**Final Report**

# Final report on thresholds and indicators of declining water quality as tools for tropical seagrass management

Catherine J. Collier, Michelle Devlin, Lucas Langlois, Len J. McKenzie,
Caroline Petus, Eduardo Teixeira da Silva, Kathryn McMahon,
Matthew Adams, Kate O'Brien, John Statton, and Michelle Waycott

# Final report on thresholds and indicators of declining water quality as tools for tropical seagrass management

**A summary of findings from Project 5.3 Vulnerability of seagrass habitats in the GBR to flood plume impacts: light, nutrients and salinity**

Catherine J Collier[1,2], Michelle Devlin[2], Lucas Langlois[2], Len J McKenzie[3], Caroline Petus[2] Eduardo Teixeira da Silva[2], Kathryn McMahon[4], Matthew Adams[5], Kate O'Brien[5], John Statton[6], and Michelle Waycott[7]

[1]School of Marine and Tropical Biology, James Cook University, Townsville 4811

[2]The Centre for Tropical & Aquatic Ecosystems Research (TropWater), James Cook University, Townsville, Queensland 4811

[3]The Centre for Tropical & Aquatic Ecosystems Research (TropWater), James Cook University, Cairns, Queensland 4878

[4]School of Natural Sciences, Edith Cowan University, Joondalup, Perth, Western Australia

[5]School of Chemical Engineering, The University of Queensland, Brisbane, Queensland 4072

[6]School of Plant Biology, The University of Western Australia, Crawley, Perth Western Australia

[7]School of Earth and Environmental Science, Australian Centre for Evolutionary Biology and Biodiversity, University of Adelaide, Adelaide, South Australia 5005

JAMES COOK UNIVERSITY
AUSTRALIA

Australian Government

Department of the Environment

# Contents

# List of Figures

# List of Tables

# Acronyms Used In This Report

**DoE** ............... Department of the Environment

**GBR** .............. Great Barrier Reef

**NERP** ............ National Environmental Research Program

**RRRC** ............ Reef and Rainforest Research Centre Limited

**RS** ................Remote sensing

**WQ** .............. Water Quality

**C** .................... Carbon

**N** ................... Nitrogen

# Abbreviations Used In This Report

**AIMS** ............ Australian Institute of Marine Science

**C** .................... Carbon

**C:N** ................ Carbon:Nitrogen

**CDOM** .......... Coloured Dissolved Organic Matter

**Chl a** ............. Chlorophyll a

**DIN** .............. Dissolved inorganic nitrogen

**DIP** .............. Dissolved inorganic phosphorus

**FRP** ............... Filterable Reactive Phosphorus

**GBR** ............. Great Barrier Reef

**GBRMP** ........ Great Barrier Reef Marine Park

**GBRMPA** ...... Great Barrier Reef Marine Park Authority

**JCU** .............. James Cook University

**$K_d$** .................. Light attenuation coefficient

**N** ................... Nitrogen

**NASA** ........... National Aeronautics and Space Administration

**$NH_4^+$** .............. Ammonium

**$NO_3^-$** ............... Nitrate

**NRM** ............. Natural Resource Management

**NTU** ............. Nephelometric Turbidity Units

**OACs** ............ Optically Active Compounds

**PAR** .............. Photosynthetically Active Radiation

**PSU** ............. Practical Salinity Units

**QLD** ............. Queensland

**RS** ................ Remote Sensing

**RRMMP** ........ Reef Rescue Marine Monitoring Program

**SD** ................ Standard Deviation

**SE** ................. Standard Error

**SPM** .............. Suspended particulate matter

**WQ**............... Water Quality

# Acknowledgements

# Synthesis

Background

The broad aims of this study were to test the vulnerability of seagrasses to declining water quality, in particular, changes associated with flooding. This project was established in response to extensive seagrass loss that occurred from 2009 to 2011 in the Great Barrier Reef when there was above average run-off for multiple wet seasons, which culminated in the passage of cyclone Yasi through the northern GBR in 2011. Dugong, which are seagrass specialists, also suffered record levels of mortality in 2011 and had low calving rates in the southern GBR following the unprecedented levels of seagrass loss. In 2014, there were signs that seagrass meadows were recovering. This project has been designed to help interpret the effects of flood water on seagrass as well as improve our capacity to detect and respond to other water quality related impacts (e.g. dredging).

Flood plumes are low in salinity (hypo-saline), have high nutrient concentrations (triggering blooms of phytoplankton) and both dissolved (e.g. CDOM) and particulate matter (e.g. suspended sediments) that create low light conditions. Floodwaters may also contain toxic levels of contaminants, such as herbicides. This was a 2-year project that included desktop analyses, analysis of in situ light logger data, analysis of remote sensing-derived water quality, a review of the literature and a number of experiments to test seagrass responses to salinity, light and nutrients and to identify thresholds associated with loss.

The objectives of this project were to determine:

- The level of exposure of seagrass meadows to broad scale and long-term changes in water quality associated with flood plumes in coastal regions of the GBR
- The influence of light, nutrients and salinity on seagrass condition
- Refined thresholds of concern for light, nutrients and salinity
- Indicators of seagrass condition to report on status
- Future trajectories for GBR ecosystems

Science summary

Plumes of floodwaters were detected using satellite imagery. Seagrass meadows in coastal and estuarine regions of the GBR were exposed to plumes of variable water quality conditions during the wet season months (Nov-April). Ocean colour information derived from remote sensing was used to develop water quality thresholds that occur when seagrasses have experienced greater than 50% loss in abundance. Different permutations of ocean colour conditions have been extracted for the four main seagrass habitats. The derived water quality thresholds all relate to the constituents (TSS, chl-a, CDOM) that influence the light attenuation. Therefore, in situ data and aquarium-based experiments were used to test seagrass responses to salinity, light and nutrients to identify which aspect of flood plumes have the greatest effect on meadow health.

The response of seagrasses to hypo-salinity was tested from 3 PSU (almost freshwater) to 36 PSU (seawater). GBR seagrasses had broad hypo-salinity tolerance with thresholds (associated with mortality) occurring at <3 PSU for Zostera muelleri and <9 PSU for Halophila ovalis and Halodule uninervis after 10 weeks (Figure 1). There was a stress-induced morphometric response at low-moderate salinities (9 – 15 PSU) whereby shoot density proliferated in response to hypo-salinity. Given the broad salinity tolerance it is highly unlikely that low salinity was the primary cause of seagrass losses associated with flooding. We did not prioritse hypo-salinity for further interactive experimental testing.

Seagrass abundance at Magnetic Island and Dunk Island was correlated to in situ light levels using data from the Reef Rescue MMP. High and significant (p<0.05) correlations between seagrass loss and low light, suggests that low light contributed to seagrass loss from 2009 to

2011. Therefore, effects of low light were prioritized for further experimental work, including the interactive effects of elevated nutrients and seasonal variations in water temperature. Four seagrass species were grown at two temperatures, warm (~27°C) and cool water (~22°C), and were exposed to light levels ranging from 0 to 70% of full sunlight (0-23 mol m$^{-2}$ d$^{-1}$) in aquaria experiments for 3 months. All species suffered faster mortality (declines in seagrass) in warm compared to cooler water, and H. ovalis and Z. muelleri were more sensitive to low light levels than C. serrulata and H. uninervis.  From this study, light thresholds for any chosen level of seagrass decline (e.g. 10, 20, 50% decline) could be calculated.  50% loss occurred at 3 to 6 mol m$^{-2}$ d$^{-1}$ after 14 weeks exposure depending on species and water temperature (Figure 1), and 20% loss occurred at 7.4 to 10.4 mol m$^{-2}$ d$^{-1}$. This experimental approach revealed a very similar light threshold for H. uninervis from Magnetic Island derived using in situ decline and in situ daily light (both approximately 4 mol m$^{-2}$ d$^{-1}$ for 50% loss after 3 months in warm water). The similarity has increased confidence in thresholds derived from experimental work for other species and verified the conclusion that low light was a large contributor to recent in situ seagrass loss in the GBR.



* for short-term exposure, varies with time of exposure (up to 3 months) and among species
# affected by temperature  $ threshold is established for seagrass loss of greater than 50%
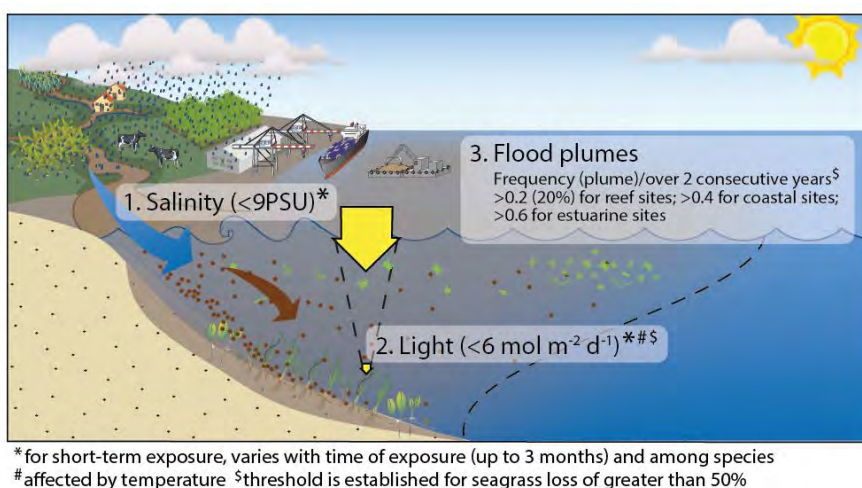
**Figure 1.**    Summary of salinity, light and flood plume thresholds identified in this study.

Indicators of light stress were also tested. Firstly, a review of global literature identified robust indicators of light stress. Secondly, some of these sub-lethal (i.e. detectable before mortality) indicators (tissue nutrients, $\delta^{13}$C and rhizome carbohydrates) were tested using two aquaria-based experiments: light only and light × nutrients. Results demonstrated that the combination of indicators (C/N, $\delta^{13}$C and rhizome carbohydrates) provide the most powerful interpretation of light and/or nutrient stress. C/N was highly sensitive to light and nutrients except at high nutrient concentration when C/N ratios did not respond to light. Rhizome carbohydrates or leaf $\delta^{13}$C are robust complimentary early-warning indicators of light availability with the most sensitive of these indicators varying among species.

This study has and will contribute to the following management outcomes:

- Identified seagrass meadows at high risk of exposure to flood waters when certain conditions, defined by the frequency of colour classes, are experienced over two consecutive years.
- Colour class frequency represents an exposure to water types experienced over the wet season.
- Different permutations of ocean colour information can provide information that can be used to develop guidelines on habitat scale seagrass loss.

- Developed new environmental thresholds (light), which will be incorporated into seagrass guidelines for protection of GBR seagrasses.
- Measured new salinity thresholds, which occur at low salinities.
- Confirmed metric selection (sub-lethal indicators) and scoring for Reef Rescue MMP reporting.

# 1. Introduction

There have been chronic declines in inshore water quality in the Great Barrier Reef (GBR) since European settlement, which have led to dramatic ecological shifts (De'ath and Fabricius, 2010; Roff et al., 2013). Intensive use of the GBR catchments for agriculture, grazing, as well as establishment of urban centers and marine based commercial activity such as ports, have placed high pressure on GBR ecosystems (Brodie et al., 2013b). Rivers discharging into the GBR lagoon are the main land-based source of key pollutants (including TSS, dissolved inorganic nitrogen (DIN) and photosystem II-inhibiting herbicides (PSII herbicides) in the coastal and marine environment (Figure 2). The levels of TSS, Colored Dissolved Organic Matters (CDOM) and chlorophyll-a (chl-a) associated with plume waters decrease from the inshore to the offshore boundaries of the River plumes. The relative concentrations of these three Optically Active components (OACs) affect the light attenuation properties of the water types (Devlin et al., 2008, 2009) and the diffuse attenuation coefficient of photosynthetically active radiation ($K_d$(PAR)) decrease from the inshore to the offshore boundaries of the River plumes (Devlin and Schaffelke, 2009; Devlin et al., 2012a, 2013a, b; Petus et al., a, b). Thirty major rivers drain into the GBR, all of which vary considerably in length, catchment area, and flow frequency and intensity. River plumes are driven by high river flow conditions, which in the GBR are the periods in the monsoonal season that are typically associated with the passage of cyclones or low pressure systems, i.e., from about December to April (Devlin and Brodie, 2005). Level of exposure of coastal to marine ecosystems (including seagrass meadows) to river plumes and land sourced pollutants is spatially and temporally dependant of the different land-uses in the GBR catchments, the local transports of pollutants, and the distance of respective ecosystems to the river mouths. Nearly all of the GBR rivers experienced a high degree of flooding during the 2010-2011 wet season due to the very strong 'La Nina' beginning early in the season in mid-2010 and three cyclones (Tasha in December 2010, Anthony in January 2011 and the most damaging: Yasi in February 2011) that crossed the North Queensland coast over a period of three months (Devlin et al., 2012b; Logan et al., 2013). The predominantly inshore distribution of seagrass meadows in the GBR makes them particularly susceptible to the direct effects of flood plumes as well as chronic water quality decline generally.
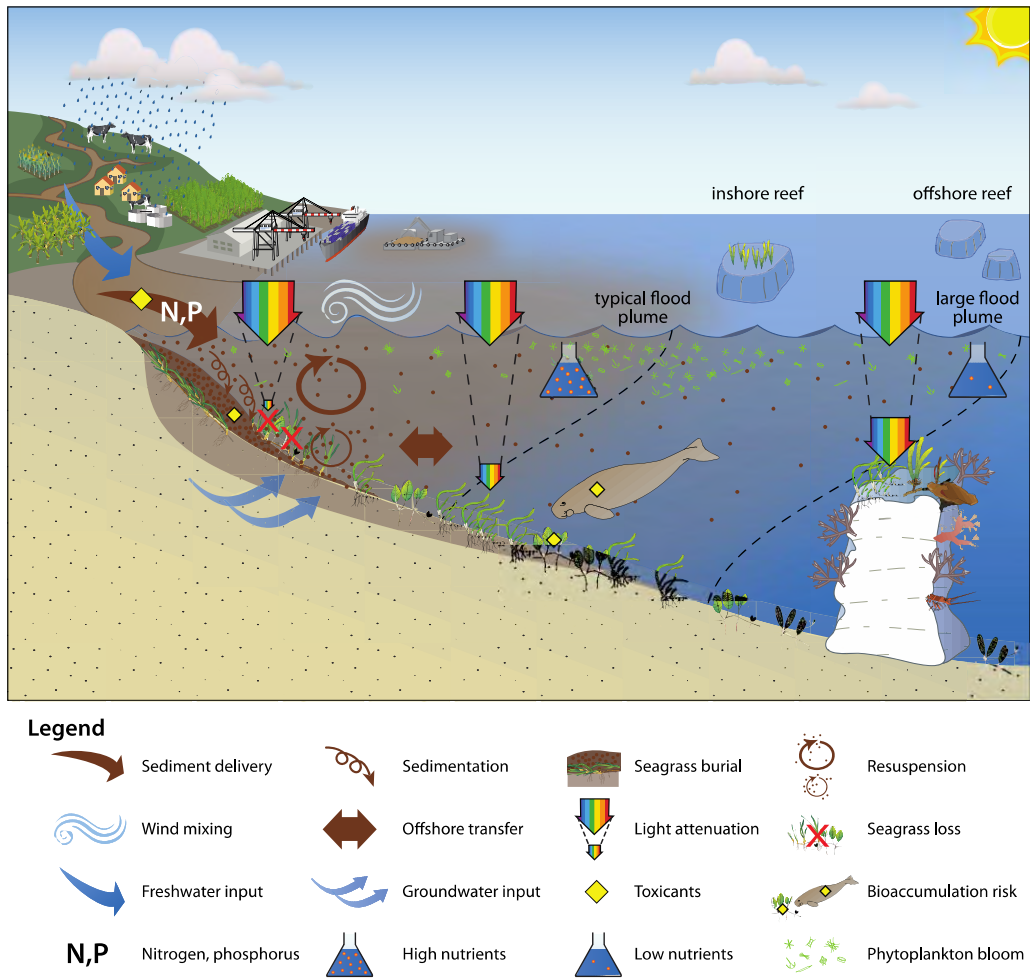
**Figure 2.** Conceptual understanding of flood water impacts on seagrasses

Seagrasses are marine flowering plants with approximately 70 species distributed globally except in polar regions (Short et al., 2011). The ecosystem services provided by seagrass meadows makes them a high conservation priority (Cullen-Unsworth and Unsworth, 2013). For example, nutrient cycling in seagrass meadows makes them one of the most economically valuable ecosystems in the world (Costanza et al., 1997). Furthermore, in the tropics, seagrass meadows support populations of turtles and dugongs, which are seagrass specialists (Marsh et al., 2011) as well as commercial (e.g. prawns) and subsistence (e.g. holothurians) fisheries (Coles et al., 1993; Cullen-Unsworth and Unsworth, 2013). They also support threatened species, with ray-finned fish including seahorses and pipefish, being the most affected group of threatened species (Hughes et al., 2008). Seagrass meadows also incorporate and retain carbon within their tissues and in the sediments, which can affect local pH and increase calcification of coral reefs, and contribute to what has become known as 'Blue Carbon' or marine carbon sequestration (Fourqurean et al., 2012; Unsworth et al., 2012).

