12). During 2024–25, extreme temperatures above 40°C were recorded only twice: once in October 2024 at Great Keppel Island in the Fitzroy region, and again in April 2025 at Low Isles. Although extreme temperature days can lead to photoinhibition, their low frequency during this period is unlikely to have resulted in burning or mortality.

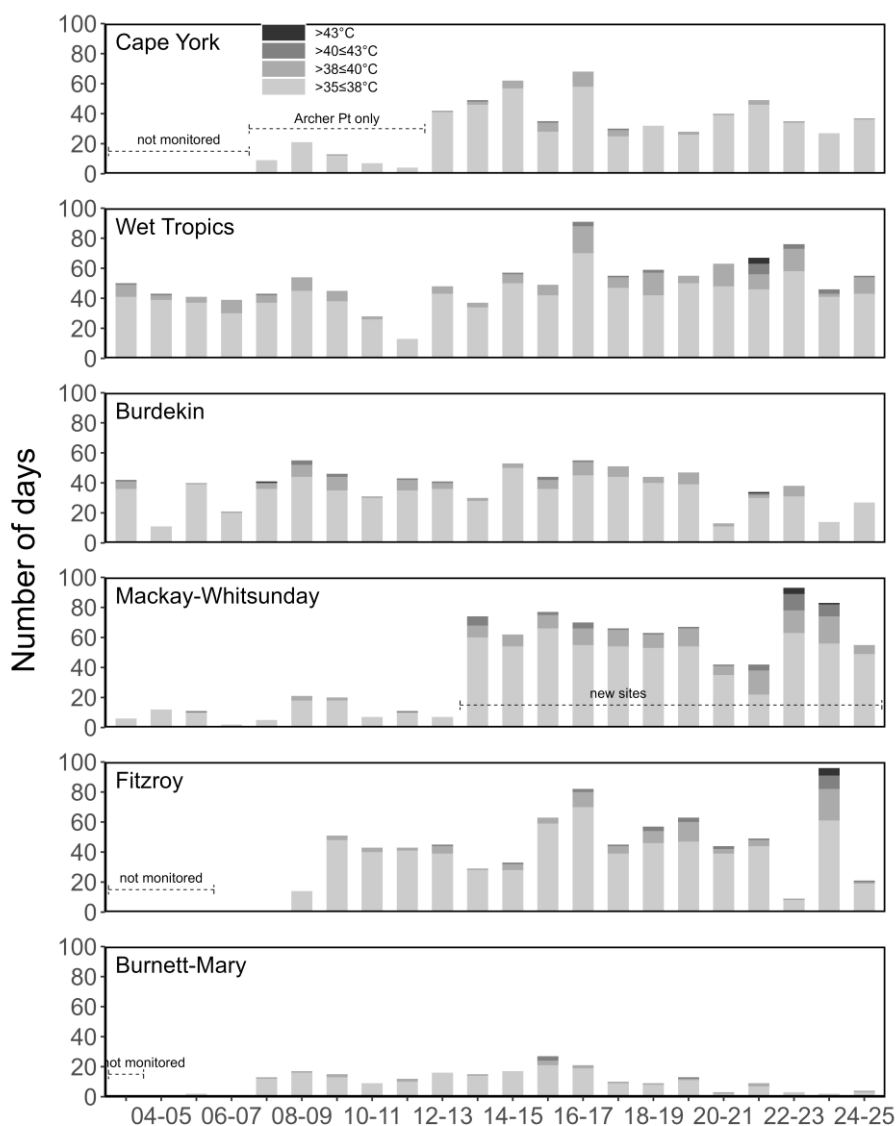


Figure 12. Number of days when inshore intertidal sea temperature exceeded 35°C, 38°C, 40°C and 43°C in each monitoring period in each NRM region. Thresholds adapted from Campbell *et al.* 2006; Collier *et al.* 2012a.

### 3.9 Seagrass meadow sediments

Estuarine and coastal subtidal seagrass habitats across the Reef had a greater proportion of fine sediments (i.e. mud) than other habitats (Table 12). Sediments at intertidal coastal habitats were predominately fine and medium sands, while reef habitats (intertidal and subtidal) were dominated by medium sands (Table 12).

Table 12. Long-term average ( $\pm$ SE) sediment composition for each seagrass habitat (pooled across regions and time) monitored within the Reef (1999–2024). \*only 7 years of data.

Habitat	Mud	Fine sand	Sand	Coarse sand	Gravel
estuarine intertidal	50.1 $\pm$ 2.3	28 $\pm$ 2.4	21.3 $\pm$ 1.6	0.5 $\pm$ 0.9	0.2 $\pm$ 0.3
coastal intertidal	29 $\pm$ 1.7	42.5 $\pm$ 2.6	23.7 $\pm$ 2.2	0.7 $\pm$ 1.1	4 $\pm$ 0.9
coastal subtidal*	41.3 $\pm$ 0.4	0 $\pm$ 0	25.6 $\pm$ 12.9	15.7 $\pm$ 13.1	17.4 $\pm$ 0.7
reef intertidal	3.3 $\pm$ 0.4	7.5 $\pm$ 1.8	35.1 $\pm$ 1.8	24.4 $\pm$ 2	29.7 $\pm$ 0.9
reef subtidal	19.8 $\pm$ 0.6	3.1 $\pm$ 3.1	38.2 $\pm$ 0	16.9 $\pm$ 0.2	21.9 $\pm$ 0.4

Throughout the 2024–25 monitoring period, the contribution of mud to sediment type decreased in all seagrass habitats compared to the previous year, with proportions in estuarine intertidal, coastal intertidal and reef subtidal habitats continuing to surpass the long-term average for the Reef (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 seagrass species, the elevated sulphide 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 (Ferguson *et al.* 2016).

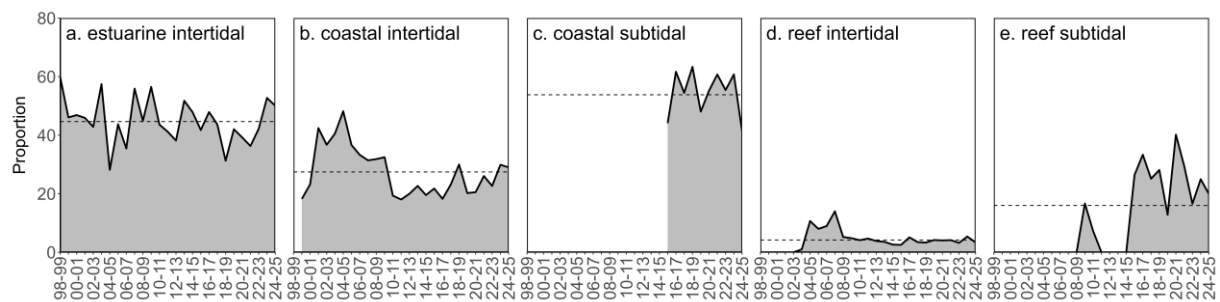


Figure 13. Proportion of sediment composed of mud (grain size  $<63 \mu\text{m}$ ) at inshore Reef seagrass monitoring habitats from 1999–2025. Dashed line illustrates the Reef long-term average for each habitat type.

## 4 Seagrass condition and trend

The following section provides detail on the overall seagrass responses for the 2024–25 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 are detailed within Appendix 2 and 3. Detailed site specific data can be accessed on the Seagrass-Watch Global Seagrass Observing Network official website at <https://bit.ly/3THVNUd>. Seagrass condition trends can also be accessed with water quality and coral condition results at the Reef Knowledge System MMP dashboard at <https://bit.ly/4aGrG5A>.

### 4.1 Overall inshore Reef seagrass condition and trend

Inshore seagrass meadows across the Reef deteriorated in overall condition in 2024–25, with the condition grade declining to **‘poor’** for the first time in 5 years (Figure 14). Cape York and the Fitzroy regions are assessed only during the late dry season, which was prior to wet season conditions of the current reporting period.

In summary, both the seagrass abundance and resilience indicators decreased:

- During the current 2024–25 period, abundance deteriorated a grade, based on the average score against the guidelines (determined at the site level), and inshore seagrass across the Reef was rated in a ‘poor’ state. Seagrass abundance has fluctuated temporally at meadows monitored in the MMP over the life of the program, displaying periods of decline and variable recovery (Figure 14). The largest declines occurred from 2009 to 2012, caused by consecutive years of above-average rainfall, and resultant discharges of poor quality water, (Figure 14, Figure 16b). Following 2012, seagrass recovery proceeded for 5 years until stalling in 2016–17 as a result of regional climatic events, after which abundances subsequently declined. From late 2020, seagrass abundances improved, although recovery appeared somewhat muted, stalling in a ‘moderate’ state with a slight decline in 2023–24.
- The resilience indicator declined in 2024–25 to ‘poor’ (Figure 14), which was the lowest level since 2012–13 when it was recovering from extreme weather events. The long-term trend in the resilience indicator is similar to the abundance indicator with large declines from 2009 to 2012 due to extreme weather events recovering to ‘good’ in 2016–17. Resilience has been on a declining trend since 2016–17, though with a two-year reprieve when it rose slightly.

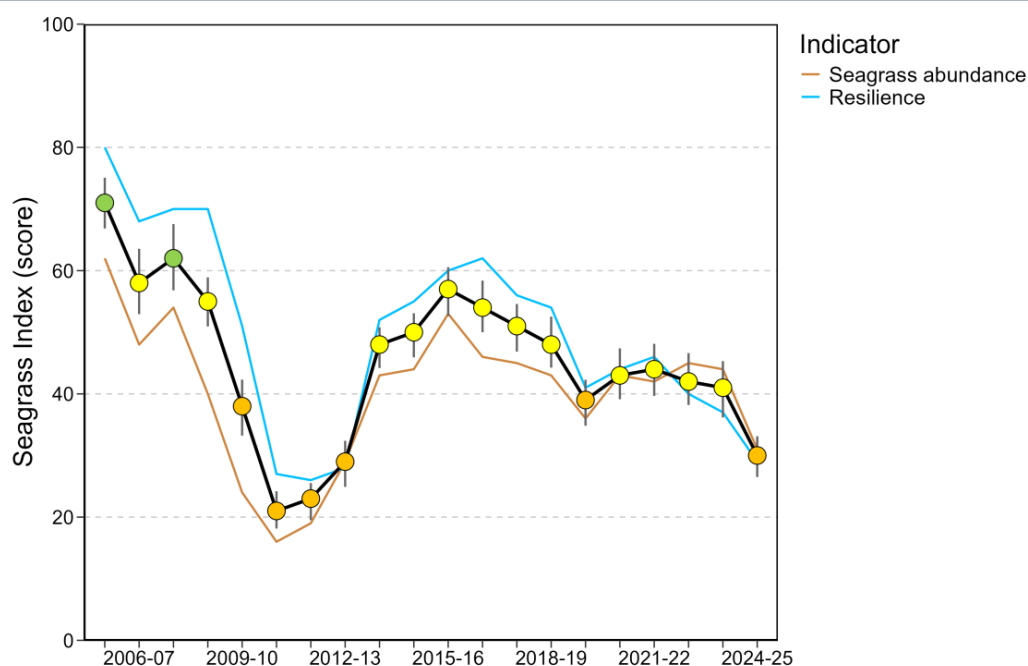


Figure 14. Overall inshore Reef Seagrass Index ( $\pm$  SE) with contributing indicator scores over the life of the MMP. The Index is derived from the aggregate of regional scores for indicators of seagrass condition: abundance and resilience. Regional scores are weighted on the percentage of GBRWHA seagrass area (shallower than 15 m) within respective regions and 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 6 NRM regions, weighted by inshore seagrass area. In 2024–25 the Index improved in 2 regions, declined in 3 and remained relatively stable in one (Figure 15). The seagrass Index improved to ‘moderate’ and ‘good’ in Fitzroy and Burnett–Mary regions (Figure 15). Improvements in the northern Wet Tropics, however, were offset by the declines in the southern Wet Tropics, resulting in no significant change overall and the regional Index remaining ‘poor’. The Index declined from ‘moderate’ to ‘poor’ in Cape York, while in the Burdekin and Mackay–Whitsunday regions the index also experienced declines, but remained ‘poor’ and ‘moderate’, respectively. It was the first time the Index in Cape York was ‘poor’ since monitoring was established. Over the long-term, the abundance and resilience 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 6 regions:

- The seagrass abundance score in 2024–25 generally followed the same trends as the overall Index. Abundance decreased greatly in Cape York (measured in the late dry, before the wet season) and increased greatly in the Fitzroy and Burnett–Mary regions. There were substantial declines in abundance from ‘moderate’ to ‘poor’ in the Burdekin and Mackay–Whitsunday regions and a marginal increase in the Wet Tropics region.
- The seagrass resilience score in 2024–25 increased in the Burnett–Mary and Fitzroy regions, was stable in the Mackay–Whitsunday region and declined in Cape York, the Wet Tropics, and Burdekin regions. In 2024–25, the resilience score trends also paralleled the overall Index except in the Mackay–Whitsunday region where there was a decline in the abundance score and overall Index, but resilience was stable.

Inshore seagrass condition scores across the regions reflect a system that is being impacted by elevated discharge from rivers, heatwaves, cyclones and local-scale disturbances.

Regional differences in condition and indicator scores appear due to the legacy of significant environmental conditions. These include: Cyclone Debbie in 2016–17, which impacted Mackay–Whitsunday; above-average riverine discharge affecting the southern and central Reef in 2016–17 and the Burdekin region in 2018–19; and a marine heatwave that occurred in the northern and central Reef in 2018–19. There had also been local-scale changes influencing regional scores, particularly in the Fitzroy region, but these influences appear to have abated. In 2024–25, extreme elevated discharge affected the Burdekin region, and to a lesser degree the southern Wet Tropics and Mackay–Whitsunday regions.

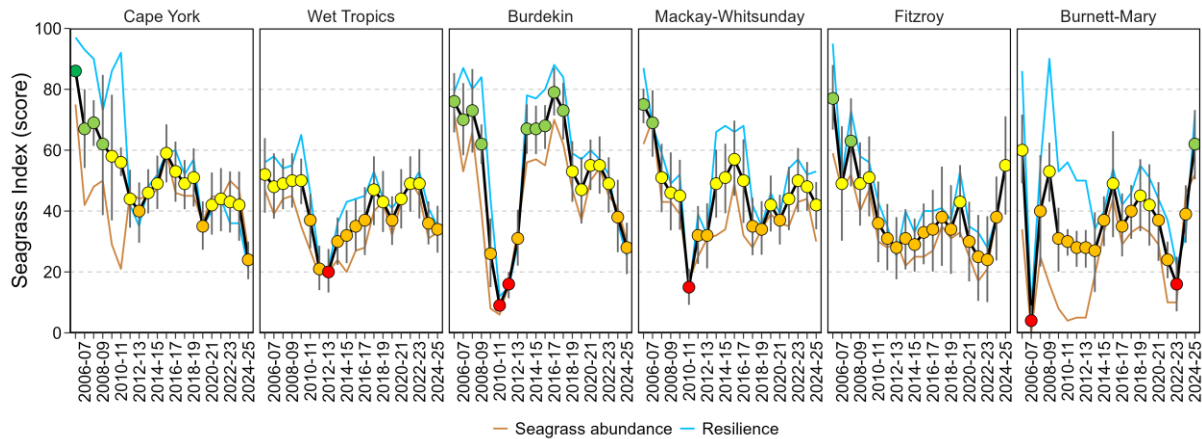


Figure 15. Seagrass Index ( $\pm$  SE) with contributing indicator scores for each NRM region over the life of the MMP. The Index is derived from the aggregate of metric scores for indicators of seagrass condition: abundance and resilience. Values are indexed scores scaled from 0–100 and graded: ● = 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 calculate 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 (per cent cover) has varied over decadal time-scales, declining in the 2009–10 through 2011–12 monitoring periods, then recovering to some extent until 2023, depending on region (Figure 16). From 2019, there had been a noticeable divergence in regional abundances between the northern and southern Reef. Notably, the southern regions (Fitzroy and Burnett–Mary) experienced declines, which were somewhat offset by improvements in the northern regions. However, since 2023, while the southern regions have made significant recoveries, the overall trend has shifted toward decline, primarily driven by the Mackay–Whitsunday, Burdekin, and Cape York regions (Figure 16).

The resilience metric has similarly varied over decadal time-scales, declining to its lowest levels in the 2010–11 through 2012–13 monitoring periods. The resilience score has been on a declining trajectory since 2016–17 and influenced heavily by large changes in the Cape York and Burdekin regions, and smaller fluctuations in other regions.

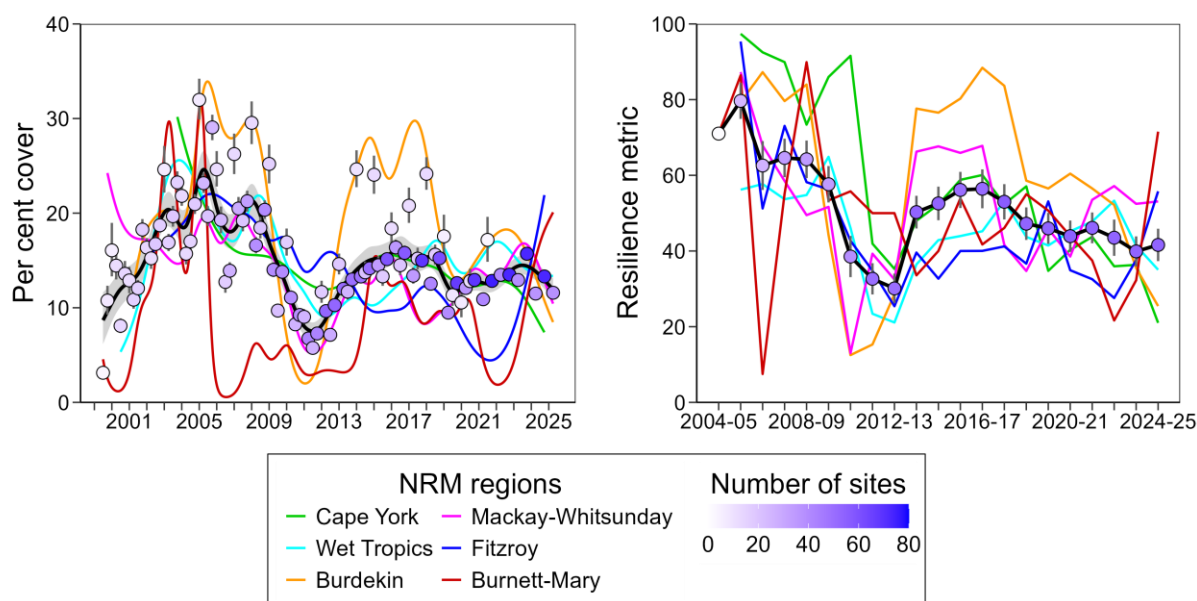


Figure 16. Trends in the seagrass indicators used to calculate the Seagrass Index including trends in Reef seagrass abundance (per cent cover,  $\pm$  SE) represented by a GAM plot (black line with shaded areas defining 95 per cent confidence interval), and coloured lines representing NRM trends (left), and trends in Reef resilience metric (black line and circles,  $\pm$  SE) and coloured lines represent trends in NRM resilience metric (right). Circle colour relates to number of sites assessed. Please note: GBR Reef resilience metric used is un-weighted.

### 4.3 Trends in seagrass condition indicators by habitat type

#### 4.3.1 Seagrass abundance, composition and extent

Seagrass abundance across the inshore Reef has varied greatly since monitoring began. A review of long-term data from inshore Reef sites shows no significant overall trend, with:

- no significant trends at 75 per cent of long-term monitoring sites assessed, although 6 per cent of sites significantly increased in abundance and 19 per cent decreased (Appendix 3, Table 22),
- the rate of change in abundance was lower for sites that were increasing ( $0.1 \pm 1.48$  per cent, per sampling event) compared to those decreasing ( $-1.4 \pm 0.6$  per cent per sampling event) (Appendix 3, Table 22).

Since 1999, the median percentage cover values for the Reef were mostly below 25 per cent cover, and depending on habitat, the 75<sup>th</sup> 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 around 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 abundance score thresholds.

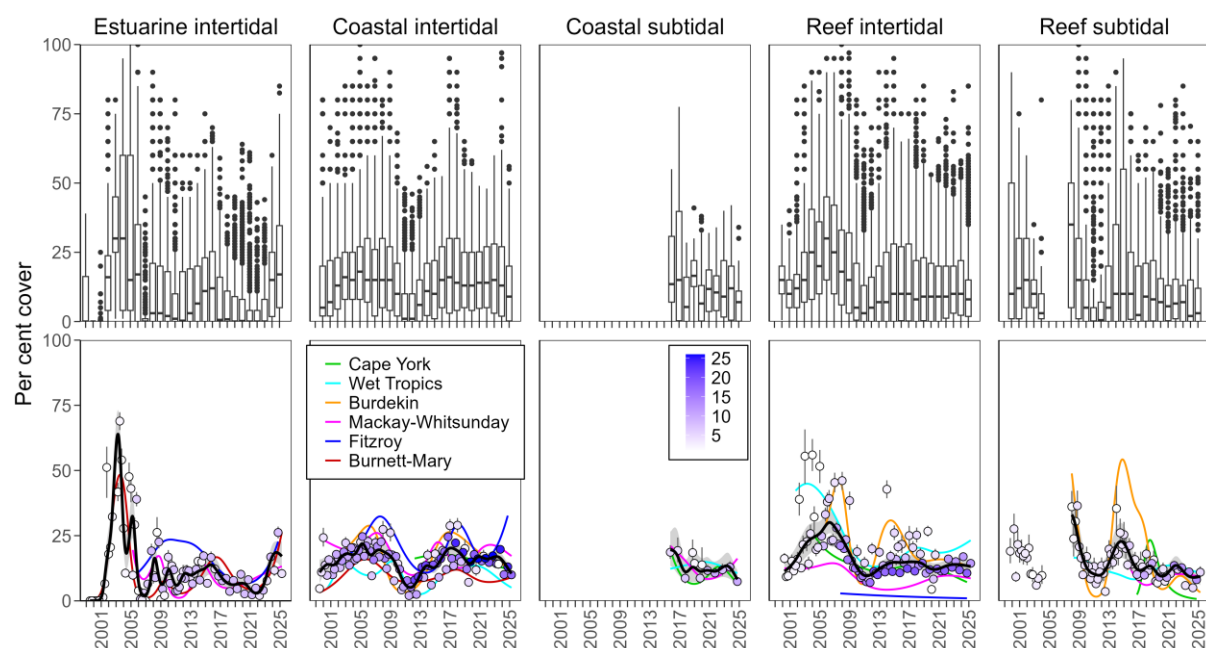


Figure 17. Seagrass per cent cover measures per quadrat from habitats monitored from June 1999 to May 2025 (sites pooled). In the whisker plots (top), the box represents the interquartile range of values, where the boundary of the box closest to zero indicates the 25<sup>th</sup> percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75<sup>th</sup> percentile. Whiskers (error bars) above and below the box indicate the 90<sup>th</sup> and 10<sup>th</sup> 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 95<sup>th</sup> 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 2024–25, estuarine intertidal habitats had the highest average seagrass abundance of all habitat types assessed, while reef subtidal habitats had the lowest (Figure 17). Estuarine habitats, which have been monitored solely in the central and southern NRM regions (Mackay–Whitsunday, Fitzroy, and Burnett–Mary), have experienced location-specific fluctuations over the past decade. These changes have been primarily driven by acute events, such as sediment deposition, reduced light availability from discharge events, and sediment movement caused by climatic pressures. Since 2016, seagrass abundances have progressively declined, reaching their lowest levels during 2021–22 (Figure 17). However, with the onset of recovery in 2022–23, seagrass abundance in estuarine habitats has continued to rise, reaching the highest levels observed since 2006–07. This resurgence is largely attributed to recovery in the Burnett–Mary and Fitzroy regions.

Over the past decade, patterns in seagrass abundance have shown similarities between intertidal locations in coastal and reef habitats. Following the losses from extreme weather events of early 2011 (e.g., Cyclone Yasi and associated flooding), abundance gradually increased until 2017. However, from 2017 to 2019, abundances declined due to the impacts of Cyclone Debbie. Afterward, abundances improved with comparable levels across intertidal habitats in both coastal and reef habitats, as well as in coastal subtidal habitats (Figure 17). However, in 2024–25, abundances subsequently declined (Figure 17).

During the 2024–25 period, the overall relative spatial extent of inshore Reef seagrass meadows did not experience the same seasonal expansion during the late dry season as it did in the previous period, when it achieved its highest recorded extent (Figure 18). During the following wet season, the relative spatial extent dropped as seasonally anticipated; however, it remained greater than the same period from the prior year. Initially, in the first few years after the establishment of the MMP in 2005, the extent of seagrass meadows remained relatively stable. However, from 2008 to early 2011, a gradual decline became evident. This was succeeded by a recovery period lasting 3–4 years, after which another decline occurred was observed from late 2016 to early 2019, before once again entering a

recovery period (Figure 18). As with seagrass abundance, these declines in relative extent are primarily linked to extreme weather events, associated flooding, and location-specific climatic factors such as the frequency of strong wind days and dynamic processes across intertidal banks.

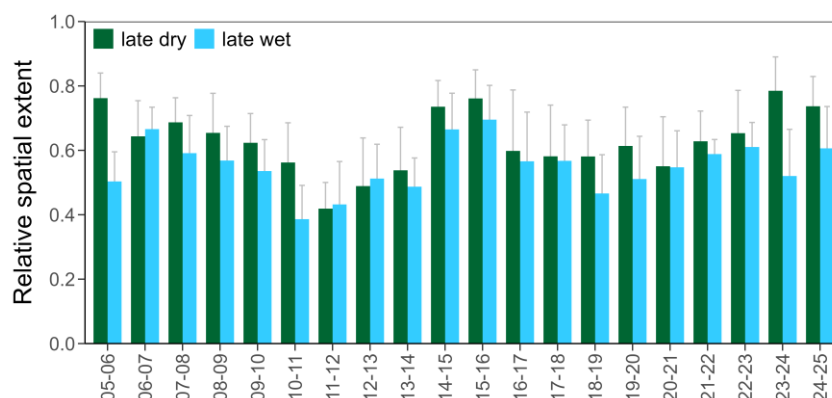


Figure 18. Average relative spatial extent of seagrass distribution at monitoring sites across inshore Reef (locations, habitats and NRM regions pooled, + SE). Green bars represent late dry and Blue bars late wet.

Following a sequence of above-average wet seasons that began in 2009 and culminated in the extreme weather events of 2011, seagrass meadows experienced widespread declines in both extent and abundance, accompanied by an increase in the colonising species *Halophila* spp. at coastal and reef sites (Figure 19). From 2012–13 to 2015–16, the prevalence of colonising species declined, before increasing again in 2016–17 and fluctuating thereafter. In 2024–25, the proportion of colonising species decreased in estuarine and coastal intertidal habitats, falling below the reef-wide average, and also declined in reef intertidal and subtidal habitats where proportions were near long-term averages. In contrast, coastal subtidal habitats had a higher mean proportion of colonising species (40%), driven by consistently high values across most sites except MI2 (Missionary Bay, Wet Tropics) and Newry Bay (Mackay–Whitsunday).

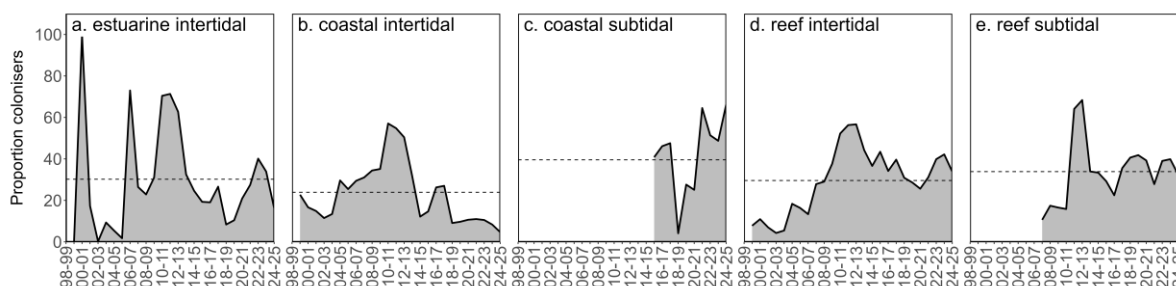


Figure 19. Proportion of total seagrass abundance composed of species displaying colonising traits (e.g. *Halophila ovalis*) in: a) estuarine intertidal, b) coastal intertidal, c) coastal subtidal, d) reef intertidal and e) reef subtidal habitats (sites pooled) of the Reef (regions pooled) for each monitoring period. Dashed line illustrates Reef average proportion of colonising species in each habitat type.

### 4.3.2 Seagrass reproductive status

Reproductive effort—measured as the number of sexual reproductive structures per unit area—and seed banks—representing intact seeds per unit area—indicate the reproductive capacity of inshore seagrasses. Patterns of maximum reproductive effort and interannual variability differ among habitats and between years. Reproductive effort across inshore meadows is typically highest in the late dry season, while seed densities fluctuate less seasonally due to seed longevity (Figures 19–20). Reproductive structures decline with

distance from shore, being most abundant in estuarine and coastal habitats and least at offshore reef sites.

In 2024–25, reproductive effort remained low or declined across most habitat types (Figure 19). Estuarine meadows were the exception, recording the third-highest dry-season levels on record—well above Reef-wide averages—though wet-season effort was very low, consistent with the reproductive timing of *Zostera muelleri*. Seed densities followed a similar pattern, reaching the third-highest recorded values before declining sharply in the wet season, largely influenced by trends in the Burnett–Mary region.

Coastal habitats recorded their sixth consecutive year of low reproductive effort, well below the historically high levels from 2016–17 to 2018–19. This pattern was consistent across regions, except in the Fitzroy, which recorded its second-highest reproductive effort on record, and the Mackay–Whitsunday, which showed near-average levels. Seed bank density continued to decline in coastal meadows, reaching the second-lowest wet-season value on record, with low levels were driven mainly by reduced reproduction and seed stocks in the Burdekin region. The apparent wet-season decline was also influenced by no monitoring occurring in Cape York and the Fitzroy regions in the late wet season (Figure 20).

Reef habitats had the lowest reproductive effort and seed densities of all habitat types (Figure 20 Figure 21), reflecting the dominance of persistent species such as *Thalassia hemprichii*, which do not form seed banks, and low reproduction rates among foundational taxa (*Halodule uninervis*, *Cymodocea rotundata*, *C. serrulata*). In 2024–25, reproductive effort decreased further across reef intertidal meadows, with limited reproductive structures observed in the dry season and none in subtidal habitats during the wet season. The southern Wet Tropics was the only region with substantial dry-season reproduction and seed presence, while the Burdekin was the only region with evidence of late wet-season reproduction.

Overall, reproductive structures were recorded at only 19 of 47 sites assessed in 2024–25. The largest declines compared to 2023–24 occurred in the Burdekin and northern Wet Tropics—where flowers were limited to colonising species—while increases were observed in the southern Wet Tropics, Fitzroy, and Burnett–Mary regions. Declines in seed density likely reflect reduced sexual reproductive output and success (e.g. seed failure) or loss of seed banks through germination or grazing, indicating potential vulnerability to future disturbance, though the seed density required to support recovery remains unknown.

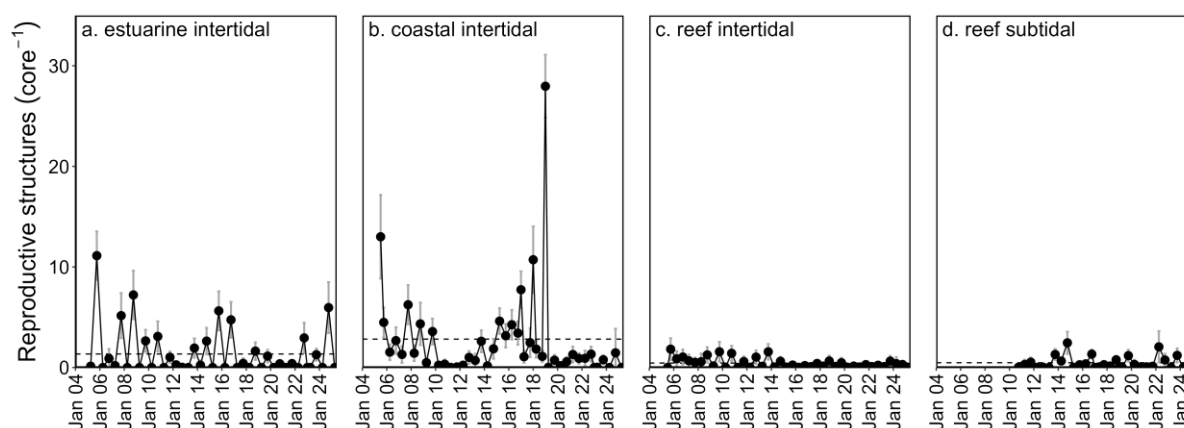


Figure 20. Seagrass reproductive effort (number of reproductive structures produced by all seagrass species,  $\pm$  SE) in Reef seagrass habitats for a) estuarine intertidal; b) coastal intertidal; c) reef intertidal; d) reef subtidal. Dashed line illustrates Reef long-term average reproductive effort in each habitat type.

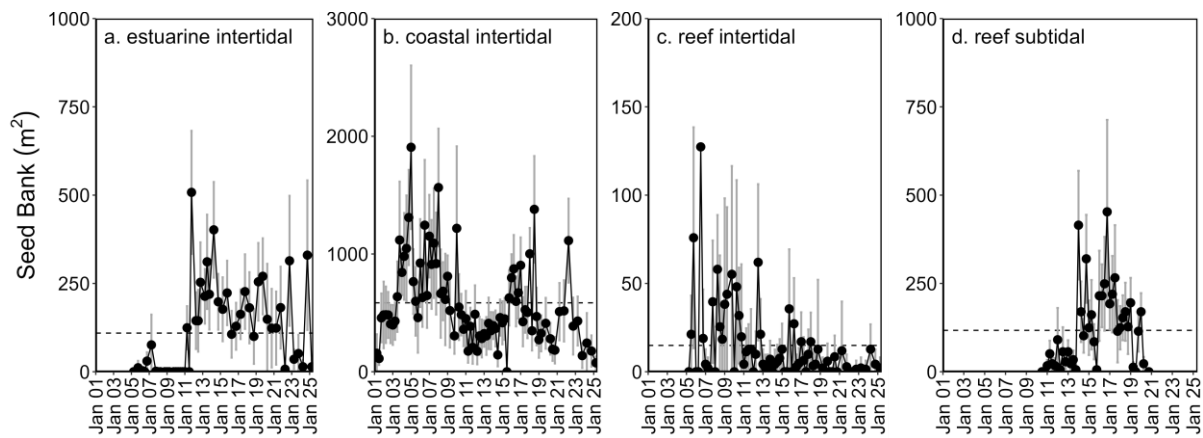


Figure 21. Average seeds banks (seeds per square metre of sediment surface, all sites and species pooled, ± SE) in Reef seagrass habitats: a) estuarine intertidal; b) coastal intertidal; c) reef intertidal; d) reef subtidal.

### 4.3.3 Resilience

The resilience score improved substantially in estuarine habitats, remained largely unchanged in reef intertidal and reef subtidal habitats, but declined in coastal intertidal habitats (Figure 22, Table 23). However, the trend for the habitat types varied among regions. The large improvement in estuarine habitats was driven by the Burnett–Mary and Fitzroy, but not the Mackay–Whitsunday NRM region, which was relatively stable. There are no estuarine sites in the other NRM regions (Figure 22, Table 23).

Resilience in coastal habitats has deteriorated in the majority of NRM regions, in particular Cape York, Wet Tropics, Burdekin and Burnett–Mary. The largest declines occurred in Cape York and the northern Wet tropics, triggered by seagrass abundance falling below resilience thresholds. Declining scores in other regions were driven by minor decreases or a total absence of reproductive effort of foundational species (Figure 22, Table 23). Conversely, resilience in coastal habitats in the Mackay–Whitsunday and Fitzroy regions improved.

Resilience in reef intertidal habitats showed a modest improvement across most NRM regions, including Cape York, northern Wet Tropics, Mackay–Whitsunday and Fitzroy regions. This enhancement was fuelled by increases in abundance above low resilience thresholds, while sexual reproduction remained low (Figure 22, Table 23). The score declined in the southern Wet Tropics and Burdekin regions driven by a change in species composition and loss of seagrass abundance, respectively.

Resilience improved in reef subtidal habitats driven by enhancements in the Burdekin and northern Wet Tropics where there were small improvements in cover and composition, while reproductive effort remained low. Resilience of reef subtidal habitats remained stable in the southern Wet Tropics and Mackay Whitsunday regions (Figure 22, Table 23).

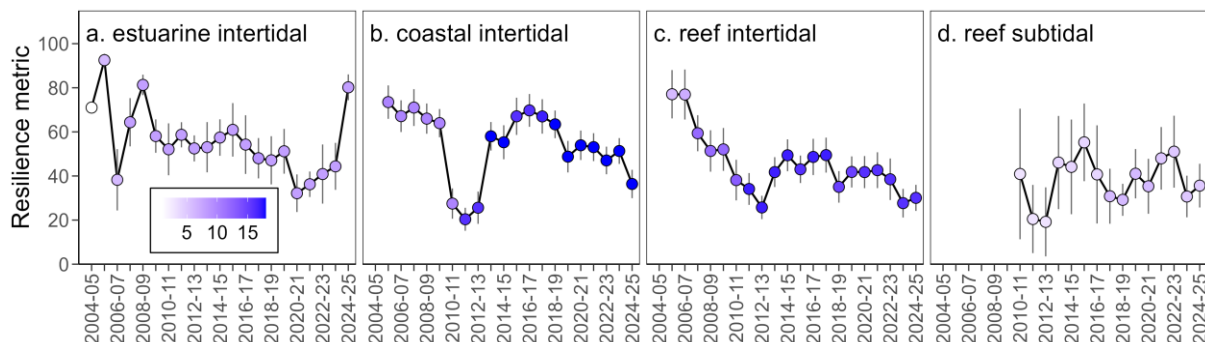


Figure 22. Trends in resilience metric summarised for each habitat type of the Reef. Blue shading of points indicates the number of sites contributing to the metric. Vertical error bars represent standard error.

### 4.3.4 Epiphytes and macroalgae

Epiphyte cover on seagrass leaves has fluctuated and often varied seasonally. For example, in 2024–25, epiphytes in estuarine habitats declined in the late dry and increased above the long-term average in the late wet season. Conversely, epiphyte cover in coastal intertidal habitats were less seasonal and often below the Reef-wide average, potentially due to exposure (drying at low tide and waves) and in 2024–25 epiphytes were below average in both seasons (Figure 23). Epiphyte cover at reef intertidal and subtidal habitats declined relative to last period and was below the average in both seasons (Figure 23).

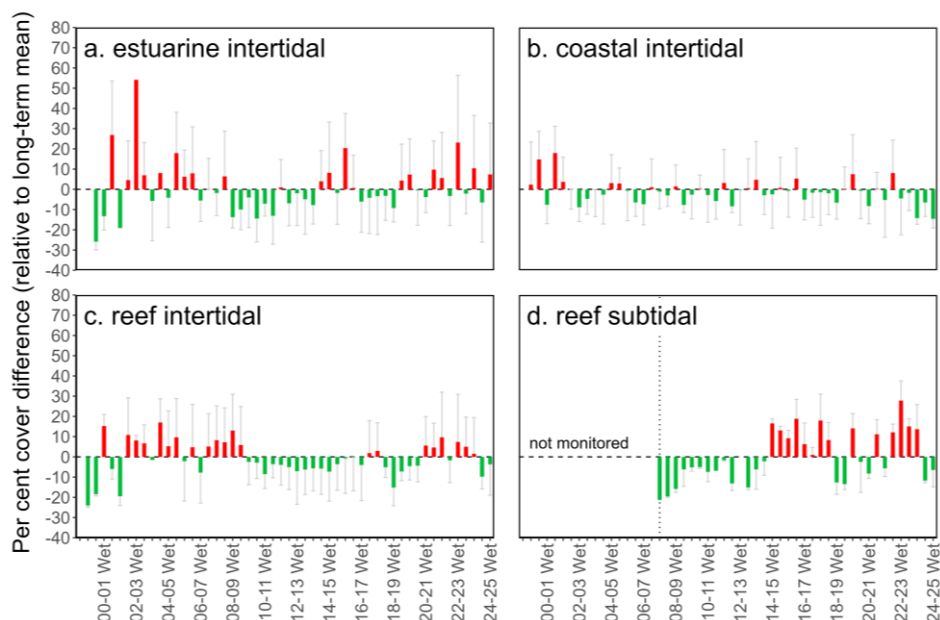


Figure 23. Epiphyte abundance (per cent cover) relative to the long-term average ( $y = 0$ ) for each Reef seagrass habitat (sites pooled,  $\pm$  SE). Reef long-term average (1999 to 2024); estuarine =  $25.9 \pm 5.1$  per cent, coastal intertidal =  $17.5 \pm 4.1$  per cent, reef intertidal =  $22.2 \pm 4.7$  per cent, reef subtidal =  $21.4 \pm 4.6$  per cent.

Macroalgae abundance in 2024–25 fell below the overall inshore Reef long-term average for all of the habitats (Figure 24).

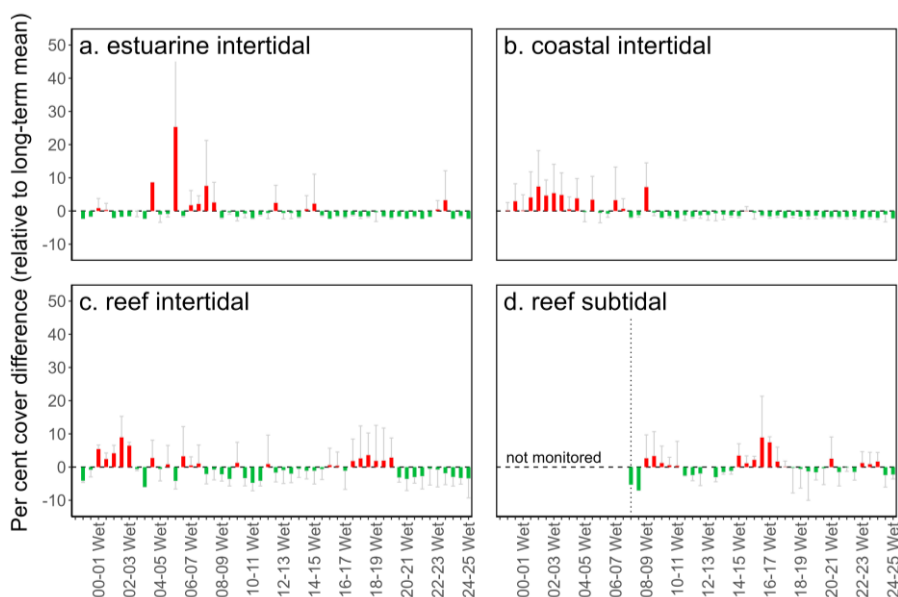


Figure 24. Macroalgae abundance (per cent cover) relative to the long-term average ( $y = 0$ ) for each inshore Reef seagrass habitat. (sites pooled,  $\pm$  SE). Reef long-term average (1999-2024); estuarine =  $1.9 \pm 1.4$  per cent, coastal intertidal =  $2.1 \pm 1.4$  per cent, reef intertidal =  $6.8 \pm 2.6$  per cent, reef subtidal =  $6.9 \pm 2.6$  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 2024–25 Summary

Wet season rainfall was around average and annual river discharges were slightly above the long-term medians, particularly in the most southern Cape York basins and the most northern. Exposure of the seagrass sites to turbid water types 1 (WT1) and 2 (WT2) was also above-average, and there was below-average daily light throughout the region. Within-canopy water temperatures were marginally warmer than in the previous period and exceeded the long-term average of the past decade.

Seagrass condition was assessed only in the late dry in Cape York, which precedes the summer when the highest rainfall, river discharge and temperatures occurred. Seagrass meadow condition across the Cape York NRM region in 2024–25 was greatly lower than 2023–24, declining to **‘poor’**. The decrease was due to the deterioration of both the abundance and resilience scores. For the indicators:

- abundance score was ‘poor’
- resilience score was ‘poor’.

Seagrass abundance (per cent cover) in 2024–25 declined from the previous period overall, to the lowest levels since monitoring commenced. This decline in seagrass abundance was driven by deterioration in all habitats, but the greatest in coastal intertidal and reef subtidal, particularly in the south of the region. Coastal intertidal abundances continued to decline for the second consecutive year, while reef subtidal abundances have continued to decline for the sixth consecutive year, with sites remaining near devoid of seagrass for the third year in a row.

Reproductive effort declined to zero across coastal and reef habitats in 2024–25, consistent with historically low reproductive activity and limited seed production in reef intertidal meadows. Seed banks remained dominated by *Halodule uninervis*, with the highest seed counts in 5 years at the coastal sites in Bathurst and Shelburne bays, though no seeds were recorded at other Cape York reef sites.

In 2024–25, the resilience score for intertidal seagrass meadows in Cape York reached the lowest on record, continuing a steady decline since 2016–17. Coastal sites recorded the second-lowest scores on record, with abundance below the 20th percentile and no reproductive structures, resulting in a resilience category of ‘1.1’. Reef intertidal sites showed a slight increase due to higher abundance and persistent species at one site (ST1), but overall resilience remained among the lowest recorded.

The number of monitored sites, their establishment dates, and the length of datasets available for each location all influence the interpretation of long-term regional trends. Prior to 2011–12, monitoring in Cape York was limited to a single location; trends from 2011–12

onward therefore incorporate data from multiple locations and a broader range of habitat types. Elevated river discharge events in 2010–11 and 2018–19 corresponded with marked declines in seagrass condition, particularly within coastal and reef subtidal habitats, which exhibited high variability in abundance and were adversely affected following the 2018–19 event. Post-flood assessments in the subsequent growing seasons consistently showed deterioration of the regional health score to ‘poor’.

Further elevated discharge occurred in 2022–23 and intensified in 2023–24. Seagrass condition in Cape York for 2024–25 was assessed during the late dry season of 2024—immediately following the most recent high-discharge period—and the observed declines are therefore likely a legacy of the cumulative impacts from the preceding wet seasons.

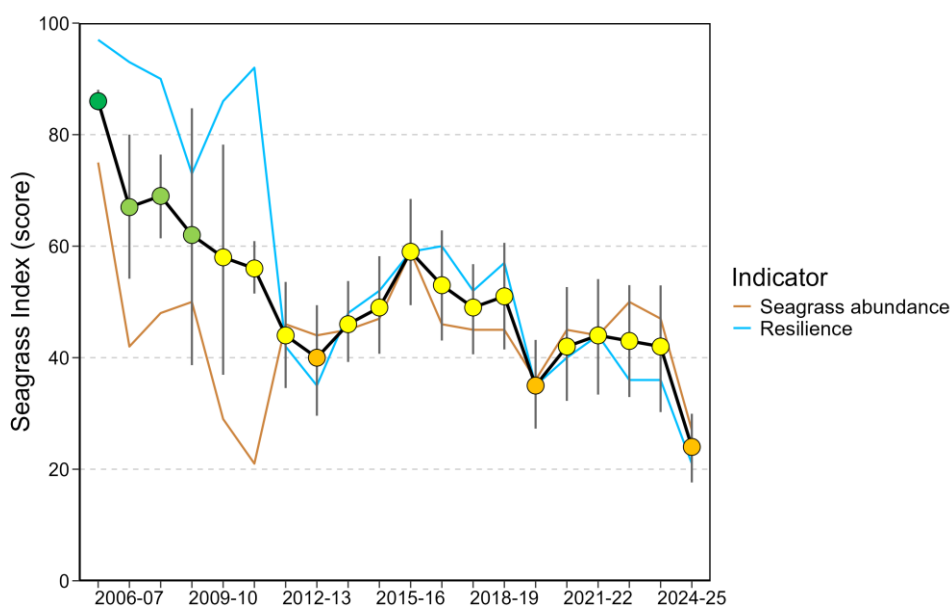


Figure 25. Temporal trend in the Seagrass Index ( $\pm$  SE) with contributing indicator scores for the Cape York NRM region (averaged across habitats and sites). Index scores scaled from 0–100 and graded: ● = very good (81–100), ● = good (61–80), ● = moderate (41–60), ● = poor (21–40), ● = very poor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

### 5.1.2 Climate and environmental pressures

In 2024-25, seagrass habitats in Cape York were predominantly affected by around average rainfall, but slightly above average river discharge, exposure to turbid waters and higher temperature. The 2024–25 wet season (December to April) had no notable cyclone influence and rainfall was below the 30-year long-term average (1961 to 1990), but this trend varied among the basins (Figure 26). The Jackie, Olive Pascoe and Stewart basins were slightly above average and the other basins were around or below average (Figure 6) (Gruber *et al.* 2026).

Annual discharge in 2024-25 was 1.4 times above median from all basins pooled in Cape York (Table 11). Those in the south (Jeannie and Endeavour Rivers) had the highest annual discharges relative to the long-term median, at 1.9 times the long-term median (Table 11). Annual discharge from Jacky Jacky Creek — the most northern basin — was 1.6 times the long-term median. Annual discharge was 1.4 times the long-term median from the Normanby River, and 1.2 times from the Olive Pascoe and Lockhart Rivers. The only discharge below the long-term median was the Stewart River.

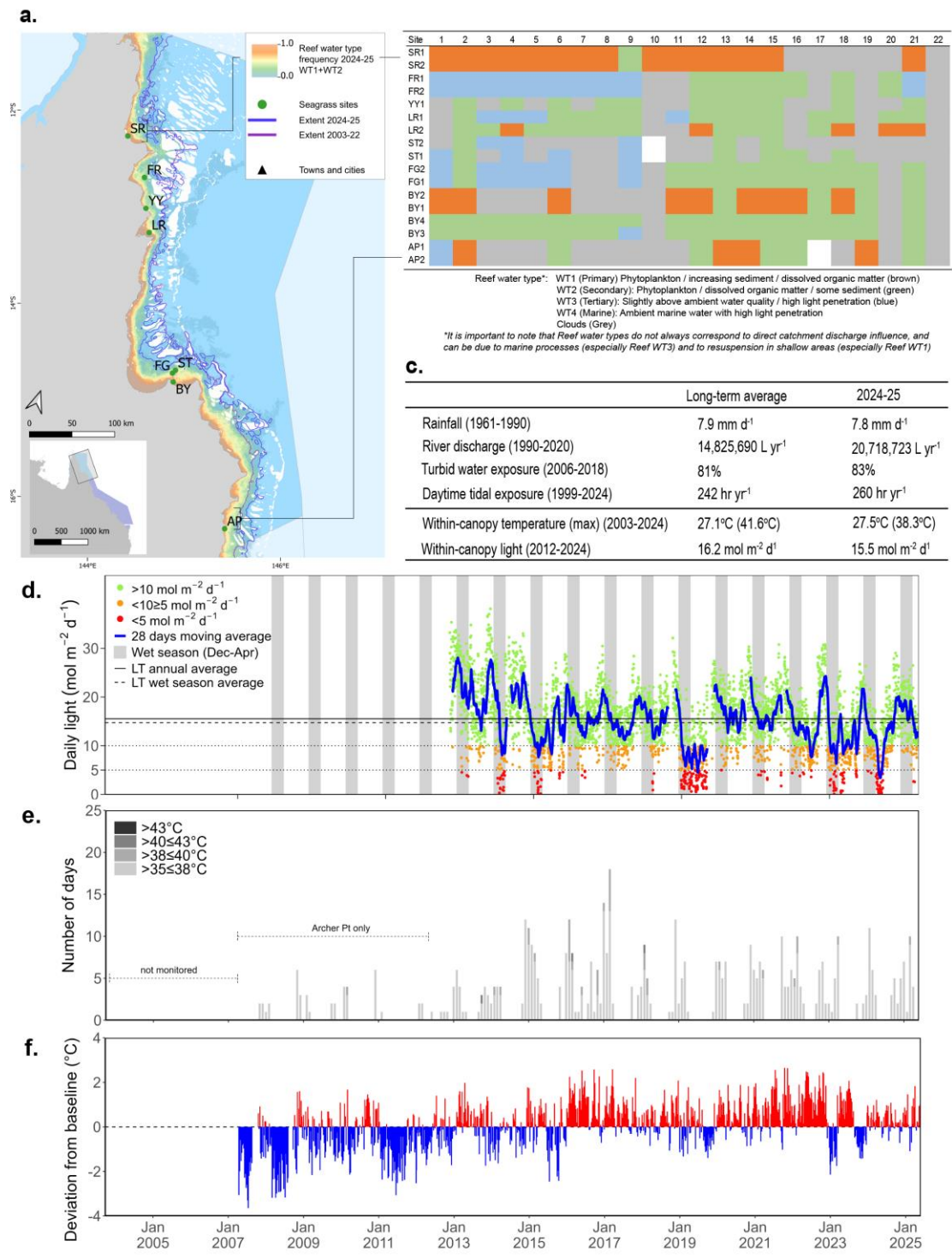


Figure 26. Environmental pressures in the Cape York region including: a. frequency of exposure to primary (WT1) and secondary (WT2) water from December 2024 to April 2025 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed) (white = no data), also showing the long-term average (2003–2018) exposure boundary (purple line), and the extent for 2024–25 (blue line) (from Gruber *et al.* 2026), b. wet season water type at each site; c. average conditions and max temperature over the long-term and in 2023–25; 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.

Exposure to water types 1 (WT1) and 2 (WT2) was slightly higher than the long-term average in Cape York; however, calculation from satellite imagery was constrained by frequent cloud cover, which often obscured the water types and prevented their classification (Figure 26, Figure 8 and Figure 9). The inshore waters of Cape York were dominated by water type 2 (WT2) during the wet season (December–April; Figure 26b). Shelburne Bay sites (SR1 and SR2), followed by Bathurst Bay sites (BY1 and BY2), exhibited the greatest exposure to turbid water type 1 (WT1) conditions. In contrast, BY3 and BY4 (further offshore, subtidal) and reef habitats at Piper Reef (FR) and Stanley Island (ST) experienced the lowest exposure to WT1 and WT2 among the inshore seagrass monitoring sites.

The potential exposure risk of seagrass to the water types are assessed in the MMP inshore water quality component (Gruber *et al.* 2026). In Cape York, the area of seagrass exposed to water quality risk was below the long-term average in 2024–25, with an overall exposure of 56 per cent, primarily falling into the low-risk category.

Daily light reaching the top of the seagrass canopy was generally high across Cape York monitoring sites, with a long-term average of  $16.2 \text{ mol m}^{-2} \text{ d}^{-1}$ . In 2024–25, mean daily light ( $15.5 \text{ mol m}^{-2} \text{ d}^{-1}$ ) was slightly below this long-term average (Figure 26d). Light loggers malfunctioned in Shelburne Bay and recorded for only a short duration at Bathurst Bay (Figure 105). Consequently, data from Piper Reef and Stanley Island were the primary contributors to the regional average, and were affected by light levels at Stanley Island remaining below both the long-term and wet-season averages for most of the year.

The 2024–25 period was warmer than the previous years and the fifth warmest year on record since intertidal within-canopy temperature monitoring was established in the region (Figure 26c). Maximum within-canopy temperatures exceeded  $35^\circ\text{C}$  for a total of 37 days (total among all sites where temperature is monitored) during 2024–25 (Figure 26e), which was higher than the 2 previous years. The highest temperature recorded was  $38.3^\circ\text{C}$  (Bathurst Bay, 30 March 2025). Daytime tidal exposure (hours water has drained from the intertidal meadow) was above the Cape York long-term median (Figure 26c, Figure 91).

In the Cape York NRM region, there was minimal alteration in the sediments of reef habitats, which continued to be primarily composed of sands and coarser grains (Appendix 2, Figure 112). During 2024–25, coastal habitats maintained their dominance of fine sand, while the rising mud levels in Bathurst Bay slightly increased above the long-term average at one site (BY2) (Appendix 2, Figure 113).

### 5.1.3 Inshore seagrass and habitat condition

There are 19 seagrass monitoring sites in Cape York across 9 locations (Table 13). Four seagrass habitat types were assessed across the Cape York region in 2024–25, with data from 16 of the 19 long-term monitoring sites (Table 13, Table 20).

Table 13. 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 5 and Table 6. Open square indicates not measured in 2024–25, 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	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	■	■					■	■
	MA1^	Margaret Bay	■	■					■	■
	MA2^	Margaret 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 Beach (Weymouth Bay)	□	□				□	□	□
Reef subtidal	FG1^	Flinders Island (Flinders Group)	■	■					■	■
	FG2^	Flinders Island (Flinders Group)	■	■					■	■

### 5.1.3.1 Seagrass Index and indicator scores

During the 2024–25 reporting period, the Seagrass Index score for the Cape York region greatly declined from the previous reporting period, with the overall grade dropping to ‘poor’ (Figure 27). This change was the result of a declines in both seagrass condition indicators.

The seagrass abundance indicator experienced a significant decline in 2024–25, following minor decreases in the prior period (Figure 27). This downturn was largely influenced by reduced abundance scores in coastal habitats, especially in intertidal areas to the south at Bathurst Bay, which dropped from ‘very good’ to ‘moderate’, and in subtidal areas to the north at Lloyd Bay and Margaret Bay, falling from ‘very good’ to ‘moderate’ and from ‘good’ to ‘poor’, respectively. Subtidal locations in both coastal and reef habitats in the south remained ‘very poor’ after the declines observed in the previous year. During 2024–25, reef intertidal habitats in the north saw a decline from ‘poor’ to ‘very poor’, while those in the south showed slight improvement, moving from ‘very poor’ to ‘poor’.

The seagrass resilience indicator score declined in 2024–25 to its lowest level since monitoring was established (Figure 27). Losses appear a consequence of reduced sexual reproductive effort and diminished seed banks. The decline was primary driven by the deterioration of the score to ‘very poor’ in coastal habitats from ‘moderate’ and ‘good’ in the north and south of the region, respectively. Conversely, seagrass resilience at reef intertidal habitats remained ‘moderate’ in the north (Piper Reef), while improving greatly in the south (Stanley Island) from ‘very poor’ to ‘moderate’. It’s important to note that resilience is only assessed for intertidal habitats in Cape York.