The Reef Rescue Marine Monitoring Program has established that seagrass meadows along the GBR were in decline in 2011 (McKenzie et al., 2012). The indicators of this decline were: 73% of sites declined in abundance (below the seagrass guidelines) from 2010-2011 and 80% showed a declining long-term trend (5-10 years); 55% of sites exhibited shrinking meadow area, majority of sites had few seeds that would enable recovery. The trends in seagrass decline were the result of changing water quality, particularly caused by flood plumes, as well as the direct impacts of cyclones in localized areas (Figure 1). Specifically, there were signs that there was an excess of nutrients and low light availability at many sites. Low salinity and herbicides

entering the GBR through floodwater may also have contributed to seagrass declines. The decline in seagrass meadow abundance and area was also associated with record dugong and turtle mortality in 2011 (Meager and Limpus, 2012).

# Seagrasses indicate changing water quality at a range of scales

As seagrasses are highly sensitive to changing water quality, they are considered "sentinels" of coastal degradation (Dennison et al., 1993; Orth et al., 2006) and, as such, they are frequently incorporated into assessments of estuarine and coastal integrity (e.g. Borja et al., 2008; Fourqurean et al., 1997; Romero et al., 2007). In addition to being good bioindicators of changing water quality, changes in seagrass health indicate likely ecological and economic flow-on effects. The advantage of measuring seagrasses as bioindicators, in addition to water quality, is that they integrate a temporal component, reflecting both the past and current environmental condition. Good bioindicators should be scientifically defensible, and the cause-effect pathway, as well as trigger levels leading to their change should be predictable and repeatable (McMahon et al., 2013). There is a plethora of potential bioindicators but ecological health assessments need to be based on simple and scientifically tested indicators (Borja et al., 2008).

One of the key causes of seagrass decline is light reduction (Waycott et al., 2009). They are particularly sensitive to light stress: many species have high light requirements and frequently occur in shallow estuarine or coastal regions, which are readily impacted by flood waters. The ways in which seagrasses can respond to changing light are reasonably well documented (Figure 2) (McMahon et al., 2013; Ralph et al., 2007). Even in the case of light stress, reliability and repeatability remains a limitation of indicator selection when developing robust monitoring programs. Indicator selection becomes even more complicated when designing programs that need to report on multiple water quality, and even climate related stressors such as the Reef Rescue Marine Monitoring Program (MMP).

There are many scales at which indicators respond, ranging from sub-lethal (physiological), through to meadow-scale (or state change) losses (Figure 3, Table 1). These indicators also respond at different temporal scales, with sublethal, physiological indicators able to respond from seconds to months, while the meadow-scale effects usually take many months to be detectable. A robust monitoring program will benefit 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 impacts. Many of the indicators listed in Figure 2 and Table 1 remain untested for GBR seagrasses or the environmental thresholds corresponding to their changes are not yet quantified for many indicators. This limits our ability to interpret results of monitoring.
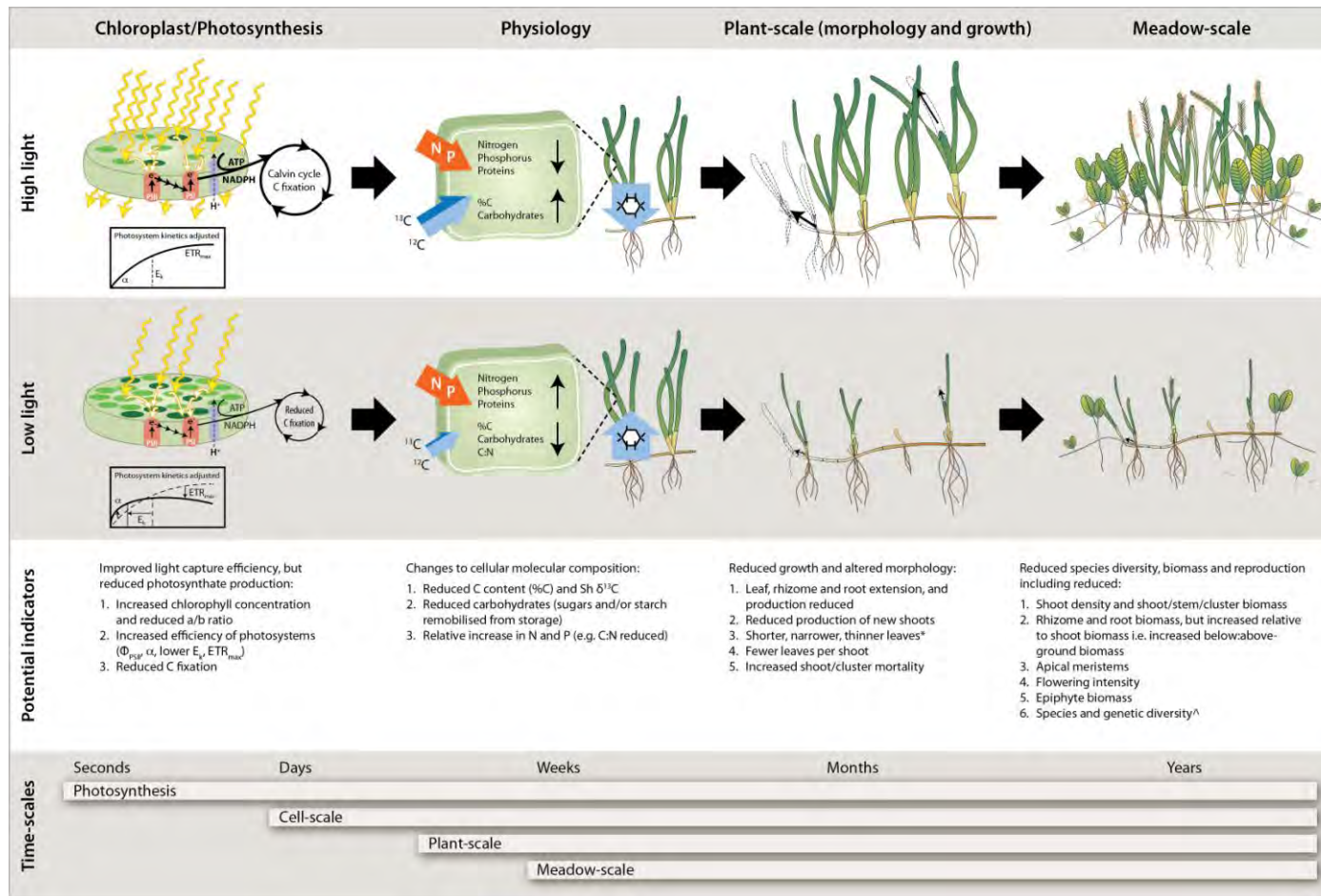
**Figure 3.** Conceptual diagram of the current understanding of the of seagrass response pathway under low light conditions separated by photosynthetic, other physiological, plant-scale (growth and morphology) and meadow-scale variables. The timescales at which the responses to light reduction generally occur are indicated at the base of the diagram. Potential bioindicators are highlighted. (McMahon et al., 2013)

**Table 1.** Response stages of seagrass meadows to external stressors and the indicator responses observed in Great Barrier Reef monitored seagrass meadows (adapted from Waycott and McKenzie 2010). * measures were utilised in Paddock to Reef reporting (McKenzie et al., 2013).

| Indicator | Sub-lethal (ecophysiological) | State change (whole plant and population scale) | Population decline (whole meadow scale) |
|---|---|---|---|
| A. Tissue nutrients | Ratios of key macronutrients change to indicate relative excesses (e.g. C:N*, C:P, N:P, $\delta^{13}$C) | Limited by species variable upper threshold | - |
| B. Chlorophyll concentrations | Rapid short term changes observed | Limited by species variable upper threshold | - |
| C. Production of reproductive structures | - | Reduced flowering and fruiting, loss of seeds for meadow recovery seen as high variability among sites* | Threshold reached where no reproduction occurs |
| D. Change in plant morphology | - | Change in meadow LAI: reduction in leaf area | Threshold where reduction in leaf area is incapable of meeting respiration demand |
| E. Community structure | - | Change in species composition | Loss of species due to the threshold being reached |
| F. Change in species abundance (population structure) | - | Change in abundance of species (i.e. % cover)* or the number of individuals in each population | Reduction in effective population size |
| G. Change in meadow area | - | - | Reduction in total meadow area (habitat loss) |
| H. Recovery time from loss | Limited or no change | Measurably delayed | Potentially no recovery if threshold reached |

# Guidelines and reporting

## Water quality guidelines

Water quality guidelines for the GBR have been developed based on correlations between coral health and water quality through long-term monitoring (De'ath and Fabricius, 2010; GBRMPA, 2009). These guidelines can be used to set targets for water quality and trigger management actions where guidelines are exceeded. The Reef Rescue Marine Monitoring Program (RRMMP) is one of the flagship programs in the GBR that can be used to detect breaches of guidelines, and identify the ecological effects of breaches, or improvements in water quality. There are currently no specific guidelines for seagrass meadows of the GBR due largely to a lack of data to support their development and identification of potential targets for detecting significant change.

## Reef Rescue MMP Report card

The RRMMP aims to report on changes in water quality and ecosystem responses to these changes, including inshore corals and seagrasses.



**Figure 4.** Seagrass abundance monitoring is carried out as part of the Reef Rescue Marine Monitoring Program, and it is used together with tissue nutrients and reproductive output to score seagrass meadow health on an annual basis.

An annual report card is generated for the status of seagrass each year for the GBR and each of the NRMs (Figure 5). The methodology used in the MMP is described in an annually updated Quality Assurance/Quality Control document (GBRMPA, 2014). The scoring for these report cards is generated from annual monitoring and the indicators listed below (McKenzie et al., 2013).
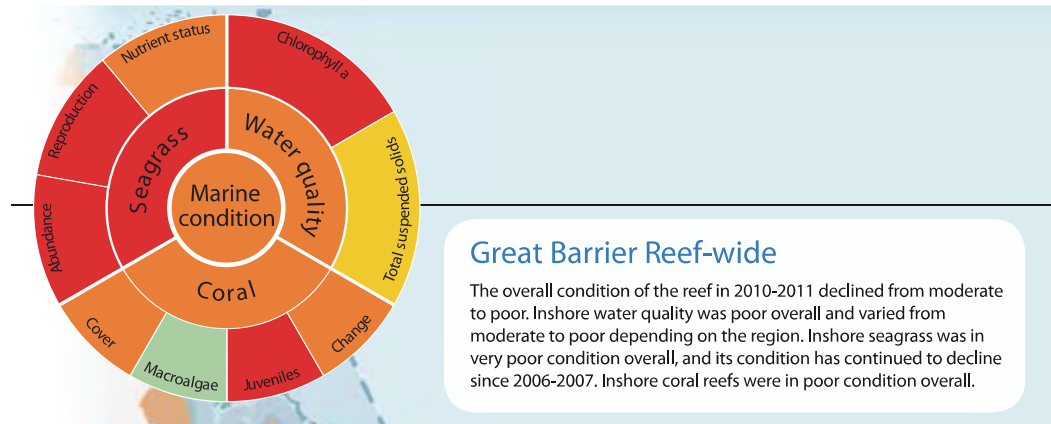
**Figure 5.** Excerpt from the 2010-2011 Great Barrier Reef Report Card from the Reef Water Quality Protection Plan showing the incorporation of three metrics (Abundance, Reproduction and Nutrient status) in the report card scoring for seagrasses.

Three indicators were selected for the report card using advice from expert working groups, by the GBRMPA, the Paddock to Reef Integration technical advisory group (McKenzie et al., 2013), as well as available evidence in the literature on their suitability for identifying changes in water quality (Fourqurean et al., 1997; McMahon et al., 2013). Seagrass abundance is used to indicate the state of the seagrass, reproductive effort to indicate the potential for the seagrass to recover from loss, and the nutrient status to indicate the condition of the environment in which the seagrass are growing in recognition of seagrass' role as a bioindicator. These indicators are scored for calculation of a seagrass status and trend metric (Table 2, Table 3) (McKenzie et al., 2013). The status of seagrass abundance was determined using the regionally-specific seagrass abundance guidelines developed by McKenzie (2009), while tissue nutrient ratios are scored using globally-derived values that have not yet been tested in the GBR. The third metric, reproduction, is not explored extensively in this project and is not as well developed in terms of thresholds. These indicators were tested in this project to verify their response to light, nutrients and salinity.

**Table 2.**     Scoring threshold table to determine seagrass abundance status.

| description | category | score | status |
|---|---|---|---|
| *very good* | 75-100 | 100 | 80 - 100 |
| *good* | 50-75 | 75 | 60 - <80 |
| *moderate* | low-50 | 50 | 40 - <60 |
| *poor* | <low | 25 | 20 - <40 |
| *very poor* | <low by >20% | 0 | 0 - <20 |

**Table 3.**     Scores for leaf tissue C:N against guideline to determine light and nutrient availability.

| description | C:N ratio range | value | score | status |
|---|---|---|---|---|
| very good | C:N ratio >30* | 30 | 100 | 80 - 100 |
| good | C:N ratio 25-30 | 25 | 75 | 60 - <80 |
| moderate | C:N ratio 20-25 | 20 | 50 | 40 - <60 |
| poor | C:N ratio 15-20 | 15 | 25 | 20 - <40 |
| very poor | C:N ratio <15* | | 0 | 0 - <20 |

*C:N ratios >35 were scored as 100, and C:N ratios <10 were scored as 0*

# 2 Summary of approach

## Goals of the NERP program

"The NERP Tropical Ecosystems Hub will address issues of concern for the management, conservation and sustainable use of the World Heritage listed Great Barrier Reef (GBR) and its catchments, tropical rainforests including the Wet Tropics World Heritage Area (WTWHA), and the terrestrial and marine assets underpinning resilient communities in the Torres Strait, through the generation and transfer of world-class research and shared knowledge. This research will be highly relevant, influential in policy, planning and management, publicly available, and value for money."