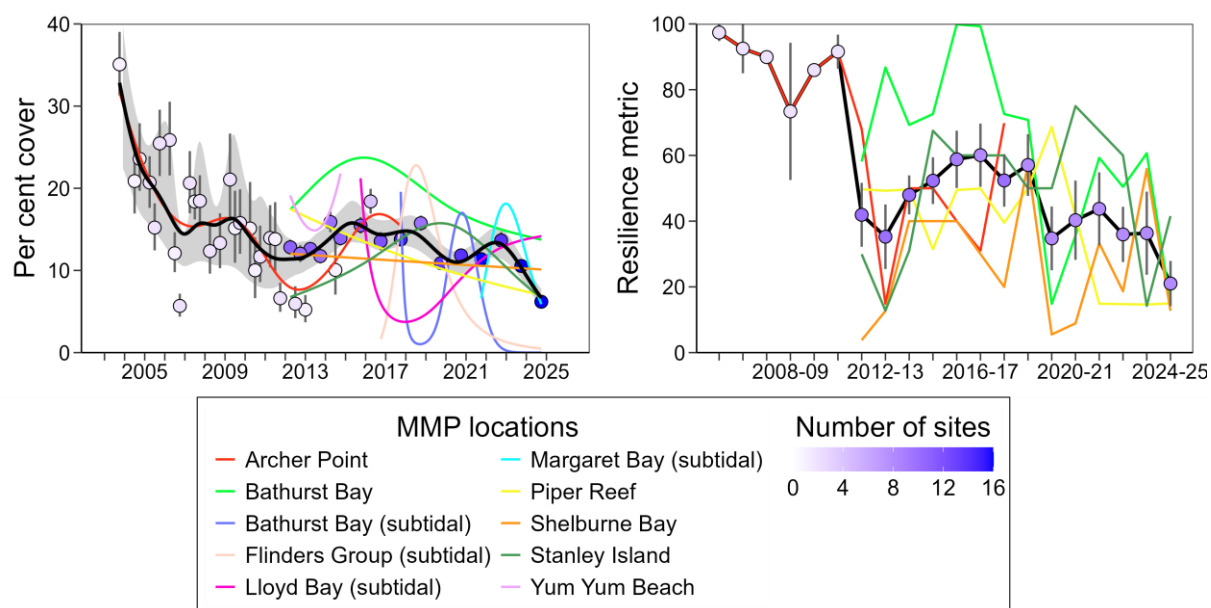


Figure 27. Temporal trends in the Cape York seagrass indicators used to calculate the Seagrass Index: a. average (circles,  $\pm$ SE) seasonal abundance (per cent cover) and GAM plots of seagrass abundance trends for each location (coloured lines) and the region (black line with grey shaded area defining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

A careful interpretation is necessary when examining the long-term trends in abundance across the Cape York NRM region, as new sites added in 2012–13 are linked to consistently lower abundance compared to the peak levels previously recorded for the region (Figure 27). Additionally, Archer Point, the sole location monitored before 2012–13, has not been included in the abundance or resilience score since October 2017, as no monitoring has been conducted.

### 5.1.3.2 Seagrass abundance, composition and extent

The Cape York NRM region's average seagrass abundance has continued to gradually decline, and in 2024–25 reached its lowest level ( $4.8 \pm 0.7$  per cent) since monitoring commenced. This decline was a consequence of deteriorating per cent cover in all habitats in 2024–25. The greatest decline was in coastal intertidal and reef subtidal habitats, which deteriorated well below their respective long-term averages to the lowest average abundances since monitoring began (Figure 28). While there was a marginal improvement in abundance at reef intertidal sites in the south (Stanley Island), for the first time in 5 years, these gains were offset by declines at Piper Reef sites in the north, following improvements in the previous year. The assessment of reef subtidal habitats is limited to the south of the region at the Flinders Group, where abundances have continued to decline since 2018–19 and consistently remain very low. In 2024–25, seagrass was found at only one site (FG1) where less than half the quadrats examined had seagrass present and abundances were less than one per cent cover (Figure 28).

Coastal intertidal seagrass abundances continued to decline for the second consecutive year, after falling below their long-term average in 2023–24. These losses were driven by Bathurst Bay located in the south, with levels in 2024–25 around a third of the long-term average. In Shelburne Bay in the north, one of the sites (SR2) fell below the long term average in 2024–25 for the first time with overall, abundances declining to about half of their long-term averages. Similarly, coastal subtidal abundances declined across the region, with the greatest losses in Margaret Bay (particularly MA2) in the north. Seagrass in Lloyd Bay, in the central part of the region, declined for the first time since 2021–22 when monitoring was established. Replicate sites (BY3 and BY4) in Bathurst Bay in the south remained pretty

much devoid of seagrass for the third year in a row, with only one shoot of *Halophila ovalis* observed across both sites in 2024–25 (Figure 28). Coastal subtidal abundances overall, remained below their long-term average for the fourth consecutive year.

As the seagrass abundance in 2024–25 was assessed before the 2024–25 wet season, the observed declines in subtidal and reef intertidal abundances are likely the legacy of the above-average rainfall and river discharges, 3 times above the long-term median, in 2023–24. The impact was greatest in the southern part of the region, particularly in Bathurst Bay and the Flinders Group, which are located adjacent to the Normanby-Kennedy river basin. This area experiences substantial sediment-laden water discharge during heavy rainfall and flow events, which can greatly affect seagrass growth in the discharge vicinity.

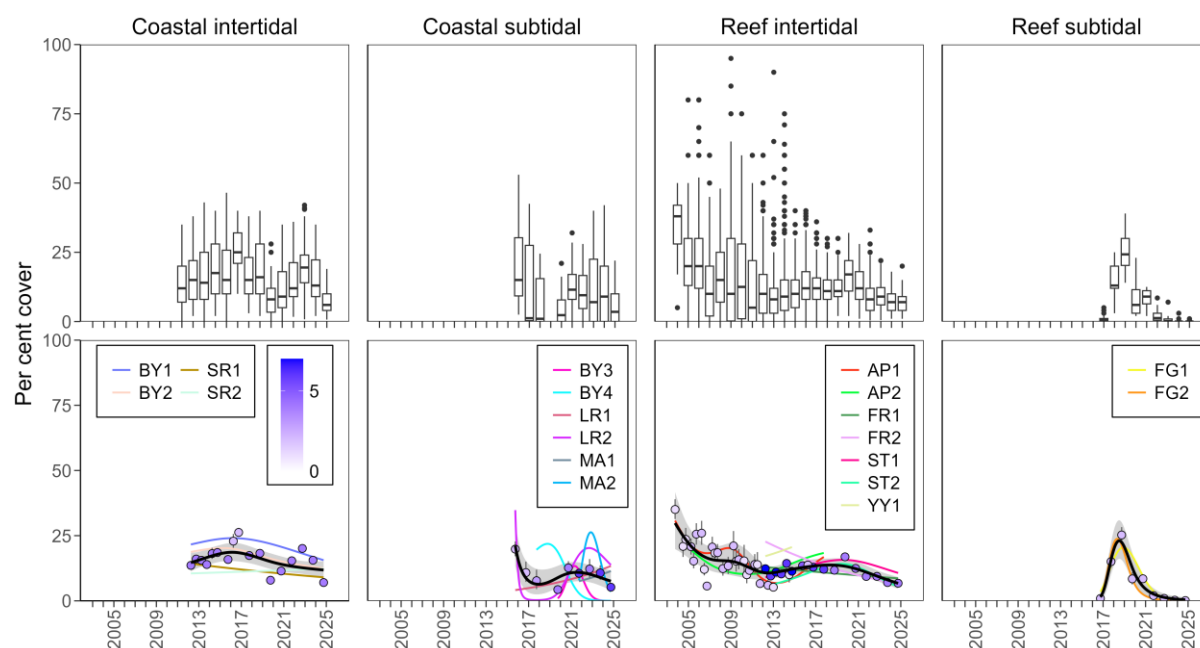


Figure 28. Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends for each habitat monitored in the Cape York region from June 2005 to May 2025. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates the 25<sup>th</sup> percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75<sup>th</sup> percentile. Whiskers (error bars) above and below the box indicate the 90<sup>th</sup> and 10<sup>th</sup> percentiles, and the dots represent outlying points. GAM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.

An examination of the long-term trend in seagrass abundance shows seagrass per cent cover progressively decreased at reef intertidal habitats throughout Cape York from 2003 to 2012. Following this period, there was a modest recovery, particularly at Stanley Island (e.g., ST2). However, abundances at the reef intertidal sites continue to remain low, and those in the northern part of the region exhibit a decreasing long-term trend (Figure 28, Table 22). The assessment of reef subtidal habitats is currently limited to the southern part of the region, specifically the Flinders Group of islands. Over the past 5 years, there has been a noticeable decline in abundances, and one of the sites is now showing a concerning long-term decreasing trend (Figure 28, Table 22).

Monitoring of coastal intertidal and subtidal habitats began in 2012 and 2015, respectively. Over the past decade, these habitats have largely exhibited no long-term trends (Figure 28, Table 22). However, a notable concern has arisen with one of the coastal intertidal sites at Shelburne Bay in the northern region, now indicating a long-term decreasing trend (Figure 28, Table 22).

In 2024–25, the proportion of species displaying colonising species traits (*Halophila ovalis*) was lower or similar to the previous reporting year in all Cape York habitats, with the exception of the reef intertidal, which saw an increase. Meanwhile, the proportion of

colonising species in subtidal habitats remained above the Reef long-term average, whereas in intertidal habitats, it remained below (Figure 29).

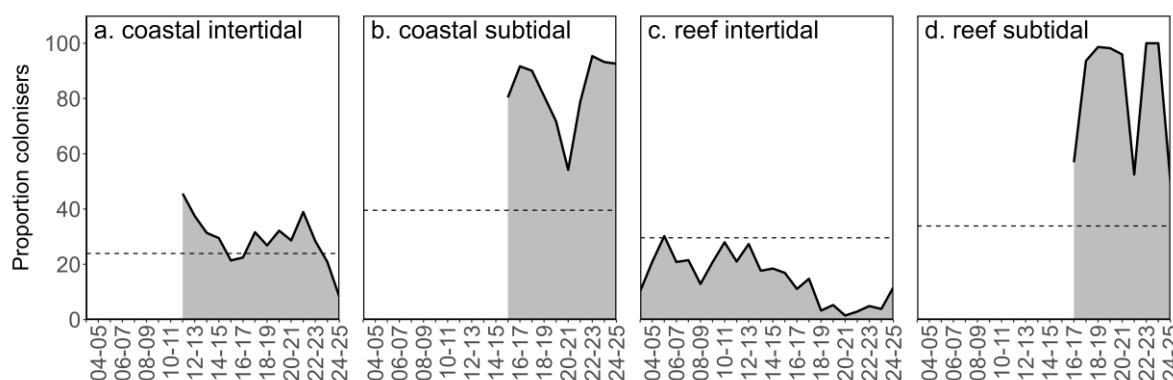


Figure 29. Proportion of seagrass abundance composed of colonising species at inshore intertidal habitats in the Cape York region, 2004 to 2025. Dashed line represents Reef long-term average proportion of colonising species for each habitat type. Seagrass spatial extent mapping was conducted within meadows 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). Only intertidal meadows are mapped across the Cape York region and prior to 2012, mapping only occurred at the reef intertidal meadows of Archer Point (Figure 30). Over the last decade, additional reef and coastal meadows in the Cape York region were included. Generally, there has been some variation in the relative meadow extent at coastal intertidal habitats over the years (Figure 30). These fluctuations appear primarily due to modifications in natural drainage channels. Meanwhile, at reef habitats, the relative meadow extent seems to be marginally higher in the current decade than the previous.

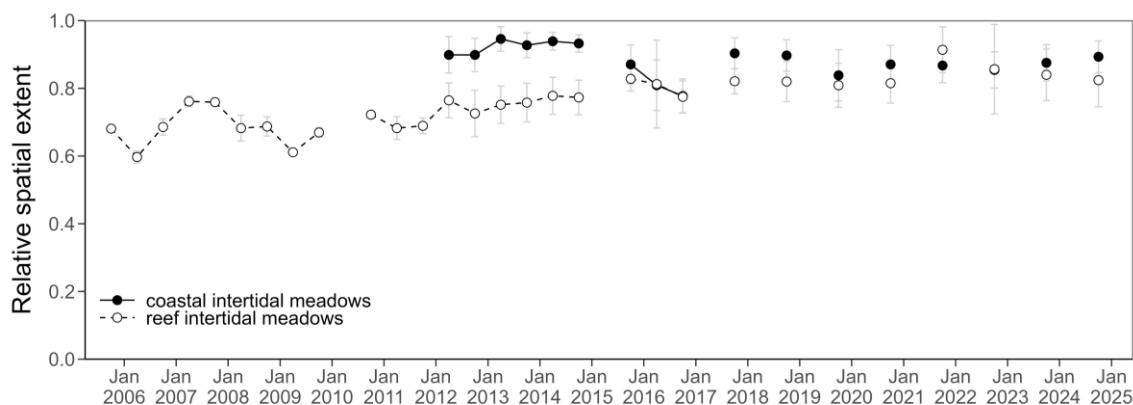


Figure 30. Change in relative spatial extent ( $\pm$  SE) of seagrass meadows within monitoring sites for each intertidal coastal and reef habitat and monitoring period across the eastern Cape York NRM region, 2005–2025.

### 5.1.3.3 Seagrass reproductive status

Reproductive effort declined to zero across coastal and reef habitats in 2024–25 (Figure 31). Historically (2006–2012), reproductive activity in reef intertidal habitats was only recorded at Archer Point, which has not been monitored since 2017. Current assessments, based on sites established in 2012, continue to show consistently low numbers of sexual reproductive structures, a trend typical of reef habitats across the Reef.

This limited reproductive effort constrains seed bank replenishment and increases meadow vulnerability to disturbance due to reduced recovery potential and genetic diversity. Seed banks remain dominated by *Halodule uninervis* at Cape York sites, with persistent seed

reserves observed in the coastal meadows of Bathurst Bay over the past decade. In 2024–25, seeds were recorded at both Bathurst Bay and Shelburne Bay, representing the highest counts in 5 years (Figure 31). Seed densities were low or absent in reef intertidal habitats, with no seeds recorded at any Cape York sites in 2024–25 (Table 5). Monitoring of reproductive effort and seed banks is currently restricted to intertidal meadows in Cape York.

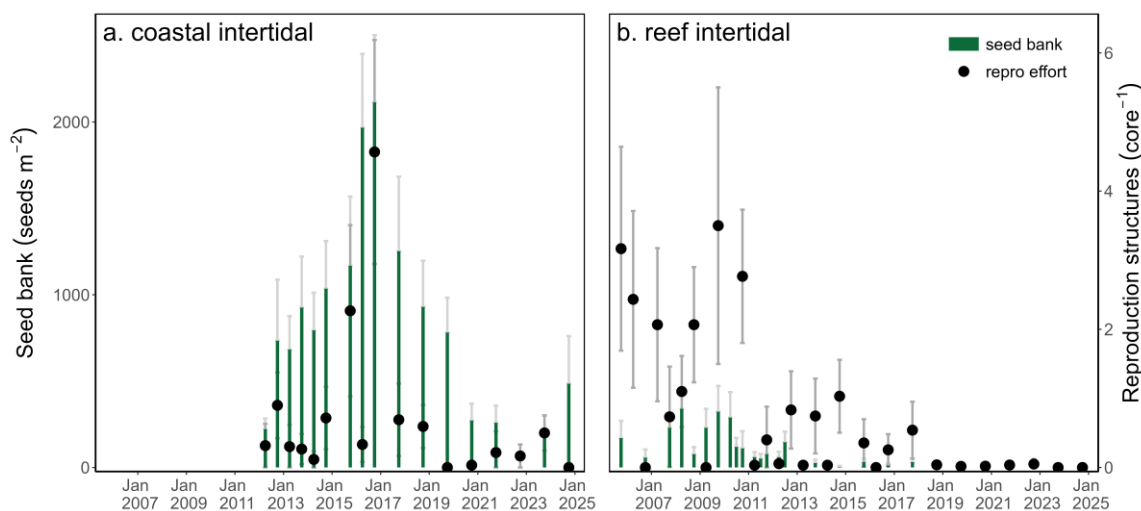


Figure 31. Seed banks and reproductive effort at inshore intertidal coastal (a) and reef (b) habitats in the Cape York region, for late dry season, 2005–25 (species and sites pooled). Seed banks (green bars,  $\pm$  SE) presented as the total number of seeds per m<sup>2</sup> sediment surface. Reproductive effort (dots,  $\pm$  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 2024–25, the resilience score was the lowest on record and has been steadily declining since 2016–17 (Figure 25).

At coastal sites in 2024–25, the score decreased to its second lowest on record. At all coastal sites, abundance declined to the lowest on record and below the 20<sup>th</sup> percentile for each site. Furthermore, there were no reproductive structures of any species. As such, the resilience score category was '1.1' for all sites. All sites were dominated by foundational species so scored at high level within that category ('11.7'–'14.9').

Resilience increased at reef intertidal sites and were the third lowest on record. For 3 of the sites (ST2, FR1 and FR2), the score category was also '1.1' because abundance was below the 20<sup>th</sup> percentile for the sites and there were no reproductive structures. At ST1, abundance and colonising species composition were above the thresholds, there no reproductive structures in 2024, but there had been in previous years. Therefore, the resilience score category was '2.1.2' and the score was 69 because of a high proportion of persistent species (*Thalassia hemprichii*). The increase in the score at this one site was the reason for the increase in the score at reef intertidal sites.

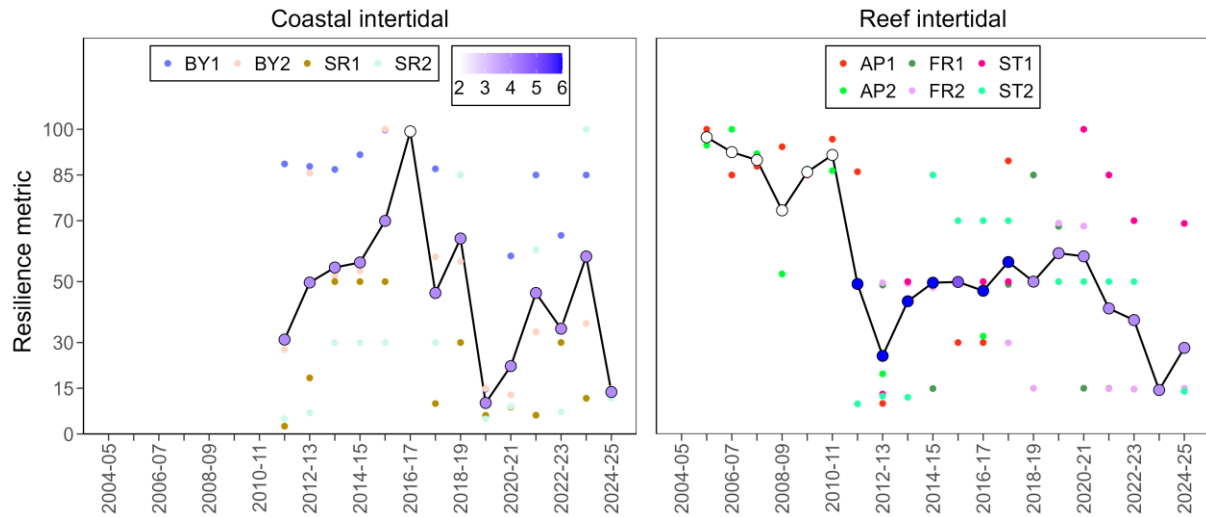


Figure 32. Temporal trend in the resilience metric for each habitat monitored in the Cape York NRM region from 2005–2025. Coloured small points represent different sites. Shades of blue for the larger points indicate the number of sites that contribute to the score.

### 5.1.3.5 Epiphytes and macroalgae

In 2024–25, epiphyte cover on seagrass leaf blades in intertidal coastal habitats declined relative to the previous period, yet it remained marginally above the long-term average. Conversely, epiphyte cover in intertidal reef habitats remains below the long-term average for the seventh consecutive year (Figure 33). In subtidal waters, epiphyte cover in both coastal and reef habitats was negligible (Figure 33). Overall, the consistently low levels of epiphyte cover across the region indicate that it is unlikely to have a significant impact on seagrass growth.

The percentage cover of macroalgae shows ongoing fluctuations among various habitats. For the seventh consecutive year, macroalgae cover in coastal intertidal habitats has remained below the long-term average (Figure 33b). In contrast, it has exceeded the average at reef intertidal habitats for the fourth consecutive year (Figure 33f and 32h). At intertidal reef habitats, macroalgae grow attached to coral rubble within the meadow, but their levels are not high enough to adversely affect seagrass. Within subtidal coastal and reef habitats, macroalgae can be variable and in 2024–25 continued to remain below the overall inshore Reef long-term average.

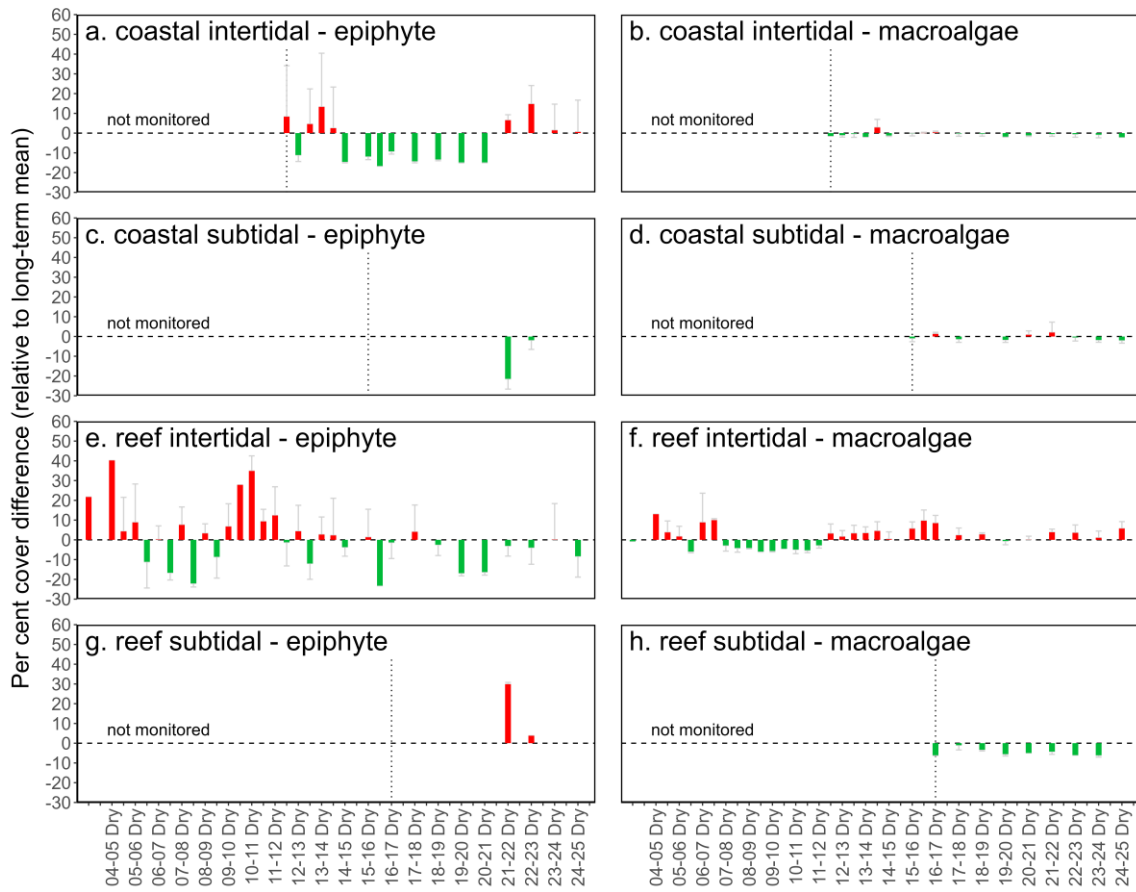


Figure 33. Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average for each inshore intertidal seagrass habitat in the Cape York, 2001–2025 (sites pooled,  $\pm$ SE). Vertical dotted lines represent the first monitoring event for each habitat type. Red and green bars indicate above and below long-term average, respectively.

## 5.2 Wet Tropics

### 5.2.1 2024–25 Summary

In the 2024–25 period, rainfall in the Wet Tropics was above the long-term average, with highest deviations in southern basins and near-median totals in the Daintree River in the north. River discharge was above the regional median in the north, ranging from 2.2 times the median in the Barron River to 1.6 times in the Mossman, but lower than in 2023–24. Turbid water exposure exceeded the long-term average, with WT2 dominating most sites except Yule Point, where WT1 was higher. Additionally, during this time, above-average rainfall and river discharge increased turbidity exposure and diminished daily light in seagrass habitats of the southern Wet Tropics. Rainfall exceeded the long-term average across all southern catchments, with discharge nearly twice the mean for the second consecutive year. Turbidity exposure in the south remained high but slightly below average, with similar exposure to WT1 and WT2 at coastal sites such as Luggar Bay and Missionary Bay. Within canopy temperatures in the 2024–25 period were around the long-term average.

Seagrass meadows throughout the Wet Tropics displayed variability across the region, with the overall Seagrass Index experiencing a marginal decline in 2024–25. Seagrass condition showed a minor drop in both the northern and southern Wet Tropics NRM regions, maintaining a status of ‘moderate’ and ‘poor’, respectively (Figure 34). Overall, the combined regional condition was ‘**poor**’ (Figure 34).

Contributing indicators in the north were:

- abundance was ‘moderate’
- resilience was ‘poor’.

Contributing indicators in the south were:

- abundance was ‘very poor’
- resilience was ‘poor’.

In the northern Wet Tropics, seagrass abundance showed a slight improvement in 2024–25 relative to the previous year, driven primarily by increases within reef habitats. In contrast, coastal habitats recorded declines in abundance, which also contributed to a modest reduction in overall resilience for the region. These coastal declines are consistent with the legacy effects of the preceding wet season, during which river discharge was substantially above average.

In contrast to the northern sub-region, seagrass abundances in the southern Wet Tropics remained relatively stable in 2024–25, with similar patterns observed across both coastal and reef habitats. Despite this stability, overall resilience declined slightly. This reduction was driven by seagrass loss at the southern coastal location and the continued absence of reproductive structures at reef sites, indicating limited recovery potential. The persistently low abundances throughout the southern Wet Tropics reflect the long-term legacy of past extreme events, particularly the severe weather, above-average rainfall and elevated discharge that occurred between 2009 and 2011. Recovery since that period has been slow, especially in southern areas, where unstable substrates, chronically poor water quality—characterised by high turbidity and restricted light availability—and limited recruitment capacity continue to constrain the regeneration of seagrass meadows.

In 2024–25, seagrass resilience in the northern and southern Wet Tropics declined to ‘poor’, continuing a three-year downward trend driven by low abundance and the absence of reproductive structures at coastal sites. Reef intertidal and subtidal resilience improved slightly but remained low overall, with persistent reproductive failure at Green Island, and limited reproductive structures but continued dominance of colonising species at Low Isles. Overall, seed densities remained well below long-term averages for the second consecutive

year, with none recorded at most sites. The continued absence of seeds and reproductive structures at the majority of sites indicates reduced reproductive capacity and recovery potential among foundational species, rendering meadows, especially in the southern sub-region, particularly vulnerable to disturbances.

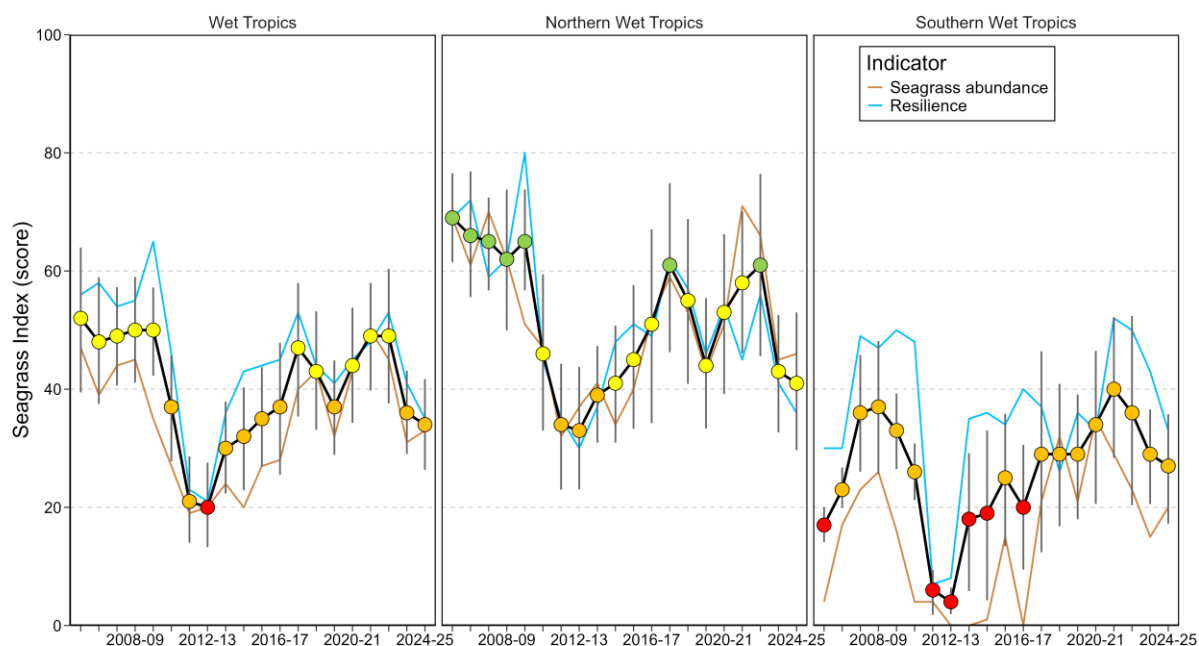


Figure 34. Temporal trend in the Seagrass Index ( $\pm$  SE) with contributing indicator scores for the Wet Tropics NRM region and sub-regions (average across habitats and sites). Values are indexed scores scaled 0–100 ( $\pm$  SE) and graded: ● = very good (81–100), ● = good (61–80), ● = moderate (41–60), ● = poor (21–40), ● = very poor (0–20). NB: Scores are unitless.

### 5.2.2 Climate and environmental pressures

In 2024-25, seagrass habitats in the northern Wet Tropics were predominantly affected by above-average rainfall, above-median river discharge, above-average exposure to turbid waters and below-average daily light.

The 2024–25 wet season (December to April) had no notable cyclone influence and annual daily rainfall in the northern Wet Tropics basins was above the long-term average (Figure 35). Annual discharge was above median from all basins in the northern Wet Tropics, and above the long-term median for the region as a whole (Table 11) although discharges were considerably lower than in 2023–24. Annual discharge was between 2.2 times the long-term median from the Barron River and 1.6 times from the Mossman River, with discharges from other rivers being closer to the long-term median (Table 11).

Exposure to primary (WT1) or secondary (WT2) turbid water was higher than the long-term average across the northern Wet Tropics during 2024–25 (Figure 35a, b). Sites were primarily exposed to WT2, except at Yule Point where there was more exposure to WT1 (Gruber *et al.* 2026).

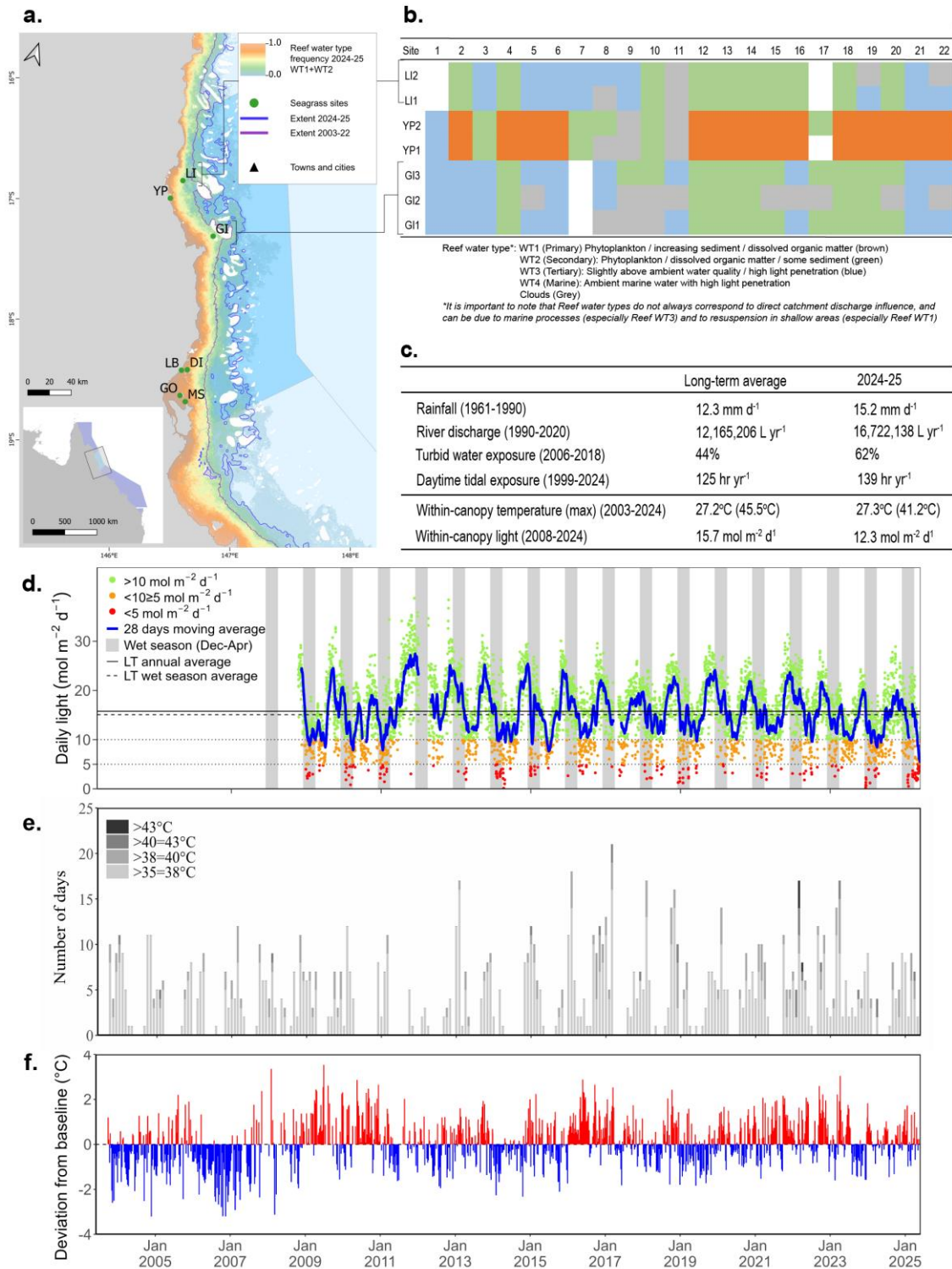


Figure 35. Environmental pressures in the northern Wet Tropics region including: a. frequency of exposure to primary (WT1) and secondary water (WT2) from December 2024 to April 2025 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed) (white = no data), also showing the long-term average (2003–2018) exposure boundary (purple line), and the extent for 2024–25 (blue line) (from Gruber *et al.* 2026); b. wet season water type at each site; c. average conditions and max temperature over the long-term and in 2024–25; 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.

Daily light levels at the intertidal sites ( $12.3 \text{ mol m}^{-2} \text{ d}^{-1}$  in 2024–25) were lower than the long-term average ( $15.7 \text{ mol m}^{-2} \text{ d}^{-1}$ ) in the northern Wet Tropics (Figure 35c, d). This was predominantly due to low light during and after the wet season at Green Island (Figure 106).

The risk of water quality exposure of seagrass are assessed in the MMP inshore water quality report (Gruber *et al.* 2026). In the Wet Tropics (north and south combined), there was no change in the overall area exposed to categories II-IV combined, in 2024–25. However, there was an increase in exposure to moderate risk (category II) and a decrease in exposure to the lowest and highest risk in 2024–25 compared to the long-term average.

Intertidal within-canopy temperature in 2024–25 in the northern Wet Tropics was around the long-term average (Figure 35e). This was due to lower-than-average temperatures recorded at Yule Point and Low Isles being offset by higher-than-average temperatures at Green Island. Maximum intertidal within-canopy temperatures exceeded  $35.0 \text{ }^{\circ}\text{C}$  for a total of 55 days during 2024–25, the sixth highest number of days in a period since monitoring commenced. Maximum temperatures also exceeded  $40.0 \text{ }^{\circ}\text{C}$ , but only during one day, the lowest number of days in the last 4 years. The highest temperature was  $41.2 \text{ }^{\circ}\text{C}$  at Low Isles (LI1) on the 28 April 2025 at 4pm. Daytime tidal exposure in the north was around or slightly above the long-term median (Figure 35c, Appendix 2, Figure 93, Figure 94.), which could affect water temperature, especially extremes (potentially increasing temperature in shallow water) and light levels.

In 2024–25, above-average rainfall and river discharge increased exposure to turbid water and reduced daily light in seagrass habitats of the southern Wet Tropics. Rainfall exceeded the long-term average in all southern basins, with the greatest anomaly observed in the Murray River basin (Figure 6). Correspondingly, river discharge was substantially elevated, reaching nearly twice the long-term mean for the second consecutive year (Figure 6). The Herbert River exhibited the highest relative increase, with flow volumes 2.7 times above the long-term average, while the magnitude of deviation declined northwards. Elevated rainfall and discharge did not translate to proportionally greater turbidity exposure, with seagrass monitoring sites experiencing turbid water on 93 per cent of wet-season weeks—slightly below the long-term average of 97 per cent (Figure 36a, c). In contrast to previous years, exposure to WT1 (suspended sediment) was comparable to exposure to WT2 (phytoplankton, dissolved organic matter, and some sediment), indicating a more balanced influence of particulate and optical constituents particularly at coastal sites at Lugger Bay (LB1, LB2) and Missionary Bay (MS1, MS2) (Figure 36b).

Average daily light reaching the seagrass canopy in 2024–25 was  $10.8 \text{ mol m}^{-2} \text{ d}^{-1}$ , substantially lower than the long-term mean of  $14.9 \text{ mol m}^{-2} \text{ d}^{-1}$  (Figure 36d, Figure 107), marking the third consecutive year of reduced light availability for the southern Wet Tropics. This sustained reduction primarily reflected low wet-season irradiance, compounded by below-average dry-season light conditions likely influenced by elevated river discharge and residual turbidity from the 2023–24 wet season (Figure 107). It is important to note that light data for the southern Wet Tropics were only available from Dunk Island intertidal habitat.

Dunk Island is also the only location where intertidal within-canopy temperatures are measured in the southern Wet Tropics, and in 2024–25, temperatures were the same as the long-term average for the third year in a row (Figure 36b). Similar to the previous period, the maximum within-canopy temperatures exceeded  $35 \text{ }^{\circ}\text{C}$  on just one day; however, this period, the peak was slightly elevated, reaching  $35.5 \text{ }^{\circ}\text{C}$  at DI1 on 12 November 2024 (Figure 36e, f). The 2024–25 period was also the second year where daytime tidal exposure has been above average (Figure 36b, Figure 93).

In the Wet Tropics region, coastal sediments mainly consisted of fine sand, while reef habitats featured a mix of sand and coarser sediments (Appendix 2, Figure 116, Figure 117). In 2024–25, sediments at the intertidal monitoring sites remained stable and similar to the long-term average (Figure 116, Figure 117). Subtidal sites, however, experienced a slight increase in finer sediments with mud sediments exceeding the long-term average at Low Isles for the first time in 6 years (Figure 109).

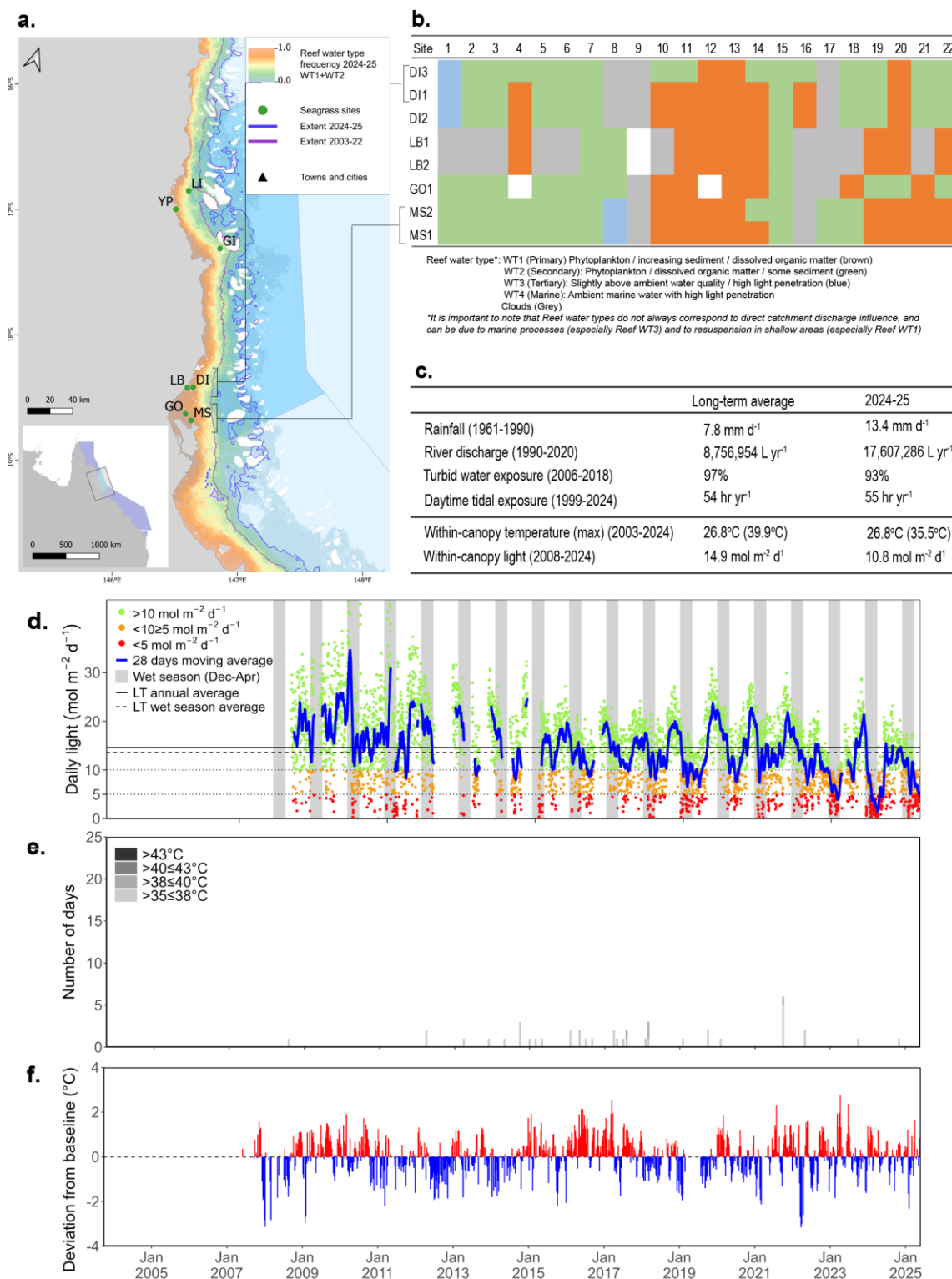


Figure 36. Environmental pressures in the southern Wet Tropics region including: a. frequency of exposure to primary (WT1) and secondary (WT2) water from December 2024 to April 2025 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed) (white = no data), also showing the long-term average (2003–18) exposure boundary (purple line), and the extent for 2024–25 (blue line) (from Gruber *et al.* 2026); b. average conditions and max temperature over the long-term and in 2024–25; 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

Four seagrass habitat types were assessed across the Wet Tropics region with data from 14 of the 15 long-term monitoring sites in 2024–25 (Table 14).

Table 14. 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 2024–25, blank cell indicates data not usually collected/measured at site. ^drop camera sampling (RJFMP), \*Seagrass-Watch. For site details see Table 5 and Table 6.

Sub region	Habitat	Site		abundance	composition	extent	reproductive effort	seed banks	meadow sediments	epiphytes	macroalgae
north	coastal intertidal	YP1	Yule Point	■	■	■	■	■	■	■	■
		YP2	Yule Point	■	■	■	■	■	■	■	■
	reef intertidal	LI1	Low Isles	■	■	■	■	■	■	■	■
		G11	Green Island	■	■	■	■	■	■	■	■
		G12	Green Island	■	■	■	■	■	■	■	■
	reef subtidal	LI2	Low Isles	■	■	■	■		■	■	■
		G13	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 2024–25 monitoring period, the Seagrass Index for the overall Wet Tropics region marginally deteriorated from the previous period but remained ‘poor’ (Figure 34). This was because the slight improvement in seagrass abundance was eclipsed by the continued decline in seagrass resilience. There were differences in the level of overshadowing of the resilience indicator between sub-regions, with the greatest declines in the southern Index.

In the northern Wet Tropics, seagrass abundance marginally improved, but the indicator remained ‘moderate’ (Figure 34). The improvement was driven by increasing scores for reef intertidal and subtidal habitats, as scores for the coastal intertidal habitat declined by a third from the previous period.

Resilience in the seagrass meadows of the northern Wet Tropics has continued to decline over the previous and current periods. While there was an improvement in the resilience of the inshore reef meadows at Green Island in 2024–25, this was overshadowed by the deterioration in the resilience of the coastal intertidal meadows at Yule Point and the reef (both intertidal and subtidal) meadows at Low Isles (Figure 37).

The lower seagrass abundance and resilience scores at Yule Point and Low Isles, suggests these meadows remain vulnerable to adverse or severe disturbances in the immediate future, with their reduced resistance and capacity to recover.

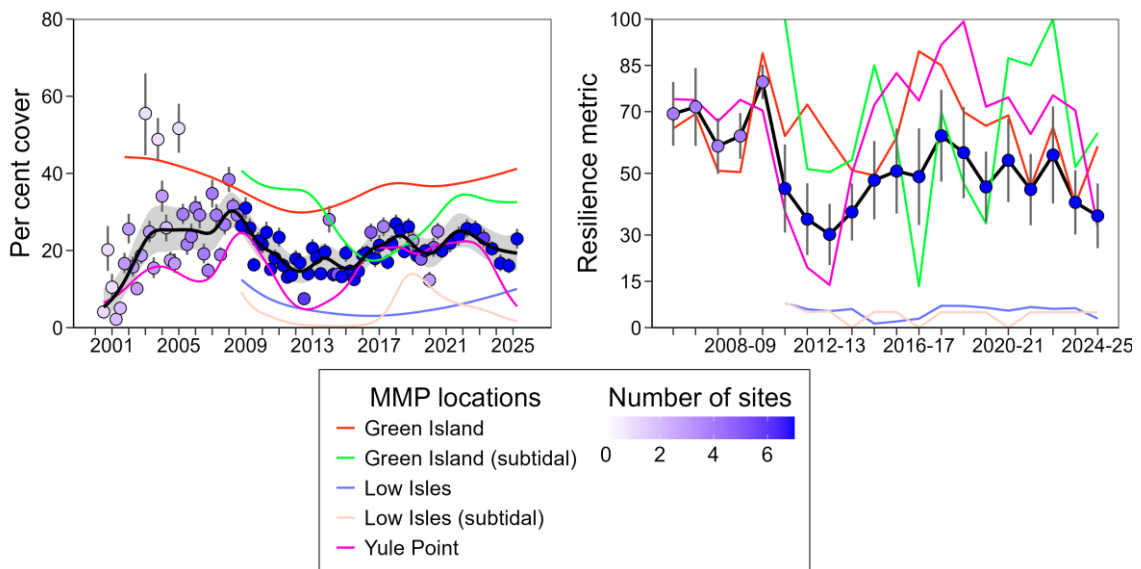


Figure 37. Temporal trends in the northern Wet Tropics seagrass indicators used to calculate the Seagrass Index: a. average (circles,  $\pm$ SE) seasonal abundance (per cent cover) and GAM plots of seagrass abundance trends for each location (coloured lines) and the region (black line with grey shaded area defining 95 per cent confidence intervals); b. average annual resilience metric ( $\pm$ 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 Index declined for the third year after reaching its highest level in 2021–22 (Figure 34). This drop was primarily driven by deterioration in the resilience indicator. Since 2012–13, both the abundance and resilience indicators have shown significant variability, often reflecting an apparent annual lag between abundance and resilience (Figure 34). The abundance indicator, which experienced its first increase in 3 years, was boosted by improved abundances at reef intertidal sites during the late wet season; however, it remained ‘very poor’ (Figure 38). The decline in the resilience indicator appears linked to decreased resilience at individual sites across all habitats, particularly at one coastal intertidal habitat site (LB1), leading to a drop in the location score from ‘poor’ to ‘very poor’ (Figure 38).

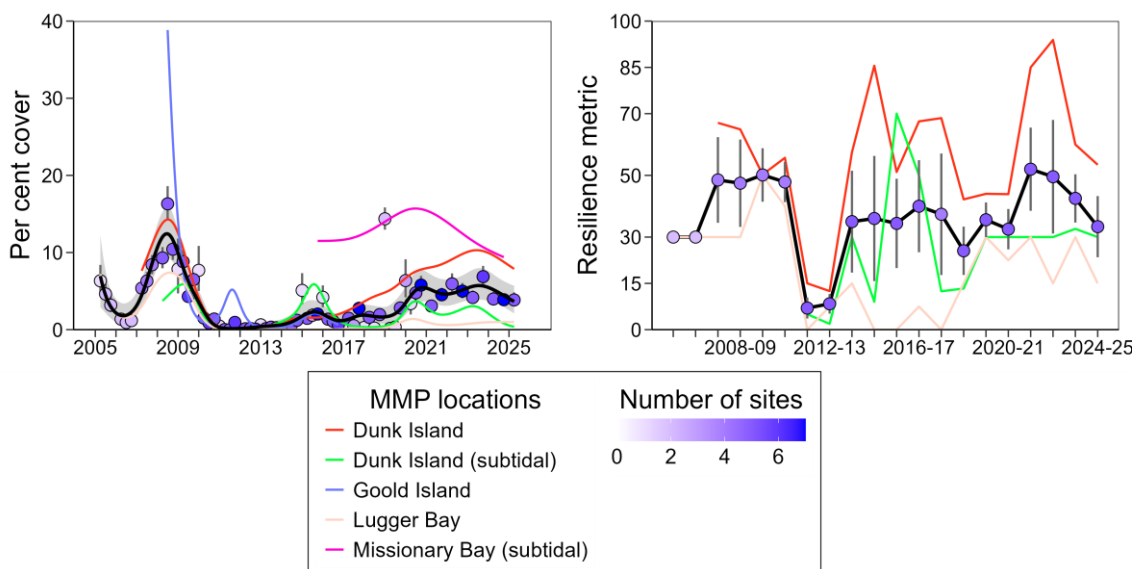


Figure 38. Temporal trends in the southern Wet Tropics seagrass indicators used to calculate the Seagrass Index: a. average (circles,  $\pm$ SE) seasonal abundance (per cent cover) and GAM plots of seagrass abundance trends for each location (coloured lines) and the region (black line with grey shaded area defining 95 per cent confidence intervals); b. average annual resilience metric ( $\pm$ 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 remain 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 also higher at intertidal reef ( $27.7 \pm 2.1$  per cent) than subtidal reef ( $17.1 \pm 2.3$  per cent) or coastal habitats ( $15.2 \pm 1.6$  per cent).

In 2024–25, seagrass abundances improved on average across the northern Wet Tropics, primarily due to minor gains in reef intertidal and subtidal habitats (Figure 39). Reef subtidal seagrass abundances at Green Island increased to above the long-term average after declining over the previous 2 years, while at Low Isles, both intertidal and subtidal abundances remained low and relatively stable (Figure 39). Meanwhile, reef intertidal abundances at Green Island remained around or above the long-term average throughout the 2024-25 period (Figure 39). The intertidal coastal meadows at Yule Point continued to remain low and below the long-term average for the second time in a decade, but shows some slight improvement in early 2025 from declines experienced in early 2024 (Figure 39).

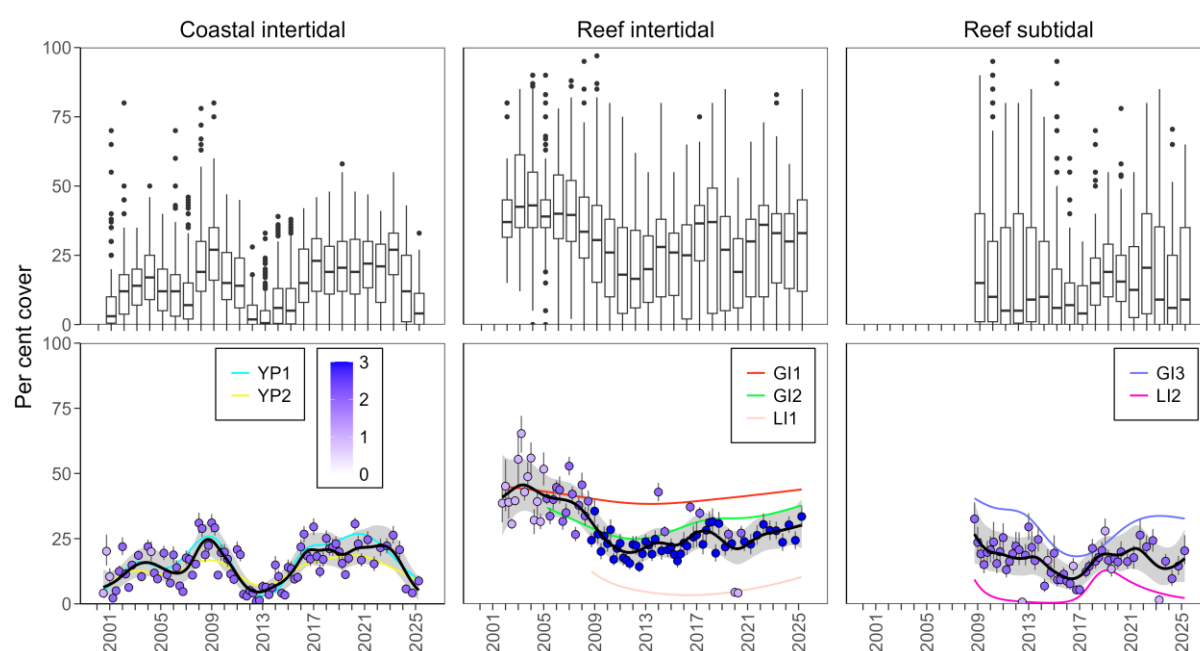


Figure 39. Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the northern Wet Tropics NRM region from 2001 to 2025. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates the 25<sup>th</sup> percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75<sup>th</sup> percentile. Whiskers (error bars) above and below the box indicate the 90<sup>th</sup> and 10<sup>th</sup> percentiles, and the dots represent outlying points. GAM plots (bottom), show trends for each habitat and coloured lines represent individual site trends

In the southern Wet Tropics, long-term seagrass abundance is notably higher at intertidal reef ( $5.0 \pm 1.1$  per cent) compared to subtidal reef ( $2.0 \pm 0.8$  per cent) or intertidal coastal habitats ( $1.6 \pm 0.6$  per cent). However, these abundances were merely a fraction of what has been recorded in the northern sub-region. This is a legacy of periods of complete loss experienced across all habitats for at least 3–6 months in early 2011. Specifically, in Luggier Bay's coastal habitats, seagrass meadows have faced consistent challenges with complete loss for years, marking 2024–25 as the 16<sup>th</sup> consecutive year of abundances significantly below pre-2011 levels (Figure 40). Although recovery has been lethargic, isolated seagrass shoots appeared at Luggier Bay sites in 2016–17, and by late 2018 small patches had established which have varied in the following years, including 2024–25. Coastal subtidal abundances have changed little since monitoring was established in 2015, experiencing a marginal decline in 2024–25 (Figure 40). The abundance of seagrass in intertidal reef

habitat, which had been on an increasing trajectory since 2012–13, slightly decreased in 2024–25 after reaching the highest levels in over a decade the previous year. While abundances at reef subtidal habitats have remained low over the long-term, deteriorating further in 2024–25 to the lowest levels recorded in 7 years (Figure 40).

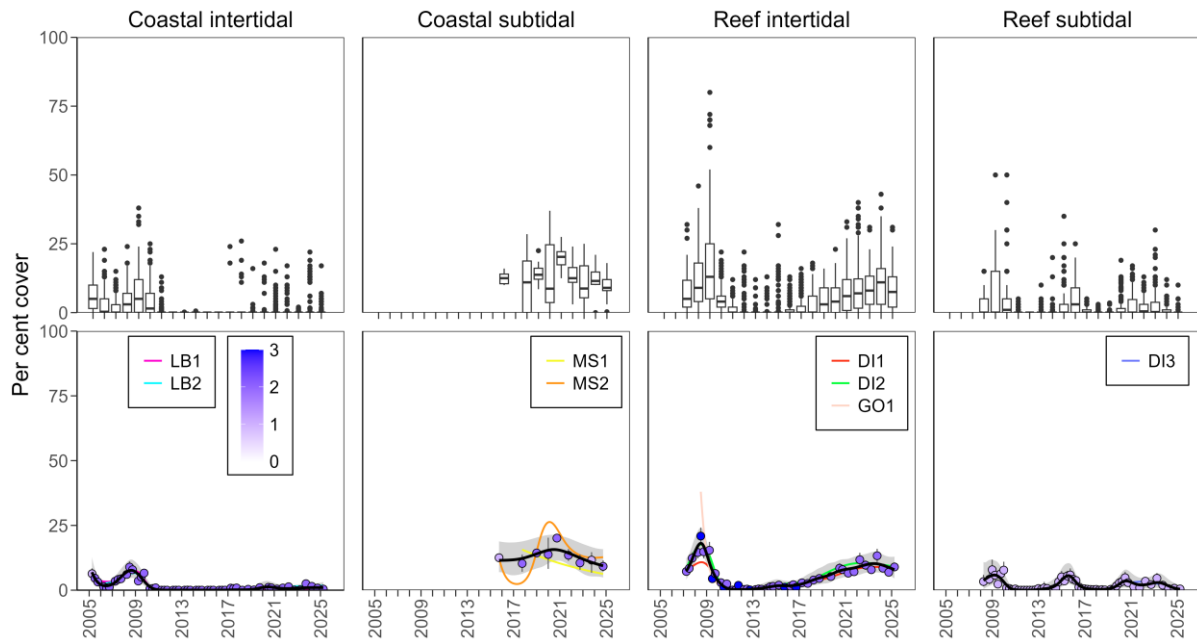


Figure 40. Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the southern Wet Tropics NRM region from 2001 to 2025. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates the 25<sup>th</sup> percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75<sup>th</sup> percentile. Whiskers (error bars) above and below the box indicate the 90<sup>th</sup> and 10<sup>th</sup> 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 (e.g., *Halophila* spp.) in the northern Wet Tropics has remained above the long-term average at reef habitats in 2024–25 (Figure 41). At coastal intertidal habitats (Yule Point), opportunistic species dominate and the proportion of colonising species declined relative to the previous period, remaining below the long-term average.