This study is part of the NERP TE hub, Theme 2, Understanding Ecosystem Function and Cumulative pressures.

"Theme 2 builds on research undertaken through the MTSRF and other programs that have identified many of the primary risks and threats to the environmental assets of North Queensland. These pressures do not occur in isolation to each other and it is clear that a greater understanding of the cumulative and synergistic impact of these pressures is required for improved management. These pressures are not static, therefore predicting and preparing for change is a significant challenge for environmental decisions makers charged with stewardship of Queensland's natural environment. Changing climates, extreme natural events, changes in natural resource use and population growth are some of the pressures facing these ecosystems. Theme 2 is comprised of four Programs that will increase the understanding of ecosystem function and the impact of synergistic and cumulative pressures on the system. This understanding is essential in developing effective management responses that promote ecosystem resilience."

Theme 2, Program 5 Cumulative impacts on benthic biodiversity" including the Great Barrier Reef, Torres Strait and adjacent catchments (Figure 6).



**Figure 6.**    NERP TE program region including the Great Barrier Reef, Torres Strait and adjacent catchments.

# Project 5.3 deliverables

- The level of exposure of seagrass meadows to broad scale and long-term changes in water quality associated with flood plumes in coastal regions of the GBR
- The influence of light, nutrients and salinity on seagrass condition
- Refined thresholds of concern for light, nutrients and salinity
- Indicators of seagrass condition to report on status
- Future trajectories for GBR ecosystems

We adopted a multi-tiered approach to achieving these goals (Figure 7):

1. Flood plume exposure mapping of in-situ data (a)
2. Analysis of existing in-situ light and seagras abundance data and review of the literature (b,c)
3. Original experimental research to test for the effects of salinity, light and nutrients. We experimentally tested combinations of salinity, light and nutrients with experimental combinations being dependant on: 1. previous knowledge; 2, management priorities; 3. research objectives; and, 4. logistical constraints (d, e, f, g).

Indicators of seagrass status were tested throughout this project and these indicators ranged from sub-lethal (physiological) through to population level (meadow-scale) indicators (Table 4). Abundance and growth were tested the most frequently owing to their ease of measurement and their importance in ecological functioning (e.g. as habitat, sediment stability); however in some projects changes in these indicators were not targeted. More specifically, sub-lethal indicators were the focus of two experiments (d,e) and changes in growth and abundance were not expected, and in fact were even avoided in order to capture the "sub-lethal" response phase. The flood plume mapping work was related to changes in seagrass abundance and species composition from the Reef Rescue Marine Monitoring Program (RRMMP). This report focuses on results that are most relevant for improved management of seagrass meadows of the GBR and all other results will be available through targeted scientific publications.

## Table outlining project 5.3 sub-components

| | Flood plumes | Thresholds | | Sub-lethal indicators | | | Total |
|---|---|---|---|---|---|---|---|
| **Light** | a. Flood plume exposure analysis | c. Light thresholds-seagrass loss* Desktop | f. light x temp (season) Experiment | b. Sub-lethal indicators-review Desktop | d. Light Experiment | e. Light x nutrients Experiment | 6 |
| **Nutrients** | Destop analysis of In situ data | | | | | | 2 |
| **Salinity** | | g. Salinity experiment | | | | | 2 |

*In-situ data generated through the reef Rescue MMP, anlysis of data in this project was a desktop analysis

**Figure 7.** Summary of NERP project 5.3 sub-components showing breakdown among in-situ (flood plume analysis), desk-top analyses and aquarium-based experimental work.

**Table 4.**     Summary of the indicators tested in each of the sub-components of Project 5.3

| Level | Parameter grouping | Parameter | a. Flood plumes | Thresholds b. Salinity | Thresholds c. *In-situ* thresholds | Thresholds d. Light x temperature | Sub-lethal indicators e. Review | Sub-lethal indicators f. Light | Sub-lethal indicators g. Light x nutrients |
|---|---|---|---|---|---|---|---|---|---|
| Physiological (sub-lethal) | Leaf tissue nutrients | %C | | | | | ▤ | ▤ | ▤ |
| | | %N | | | | | ▤ | ▤ | ▤ |
| | | C/N | | | | | ▤ | ▤ | ▤ |
| | | Del$^{13}$C | | | | | ▤ | ▤ | ▤ |
| | Energy reserves | Rhizome carbohydrates | | | | | ▤ | ▤ | ▤ |
| | Photosynthesis | PAM | | | | ▤ | ▤ | ▤ | ▤ |
| | | O$_2$ production | | | | ▤ | ▤ | | ▤ |
| Plant-scale (state change) | Growth | Leaf | | ▤ | | ▤ | ▤ | ▤ | ▤ |
| | | Rhizome | | | | ▤ | ▤ | | |
| | Morphology | Morphology | | | ▤ | ▤ | ▤ | | |
| | Sexual reproduction | Sexual reproduction | | | ▤ | | ▤ | | |
| Meadow-scale (population level) | Abundance | Shoot density | | ▤ | | ▤ | ▤ | ▤ | ▤ |
| | | Percent cover | ▤ | | ▤ | | | | |
| | | Biomass | | ▤ | | ▤ | ▤ | ▤ | ▤ |
| | Species composition | Species composition | ▤ | | | | ▤ | | |

This project extends across a large proportion of the GBR south of Cooktown for the flood plume exposure analysis (a), in situ assessment of light and seagrass loss (c) was from wet tropics and Burdekin Dry Tropics sites in the northern GBR, and collection of seagrasses for experimental work (b,d,f,g) also included collections from Green Island off Cairns (Wet Tropics NRM) down to Gladstone Harbour (Fitzroy NRM), with different species occurring at each site (Table 5 Figure 8).

The remainder of this report has been structured around the objectives listed above. Additional detail on each of the components can be found in associated publications, and these are listed, where relevant.
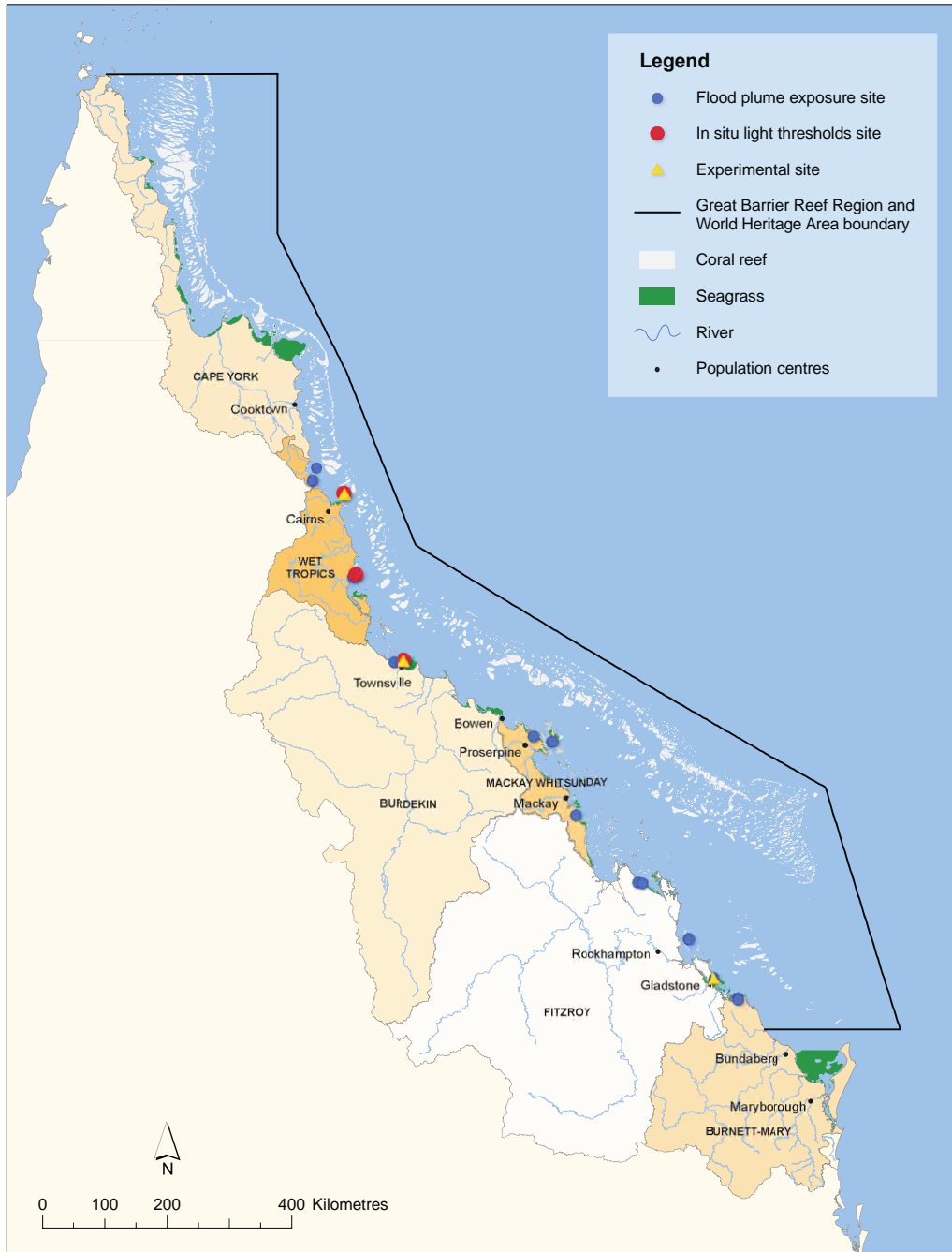


**Figure 8.**    Map of the seagrass sampling sites referred to in this report.

**Table 5.**        Sites referred to in this report (also see Figure 8).

| NRM region (*Board*) | Location | Site | | Seagrass community type |
|---|---|---|---|---|
| **Wet Tropics** (*Terrain NRM*) | Low Isles *reef* | LI1* | Low Isles | *H.ovalis/H.uninervis* |
| | | LI2^ | Low Isles | *H.ovalis/H.uninervis* |
| | Cairns | YP1* | Yule Point | *H. uninervis* with *H. ovalis* |
| | *coastal* | YP2* | Yule Point | *H. uninervis* with *H. ovalis* |
| | Green Island | GI1* | Green Island | *C. rotundata/T. hemprichii* with *H. uninervis/H. ovalis* |
| | | GI2* | Green Island | *C. rotundata/T. hemprichii* with *H. uninervis/H. ovalis* |
| | *reef* | GI3^ | Green Island | *C. rotundata/ H. uninervis/C.serrulata/S.isoetifolium* |
| | Mission Beach | LB1* | Lugger Bay | *H. uninervis* |
| | *coastal* | LB2* | Lugger Bay | *H. uninervis* |
| | Dunk Island | DI1* | Dunk Island | *H. uninervis* with *T. hemprichii/ C. rotundata* |
| | | DI2* | Dunk Island | *H. uninervis* with *T. hemprichii/ C. rotundata* |
| | *reef* | DI3^ | Dunk Island | *H. uninervis / H. ovalis/H.decipiens/C. serrulata* |
| **Burdekin** (*NQ Dry Tropics*) | Magnetic island *reef* | MI1* | Picnic Bay | *H. uninervis* with *H. ovalis* & *Zostera/T. hemprichii* |
| | | MI2* | Cockle Bay | *C. serrulata/ H. uninervis* with *T. hemprichii/H. ovalis* |
| | | MI3^ | Picnic Bay | *H. uninervis* with *H. ovalis* & *Zostera/T. hemprichii* |
| | Townsville | SB1* | Shelley Beach | *H. uninervis* with *H. ovalis* |
| | *coastal* | BB1* | Bushland Beach | *H. uninervis* with *H. ovalis* |
| **Mackay Whitsunday** (*Reef Catchments*) | Whitsundays *coastal* | PI2* | Pioneer Bay | *H. uninervis/ Zostera* with *H. ovalis* |
| | | PI3* | Pioneer Bay | *H. uninervis* with *Zostera/H. ovalis* |
| | Whitsundays | HM1* | Hamilton Island | *H. uninervis* with *H. ovalis* |
| | *reef* | HM2* | Hamilton Island | *Z. muelleri* with *H. ovalis/H. uninervis* |
| | Mackay | SI1* | Sarina Inlet | *Z. muelleri* with *H. ovalis* (*H. uninervis*) |
| | *estuarine* | SI2* | Sarina Inlet | *Z. muelleri* with *H. ovalis* (*H. uninervis*) |
| **Fitzroy** (*Fitzroy Basin Association*) | Shoalwater Bay *coastal* | RC1* | Ross Creek | *Z. muelleri* with *H. ovalis* |
| | | WH1* | Wheelans Hut | *Z. muelleri* with *H. ovalis* |
| | Keppel Islands | GK1* | Great Keppel Is. | *H. uninervis* with *H. ovalis* |
| | *reef* | GK2* | Great Keppel Is. | *H. uninervis* with *H. ovalis* |
| | Gladstone Harbour | GH1* | Gladstone Hbr | *Z. muelleri* with *H. ovalis* |
| | | GH2* | Gladstone Hbr | *Z. muelleri* with *H. ovalis* |
| | *estuarine* | | | |

.
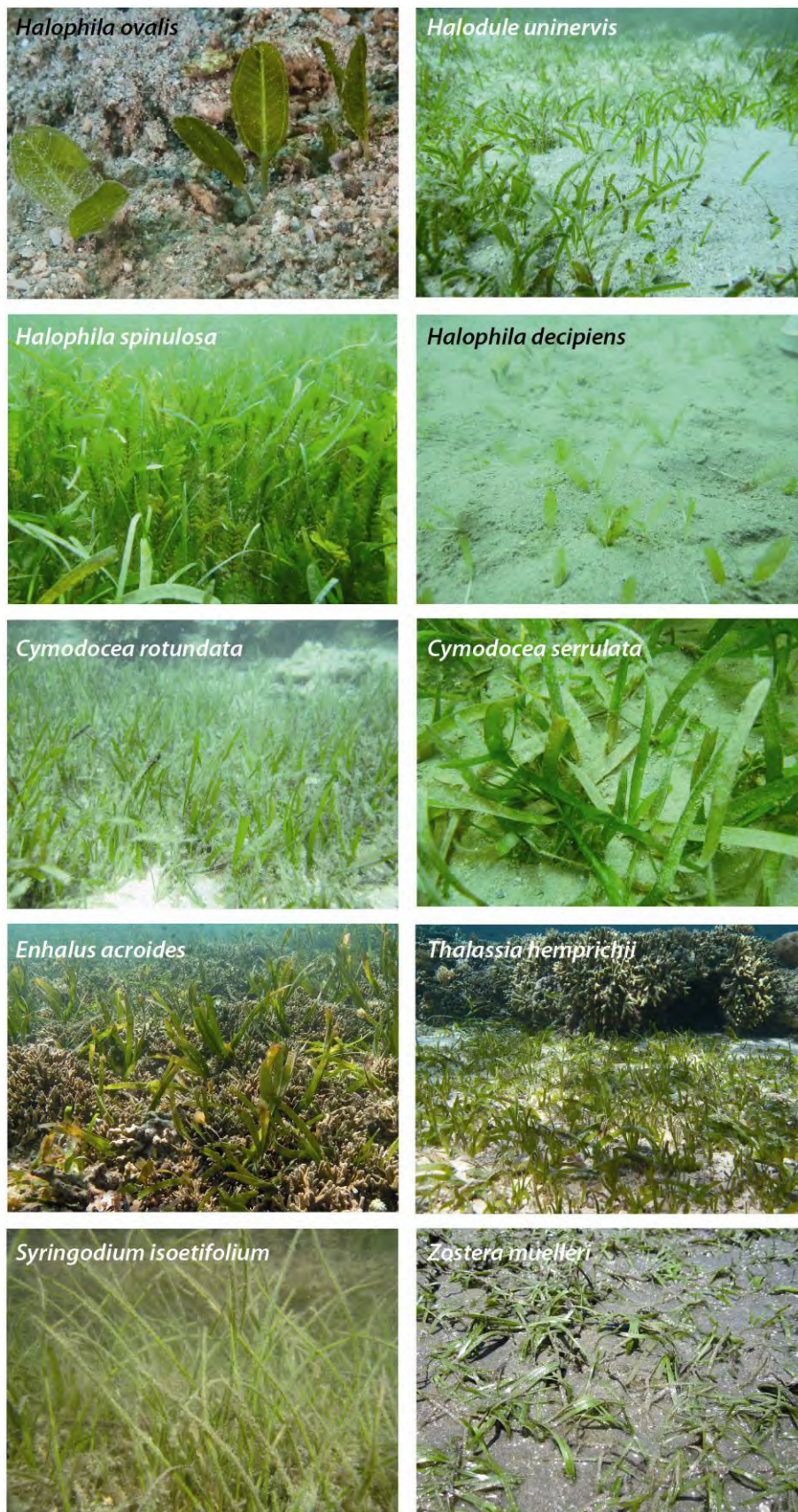
**Figure 9.** Species described in this report and which are common throughout the Great Barrier Reef.

water types (Primary, Secondary, and Tertiary). The Primary waters corresponded to CC1 to CC4 and were defined by low salinity and high turbidity; the Secondary corresponded to CC5 and were defined by intermediate salinity and a region where reduced TSS due to sedimentation and high nutrient availability prompt primary production; and the Tertiary waters corresponded to CC6 and were defined as being the transition between secondary waters and marine ambient. In this study we chose to keep the original 6 colour classes of Álvarez-Romero et al. (2013) as we wanted to preserve the subscale information in the coastal plume waters available through the mapping of CC1, CC2, CC3 and CC4.

**Table 6.** Terminology and characteristics of ocean colour classes measured during wet season conditions in the Great Barrier Reef.

| Water Type | Ocean colour | Characteristics |
|---|---|---|
| "Primary" Water Types | CC1, CC2, CC3, CC4<br><br>Four distinct colour classes within the primary waters reflecting changes in light attenuating properties (high TSS – high Chl) | Very high turbidity, low salinity (0 to 10; Devlin et al., 2010), very high values of CDOM and Total Suspended Sediment (TSS). Turbidity levels limit light penetration in Primary waters, inhibiting primary production and limiting Chl-a concentration. |
| "Secondary" Water Types | S, CC5 | Characterised by intermediate salinity, elevated CDOM concentrations, and reduced TSS due to sedimentation (Bainbridge et al., 2012). Middle salinity range: 10 to 25; Devlin et al., 2012b). Phytoplankton growth is prompted by the increased light (due to lower TSS) and high nutrient availability (delivered by the river plume). |
| "Tertiary" Water Types | T (Ft), CC6 | Occupies the external region of the river plume. It exhibits no or low TSS associated with the river plume, but above-ambient concentrations of Chl-a and CDOM. Described as the transition between Secondary water and marine ambient water, presenting salinity lower than the former one (typically defined by salinity ≥ 35; e.g., Pinet, 2000). |

The supervised classification of Álvarez-Romero et al. (2013) was used to classify 10 years of MODIS images (from 2005 to 2014, focused on the summer wet season i.e., December to April inclusive), and to produce daily 6CC water type maps over the 2005 (i.e., December 2004 to April 2005) to 2014 (i.e., December 2013 to April 2014) wet seasons. Coral reefs and Land areas were masked out and weekly 6CC composites (22 composites per wet season) were created to minimize the amount of area without data per image due to masking of dense cloud cover, common during the wet season (Brodie et al., 2010), and intense sun glint (Álvarez-Romero et al., 2013).

Comparisons between weekly six-colour class (CC) composites and in-situ water quality measurements collected during the wet seasons 2004 to 2014 as part of the GBR Marine Monitoring Program were performed. The design of the flood monitoring program is detailed in Devlin et al. (2012a) with QC/QA procedures documented in GBRMPA (2014). In-situ values were assigned to weekly CC (1 to 6) based on their location. Data extraction was performed using extract in the raster package (Hijmans 2014) with bilinear method in R 3.1 (R Development Core Team, 2013).

## Seagrass abundance

Seagrass monitoring occurred at 23 locations from Archer Point in the north of the GBR to Urangan (Figure 8). Field survey methodology is detailed in GBRMPA (2014) and McKenzie et al (2013). Sites were monitored for seagrass abundance (reported as a % annual cover) and

species composition. All sites have been sampled up to and including 2013, however we focused on a 2-year period (2005–2007) of below median rainfall followed by a five-year period (2008–2012) of above-median rainfall and flooding to test seagrass health responses to river plume exposure in the GBR.

Different seagrass measurements were selected to describe changes in seagrass health in the GBR: the annual abundance (% cover), the mean annual and multi-annual (2004-2012) seagrass abundance and the changes in seagrass abundance relative to a baseline value ([Δ(Aref]). This latter seagrass measurement was calculated as the average change between a baseline value (calculated from the average abundance across the dry period 2003 to 2007) and annual abundance for each wet year (2008 – 2012, inclusive).

## Relationships between River plume exposure and seagrass health

River plume exposure at the seagrass sites was extracted from each weekly 6CC composites using ArcMap Spatial Analyst (ESRI, 2010). A 3km buffer was created around each seagrass monitoring sites and the ArcMap Zonal Statistic tool was used to extract CC values (1 to 6) of each pixel of the buffered area. Each seagrass monitoring site was thus assigned a weekly CC value (22 values per wet season between 2005 and 2014) based on the majority CC value extracted. The total annual and multi-annual (2005-2012) frequency of exposure of each seagrass site to each CC (1 to 6) was finally computed (in week per wet season) and normalised (0-1) by the maximum number of satellite information available per site (22 being the maximum). Missing satellite information is related to cloud cover, or masked reefs and lands. We didn't consider the 2013 and 2014 weekly data in the multi-annual frequency calculation as seagrass health measurements were not available after 2012.

The susceptibility of the GBR seagrass meadows to river plume exposure were investigated by correlating seagrass health measurements with the river plume exposure measurements at the habitat scale (i.e., estuarine intertidal, coastal intertidal, reef subtidal, reef intertidal). Mean river plume exposure at each seagrass habitat was calculated and river plume exposure thresholds indicative of significant decline in seagrass abundance (50%) were developed.

# Results

## Great Barrier Reef River flow

In the 5 years from 2007 to 2011, the total annual discharge for 35 rivers distributed throughout the GBR exceeded the total annual long-term median calculated for the hydrological year (i.e., 1 October to 30 September) covering the period 1970–2000. Exceedances ranged from 66 to 620% higher than the long-term median flow (Figure 2). Record flow conditions were measured for 2010–2011, where a combination of three cyclones produced record flows in nearly all GBR rivers, particularly in the southern half of the GBR. Flows in the latter 3 years have been dominated by large floods out of the southern rivers, particularly the large dry tropic rivers, Burdekin and Fitzroy, and by the southern influence of flow from the Burnett-Mary (Devlin et al. 2012a; da Silva et al. 2013).

## Correlation between water quality and satellite data

Decreased mean salinity and depth values from CC6 to CC1 underlined the spatial distribution of the six river plume CC i.e., a relative offshore location for CC6 (Sal = 32.44 ± 4.70 and depth 21.30 ± 6.39 m) in comparison to inshore distribution for CC1 and 2 (Table 7). Most WQ parameters including $K_d$(PAR), TSS, DIN and DIP, followed decreasing trends from the CC1 ($K_d$(PAR)= 1.10 ±0.97 m$^{-1}$; TSS= 51.03±74.41 mg L$^{-1}$; DIN=5.10±4.14 mg L$^{-1}$ and DIP=0.73±0.80 mg L$^{-1}$) to the CC6 ($K_d$(PAR)= 0.20±0.26m$^{-1}$; TSS= 8.37± 8.29mg L$^{-1}$; DIN 1.68±2.23 mg L$^{-1}$ and DIP 0.17±0.14 mg L$^{-1}$). Higher mean WQ concentrations were also generally measured in plume waters (CC1 to CC5) than in marine waters. Analyses of relative

proportion of optically active components [CDOM, Chl-a and TSS] across each 6CC suggest that CC3, CC4 and CC5 are chl-a dominated and with highest/lowest light availability in comparison to CC1-CC2/CC6 (Figure 10).

**Table 7.** Mean and standard deviation (stdv) of in-situ WQ measurements in each plume colour class. TSS in mg $L^{-1}$, CDOM in $m^{-1}$, Chl-a in µg $L^{-1}$, $K_d$(PAR) in $m^{-1}$, DIN in µg $L^{-1}$ and DIP in µg $L^{-1}$

|  |  | Depth | Sal. | TSS | CDOM | Chl-a | Kd(PAR) | DIN | DIP |
|---|---|---|---|---|---|---|---|---|---|
| CC1 | mean | 11.45 | 17.63 | 51.03 | 1.73 | 2.91 | 1.1 | 5.1 | 0.73 |
|  | stdv | 7.86 | 11.19 | 74.41 | 1.28 | 4.16 | 0.97 | 4.14 | 0.8 |
| CC2 | mean | 8.89 | 21.43 | 19.04 | 1.39 | 2.1 | 0.87 | 4.23 | 0.57 |
|  | stdv | 8 | 9.76 | 36.87 | 1.32 | 3.07 | 0.66 | 3.56 | 0.67 |
| CC3 | mean | 13.89 | 27.64 | 17.22 | 0.71 | 2.28 | 0.71 | 3.04 | 0.41 |
|  | stdv | 16.68 | 7.16 | 16.27 | 0.77 | 2.79 | 0.49 | 2.97 | 0.36 |
| CC4 | mean | 15.79 | 26.82 | 10.46 | 0.62 | 1.48 | 0.68 | 2.71 | 0.33 |
|  | stdv | 18.34 | 7.06 | 10.47 | 0.69 | 1.78 | 0.54 | 2.26 | 0.24 |
| CC5 | mean | 17.08 | 30.27 | 8.44 | 0.36 | 0.88 | 0.36 | 1.87 | 0.28 |
|  | stdv | 15.62 | 6.28 | 8.21 | 0.49 | 0.88 | 0.27 | 1.99 | 0.24 |
| CC6 | mean | 21.3 | 32.44 | 8.37 | 0.25 | 0.5 | 0.2 | 1.68 | 0.24 |
|  | stdv | 6.39 | 4.7 | 8.29 | 0.26 | 0.61 | 0.26 | 2.23 | 0.18 |
| Marine waters | mean | 25 | 32.86 | 2.61 | 1.82 | 0.38 | NA | 1.9 | 0.17 |
|  | stdv | 11.31 | 1.72 | 1.99 | NA | 0.93 | NA | 1.03 | 0.14 |
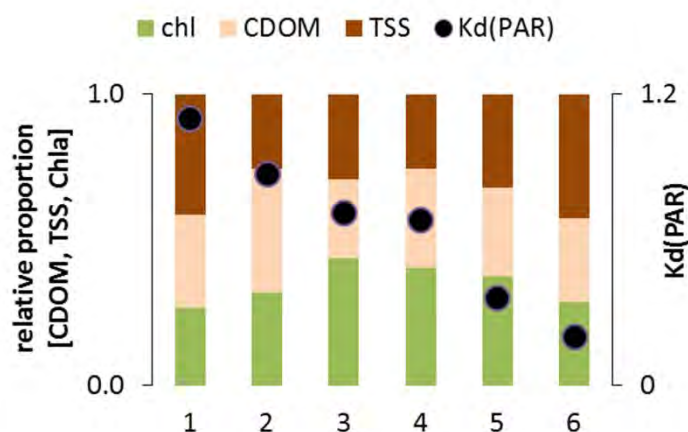


**Figure 10.** Relative proportion of optically active components [CDOM, Chl-a and TSS] across the 6 colour classes.

## Relationships between River plume exposure and seagrass health at the habitat scale

Relationships were investigated by correlating seagrass health measurements with the river plume exposure measurements at the habitat scale (i.e., estuarine intertidal, coastal intertidal, reef subtidal, reef intertidal).

Seagrass species richness differed between habitats across the inshore GBR, with higher number of species at reef than coastal or estuarine habitats. Reef habitats are typically dominated by H. uninervis, T. hemprichii, Cymodocea spp. and H. ovalis, coastal were dominated by H. uninervis and H. ovalis, and estuary dominated by Z. muelleri. (Table 5 Figure 9). The multi-annual seagrass abundance (reported as % cover) at each of the inshore GBR habitats were 10% and 13% for the estuarine and coastal intertidal meadows and 15% and 13% for the reef subtidal and intertidal meadows (Figure 11). These habitats changed by -42.6, -43.5, -53.7 and -49.8% respectively from 2007 to 2012.

Total number of satellite information available (i.e. not classified as clouds, reef or land) over the 2005-2012 period was function of the habitat considered, but relatively consistent across the habitats Figure 11a). The annual and multi-annual frequency of exposure to plume waters ($F_{cc}$) were normalised against the number of satellite information available per habitat (Figure 11b and c).

Over 2005-2014, exposure to the most turbid river plume waters (CC1, CC2 and CC3) decreased from the estuarine intertidal, to the coastal intertidal and to the reef subtidal habitats (Figure 11b, $F_{CC1}$= 10%, 5% and 0% of the time we have satellite information). Coastal intertidal and both subtidal and intertidal reef habitats were predominantly exposed to CC4 ($F_{CC4}$ =17%, 15% and 22% of the time, respectively) and CC5 ($F_{CC5}$=38%, 72% and 48%), though exposure of coastal intertidal sites to CC3 was also about 15% of the time.