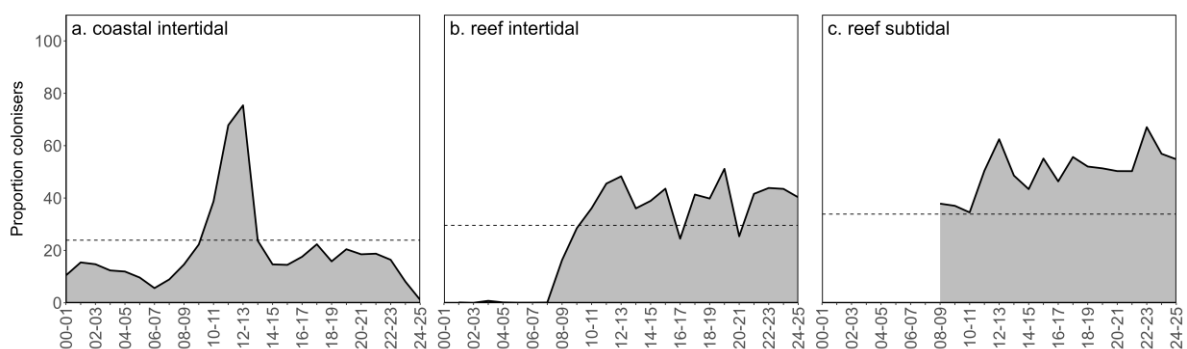


Figure 41. Proportion of seagrass abundance composed of colonising species at inshore intertidal habitats in the northern Wet Tropics region, 2001 to 2025. Dashed line represents Reef long-term average proportion of colonising species for each habitat type.

In the southern Wet Tropics, the proportion of colonising species remains variable across habitats (Figure 42). Coastal intertidal habitats appear unchanged, primarily dominated by opportunistic species, while the subtidal habitats show a greater proportion of colonising

species. Although colonising species remained in low proportions in reef habitats, their proportion slightly increased subtidally during 2024–25.

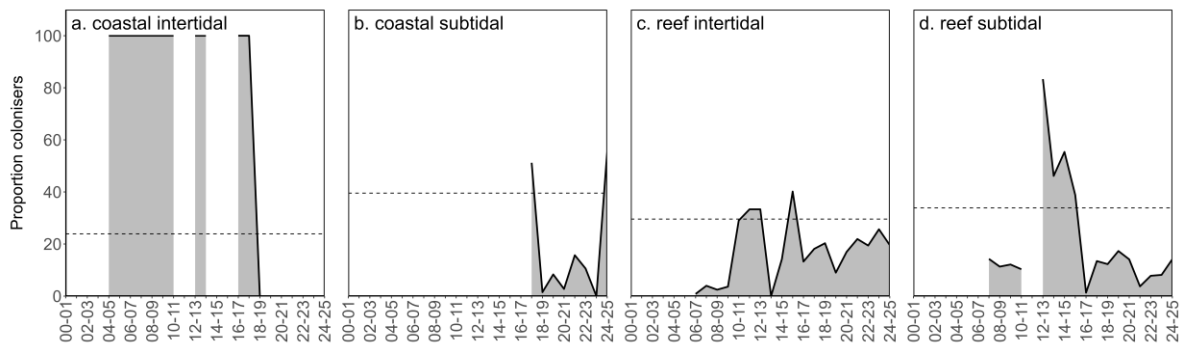


Figure 42. Proportion of seagrass abundance composed of colonising species at inshore intertidal habitats in the southern Wet Tropics region, 2001 to 2025. Dashed line represents Reef long-term average proportion of colonising species for each habitat type.

The spatial extent of seagrass meadows at all monitoring sites continues to show fluctuations both within and across years. In the intertidal reef habitats of the northern Wet Tropics, the relative extent of meadows has remained stable over the last 12 months (Figure 43). Over the long term, the meadow at Green Island shows minimal annual variation, while the fluctuations in the intertidal reef meadows in the northern Wet Tropics are largely influenced by changes at the Low Isles meadow. The most considerable changes in 2024-25 occurred in the coastal intertidal meadows towards the end of period, in the late wet season (Figure 43), where the meadows showed to greatest recovery since Cyclone Jasper and the associated flooding in late 2023.

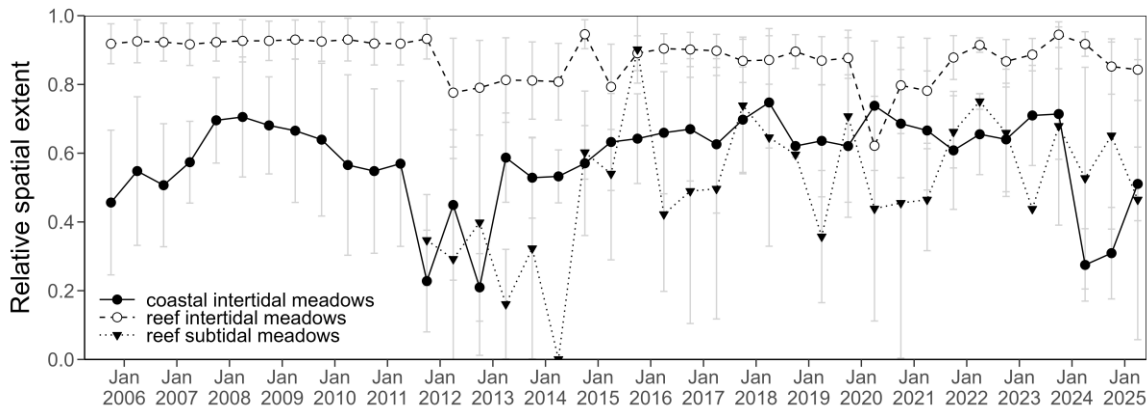


Figure 43. Change in relative spatial extent ( $\pm$ SE) of seagrass meadows within monitoring sites for each habitat and monitoring period across the northern Wet Tropics NRM region, 2005–2025.

In the southern Wet Tropics, seagrass meadows across all habitats suffered extensive damage due to Cyclone Yasi in early 2011 (Figure 44). Since then, intertidal reef meadows have progressively improved, achieving their greatest extent since 2011 in 2024–25 (Figure 44). Meanwhile, subtidal reef meadows have also increased in extent throughout 2024–25, following a significant decline in the previous period (Figure 44). For intertidal coastal habitats, recovery has been severely protracted since 2011, with meadow establishment absent until late 2018 (Figure 44). Following this, isolated seagrass patches have faced challenges in expanding and consolidating within a highly dynamic environment characterised by mobile sediments (pers. obs.).

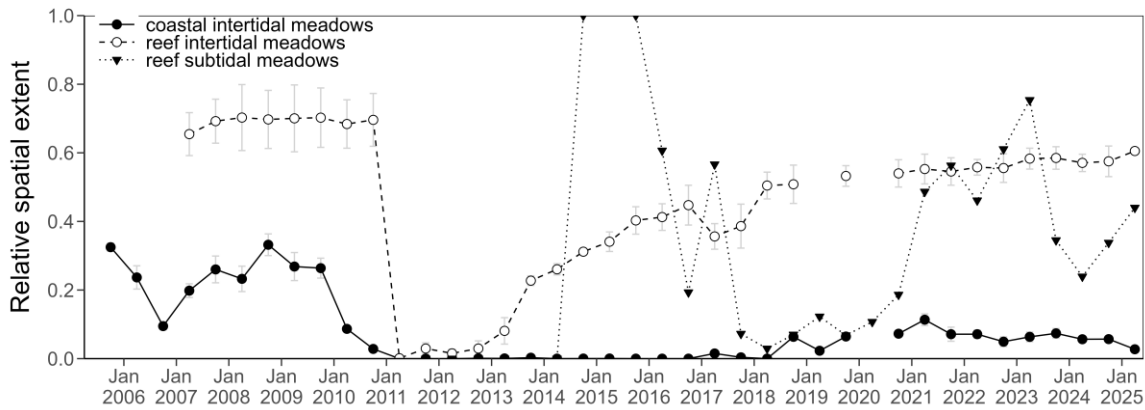


Figure 44. Change in relative spatial extent ( $\pm$ SE) of seagrass meadows within monitoring sites for each habitat and monitoring period across the southern Wet Tropics NRM region, 2005–2025.

5.2.3.3 Seagrass reproductive status

Reproductive effort was measured in both the late dry and late wet seasons across the Wet Tropics. In 2024–25, reproductive effort was the lowest on record for the northern Wet Tropics but not for the southern sub-region (Figure 45 Figure 46).

At the northern coastal intertidal site (Yule Point), no reproductive structures were recorded in 2024–25, an uncommon result given the variability and periodic peaks observed in 2018–19 and 2019–20 (Figure 45). No reproductive structures were detected in intertidal reef habitats, with the only observations occurring in reef subtidal habitats at Low Isles and they were on colonising species (Figure 45).

Seed densities at northern coastal intertidal habitats were well below the long-term average for the second consecutive year. No seeds were recorded at YP2 during either season, although they were present at YP1 in the late wet season (Figure 45). Seed banks remained very low in reef habitats, with no seeds recorded at intertidal or subtidal reef sites in 2024–25. This likely reflects severely reduced reproductive effort among foundational species, as sexual reproductive structures were mainly observed on colonising species whose small seeds are not included in seed bank assessments. Additional contributing factors may include failure to set seed in sparse dioecious populations (Shelton 2008) or rapid seed loss following release due to germination or grazing (Heck and Orth 2006). Seed banks are not assessed at reef subtidal sites.

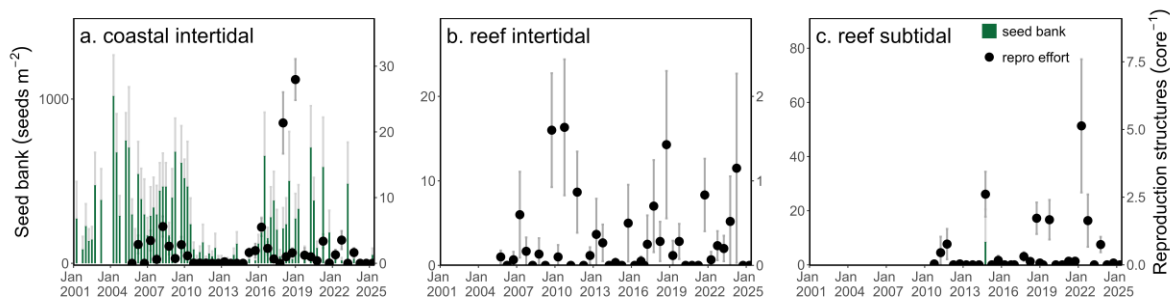


Figure 45. Seed bank and reproductive effort at inshore coastal intertidal and reef intertidal and subtidal habitats in the northern Wet Tropics region, 2001 to 2025. Seed banks presented as the total number of seeds per  $m^2$  of sediment surface (green bars  $\pm$ SE). Reproductive effort presented as the average number of reproductive structures per core (species and sites pooled) (dots  $\pm$ SE). Y-axis labels are different in panel a to those in panels b and c. Seed banks were not assessed at subtidal sites.

In the southern Wet Tropics, sexual reproductive structures were absent from coastal intertidal meadows for the 12th consecutive year (Figure 46). There were also no reproductive structures at reef subtidal habitats for the fifth year in a row. The only habitat with reproductive structures were of colonising species at reef intertidal habitats at DI1 and DI2 in the late dry, but not in the late wet season (Figure 46).

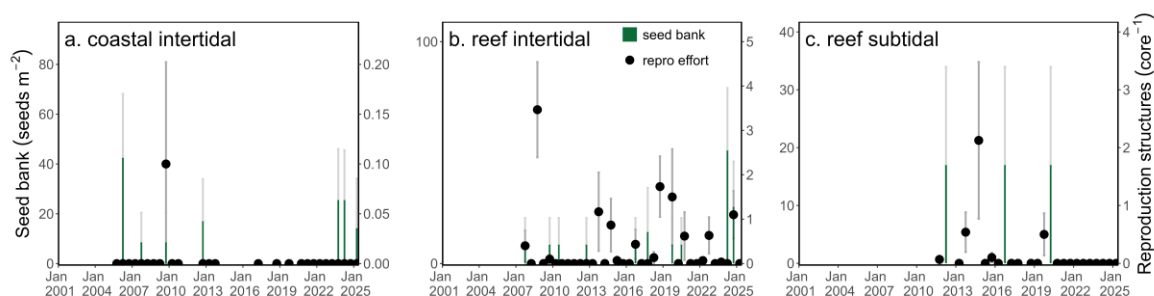


Figure 46. Seeds banks and reproductive effort for inshore coastal intertidal and reef intertidal and subtidal habitats in the southern Wet Tropics region for the late dry and late wet season, 2001 to 2025. Seed banks presented as the total number of seeds per  $m^2$  sediment surface (green bars  $\pm$ SE). Reproductive effort presented as the average number of reproductive structures per core (species and sites pooled) (dots  $\pm$ SE).

#### 5.2.3.4 Resilience

Overall resilience in the northern Wet Tropics declined to poor in 2024–25, down from moderate in 2023–24; continuing a three-year downward trend (Figure 47). The decline was primarily driven by coastal habitats, while resilience improved slightly at reef intertidal and subtidal sites (Table 23).

At Yule Point, abundance at YP2 fell below the low-cover threshold and reproductive structures were absent, resulting in a score category of '1.1' and driving the decline at coastal habitats. At YP1, abundance remained above the 20th percentile but reproductive structures were absent; however, a recent history of reproduction contributed to a score of 50, the highest within category '2.1.2'. Reef intertidal resilience increased compared to the previous year but varied among sites. At Green Island, seagrass cover exceeded thresholds for abundance and composition, though reproductive structures remained absent for over 3 years at GI1 (category '2.1.1'), while GI2 retained category '2.1.2' following reproduction recorded in 2022–23. At Low Isles, colonising species continued to dominate and increase in relative abundance, maintaining a low score within category '1.1', consistent since monitoring began, with a further decline in 2024–25. Reef subtidal resilience improved slightly due to higher scores at Green Island (GI3), where *Thalassia hemprichii* increased in relative abundance despite the absence of reproductive structures, maintaining category '2.1.2'. At Low Isles, the subtidal meadow remained dominated by colonising species, resulting in a persistently low resilience score and continued vulnerability to disturbances such as elevated discharge.

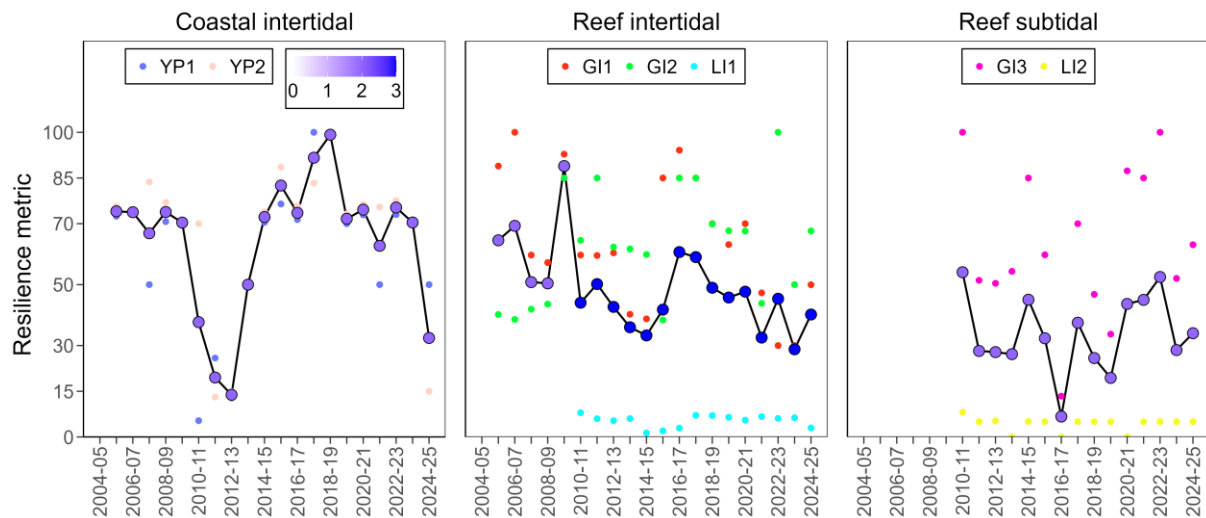


Figure 47. Resilience metric for each habitat type in the northern Wet Tropics. Coloured small points represent different sites. Shades of blue for the larger points indicate the number of sites that contribute to the score.

In the southern Wet Tropics, resilience declined for the third consecutive year in 2024–25, decreasing from ‘moderate’ to ‘poor’. At the coastal intertidal site at Luger Bay (LB2), the meadow exceeded critical per cent cover thresholds and was dominated by opportunistic species, but no flowering was observed, nor was there any recent history of reproduction (Figure 48 Table 23). In contrast, no seagrass was present at LB1, resulting in a site score of zero and contributing to the overall decline. Resilience at reef intertidal sites at Dunk Island decreased slightly at both locations due to the continued absence of reproductive structures of foundational species, despite reproductive activity being recorded within the past 3 years. At DI2, a small reduction in the proportion of persistent species led to a lower score within category ‘2.1.2’, while DI1 recorded the lowest score within the same category (50) due to minimal persistent species. At reef subtidal sites, resilience remained consistent with the previous 4 years, as species composition and per cent cover stayed above critical thresholds, although no reproductive structures of foundational species were observed in 2024–25 or the preceding 3 years.

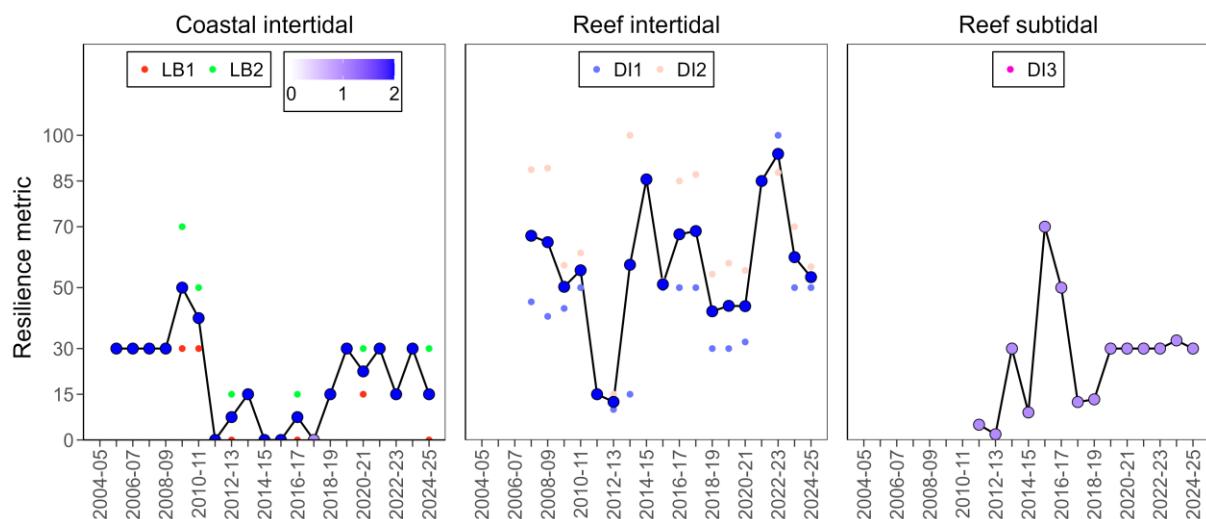


Figure 48 Resilience metric for each habitat type in the southern Wet Tropics. Coloured small points represent different sites. Shades of blue for the larger points indicate the number of sites that contribute to the score. *Epiphytes and macroalgae*

In 2024-25, the epiphyte cover on seagrass leaves in the northern Wet Tropics continued to remain below the long-term average in coastal intertidal habitats for the second consecutive

year. Meanwhile, epiphyte cover in inshore reef intertidal and subtidal habitats fell below the long-term average in the late dry season, before increasing back above the average in the late wet (Figure 49).

Macroalgae cover has remained below the Reef long-term average in both coastal intertidal and reef subtidal habitats during the wet and dry seasons for the eighth consecutive year (Figure 49). In contrast, reef intertidal habitats typically exhibit higher macroalgae coverage, as the algae attach to coarser sediments and coral rubble, sustaining levels above the long-term average for over a decade (Figure 49). This coverage only dips below the long-term average during the occasional wet season as a consequence of increased freshwater and reduced light. In 2024–25, macroalgae cover in reef intertidal habitats was slightly lower than the previous period (Figure 49).

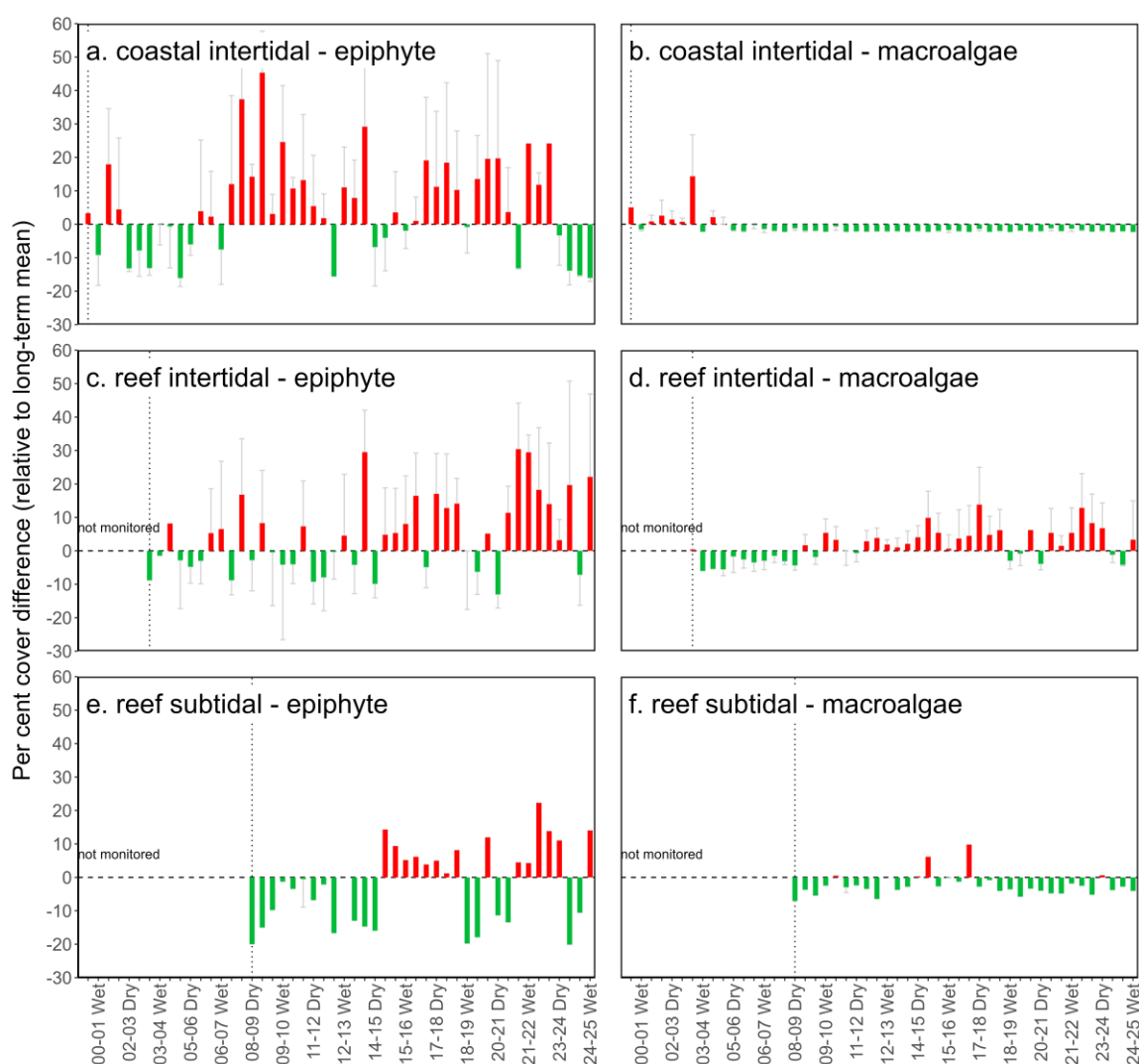


Figure 49. Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average for each inshore intertidal seagrass habitat in the northern Wet Tropics region, 2001–2025 (sites pooled,  $\pm$ SE). Vertical dotted lines represent the first monitoring event for each habitat type. Red and green bars indicate above and below long-term average, respectively.

In the southern Wet Tropics, epiphyte cover in coastal intertidal habitats remained well below the Reef long-term average during 2024–25, similar to the previous period (Figure 50a). Both subtidal coastal and reef habitats also remained below or around the long-term average over the same period (Figure 50c,g). In contrast, epiphyte cover in reef intertidal

habitats was above the long-term average in both the late dry and late wet seasons (Figure 50e).

Macroalgae cover is generally below the Reef long-term average in all habitats in the southern Wet Tropics (Figure 50). Macroalgae cover at the reef subtidal site has varied greatly over the last decade and prior to 2024–25 had remained above the long-term average for the last few years (Figure 50g).

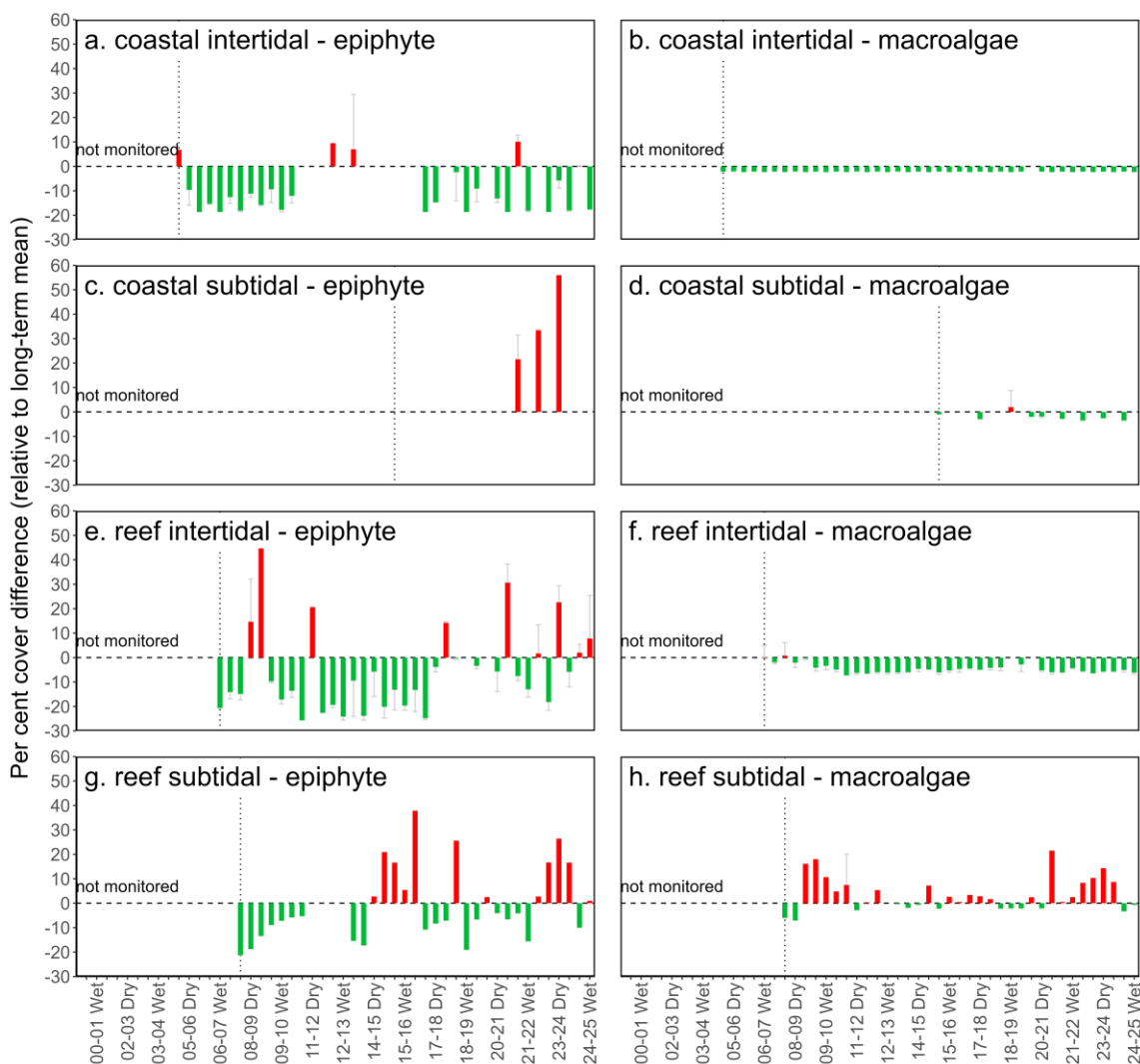


Figure 50. Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average for each inshore intertidal seagrass habitat in the southern Wet Tropics region, 2001–2025 (sites pooled,  $\pm$ SE). Vertical dotted lines represent the first monitoring event for each habitat type. Red and green bars indicate above and below long-term average, respectively.

## 5.3 Burdekin

### 5.3.1 2024–25 Summary

Wet-season rainfall across the Burdekin region was well above the long-term average, and annual river discharge was approximately 7 times the long-term median, dominated by the Burdekin River, which carried nearly ten times the volume of other rivers. Seagrass habitats experienced turbid waters through most of the wet season, with coastal sites primarily influenced by turbid sediment laden waters (WT1) and reef sites around Magnetic Island by turbid algae enriched waters (WT2).

The condition of seagrass meadows throughout the Burdekin NRM region continued to decline in 2024–25, ultimately remaining in a **‘poor’** state (Figure 51). Condition indicators contributing to this were:

- abundance score was ‘poor’
- resilience score was ‘poor’.

The abundance indicator declined for the third consecutive year in a row, resulting in a drop from ‘moderate’ to ‘poor’ in 2024–25. This decline was a consequence of the deterioration in coastal intertidal and reef intertidal habitats, particularly those adjacent to Townsville and Magnetic Island. Conversely, reef subtidal abundance has increased at Magnetic Island. Overall, seagrass abundances have remained below the long-term average for the seventh consecutive year.

The resilience indicator declined in 2024–25, reaching its third lowest score on record and the lowest since 2013. This decline was driven largely by reduced abundance and lack of sexual reproduction among foundational species at reef intertidal habitats. Coastal meadows remained mostly stable, except for Bushland Beach, where abundance dropped below thresholds and sexual reproductive structures were absent. While seed banks at coastal habitats persisted, they were lower than historical levels and absent from reef habitats throughout the period. Reef subtidal resilience improved slightly due to increased cover, but the continued absence of reproductive structures from foundational species has limited the recovery potential of the meadows.

Seagrass meadows in the Burdekin region have demonstrated strong resilience since monitoring began, particularly in their capacity to recover following disturbance. This resilience is likely supported by high species diversity and a well-developed seed bank, both of which enhance the meadows’ ability to adapt to changing conditions. The primary disturbances—episodic wind and storm events (e.g., tropical cyclones) and Burdekin River discharges—are intermittent in nature, allowing periods of recovery between events. However, there are concerning signs for the region’s resilience.

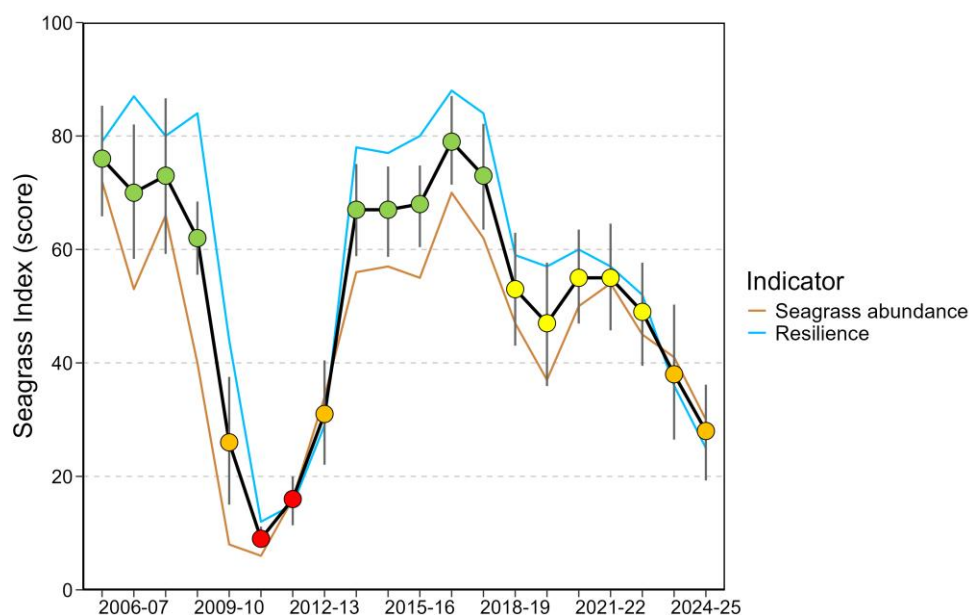


Figure 51. Temporal trend in the Seagrass Index ( $\pm$  SE) with contributing indicator scores for the Burdekin NRM region (averages across habitats and sites). Values are indexed scores scaled from 0–100 ( $\pm$  SE) and graded: ● = very good (81–100), ● = good (61–80), ● = moderate (41–60), ● = poor (21–40), ● = very poor (0–20). NB: Scores are unitless. **Climate and environmental pressures**

Wet-season rainfall across the region was substantially above the long-term average (Figure 51), with the largest anomalies recorded in the northern basins (Figure 6). Annual river discharge was approximately 7 times the long-term median across the region, with consistently high flows from all major rivers (Figure 52). The Burdekin River contributed the majority of discharge—nearly ten times the volume of other rivers—which is characteristic for this region. The Burdekin River experienced 2 considerable flow events during 2024–25. The first, occurring in early February 2025, was the largest, while the second peaked in early April. Of particular note was the length of the February discharge event, where the Burdekin River remained above the moderate flood level for 7 consecutive days and above the minor flood level for 13 consecutive days (Gruber *et al.* 2026). This substantial discharge led to a large flood plume that stretched along the inshore Reef from the mouth of the Burdekin River to the Palm Islands in the north.

Water quality loggers deployed by AIMS at Geoffrey Bay, located 3.8 km from Picnic Bay, recorded 3 notable spikes in turbidity during 2024–25 (Gruber *et al.* 2026). While the timing of the initial spikes aligns with the Burdekin River's discharges, the peak turbidity (~100 NTU) is more indicative of local runoff into the bay, as Burdekin River plumes in Cleveland Bay typically run at about a tenth of the River peak. The turbidity spikes following the major discharge event were likely associated with the resuspension of both newly delivered and pre-existing sediments (Gruber *et al.* 2026).

Inshore seagrass sites were exposed to turbid waters (WT1 and WT2) during all observable wet-season weeks except one at Magnetic Island (MI2); however, persistent cloud cover limited remote sensing observations for much of the season. In 2024–25, exposure to turbid water was close to the long-term average, with coastal sites (BB, SB, and JR) primarily influenced by turbid brownish waters (enriched in sediment and dissolved organic matter) (WT1), and reef sites around Magnetic Island dominated by greenish waters (enriched in algae and dissolved organic matter) (WT2), though data gaps remained due to cloud interference (Figure 52a, b). In the Burdekin NRM region, there was a slight increase (1%) in exposure to risk categories II-IV combined. However, there was an increase in the area of seagrass exposed to category III and IV (moderate and high risk) and a decrease in the area exposed to the lowest risk category.

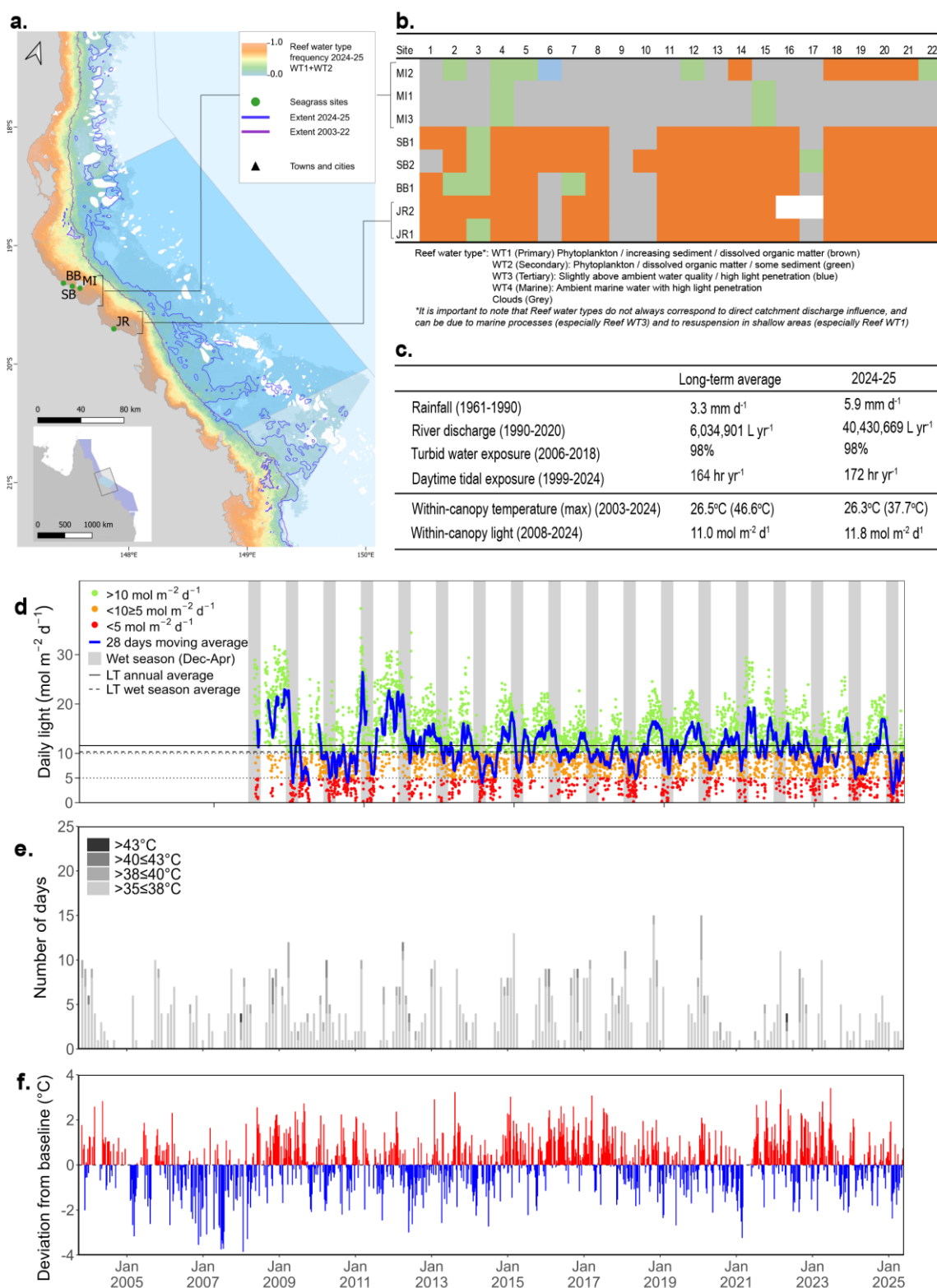


Figure 52. Environmental pressures in the Burdekin region including: a. frequency of exposure to primary (WT1) and secondary (WT2) water from December 2024 to April 2025 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed) (white = no data), also showing the long-term average (2003–2018) exposure boundary (purple line), and the extent for 2024–25 (blue line) (from Gruber *et al.* 2026); b. wet season water type at each site; c. average conditions and max temperature over the long-term and in 2024–25; 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.

Daily light levels at intertidal locations in the Burdekin region averaged 11.8 mol m<sup>-2</sup> d<sup>-1</sup> in 2024–25, exceeding the long-term average of 11.0 mol m<sup>-2</sup> d<sup>-1</sup> (Figure 52c, d). However,

logger malfunction at the coastal sites meant that the 2024–25 regional average was derived from a single reef site (MI1); which typically records higher light levels than the coastal sites that contribute to the long-term average (Figure 108). At MI1, daily light levels peaked immediately prior to the wet season and then declined sharply during the wet season—the second consecutive year in which this pronounced drop has occurred (Figure 108).

Intertidal within-canopy temperatures decreased again this year and were below the long-term average (Figure 52c, f). Maximum intertidal within-canopy temperatures exceeded 35°C for a total of 27 days, which was more than the previous period, but the fifth lowest count of exceedances since 2003–04. The highest temperature for the period was 37.7°C (Magnetic Island, 28 September 2024) (Figure 52).

In 2024–25, daytime tidal exposure exceeded the long-term median at all sites except Jerona (Barratta Creek, Bowling Green Bay) (Figure 52c, Figure 95, Figure 96). This was the first time in 8 years that daytime tidal exposure has surpassed the regional long-term median, potentially heightening certain stresses (such as carbon limitation and desiccation) encountered by seagrasses throughout the region.

The proportion of mud at Jerona (Barratta Creek, Bowling Green Bay) coastal meadows remains much higher than Townsville meadows (Bushland Beach and Shelley Beach) and has persisted well above the Reef long-term average (Figure 120). Following 2018, Townsville coastal meadows have largely been characterised by sand dominated sediments, such as medium sand. Between 2018 and 2020, although the mud levels at Bushland Beach fluctuated, they remained significantly below the long-term average until early 2025, when the mud content surpassed that average (Figure 120). In contrast, reef habitats continue to be primarily dominated by sand sediments, with the mud composition at Cockle Bay (MI2) remaining relatively stable (Figure 121, Figure 122).

### 5.3.3 Inshore seagrass and habitat condition

Three seagrass habitat types were assessed across the Burdekin region in 2024–25, with data from 10 sites (Table 15, Table 20).

Table 15. 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 5 and Table 6.

Habitat	Site code and location		abundance	composition	distribution	reproductive effort	seed banks	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 2024–25 monitoring period, the Seagrass Index for the Burdekin region declined while the grade remained ‘poor’ (Figure 51). The grade is partially shaped by the legacy of previous monitoring periods, which experienced above-average wet season rainfall and river discharge across the region in early 2019 and the effects of which have persisted. Additionally, it reflects the impacts of Cyclone Kirrily during the wet season of 2023–24, along with the elevated rainfall and river discharges during the current period. Over the long-term, the Seagrass Index in the Burdekin NRM has been highly variable as it responds rapidly to changing pressures. Since monitoring commenced in 2005, it has been one of the few regions with scores ranging from ‘good’ to ‘very poor’.

Both indicators declined in 2024–25 compared to the previous year. The 3 successive declines in the annual abundance indicator, leading to a drop from ‘moderate’ to ‘poor’ in 2024–25, appear to be influenced by lower-than-expected abundances during the late dry seasons. Additionally, the impact of Cyclone Kirrily during the wet season of 2023–24, coupled with elevated rainfall and river discharges during 2024–25, has played a significant role.

The resilience indicator has also seen a decline in recent years, falling from ‘moderate’ to ‘poor’ during the previous period, and continuing to decline further throughout 2024–25, to its lowest level in over a decade. The decline in the resilience score for 2024–25 is primarily attributed to significant reductions in the Townsville coastal intertidal habitat, especially at Bushland Beach, as well as the Magnetic Island reef intertidal habitat, particularly at Picnic Bay M11.

Examination of the indicators over the long-term show declines from 2009–2011 as a consequence of the multiple years of above-average rainfall and severe weather, followed by rapid recovery. However, based on these historical trends, the recovery of seagrass habitats abated in 2022, followed by declines at several sites that fell significantly below historical peaks (Figure 53).

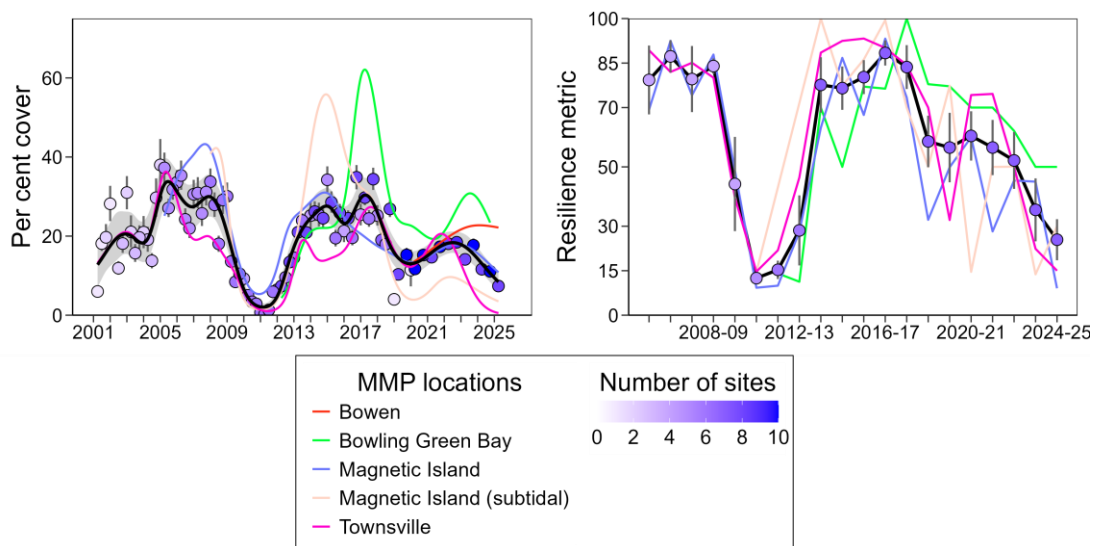


Figure 53. Temporal trends in the Burdekin seagrass indicators used to calculate the Seagrass Index: a. average (circles,  $\pm$ SE) seasonal abundance (per cent cover) and GAM plots of seagrass abundance trends for each location (coloured lines) and the region (black line with grey shaded area defining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

In the Burdekin region, overall seagrass abundance declined for the second consecutive year in 2024–25. This decline was a consequence of deteriorations in coastal intertidal and reef intertidal habitats. Conversely, reef subtidal abundance increased overall for the region (Figure 54).

The abundance of seagrass in the Burdekin region has exhibited a pattern of loss followed by recovery throughout the duration of the MMP. Between 2008–09 and 2010–11, losses occurred as a result of multiple consecutive years of above-average rainfall (incl. river discharge) and severe weather (Cyclone Yasi). From 2011, seagrass rapidly recovered. However, since 2014, recovery has varied across different habitats (Figure 54). In the 2017–18 period, coastal habitats increased to their highest abundance since 2001, but this was quickly followed by significant declines in 2018–19. During that year, the region experienced notable decreases in abundance, particularly in reef subtidal and coastal intertidal habitats. The onset of recovery occurred in coastal intertidal habitats within 12 to 18 months, with abundances continuing to improve above the long-term average in 2022, but subsequently declining in the late wet 2022–23. Similarly, recovery in reef intertidal and subtidal habitats occurred for the first few of years following the 2017 to 2019 declines, however, the improvement over time became more protracted, with only minimal changes in abundance over the years. In 2022–23, intertidal abundances started deteriorating and in the following year there was a significant loss of subtidal abundances (Figure 54). In 2024–25, abundances in reef intertidal habitats continued to decline, despite improvements in subtidal habitats.

An examination of the long-term abundances across the Burdekin region indicates no significant regional trend (from first measure to 2024–25). However, a significant decline has been observed at one of the reef intertidal sites (MI2) and one coastal intertidal site (SB2) since monitoring commenced (Table 22).

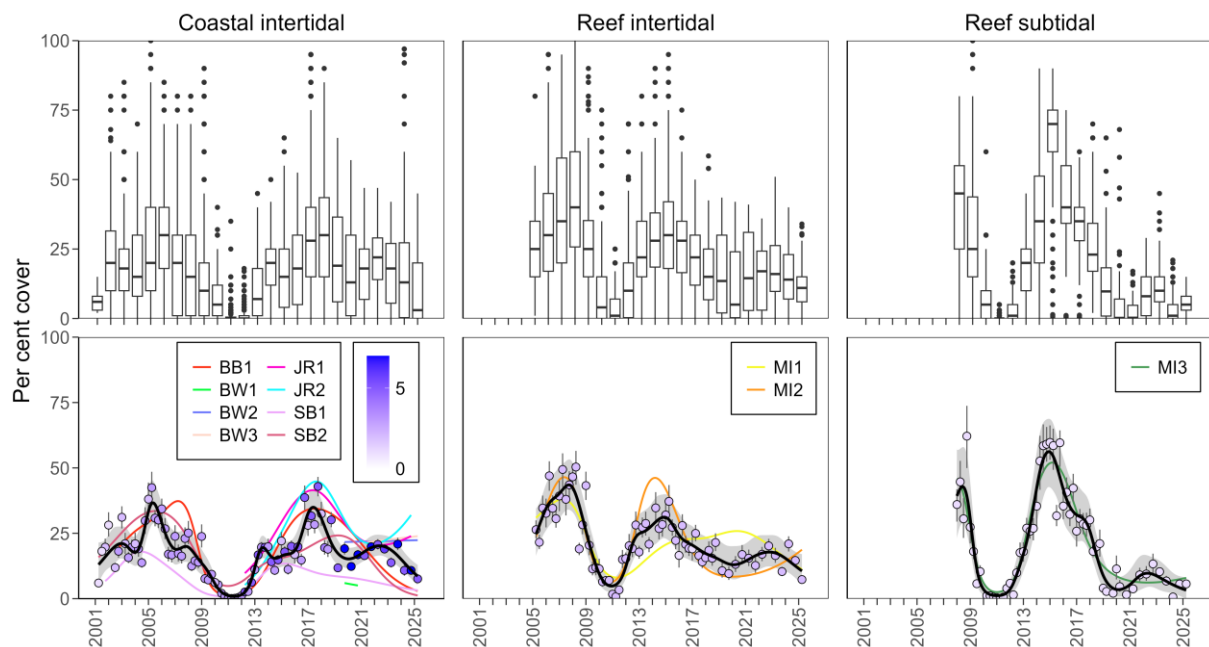


Figure 54. Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the Burdekin NRM region from 2001 to 2025. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates the 25<sup>th</sup> percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75<sup>th</sup> percentile. Whiskers (error bars) above and below the box indicate the 90<sup>th</sup> and 10<sup>th</sup> percentiles, and the dots represent outlying points. GAM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.

Foundational seagrass species maintained dominance in coastal intertidal and reef subtidal meadows during 2024–25. The proportion of species (*Halophila ovalis*) with colonising traits, remained above the long-term average for reef intertidal habitats in the region (Figure 55). The intertidal reef meadow at Cockle Bay (M12) has been dominated by *H. ovalis* since early 2019, following severe flood-related impacts, while *Cymodocea serrulata* has been absent since 2021–22. Colonising species play a key role in recovery after disturbance (Kilminster *et al.* 2015); however, their increased prevalence indicates ongoing localised disturbance delaying recovery. In contrast, coastal and reef subtidal habitats continue to be dominated by opportunistic species (*Halodule uninervis*, *Zostera muelleri*, and *C. serrulata*). Opportunistic foundational species exhibit strong tolerance to acute disturbances through resource reallocation (Collier *et al.* 2012b), suggesting that the current species composition in these habitats contributes to the sustained resilience of Burdekin seagrass meadows.

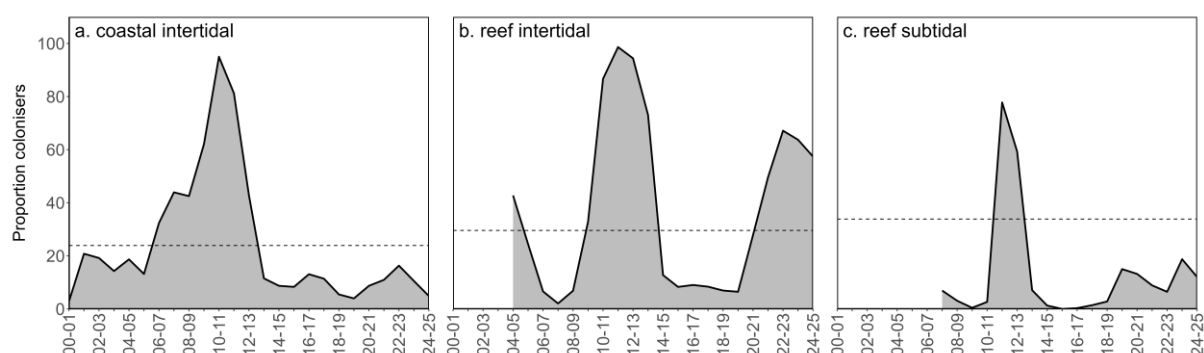


Figure 55. Proportion of seagrass abundance composed of colonising species at inshore intertidal habitats in the Burdekin region, 2001 to 2025. Dashed line represents Reef long-term average proportion of colonising species for each habitat type.

Prior to the 2023–24 wet season, the spatial extent of seagrass meadows in the Burdekin region had either been increasing or remained relatively stable for approximately 5 years. However, following the impact of Cyclone Kirrily in early 2024, these extents declined to their lowest levels in years (Figure 56). The most substantial losses occurred in the coastal meadows, particularly BB1 and SB1, where physical disturbances from wave action on the intertidal banks led to the smallest meadow extents since monitoring began. Reef subtidal meadows similarly fell to their lowest extents in a decade. Conversely, the reef intertidal meadows were the least affected (Figure 56), likely benefiting from their location in small bays that offered greater shelter from the damaging waves. During the 2024–25 period, meadow extents increased in the late dry season, but saw another decline in the wet season, with the exception of the reef subtidal meadows, which continued to show improvement (Figure 56).

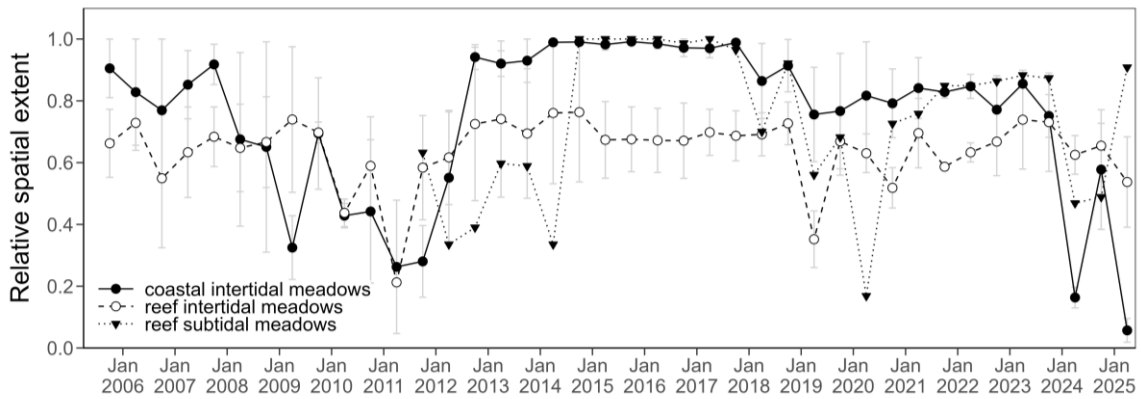


Figure 56. Change in spatial extent ( $\pm$  SE) of seagrass meadows within monitoring sites for each inshore intertidal habitat and monitoring period across the Burdekin region, 2005–2025.

### 5.3.3.3 Seagrass reproductive status

Reproductive effort has been highly variable across habitats in the Burdekin region over the long term, particularly in coastal meadows where anomalously high levels are occasionally recorded during periods of high abundance (Figure 57). Reproductive effort increased at coastal intertidal sites in 2023–24 relative to 2022–23, but remained well below the long-term average. In 2024–25, reproductive effort was low and confined to the coastal sites at Jerona, where it was limited to colonising species (*Halophila ovalis*), with no contribution from foundational species. At reef intertidal sites, reproductive effort during the late dry season reached the highest level in 11 years but was again solely from colonising species, while no reproductive structures were recorded at the reef subtidal site in either season.

Seed banks persisted across the region, ranging from 272 to 1103 seeds  $m^{-2}$  at coastal intertidal sites near Townsville—higher than most other Reef regions but lower than historical levels for these habitats—and were absent from some sites in the late dry season. Seeds were not detected in reef intertidal habitats during either the wet or dry season (Figure 56a), and seed banks are no longer measured at subtidal sites.

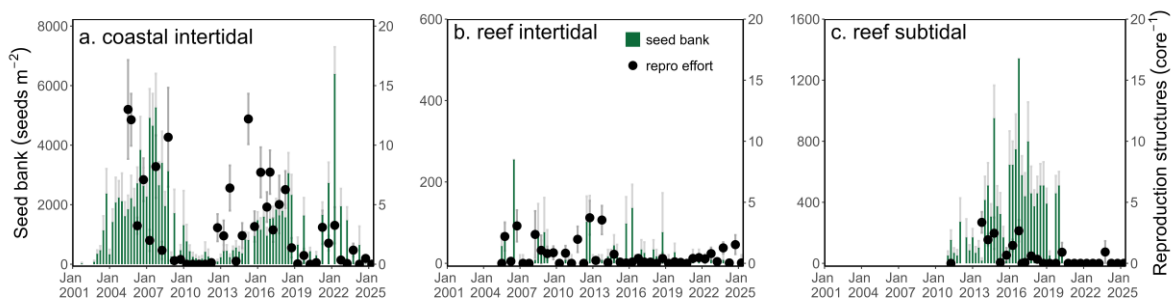


Figure 57. Seedbank and reproductive effort at inshore coastal intertidal and reef subtidal and intertidal habitats in the Burdekin region, for late dry season, 2002–25 Seed bank presented as the total number of seeds per  $m^2$  sediment surface (green bars  $\pm$ SE). Reproductive effort for the late dry season and late wet season presented as the average number of reproductive structures per core (species and sites pooled) (dots  $\pm$ SE). NB: Y-axis scale for seed banks differs between habitats and seeds not assessed at subtidal sites since 2020.

### 5.3.3.4 Resilience

The overall resilience score for the Burdekin region declined in 2024–25 to the third lowest on record and the lowest since 2013 (Figure 51). This decline was primarily driven by reef intertidal sites (Figure 58). At coastal sites, resilience decreased slightly due to a decline at

Bushland Beach (BB1), where seagrass abundance fell below the low-cover threshold and no reproductive structures were recorded, resulting in category ‘1.1’. Scores at other coastal sites remained unchanged from the previous year. A substantial reduction occurred at reef intertidal sites, particularly at MI1, where abundance fell below the low-cover threshold but limited reproductive structures from colonising species maintained category ‘1.2’. At MI2, colonising species continued to dominate, maintaining the lowest resilience score on record for the site. In contrast, the reef subtidal site showed a slight increase in resilience, with cover rising above the low-abundance threshold and moving into category ‘2’. However, reproductive structures remained absent, there was no recent history of reproduction, and foundational species were present only in very low proportions.

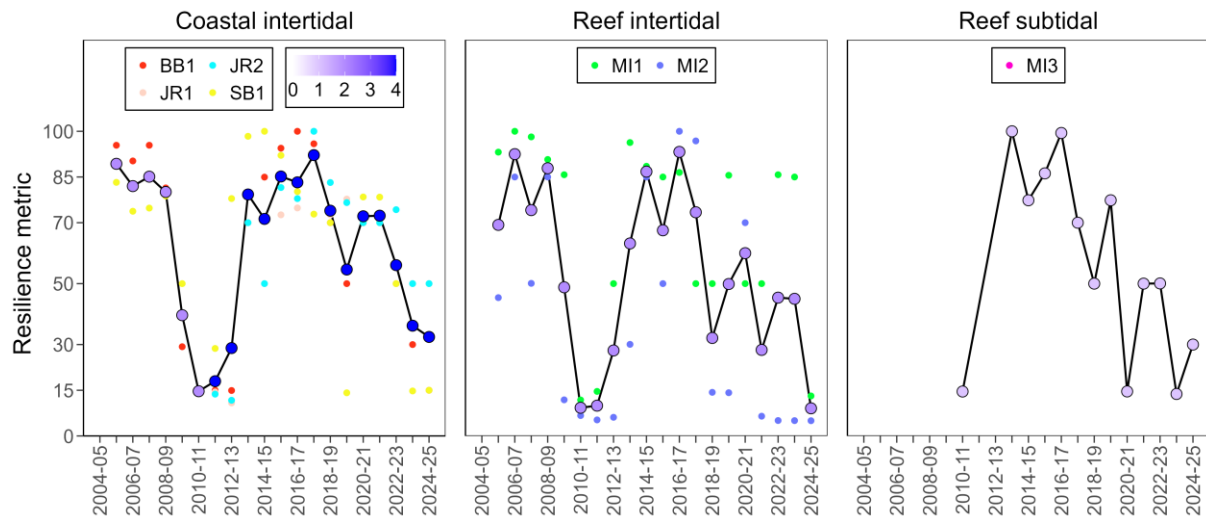


Figure 58. Resilience metric in each habitat in the Burdekin, 2006 to 2025. Coloured small points represent different sites. Shades of blue for the larger points indicate the number of sites that contribute to the score.

### 5.3.3.5 Epiphytes and macroalgae

Epiphyte cover on seagrass leaf blades during the 2024–25 period dropped below the long-term average in all habitats across the region (Figure 59). Meanwhile, macroalgae abundance in 2024–25 continued to remain low, staying below the long-term average for the fifth consecutive year across all seagrass habitats (Figure 59).

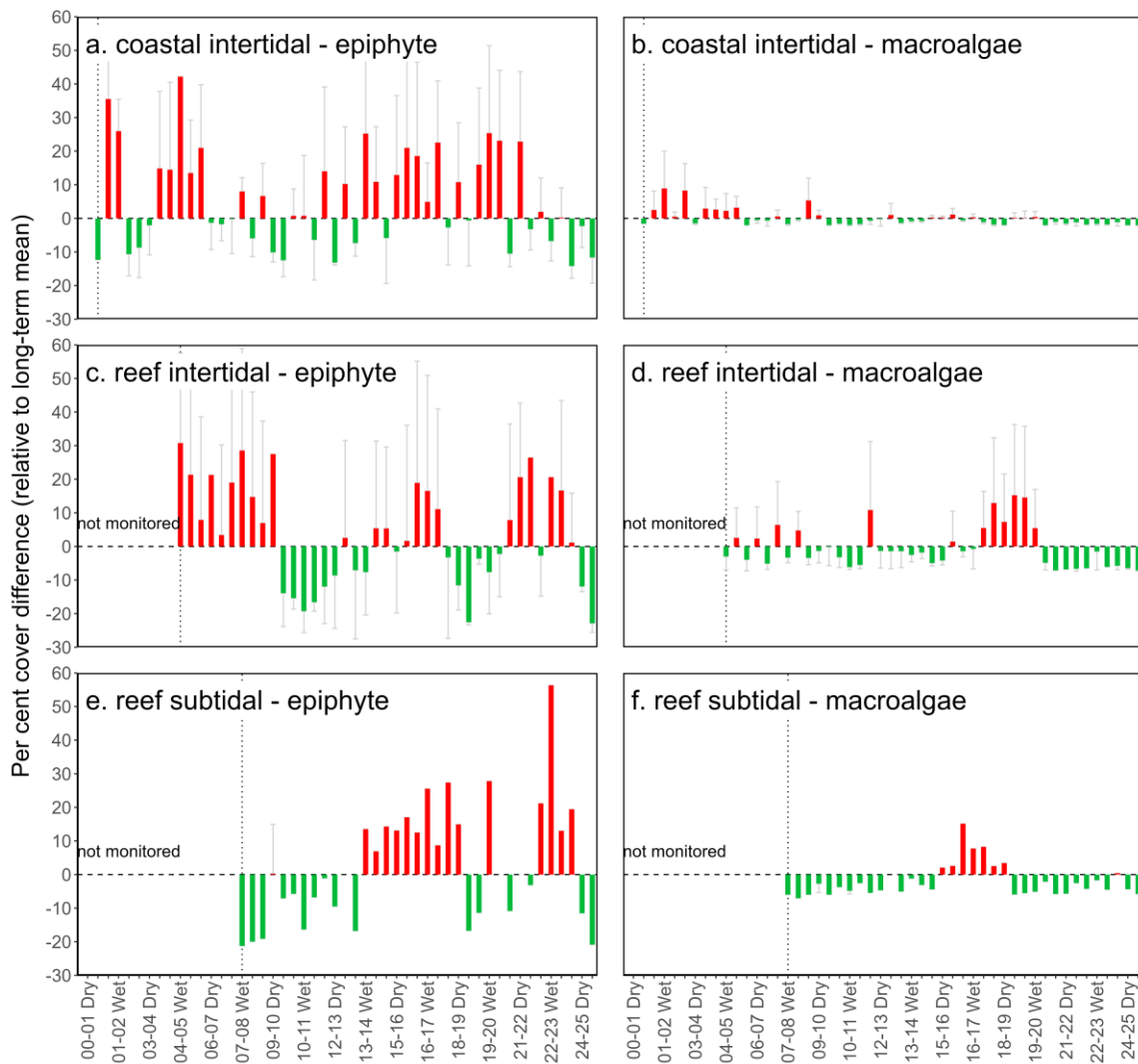


Figure 59. Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average for each inshore intertidal seagrass habitat in the Burdekin region, 2001–2025 (sites pooled,  $\pm$ SE). Vertical dotted lines represent the first monitoring event for each habitat type. Red and green bars indicate above and below long-term average, respectively.

## 5.4 Mackay–Whitsunday

### 5.4.1 2024–25 Summary

Environmental conditions were less optimal for seagrass than the long-term average in the Mackay–Whitsunday region in 2024–25. In 2024–25, rainfall and river discharge in the Mackay–Whitsunday region were above the long-term average, with the greatest deviations in the northern catchments and discharge 2.6 times the regional mean. Exposure of inshore seagrass to turbid waters was below the long-term average, with northern sites such as Hideaway Bay and Hamilton Island more frequently influenced by WT2, while southern sites were dominated by WT1. Daily light levels were lower than average across the region, with the lowest values ( $<5 \text{ mol m}^{-2} \text{ d}^{-1}$ ) recorded at Midge Point during the wet season.

Inshore seagrass meadows across the Mackay–Whitsunday NRM region marginally declined in overall condition in 2024–25, and the condition grade remained **'moderate'** (Figure 60). Indicators for the overall condition score were:

- abundance was 'poor'
- resilience was 'moderate'.

Seagrass condition in the Mackay–Whitsunday region has fluctuated between 'poor' and 'moderate' since 2010–11, which appears to be due to a range of environmental pressures at both regional and local scales.

The seagrass abundance score deteriorated to 'poor' in 2024–25, after showing improvements in the previous 2 years. Declines were driven by deterioration in estuary intertidal and coastal subtidal habitats. The only habitat that maintained its abundance score was coastal intertidal, where any improvements at the site level were counterbalanced by declines.

Resilience in the Mackay–Whitsunday region remained moderate and stable for the fourth consecutive year in 2024–25, with slight increases at coastal sites and declines at estuarine habitats. Reef intertidal and subtidal sites remained stable but continued to lack reproductive structures, indicating limited recovery potential despite adequate abundance and species composition. Seeds of foundational species were also absent from reef sites.

Up until 2016–17 period, the condition of seagrass in the Mackay–Whitsunday region had been improving from its lowest levels in 2010–2011. However, following Cyclone Debbie in March 2017, the condition deteriorated to 'poor'. Since then, the region's recovery from previous disturbances has faced challenges, with the Index fluctuating between 'poor' and 'moderate', though showing gradual improvement. Nevertheless, in the previous period, the trend of recovery abated and condition has continued to decline, albeit slightly, in 2024–25. While this decline is mainly influenced by the abundance indicator, the moderate and stable resilience of the meadows indicates a reasonable potential for recovery across most habitats, provided that favourable conditions arise during the wet season of 2025–26.

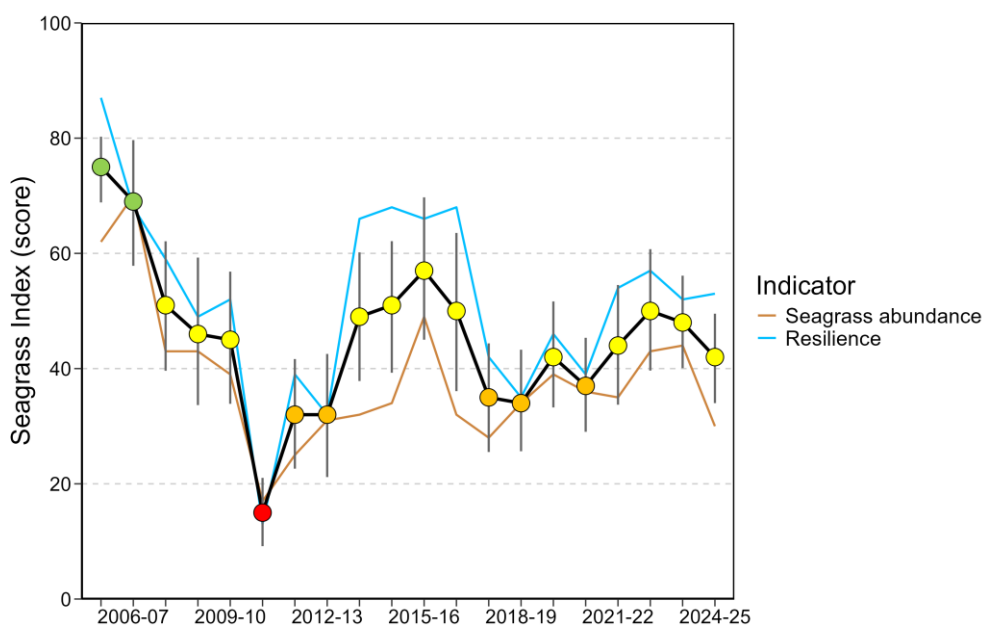


Figure 60. Temporal trend in the Seagrass Index ( $\pm$  SE) with contributing indicator scores for the Mackay–Whitsunday NRM region (averages across habitats and sites). Values are indexed scores scaled from 0–100 ( $\pm$  SE) and graded: ● = very good (81–100), ● = good (61–80), ● = moderate (41–60), ● = poor (21–40), ● = very poor (0–20). NB: Scores are unitless.

#### 5.4.2 Climate and environmental pressures

Environmental conditions were less optimal for seagrass than the long-term average in the Mackay–Whitsunday region in 2024–25. While no cyclones impacted the region during this time, rainfall was above the long-term average across all basins, with the largest anomalies recorded in the northern catchments (Figure 6). Regional river discharge was 2.6 times the long-term median, with all basins showing above-average flows (Table 11). The Don River, the northernmost and smallest of the region’s rivers, exhibited the highest relative deviation despite its low total discharge volume.

Exposure of inshore seagrass to turbid waters during the wet season was below long-term averages (Figure 61a, c). Estuarine and coastal sites from Lindeman Island southwards were exposed to turbid sediment laden waters (WT1) and turbid algae enriched waters (WT2) waters for most of the wet season, with considerable variability among habitats (Figure 60b). Northern sites, including Hideaway Bay (HB2), Hamilton Island (HM3), Midge Point (MP2), and Newry Bay (NB1), were more frequently exposed to WT2 than WT1, while southern sites were predominantly influenced by WT1 (Figure 8, Figure 60b).

The risk of exposure of seagrass to the water types are assessed in the MMP inshore water quality report (Gruber *et al.* 2026). In the Mackay–Whitsunday NRM region in 2024–25, there was a decrease (-7 %) in the area that seagrass was at risk to water quality exposure compared to the long-term average. This was driven by declines in risk of exposure to the lowest risk category (II), while there was a small increase in exposure to the moderate level of exposure (III).

Daily light levels across the region were lower than the long-term average (Figure 9, Figure 61c, Figure 109). Light logger malfunctions, particularly during the wet season, limited data availability, although Midge Point provided the most complete records, showing the lowest light levels ( $<5 \text{ mol m}^{-2} \text{ d}^{-1}$ ) for much of the wet season. These low-light periods were partly offset by near-average dry-season values recorded at several other sites.

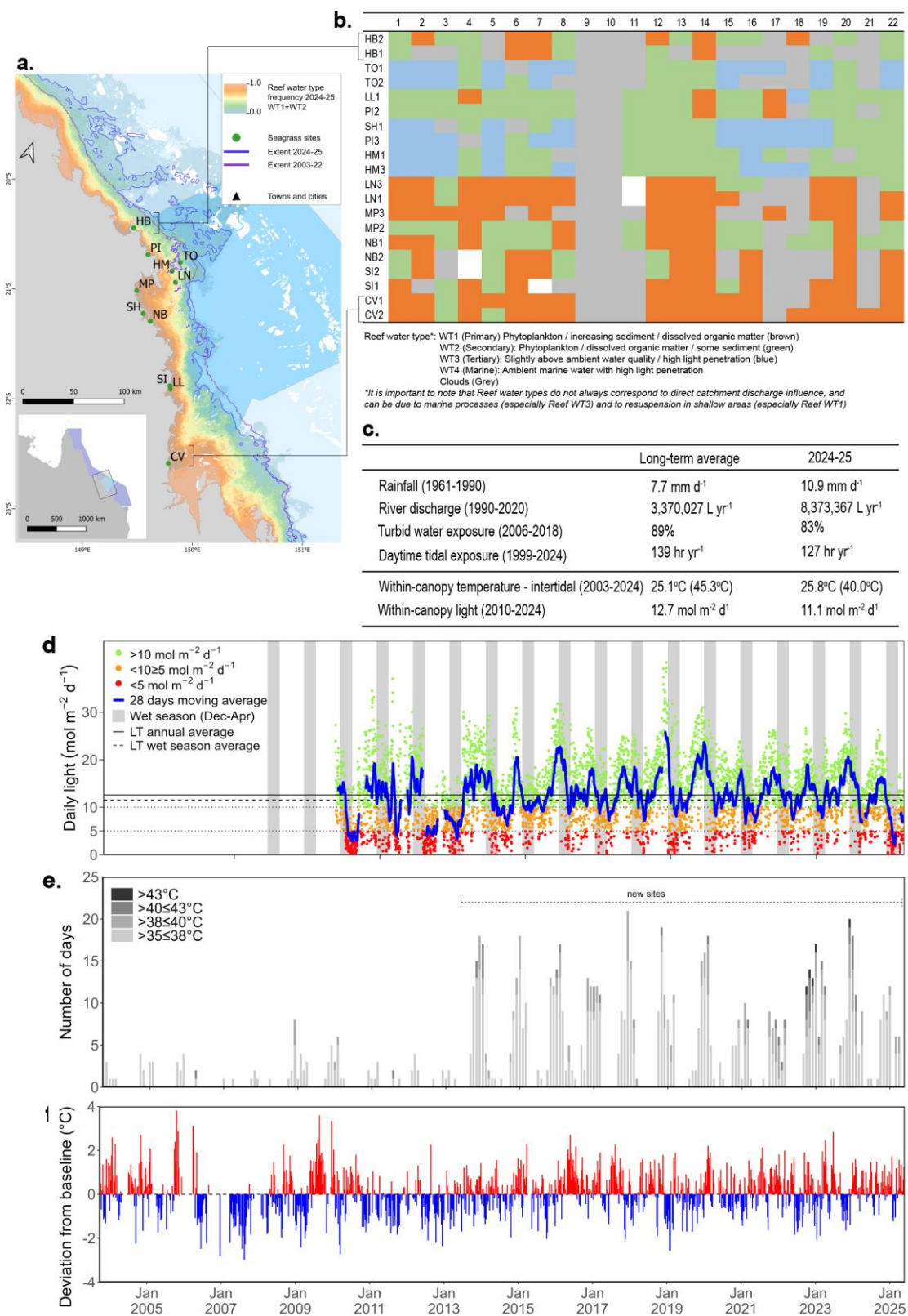


Figure 61. Environmental pressures in the Mackay–Whitsunday NRM region including: a. frequency of exposure to primary (WT1) and secondary (WT2) water from December 2024 to April 2025 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed) (white = no data), also showing the long-term average (2003–2018) exposure boundary (purple line), and the extent for 2024–25 (blue line) (from Gruber *et al.* 2026); b. wet season water type at each site; c. average conditions and max temperature over the long-term and in 2024–25; 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.

During the 2024–25 reporting period, intertidal within-canopy temperatures were above the long-term average (Figure 61c,f). Maximum temperatures exceeded 35°C for a total of 55 days, including reaching a maximum of 40.0°C (Midge Point, 23 January 25 at 2:30pm), which was much less than the previous 2 years (Figure 61e, f). Daytime tidal exposure for the region in 2024–25 was below the long-term average (Figure 61c, Figure 97), and lower than the previous period, which may have resulted in decreased desiccation stresses at the intertidal sites.

Monitoring of sediment grain size in Mackay–Whitsunday seagrass habitats revealed notable short- and long-term changes in 2024–25. Mud content increased in estuarine and coastal habitats compared with the previous period (Figure 123, Figure 124). Estuarine sites at Sarina Inlet became unusually sandy in 2022, with the proportion of mud well below the long-term average, but mud levels have since been gradually increasing although remaining below average in 2024–25. In contrast, coastal habitats now exceed the long-term average mud content at all sites except Clairview (Figure 124). Fine sediments across the region decrease with distance from shore, and reef habitats generally remain dominated by fine to medium sands, with little change in 2024–25 (Figure 125).

### 5.4.3 Inshore seagrass and habitat condition

Five seagrass habitat types were assessed across the Mackay–Whitsunday region this year, with data from 23 of the 24 long-term monitoring sites in 2024–25 (Table 16, Table 20).

Table 16. List of data sources of seagrass and environmental condition indicators for each seagrass habitat type in the Mackay–Whitsunday NRM region. Open square indicates not measured in 2024–25, blank cells indicate data not usually collected/measured at site. ^drop camera sampling (QPWS), \*Seagrass-Watch. For site details see Table 5 and Table 6.

Habitat	Site		abundance	composition	distribution	reproductive effort	seed banks	meadow sediments	epiphytes	macroalgae
estuarine intertidal	SI1	Sarina Inlet	■	■	■	■	■	■	■	■
	SI2	Sarina Inlet	■	■	■	■	■	■	■	■
coastal intertidal	CV1*	Clairview	■	■			■	■	■	■
	CV2*	Clairview	■	■			■	■	■	■
	LL1*	Llewellyn Bay	■	■				■	■	■
	MP2	Midge Point	■	■	■	■	■	■	■	■
	MP3	Midge Point	■	■	■	■	■	■	■	■
	PI2*	Pioneer Bay	■	■				■	■	■
	PI3*	Pioneer Bay	■	■				■	■	■
	SH1*	St Helens	□	□				□	□	□
coastal subtidal	NB1^	Newry Bay	■	■					■	■
	NB2^	Newry Bay	■	■					■	■
reef intertidal	HM1	Hamilton Island	■	■	■	■	■	■	■	■
	HM3	Hamilton Island	■	■	■	■	■	■	■	■
	HB1*	Hydeaway Bay	■	■				■	■	■
	HB2*	Hydeaway Bay	■	■				■	■	■
	LN3	Lindeman Is	■	■	■	■	■	■	■	■
reef subtidal	CH4^	Cid Harbour	■	■					■	■
	CH5^	Cid Harbour	■	■					■	■
	LN1	Lindeman Is	■	■	■	■	■	■	■	■
	TO1^	Tongue Bay	■	■					■	■
	TO2^	Tongue Bay	■	■					■	■
	WB1^	Whitehaven Bch	■	■					■	■
	WB2^	Whitehaven Bch	■	■					■	■

### 5.4.3.1 Seagrass Index and indicator scores

In the 2024–25 monitoring period, the Mackay–Whitsunday region Seagrass Index marginally decreased from the previous year, and remained **‘moderate’** (Figure 62). The decrease was due to a decline in the abundance indicator offsetting the marginal increase in resilience indicator. The resilience score marginally improved and remained ‘moderate’, largely due to improvements in coastal intertidal habitats. While the resilience scores for estuary intertidal and coastal subtidal remained fairly stable, there was a marginal decline in the estuarine intertidal score.

The abundance score declined from ‘moderate’ to ‘poor’, with declines observed throughout the region to differing degrees across various habitats and sites. The most significant reductions occurred in estuary intertidal and coastal subtidal habitats, which dropped by three-quarters and three-fifths, respectively. The only habitat that maintained its abundance score was coastal intertidal, where any improvements at the site level were counterbalanced by declines.

The Index has been varying between poor and moderate since 2011–12 when it recovered from the impacts of the 2010–11 extreme weather events. In 2016–17 the improving trend abated and abundance declined as a consequence of Cyclone Debbie (Figure 62). The following year both abundance and resilience declined, and in 2018–19 the Index reached its lowest level since 2012–13. Since then, the region's recovery has faced challenges, with the Index fluctuating between ‘poor’ and ‘moderate’, though showing gradual improvement. In the previous period, the Index marginally declined and condition has continued to decline in 2024–25.

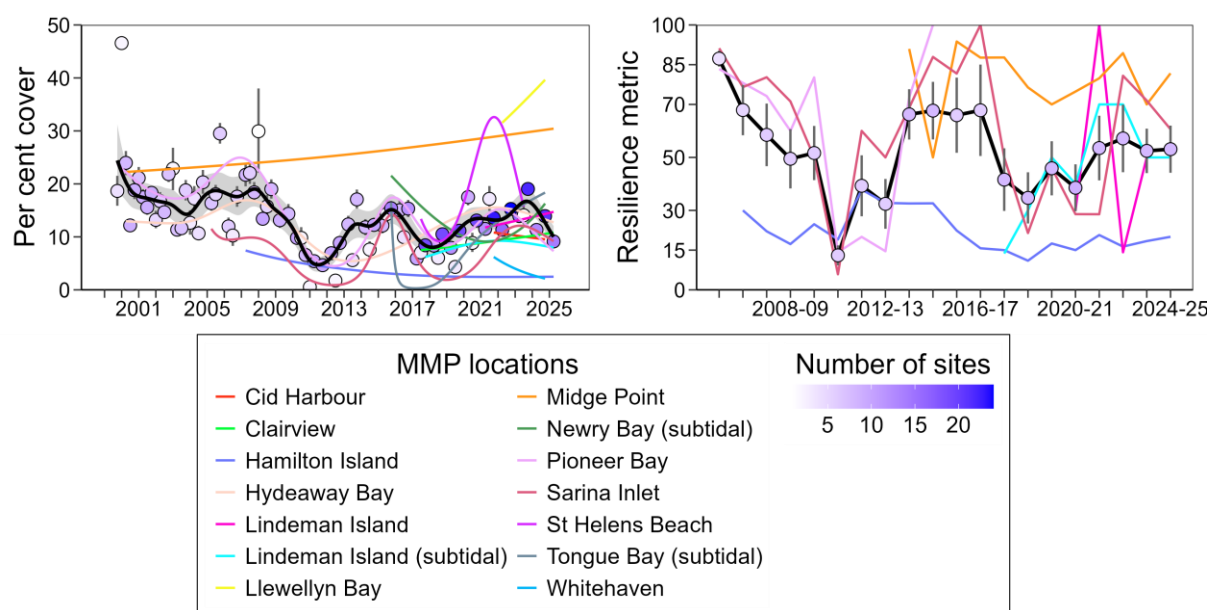


Figure 62. Temporal trends in the Mackay–Whitsunday seagrass indicators used to calculate the Seagrass Index: a. average (circles,  $\pm$ SE) seasonal abundance (per cent cover) and GAM plots of seagrass abundance trends for each location (coloured lines) and the region (black line with grey shaded area defining 95 per cent confidence intervals); b. average annual resilience metric ( $\pm$ 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, the average seagrass abundance in the Mackay–Whitsunday region declined in 2024–25, following 3 years of improvement. While coastal intertidal habitats improved overall, all other habitats experienced varying degrees of decline (Figure 63). Inshore reef

intertidal habitats declined the most overall, falling below the long-term average, with abundances in 2024–25 reaching less than two-thirds of those observed in the previous period. Coastal subtidal and estuarine intertidal habitats followed closely, with abundances decreasing by nearly one-third. Meanwhile, reef subtidal habitats experienced the least decline, losing approximately a quarter of their abundance, yet still remaining significantly lower than levels observed prior to 2005 (Figure 63).

Seagrass abundance (per cent cover) in the Mackay–Whitsunday region in 2024–25 was higher in coastal habitats (intertidal =  $19.9 \pm 1.1$  per cent, subtidal =  $12.2 \pm 2.8$  per cent) than estuarine ( $8.5 \pm 1.4$  per cent) or reef habitats (intertidal =  $7.8 \pm 1.0$  per cent, subtidal =  $6.5 \pm 1.0$  per cent), respectively.

Seagrass abundance in estuarine and coastal habitats has fluctuated greatly over the years, with some sites experiencing total or near total loss, followed by recovery (Figure 63). The long-term regional trend indicates a downward trajectory (Table 22), although coastal habitats are slowly recovering from repeated losses over the past decade. Nevertheless, localised declining trends continue to persist in intertidal and subtidal reef habitats.

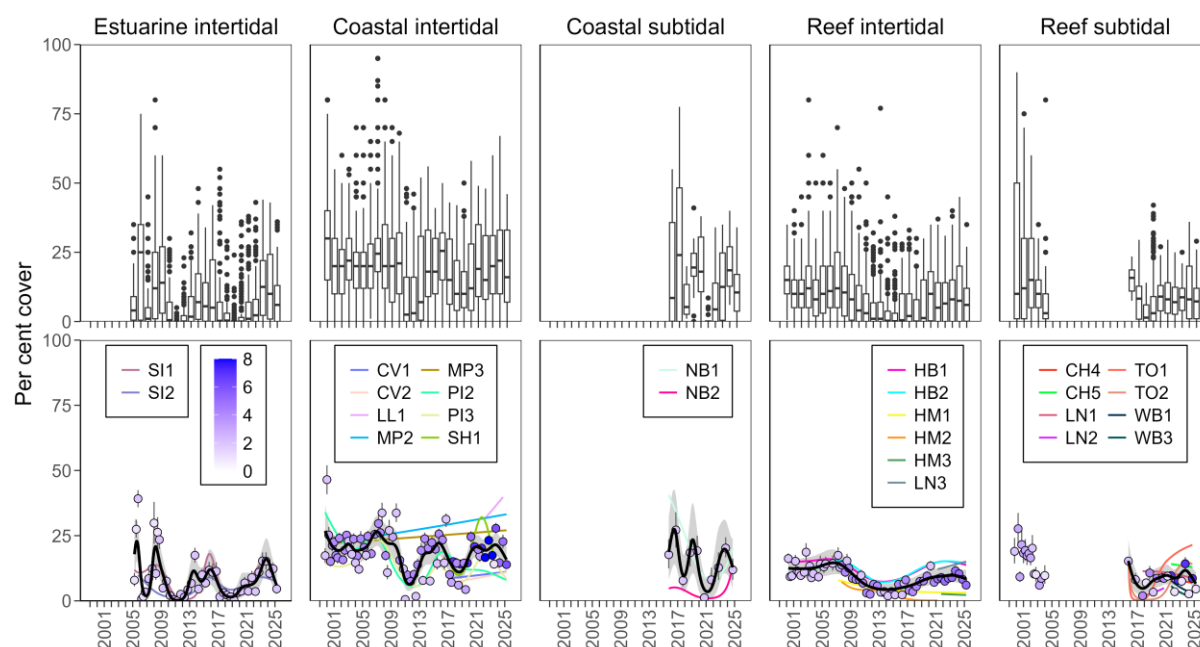


Figure 63. Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the Mackay–Whitsunday NRM region from 1999 to 2025. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates the 25<sup>th</sup> percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75<sup>th</sup> percentile. Whiskers (error bars) above and below the box indicate the 90<sup>th</sup> and 10<sup>th</sup> percentiles, and the dots represent outlying points. GAM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.

The predominant seagrass species found throughout various habitats in the Mackay–Whitsunday NRM region include *Halodule uninervis* and *Zostera muelleri*, often accompanied by the colonising species *Halophila ovalis*. In the years immediately following extreme weather events (such as those in 2011 and 2017), colonising species typically dominate in intertidal meadows across the Mackay–Whitsunday region; however, variations can occur between different habitats. Estuaries can experience significant fluctuations both between and within years, and in the last four years, the proportion of colonisers has exceeded the Reef long-term average (Figure 64).

Coastal subtidal habitats have been monitored for only 10 years, and they have exceeded the reef average for colonising species for the last 6 years. In reef intertidal habitats, the proportion of colonising species has risen in recent years, although it still remains below the

long-term average. Conversely, colonising species have remained low in coastal and reef intertidal habitats in recent years, but increased slightly in reef subtidal habitats (Figure 64).

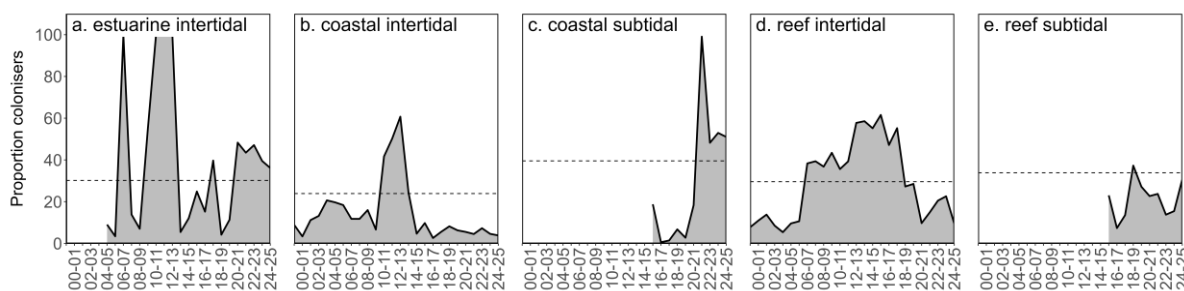


Figure 64. Proportion of seagrass abundance composed of colonising species at inshore intertidal habitats in the Mackay–Whitsunday region, 1999 to 2025. Dashed line represents Reef long-term average proportion of colonising species for each habitat type.

Seagrass meadow landscape mapping was conducted within all sentinel monitoring sites in October 2024 and May 2025 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 last year, the spatial extent has either increased or stayed constant throughout the region compared to the prior period, although there were seasonal decreases during the late wet season, especially in estuarine intertidal and reef subtidal habitats (Figure 65).

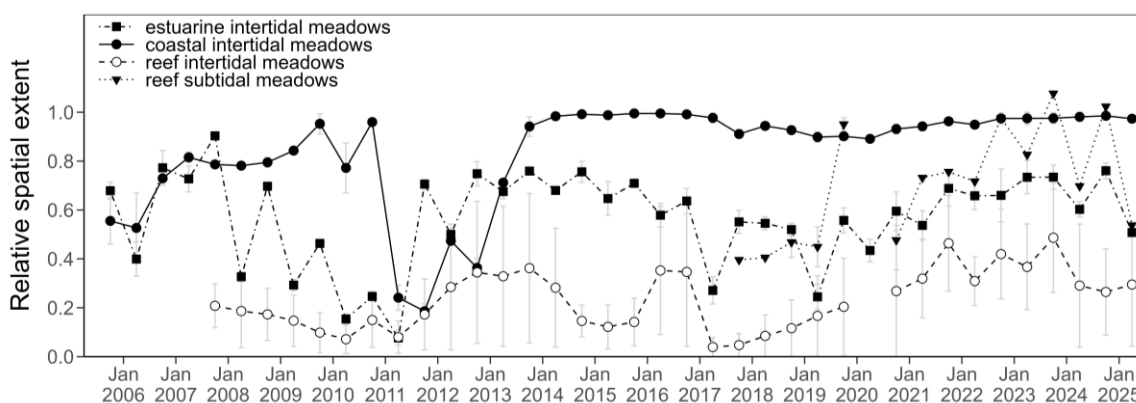


Figure 65. Change in spatial extent ( $\pm$  SE) of seagrass meadows within monitoring sites for each inshore intertidal habitat and monitoring period across the Mackay–Whitsunday NRM region, 2005-2025.

### 5.4.3.3 Seagrass reproductive status

Reproductive effort in the Mackay–Whitsunday NRM region was reduced in 2024–25 relative to the long-term average and compared to 2023–24 (Figure 66). There was no sexual reproduction in the late wet season as is typical for habitats predominantly composed of foundational species with seasonally distinct reproductive seasons (e.g., *Zostera muelleri*). Reproductive effort was reduced in the late dry season except at the coastal intertidal sites at Midge Point where reproductive effort remained around average.

The reproductive effort was composed almost entirely of foundational species at both estuarine and coastal sites. Reproductive effort was absent at reef intertidal and subtidal sites after large rises in the previous years.

There were no seeds of foundational species at reef intertidal and subtidal habitats, and the small seeds of colonising species are not quantified (Figure 66). There were seeds at intertidal reef and estuarine sites in 2024–25 but these were lower than historical peaks.

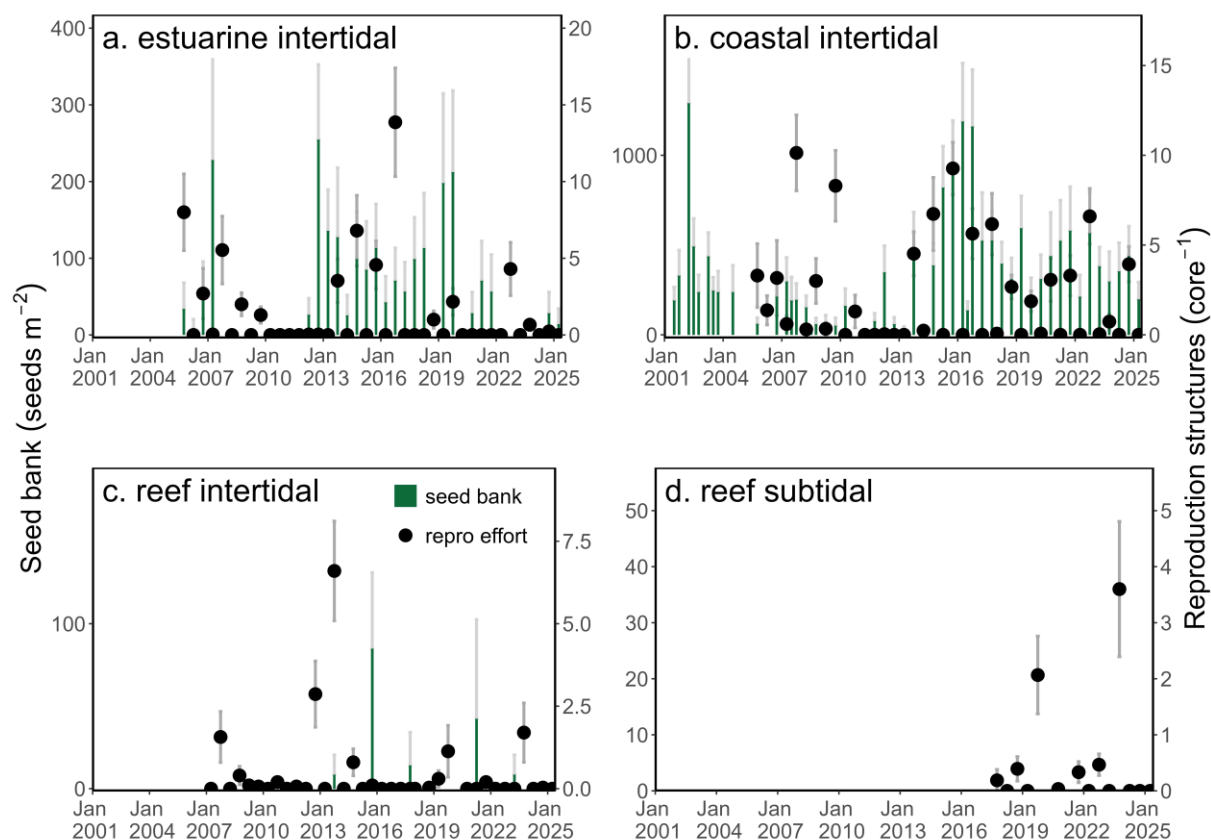


Figure 66. Seed bank and reproductive effort at inshore estuarine intertidal, coastal intertidal and reef intertidal and subtidal habitats in the Mackay–Whitsunday region, 2001–2025. Seed bank presented as the total number of seeds per m<sup>2</sup> sediment surface (green bars  $\pm$ SE), and late dry season reproductive effort presented as the average number of reproductive structures per core (species and sites pooled) (dots  $\pm$ 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 remained ‘moderate’ and stable for the fourth consecutive year in 2024–25 (Figure 59). Scores declined at estuarine sites, increased at coastal sites, and remained stable at reef intertidal and subtidal habitats (Figure 66). At Sarina Inlet, both estuarine sites met thresholds for abundance and composition, and reproductive structures of the foundational species *Zostera muelleri* were present at SI1 but absent at SI2, resulting in a category ‘2.1.2’ classification. At coastal intertidal sites, resilience increased to over 80 due to higher counts of reproductive structures compared with 2023–24. Resilience remained low at reef intertidal sites; HM3 fell below the low-abundance threshold and lacked reproductive structures (category ‘1.1’), while HM1 and LN3 exceeded abundance thresholds but remained in category ‘2.1’ due to the absence of reproduction. At the reef subtidal site, resilience was stable, with abundance and composition above thresholds; however, no reproductive structures were present in 2024–25, and the score was at the lower margin of category ‘2.1.2’. Several additional sites in the region are monitored through the Reef Joint Field Management Program and Seagrass–Watch, where resilience cannot currently be assessed.

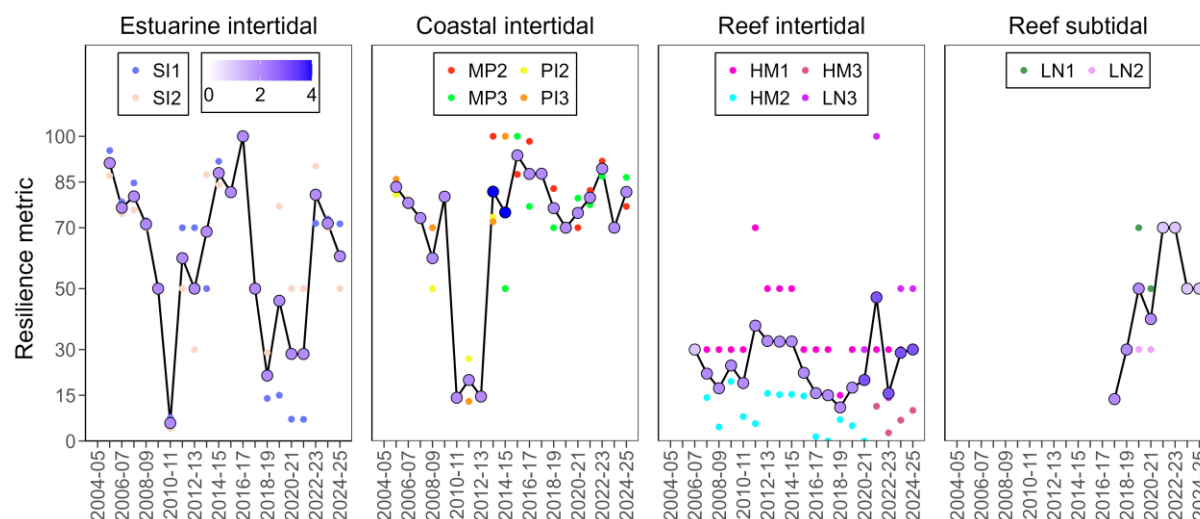


Figure 67. Resilience metric for each habitat type in the Mackay–Whitsunday region, 2006 to 2024. Coloured small points represent different sites. Shades of blue for the larger points indicate the number of sites that contribute to the score.

#### 5.4.3.5 Epiphytes and macroalgae

Epiphyte cover on seagrass leaf blades during the 2024–25 period remained at or below the long-term average throughout both the dry and wet seasons across all habitats (Figure 68). During this period, the percentage cover of macroalgae stayed consistent, remaining at or below the overall long-term average for estuarine and inshore reef intertidal habitats (Figure 68). In coastal intertidal habitats, however, macroalgae coverage experienced a slight increase above the long-term average in the late dry season. Similarly, in subtidal habitats within both coastal and inshore reef areas, macroalgae coverage rose above the long-term average during the late dry season; yet, the levels observed in 2024–25 were lower than those recorded the previous year.

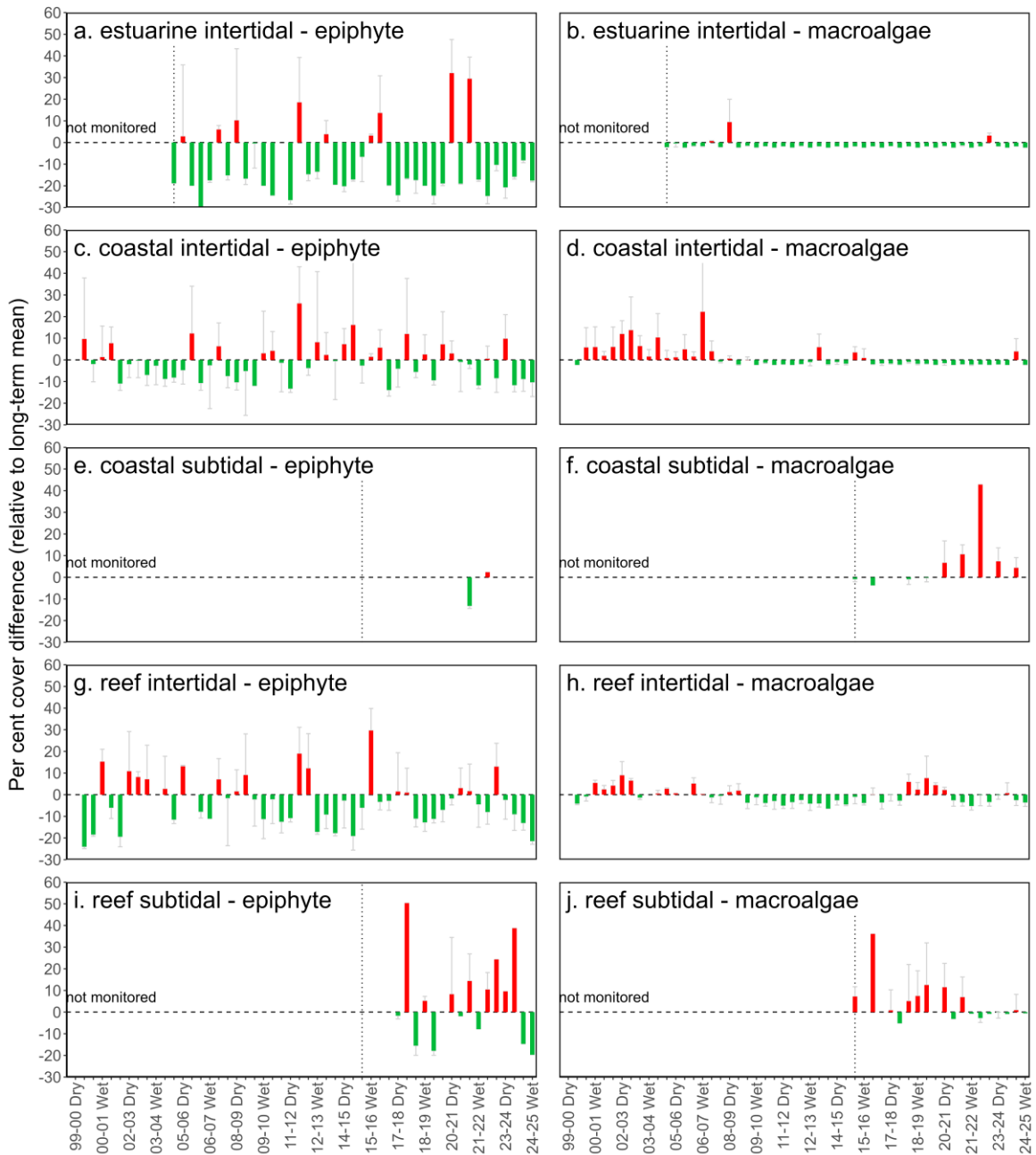


Figure 68. Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average for each inshore intertidal seagrass habitat in the Mackay–Whitsunday region, 1999–2025 (sites pooled,  $\pm$ SE). Vertical dotted lines represent the first monitoring event for each habitat type. Red and green bars indicate above and below long-term average, respectively.

## 5.5 Fitzroy

### 5.5.1 2024–25 Summary

Environmental pressures in the Fitzroy region during 2024–25 were generally near long-term averages for discharge and water quality, although seawater temperatures were above average. Coastal and estuarine seagrass habitats experienced high exposure to turbid sediment laden and algae enriched waters (WT1 and WT2) for most of the wet season, while reef sites around Great Keppel Island recorded relatively more WT2 and occasional less turbid clearer water (WT3) conditions. Annual average daily light availability could not be assessed due to widespread logger malfunctions.

The Fitzroy NRM region is only surveyed in the late dry season prior to the wet season, meaning the Seagrass Index primarily reflects legacy effects from the previous year's environmental conditions. In 2024–25, river discharge and water quality levels were relatively benign; however, within-canopy seawater temperatures were above average.

Overall, the seagrass condition score for the Fitzroy NRM region improved to '**moderate**' in 2024–25 (Figure 69). There were improvements in both indicators. Condition indicators contributing to this were:

- abundance score was 'moderate'
- resilience was 'moderate'.

The seagrass abundance score significantly improved from the previous period, achieving its highest score since 2007–08. The increase can be attributed to improved conditions in estuarine and coastal habitats, while the status of reef habitats has remained unchanged. The resilience score in the Fitzroy region improved to 'moderate' in 2024–25, the highest level since 2009–10, with increases observed across all habitat types. Estuarine and coastal sites showed strong gains due to increased abundance of sexual reproductive structures, while modest improvements at Great Keppel Island were driven by slight increases in cover despite continued dominance of colonising species. Seed banks persisted across most habitats, with high densities at estuarine sites but reduced levels at coastal sites compared to previous years. In contrast, reef meadows at Great Keppel Island again showed no reproductive structures or seeds of foundational species (*Halodule uninervis*), indicating severely limited recovery capacity and heightened vulnerability to disturbance.

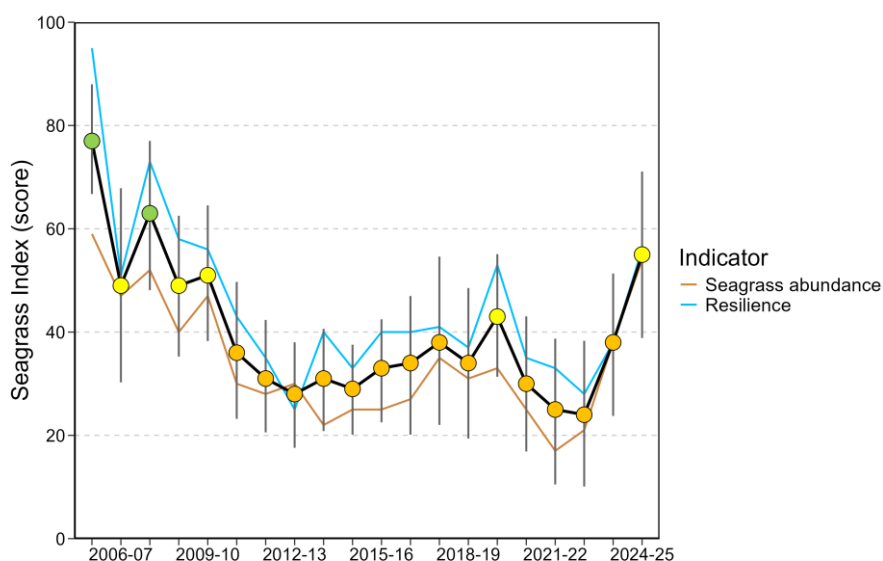


Figure 69. Temporal trend in the Seagrass Index ( $\pm$  SE) with contributing indicator scores for the Fitzroy NRM region (averages across habitats and sites). Values are indexed scores scaled from 0–100 ( $\pm$  SE) and graded: ● = very good (81–100), ● = good (61–80), ● = moderate (41–60), ● = poor (21–40), ● = very poor (0–20). NB: Scores are unitless.

### 5.5.2 Climate and environmental pressures

Environmental pressures in the Fitzroy NRM region were around the long-term average for discharge and water quality, but seawater temperature was above average.

Wet season rainfall and discharge from the Fitzroy basins in 2024–25 was around the long-term average for the region (Figure 70c). Inshore coastal and estuarine seagrass habitats were exposed to turbid sediment laden waters (WT1) or turbid algae enriched waters (WT2) for 90% of weeks during the wet season, which was below the long-term mean (Figure 70c). There was relatively more WT2 over the reef sites at Great Keppel Island and a few weeks when there was less turbid clearer water (WT3) (Figure 70a, b). Water type exposure can be influenced by inshore processes such as resuspension as well as river discharge.

Annual averaged daily light availability in 2024–25 was unable to be calculated due to logger malfunctions resulting in major portions of data loss through the period (Figure 110). As sites are only sampled once a year, the 12-month logger deployments increase the risk of data loss.

During the 2024–25 period, within-canopy temperatures were above the long-term average for the region (Figure 70c,f), although they were less extreme compared to the previous period. Maximum intertidal within-canopy temperatures only surpassed 35°C on 21 days, a quarter of the total days recorded in the prior period, which had the highest number since 2003–04. The peak temperature reached 40.02°C (Great Keppel Island, 17 October 24), which was 10°C lower than the previous maximum in the region in 2023–24 (Figure 70e). Additionally, daytime tidal exposure in 2024–25 was lower than both the previous period and the long-term average (Figure 70c, Figure 97), potentially providing some relief from stressors such as carbon limitation and desiccation to seagrasses throughout the region.

In the 2024–25 period, sediments in the estuarine habitat sites continue to be primarily composed of finer grains. Although the mud content has slightly decreased, it still remains above the long-term average at both sites (Figure 128). Furthermore, the sediments in the reef habitat have become coarser over the year due to the loss of finer grains. In contrast, the mud content in coastal habitat sediments has continued to rise at both sites within Shoalwater Bay (Figure 129, Figure 130).

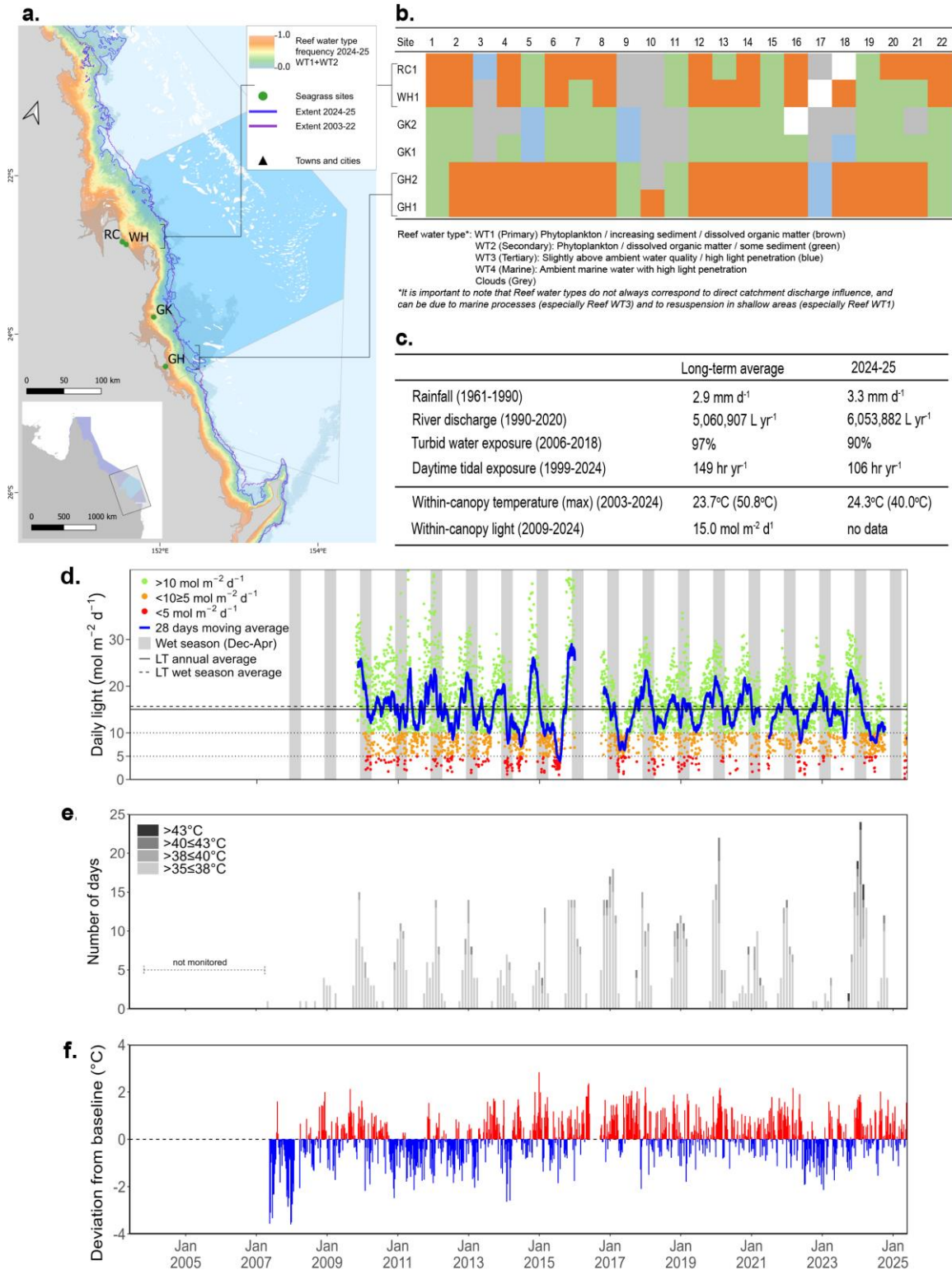


Figure 70. Environmental pressures in the Fitzroy region including: a. frequency of exposure to primary (WT1) and secondary (WT2) water from December 2024 to April 2025 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed) (white = no data), also showing the long-term average (2003–2018) exposure boundary (purple line), and the extent for 2024–25 (blue line) (from Gruber *et al.* 2026); b. wet season water type at each site; c. average conditions and max temperature over the long-term and in 2024–25; 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 NRM region in 2024–25, with data from all 6 long-term monitoring sites (Table 17).

Table 17. 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 5.

Habitat	Site		abundance	composition	distribution	reproductive effort	seed banks	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 2024–25 monitoring period, the Seagrass Index continued to improve, with a grade increase to **'moderate'** (Figure 69). The Index was the highest in the last 17 years and the third highest since monitoring began in the Fitzroy NRM region.

In 2024–25, the abundance score increased significantly from the previous period, reaching its highest level since 2005-06; resulting in a grade change from 'poor' to 'moderate' (Figure 71). The increase stemmed from the improved abundances in Gladstone Harbour and Shoalwater Bay (estuarine and coastal habitats, respectively), while Great Keppel Island reef habitats continued their declining trajectory (Figure 71).

The resilience score similarly increased, reaching its highest level in fourteen years (Figure 71). This improvement was primarily due to enhanced resilience in the estuarine meadows of Gladstone Harbour, and to a lesser extent, the coastal meadows in Shoalwater Bay. Additionally, the resilience of reef habitats at Great Keppel Island improved following a record lows in the previous year (Figure 71)

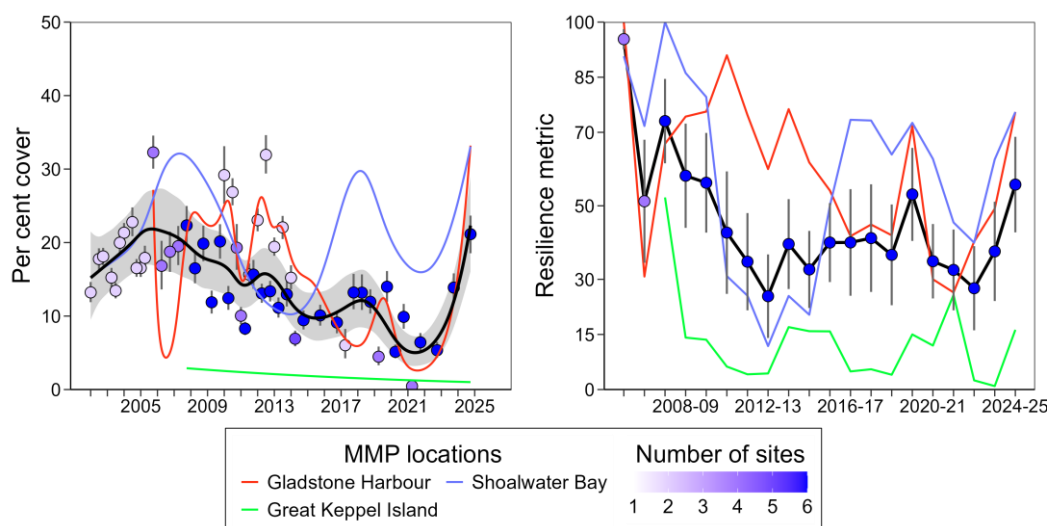


Figure 71. Temporal trends in the Fitzroy seagrass indicators used to calculate the Seagrass Index: a. average (circles,  $\pm$ SE) seasonal abundance (per cent cover) and GAM plots of seagrass abundance trends for each location (coloured lines) and the region (black line with grey shaded area defining 95 per cent confidence intervals); b. average annual resilience metric ( $\pm$ 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

Seagrass abundance (per cent cover) in the Fitzroy region reached its highest level in 2 decades in 2024–25. The strongest gains occurred in estuarine habitats (Figure 72), especially at site GH2, where abundance nearly quadrupled to its highest level in 15 years; recovering rapidly from the near-total loss in 2022–23 (the lowest since monitoring was established). Similarly, site GH1 also rose to its highest level in more than a decade, supported by steady improvement over the past four years.