All seagrass habitats have been declining since 2009 (Figure 11c) with a maximum decline recorded for the reef subtidal habitats ($\Delta$(Aref) =-54). Influence of cyclone Yasi (late January 2011) was registered in the frequency of exposure to CC1 to CC5 faced by all seagrass habitats (Figure 11c, "Y"). Nevertheless, most seagrass meadows were already at low abundance in 2009 or 2010 and the wet season 2011 wasn't characterised by further decrease in seagrass abundances.
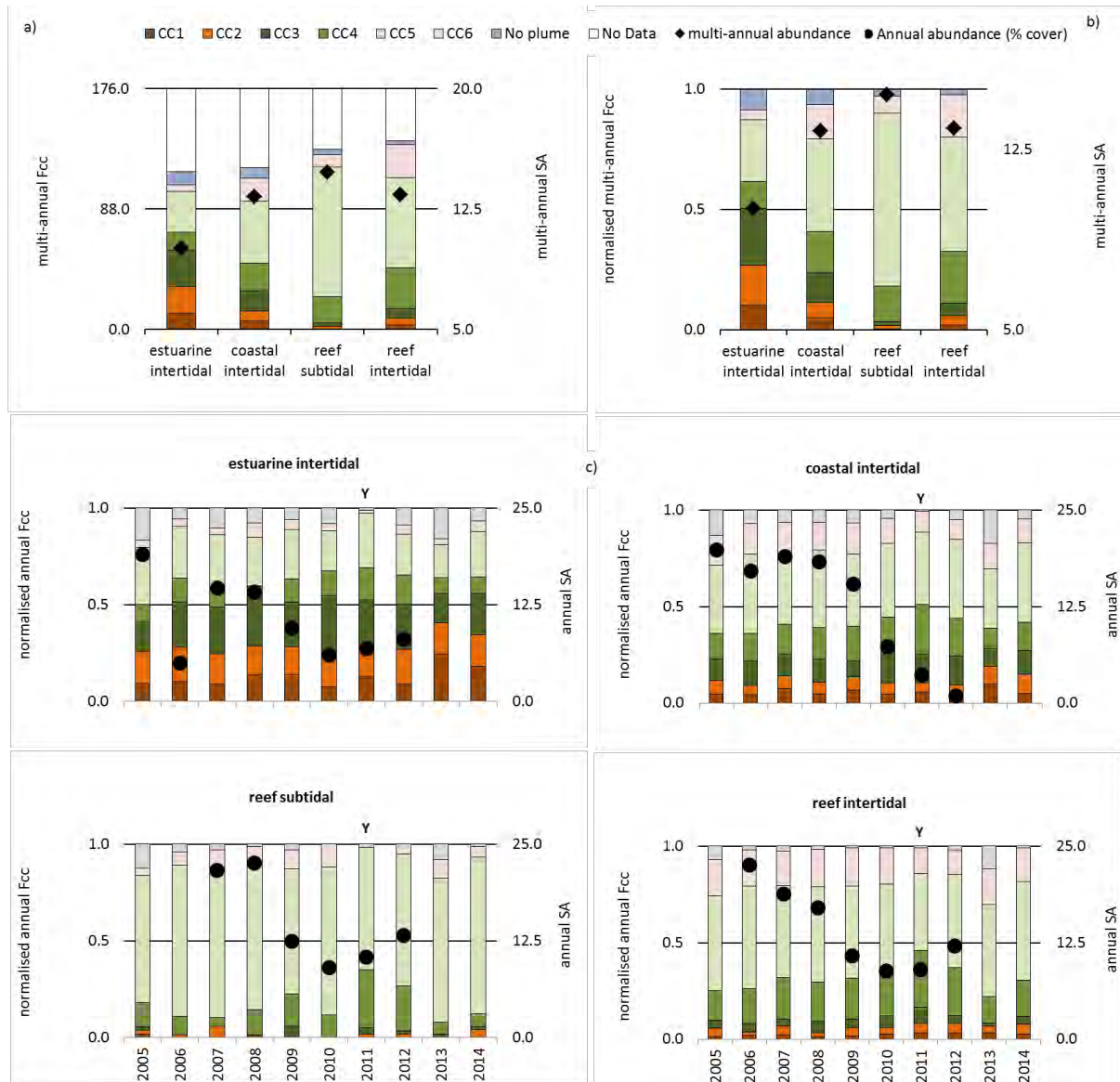
**Figure 11.** Plume conditions at each different seagrass habitats measured via the frequency of exposure to the plume colour classes ($F_{cc}$). Multi-annual (2005-12) seagrass abundance (SA, % cover) against (a) multi-annual $F_{cc}$ and b) multi-annual normalised $F_{cc}$. c) Annual seagrass abundance (SA) against the annual normalised $F_{cc}$ (% cover).

Linear correlation analyses were performed between the seagrass and plume measurements in order to determine if changes in SA could be determined by the frequency of exposure to a specific river plume CC or combination of CC. Strong correlation obtained between the annual seagrass abundance and the frequencies of occurrence of river plumes colour classes underlined that fluctuation in seagrass abundance was a response to river plume exposure in each seagrass habitat (Figure 12, Table 8). At the estuarine intertidal sites, 84% of the annual variability in abundance was explained by the total annual exposure to CC1, CC2, CC3 and CC4 (i.e. Primary water type of e.g. Devlin et al., 2013b; Petus et al., 2014b). At the coastal intertidal sites, the total annual exposure to CC2, CC3 and CC4 explained 88% of the variation in abundance. Finally, variations in abundance at the reef subtidal and reef intertidal habitats were mainly explained by the annual exposure to CC4 (34 % but non-significant and 74%, respectively).
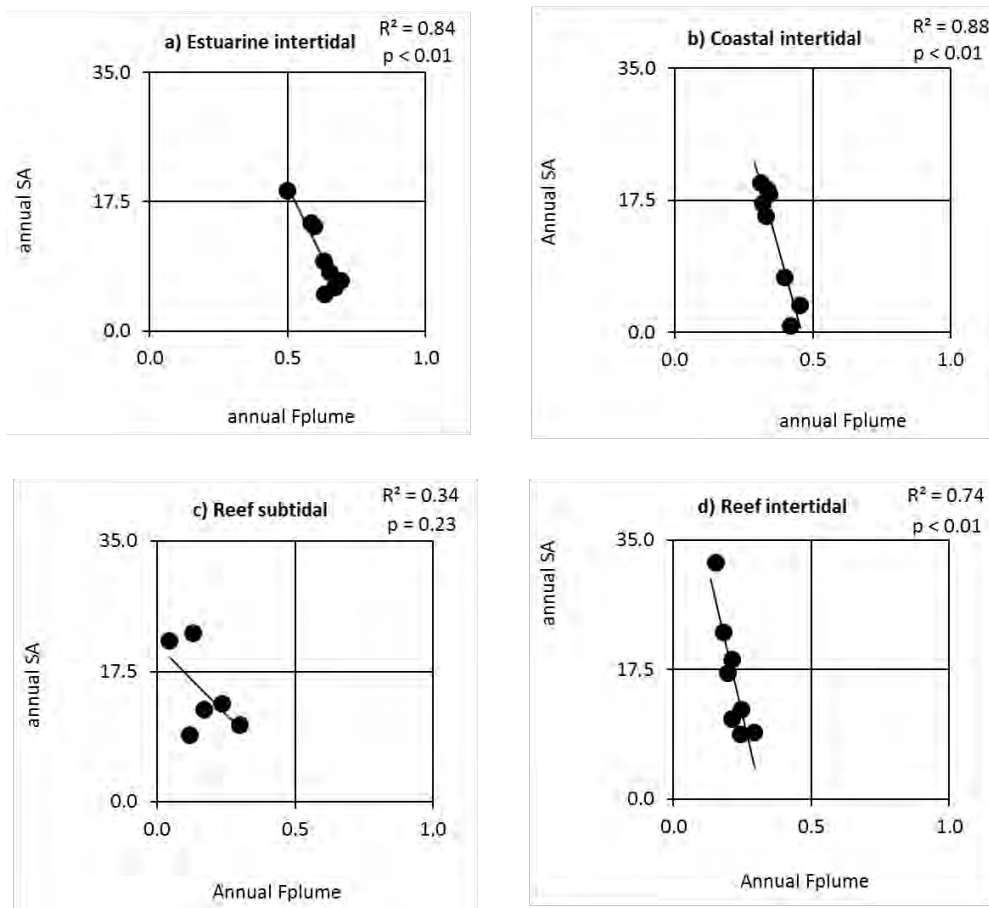
**Figure 12.** Correlation between the normalised annual $F_{plume}$ and annual SA at the four seagrass habitats. $F_{plume}$ is a proxy measurement that represents a colour class or combination of colour classes that relate directly to the plume condition at each seagrass locations.  a) Estuarine intertidal: $F_{plume} = CC1 + CC2 + CC3 + CC4$; b) costal intertidal : $F_{plume} = CC2 + CC3 + CC4$ and reef c) subtidal and d)  intertidal : $F_{plume} = CC4$

**Table 8.** Multi-annual (2005-2012) normalised frequency of exposure to the plume colour classes ($F_{cc}$) and multi-annual seagrass abundance (SA). Seagrass loss greater than 50% in red text and blue text highlights $F_{(plume)}$ in preceding two years.

| Habitat | Annual | Baseline (2005-2007) | 2008 | 2009 | 2010 | 2011 | 2012 | 2013 | 2014 |
|---|---|---|---|---|---|---|---|---|---|
| Estuarine intertidal | F(plume) | 17.12 | **0.60** | **0.63** | 0.67 | 0.69 | 0.65 | 0.64 | 0.64 |
| | SA | 0.57 | 14.16 | 9.43 | **5.91** | 6.76 | 7.97 | | |
| Coastal intertidal | F(plume) | 17.10 | **0.34** | **0.33** | 0.40 | 0.45 | 0.42 | 0.29 | 0.37 |
| | SA | 0.32 | 18.34 | 15.43 | **7.29** | 3.61 | 0.90 | | |
| Reef subtidal | F(plume) | 21.60 | **0.13** | **0.17** | 0.12 | 0.30 | 0.23 | 0.06 | 0.07 |
| | SA | 0.09 | 22.59 | 12.42 | **9.01** | 10.35 | 13.16 | | |
| Reef intertidal | F(plume) | 18.83 | **0.20** | **0.21** | 0.24 | 0.29 | 0.25 | 0.13 | 0.19 |
| | SA | 0.18 | 17.06 | 10.80 | **8.82** | 9.02 | 12.05 | | |

The correlation between colour classes and water quality information across habitat type can be used to develop water quality thresholds. The correlation between the normalised annual Fplume and annual SA were plotted at the four seagrass habitats (Figure 12).  Using a seagrass

threshold of equal or more than 50% loss, the corresponding colour class data was extracted as a frequency value calculated over two consecutive years (Table 8). We used Fplume as a proxy measurement that represents a colour class or combination of colour classes which related directly to the plume condition at that specific habitat and most likely to impact on the seagrass. The corresponding Fplume frequency was extracted at the threshold of 50% loss (Table 8). Then for each habitat, we have extracted a TSS, Chl-a and $K_d$ threshold value that aligned with the Fplume (colour class) frequency (Table 9) based on water quality values in Table 7. Exceedance of these values at the reported frequency in Table 9 would result in a loss of seagrass cover of greater than 50%. Note that all habitats, except the coastal intertidal have shown signs of recovery after 2010 so additional information would be required before thresholds and timing could be developed to predict recovery after event driven losses.

**Table 9.** Water quality occurring in each habitat where seagrass loss was greater than 50% (over 2 consecutive years). These values are analogous to water quality thresholds associated with loss and are relevant to conditions of above average wet season from 2008-2011.

| Habitat | Colour class (CC) 1 – 6 | Frequency | TSS (mg L$^{-1}$) | Chl-a* (ug L$^{-1}$) | $K_d$ (m$^{-1}$) |
|---|---|---|---|---|---|
| Estuarine intertidal | F (CC 1 – 4) | 0.6 | > 10 | > 1.5 | > 0.7 |
| Coastal intertidal | F (CC 2 – 4) | 0.3 | 10 - 20 | 1.5 - 2.1 | 0.7 - 0.9 |
| Reef subtidal | F (CC – 4) | 0.1 | ~ 10 | ~ 1.5 | ~ 0.7 |
| Reef intertidal | F (CC – 4) | 0.2 | ~ 10 | ~ 1.5 | ~ 0.7 |

# Discussion

Increasing severity, frequency and spatial extent (breaking connectivity) of water quality impacts will reduce seagrass resilience and challenge recovery processes. This work examines the correlation between remote sensed measurements correlated with water quality over consecutive wet seasons and the corresponding changes in seagrass meadows in the central Great Barrier Reef. Utilising both remote sensing ocean colour information plus in-situ water quality has provided data over the appropriate spatial and temporal scales to allow the long term comparison of water quality and biological measurements. This analysis has demonstrated that exposure to turbid primary waters (characterised by high TSS, moderate chl and CDOM), can predict seagrass abundance as well as seagrass responses to event-based conditions. Seagrass areas exposed to high frequency of primary turbid waters (during consecutive wet seasons) were characterised by low cover and low biomass, with consistently low measures of seagrass health. Seagrass areas with little or no exposure to primary turbid waters tend to be relatively intact with high biomass and little annual change. The seagrass areas that were exposed to primary water (CC 1 - 4) with high annual biomass prior to 2007 declined and had the greatest annual change. Intermittent exposure to reduced water quality can result in relatively high biomass meadows but slight change in water quality can shift the balance in these seagrass communities. Large-scale water quality mapping can help define the type of seagrass communities and identify the main water types, which shape and drive seagrass response. Thus long term water quality data, both in-situ and through remote sensing can provide measures of risk relative to the seagrass community health, including measures of seagrass biomass, cover and species.

Changes in seagrass abundance in the GBR have been previously documented (McKenzie et al., 2012, 2013). However, these studies included limited environmental data, making interpretation of seagrass changes across the GBR difficult. Repeated and extended above average wet seasons were monitored in the GBR between 2008 and 2012. The present study, which combines MODIS measurements and ecological information across years, extended the case

study of Petus et al. (2014a). It provided further information about interactions between seagrass health (as measured by the % cover of seagrasses) and reduced WQ levels (caused by the increased exposure to River plume waters) at a greater spatial scale.

The decline in meadow % cover in the GBR over the monitored period confirm results of Petus et al. (2014a) and indicates an advanced state of exposure to stressful conditions throughout the GBR from 2005/6 to 2011 (Collier et al., 2012; McKenzie et al., 2012; Pollard and Greenway, 2013; Rasheed et al., 2014). These widespread impacts can be attributed largely to the extreme weather conditions occurring during the monitoring wet years in which above-average rainfall exposed meadows to turbid plume waters that reduce light penetration and increase exposure to sourced pollutants. Seagrass meadows of the GBR typically undergo a period of senescence in the wet season when meadow abundance and productivity is reduced (McKenzie et al., 2012). The meadows then recover during the dry season as water warms and light increases. However, during the study period, the repeated and extended above average wet seasons were followed by shortened dry season recovery periods (McKenzie et al., 2013).

The limited response in of the seagrasses to the small to moderate flooding events between 2005 and 2008 demonstrates that seagrasses possess an inherent resistance to moderate reductions in water quality. However, the influence of the cumulative impact of the river floods was evident in the % cover records with a maximum decrease recorded in 2009 for all seagrass habitats.

**For further information see:**

Petus, C., Collier, C.J., Devlin, M., Rasheed, M., and McKenna, S. (2014). Using MODIS data for understanding changes in seagrass meadow health: a case study in the Great Barrier Reef (Australia). Mar Environ Res.

Devlin, M., Petus, C., Collier, C., Mckenzie, L., Waycott, M., and Teixera da Silva, E. (In Prep). Monitoring large scale water quality impacts on seagrass communities in the Great Barrier Reef, through integration of remote sensed data, in-situ water quality and long term seagrass monitoring data

**Key finding:** Impacts of water quality in seagrass meadows are habitat dependent. Water Quality thresholds derived from correlations between ocean colour class and in-situ water quality data are specific to the four different habitats of seagrass in the GBR. Thresholds developed correlated to periods of extreme weather, and can be used to predict extended response (loss of cover > 50%) of seagrass to elevated wet season conditions.

**Knowledge gap:** Establish thresholds that link to recovery and resilience of seagrass meadow.

# 4 Impacts of light, nutrients and salinity on seagrass health

Flood plume mapping showed that seagrass loss could be predicted from the frequency of exposure to flood plumes. These flood plumes have high light attenuation properties ($K_d$) due to high turbidity, chlorophyll a and CDOM concentrations, as well as being elevated in nutrients and having low salinity. Exposure to flood plumes affected seagrass meadow abundance and species composition in a manner that could be predicted from the flood plume data.

We incrementally tested the responses of seagrasses to low light, elevated nutrients and low salinity. This two-year project was necessarily limited in the number of interactions that could be tested. We first tested sensitivity to salinity because there was no previous information on salinity thresholds for most species making it difficult to assess importance, or appropriate salinities to test in interactive experiments. There was broad salinity tolerance, which meant that this was not considered the highest priority for follow-up experimental work. There was a larger focus on light in these studies because of the finding that light was highly correlated to changes in seagrass abundance during recent loss in the GBR. Nutrients were tested in an experiment with light (light x nutrients) at relatively broad levels.

## Light

Light reaching seagrass leaves is used in photosynthesis and the energy arising from this is basis for seagrass growth. Light was previously known to be very important for seagrass meadow productivity and biomass and in many locations around the world insufficient light availability caused by declining water quality (from turbidity, plankton blooms or epiphytes) has been implicated in declining meadow health (Cambridge et al., 1986; Dennison et al., 1993; Longstaff and Dennison, 1999; Preen et al., 1995; Waycott et al., 2009; Waycott et al., 2005). Our conceptual understanding of seagrass responses to low light was quite well developed. None-the-less, there were considerable knowledge gaps that prevented the management of seagrasses on the basis of their light requirements. Firstly, we needed to know what aspects of seagrass biology were the most sensitive to changes in light and could be used as indicators that seagrass meadows were under light stress. Furthermore, there were no data on light levels (thresholds) leading to changes in seagrass meadow abundance or growth.

Using in-situ seagrass abundance data and long-term light monitoring data, we identified reasonably good correlations between changes in seagrass abundance and light at sites where seagrass loss had occurred (c) (Collier et al., 2012). This demonstrated the important role of low light as a driver of seagrass abundance in the GBR and provided a basis for developing light thresholds, that could be applicable to water quality guidelines for seagrasses of the northern GBR. The role of light in these seagrass losses was later verified using experimental work (d) that found similar light thresholds leading to 50% loss in situ and in aquaria (discussed further below) indicating that in situ losses where light thresholds were breached were not just correlative, but causative. Furthermore, in a recent Risk Assessment, suspended sediments and their light attenuating properties were identified as the greatest risk to GBR seagrasses (Brodie et al., 2013a). For these reasons we placed a large focus on understanding indicators of light stress and deriving light thresholds, while also exploring interactive effects of nutrients and seasonal changes in water temperature.
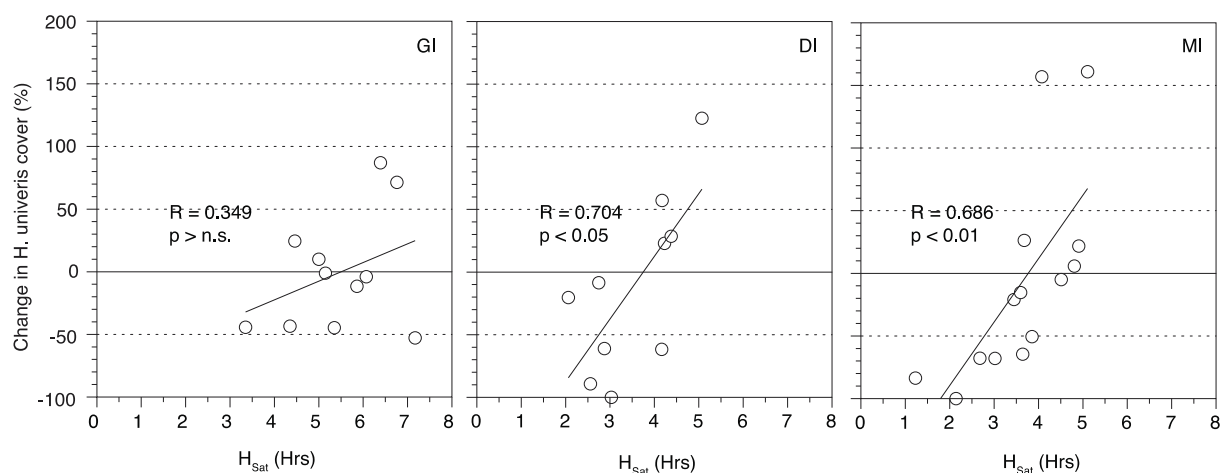
**Figure 13.** Change in the abundance of the seagrass Halodule uninervis and the hours of light saturated photosynthesis at three sites including Green Island (GI, stable meadow), Dunk Island (DI, complete seagrass loss in 2011), and Magnetic Island (MI, complete seagrass loss in 2011) from 2008 to 2011. Each point represents change in seagrass percent cover (33 quadrats in 3 replicate transects) over ~three months. Light at seagrass canopy height was continuously monitored for the same period, with the hours of light saturated photosynthesis calculated from incoming light and published values on saturating light levels ($E_k$). From Collier et al 2012.

Major knowledge gaps for GBR seagrasses included: (1) light thresholds, (2) robust indicators of light stress, (3) species differences, and (4) role of low light relative to other flood plume impacts. In this study we tested impacts of light on seagrasses in 6 different sub-projects, including flood plume mapping (a), a desktop analysis of in situ light and seagrass abundance data (c), three experiments (d, f and g) and a desk-top analysis of indicators (e). These approaches all explored different aspects of seagrass light responses and ranged in scale from rapidly responding physiological indicators (f,g) through to broad scale analysis of changing water quality (a). Light levels that were tested experimentally largely depended on incoming irradiance (time of year, clouds etc.).

This study has confirmed that:

- Seagrasses in the GBR are highly sensitive to reductions in light.
- Most seagrass meadows in the inshore GBR are light limited.
- There are species differences in sensitivity to low light and these thresholds were identified (see below).
- Seagrass loss in the GBR between 2009 and 2011 was highly correlated to light. Comparison of the light levels occurring during these events and the thresholds derived in aquaria experiments confirms that light levels exceeded thresholds and confirmed that low light availability was most likely a primary cause of loss.
- The amount of light required for seagrass growth or to maintain abundance is affected by water temperature.
- Seagrasses show sub-lethal indications of light stress that are measurable and relate to specific levels of light stress.

More information can be found in the scientific papers and publications arising from this work:

Collier, C.J., Waycott, M., and McKenzie, L.J. (2012). Light thresholds derived from seagrass loss in the coastal zone of the northern Great Barrier Reef, Australia. Ecological Indicators 23, 211-219.

Collier, C.J., Langlois, L., Adams, M., O'Brien, K., Maxwell, P. Waycott, M. and McKenzie, L., (In Prep). Seagrass protection guidleines developed from experimental light response curves.

McMahon, K.M., Collier, C.J., and Lavery, P.S. (2013). Identifying robust bioindicators of light stress in seagrasses: A review. Ecological Indicators 30, 7-15.

**Table 10.** Summary of findings in relation to the environmental variable, light

| Approach | Environmental parameter | Range tested | Finding |
|---|---|---|---|
| a. Flood plume exposure | Primary and secondary water types. Contains: high turbidity, high chlorophyll *a* high CDOM all attenuating light | Primary and secondary water types | Meadows exposed to moderate frequency of both primary (Fp) and secondary (Fs) waters are the most sensitive and both water types have high light attenuating properties (0.2 – 0.6) |
| c. Event-based *in situ* seagrass loss/*in situ* light loggers | Daily light (mol m$^{-2}$ d$^{-1}$) | 2.3 – 14.2 mol m$^{-2}$ d$^{-1}$ | • In-situ changes in seagrass abundance (*H. uninervis*-dominated communities) were correlated with *in-situ* light |
|  | Frequency of low light days | % days <14 mol m$^{-2}$ d$^{-1}$ | • This relationship enabled identification of light thresholds for >50% loss |
|  | Hsat | 3.6 – 10.56 hrs | • Loss of Halodule-dominated meadows occurred at 4 mol m$^{-2}$ d$^{-1}$ over 3 months |
| f, Light experiment | Daily light (mol m$^{-2}$ d$^{-1}$) | 2.8 – 11.1 mol m$^{-2}$ d$^{-1}$ | Sub-lethal (early warning) indicators tested (prior to shoot loss) |
|  | Percent of surface light (%SI) | 10 – 40% | C/N, $\delta^{13}$C and rhizome carbohydrates were sensitive to light but: |
| g. Light x nutrients | Daily light (mol m$^{-2}$ d$^{-1}$) | 2.2 – 8.7 mol m$^{-2}$ d$^{-1}$ | • C/N only sensitive at low-moderate nutrient concentrations |
|  | Percent of surface light (%SI) | 10 – 40% | • $\delta^{13}$C and rhizome carbohydrates were less sensitive to nutrients, and sensitivity was species dependant |
| d. Light x temperature | Daily light (mol m$^{-2}$ d$^{-1}$) | 0 – 23 mol m$^{-2}$ d$^{-1}$ | • Shoot loss and growth reductions at low light (0 – 10% SI) over 14 weeks |
|  | Percent of surface light (%SI) | 0 – 70% | • Time to complete seagrass measured: Ho<Zm<Cs<Hu<br>• Light thresholds developed e.g. 3.8 mol m$^{-2}$ d$^{-1}$ for 14 weeks leads to 50% loss of seagrass<br>• Experimental light thresholds were comparable to in-situ thresholds (*H. uninervis, Z. muelleri*) |

## Nutrients

The effect of elevated nutrients on seagrasses was directly tested in a light x nutrients experiment (g). The range tested was 0.8 to 40.4 µmol DIN ($NH_4^+$ and $NO_3^-$), and 0.14, 0.18, and 0.28 µmol filterable reactive phosphate (FRP) for 8 weeks. The highest DIN level tested was far in excess of concentrations measured within plumes of the in-shore GBR, however DIP values were around average for flood plumes (Devlin and Brodie, 2005). The high nutrient treatment did not affect seagrass growth or biomass after eight weeks exposure.

Typically, to identify nutrient limitation, nutrients are added and a positive change in growth or biomass would indicate that the seagrass had been nutrient limited (e.g. Udy et al., 1999). There was no significant effect of these nutrient levels on growth or biomass in this experiment, and this probably occurred because the duration of the experiment (8 weeks) was insufficient to induce plant and meadow-scale changes. This was intentional, as the sub-lethal indicators were the target of this work. There was also no evidence for nutrient toxicity as necrosis from nutrient toxicity is visible after just 5 days (Van der Heide et al., 2008), therefore DIN concentrations that exceed current flood plume concentrations are not likely to be directly hazardous for seagrass plant health.

There was, however, a large change in the concentration of carbon and nitrogen in the leaf tissue of all three species in response to nutrient enrichment (Figure 14) as well as the concentration of storage carbohydrate reserves in the rhizomes. This highlights that seagrasses are sensitive to water quality, in particular, elevated nutrients, and that these physiological changes (C/N or carbohydrates) may be useful as indicators of changing water quality, such as nutrient concentration (see next section).
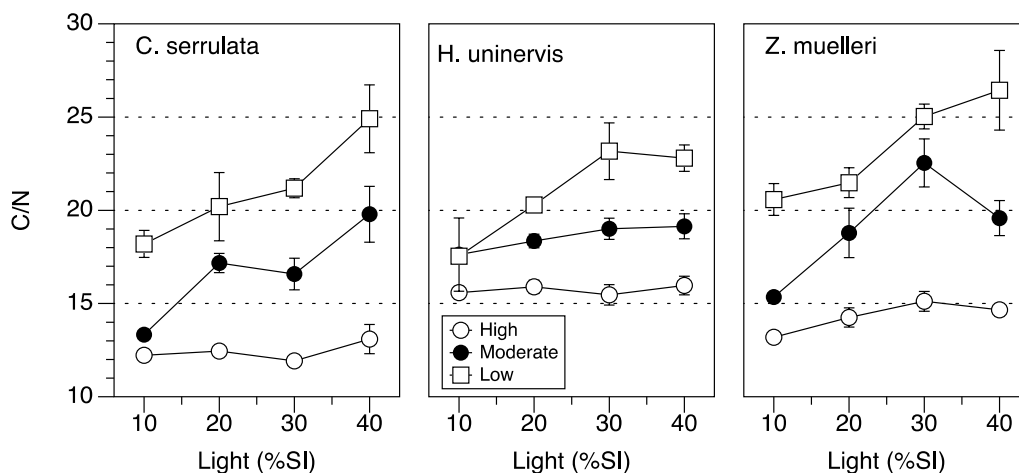


**Figure 14.** Tissue nutrient ratio of carbon (C) and nitrogen (N) at light levels ranging from 0-40% surface light at low (white square), moderate (black circle) and high nutrient (white circle) concentrations after 7 weeks in three seagrass species. C/N was highly sensitive to nutrient concentration and also increased with light, but not at high nutrient concentrations. Adapted from Collier et al, In Prep.