In 2024–25, seagrass abundance was significantly higher in coastal and estuarine habitats, measuring  $33.2 \pm 1.3$  per cent and  $32.2 \pm 1.8$  per cent, respectively, compared to reef habitats, which only reached  $0.5 \pm 0.3$  per cent (Figure 72). Over the years of the monitoring program, seagrass abundance in estuarine and coastal intertidal habitats has varied significantly, with some sites experiencing complete or nearly complete loss followed by recovery (Figure 72). Seagrass abundance in the intertidal reef habitat at Great Keppel Island has consistently been low since the onset of monitoring in 2007. While it experienced a rise in 2023–24, reaching its highest level in 5 years, it saw a significant drop, resulting in the fourth lowest level recorded in 2024–25.

Despite these recent increases, examination of the long-term (2002 to 2024) trend in seagrass abundance throughout the region reveals a significant decrease (Figure 71, Table 22). This decrease is mainly influenced by individual sites in estuarine and reef habitats (GH1 and GK1, respectively), although two-thirds of all monitoring sites in the region, including coastal habitats, show no significant trend (Table 22).

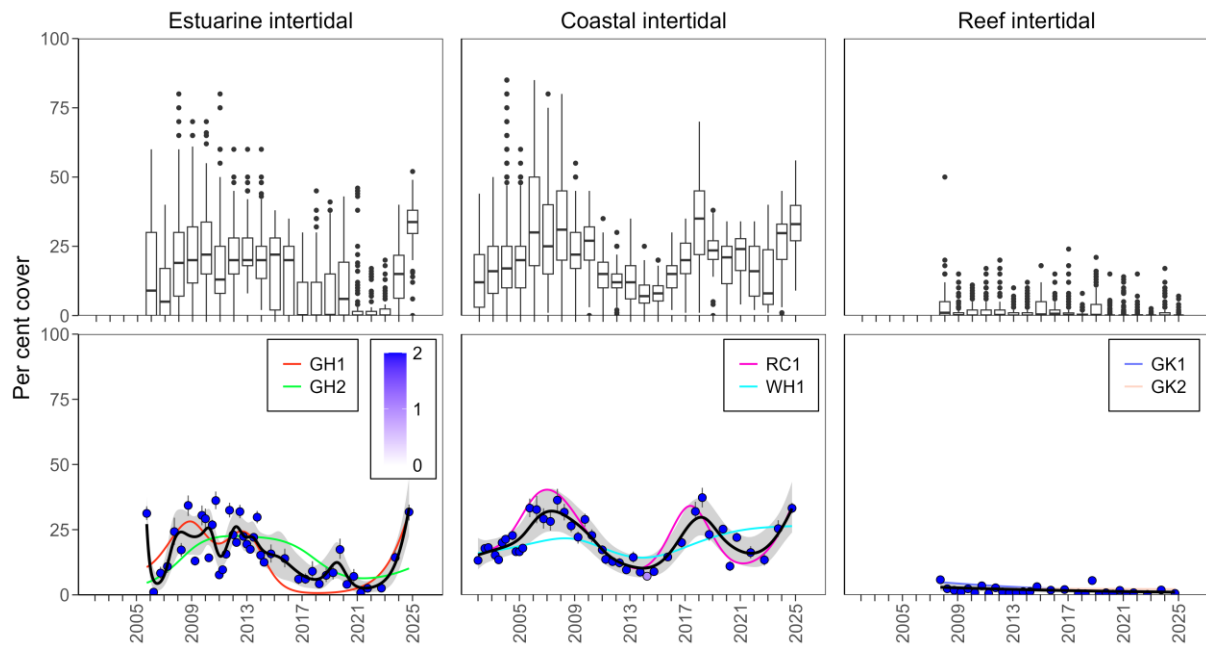


Figure 72. Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the Fitzroy NRM region from 2002 to 2025. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates the 25<sup>th</sup> percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75<sup>th</sup> percentile. Whiskers (error bars) above and below the box indicate the 90<sup>th</sup> and 10<sup>th</sup> percentiles, and the dots represent outlying points. GAM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.

With an increase in seagrass abundance, the estuarine meadows at Pelican Banks (Gladstone Harbour) have shown a rise in the dominance of the foundational seagrass *Zostera muelleri*, while the presence of colonising species (*Halophila ovalis*) has continued to diminish. In the coastal meadows of Shoalwater Bay (Ross Creek and Wheelans Hut), *Z. muelleri* continues to dominate, also with low proportions of colonising *H. ovalis*. Conversely, colonising species have continued to dominate in the reef habitat sites, exceeding the overall inshore Reef long-term average (Figure 73).

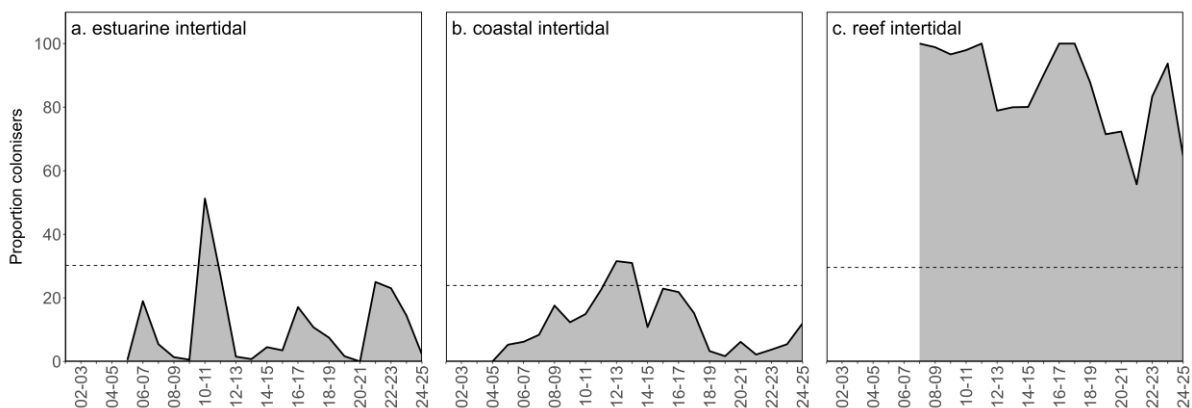


Figure 73. Proportion of seagrass abundance composed of colonising species in inshore intertidal habitats of the Fitzroy region, 2001–2025. Dashed line represents Reef long-term average proportion of colonising species for each habitat type.

Despite a minor decrease in the extent of coastal intertidal meadows at the end of 2022, monitoring sites in Shoalwater Bay have demonstrated overall improvement over the past 2 years (Figure 74). The estuarine meadows at Pelican Banks in Gladstone Harbour have experienced significant fluctuations in extent since 2015–16, when one site underwent a

major reduction due to extensive scarring and sediment deposition. By 2019–20, the sediment deposition abated, enabling the meadow to show signs of recovering through shoot extension and enhanced meadow cohesion. However, between 2020–21 and 2021–22, increased erosion along tidal drainage channels and further scarring throughout the meadow contributed to a reduction in overall extent (Figure 74). In 2022–23, the deterioration of the meadow seascape subsided, leading to the highest overall meadow extent in a decade by late 2023, where it has remained since. Conversely, the meadows on the sand reef flat at Great Keppel Island have remained highly fragmented since the losses in 2015–16 and have experienced considerable fluctuations over the last 6 years, with a slight increase in 2024–25 after experiencing a decline in the preceding period (Figure 74).

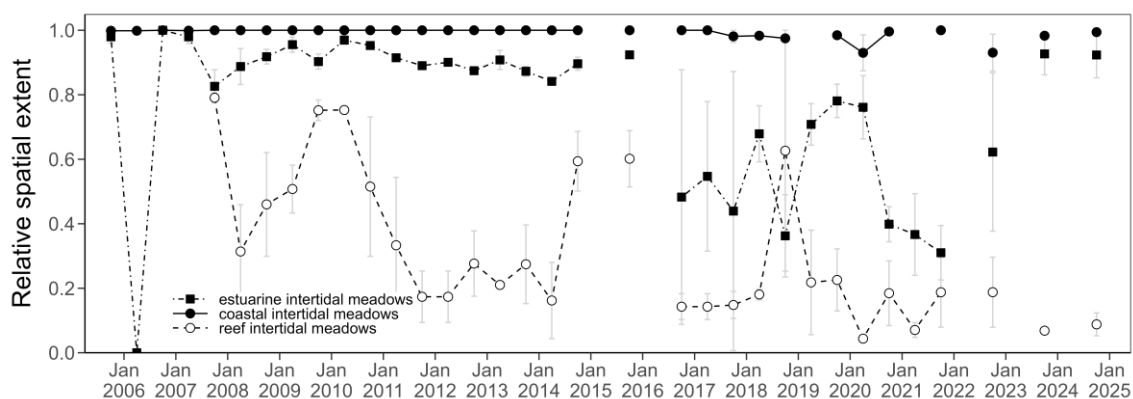


Figure 74. Change in spatial extent ( $\pm$  SE) of seagrass meadows within monitoring sites for each inshore intertidal habitat across the Fitzroy NRM region, 2005–2025.

### 5.5.3.3 Seagrass reproductive status

The abundance of reproductive structures in the Fitzroy region has shown strong seasonal and interannual variability across habitats over the monitoring period, particularly in estuarine and coastal areas (Figure 75). Since 2021, seagrass assessments have been undertaken only during the late dry season. In 2024–25, overall reproductive effort increased, with estuarine sites in Gladstone Harbour recording the second-highest levels since 2010 and coastal sites in Shoalwater Bay reaching their second-highest on record (Figure 75). This will likely hasten recovery of the seed banks.

Over the past decade, seed banks have persisted in both estuarine and coastal intertidal habitats (Figure 75). At estuarine sites, seed density was the second-highest on record due to a large increase at GH2, while seeds remained present but lower than average at coastal sites in Shoalwater Bay. For the third consecutive year, reproductive structures of the foundational species *Halodule uninervis* were absent at reef sites, and seeds have never been recorded in reef meadows at Great Keppel Island. This continued absence reduces the capacity for natural recovery and increases vulnerability to future disturbances. The lack of seeds likely reflects chronically low reproductive output but may also result from failed seed set or rapid post-release loss through germination or grazing (Heck and Orth 2006). The persistent absence of seeds further limits recovery potential for foundational species, which are already in low abundance.

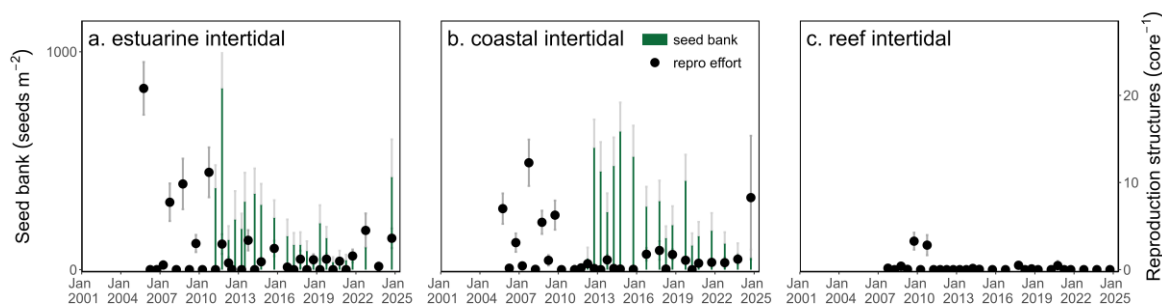


Figure 75. Seedbank and reproductive effort at inshore intertidal coastal, estuarine and reef habitats in the Fitzroy region, 2005–2025. Seed bank presented as the total number of seeds per m<sup>2</sup> of sediment surface (green bars ±SE). Reproductive effort for the late dry season presented as the average number of reproductive structures per core (species and sites pooled) (circles ±SE).

### 5.5.3.4 Resilience

Overall resilience in the Fitzroy region improved to moderate in 2024–25, reaching the highest level since 2009–10 (Figure 71). Increases were recorded across all habitat types (Figure 76). At estuarine intertidal habitats in Gladstone and coastal intertidal sites in Shoalwater Bay, reproductive structures were present and all sites were classified within category ‘2.2.1’, with the greatest increases observed at GH2 and RC1. At reef intertidal sites at Great Keppel Island, resilience improved following a record low in the previous year. GK1 remained dominated by colonising species and below the low-cover threshold (category ‘1.1’), while GK2, despite very low abundance, exceeded the low-cover threshold and was classified as category ‘2.1.1’, driving the overall increase in regional resilience.

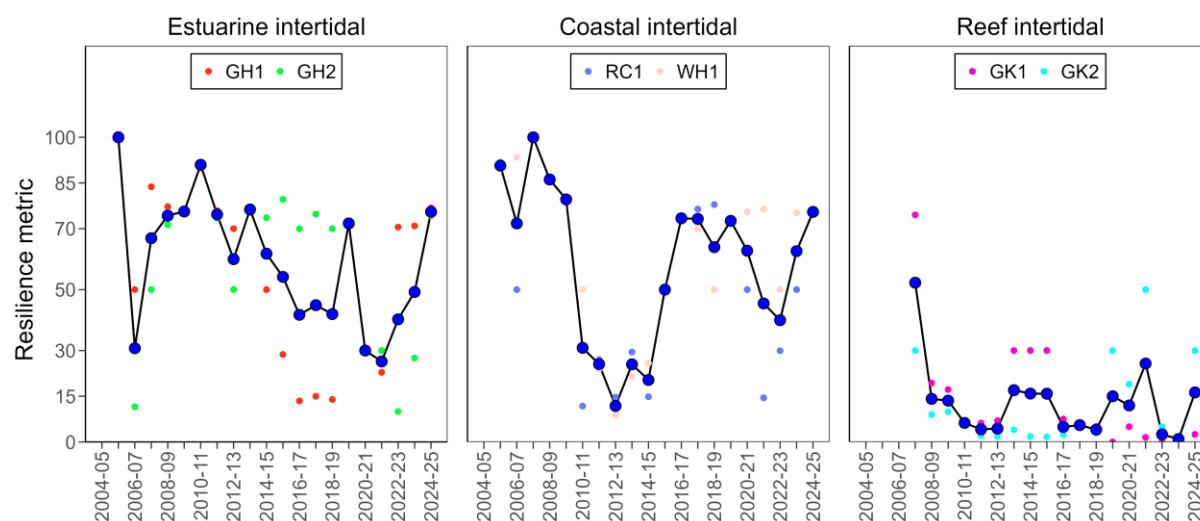


Figure 76. Resilience in each habitat in the Fitzroy region 2006 to 2025. Coloured small circles represent different sites. Shades of blue for the larger circles indicates the number of sites that contributed to the metric.

### 5.5.3.5 Epiphytes and Macroalgae

In the 2024–25 period, epiphyte cover on seagrass leaves in estuarine habitats dropped below the long-term average. Meanwhile, it has continued to remain below the inshore Reef long-term average for intertidal coastal and reef habitats for the fifth and fourth consecutive years, respectively (Figure 77).

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

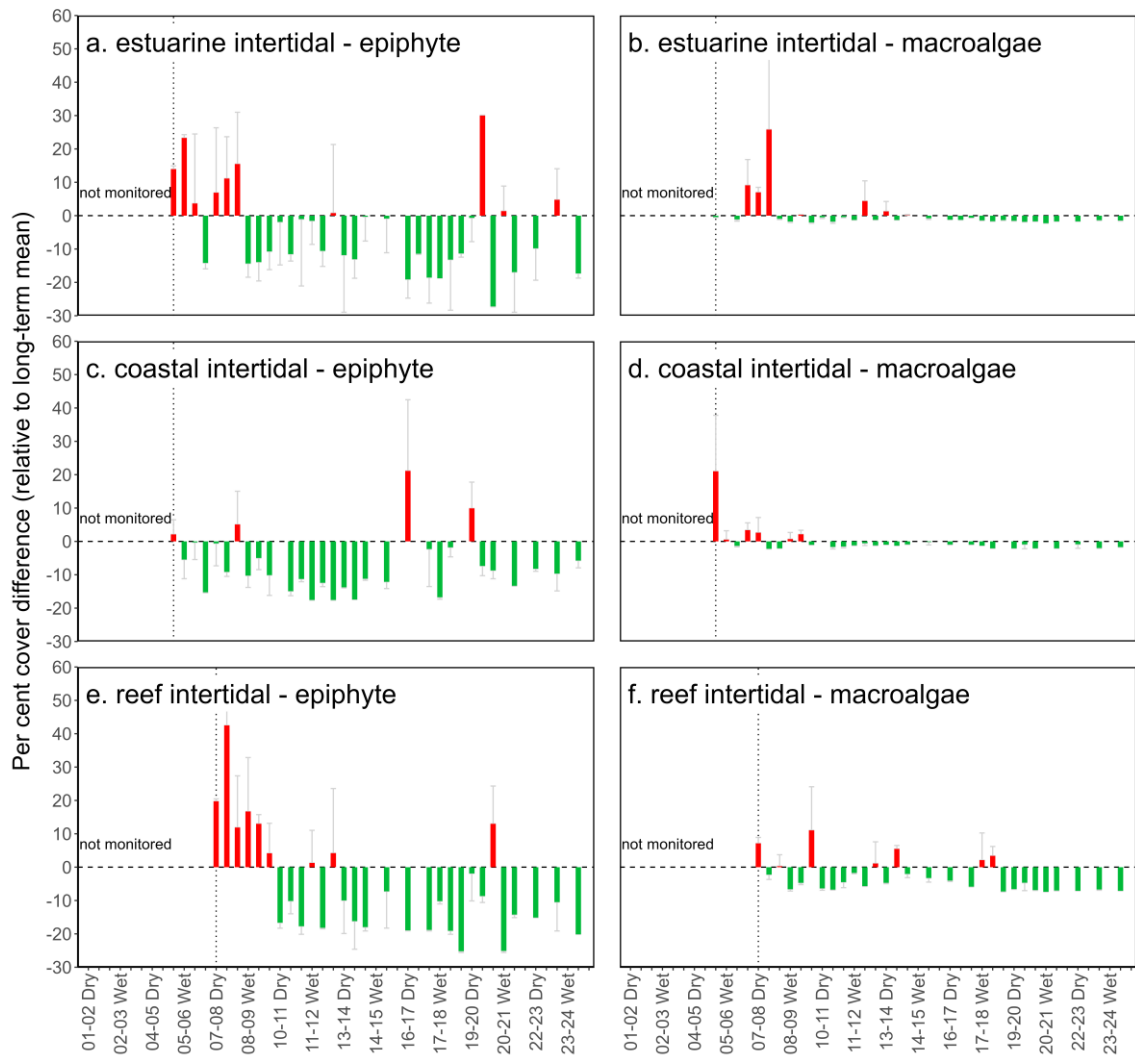


Figure 77. Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average for each inshore intertidal seagrass habitat in the Fitzroy region, 2005–2025 (sites pooled,  $\pm$ SE). Vertical dotted lines represent the first monitoring event for each habitat type. Red and green bars indicate above and below long-term average, respectively.

## 5.6 Burnett–Mary

### 5.6.1 2024–25 Summary

Rainfall and river discharge in 2024–25 were above the long-term average in the Burnett–Mary region, with the 3 southern basins recording more than four times their typical discharge and elevated flows occurring in 3 of the past four years. Coastal and estuarine monitoring sites were exposed to turbid sediment laden waters (WT1) and turbid algae enriched waters (WT2) for most of the wet season, and within-canopy daily light levels in 2024–25 well below long-term averages across all sites. Light remained lower than average through the dry season and declined sharply during the wet season, reflecting the legacy of recent high-discharge years. Within-canopy temperatures in 2024–25 were 0.6 °C above the long-term average, marking the second consecutive warm year, with intertidal temperatures exceeding 35 °C on four days and reaching a maximum of 38.7 °C at Rodds Bay.

Inshore seagrass meadows across the Burnett–Mary NRM region improved in overall condition in 2024–25, with the Seagrass Index increasing to a **‘good’** grade (Figure 78). Contributing indicators to the overall score were:

- abundance score was ‘moderate’
- resilience score was ‘good’.

The seagrass abundance score for the NRM region increased to its highest level since the MMP commenced, remaining ‘moderate’ in 2024–25. This improvement was observed throughout the region, covering all locations, with abundances surpassing long-term averages for both estuarine and coastal habitats.

Resilience in the Burnett–Mary region improved to ‘good’ in 2024–25, up from ‘poor’ in 2023–24, driven largely by strong recovery in estuarine habitats. High reproductive output of foundational species at Rodds Bay and Urangan sites contributed most to the improvement. In contrast, coastal sites at Burrum Heads remained constrained by a continued lack of reproductive effort, leading to a slight local decline. Despite that, seeds banks remain at estuarine and coastal habitats, but not at all sites.

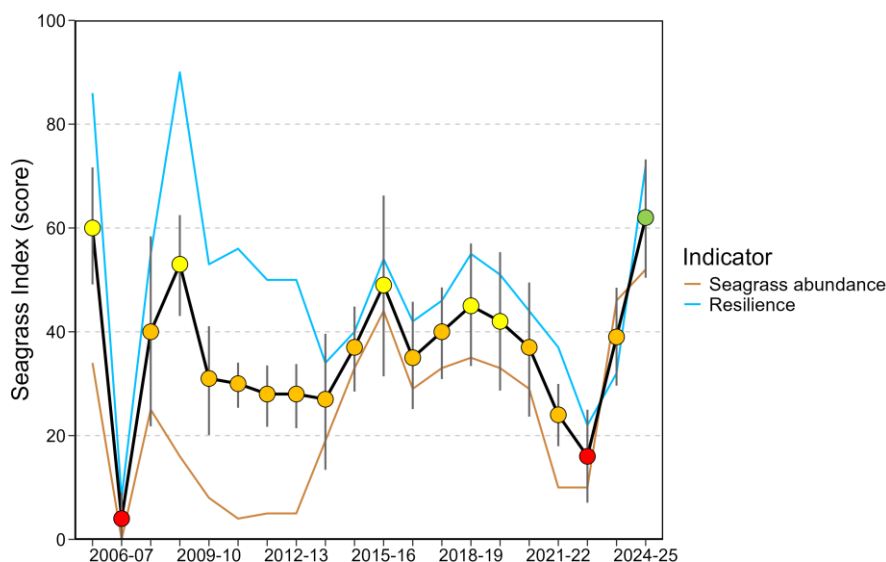


Figure 78. Temporal trend in the Seagrass Index ( $\pm$  SE) with contributing indicator scores for the Burnett–Mary region (averages across habitats and sites). Values are indexed scores scaled from 0–100 ( $\pm$  SE) and graded: ● = very good (81–100), ● = good (61–80), ● = moderate (41–60), ● = poor (21–40), ● = very poor (0–20). NB: Scores are unitless.

### 5.6.2 Climate and environmental pressures

Rainfall in 2024–25 was higher than the long-term average for the region (Figure 79), particularly across the southern basins of the Burnett–Mary NRM region (Figure 6). River discharge was also above the long-term median, with elevated flows recorded in all basins except Baffle Creek in the northern part of the region (Table 11). Discharge in the 3 southern basins exceeded four times the long-term median, and flows have been elevated in 3 of the past 4 years, contributing to sustained water-quality pressures in coastal habitats.

In the Burnett–Mary region, monitoring is limited to intertidal estuarine and coastal sites, which are typically exposed to high frequencies of turbid sediment laden and algae enriched waters (WT1 and WT2). In 2024–25, these sites experienced WT1 or WT2 conditions for 97% of wet-season weeks, consistent with the long-term average of 100% (Figure 79a, b). Within-canopy daily light levels averaged  $7.6 \text{ mol m}^{-2} \text{ d}^{-1}$  in 2024–25, well below the long-term mean of  $11.7 \text{ mol m}^{-2} \text{ d}^{-1}$ . This reduction was driven by particularly low light at Urangan ( $6.3 \text{ mol m}^{-2} \text{ d}^{-1}$ ) and Burrum Heads ( $8.7 \text{ mol m}^{-2} \text{ d}^{-1}$ ), relative to their long-term averages of  $11.2$  and  $13.7 \text{ mol m}^{-2} \text{ d}^{-1}$ , respectively. Daily light remained lower than average throughout the dry season and declined sharply during the wet season, staying consistently below wet-season long-term averages (Figure 111). These patterns are consistent with the legacy effects of consecutive years of above-average discharge.

Within-canopy temperature in 2024–25 was  $0.6^\circ\text{C}$  warmer than the long-term average. This is the second warm year following 2022–23 when temperatures were marginally below the long-term average for the first time in nearly a decade (Figure 79c,f). Maximum intertidal within-canopy temperatures exceeded  $35^\circ\text{C}$  for a total of 4 days during 2024–25 (Figure 79e), with the highest temperature recorded at  $38.7^\circ\text{C}$  (Rodds Bay, 17 October 2024).

Daytime tidal exposure in 2024–25 was well below the regional long-term average for the eighth consecutive year (Figure 79c, Figure 103). At the Burrum Heads coastal habitat, the number of daylight hours exposed was the lowest recorded since monitoring began at the location. Additionally, across estuarine habitats, the exposure duration was among the lowest ever, remaining below the long-term averages for each site for a decade. While the increased hours of seagrass submersion during daylight (i.e., fewer hours exposed) may have mitigated the risk of desiccation stress, it might also have heightened the risk of light limitation in areas with turbid water.

Sediments within the estuarine seagrass habitats of the Burnett–Mary region are generally dominated by mud. Throughout 2022 and 2023, the sediments at the southernmost estuarine location (Urangan) were primarily composed of fine sands, featuring a surface layer of dispersive soils that were deposited across the intertidal banks due to the floods in early 2022. In the 2023–24 period, the proportion of mud in the estuarine habitats increased above the Reef long-term average, and the dispersive soils at Urangan dissipated and moved offshore. In 2024–25, the proportion of mud remained above long-term average, and the location more dominated by mud. Estuarine meadows in the north of the region varied, with a noticeable increase in sand content at one site (RD3), while the other site remained unchanged and dominated by mud (Figure 131). In 2024–25, coastal habitat meadows continued to be primarily composed of fine sand; however, during the late wet season, an accumulation of mud was noted at the site (BH1) immediately adjacent to the mouth of the Burrum River.

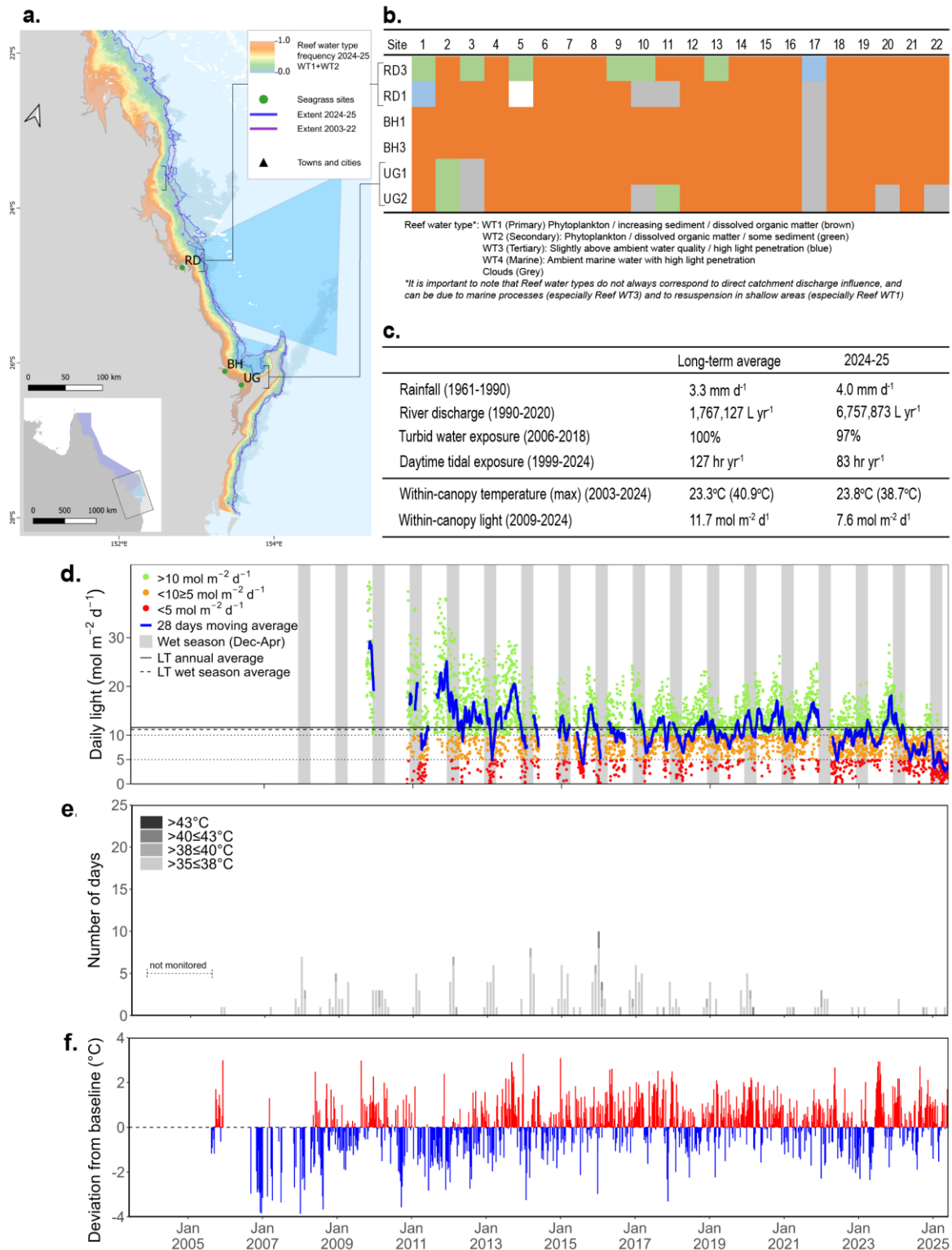


Figure 79. Environmental pressures in the Burnett–Mary region including: a. frequency of exposure to primary (WT1) and secondary (WT2) waters from December 2024 to April 2025 ranging from frequency of 1 (orange, always exposed) to 0 (pale blue, never exposed) (white = no data), also showing the long-term average (2003–2018) exposure boundary (purple line), and the extent for 2024–25 (blue line) (from Gruber *et al.* 2026); b. wet season water type at each site; c. average conditions and max temperature over the long-term and in 2024–25; 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 2024–25, with data from all 6 long-term monitoring sites (Table 18).

Table 18. 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 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

During the 2024–25 monitoring period, the Seagrass Index for the Burnett–Mary region elevated from a 'poor' to a '**good**' rating, achieving its highest score since the establishment of the MMP (Figure 78). This improvement reinforces the recovery and reversal of the declining trend noted between 2019 and 2023, with improvements in both indicators contributing to this positive result (Figure 80).

Over the long term, the seagrass abundance indicator in the region has shown significant fluctuations, characterised by phases of both loss and recovery. Between 2012 and 2016, the estuarine meadows at Urangan experienced a notable increase, which sharply declined in early 2017 and was completely lost by 2022, due to severe flooding in the region. Similar declines were observed at other estuarine and coastal locations starting in 2019. However, the improvement in 2024–25 resulted in the highest abundance score recorded since the MMP commenced in 2005–06. This recovery spanned the region, encompassing all locations and habitats (Figure 80).

Seagrass resilience in the Burnett–Mary region improved during 2024–25 to its highest score since 2008–09 (Figure 80). This improvement was primarily due to substantial increases recorded at all estuarine habitat sites in Rodds Bay and Urangan, in the north and south, respectively. These gains outweighed a slight decline in the coastal habitat resilience, particularly in site BH1 at Burrum Heads.

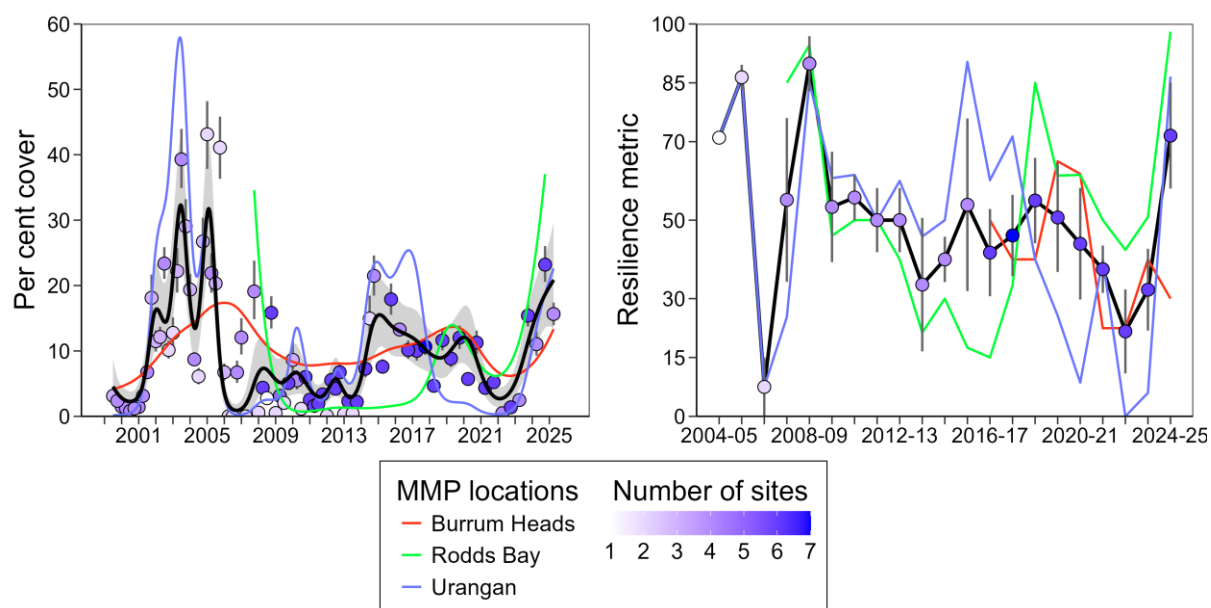


Figure 80. Temporal trends in the Burnett–Mary seagrass indicators used to calculate the Seagrass Index: a. average (circles,  $\pm$ SE) seasonal abundance (per cent cover) and GAM plots of seagrass abundance trends for each location (coloured lines) and the region (black line with grey shaded area defining 95 per cent confidence intervals); b. average annual resilience metric ( $\pm$ 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

Since the initiation of monitoring, the estuarine meadows across the Burnett–Mary region have come and gone on an irregular basis, showing no clear long-term trend as of 2024–25 (Table 22). In contrast, the coastal meadows at Burrum Heads have displayed slightly more consistency, although one site (BH3) has experienced a significant long-term increase (Table 22).

Historically, seagrass abundances (per cent cover) across the Burnett–Mary region has generally been higher in estuarine habitats compared to coastal ones, with long-term averages of  $10.6 \pm 1.5$  per cent and  $9.4 \pm 0.9$  per cent, respectively. During the 2024–25 period, seagrass abundance exceeded long-term averages for both habitats, reaching  $23.6 \pm 1.8$  per cent and  $11.8 \pm 1.1$  per cent in estuarine and coastal habitats, respectively (Figure 81). Overall, the Burnett–Mary region experienced its second consecutive year of significant improvement in seagrass abundance after a period of significant losses due to multiple years of above-average rainfall and river discharge, culminating in record floods in early 2022.

The largest improvement was in the estuarine meadows at Urangan, which had been completely lost during the late dry season of 2022, with no shoots present across the entire bank. The onset of seagrass recovery in the Urangan meadows was observed in the late wet season of 2023 (12 month post floods), with substantial gains occurring in 2023–24 and continuing throughout 2024–25 (Figure 81). During the last monitoring period, the observed increases were primarily attributed to the resurgence of the foundation species, *Zostera muelleri*, which began to dominate (Figure 82). The rise of this foundation species indicates that the meadows are likely to have a considerably enhanced ability to resist moderate disturbances in the near future.

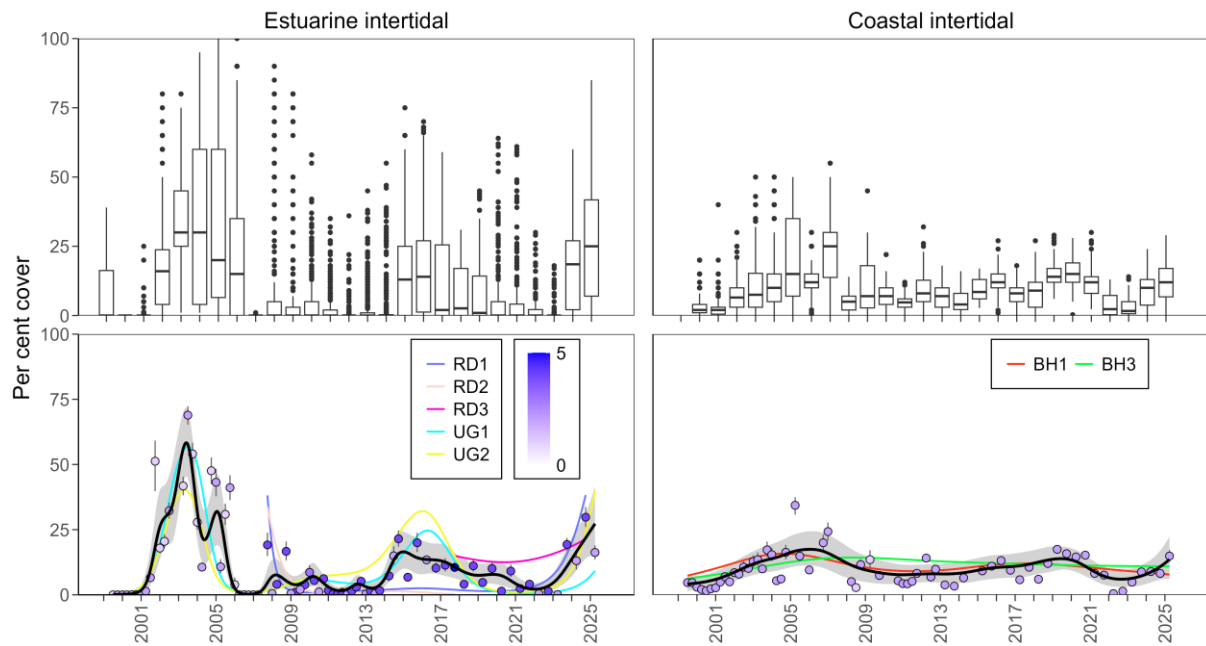


Figure 81. Seagrass per cent cover measures per quadrat (sites pooled) and long-term trends, for each habitat monitored in the Burnett–Mary NRM region from 1999 to 2025. Whisker plots (top) show the box representing the interquartile range of values, where the boundary of the box closest to zero indicates the 25<sup>th</sup> percentile, a line within the box marks the median, and the boundary of the box farthest from zero indicates the 75<sup>th</sup> percentile. Whiskers (error bars) above and below the box indicate the 90<sup>th</sup> and 10<sup>th</sup> percentiles, and the dots represent outlying points. GAM plots (bottom), show trends for each habitat and coloured lines represent individual site trends.

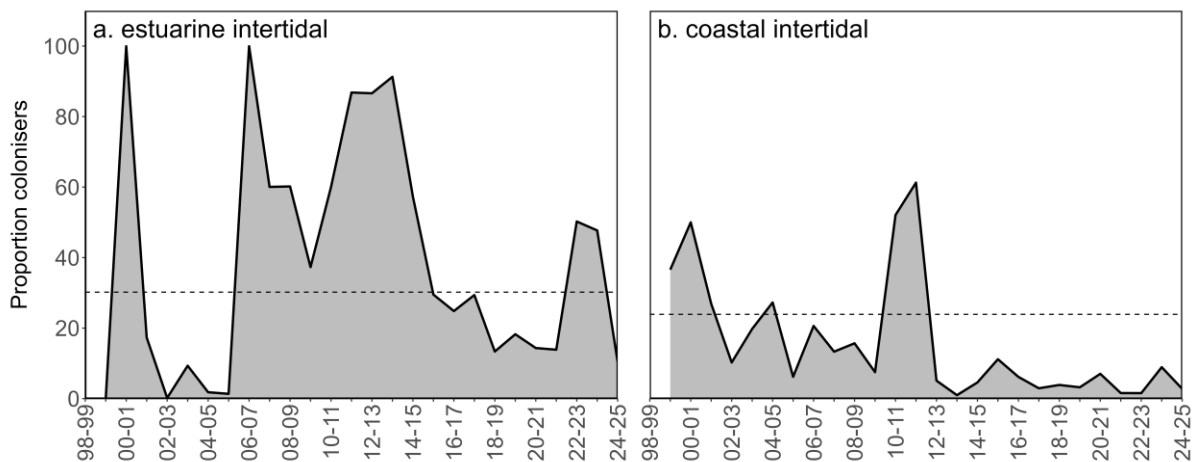


Figure 82. Proportion of seagrass abundance composed of colonising species at: a. estuarine and b. coastal habitats in the Burnett–Mary region, 1998 to 2025. Dashed line represents Reef long-term average proportion of colonising species for each habitat type.

Meadow spatial extent continued to improve across all habitats, increasing during both the late dry and late wet seasons of 2024–25 (Figure 83). Estuarine meadows showed the greatest improvement overall, particularly at the RD1 meadow in Rodds Bay in the north and UG1 at Urangan in the south. The ongoing improvement in the south can be attributed to the meadows becoming less fragmented and more cohesive, as they recover from complete loss in November 2022, which was a consequence of the severe flooding from the nearby Mary River earlier that year.

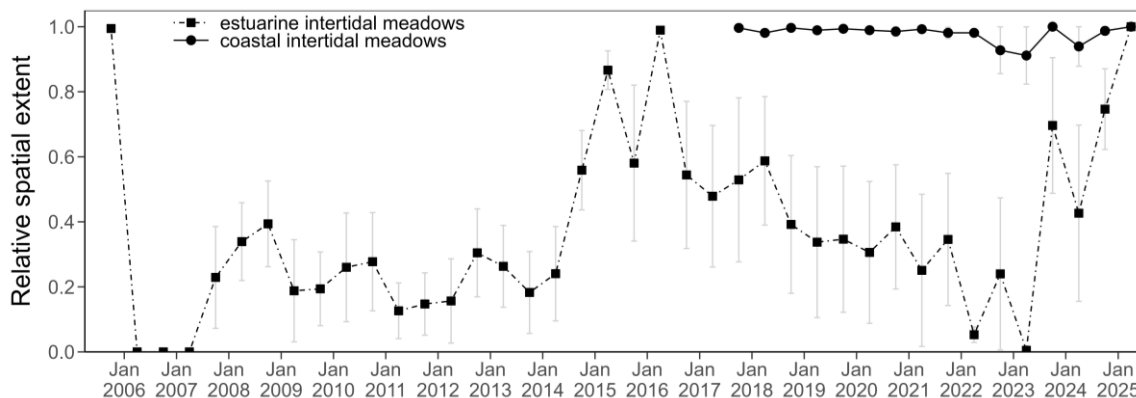


Figure 83. Change in spatial extent ( $\pm$  SE) of estuarine seagrass meadows within monitoring sites for each habitat and monitoring period across the Burnett–Mary NRM region, 2005–2025.

### 5.6.3.3 Seagrass reproductive status

During the 2024–25 period, reproductive effort peaked at its highest level recorded at estuarine sites (Figure 84). Notably, this effort was observed in estuarine habitats in both the northern and southern regions, during the dry season with the largest effort at RD3. Despite these observations, no sexual reproductive structures were detected in the coastal meadows at Burrum Heads again, continuing the trend since 2021. Nevertheless, a seed bank is still present at BH1, indicating that reproduction has been taking place at some time. Seed banks were also found in estuarine meadows in both the northern and southern regions during the 2024–25 period. A significant seed bank ranging from 0 to 2094 seeds per square metre persists at Rodds Bay in the north, while a smaller seed bank of 28 to 57 seeds per square metre remains at Urangan in the south (Figure 84).

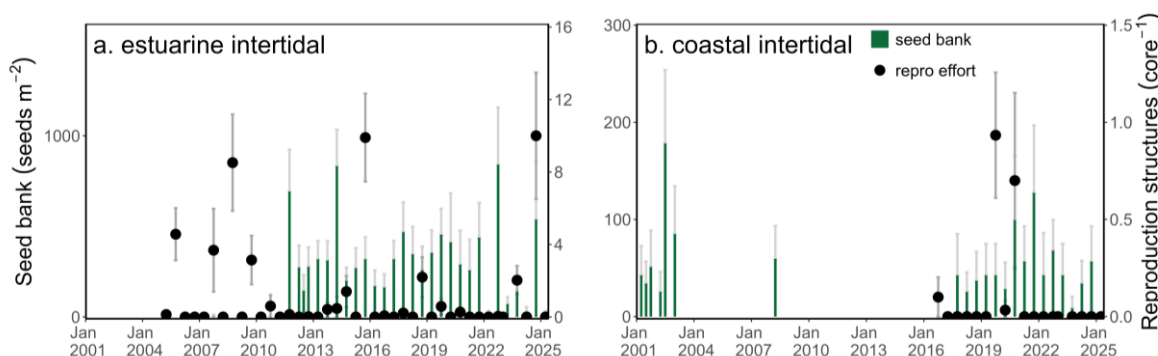


Figure 84. Seedbank and reproductive effort at inshore estuarine (a.) and coastal (b.) intertidal habitats in the Burnett–Mary region, 2001–2025. Seed bank presented as the total number of seeds per  $m^2$  sediment surface (green bars  $\pm$ SE). Reproductive effort for late dry season presented as the average number of reproductive structures per core (species and sites pooled) (circles  $\pm$ SE). NB: Y-axis scale for seed banks and reproductive structures differ between the 2 habitats.

### 5.6.3.4 Resilience

The resilience score was ‘good’ overall in the Burnett–Mary NRM region in 2024–25, increasing from ‘poor’ in 2023–24. The improvements were driven by estuarine habitats (Figure 85).

At Rodds Bay sites, the scores were 96 and 100 due to high counts of reproductive structures of foundational species. Urangan sites increased above thresholds for resistance with abundance increasing and the proportion of colonising species remaining low. There was also a large reproductive effort for the sites. These led to the large rise in the resilience score.

At coastal intertidal sites at Burrum Heads, abundance was above the threshold indicative of resistance. However, the absence of reproductive structures, along with more than 3 years without any reproductive effort, resulted in a low score in category ‘2.1.1’ for both sites. This contributed to a slight overall decline in the score, marking it as the third lowest on record.

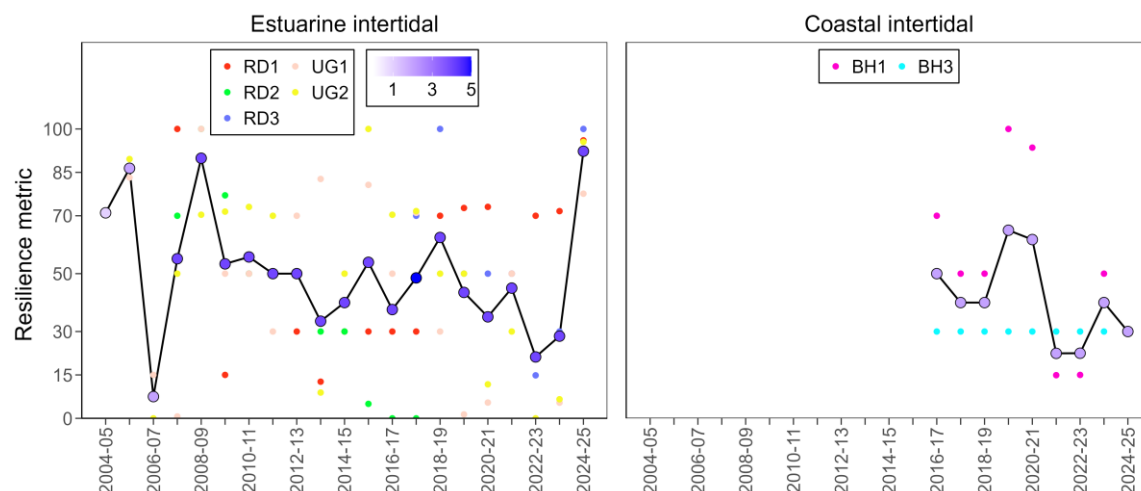


Figure 85. Resilience metric in each habitat in the Burnett–Mary region from 2006 to 2025. Coloured small circles represent different sites. Shades of blue for the larger circles indicate the number of sites that contributed to the metric.

### 5.6.3.5 Epiphytes and macroalgae

Epiphyte cover on seagrass leaf blades in 2024–25 generally decreased, but remained above the long-term average for the eleventh consecutive year at estuarine habitats (Figure 86a). At coastal habitats, epiphyte abundance remained below the long-term average (Figure 86c).

Over the last year per cent cover of macroalgae declined to below the long-term average in estuarine habitats (Figure 86b), while in coastal habitats it remained below the long-term average for the tenth consecutive year (Figure 86d).

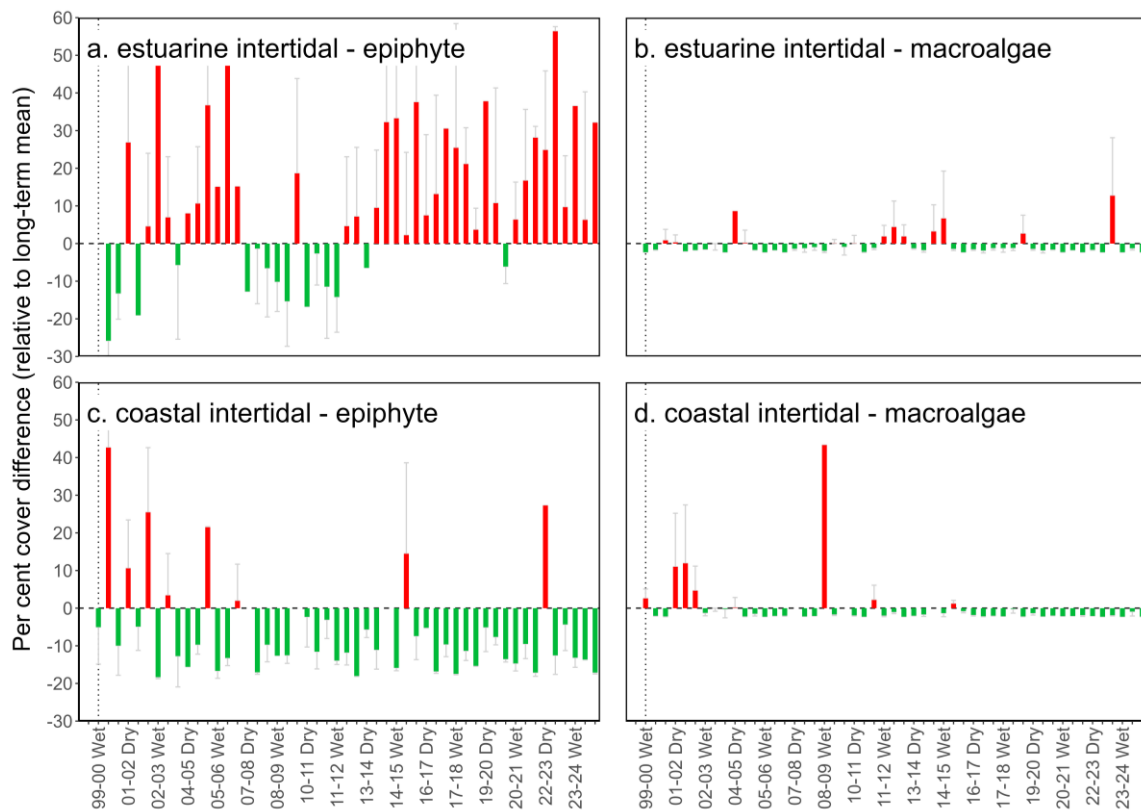


Figure 86. Long-term trend in mean epiphyte and macroalgae abundance (per cent cover) relative to the long-term average for each inshore intertidal seagrass habitat in the Burnett–Mary region, 2000–2025 (sites pooled,  $\pm$ SE). Vertical dotted lines represent the first monitoring event for each habitat type. Red and green bars indicate above and below long-term average, respectively.

## 6 Discussion

The Reef inshore seagrass condition declined by 11 Score points to **‘poor’** in 2024–25, with decreases in both indicators. There were, however, regional differences with differing levels of deterioration in the northern NRM regions (Cape York, Wet Tropics, Burdekin and Mackay–Whitsunday), and improvements in the southern regions (Fitzroy and Burnett–Mary). In 2024–25, inshore seagrass across all northern regions was classified as being in ‘poor’ condition, with Cape York in particular experiencing considerable deterioration. This current and declining state reflects a history of less favourable conditions in preceding years. Therefore, based on previous recovery patterns, recovery rates following 2024–25 may vary by NRM region.

In Cape York, the recovery of these meadows continues to face challenges of below average daily light availability and slightly higher than average seawater temperatures. Abundances being the lowest since monitoring commenced, particularly in the south of the region, and resilience having fallen to an unprecedented low, with sexual reproductive effort dropping to zero across coastal and reef habitats, renders the meadows increasingly vulnerable to future disturbances. Nevertheless, there are glimmers of hope if conditions improve soon: the extent of the meadows has remained relatively stable, foundational species continue to dominate the meadows, and seed banks in coastal meadows are at their highest numbers in 5 years.

In the Burdekin region, prior to the events of 2024–25, seagrass was in a ‘poor’ and declining state. This deterioration follows the extreme climatic events in 2023–24 and the sustained moderate condition in the preceding years. Driven largely by reduced abundance and lack of sexual reproduction, particularly among foundational species at reef intertidal habitats, the depressed resilience (lowest in over a decade) may hinder inshore seagrass recovery in this region, consequently increasing the region’s vulnerability to large disturbances in the near future.

Conversely, although condition remains ‘poor’ in the Wet Tropics, there are promising signs with the onset of recovery across half the locations. These improvements are driven by slightly increasing abundances in reef intertidal and subtidal habitats throughout the region, along with some improvement in the coastal intertidal meadows in the northern part of the region. Resilience of meadows across the region were mixed, as it is likely seagrasses are allocating resources to vegetative growth rather than reproduction, indicated by the lower reproductive effort and seed banks. Drawing from recovery patterns following the extreme weather events from 2009 to 2011, there is potential for recovery within the next year or so, but this is contingent on ongoing environmental conditions and improved reproductive effort.

Seagrasses remain in a ‘moderate’ condition in the Mackay–Whitsunday region, having declined slightly over the last couple of years. Seagrass condition across the region has fluctuated between ‘poor’ and ‘moderate’ since 2010–11, which appears to be due to a range of environmental pressures at both regional and local scales. After 3 years of improvement, abundances declined this reporting period across most habitats, particularly in reef intertidal and subtidal habitats as a result of above-average rainfall and river discharge. Resilience, however, remained ‘moderate’ and stable for the fourth consecutive year in 2024–25, with slight increases at coastal sites and declines at estuarine habitats. Reef intertidal and subtidal sites remained stable but continue to lack reproductive structures, indicating limited recovery potential despite adequate abundance and species composition.

Improvements in the seagrass Index in 2024–25 occurred in the Fitzroy (the highest since 2007–08) and Burnett–Mary regions (highest since MMP established) with both climbing condition grades to ‘moderate’ and ‘good’, respectively. The resilience in the meadows across the regions was buoyed by estuarine and coastal habitats showing strong gains in abundance, coupled with increased prevalence of sexual reproductive structures and persistent seed banks. The Fitzroy reef meadows at Great Keppel Island showed only modest improvements, driven by slight increases in cover of the colonising species which

continue to dominate. The continuing lack of reproductive structures or seeds of foundational species at reef habitats, indicates severely limited recovery capacity and heightened vulnerability to future disturbances.

The resilience of seagrass ecosystems along the roughly fourteen degrees of latitude that the Reef encompasses varies, and faces an escalating array of local and global pressures (Waterhouse *et al.* 2024b). Of concern, is that rising temperatures and heatwaves are also becoming more prevalent. These anomalous events not only appear to be increasing in frequency over the past decade but are also occurring earlier—shifting into the main seagrass growing season, which precedes the wet season. The effects of temperature on biological processes critical to seagrass resilience remain largely unknown. For instance, temperature likely affects factors such as the timing and density of flowering, seed development, sediment condition, seed viability, and germination. Addressing these knowledge gaps are also becoming increasingly urgent as they may influence resilience of seagrass to other pressures such as water quality.

### *Program revision and improvements*

The continuous revision and examination of opportunities for improvement of the monitoring program will ensure that the information is current, relevant, and makes the most for emerging technologies. Immediately prior to the formal commencement of the MMP in 2005 (Haynes *et al.* 2005), available inshore seagrass data was reviewed (De'ath 2005) and the inshore seagrass component was implemented as recommended by the RWQPP Seagrass Expert Panel and the independent reviewers (Great Barrier Reef Marine Park Authority 2004). Over the next 20 years, the MMP has undergone a number of reviews, including during development of the Paddock to Reef report card in 2007–08, and the Paddock to Reef 5-year review in 2018 (Waterhouse *et al.* 2018). Each year the MMP annual reports are independently externally scientifically peer reviewed. The most significant MMP review was in 2013–14, after which a number of recommendations were implemented (Kuhnert *et al.* 2014). With the next review scheduled for early 2026, it is timely to explore fresh opportunities and propose enhancements to strengthen the program to ensure value-for-money and best practice in marine monitoring.

Currently, the MMP inshore seagrass monitoring annually reports condition across a maximum of 80 sites, including 47 sentinel sites assessed by JCU which report on both indicators. For the remaining sites, JCU works in partnership with the RJFMP, along with various Citizen Science and First Nation groups, who submit valuable contributory data and/or assist with field assessments. To maintain the quality of the contributory data, Citizen scientists participate in capacity-building initiatives through the Seagrass-Watch global seagrass observing network where they must demonstrate competency in globally standardised methods before their data can be integrated into the MMP. QPWS rangers, in collaboration with the RJFMP, as well as Indigenous Rangers, also receive capacity-building support where they are trained to collect standardised photoquadrat imagery using a drop-camera in subtidal habitats. The collected imagery is submitted to Seagrass-Watch, where quantitative data is extracted from the photoquadrats and integrated into the MMP. All data used for reporting, including that gathered in collaboration with our partners, undergoes rigorous quality control ensuring accuracy and rigor. Currently, contributory data is captured from approximately forty percent of the long-term monitoring sites used to report on the condition of inshore seagrass in the Reef.

Capturing data from additional monitoring sites or other seagrass assessment programs would improve the spread and resolution of information to assist with condition reporting, particularly in more remote locations or locations at greatest risk from human activities.