The effects of nutrients on other ecological processes remain a priority concern (Figure 15). For example, nutrient enrichment can lead to epiphytic and macroalgal overgrowth, and these are likely to reach even higher abundances in the absence of grazing pressure (Unsworth et al., Subm). Nutrient enrichment also triggers blooms of phytoplankton (detected as chlorophyll a), and elevated chlorophyll a concentrations in flood plumes contribute to the high light attenuating properties of inshore water. Not only does this reduce total light penetration to seagrasses, but the wavelengths absorbed by chlorophyllous phytoplankton are similar to that required by seagrasses (Larkum et al., 2006). The spectral shift caused by "green water" could affect the usability of the incoming irradiance. The ecological flow-on effects of nutrient enrichment including epiphytic over-growth and top-down grazer effects as well as the effects of spectral shifts on light requirements and light thresholds remain critical knowledge gaps.
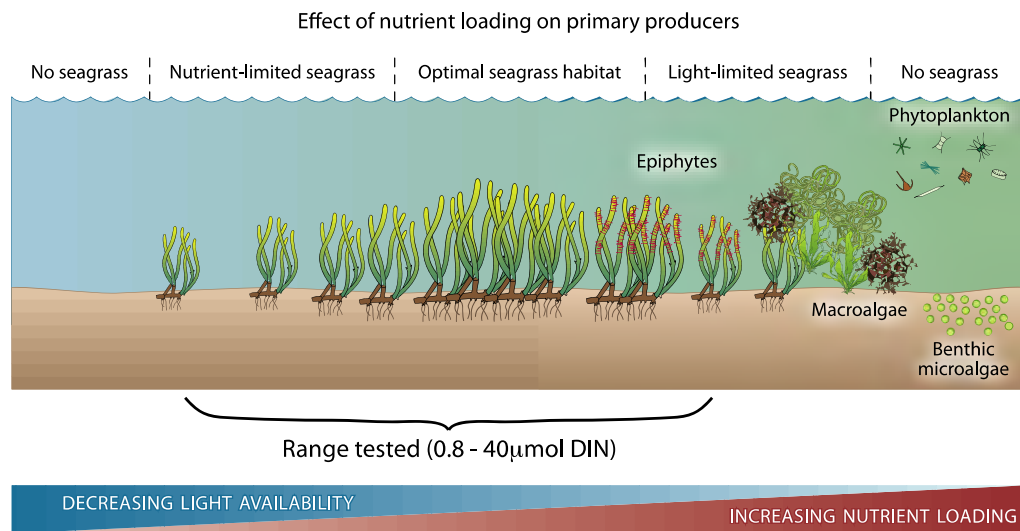
**Figure 15.** Nutrient response model for benthic primary producers showing the range in DIN ($NH_4^{++}$ $NO_3^-$) tested in this study over the short-term (8 weeks). Plants were likely to be nutrient limited in the low (0.5 umol) and nutrient replete in the high nutrient treatment (40 umol).

**Key finding**: Water column nitrogen concentrations in excess of flood plume concentrations do not negatively affect seagrasses in a direct manner.

**Knowledge gap:** Ecosystem-scale responses to elevated nutrients.

## Salinity

The effect of low salinity on Indo-Pacific seagrasses was not known prior to this project. This made it difficult to assess the role of low salinity in flood-related loss of seagrass. We tested the effects of low salinity ranging from 3 to 36 PSU on three seagrass species (b) and found broad salinity tolerance after 10 weeks exposure, except at the lowest salinities (<6PSU) where there was mortality (shoots were lost) (Figure 38). These low salinity thresholds are described in further detail below. Because of the broad tolerance, salinity was not prioritized for further experimental work within this short project. Despite the broad tolerance, there was a stress signal whereby shoots proliferated prior to complete loss indicating a redistribution of resources to lateral branching (shoot proliferation), and away from sexual reproduction (for Halophila ovalis). This study is the first to identify this stress induced morphometric response (SIMR) whereby shoot proliferation can occur in seagrasses during exposure to hypo-salinity. Since the shoot proliferation appeared to be a mild salinity stress, this suggests that interactive stressors such as high turbidity, elevated nutrients and elevated herbicides, which also occur in flood plumes, could exacerbate the stress response and lead to more and faster rates of mortality. The interactive effect of low salinity and other flood plume stressors remains a knowledge gap, which could not be addressed within the time-frame of this project.

More detail can also be found in Collier, C.J., Villacorta-Rath, C., van Dijk, K.-j., Takahashi, M., and Waycott, M. (2014). Seagrass Proliferation Precedes Mortality during Hypo-Salinity Events: A Stress-Induced Morphometric Response. PLoS ONE 9, e94014. 10.1371/journal.pone.0094014
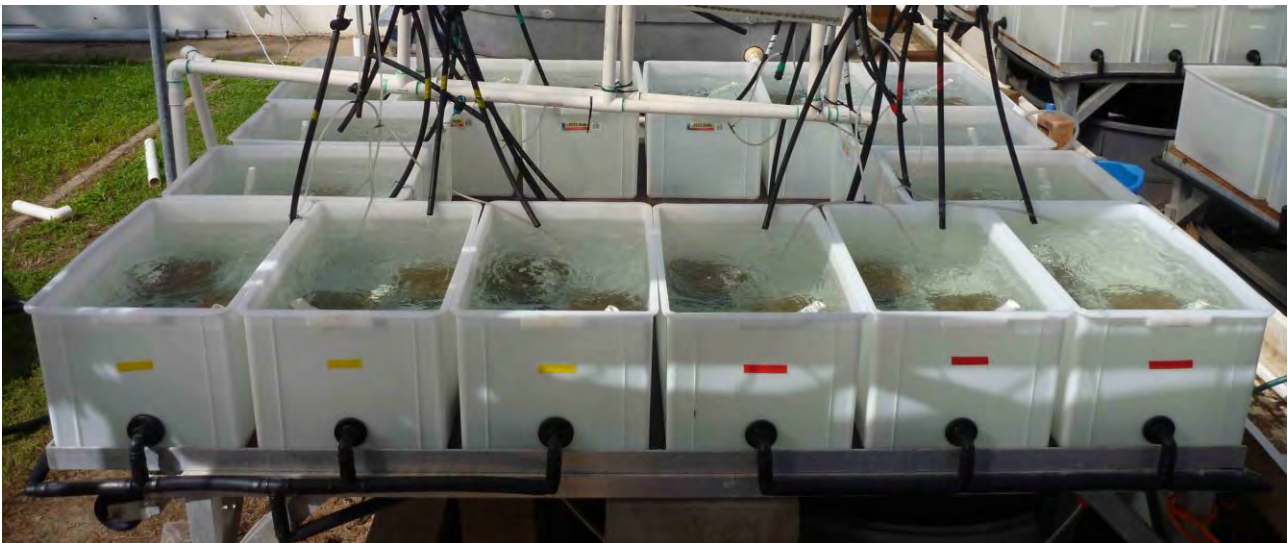


**Figure 16.**  Experimental set-up for testing responses of seagrass to seagrass at a fine-scale resolution with salinity ranging from 3 PSU (almost freshwater) to 36 PSU (seawater) at 3 PSU increments. This was run for 10 weeks and three species were tested. n=4.

**Key finding:** Seagrasses have broad salinity tolerance.

**Knowledge gap:** Interactive effects of salinity and other stressors.

# 5 Refined thresholds of concern for light, nutrients and salinity

## Light

Light thresholds were developed using two different approaches: in situ seagrass loss and experimental approaches where the interactive effects of seasonal water temperature were also explored and on a greater number of species (i.e. four species). There was close agreement in results between the two approaches (both showing 50% loss at ~4 mol m$^{-2}$ d$^{-1}$) for H. uninervis indicating a relatively high degree of confidence in experimental light thresholds (36). Using the experimental approach we can calculate light thresholds associated with any chosen level of seagrass loss/protection and we have focused on 50% loss for illustrative purposes. Reductions in shoot density of >50% relative to reference sites also demarcates a change from moderate to poor in the calculation of RRMMP report card scoring (McKenzie et al., 2013). However, the more aspirational levels of protection (e.g. 80% protection/20% loss) could be applied in very discrete impact events, such as dredging where there are known reference values for shoot density (e.g. before-after, or reference sites).
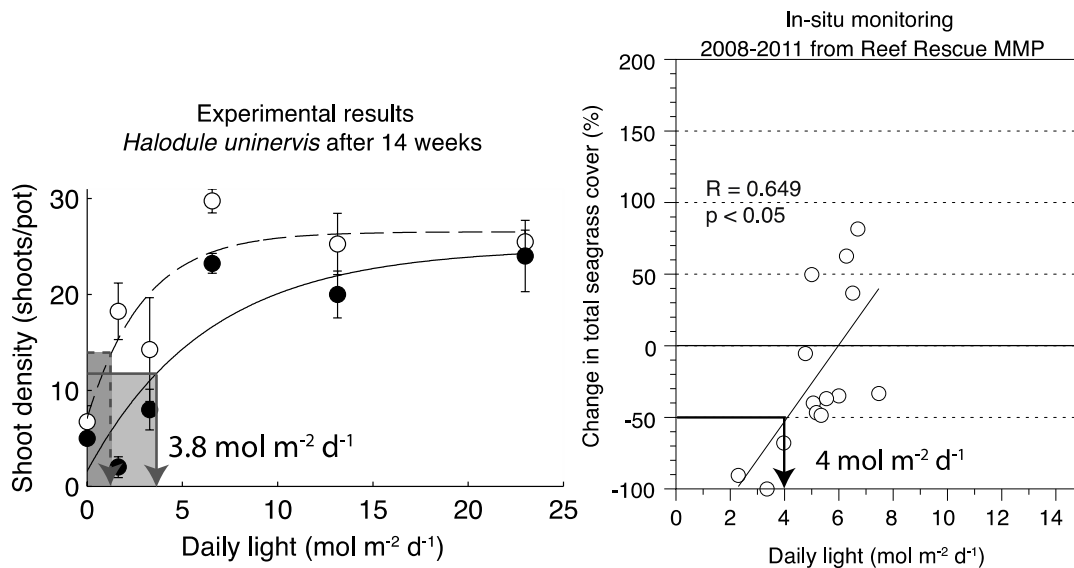


**Figure 17.** Halodule uninervis shoot density after 14 weeks in light levels ranging from 0 to 23 mol m$^{-2}$ d$^{-1}$ (70% sunlight) in warm (~27°C black circles) and cool (~22°C white circles) water. After 14 weeks, 50% loss of shoots was related to 3.8 mol m$^{-2}$ d$^{-1}$ (left). Change in seagrass cover (Halodule uninervis-dominated meadows) at Magnetic Island from 2008 to 2011 and Daily light (I$_d$ mol m$^{-2}$ d$^{-1}$). Each point represents an approximately 3 month period between seagrass cover measures (n=33 quadrats), and average daily light over the same period. 50% loss of seagrass occurred at an average of 4 mol m$^{-2}$ d$^{-1}$ and these losses occurred during the wet season in warm water conditions. From Collier et al 2012 and Collier et al In Prep.

The differences in thresholds depending on species and water temperature indicate that a single light threshold or guideline could over- or under-estimate the amount of light needed for seagrass protection (Figure 17 Figure 18). An adaptable light threshold could be more appropriate, particularly as increasing evidence for species differences in light thresholds amounts. For example higher in situ light thresholds developed for Z. muelleri (e.g. Chartrand et al., 2012) compared to H. uninervis (Collier et al., 2012) have been confirmed in this study using experimental approaches. Z. muelleri required 5.7 mol m$^{-2}$ d$^{-1}$, while H. uninervis 3.8 mol m$^{-2}$ d$^{-1}$ after 3 months exposure. Application of different thresholds for communities dominated by H. uninervis or Z. muelleri are relatively straightforward given that species dominance is also separated latitudinally, with Z. muelleri being more dominant in southern GBR meadows.
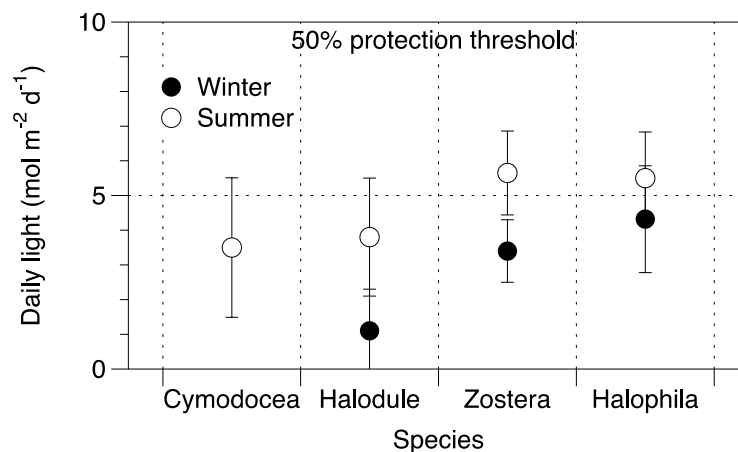
**Figure 18.** 50% decline threshold calculated from experimental light response curves (see 36). All species were exposed to a range of light treatments, and the light level causing 50% loss of shoots was calculated. This plot is the maximum threshold (generally 3 months) calculated over the experimental duration. These thresholds can be modelled from this data for short-medium term exposure to low light, and they do not represent the long-term minimum light requirement. NB a threshold could not be calculated for C. serrulata, as there was not 50% shoot loss in cool water even after 14 weeks.

Furthermore, there are different thresholds depending on the duration of exposure as generally, lower light levels can be tolerated for shorter periods of time (Collier et al, in prep). Light thresholds for 50% loss were 0 mol m$^{-2}$ d$^{-1}$ for C. serrulata and H. uninervis after 1 month but after 3 months (10-14 weeks) were 3.5 – 3.8 mol m$^{-2}$ d$^{-1}$. Using the models developed in this study, light thresholds can be calculated for any period of exposure up to 3 months for all species at different water temperature, and for any desired level of protection or loss. Reporting of in situ light levels against different thresholds is already in place in the Reef Rescue MMP, and this has been possible as a direct outcome of this project (McKenzie et al., 2013). It was not possible to test all interactive factors and so in cases where additional stressors are expected (e.g. possible herbicide or metal toxicity) a conservative interpretation of the thresholds is warranted. It is anticipated that these findings can contribute to the development of water quality guidelines for seagrasses, in particular, in relation to short-term water quality compliance (e.g. dredging) for the GBR. These light thresholds were developed under "subtidal" conditions. The transferability of these thresholds to intertidal habitats where daily light is swamped by extremely high, and largely unusable light levels (photoinhibition reduces photosynthetic efficiency) in the middle of the day (Petrou et al., 2013) remains a knowledge gap requiring attention.

Further detail on light thresholds and the methodology used to develop them can be found in Collier et al (2012) and Collier et al (In Prep-a).

**Key finding:** Event-based light thresholds have been developed for four GBR seagrasses with time and temperature-dependent values.

**Knowledge gap:** Effect of interactive stressors on light thresholds, light thresholds and how to report them in intertidal habitats, minimum light requirements, spectral light shifts, turbidity/chlorophyll a thresholds (sensu. De'ath and Fabricius, 2010), secondary effects of light limitation (e.g. changes in sediment biogeochemistry).

## Salinity

Threshold salinities associated with either proliferation or die-off were different among species with H. ovalis being the most sensitive, and Z. muelleri the most tolerant of hypo-salinity. Even after 10 weeks exposure, Z. muelleri had survived to salinities as low as 3PSU, and therefore has a hypo-salinity threshold <3 PSU. In contrast H. ovalis and H. uninervis remained abundant at 9PSU, but there were very large reductions in shoot density at lower salinities (3 and 6 PSU) indicating a hypo-salinity threshold of <9 PSU in these species after 10 weeks exposure. The longer the exposure, the higher the salinity threshold became. For example, the peak leaf pair density in H. ovalis shifted from 9 PSU after 2 weeks exposure to 12 PSU and finally to 15 PSU after 10 weeks exposure (Collier et al 2012). This time-dependency of the response could affect long-term species distributions i.e. they cannot survive low salinity over the long-term.