The Queensland Ports Seagrass Monitoring Program (QPSMP) has been monitoring seagrass meadows in a number of high risk locations, e.g., ports, since 1993 (Lee Long *et al.* 1996; McKenzie *et al.* 1996; McKenzie *et al.* 1998; Rasheed *et al.* 2017). Although the

program also presents a tailored report card (Carter *et al.* 2023), there are unfortunately, a number of matters which to date exclude the integration of data into the MMP, these include:

- *sampling unit measure standardisation*: the QPSMP reports visual estimates of seagrass biomass, which are transformed measurements derived from raw field ranks and not classified as one of the global seagrass Essential Ocean Variables (EOV) (Duffy *et al.* 2019; Duffy *et al.* 2026). Additionally, the QPSMP does not capture geolocated or geotagged photoquadrats as part of its field protocol, relying solely on human observation, which provides no opportunity for upcycling, data validation or accuracy assessments as part of the MMP quality assurance and quality control (QAQC) process (Gruber *et al.* 2025).
- *selection of indicators*: the QPSMP measures a variety of indicators, including the area of each 'monitoring meadow'. The definition of a meadow is unclear and subjective, as meadowscape, such as patches and fragmentation, are disregarded in the measure. The method used to delineate 'monitoring meadow' boundaries may also result in potentially overestimating the area of seagrass by 6% to 80% (McKenzie *et al.* 2019).
- *abundance baseline determination*: the MMP uses all available sampling events until abundance variance plateaus to provide estimates of true percentile values to define thresholds (Appendix 1). The QPSMP uses the average of 10 annual sampling events and CV of 40% to define thresholds (Carter *et al.* 2023).
- *reporting unit Minimal Detectable Difference*: the MMP requires that the MDD of the abundance indicator for a reporting unit must not exceed 20 per cent (at the 5 per cent level of significance with 80 per cent power) (see 2.2.1) The variability of the QPSMP abundance indicator results in MDDs ranging from 30% to 75% during the late dry season at the 90% level with 90% assurance (Lee Long *et al.* 1996; McKenzie *et al.* 1998).
- *reporting units not fixed in size and position*: MMP reporting units are fixed 5.5 ha sites which are equally weighted, while the QPSMP reports on 'monitoring meadows' which are both spatially and temporally variable in size and position, but equally weighted. This approach compromises regional condition reporting where the condition of a few large meadows dominating an Area of Interest may be overwhelmed by the condition of many minor meadows (see Carter *et al.* 2023).

If these obstacles could be overcome, then it may be possible to integrate QPSMP data if the data owners are willing to share their data.

To further strengthen the program, implementing several key enhancements will create a more integrated, predictive, and scalable approach to help understand the resilience and long-term health of the inshore seagrass habitats, given the escalating pressures they face. The most urgent improvements include:

1. **Developing methods to summarise cumulative pressures and indices of pressure** is crucial. With multiple pressures occurring simultaneously or successively, it is often difficult to pinpoint the cause of damage. A more integrated approach bringing multiple stressors together (e.g. light limitation, temperature, nutrients, pesticides, sedimentation) would provide critical insights into the combined effects of these pressures, enabling monitoring to reflect real-world conditions, where impacts rarely occur in isolation. This improves the ability to diagnose causes of decline, assess resilience, and prioritise management actions rather than responding to single pressures in isolation.
2. **Developing a spatial inshore thermal stress risk model** (currently underway) and refining temperature thresholds to better understand their impact on resilience. This is vital for addressing the growing threat of temperature-induced stress.
3. **Updating light indicators and thresholds** to enhance the effectiveness of *in situ* light monitoring for intertidal habitats while exploring ways to leverage existing tools

(e.g., eReefs) to improve pressures reporting for inshore subtidal habitats. These are essential for increasing accuracy and consistency of assessments of light stress across inshore seagrass habitats, which leads to more reliable detection of stress and recovery trajectories.

4. **Creating a fragmentation index for meadow-scape condition** based on current and historical seagrass extent data and establishing protocols for drone use in spatial extent and fragmentation monitoring. This is critical because patchiness influences connectivity, resilience, and ecological function. This is key to better understanding when habitats approach ecological tipping points that could accelerate loss.
5. **Incorporating quantitative indicators of local-scale processes** into routine pressure analyses such as sediment movement or grazing pressures, however, the practicalities of measuring these need to be explored. This would enable a more comprehensive evaluation of the factors affecting inshore seagrass habitats and to identify local versus broad-scale management actions.
6. **Scaling monitoring efforts** to broader levels to fully capture habitat decline and recovery. 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. This allows for high-resolution, repeatable surveys over large areas, improves detection of change, and links local observations to regional patterns—enhancing both efficiency and ecological interpretation.

Together, these approaches improve sensitivity (detecting early change), attribution (linking change to drivers and pressures), prediction (forecasting risk and recovery), and scalability (from site to regional assessments). In short, they could transform the MMP inshore seagrass monitoring into a more holistic, data-rich, and decision-relevant system. Addressing these enhancements is not just important—it is imperative for safeguarding the future of the Reef’s inshore seagrass ecosystems and their ecological functions.

## 7 Conclusion

In 2024–25 inshore seagrass meadows across the Reef declined in overall condition, with a grade change in the Seagrass Index from ‘moderate’ to ‘poor’. Reef-wide inshore seagrass abundance declined in 2024–25, resulting in a grade change from ‘moderate’ to ‘poor’, following only a marginal decline in the previous period score. Resilience continued to decline in 2024–25, mirroring the overall Index, and reached its lowest score in more than a decade while remaining ‘poor’.

Environmental conditions varied across the Reef, with rainfall and elevated discharges affecting all NRM regions, except the Fitzroy. The largest discharge anomalies occurred in the southern Wet Tropics, Burdekin, Mackay–Whitsunday and Burnett–Mary NRM regions. The extent of exposure to turbid coloured waters was elevated in those regions, and there was a shift towards more seagrass exposed to higher risk categories. Overall, daily average benthic light for seagrass was below the long-term average where recorded, in particular in the regions affected by elevated discharge. Plants continued to be exposed to above average water temperatures, but with fewer days of extreme heat (>40 °C).

In 2024–25, the inshore seagrass was in a ‘poor’ condition in the northern NRM regions, and ‘moderate’ to ‘good’ in the southern regions. Seagrass condition in the far northern region (Cape York) declined to ‘poor’, whereas condition in the 2 southern most regions (Fitzroy and Burnett–Mary) improved to ‘moderate’ and ‘good’, respectively. Condition in the northern (Wet Tropics and Burdekin) and central (Mackay–Whitsunday) regions remained largely unchanged (but with declines in score) in ‘poor’ and ‘moderate’ states, respectively.

The inshore Reef seagrass meadows 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 drive these changes (i.e. declines and subsequent recovery) varies between habitats, communities and regions. Inshore seagrass meadows of the Reef were in an overall ‘good’ state in 2007–08. In particular, 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 87).

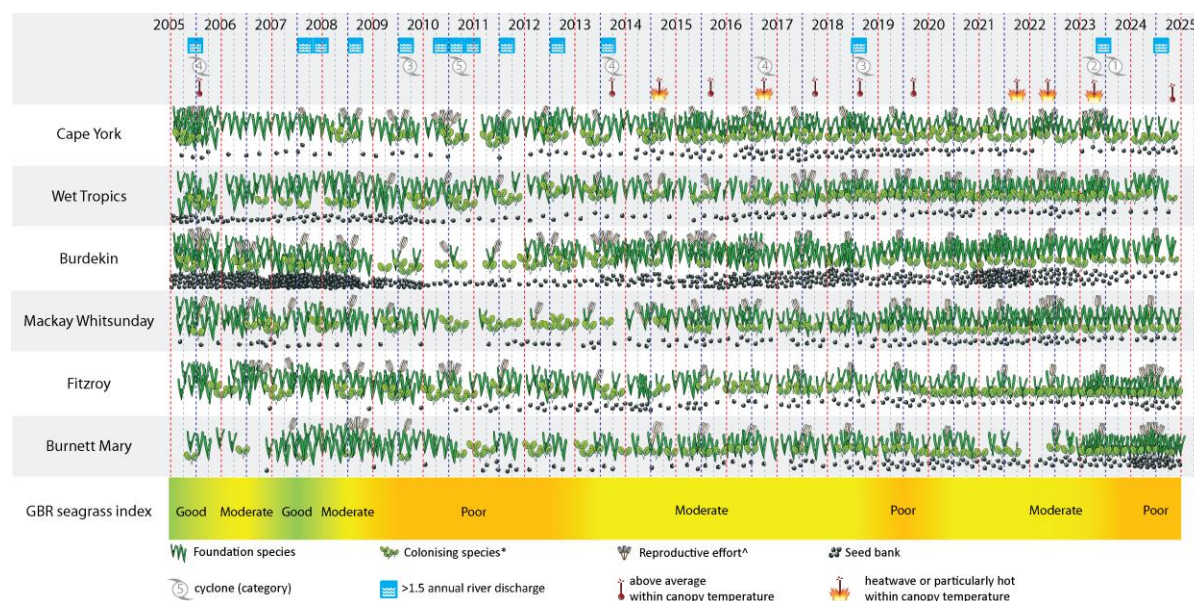


Figure 87. Summary of inshore seagrass state illustrating pressures, abundance of foundation / colonising species, seed bank and reproductive effort in each NRM from 2005 to 2025. \* colonising species are represented by the genus *Halophila*, however, *Zostera* and *Halodule* can be both colonising and foundational species depending on meadow state. ^ not conducted in 2005.

In late 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 began the gradual decline. Cumulative pressures, including severe climatic events (Cyclone Debbie), continued to undermine the resilience of inshore seagrass meadows of the Reef. Frequent and repeated disturbances seemed 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. By 2019–20, the inshore Reef seagrass had fallen back to a ‘poor’ state. Since then, recovery had been buoyed across northern and central regions by a few years of low to negligible climatic pressures, while in the most southern regions (Fitzroy and Burnett–Mary) consistent declines undermined improvements. Nevertheless, the events of 2023–24 reversed those trends, resulting in considerable deterioration in the northern regions due to cyclones and associated flooding, while the southern regions experienced significant recovery with more favourable conditions for seagrass growth.

The sustained improvement of the Reef’s inshore seagrass meadows depends on various factors, such as favourable growth conditions and effective environmental protection measures. While we cannot control weather patterns, we can mitigate their impact on seagrasses by implementing initiatives like the Paddock to Reef Program that reduce land-based runoff to the Reef. It is essential to prioritise the resilience of seagrass meadows, particularly their capacity to recover from damage, in our research and management strategies.

Practicable conservation opportunities also exist, which can make substantial and quantifiable improvements to seagrass condition. In addition to managing pressures, there are direct actions that can be taken to facilitate recovery and build resilience. Some of these include:

1. Developing accurate models of seagrass recovery to identify when recovery is on track or when intervention actions may be required.
2. Active seagrass restoration or enhancement of resilience may be of benefit, but significant research is required before techniques can be operationalised at scale (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.
3. Establishing Seagrass Resilience Centres/Nurseries to facilitate research which underpins real-world restoration efforts, ensuring that restored meadows have the best chance of thriving. These facilities provide an opportunity to experiment with

different restoration techniques, test different strategies for enhancing seed production and germination rates, or explore the benefits of symbiotic relationships with marine organisms. This hands-on approach will aid in identifying the most effective strategies for enhancing seagrass resilience and facilitating natural recovery.

4. Proactively building resilience in seagrass meadows to elevated temperatures by facilitating adaptation to future conditions by introducing 'pre-adapted individuals' into meadows within the current distribution range; thermally priming seedlings prior to restoration initiatives; and, selecting genetic variants of species that exhibit greater thermal tolerance for upcoming restoration projects.

To ensure the long-term health of the Reef's seagrass ecosystems, it is vital to advance our understanding of ecosystem science related to resilience and recovery. In addition to comprehensive research, adaptive resilience-based management is crucial. This approach should prioritize forecasting tools to guide planning and actions, along with monitoring and diagnostic tools to refine and implement strategies that enhance resilience, optimise recovery, and lessen disturbances or impacts.

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## **Appendix 1 Seagrass condition indicator guidelines**

## A1.1 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 80<sup>th</sup>, 50<sup>th</sup> and 20<sup>th</sup> percentiles were used to define the guideline values as these are recommended for water quality guidelines (DEHP 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 80<sup>th</sup> percentile was changed to 75<sup>th</sup> 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 88).

Across the majority of reference sites, variance for the 50<sup>th</sup> and 20<sup>th</sup> 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 plateaued, the percentile values at 24 sampling events was selected to best represent the variance as being captured. This conforms with Kilminster *et al.* (2015) definition where an enduring meadow is present for 5 years.

Nonlinear regressions (exponential rise to maximum, 2 parameter) were then fitted to per cent cover percentile values at each number of sampling events using the following model:

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

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 19).

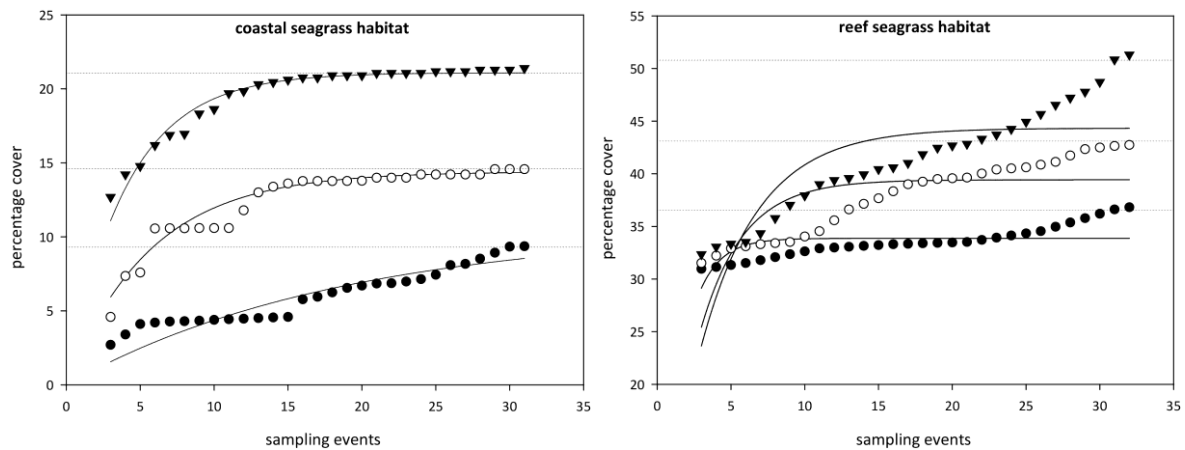


Figure 88. Relationship between sample size and the error in estimation of percentile values for seagrass abundance (per cent cover) in coastal and reef seagrass habitats in the Wet Tropics NRM. ▼ = 75<sup>th</sup> percentile, ○ = 50<sup>th</sup> percentile, ● = 20<sup>th</sup> percentile. Horizontal 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 20<sup>th</sup> percentile at a reference site should be based on a minimum of 18 samples collected over at least 3 years. For the 50<sup>th</sup> 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 20<sup>th</sup> percentile anyway. For seagrass habitats with low variability, a more appropriate guideline was the 10<sup>th</sup> percentile primarily the result of seasonal fluctuations (as nearly every seasonal low would fall below the 20<sup>th</sup> percentile). Percentile variability was further reduced within a habitat type of each region by pooling at least 2 (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 50<sup>th</sup> percentile)
- moderate (median abundance below 50<sup>th</sup> percentile and at or above 20<sup>th</sup> percentile)
- poor (median abundance below 20<sup>th</sup> or 10<sup>th</sup> percentile).

For example, when the median seagrass abundance for Yule Point is plotted against the 20<sup>th</sup> and 50<sup>th</sup> percentiles for coastal habitats in the Wet Tropics (Figure 89), it indicates that the meadows were in a poor condition in mid-2000, mid-2001 and mid-2006 (based on abundance).

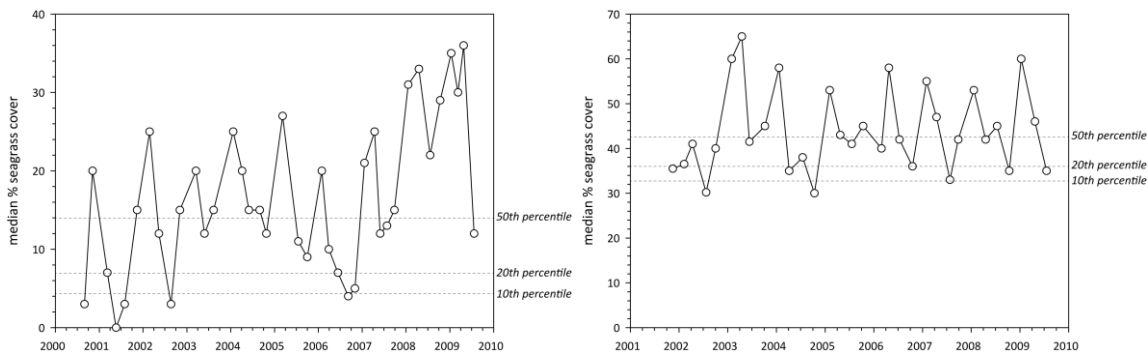


Figure 89. Median seagrass abundance (per cent cover) plotted against the 50<sup>th</sup>, 20<sup>th</sup> and 10<sup>th</sup> percentiles at intertidal coastal (Yule Point, left) and intertidal reef habitats (Green Island, right) in the Wet Tropics.

Similarly, when the median seagrass abundance for Green Island is plotted against the 20<sup>th</sup> and 50<sup>th</sup> 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 10<sup>th</sup> rather than the 20<sup>th</sup> percentile.

Using this approach, subregional seagrass abundance guidelines (hereafter known as “the seagrass guidelines”) were developed for each seagrass habitat type where possible (Table 19). 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 Reef Authority 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 75<sup>th</sup> percentile)
- very poor (median abundance below 20<sup>th</sup> or 10<sup>th</sup> percentile and declined by >20 per cent since previous sampling event).

Seagrass state was then rescaled to a 5 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 19. 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			
			10 <sup>th</sup>	20 <sup>th</sup>	50 <sup>th</sup>	75 <sup>th</sup>
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	<b>reef intertidal</b>	<b>11</b>	<b>16.8</b>	<b>18.9</b>	<b>23.7</b>
	FG	reef subtidal		26	33	39.2
	NRM	<b>reef subtidal*</b>	<b>22</b>	<b>26</b>	<b>33</b>	<b>39.2</b>
	BY*	coastal intertidal		6.6	12.9	14.8
	SR*	coastal intertidal		6.6	12.9	14.8
	NRM	<b>coastal intertidal*</b>	<b>5</b>	<b>6.6</b>	<b>12.9</b>	<b>14.8</b>
	BY*	coastal subtidal		6.6	12.9	14.8
	LR*	coastal subtidal		6.6	12.9	14.8
MA*	coastal subtidal		6.6	12.9	14.8	
NRM	<b>coastal subtidal*</b>		<b>6.6</b>	<b>12.9</b>	<b>14.8</b>	
Wet Tropics	LB	coastal intertidal		6.6	12.9	14.8
	YP1^	coastal intertidal	4.3	7	14	15.4

NRM region	site/ location	Habitat	percentile guideline			
			10 <sup>th</sup>	20 <sup>th</sup>	50 <sup>th</sup>	75 <sup>th</sup>
NRM	YP2 <sup>^</sup>	coastal intertidal	5.7	6.2	11.8	14.2
	NRM	<b>coastal intertidal</b>	<b>5</b>	<b>6.6</b>	<b>12.9</b>	<b>14.8</b>
	MS	coastal subtidal		6.6	12.9	14.8
	NRM	<b>coastal subtidal</b>		<b>6.6</b>	<b>12.9</b>	<b>14.8</b>
	DI	reef intertidal	27.5		37.7	41
	GI1 <sup>^</sup>	reef intertidal	32.5	38.2	42.7	45.5
	GI2 <sup>^</sup>	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	<b>reef intertidal</b>	<b>27.5</b>	<b>31.9</b>	<b>37.7</b>	<b>41</b>
	DI3	reef subtidal		26	33	39.2
	GI3 <sup>^</sup>	reef subtidal	22	26	33	39.2
	LI2	reef subtidal		26	33	39.2
	NRM	<b>reef subtidal</b>	<b>22</b>	<b>26</b>	<b>33</b>	<b>39.2</b>
Burdekin	BB1 <sup>^</sup>	coastal intertidal	16.3	21.4	25.4	35.2
	SB1 <sup>^</sup>	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		13.2	19.1	22.2
	NRM	<b>coastal intertidal</b>	<b>11.9</b>	<b>15.7</b>	<b>21.1</b>	<b>28.6</b>
	MI1 <sup>^</sup>	reef intertidal	23	26	33.4	37
	MI2 <sup>^</sup>	reef intertidal	21.3	26.5	35.6	41
	NRM	<b>reef intertidal</b>	<b>22.2</b>	<b>26.3</b>	<b>34.5</b>	<b>39</b>
	MI3 <sup>^</sup>	reef subtidal	18	22.5	32.7	36.7
	NRM	<b>reef subtidal</b>	<b>18</b>	<b>22.5</b>	<b>32.7</b>	<b>36.7</b>
	Mackay–Whitsunday	SI	estuarine intertidal		18	34.1
NRM		<b>estuarine intertidal</b>	<b>10.8*</b>	<b>18*</b>	<b>34.1*</b>	<b>54*</b>
PI2 <sup>^</sup>		coastal intertidal	18.1	18.7	25.1	27.6
PI3 <sup>^</sup>		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
LL		coastal intertidal		13.2	19.1	22.2
SH1		coastal intertidal		13.2	19.1	22.2
NRM		<b>coastal intertidal</b>	<b>12.1</b>	<b>13.2</b>	<b>19.1</b>	<b>22.2</b>
NB		coastal subtidal		13.2	19.1	22.2
NRM		<b>coastal subtidal</b>	<b>12.1</b>	<b>13.2</b>	<b>19.1</b>	<b>22.2</b>
HB1 <sup>^</sup>		reef intertidal		10.53	12.9	14.2
HB2 <sup>^</sup>		reef intertidal		7.95	11.59	13.4
HM		reef intertidal		9.2	12.2	13.8
LN3		reef intertidal		9.2	12.2	13.8
NRM		<b>reef intertidal</b>		<b>9.2</b>	<b>12.2</b>	<b>13.8</b>
CH		reef subtidal		22.5	32.7	36.7
LN		reef subtidal		22.5	32.7	36.7
TO		reef subtidal		22.5	32.7	36.7
WB		reef subtidal		22.5	32.7	36.7
NRM		<b>reef subtidal*</b>	<b>18*</b>	<b>22.5*</b>	<b>32.7*</b>	<b>36.7*</b>
Fitzroy		GH	estuarine intertidal		18	34.1
	NRM	<b>estuarine intertidal</b>	<b>10.8*</b>	<b>18*</b>	<b>34.1*</b>	<b>54*</b>
	RC1 <sup>^</sup>	coastal intertidal	18.6	20.6	24.4	34.5
	WH1 <sup>^</sup>	coastal intertidal	13.1	14.4	18.8	22.3
	NRM	<b>coastal intertidal</b>	<b>15.85</b>	<b>17.5</b>	<b>21.6</b>	<b>28.4</b>
	GK	reef intertidal		9.2	12.2	13.8
NRM	<b>reef intertidal</b>		<b>9.2*</b>	<b>12.2*</b>	<b>13.8*</b>	
Burnett–Mary	RD	estuarine intertidal		18	34.1	54
	UG1 <sup>^</sup>	estuarine intertidal	10.8	18	34.1	54
	UG2	estuarine intertidal		18	34.1	54
	NRM	<b>estuarine intertidal</b>	<b>10.8</b>	<b>18</b>	<b>34.1</b>	<b>54</b>
	BH1 <sup>^</sup>	coastal intertidal		7.8	11.9	21.6
BH3	coastal intertidal		7.8	11.9	21.6	
NRM	<b>coastal intertidal</b>		<b>7.8</b>	<b>11.9</b>	<b>21.6</b>	

## A1.2 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 2 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 90). 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 2 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).

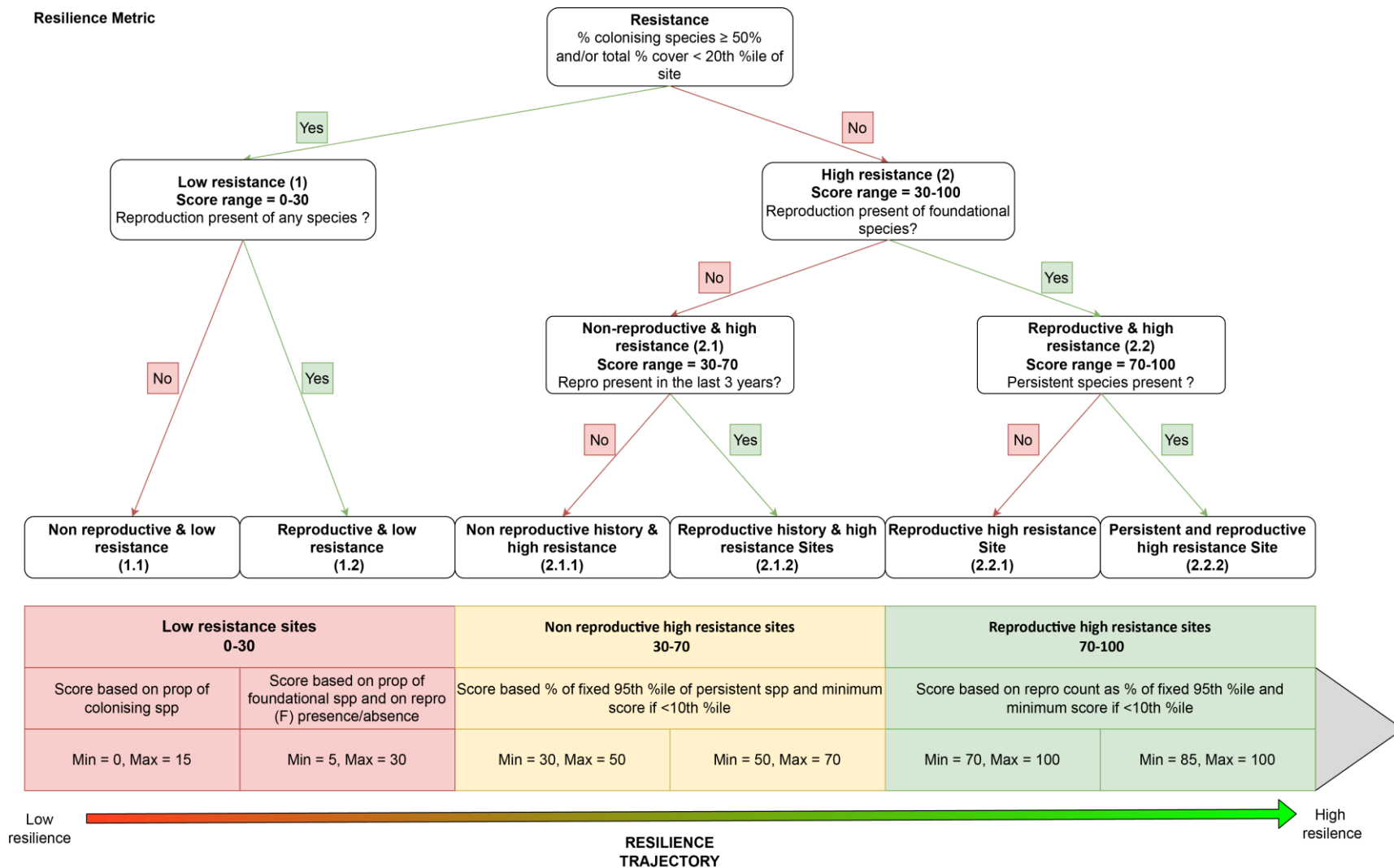


Figure 90. Overall structure of the proposed MMP resilience metric. The score ranges from 0 to 100. Splits in the tree are used to place a site in a grouping (red, yellow, or green), with grading within each grouping based on species composition and reproductive effort. Reproduction refers to sexual reproduction. From Collier *et al.* (2021a).

## **Appendix 2 Detailed data**

Table 20. Samples collected at each inshore monitoring site per parameter for each season. Activities include: SG = seagrass cover & composition, SB=seed bank monitoring, EM=edge mapping, RH=reproductive effort, TL=temperature loggers, LL=light loggers. ^=subtidal.

Reef region	NRM region	Basin	Monitoring location	late dry Season (2024)						late wet Season (2025)							
				SG	SB	EM	RH	TL	LL	SG	SB	EM	RH	TL	LL		
Far Northern	Cape York	Jacky Jacky / Olive Pascoe	Shelburne Bay	SR1	33	30	✓	15	✓								
				SR2	33	30	✓	15	✓	✓							
			Margaret Bay	MA1	10												
				MA2	10												
		Piper Reef	FR1	33	30	✓	15	✓									
			FR2	33	30	✓	15	✓	✓								
		Lockhart	Weymouth Bay	YY1													
			Lloyd Bay	LR1^	10												
				LR2^	10												
		Flinders Group	ST1	33	30	✓	15	✓	✓								
			ST2	33	30	✓	15	✓									
			FG1^	10													
			FG2^	10													
		Normanby / Jeanie	Bathurst Bay	BY1	33	30	✓	15	✓								
				BY2	33	30	✓	15	✓	✓							
				BY3^	10												
	BY4^		10														
Endeavour	Archer Point	AP1															
		AP2															
Northern	Wet Tropics	Daintree	Low Isles	LI1	33	30	✓	15	✓	✓	33	30	✓	15	✓	✓	
				LI2^	33			15			33			15			
		Mossman / Barron / Mulgrave - Russell / Johnstone	Yule Point	YP1	33	30	✓	15	✓				33	30	✓	15	✓
				YP2	33	30	✓	15	✓	✓				33	30	✓	15
		Green Island	GI1	33	30	✓	15	✓	✓				33	30	✓	15	✓
			GI2	33	30	✓	15	✓					33	30	✓	15	✓
			GI3^	33			15										
		Mission Beach	LB1	33	30	✓	15						33	30	✓	15	
			LB2	33	30	✓	15						33	30	✓	15	
		Tully / Murray / Herbert	Dunk Island	DI1	33	30	✓	15	✓				33	30	✓	15	✓
				DI2	33	30	✓	15	✓	✓				33	30	✓	15
				DI3^	33			15						33			15
Rockingham Bay	GO1																
Missionary Bay	MS1^	10															
	MS2^	10															
Central	Burdekin	Ross / Burdekin	Magnetic Island	MI1	33	30	✓	15	✓		33	30	✓	15	✓		
				MI2	33	30	✓	15	✓	✓				33	30	✓	
				MI3^	43			15						33			15
		Townsville	SB1	33	30	✓	15	✓	✓				33	30	✓	15	✓
			SB2	33				✓					33	30			✓
			BB1	33	30	✓	15	✓	✓				33	30	✓	15	✓
		Bowling Green Bay	JR1	33	30	✓	15	✓	✓							✓	✓
			JR2	33	30	✓	15	✓								✓	
Don	Bowen	BW2	33														

Reef region	NRM region	Basin	Monitoring location	late dry Season (2024)						late wet Season (2025)						
				SG	SB	EM	RH	TL	LL	SG	SB	EM	RH	TL	LL	
Southern	Mackay–Whitsunday	Don	Shoal Bay	BW3	33											
				HB1	33				✓						✓	
				HB2	33				✓						✓	
		Proserpine	Pioneer Bay	PI2	33										✓	
				PI3	33					✓					✓	
		Repulse Bay		MP2	33	30	✓	15	✓	✓	33	30	✓	15	✓	✓
				MP3	33	30	✓	15	✓	✓	33	30	✓	15	✓	✓
		Hamilton Is.		HM1	33	30	✓	15	✓	✓	33	30	✓	15	✓	✓
				HM2	30	30	✓	15	✓	✓	30	30	✓	15	✓	✓
		Proserpine / O'Connell	Cid Harbour	CH4^	10											
				CH5^	10											
			Whitsunday Island	TO1^	10											
				TO2^	10											
				WB1^	10											
			WB3^	10												
		Lindeman Island		LN1^	37		✓	15			33	30	✓	15	✓	✓
				LN3	33	30	✓	15	✓	✓	33	30	✓	15	✓	✓
		O'Connell	St Helens Bay	SH1												
			Newry Islands	NB1^	10											
				NB2^	10											
Plane	Sarina Inlet	SI1	33	30	✓	15	✓	✓	33	30	✓	15	✓	✓		
		SI2	33	30	✓	15	✓	✓	33	30	✓	15	✓	✓		
Llewellyn Bay		LL1						✓	33				✓			
		CV1						✓	33	30			✓			
	Clairview	CV2							33	30						
Southern	Fitzroy	Shoalwater Bay	RC1	33	30	✓	15	✓						✓	✓	
			WH1	33	30	✓	15	✓	✓					✓	✓	
		Great Keppel Island	GK1	33	30	✓	15	✓	✓					✓	✓	
			GK2	33	30	✓	15	✓	✓					✓	✓	
		Boyerne	Gladstone Harbour	GH1	33	30	✓	15	✓	✓					✓	✓
				GH2	33	30	✓	15	✓	✓					✓	✓
	Burnett	Rodds Bay	RD1	33	30	✓	15	✓	✓					✓	✓	
			RD3	33	30	✓	15	✓	✓					✓	✓	
	Burnett–Mary	Burrum	Burrum Heads	BH1	33	30	✓	15	✓		33	30	✓	15	✓	
				BH3	33	30	✓	15	✓		33	30	✓	15	✓	
Mary	Hervey Bay	UG1	33	30	✓	15	✓		33	30	✓	15	✓			
		UG2	33	30	✓	15	✓	✓	33	30	✓	15	✓	✓		

## A2.1 Environmental pressures

### A2.1.1 Tidal exposure

Table 21. Height of intertidal monitoring meadows/sites above lowest astronomical tide (LAT) relative to the reference port (tidal gauge or tidal predictions) and annual daytime tidal exposure (total hours) when meadows become exposed at a low tide. Year is June–May. Observed and predicted tidal heights courtesy Maritime Safety Queensland and BOM, 2025. \* Sites were tidal predictions are used instead of tidal gauge observations. NB: Relative meadow heights have been approximated using various methods and are subject to change over time.

NRM	Site	Meadow height (above LAT) relative to Reference Port	Reference Port name	Annual median hours exposed during daylight (long-term)	Per cent of annual daylight hours meadow exposed (long-term)	Annual daytime exposure 2023–24 (hrs)	Per cent of annual daylight hours meadow exposed (2023–24)
Cape York	SR1*	0.8	Round Point	479.8	10.7	491.7	11.1
	SR2*	0.8	Round Point	479.8	10.7	491.7	11.1
	FR1*	0.44	Kay Reef	140.5	3.1	148.2	3.3
	FR2*	0.44	Kay Reef	140.5	3.1	148.2	3.3
	ST2*	0.25	Stanley Island	54.7	1.2	65.5	1.5
	ST1*	0.3	Stanley Island	35	0.8	43	1
	BY1*	0.75	Cape Melville	310	6.9	344.8	7.8
	BY2*	0.75	Cape Melville	310	6.9	344.8	7.8
Wet Tropics	LI1*	0.65	Low Islets	181.5	4.1	208.5	4.7
	YP1*	0.52	Port Douglas	71.8	1.6	88.7	2
	YP2*	0.64	Port Douglas	132.2	2.9	158.3	3.6
	GI1	0.6	Cairns	98.7	2.3	100.7	2.3
	GI2	0.67	Cairns	138.7	3.1	137.7	3.1
	DI1	0.54	Clump Point	97.2	2.1	100.5	2.3
	DI2	0.44	Clump Point	60.5	1.3	57.5	1.3
	LB1	0.31	Clump Point	27.8	0.6	26.5	0.6
	LB2	0.35	Clump Point	36.2	0.8	35.3	0.8
Burdekin	BB1	0.58	Townsville	60.5	1.3	68.8	1.6
	SB1	0.57	Townsville	86.3	2	107.2	2.4
	SB2	0.8	Townsville	53	1.2	59.7	1.3
	MI1	0.67	Townsville	56.3	1.3	65.2	1.5
	MI2	0.56	Townsville	151.5	3.5	175.5	4
	JR1	0.55	Cape Ferguson	81.5	1.8	78.2	1.8
	JR2	0.55	Cape Ferguson	81.5	1.8	78.2	1.8
	BW1	0.9	Bowen	353.2	8	361.3	8.1
	BW3	0.9	Bowen	353.2	8	361.3	8.1
Mackay–Whitsunday	PI2*	0.44	Shute Harbour	35	0.8	33.8	0.8
	PI3*	0.33	Shute Harbour	35	0.8	33.8	0.8
	HM1*	0.38	Shute Harbour	52.3	1.2	57.2	1.3
	HM3*	0.38	Shute Harbour	24.3	0.5	23.8	0.5
	LN3*	0.65	Shute Harbour	173.8	4	175.5	4
	MP2	0.8	Half tide	62.3	1.4	50.3	1.1
	MP3	0.8	Half tide	62.3	1.4	50.3	1.1
	SI1	0.6	Half tide	128.3	2.8	87	2
	SI2	0.6	Half tide	24.2	0.6	21.5	0.5
	LL1	1	Half tide	24.2	0.6	21.5	0.5

	CV1	1.7	Half tide	534.7	11.8	485.8	10.9
	CV2	1.7	Half tide	534.7	11.8	485.8	10.9
Fitzroy	RC1	1.22	Half tide	241.8	5.4	178.3	4
	WH1	1.35	Half tide	329.8	7.3	253.5	5.7
	GK1	0.59	Rosslyn Bay	75.5	1.6	46	1
	GK2	0.62	Rosslyn Bay	89.7	1.9	51.5	1.2
	GH1	0.69	Gladstone	91.7	2	53.7	1.2
	GH2	0.69	Gladstone	91.7	2	53.7	1.2
	Burnett–Mary	RD3	0.56	Gladstone	95	2.1	56.7
RD1		0.63	Gladstone	42.3	1	30.2	0.7
BH1		0.75	Urangan	149.3	3.4	93.2	2.1
BH3		0.9	Urangan	281.7	6.4	195.2	4.4
UG1		0.70	Urangan	114	2.6	70.2	1.6
UG2		0.64	Urangan	76.7	1.7	53.3	1.2

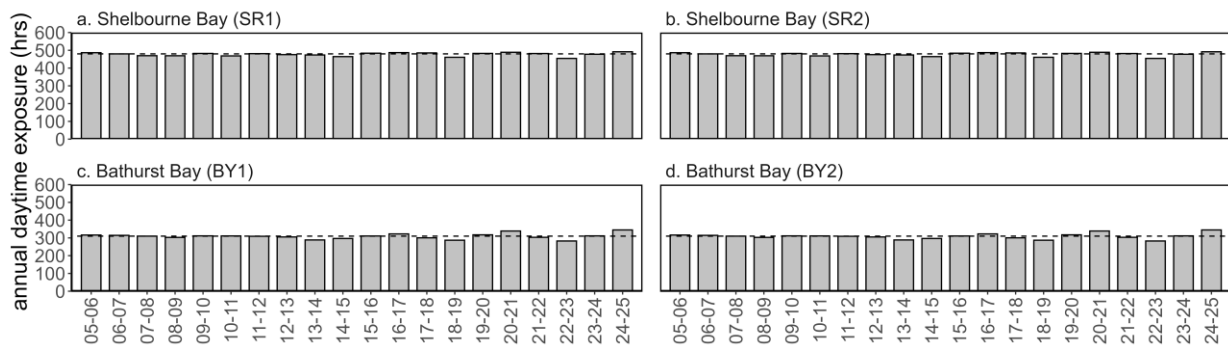


Figure 91. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of coastal intertidal seagrass meadows in Cape York NRM region; 1999–2025. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 21. Observed and predicted tidal heights courtesy Maritime Safety Queensland and Bureau of Meteorology, 2025.

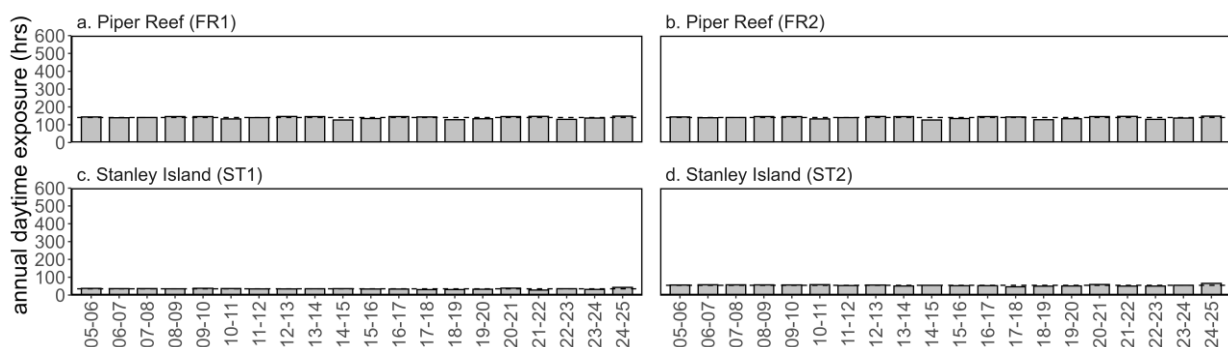


Figure 92. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of reef intertidal seagrass meadows in Cape York NRM region; 1999–2025. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 21. Observed and predicted tidal heights courtesy Maritime Safety Queensland and Bureau of Meteorology, 2025.

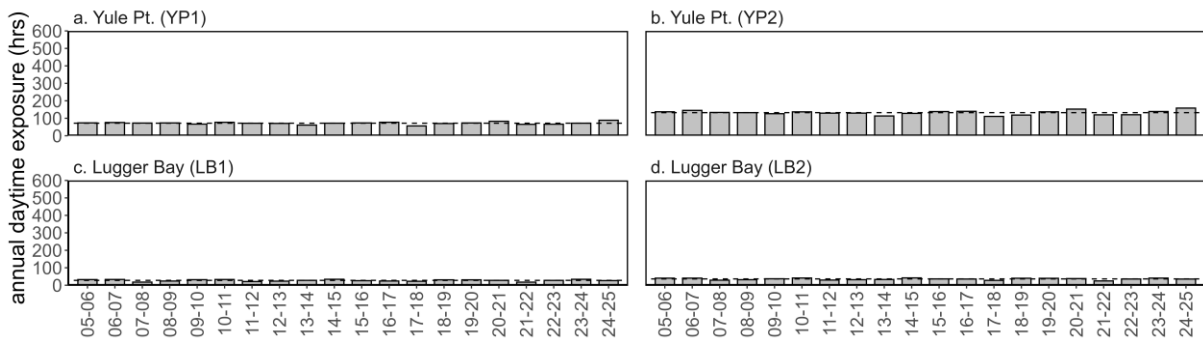


Figure 93. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of coastal intertidal seagrass meadows in Wet Tropics NRM region; 1999–2025. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 21. Observed and predicted tidal heights courtesy Maritime Safety Queensland and Bureau of Meteorology, 2025.

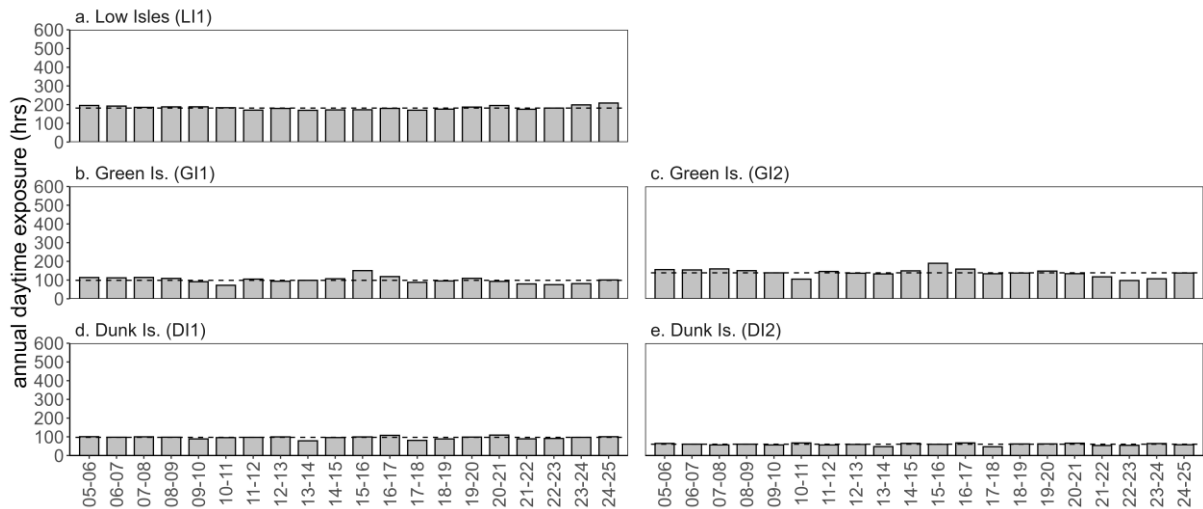


Figure 94. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of reef intertidal seagrass meadows in the Wet Tropics NRM region; 1999–2025. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 21. Observed and predicted tidal heights courtesy Maritime Safety Queensland and Bureau of Meteorology, 2025.

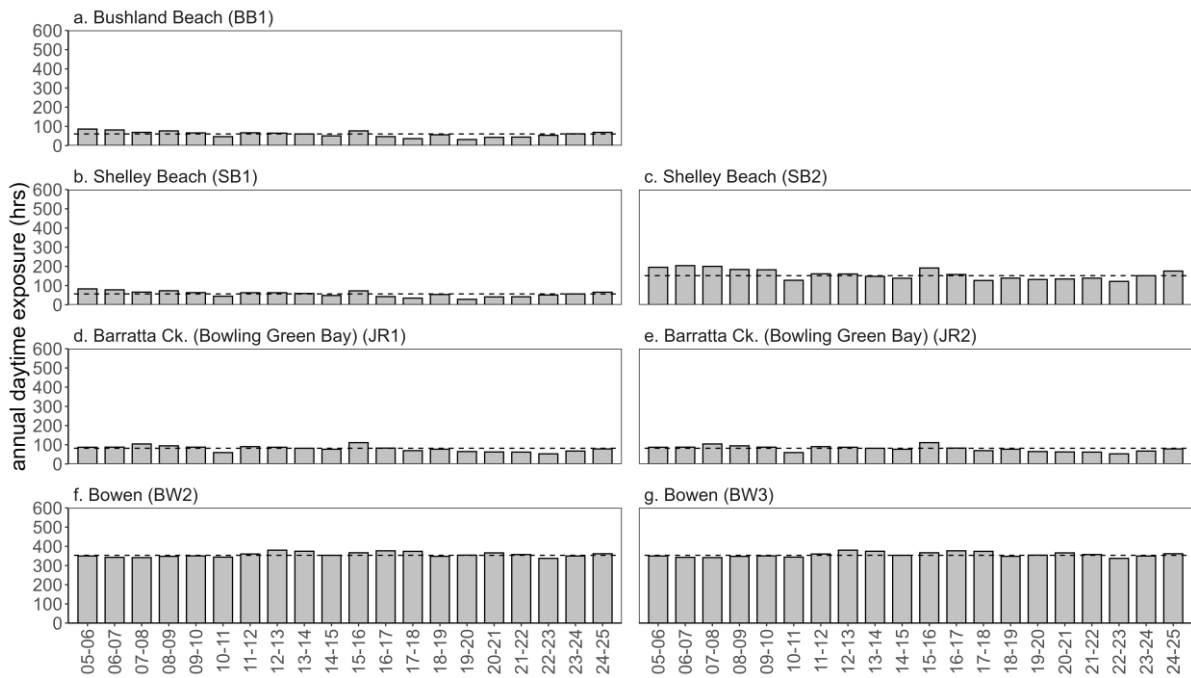


Figure 95. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of coastal intertidal seagrass meadows in Burdekin NRM region; 2000–2025. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 21. Observed and predicted tidal heights courtesy Maritime Safety Queensland and Bureau of Meteorology, 2025.

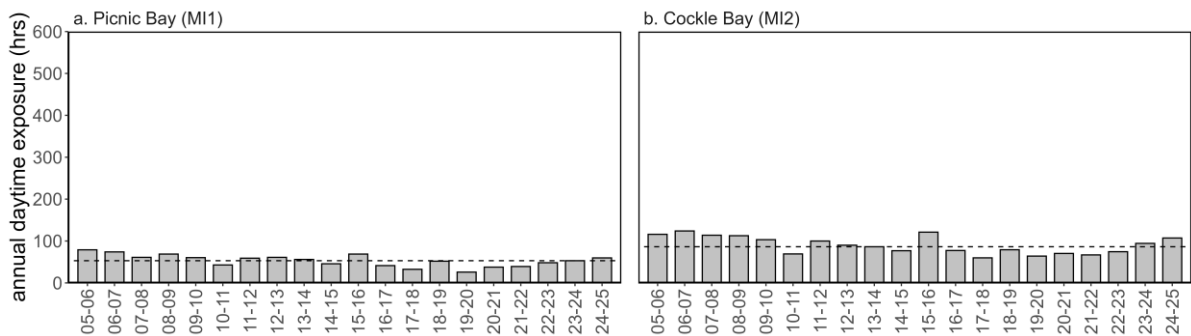


Figure 96. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of reef intertidal seagrass meadows in Burdekin NRM region; 2000–2025. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 21. Observed and predicted tidal heights courtesy Maritime Safety Queensland and Bureau of Meteorology, 2025.

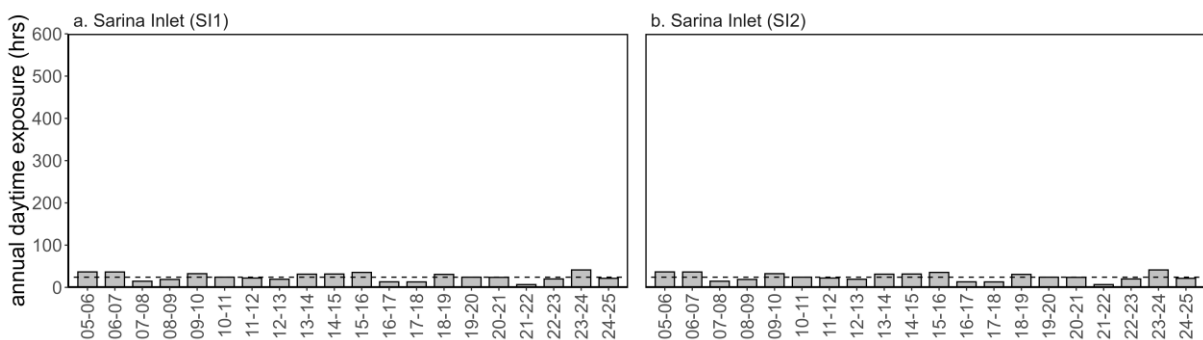


Figure 97. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of estuarine intertidal seagrass meadows in Mackay–Whitsunday NRM region; 1999–2025. Year is June–May. For tidal exposure (when intertidal banks

become exposed at a low tide) height at each site, see Table 21. Observed and predicted tidal heights courtesy Maritime Safety Queensland and Bureau of Meteorology, 2025.

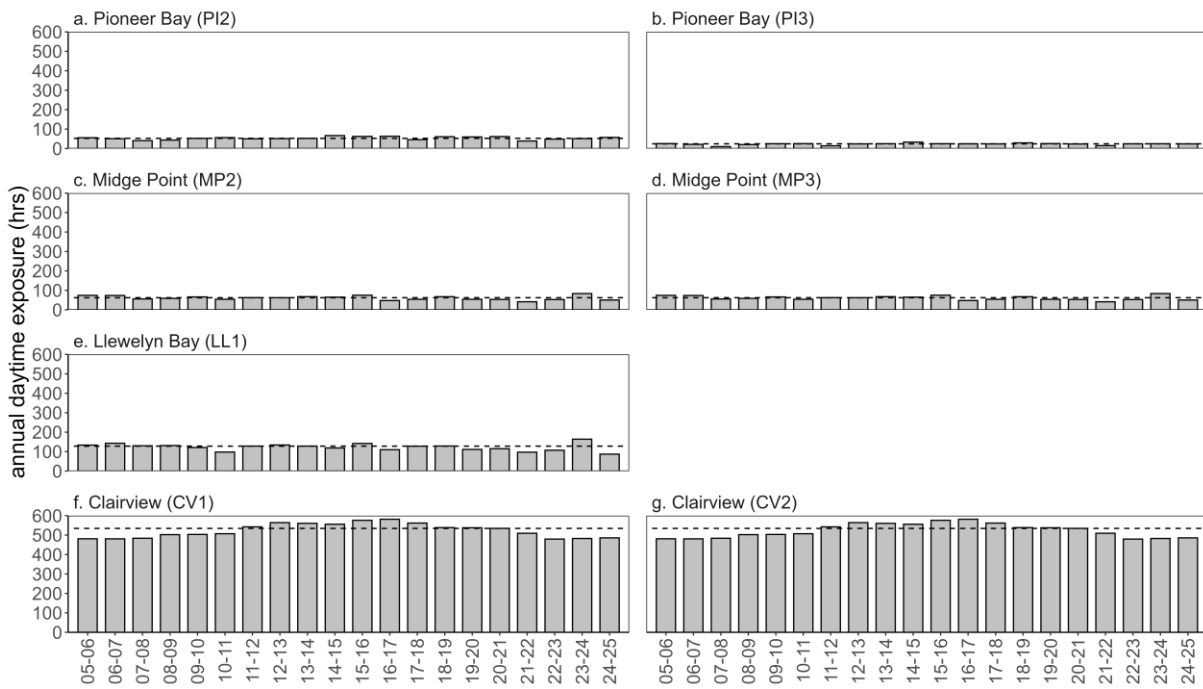


Figure 98. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of coastal intertidal seagrass meadows in Mackay–Whitsunday NRM region; 1999–2025. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 21. Observed and predicted tidal heights courtesy Maritime Safety Queensland and Bureau of Meteorology, 2025.

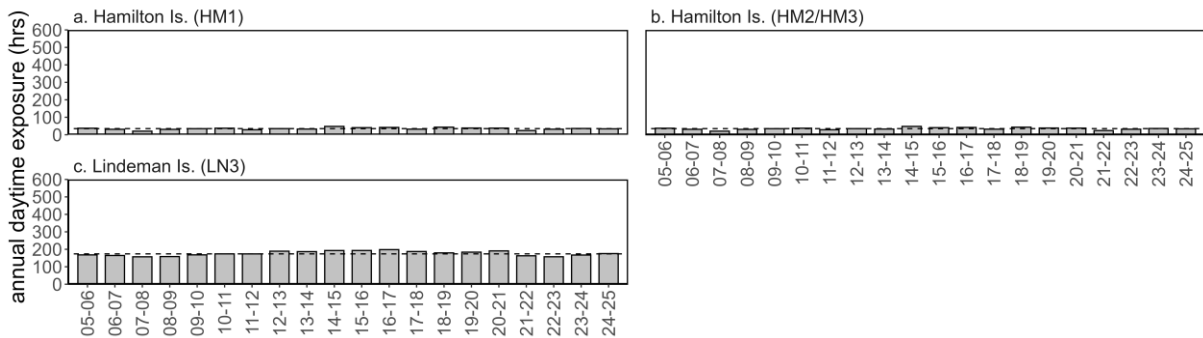


Figure 99. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of reef intertidal seagrass meadows in Mackay–Whitsunday NRM region; 1999–2024. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 21. Observed and predicted tidal heights courtesy Maritime Safety Queensland and Bureau of Meteorology, 2025.

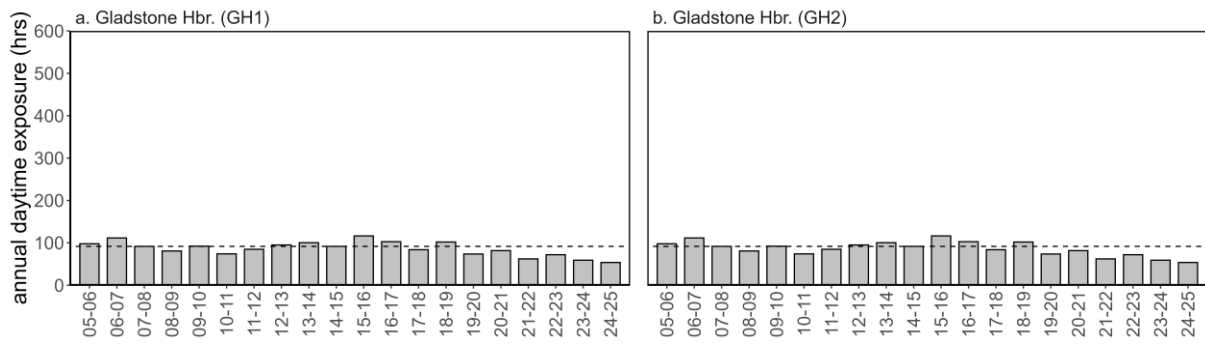


Figure 100. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of estuarine intertidal seagrass meadows in the Fitzroy NRM region; 1999–2025. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 21. Observed and predicted tidal heights courtesy Maritime Safety Queensland and Bureau of Meteorology, 2025.

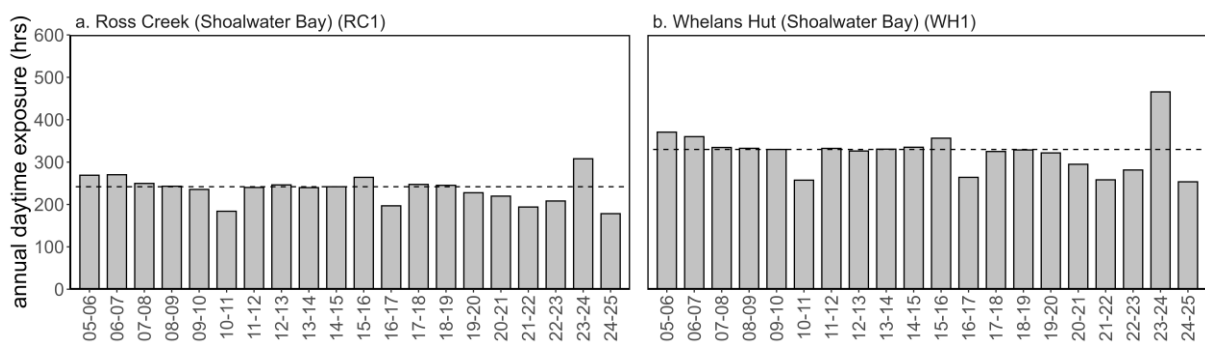


Figure 101. Annual daytime tidal exposure (total hours) and long-term median (dashed line) coastal intertidal seagrass meadows in the Fitzroy NRM region; 1999–2025. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 21. Observed and predicted tidal heights courtesy Maritime Safety Queensland and Bureau of Meteorology, 2025.

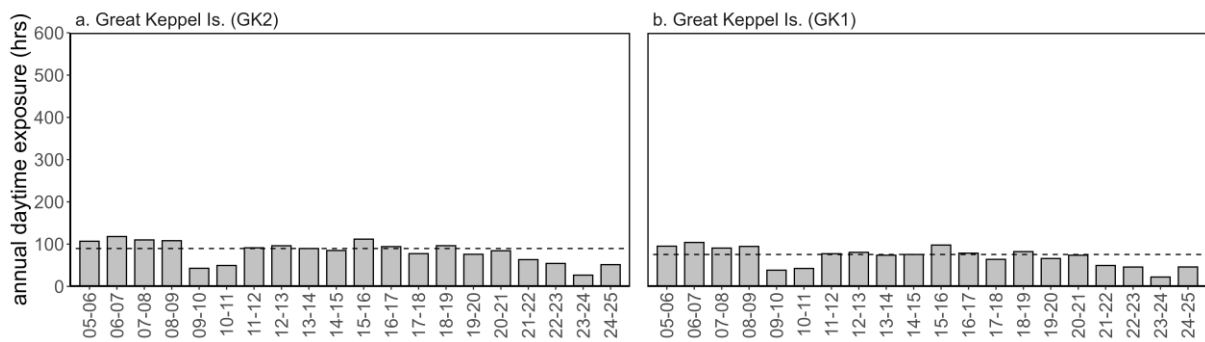


Figure 102. Annual daytime tidal exposure (total hours) and long-term median (dashed line) reef intertidal seagrass meadows in the Fitzroy NRM region; 1999–2025. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 21. Observed and predicted tidal heights courtesy Maritime Safety Queensland and Bureau of Meteorology, 2025.

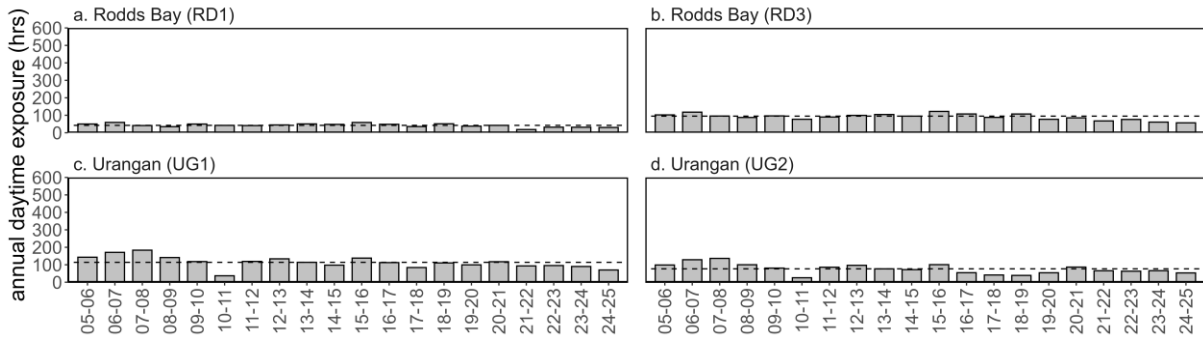


Figure 103. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of estuarine intertidal seagrass meadows in the Burnett–Mary NRM region; 1999–2024. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 21. Observed and predicted tidal heights courtesy Maritime Safety Queensland and Bureau of Meteorology, 2025.

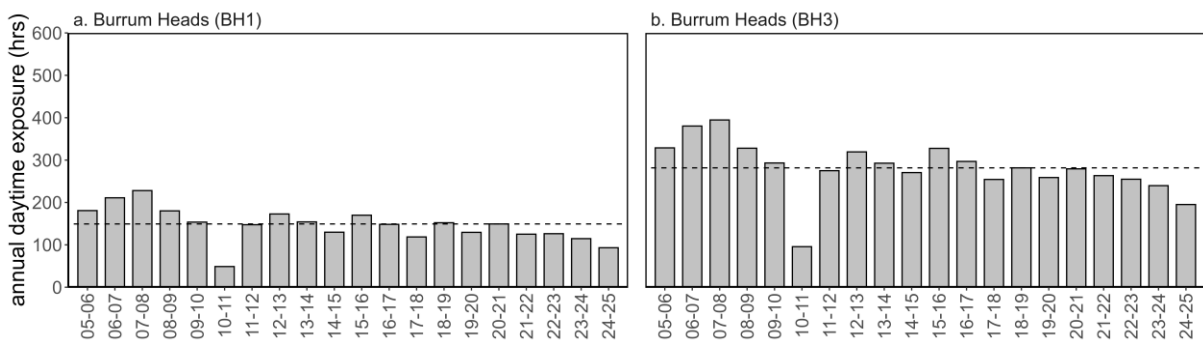


Figure 104. Annual daytime tidal exposure (total hours) and long-term median (dashed line) of coastal intertidal seagrass meadows in the Burnett–Mary NRM region; 1999–2024. Year is June–May. For tidal exposure (when intertidal banks become exposed at a low tide) height at each site, see Table 21. Observed and predicted tidal heights courtesy Maritime Safety Queensland and Bureau of Meteorology, 2025.

### A2.1.2 Light at seagrass canopy

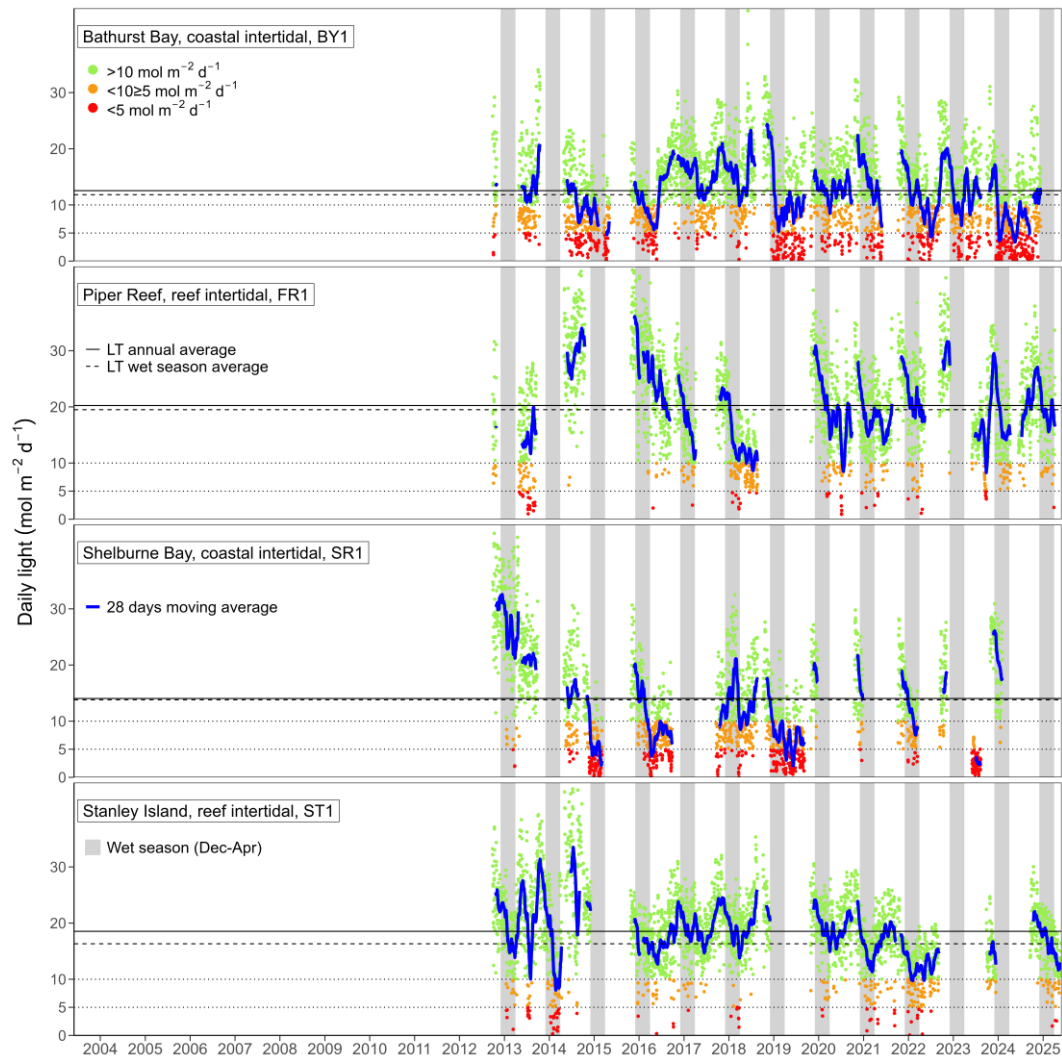


Figure 105. Daily light (coloured points), 28-day rolling average (blue, bold line) and long-term average (annual and wet season, solid and dashed lines) at monitoring locations in the Cape York NRM region.

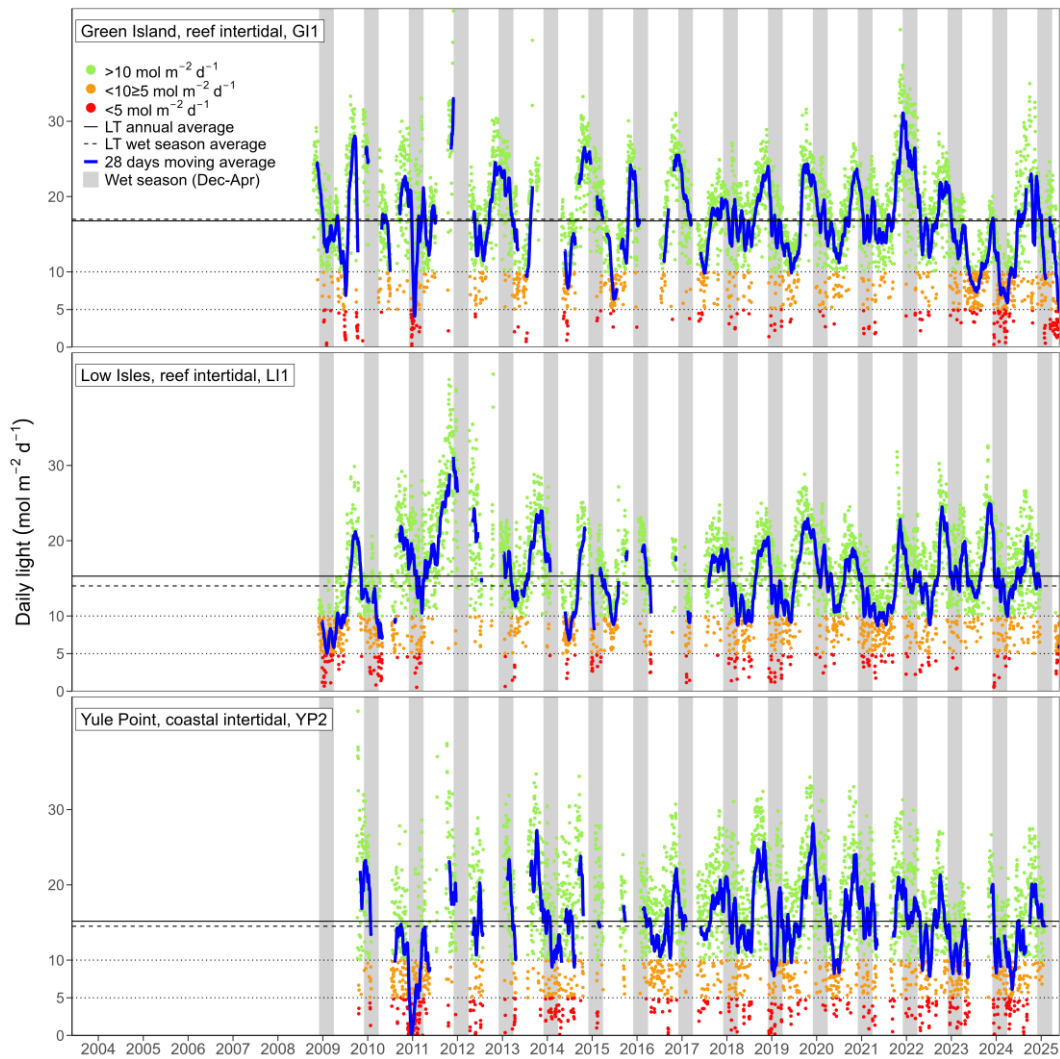


Figure 106. Daily light (coloured points), 28-day rolling average (blue, bold line) and long-term average (annual and wet season, solid and dashed lines) at monitoring locations in northern Wet Tropics region.

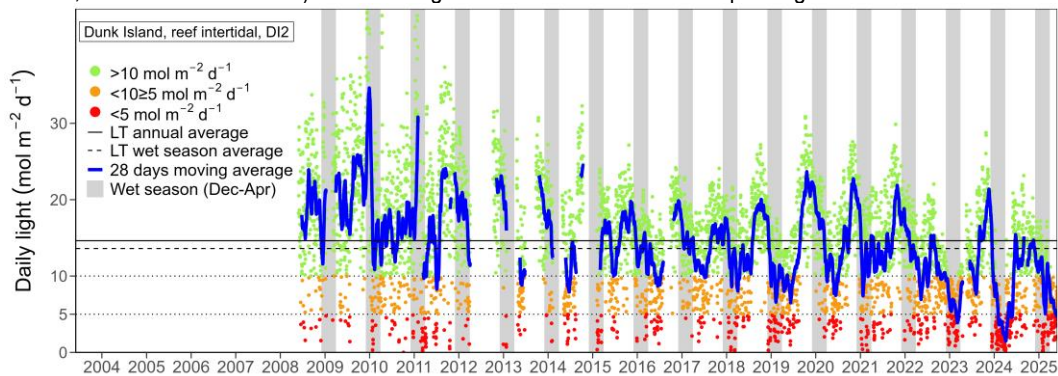


Figure 107. Daily light (coloured points), 28-day rolling average (blue, bold line) and long-term average (annual and wet season, solid and dashed lines) at monitoring locations in the southern Wet Tropics region.

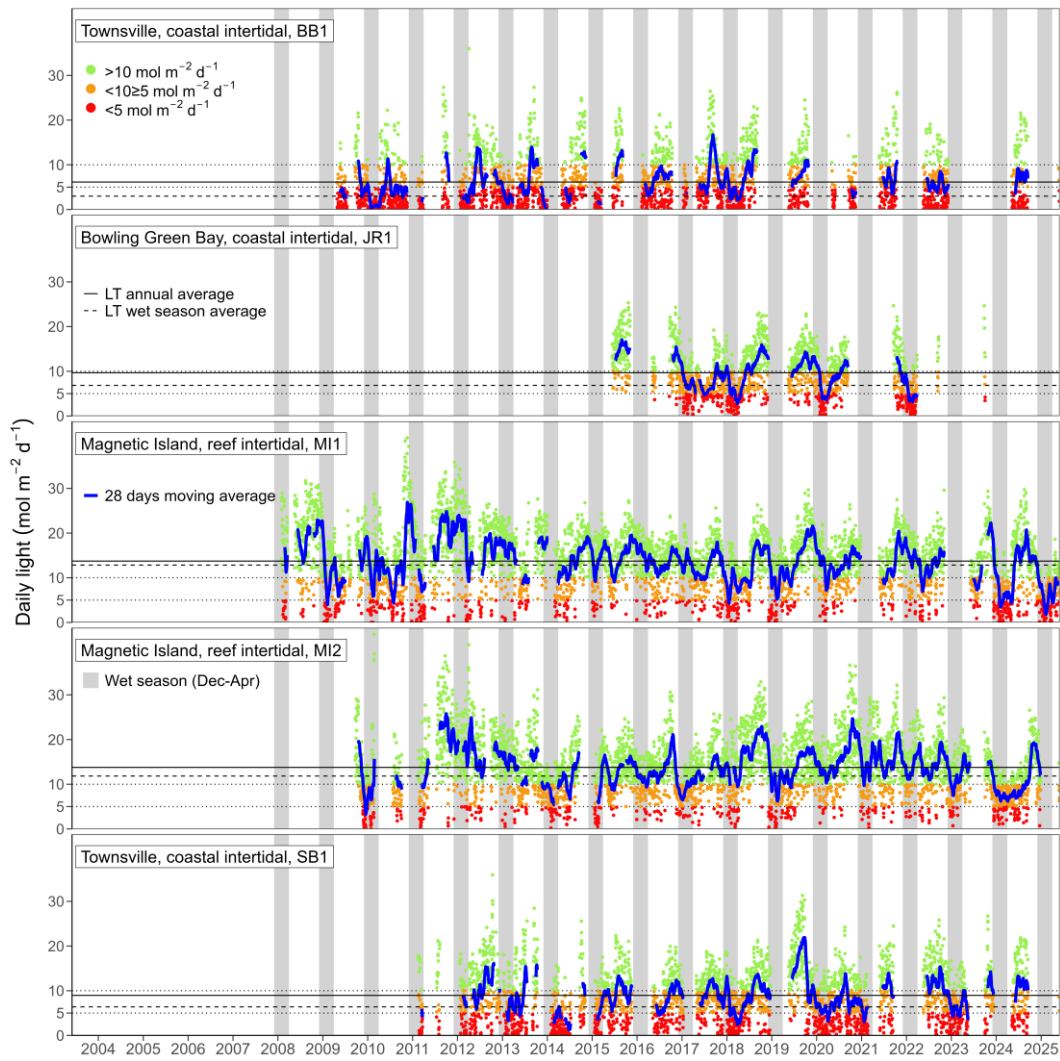


Figure 108. Daily light (coloured points), 28-day rolling average (blue, bold line) and long-term average (annual and wet season, solid and dashed lines) at monitoring locations in the Burdekin region.

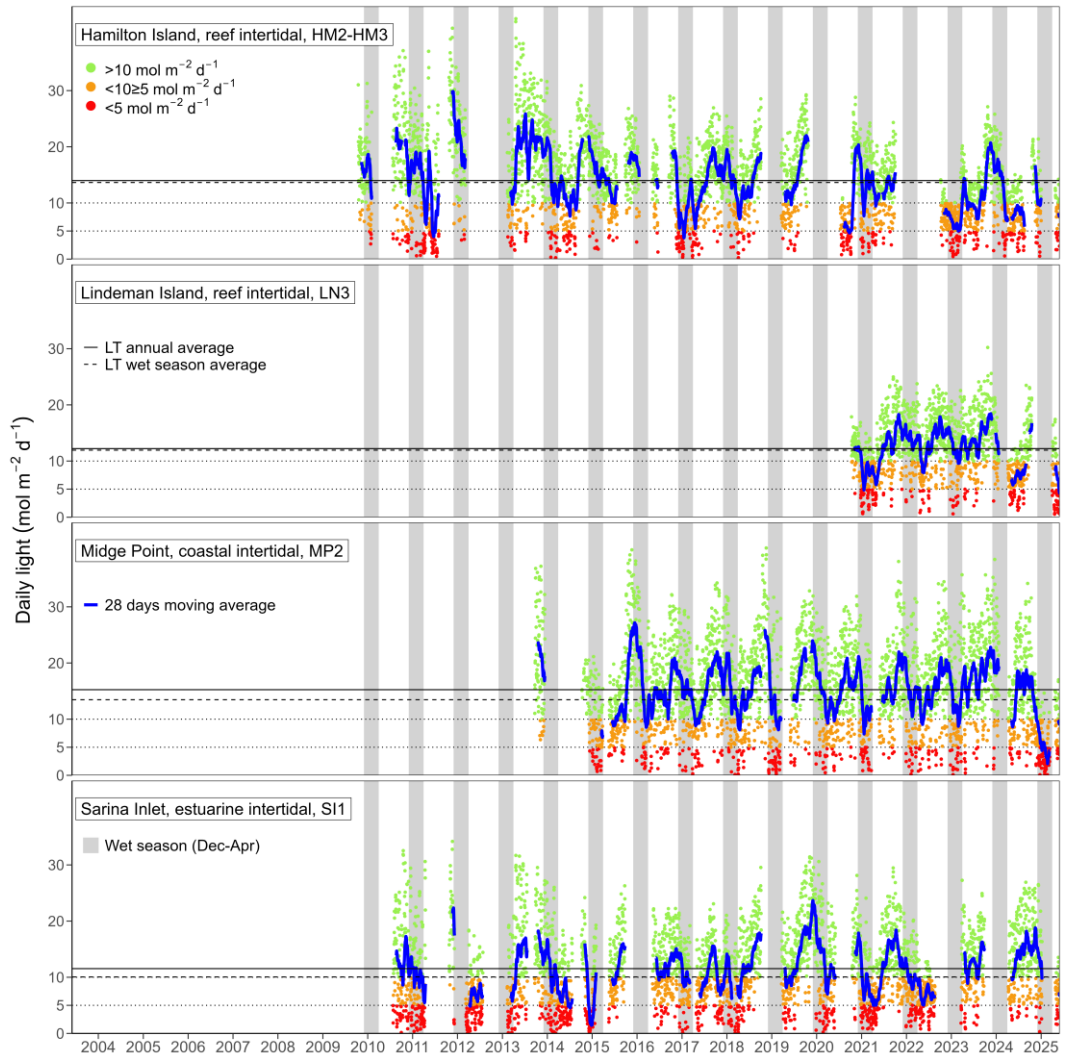


Figure 109. Daily light (coloured points), 28-day rolling average (blue, bold line) and long-term average (annual and wet season, solid and dashed lines) at monitoring locations in the Mackay–Whitsunday NRM region.

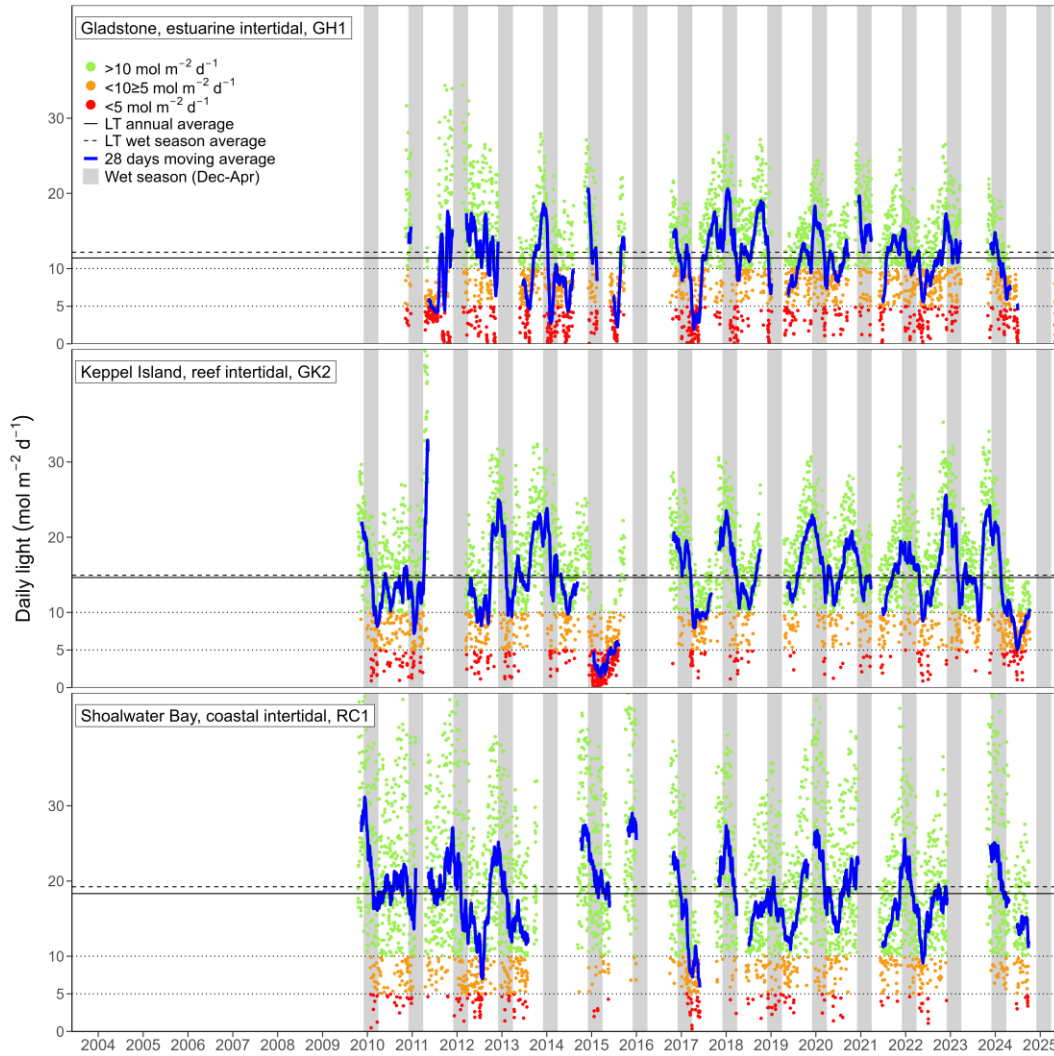


Figure 110. Daily light (coloured points), 28-day rolling average (blue, bold line) and long-term average (annual and wet season, solid and dashed lines) at monitoring locations in the Fitzroy NRM region.

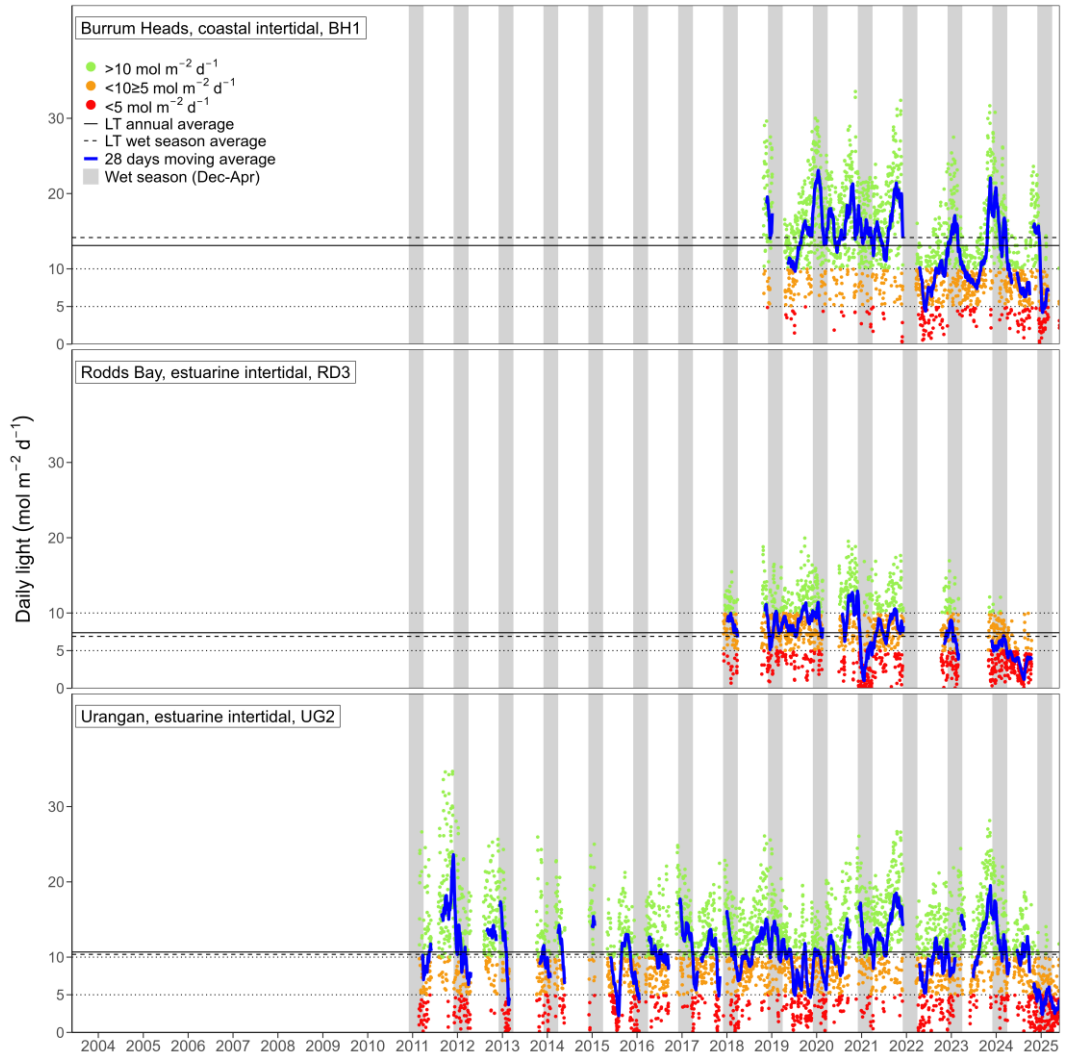


Figure 111. Daily light (coloured points), 28-day rolling average (blue, bold line) and long-term average (annual and wet season, solid and dashed lines) at monitoring locations in the Burnett–Mary NRM region.

## A2.2 Seagrass habitat condition: Sediments composition

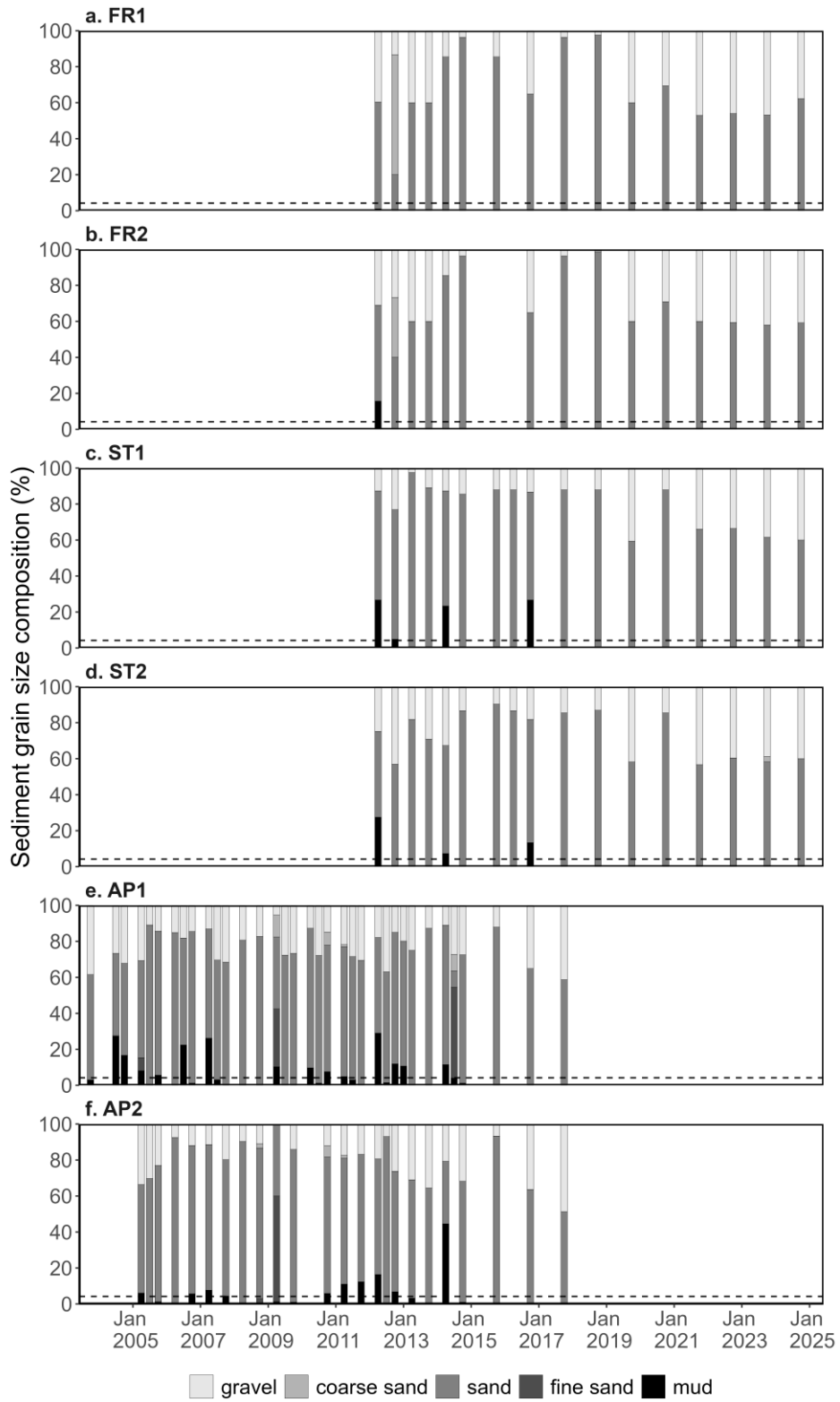


Figure 112. Sediment grain size composition at intertidal reef habitat monitoring sites in the Cape York region, 2003–2025. Dashed line is the Reef long-term average proportion of mud.

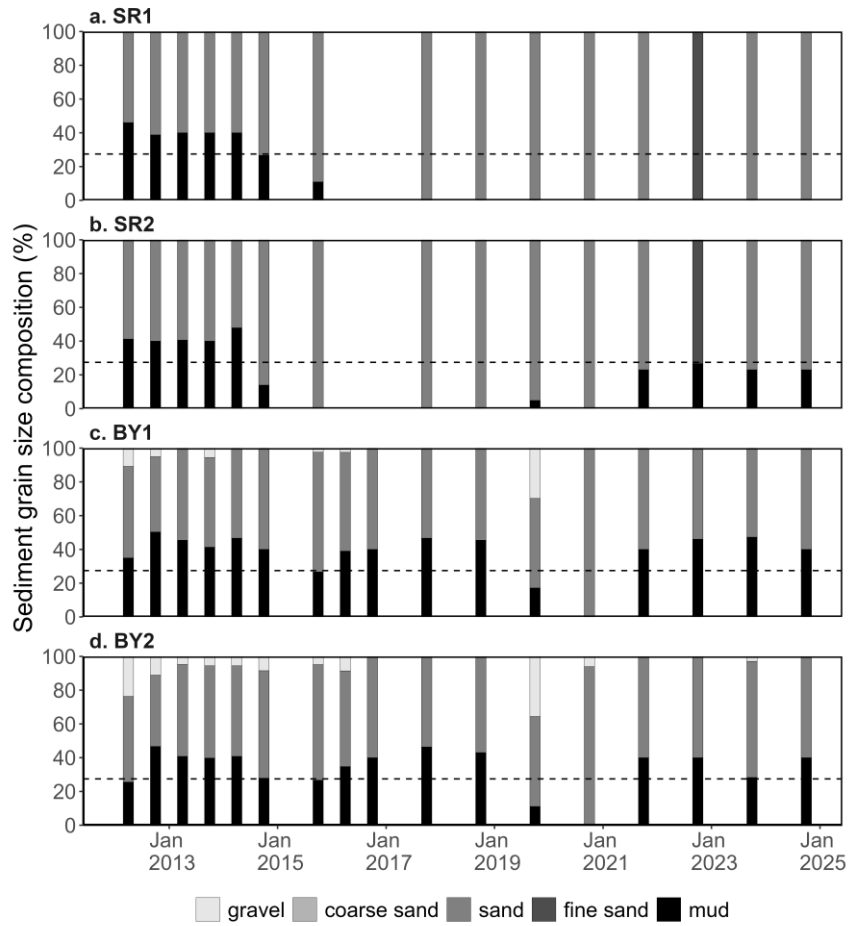


Figure 113. Sediment grain size composition at intertidal coastal habitat monitoring sites in the Cape York region, 2012–2024. Dashed line is the Reef long-term average proportion of mud.

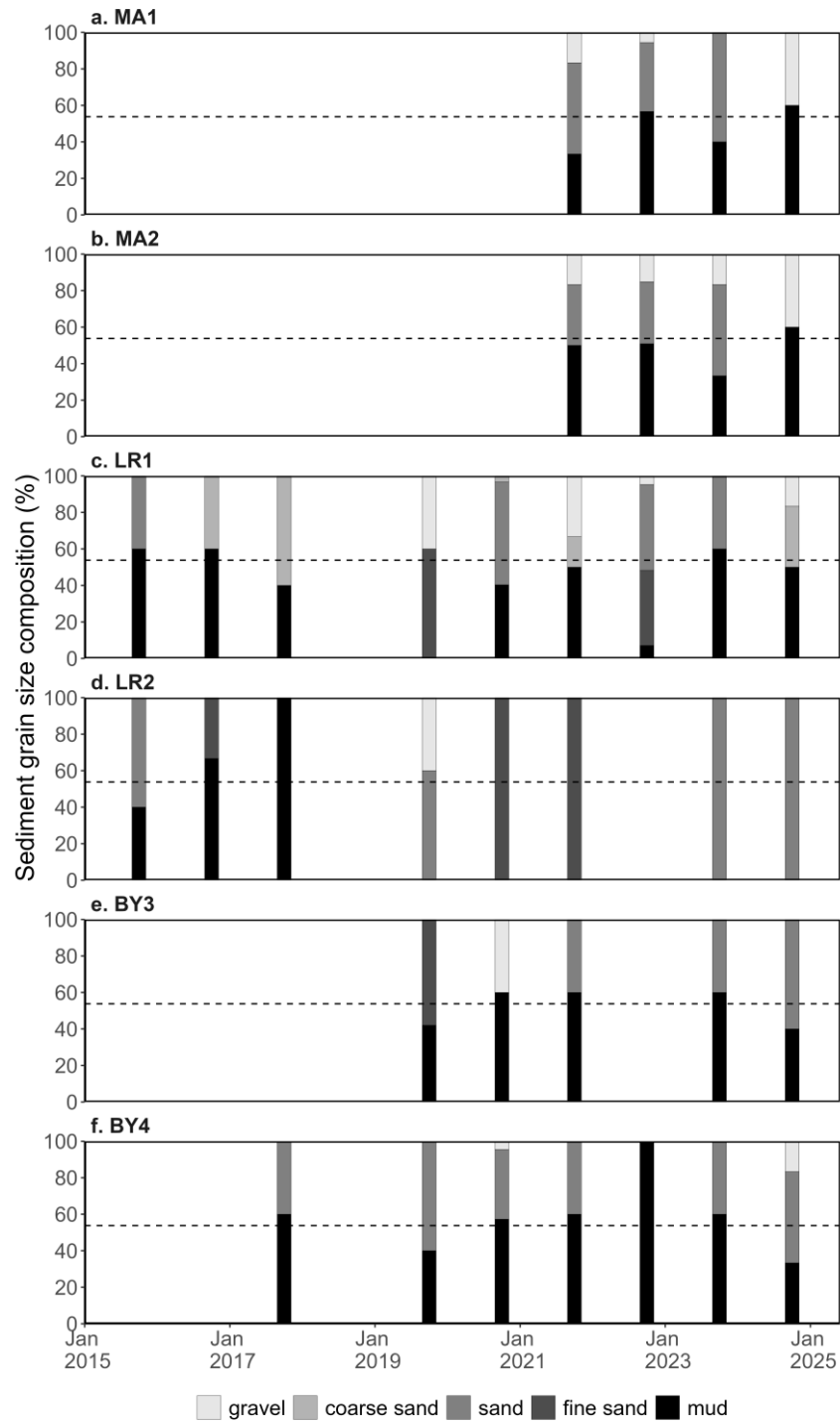


Figure 114. Sediment grain size composition at subtidal coastal habitat monitoring sites in the Cape York region, 2015–2025. Dashed line is the Reef long-term average proportion of mud.

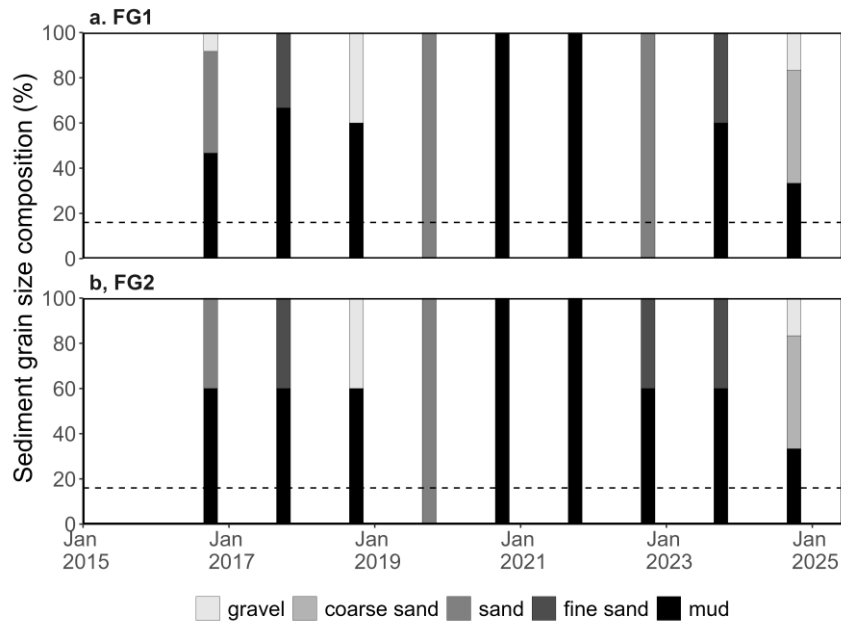


Figure 115. Sediment grain size composition at subtidal reef habitat monitoring sites in the Cape York region, 2016–2025. Dashed line is the Reef long-term average proportion of mud.

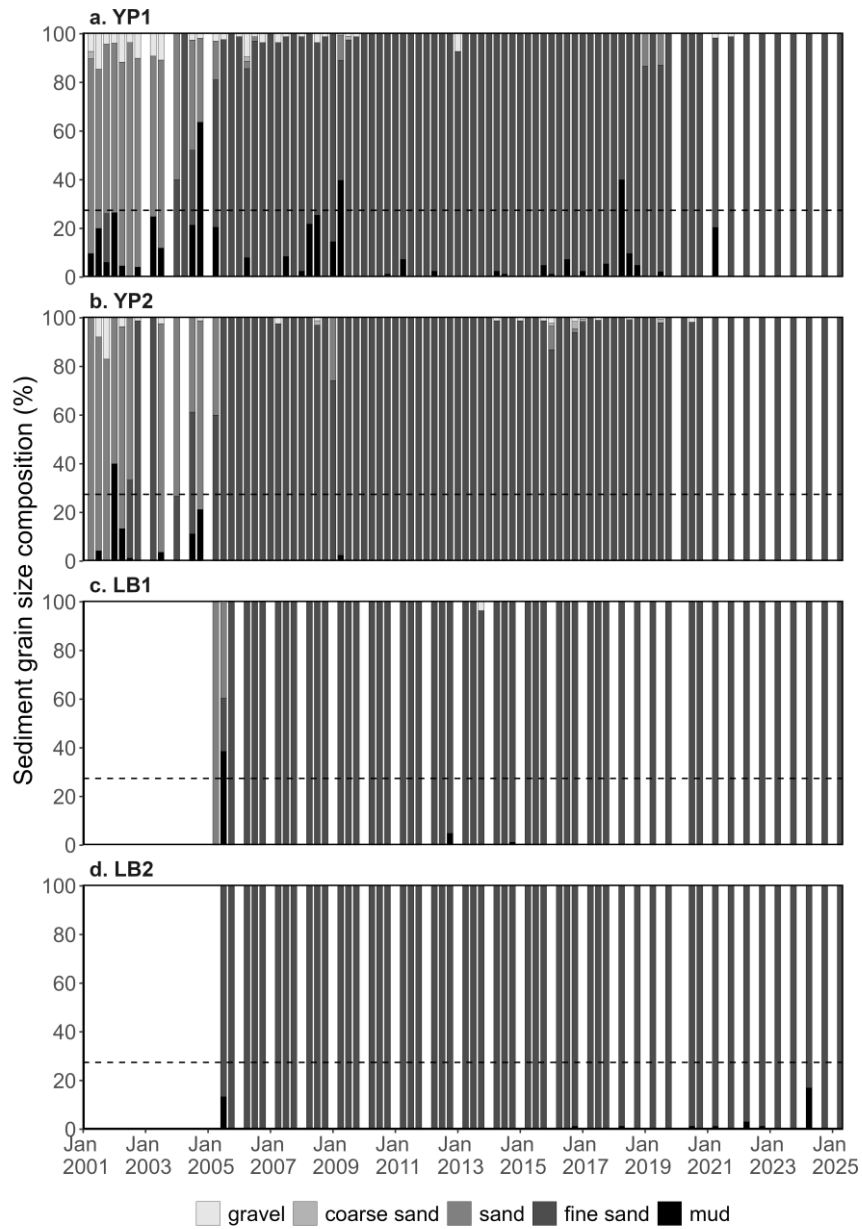


Figure 116. Sediment grain size composition at intertidal coastal habitat monitoring sites in the Wet Tropics region, 2001–2025. Dashed line is the Reef long-term average proportion of mud.

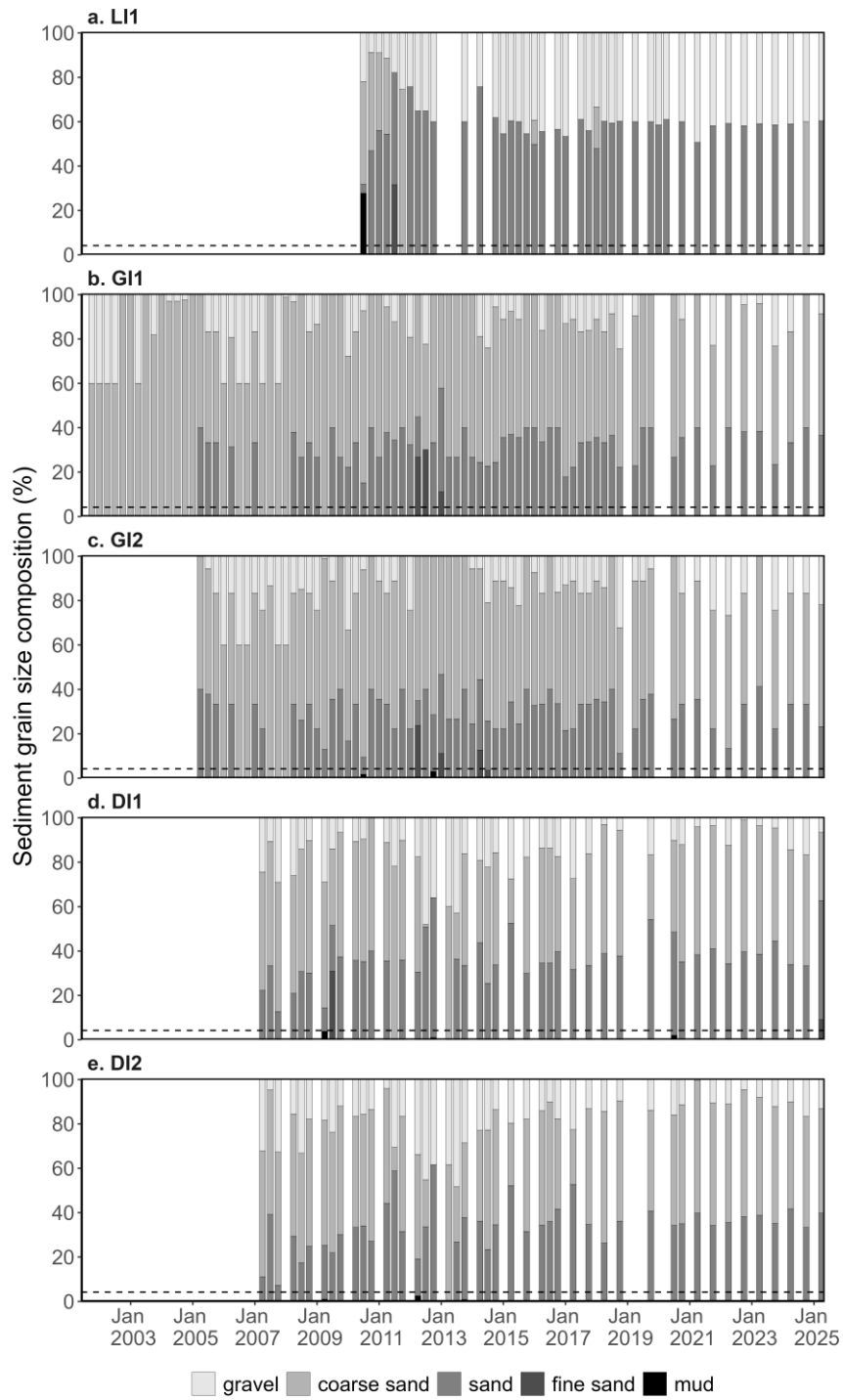


Figure 117. Sediment grain size composition at intertidal reef habitat monitoring sites in the Wet Tropics region, 2001–2025. Dashed line is the Reef long-term average proportion of mud.

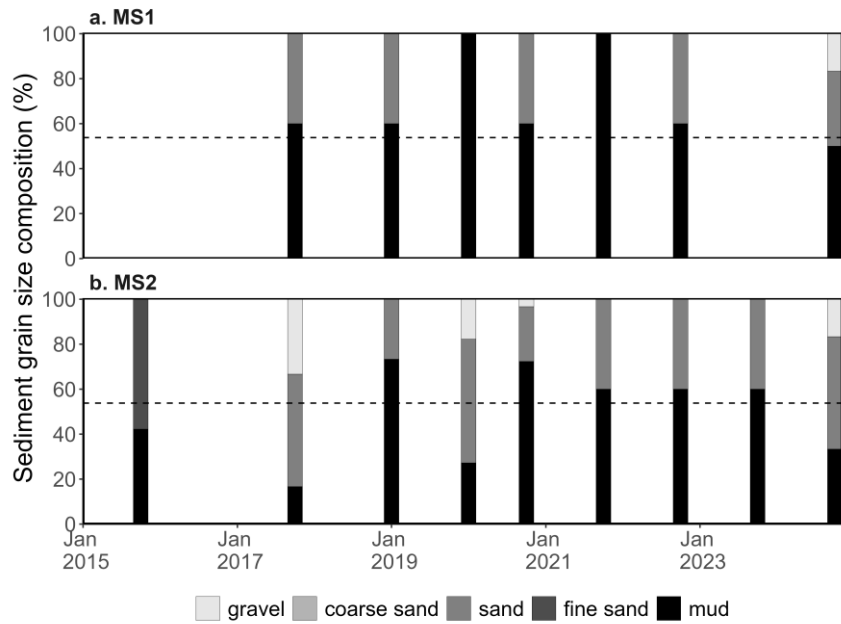


Figure 118. Sediment grain size composition at subtidal coastal habitat monitoring sites in the Wet Tropics region, 2015–2025. Dashed line is the Reef long-term average proportion of mud.

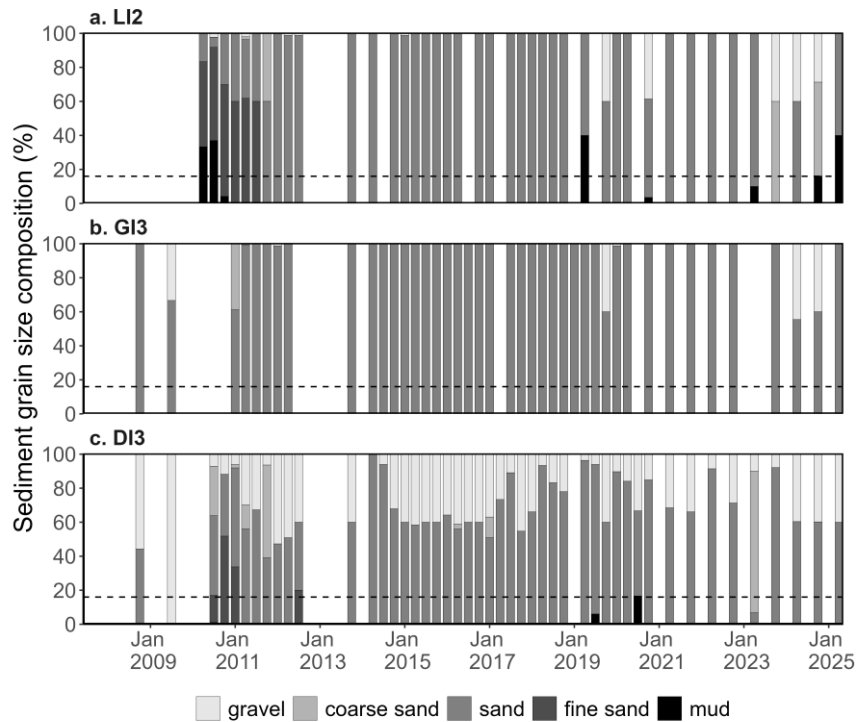


Figure 119. Sediment grain size composition at subtidal reef habitat monitoring sites in the Wet Tropics region, 2008–2024. Dashed line is the Reef long-term average proportion of mud.

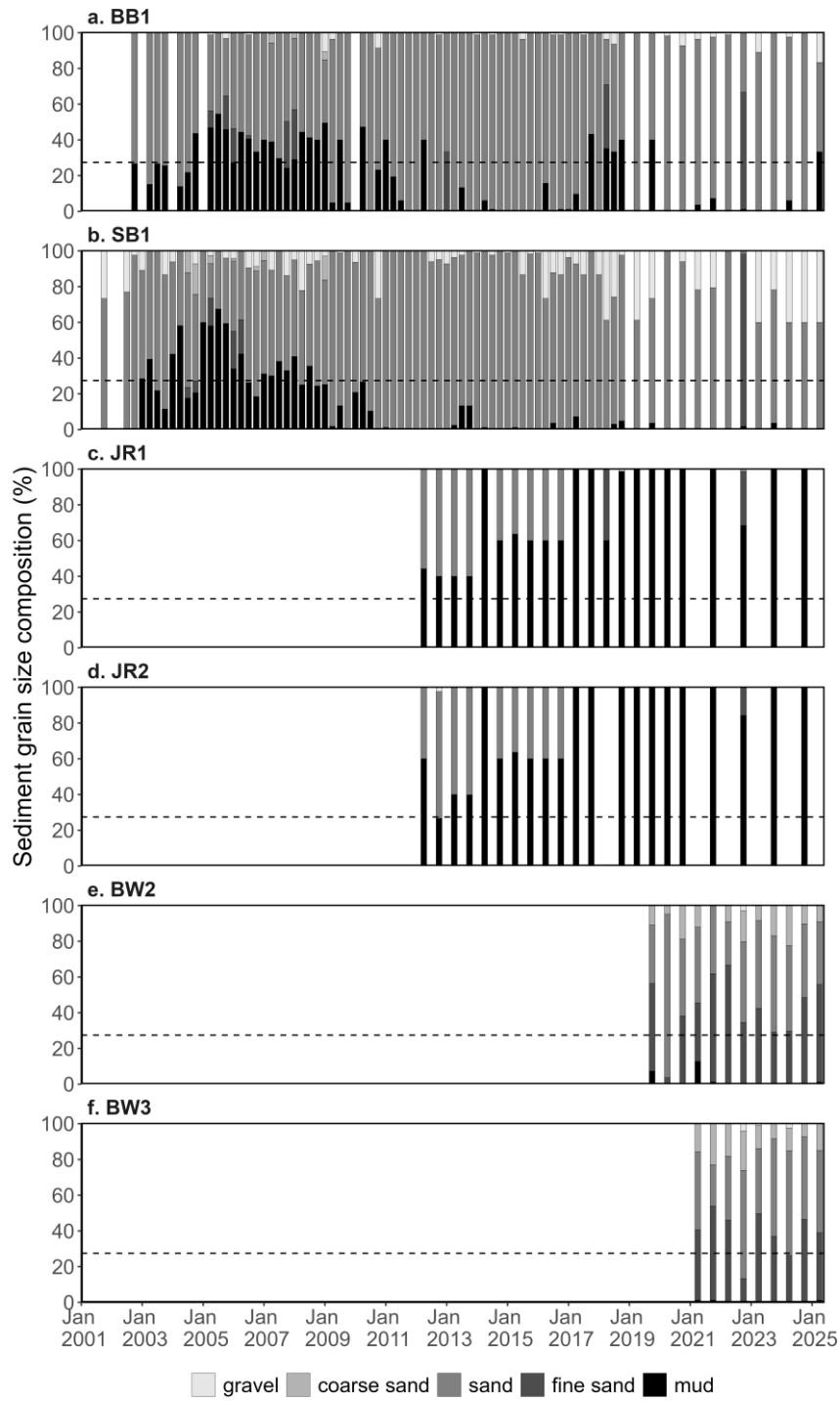


Figure 120. Sediment grain size composition at coastal intertidal habitat monitoring sites in the Burdekin region, 2001–2025. Dashed line is the Reef long-term average proportion of mud.

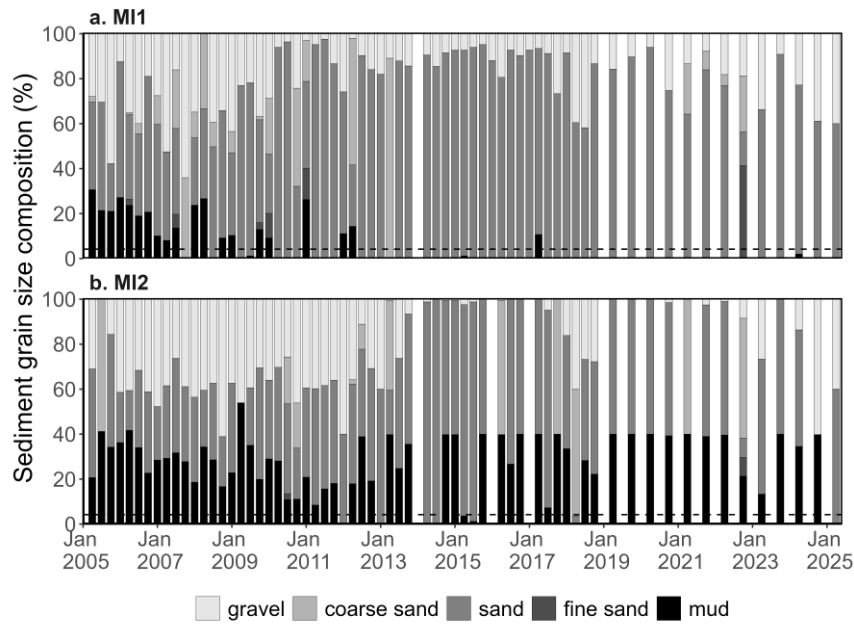


Figure 121. Sediment grain size composition at intertidal reef habitat monitoring sites in the Burdekin region, 2004–2025. Dashed line is the Reef long-term average proportion of mud.

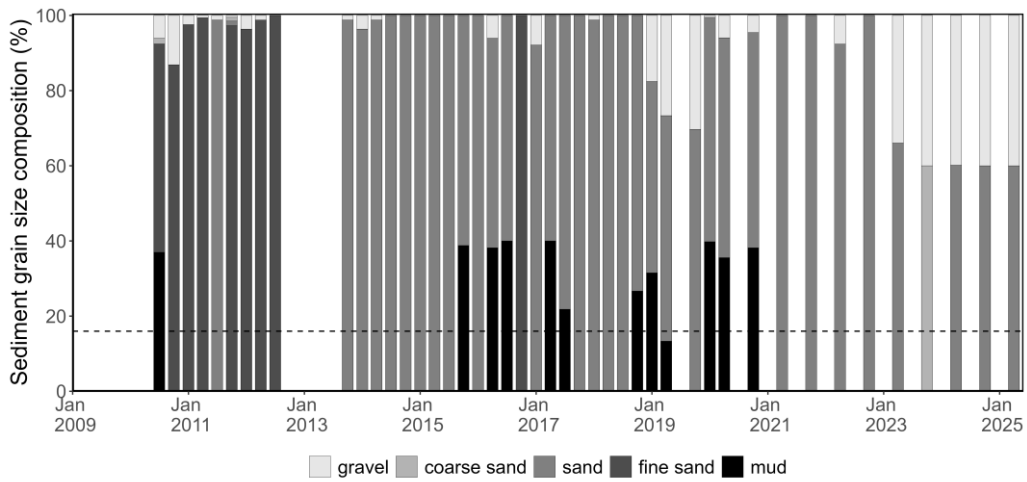


Figure 122. Sediment grain size composition at subtidal reef habitat monitoring site (MI3) in the Burdekin region, 2010–2025. Dashed line is the Reef long-term average proportion of mud.

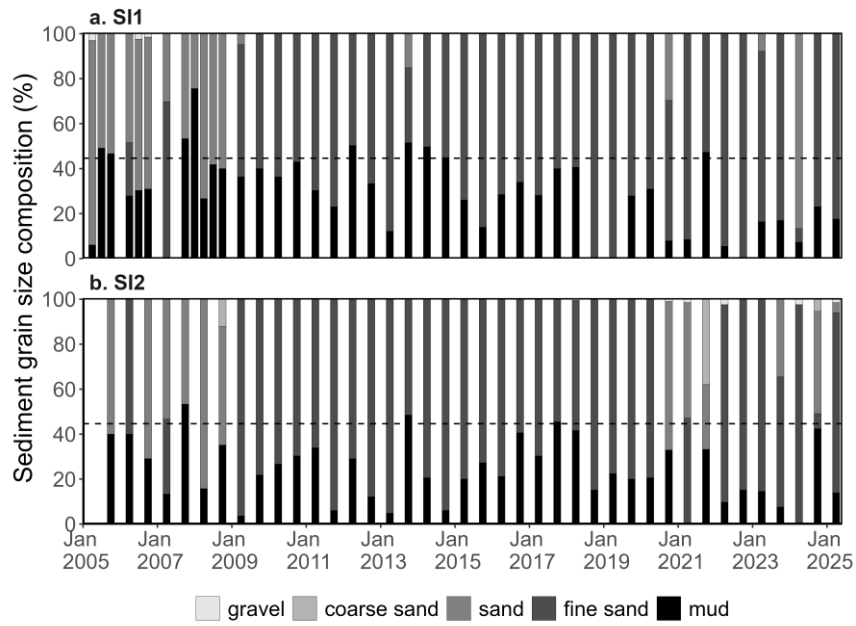


Figure 123. Sediment grain size composition at intertidal estuarine habitat monitoring sites in the Mackay–Whitsunday region, 2005–2025. Dashed line is the Reef long-term average proportion of mud.

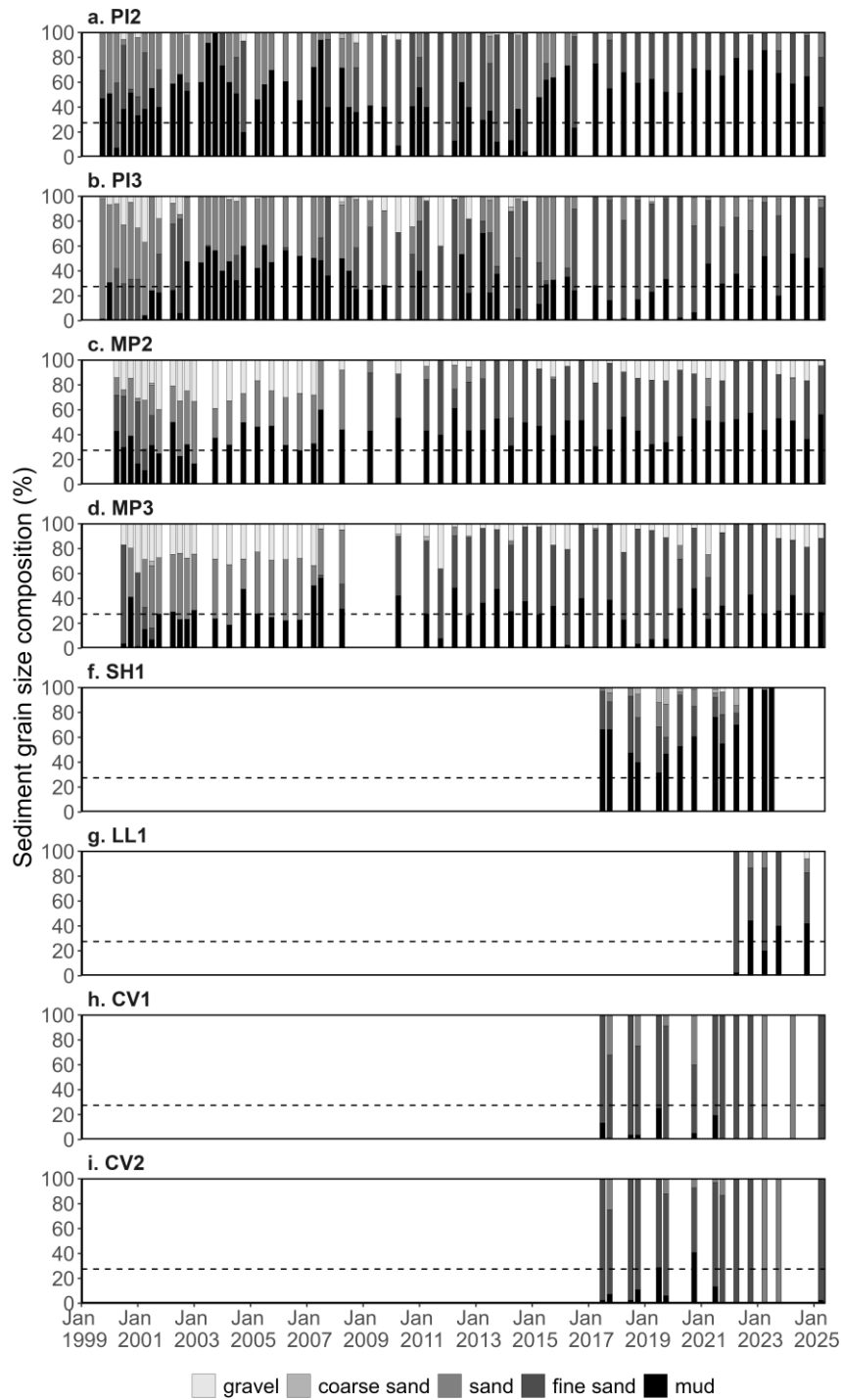


Figure 124. Sediment grain size composition at coastal intertidal habitat monitoring sites in the Mackay–Whitsunday region, 1999–2025. Dashed line is the Reef long-term average proportion of mud.

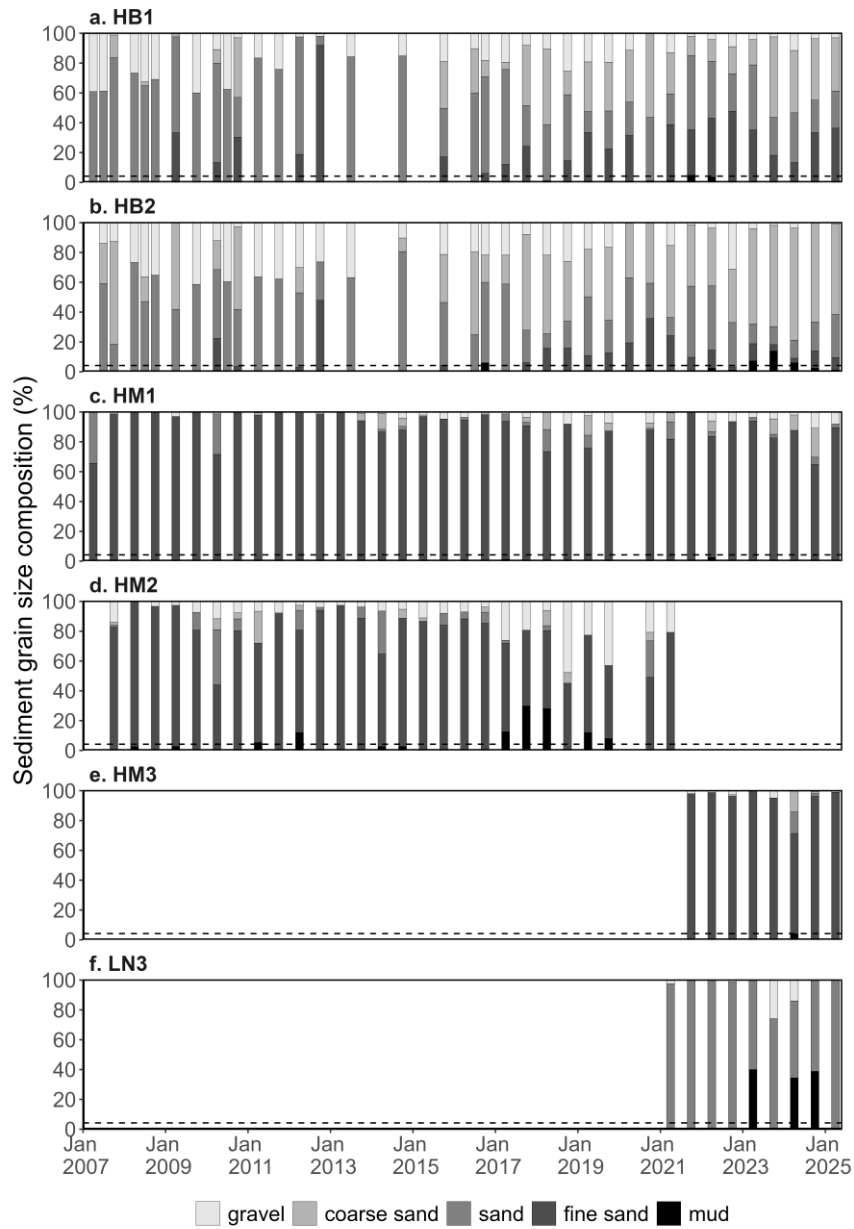


Figure 125. Sediment grain size composition at reef intertidal habitat monitoring sites in the Mackay–Whitsunday region, 2007–2025. Dashed line is the Reef long-term average proportion of mud.

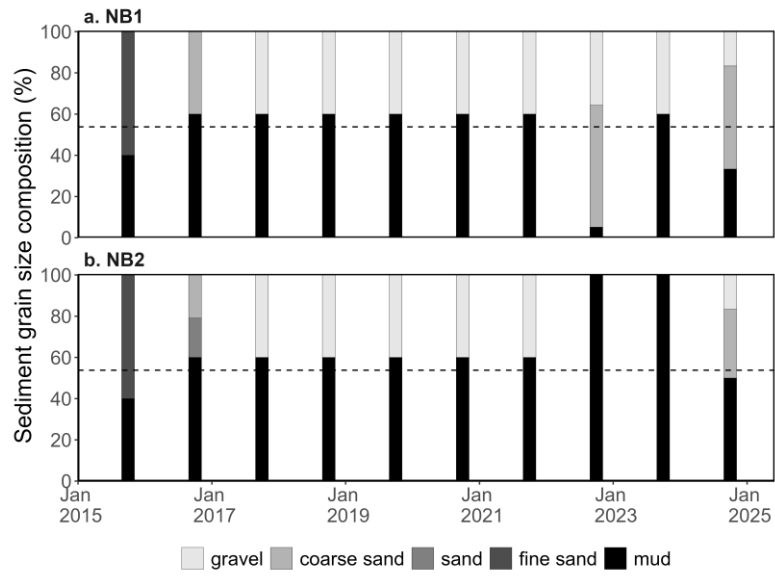


Figure 126. Sediment grain size composition at subtidal coastal habitat monitoring sites in the Mackay–Whitsunday region, 2015–2025. Dashed line is the Reef long-term average proportion of mud.

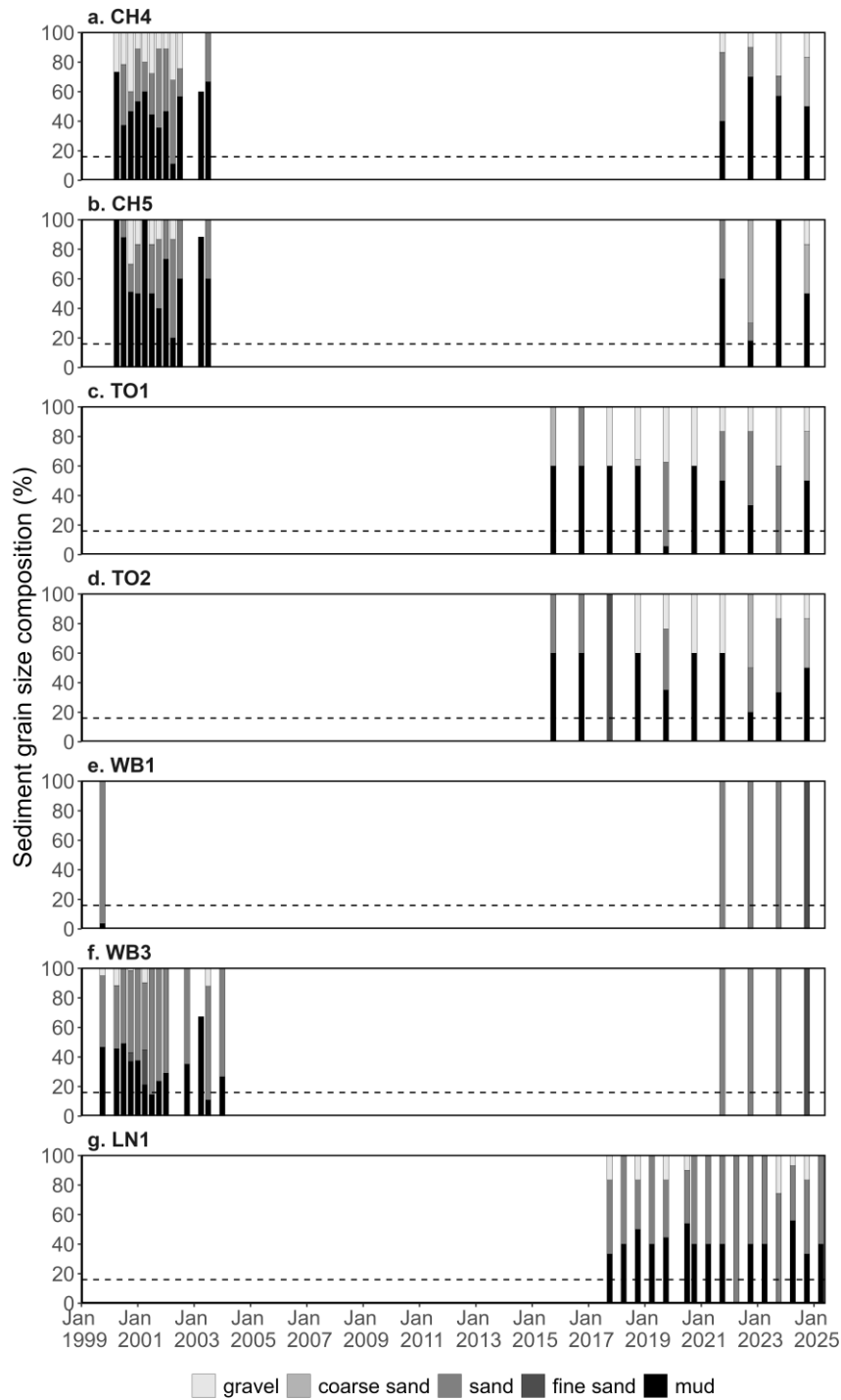


Figure 127. Sediment grain size composition at reef subtidal habitat monitoring sites in the Mackay–Whitsunday region, 2000–2025. Dashed line is the Reef long-term average proportion of mud.

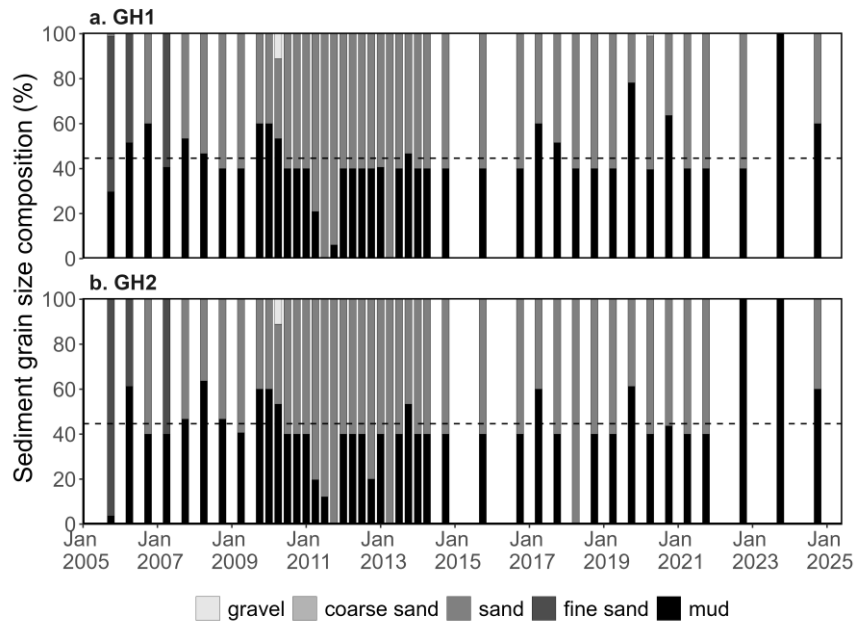


Figure 128. Sediment grain size composition at estuarine intertidal habitat monitoring sites in the Fitzroy region, 2005–2025. Dashed line is the Reef long-term average proportion of mud.

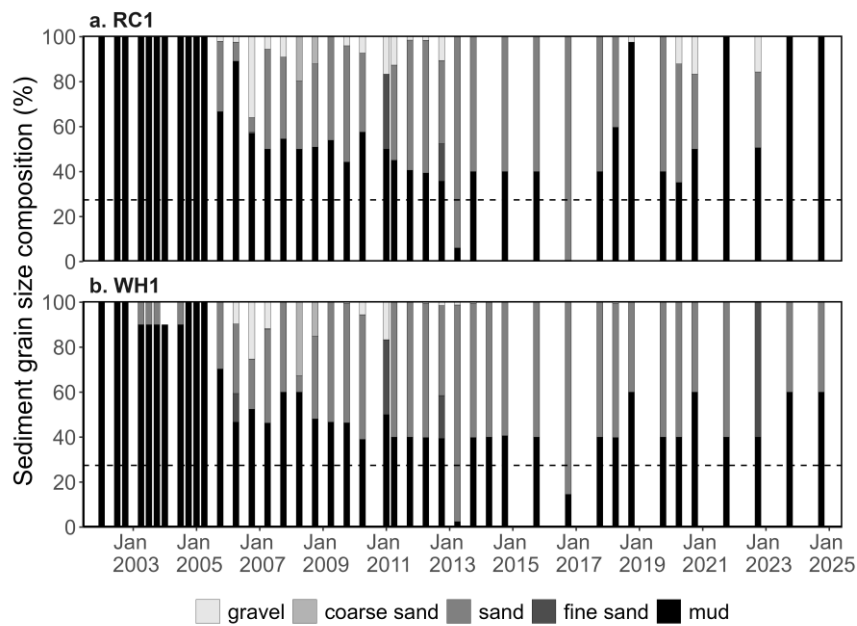


Figure 129. Sediment grain size composition at coastal intertidal habitat monitoring sites in the Fitzroy region, 2002–2025. Dashed line is the Reef long-term average proportion of mud.

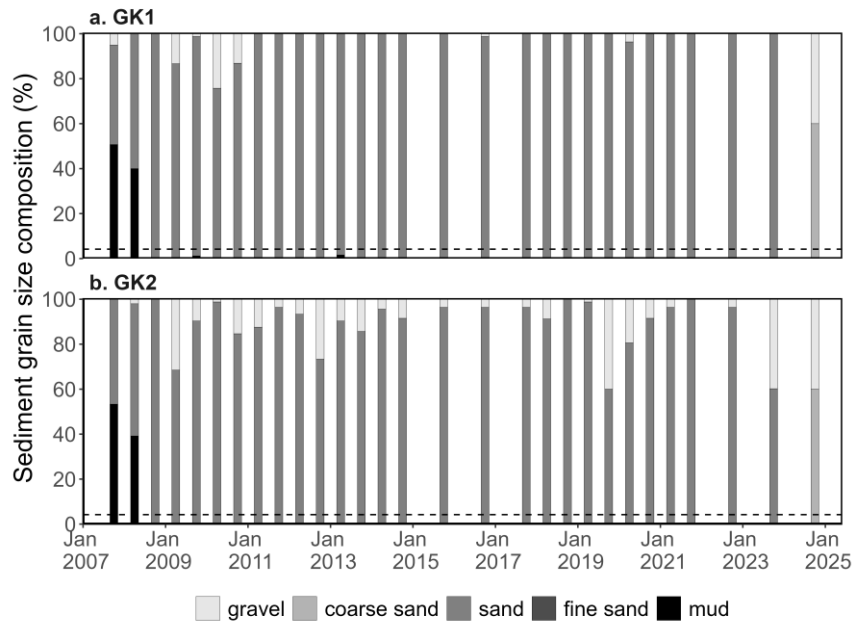


Figure 130. Sediment grain size composition at reef intertidal habitat monitoring sites in the Fitzroy region, 2007–2025. Dashed line is the Reef long-term average proportion of mud.

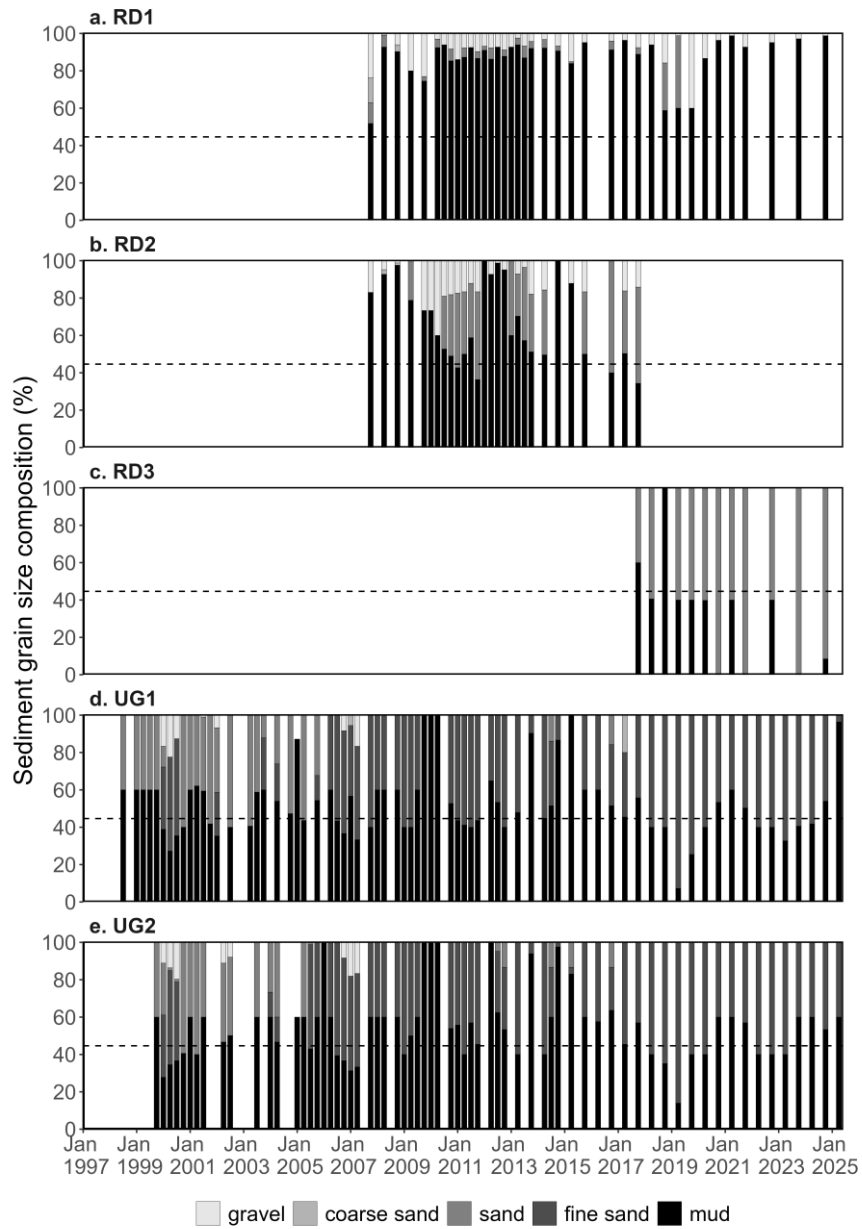


Figure 131. Sediment grain size composition at estuarine intertidal habitat monitoring sites in the Burnett–Mary region, 1999–2025. Dashed line is the Reef long-term average proportion of mud.

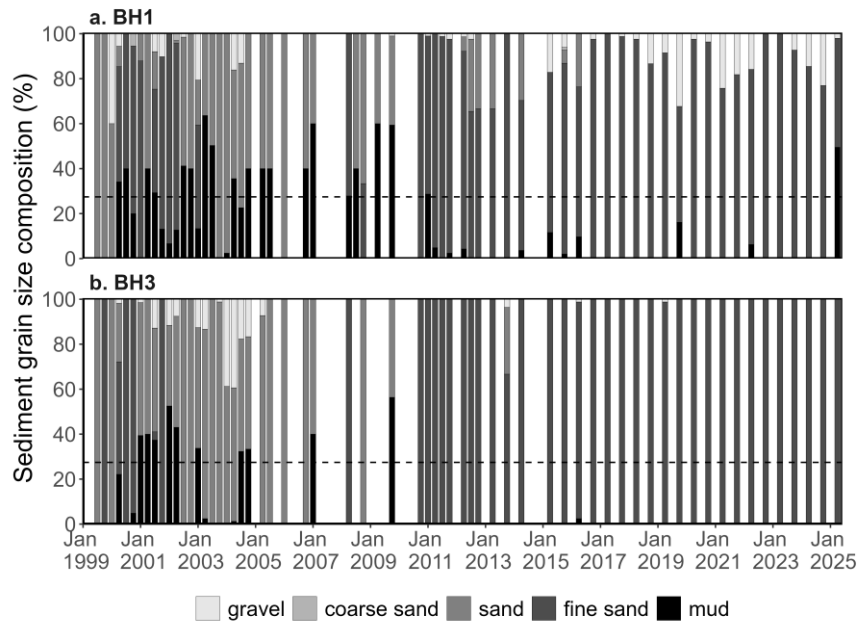


Figure 132. Sediment grain size composition at coastal intertidal habitat monitoring sites in the Burnett–Mary region, 1999–2025. Dashed line is the Reef long-term average proportion of mud.

## **Appendix 3 Results of statistical analysis**

Table 22. 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- $\tau$ ), two-sided p-value (significant at  $\alpha = 0.05$  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	<i>n</i>	Kendall $\tau$	<i>p</i> (2-sided)	Sen's slope (confidence interval)	trend
Cape York	coastal intertidal	BY1	2012	2024	17	-0.155	0.41	-0.345	no trend
		BY2	2012	2024	17	-0.059	0.773	-0.218	no trend
		<b>SR1</b>	<b>2012</b>	<b>2024</b>	<b>15</b>	<b>-0.448</b>	<b>0.023</b>	<b>-0.485 (-0.815 to -0.144)</b>	<b>decrease</b>
		SR2	2012	2024	15	0	1	0	no trend
	coastal subtidal	BY3	2019	2024	5	-0.316	0.613	-0.01	no trend
		BY4	2017	2024	7	-0.488	0.172	-3.632	no trend
		LR1	2015	2024	9	0.389	0.175	1.234	no trend
		LR2	2015	2024	8	-0.143	0.711	-2.546	no trend
		MA1	2021	2024	4	0.667	0.308	2.028	no trend
		MA2	2021	2024	4	-0.333	0.734	-5.839	no trend
	reef intertidal	AP1	2003	2017	35	-0.459	<0.001	-0.533 (-0.763 to -0.283)	decrease <sup>(2017)</sup>
		AP2	2005	2017	24	-0.022	0.901	-0.03	no trend
		<b>FR1</b>	<b>2012</b>	<b>2024</b>	<b>16</b>	<b>-0.377</b>	<b>0.047</b>	<b>-0.327 (-0.552 to 0)</b>	<b>decrease</b>
		<b>FR2</b>	<b>2012</b>	<b>2024</b>	<b>15</b>	<b>-0.6</b>	<b>0.002</b>	<b>-1.116 (-1.623 to -0.712)</b>	<b>decrease</b>
		ST1	2012	2024	17	0.265	0.149	0.287	no trend
		ST2	2012	2024	17	0.332	0.07	0.556	no trend
		YY1	2012	2014	3	0.333	1	1.045	no trend <sup>(2014)</sup>
reef subtidal	<b>FG1</b>	<b>2016</b>	<b>2024</b>	<b>9</b>	<b>-0.556</b>	<b>0.048</b>	<b>-1.813 (-3.93 to -0.023)</b>	<b>decrease</b>	
	FG2	2016	2024	9	-0.479	0.093	-1.146	no trend	
pooled		2012	2024	17	-0.338	0.064	-0.237	no trend	
Wet Tropics	coastal intertidal	<b>LB1</b>	<b>2005</b>	<b>2025</b>	<b>54</b>	<b>-0.338</b>	<b>&lt;0.001</b>	<b>-0.013 (-0.046 to 0)</b>	<b>decrease</b>
		LB2	2005	2025	53	-0.112	0.258	-0.001	no trend
		<b>YP1</b>	<b>2000</b>	<b>2025</b>	<b>87</b>	<b>0.161</b>	<b>0.028</b>	<b>0.097 (0.012 to 0.188)</b>	<b>increase</b>
	YP2	2001	2025	83	0.123	0.101	0.051	no trend	
	coastal subtidal	MS1	2017	2024	7	-0.333	0.368	-1.824	no trend

NRM region	Habitat	Site	First Year	Last Year	<i>n</i>	<i>Kendall</i> <i>-T</i>	<i>p</i> (2-sided)	Sen's slope (confidence interval)	trend
	reef intertidal	MS2	2015	2024	9	-0.111	0.754	-0.235	no trend
		D11	2007	2025	45	0.19	0.067	0.083	no trend
		DI2	2007	2025	45	0.181	0.082	0.121	no trend
		GI1	2001	2025	83	-0.043	0.569	-0.021	no trend
		GI2	2005	2025	69	0.08	0.335	0.055	no trend
		GO1	2008	2016	7	-0.429	0.23	-1.682	no trend
		LI1	2008	2025	51	-0.078	0.426	-0.038	no trend
	reef subtidal	DI3	2008	2025	57	0.016	0.863	0.001	no trend
		GI3	2008	2025	53	-0.183	0.054	-0.238	no trend
		LI2	2008	2025	51	0.115	0.239	0.039	no trend
pooled		2000	2025	96	-0.126	0.069	-0.049	no trend	
Burdekin	coastal intertidal	BB1	2002	2025	74	-0.087	0.275	-0.082	no trend
		SB1	2001	2025	80	-0.067	0.383	-0.036	no trend
		<b>SB2</b>	<b>2001</b>	<b>2025</b>	<b>78</b>	<b>-0.236</b>	<b>0.002</b>	<b>-0.188 (-0.321 to -0.057)</b>	<b>decrease</b>
		JR1	2012	2024	22	0.117	0.463	0.305	no trend
		JR2	2012	2024	21	0.305	0.057	0.939	no trend
		BW1	2019	2020	3	-1	0.296	-0.445	no trend
		BW2	2019	2025	12	0.061	0.837	0.165	no trend
	BW3	2021	2025	9	0.222	0.466	0.197	no trend	
	reef intertidal	MI1	2005	2025	67	-0.121	0.148	-0.122	no trend
		<b>MI2</b>	<b>2005</b>	<b>2025</b>	<b>65</b>	<b>-0.261</b>	<b>0.002</b>	<b>-0.394 (-0.614 to -0.144)</b>	<b>decrease</b>
reef subtidal	MI3	2008	2025	58	-0.116	0.202	-0.233	no trend	
pooled		2001	2025	85	-0.096	0.196	-0.064	no trend	
Mackay Whitsunday	estuarine intertidal	SI1	2005	2025	45	-0.085	0.417	-0.078	no trend
		SI2	2005	2025	40	0.133	0.23	0.071	no trend
	coastal intertidal	CV1	2017	2025	14	0.165	0.443	0.236	no trend
		CV2	2017	2024	14	0.253	0.228	0.261	no trend

NRM region	Habitat	Site	First Year	Last Year	<i>n</i>	<i>Kendall</i> <i>-τ</i>	<i>p</i> (2-sided)	Sen's slope (confidence interval)	trend	
		LL1	2022	2024	5	0.4	0.462	2.659	no trend	
		<b>MP2</b>	<b>2000</b>	<b>2025</b>	<b>52</b>	<b>0.366</b>	<b>&lt;0.001</b>	<b>0.23 (0.127 to 0.327)</b>	<b>increase</b>	
		<b>MP3</b>	<b>2000</b>	<b>2025</b>	<b>50</b>	<b>0.221</b>	<b>0.024</b>	<b>0.14 (0.02 to 0.257)</b>	<b>increase</b>	
		<b>PI2</b>	<b>1999</b>	<b>2025</b>	<b>68</b>	<b>-0.357</b>	<b>&lt;0.001</b>	<b>-0.266 (-0.376 to -0.164)</b>	<b>decrease</b>	
		PI3	1999	2025	68	-0.115	0.165	-0.072	no trend	
		SH1	2017	2024	14	0.385	0.063	1.251	no trend	
		coastal subtidal	NB1	2015	2024	10	-0.289	0.283	-2.8	no trend
			NB2	2015	2024	10	0.289	0.283	1.011	no trend
		reef intertidal	HB1	2000	2025	54	-0.08	0.399	-0.041	no trend
			HB2	2000	2025	53	0.096	0.315	0.046	no trend
	<b>HM1</b>		<b>2007</b>	<b>2025</b>	<b>36</b>	<b>-0.311</b>	<b>0.008</b>	<b>-0.115 (-0.206 to -0.032)</b>	<b>decrease</b>	
	HM2		2007	2021	27	-0.448	0.001	-0.141 (-0.282 to -0.054)	decrease <sup>(2021)</sup>	
	HM3		2021	2025	8	0.071	0.902	0.005	no trend	
	reef subtidal	LN3	2021	2025	9	0	1	-0.014	no trend	
		<b>CH4</b>	<b>2000</b>	<b>2024</b>	<b>17</b>	<b>-0.603</b>	<b>&lt;0.001</b>	<b>-2.48 (-3.278 to -0.888)</b>	<b>decrease</b>	
		<b>CH5</b>	<b>2000</b>	<b>2024</b>	<b>17</b>	<b>-0.544</b>	<b>0.003</b>	<b>-1.457 (-2.641 to -0.285)</b>	<b>decrease</b>	
		LN1	2017	2025	16	-0.133	0.499	-0.221	no trend	
		LN2	2017	2020	6	0.333	0.452	0.313	no trend	
		TO1	2015	2024	10	0.422	0.107	1.05	no trend	
		TO2	2015	2024	10	0.333	0.21	0.98	no trend	
<b>WB1</b>		<b>1999</b>	<b>2024</b>	<b>5</b>	<b>-1</b>	<b>0.027</b>	<b>-2.998 (-20.475 to -0.77)</b>	<b>decrease</b>		
WB3	2000	2024	18	-0.302	0.088	-0.393	no trend			
pooled		<b>1999</b>	<b>2025</b>	<b>83</b>	<b>-0.251</b>	<b>&lt;0.001</b>	<b>-0.089 (-0.141 to -0.043)</b>	<b>decrease</b>		
Fitzroy	estuarine intertidal	<b>GH1</b>	<b>2005</b>	<b>2024</b>	<b>43</b>	<b>-0.355</b>	<b>&lt;0.001</b>	<b>-0.543 (-0.862 to -0.221)</b>	<b>decrease</b>	
		GH2	2005	2024	43	-0.146	0.17	-0.256	no trend	
	coastal intertidal	RC1	2002	2024	42	-0.084	0.442	-0.133	no trend	
		WH1	2002	2024	43	0.177	0.096	0.141	no trend	

NRM region	Habitat	Site	First Year	Last Year	<i>n</i>	<i>Kendall</i> <i>-T</i>	<i>p</i> (2-sided)	Sen's slope (confidence interval)	trend
	reef intertidal	<b>GK1</b>	<b>2007</b>	<b>2024</b>	<b>29</b>	<b>-0.545</b>	<b>&lt;0.001</b>	<b>-0.089 (-0.135 to -0.051)</b>	<b>decrease</b>
		GK2	2007	2024	29	-0.062	0.653	-0.007	no trend
	pooled		<b>2002</b>	<b>2024</b>	<b>55</b>	<b>-0.356</b>	<b>&lt;0.001</b>	<b>-0.205 (-0.3 to -0.111)</b>	<b>decrease</b>
Burnett Mary	estuarine intertidal	RD1	2007	2024	38	0.179	0.119	0.018	no trend
		<b>RD2</b>	<b>2007</b>	<b>2017</b>	<b>28</b>	<b>-0.409</b>	<b>0.003</b>	<b>-0.009 (-0.096 to -0.001)</b>	<b>decrease</b>
		RD3	2017	2024	12	0	1	-0.043	no trend
		UG1	1998	2025	73	0.034	0.674	0	no trend
		<b>UG2</b>	<b>1999</b>	<b>2025</b>	<b>69</b>	<b>0.181</b>	<b>0.029</b>	<b>0.028 (0 to 0.146)</b>	<b>increase</b>
	coastal intertidal	BH1	1999	2025	64	0.03	0.732	0.015	no trend
		<b>BH3</b>	<b>1999</b>	<b>2025</b>	<b>62</b>	<b>0.301</b>	<b>&lt;0.001</b>	<b>0.126 (0.064 to 0.178)</b>	<b>increase</b>
pooled		1998	2025	86	0.03	0.69	0.01	no trend	

Table 23. Resilience score and resilience score category for each site in 2024–25.

MMP Site	Score	Score category	% colonising species > 50%	% cover < low cover threshold	Repro structures present (all species)	Repro structures present (foundational species)	Repro history (last 3 years)	Persistent species present
<b>Cape York</b>								
BY1	15	1.1	FALSE	TRUE	FALSE	FALSE	TRUE	TRUE
BY2	15	1.1	FALSE	TRUE	FALSE	FALSE	FALSE	TRUE
FR1	15	1.1	FALSE	TRUE	FALSE	FALSE	FALSE	TRUE
FR2	15	1.1	FALSE	TRUE	FALSE	FALSE	FALSE	TRUE
SR1	14	1.1	FALSE	TRUE	FALSE	FALSE	FALSE	TRUE
SR2	12	1.1	FALSE	TRUE	FALSE	FALSE	TRUE	TRUE
ST1	69	2.1.2	FALSE	FALSE	FALSE	FALSE	TRUE	TRUE
ST2	14	1.1	FALSE	TRUE	FALSE	FALSE	FALSE	TRUE
<b>Northern Wet Tropics</b>								
GI1	50	2.1.1	FALSE	FALSE	FALSE	FALSE	FALSE	TRUE
GI2	68	2.1.2	FALSE	FALSE	FALSE	FALSE	TRUE	TRUE
GI3	63	2.1.2	FALSE	FALSE	FALSE	FALSE	TRUE	TRUE
LI1	3	1.1	TRUE	FALSE	FALSE	FALSE	FALSE	TRUE
LI2	5	1.2	TRUE	FALSE	TRUE	FALSE	FALSE	FALSE
YP1	50	2.1.2	FALSE	FALSE	FALSE	FALSE	TRUE	FALSE
YP2	15	1.1	FALSE	TRUE	FALSE	FALSE	TRUE	FALSE
<b>Southern Wet Tropics</b>								
DI1	50	2.1.2	FALSE	FALSE	TRUE	FALSE	TRUE	TRUE
DI2	57	2.1.2	FALSE	FALSE	TRUE	FALSE	TRUE	TRUE
DI3	30	2.1.1	FALSE	FALSE	FALSE	FALSE	FALSE	TRUE
LB1	0	1.1	FALSE	TRUE	FALSE	FALSE	FALSE	FALSE
LB2	30	2.1.1	FALSE	FALSE	FALSE	FALSE	FALSE	FALSE
<b>Burdekin</b>								
BB1	15	1.1	FALSE	TRUE	FALSE	FALSE	TRUE	FALSE
JR1	50	2.1.2	FALSE	FALSE	TRUE	FALSE	TRUE	FALSE
JR2	50	2.1.2	FALSE	FALSE	TRUE	FALSE	TRUE	FALSE
MI1	13	1.2	FALSE	TRUE	TRUE	FALSE	TRUE	TRUE
MI2	5	1.2	TRUE	FALSE	TRUE	FALSE	FALSE	TRUE
MI3	30	2.1.1	FALSE	FALSE	FALSE	FALSE	FALSE	TRUE
SB1	15	1.1	FALSE	TRUE	FALSE	FALSE	TRUE	FALSE
<b>Mackay-Whitsunday</b>								
HM1	30	2.1.1	FALSE	FALSE	FALSE	FALSE	FALSE	FALSE
HM3	10	1.1	FALSE	TRUE	FALSE	FALSE	FALSE	FALSE
LN1	50	2.1.2	FALSE	FALSE	FALSE	FALSE	TRUE	FALSE
LN3	50	2.1.2	FALSE	FALSE	TRUE	FALSE	TRUE	FALSE
MP2	77	2.2.1	FALSE	FALSE	TRUE	TRUE	TRUE	FALSE

<b>MMP Site</b>	<b>Score</b>	<b>Score category</b>	<b>% colonising species &gt; 50%</b>	<b>% cover &lt; low cover threshold</b>	<b>Repro structures present (all species)</b>	<b>Repro structures present (foundational species)</b>	<b>Repro history (last 3 years)</b>	<b>Persistent species present</b>
MP3	87	2.2.1	FALSE	FALSE	TRUE	TRUE	TRUE	FALSE
SI1	71	2.2.1	FALSE	FALSE	TRUE	TRUE	TRUE	FALSE
SI2	50	2.1.2	FALSE	FALSE	FALSE	FALSE	TRUE	FALSE
<b>Fitzroy</b>								
GH1	77	2.2.1	FALSE	FALSE	TRUE	TRUE	TRUE	FALSE
GH2	74	2.2.1	FALSE	FALSE	TRUE	TRUE	TRUE	FALSE
GK1	3	1.1	TRUE	TRUE	FALSE	FALSE	FALSE	FALSE
GK2	30	2.1.1	FALSE	FALSE	FALSE	FALSE	FALSE	FALSE
RC1	76	2.2.1	FALSE	FALSE	TRUE	TRUE	TRUE	FALSE
WH1	75	2.2.1	FALSE	FALSE	TRUE	TRUE	TRUE	FALSE
<b>Burnett-Mary</b>								
BH1	30	2.1.1	FALSE	FALSE	FALSE	FALSE	FALSE	FALSE
BH3	30	2.1.1	FALSE	FALSE	FALSE	FALSE	FALSE	FALSE
RD1	96	2.2.1	FALSE	FALSE	TRUE	TRUE	TRUE	FALSE
RD3	100	2.2.1	FALSE	FALSE	TRUE	TRUE	FALSE	FALSE
UG1	78	2.2.1	FALSE	FALSE	TRUE	TRUE	FALSE	FALSE
UG2	96	2.2.1	FALSE	FALSE	TRUE	TRUE	FALSE	FALSE

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

<b>MODELS - REEF</b>	<b>N</b>	<b>EDF</b>	<b>CHI-SQ</b>	<b>P-VALUE</b>	<b>R-SQ (ADJ)</b>	<b>DEVIANCE EXPLAINED</b>
<b>% COVER = S(DATE)</b>	96	23.965	6510.347	<0.001	0.581	0.719
<b>% COVER = S(DATE) + HABITAT</b>	353				0.491	0.808
<b>ESTUARINE</b>						
<b>INTERTIDAL</b>		24.157	1562.872	<0.001		
<b>COASTAL</b>						
<b>INTERTIDAL</b>		21.494	1080.505	<0.001		
<b>COASTAL SUBTIDAL</b>		6.287	38.568	<0.001		
<b>REEF INTERTIDAL</b>		14.91	1173.877	<0.001		
<b>REEF SUBTIDAL</b>		16.559	438.243	<0.001		
<b>% COVER = S(DATE) + NRM</b>	434				0.593	0.792
<b>CAPE YORK</b>		9.079	126.447	<0.001		
<b>WET TROPICS</b>		17.51	663.585	<0.001		
<b>BURDEKIN</b>		20.942	1491.557	<0.001		
<b>MACKAY</b>						
<b>WHITSUNDAY</b>		20.269	617.494	<0.001		
<b>FITZROY</b>		14.121	317.902	<0.001		
<b>BURNETT MARY</b>		24.052	1405.035	<0.001		

Table 25 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
<b>CAPE YORK</b>						
% COVER = S(DATE)	43	13.23	440.038	<0.001	0.526	0.678
% COVER = S(DATE) + HABITAT	78				0.6	0.807
COASTAL INTERTIDAL		2.957	21.19	<0.001		
COASTAL SUBTIDAL		2.934	12.715	0.004		
REEF INTERTIDAL		6.284	146.136	<0.001		
REEF SUBTIDAL		2.954	113.249	<0.001		
% COVER = S(DATE) + LOCATION	132				0.669	0.945
COASTAL INTERTIDAL [BY]		2.867	15.382	0.001		
COASTAL INTERTIDAL [SR]	1		1.003	0.317		
COASTAL SUBTIDAL [BY]		2.964	311.228	<0.001		
COASTAL SUBTIDAL [LR]		2.916	14.906	0.002		
COASTAL SUBTIDAL [MA]		1.946	2.944	0.24		
REEF INTERTIDAL [AP]		6.09	89.697	<0.001		
REEF INTERTIDAL [FR]	1		27.495	<0.001		
REEF INTERTIDAL [ST]		2.926	27.891	<0.001		
REEF INTERTIDAL [YY]		1.598	0.336	0.771		
REEF SUBTIDAL [FG]		2.939	83.784	<0.001		
% COVER = S(DATE) + SITE	246				0.539	0.999
AP1		4.877	46.678	<0.001		
AP2		2.634	9.272	0.033		
BY1		1.805	2.741	0.235		
BY2		1.894	3.77	0.172		
BY3		1.97	37.143	<0.001		
BY4		1.998	95.027	<0.001		
FG1		2.599	70.43	<0.001		
FG2		2.874	70.897	<0.001		
FR1	1		1.417	0.234		
FR2	1		15.142	<0.001		
LR1	1		5.048	0.025		
LR2		2.952	8.495	0.037		
MA1	1		0.442	0.506		
MA2		1.925	3.221	0.216		
SR1	1		1.93	0.165		
SR2	1		0.233	0.63		
ST1		2.045	2.644	0.318		
ST2		2.442	8.225	0.041		
YY1		1.004	0.14	0.705		
<b>NORTHERN WET TROPICS</b>						
% COVER = S(DATE)	91	17.375	412.537	<0.001	0.348	0.512
% COVER = S(DATE) + HABITAT	227				0.686	0.74
COASTAL INTERTIDAL		14.32	226.169	<0.001		
REEF INTERTIDAL	227	11.03	241.689	<0.001		
REEF SUBTIDAL	227	8.829	56.839	<0.001		
% COVER = S(DATE) + LOCATION	325				0.827	0.909
COASTAL INTERTIDAL [YP]		13.636	197.727	<0.001		
REEF INTERTIDAL [GI]		5.595	47.516	<0.001		
REEF INTERTIDAL [LI1]		3.368	29.264	<0.001		

MODELS PER NRM REGIONS	N	EDF	CHI-SQ	P-VALUE	R-SQ (ADJ)	DEVIANCE EXPLAINED
REEF SUBTIDAL [GI3]		6.001	57.911	<0.001		
REEF SUBTIDAL [LI2]		7.565	139.022	<0.001		
<b>% COVER = S(DATE) + SITE</b>	<b>477</b>				<b>0.813</b>	<b>0.912</b>
G11		2.441	5.85	0.123		
G12		3.725	17.547	0.004		
G13		4.604	46.038	<0.001		
LI1		3.034	20.123	<0.001		
LI2		5.377	114.052	<0.001		
YP1		8.435	125.19	<0.001		
YP2		7.117	36.757	<0.001		
<b>SOUTHERN WET TROPICS</b>						
<b>% COVER = S(DATE)</b>	<b>68</b>	<b>16.722</b>	<b>1286.39</b>	<b>&lt;0.001</b>	<b>0.716</b>	<b>0.913</b>
<b>% COVER = S(DATE) + HABITAT</b>	<b>165</b>				<b>0.907</b>	<b>0.98</b>
COASTAL INTERTIDAL		13.588	909.182	<0.001		
COASTAL SUBTIDAL		3.02	10.409	0.024		
REEF INTERTIDAL		12.683	697.929	<0.001		
REEF SUBTIDAL		12.849	244.584	<0.001		
<b>% COVER = S(DATE) + LOCATION</b>	<b>172</b>				<b>0.916</b>	<b>0.987</b>
COASTAL INTERTIDAL [LB]		14.909	1083.075	<0.001		
COASTAL SUBTIDAL [MS]		3.087	11.424	0.016		
REEF INTERTIDAL [DI]		13.359	556.992	<0.001		
REEF INTERTIDAL [GO]		5.458	174.879	<0.001		
REEF SUBTIDAL [DI3]		12.948	283.256	<0.001		
<b>% COVER = S(DATE) + SITE</b>	<b>277</b>				<b>0.882</b>	<b>1.118</b>
D11		7.863	178.703	<0.001		
D12		8.13	235.698	<0.001		
D13		7.773	164.069	<0.001		
GO1		3.468	102.789	<0.001		
LB1		8.119	456.099	<0.001		
LB2		7.923	322.571	<0.001		
MS1		1.001	5.707	0.017		
MS2		3.837	10.202	0.034		
<b>BURDEKIN</b>						
<b>% COVER = S(DATE)</b>	<b>85</b>	<b>21.034</b>	<b>2097.452</b>	<b>&lt;0.001</b>	<b>0.789</b>	<b>0.901</b>
<b>% COVER = S(DATE) + HABITAT</b>	<b>208</b>				<b>0.791</b>	<b>0.904</b>
COASTAL INTERTIDAL		20.359	861.055	<0.001		
REEF INTERTIDAL		14.819	502.311	<0.001		
REEF SUBTIDAL		13.117	566.338	<0.001		
<b>% COVER = S(DATE) + LOCATION</b>	<b>242</b>				<b>0.755</b>	<b>0.895</b>
COASTAL INTERTIDAL [BW]		1.893	5.642	0.041		
COASTAL INTERTIDAL [JR]		9.085	185.615	<0.001		
COASTAL INTERTIDAL [TSV]		19.409	627.484	<0.001		
REEF INTERTIDAL [MI]		13.788	372.202	<0.001		
REEF SUBTIDAL [MI3]		12.382	433.634	<0.001		
<b>% COVER = S(DATE) + SITE</b>	<b>488</b>				<b>0.652</b>	<b>0.837</b>
BB1		8.205	187.165	<0.001		
BW1		1	0.024	0.876		
BW2		1	0.007	0.935		
BW3		1	0.059	0.808		
JR1		3.349	15.789	0.002		
JR2		3.607	29.627	<0.001		
MI1		6.511	55.776	<0.001		
MI2		7.419	148.258	<0.001		

MODELS PER NRM REGIONS	N	EDF	CHI-SQ	P-VALUE	R-SQ (ADJ)	DEVIANCE EXPLAINED
MI3		6.503	199.234	<0.001		
SB1		7.822	95.701	<0.001		
SB2		7.587	109.965	<0.001		
<b>MACKAY WHITSUNDAY</b>						
% COVER = S(DATE)	80	20.828	1016.572	<0.001	0.471	0.663
% COVER = S(DATE) + HABITAT	207				0.645	0.851
ESTUARINE						
INTERTIDAL		16.631	271.196	<0.001		
COASTAL INTERTIDAL		18.568	261.894	<0.001		
COASTAL SUBTIDAL		7.16	36.721	<0.001		
REEF INTERTIDAL		7.803	147.226	<0.001		
REEF SUBTIDAL		5.701	25.277	0.001		
% COVER = S(DATE) + LOCATION	342				0.657	0.809
ESTUARINE						
INTERTIDAL [SI]	1		1.4	0.236		
COASTAL INTERTIDAL						
[CV]	1		0.638	0.424		
COASTAL INTERTIDAL						
[LL]	1.333		11.956	0.001		
COASTAL INTERTIDAL						
[MP]	8.526		153.268	<0.001		
COASTAL INTERTIDAL						
[PI]	2.817		18.004	0.001		
COASTAL INTERTIDAL						
[SH1]	2.39		11.808	0.034		
COASTAL SUBTIDAL						
[NB]	6.91		110.31	<0.001		
REEF INTERTIDAL [HB]	2.067		16.703	<0.001		
REEF INTERTIDAL						
[HM]	6.53		42.054	<0.001		
REEF INTERTIDAL						
[LN3]	1		0.285	0.593		
REEF SUBTIDAL [CH]	1		0.083	0.773		
REEF SUBTIDAL [LN]	1.621		0.768	0.709		
REEF SUBTIDAL [TO]	3.887		44.432	<0.001		
REEF SUBTIDAL [WB]	1		2.603	0.106		
% COVER = S(DATE) + SITE	635				0.627	0.912
CH4	1		0.064	0.8		
CH5	1		0.021	0.886		
CV1	1		0.279	0.597		
CV2	1		0.754	0.385		
HB1	4.469		14.584	0.023		
HB2	4.852		13.227	0.056		
HM1	1.595		6.66	0.035		
HM2	3.316		58.832	<0.001		
HM3	1		0.03	0.863		
LL1	1		0.461	0.497		
LN1	1.215		0.211	0.808		
LN2	1		0.868	0.352		
LN3	1		0.211	0.646		
MP2	1.001		8.95	0.003		
MP3	1.001		1.921	0.166		
NB1	2.616		21.286	<0.001		
NB2	2.74		14.044	0.009		
PI2	7.512		82.878	<0.001		
PI3	7.825		44.771	<0.001		
SH1	2.732		13.222	0.013		
SI1	6.833		74.176	<0.001		

MODELS PER NRM REGIONS	N	EDF	CHI-SQ	P-VALUE	R-SQ (ADJ)	DEVIANCE EXPLAINED
SI2		5.564	25.488	<0.001		
TO1		3.817	22.632	<0.001		
TO2		3.15	16.212	0.001		
WB1		1	1.063	0.303		
WB3		1	1.572	0.21		
<b>FITZROY</b>						
% COVER = S(DATE)	54	12.167	269.902	<0.001	0.425	0.634
% COVER = S(DATE) + HABITAT	114				0.806	0.925
ESTUARINE INTERTIDAL		10.262	145.419	<0.001		
COASTAL INTERTIDAL		15.997	250.208	<0.001		
REEF INTERTIDAL		1.001	9.876	0.002		
% COVER = S(DATE) + LOCATION	114				0.806	0.925
ESTUARINE INTERTIDAL [GH]		10.259	145.548	<0.001		
COASTAL INTERTIDAL [SWB]		16.009	250.471	<0.001		
REEF INTERTIDAL [GK]		1.001	9.876	0.002		
% COVER = S(DATE) + SITE	227				0.695	0.869
GH1		6.431	116.88	<0.001		
GH2		4.673	38.963	<0.001		
GK1		1	7.952	0.005		
GK2		1	0.049	0.825		
RC1		7.237	57.466	<0.001		
WH1		4.182	11.397	0.056		
<b>BURNETT MARY</b>						
% COVER = S(DATE)	81	23.036	675.88	<0.001	0.511	0.725
% COVER = S(DATE) + HABITAT	141				0.567	0.881
ESTUARINE INTERTIDAL		9.35	55.632	<0.001		
COASTAL INTERTIDAL		22.421	794.04	<0.001		
% COVER = S(DATE) + LOCATION	176				0.631	0.896
ESTUARINE INTERTIDAL [RD]		9.03	52.903	<0.001		
ESTUARINE INTERTIDAL [UG]		8.974	292.288	<0.001		
COASTAL INTERTIDAL [BH]		21.606	780.244	<0.001		
% COVER = S(DATE) + SITE	340				0.447	0.84
BH1		4.151	8.302	0.141		
BH3		2.82	6.905	0.145		
RD1		5.44	76.336	<0.001		
RD2		3.657	65.433	<0.001		
RD3		1.438	0.584	0.741		
UG1		8.569	249.436	<0.001		
UG2		8.433	199.626	<0.001		

Table 26. 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
<b>ESTUARINE INTERTIDAL</b>						
% COVER = S(DATE) + NRM	165				0.45	0.787
MACKAY WHITSUNDAY		6.786	42.882	<0.001		
FITZROY		4.73	46.512	<0.001		
BURNETT MARY		8.829	451.66	<0.001		
<b>COASTAL INTERTIDAL</b>						
% COVER = S(DATE) + NRM	368				0.548	0.701
CAPE YORK		2.083	4.825	0.085		
WET TROPICS		8.618	200.683	<0.001		
BURDEKIN		8.376	526.86	<0.001		
MACKAY WHITSUNDAY		8.57	137.433	<0.001		
FITZROY		7.372	88.572	<0.001		
BURNETT MARY		6.709	70.478	<0.001		
<b>COASTAL SUBTIDAL</b>						
% COVER = S(DATE) + NRM	28				0.171	0.519
CAPE YORK		3.482	24.921	<0.001		
WET TROPICS		1.905	2.053	0.381		
MACKAY WHITSUNDAY		2.831	18.75	0.001		
<b>REEF INTERTIDAL</b>						
% COVER = S(DATE) + NRM	283				0.783	0.863
CAPE YORK		4.889	81.866	<0.001		
WET TROPICS		7.412	718.569	<0.001		
BURDEKIN		7.746	592.053	<0.001		
MACKAY WHITSUNDAY		6.682	153.824	<0.001		
FITZROY		1.002	10.56	0.001		
<b>REEF SUBTIDAL</b>						
% COVER = S(DATE) + NRM	143				0.77	0.78
CAPE YORK		8.523	387.28	<0.001		
WET TROPICS		3.704	59.045	<0.001		
BURDEKIN		3.423	7.199	0.159		
MACKAY WHITSUNDAY		5.241	38.057	<0.001		