Collier, C.J., Villacorta-Rath, C., van Dijk, K.-j., Takahashi, M., and Waycott, M. (2014). Seagrass Proliferation Precedes Mortality during Hypo-Salinity Events: A Stress-Induced Morphometric Response. PLoS ONE 9, e94014. 10.1371/journal.pone.0094014
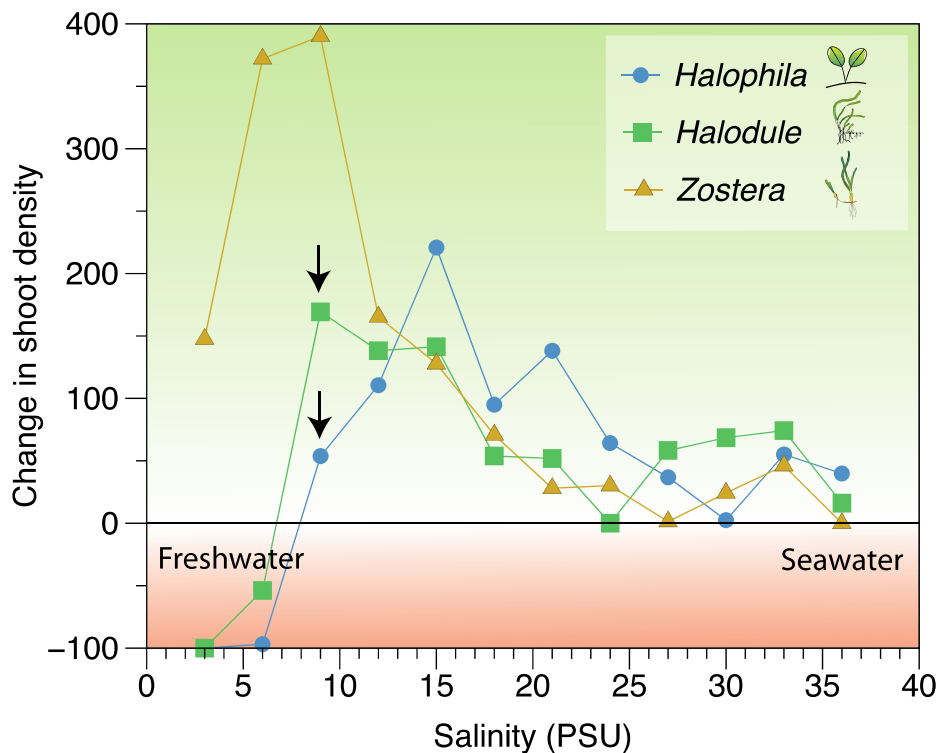


**Figure 19.**   Response of seagrass shoot density to reduced salinity ranging from almost freshwater (3PSU) to seawater (36PSU) after 10 weeks exposure. At mid- to low-salinity (6-15) there were increases in shoot density, by up to 400%, which indicates mild hypo-salinity stress. Below hypo-salinity thresholds (indicated by downward arrow at <9PSU) there was rapid mortality. Adapted from Collier et al 2014.

**Key finding:** Salinity thresholds have been developed for three GBR seagrasses over wet season exposures.

**Knowledge gap:** Effect of interactive stressors on salinity thresholds, long-term salinity thresholds.

# 6 Indicators of seagrass status

We tested a range of different indicators of seagrass status that fall into 8 groups (Table 4) including those currently used in Reef Rescue MMP report card generation such as seagrass abundance and tissue nutrient ratios, (Figure 5, Table 2, Table 3), indicators that could support generation of MMP report cards (e.g. $\delta^{13}C$ and carbohydrates), and supporting indicators that could be used in other monitoring programs or research projects (all others). They were all sensitive to water quality but their sensitivity was dependent on:

- Environmental parameter (e.g. light, nutrients, salinity)
- Duration of exposure to changed environmental parameter
- Species

Changes in an indicator need to be separated from natural background variation (daily, seasonal, inter-annual). Therefore, multiple indicators and appropriate design of monitoring is critical. The environmental conditions are very important, as for example, carbon/nitrogen ratios in seagrass leaves were very sensitive to nutrient concentration and to light, but at very high nutrient concentrations, they were no longer sensitive to light (Figure 14). Under conditions that nutrients are likely to be high, complementary indicators that are not sensitive to environmental nutrients, such as $\delta^{13}C$ could be appropriate (Figure 20) (Collier et al., In Prep-b). Timescales of response are also important, as over 6 – 8 weeks we found that sub-lethal indicators were sensitive to environmental change with no significant change in abundance or growth, even though abundance and growth are the most robust indicators over the longer-term (McMahon et al., 2013). Some indicators may not respond to very short-term environmental change as responses need to be sufficiently large to be detectable against background noise. In situ changes in shoot density occurred at low light intensities after ~ 3 months, suggesting that the time-frame of 2 to 3 months is a good period of time for monitoring sub-lethal through to lethal impacts for a number of species. Over shorter time-frames it may be appropriate to consider photosynthetic changes as a suitable indicator (McMahon et al., 2013; Ralph et al., 2007), but because they are highly sensitive they need to be applied with caution. They also require specialized equipment and training. Finally, there are considerations around resources including both expertise and budget, interactive factors, and management goals (McMahon et al., 2013). Table 11 summarises the indicators tested in this project.
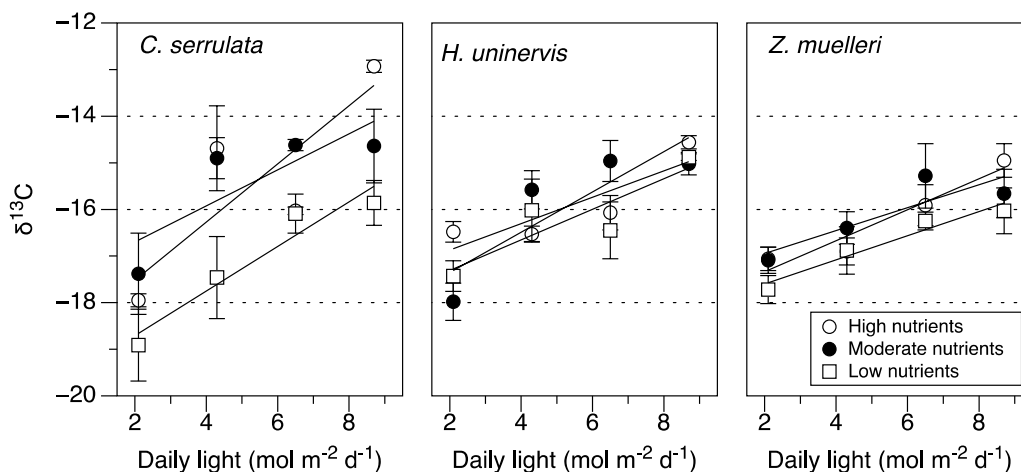


**Figure 20.**   $\delta^{13}C$ at light levels ranging from 0-40% surface light at low (white square), moderate (black circle) and high nutrient (white circle) concentrations after 7 weeks in three seagrass species. $\delta^{13}C$ reduced in low light conditions in all nutrient treatments. Adapted from Collier et al, In Prep.

**Table 11.**    Summary of indicators and how they responded to light, nutrients and salinity. A ✓ indicates that they were sensitive to the environmental parameter, a x means they were not, and a - means it was not tested.

| Level | Parameter grouping | Parameter | Light | Nutrients | Salinity | Flood plumes |
|---|---|---|---|---|---|---|
| Physiological (Sub-lethal) | Leaf Tissue nutrients | %C | ✓ | ✓ | - | - |
| | | %N | ✓ | ✓ | - | - |
| | | C:N | ✓ | ✓ | - | - |
| | | $\delta^{13}C$ | ✓ | ✓ | - | - |
| | Energy reserves | Rhiz carbohydrates | ✓ | ✓ | - | - |
| | Photosynthesis | PAM | ✓ | ✓ | x | - |
| | | $O_2$ production | ✓ | ✓ | - | - |
| Plant-scale (state change) | Growth | Leaf | ✓ | - | x | - |
| | | Rhizome | ✓ | - | - | - |
| | Morphology | | ✓ | - | x | - |
| | Sexual reproduction | | - | - | ✓ | - |
| Meadow-scale (population level) | Abundance | Shoot density | ✓ | - | ✓ | ✓ |
| | | Percent cover | ✓ | - | - | ✓ |
| | | Biomass | ✓ | - | ✓ | - |
| | Species composition | | ✓ | - | ✓ | ✓ |

**Key finding:** sub-lethal through to meadow-scale responses were sensitive to water quality.

**Knowledge Gap:** Very short-term responses, community shifts, interactive effects and long-term population-scale responses (e.g. changes in genetic diversity).

# 7 Future trajectories for GBR ecosystems

Water quality is a management priority for the long-term health of GBR seagrasses and for the animals (e.g. dugong) and ecological processes (e.g. nutrient uptake) that are dependent on healthy, diverse and productive meadows. Pulsed run-off from flood plumes, localized water quality impacts (e.g. dredge plumes) as well as chronic inshore water quality declines are all priority management concerns.

Water quality risks to seagrasses vary spatially, and for seagrasses, the highest risks occur in the Burdekin and Fitzroy NRM regions primarily due to high loads of catchment sourced suspended sediments (Brodie et al., 2013a). This risk assessment has provided a means by which management for improved water quality can be targeted for the greatest benefits. High turbidity and low light are not the only risks to seagrasses from declining water quality. Nutrients, and herbicides in particular affect seagrasses either directly or indirectly (Brodie et al., 2013a; Flores et al., 2013). Interaction between these stressors, particularly where synergistic or cumulative impacts occur, is a further concern. The chronic effect of exposure to herbicides on seagrasses is being explored in a separate NERP project (4.3 Negri et al) with some findings now available (Flores et al., 2013).

Flood plume impacts have led to recent unprecedented levels of seagrass loss. There is, however, signs of recovery in a number of the impacted seagrass meadows with colonizing species increasing in abundance (McKenzie et al., In Prep). It will take a number of years (up to 10 years) before the foundational species return to their former abundances. Disturbances (such as cyclones) resulting in loss and recovery of seagrass have been a part of the GBR seagrass ecosystems (Birch and Birch, 1984); however, as the types and frequency of disturbances increase, recovery processes (such as recruitment and germination) may be affected. The effect of pollutants on seagrass meadows is an especially important topic due to their close proximity of the coast. As agriculture and coastal development have increased and are expected to further increase over the coming years, it is particularly important to develop methods to interpret ecological change where environmental and WQ in-situ data are not available or limited. This study reaffirmed the strong potential of satellite images for understanding ecological change, as well as in-situ, site-specific logger data, and experimental approaches (aquaria) to investigate effects of water quality on seagrasses.

Recovery times, which depend on reproduction and connectivity will be critical to the capacity for seagrasses to recover from future and ongoing water quality impacts. Declining water quality is not the only effect on seagrasses; physical disturbance (e.g. cyclones Birch and Birch, 1984; McKenzie et al., 2012) and climate change, in particular increasing temperature, both chronic (Collier et al., 2011) as well as event-based extremes (Campbell et al., 2006; Collier and Waycott, 2014; Rasheed and Unsworth, 2011) threaten the resilience of seagrasses for coping with changing water quality. This research has improved our ability to manage ongoing water quality impacts to seagrasses and in doing so, increase their resilience in the face of these cumulative pressures.

The Indo-Pacific seagrasses tested in this project differed in their sensitivity to flood plumes, light, nutrients and salinity. For example, H. ovalis was the most sensitive to both hypo-salinity (b) and light stress (g). Therefore, it is expected that this species will be the first to disappear following an impact involving these environmental stressors. This species is a disturbance specialist: it produces large seed banks and regrows quickly from seed after a mortality event (Collier and Waycott, 2009). In contrast, the slightly less sensitive species (e.g. H. uninervis), may initially resist stress, but then recover more slowly than other species. Diverse seagrass meadows that include these climax species will not be sustained in a scenario with chronic water quality decline, as well as event-based impacts.

# References

Álvarez-Romero, J.G., Devlin, M., Teixeira da Silva, E., Petus, C., Ban, N.C., Pressey, R.L., Kool, J., Roberts, J.J., Cerdeira-Estrada, S., Wenger, A.S., et al. (2013). A novel approach to model exposure of coastal-marine ecosystems to riverine flood plumes based on remote sensing techniques. J Environ Manag 119, 194-207.

Bainbridge, Z.T., Wolanski, E., Álvarez-Romero, J.G., Lewis, S.E., and Brodie, J.E. (2012). Fine sediment and nutrient dynamics related to particle size and floc formation in a Burdekin River flood plume, Australia. Mar Pollut Bull 65, 236-248.

Birch, W.R., and Birch, M. (1984). Succession and pattern of tropical intertidal seagrases in Cockle Bay, Queensland, Australia: A decade of observations. Aquat Bot 19, 343-367.

Borja, A., Bricker, S.B., Dauer, D.M., Demetriades, N.T., Ferreira, J.G., Forbes, A.T., Hutchings, P., Jia, X., Kenchington, R., Marques, J.C., et al. (2008). Overview of integrative tools and methods in assessing ecological integrity in estuarine and coastal systems worldwide. Mar Pollut Bull 56, 1519-1537.

Brando, V.E., Dekker, A.G., Park, Y.J., and Schroeder, T. (2012). Adaptive semianalytical inversion of ocean color radiometry in optically complex waters. Applied optics 51, 2808-2833.

Brodie, J., Schroeder, T., Rohde, K., Faithful, J., Masters, B., Dekker, A., Brando, V., and Maughan, M. (2010). Dispersal of suspended sediments and nutrients in the Great Barrier Reef lagoon during river-discharge events: conclusions from satellite remote sensing and concurrent flood-plume sampling. Mar Freshw Res 61, 651-664.

Brodie, J., Waterhouse, J., Schaffelke, B., Furnas, M., Maynard, J., Collier, C., Lewis, S.E., Warne, M., Fabricius, K., Devlin, M.J., et al. (2013a). Relative risks to the Great Barrier Reef from degraded water quality. In Synthesis of evidence to support the Reef Water Quality Scientific Consensus Statement 2013 (Brisbane: Department of the Premier and Cabinet, Queensland Government).

Brodie, J., Waterhouse, J., Shaffelke, B., Johnson, J., Kroon, F.J., Thorburn, P., Rolfe, J., Lewis, S.E., Warne, M., Fabricius, K., et al. (2013b). Reef Water Quality Scientific Consensus Statement 2013 (Brisbane: Department of the Premier and Cabinet, Queensland Government).

Cambridge, M.L., Chiffings, A.W., Brittain, C., Moore, L., and McComb, A.J. (1986). The loss of seagrass in Cockburn Sound, Western Australia. II. Possible causes of seagrass decline. Aquat Bot 24, 269-285.

Campbell, S.J., McKenzie, L.J., and Kerville, S.P. (2006). Photosynthetic responses of seven tropical seagrasses to elevated seawater temperature. J Exp Mar Biol Ecol 330, 455-468.

Chartrand, K.M., Ralph, P.J., Petrou, K., and Rasheed, M.A. (2012). Development of a light-based seagrass management approach for the Gladstone Western basin dredging program (Cairns: Fisheries Queensland, Department of Agriculture Fisheries and Forestry), pp. 126.

Coles, R.G., Lee Long, W.J., Watson, R.A., and Derbyshire, K.J. (1993). Distribution of seagrasses and their fish and penaeid prawn communities in cairns harbour, a tropical estuary, Northern Queensland. Mar Freshw Res 44, 193-201.

Collier, C., and Waycott, M. (2009). Drivers of change to seagrass distributions and communities on the Great Barrier Reef: Literature review and gaps analysis. In Report to the Marine and Tropical Sciences Research Facility (Cairns: Reef and Rainforest Research Centre Limited).

Collier, C.J., Langlois, L., Adams, M., O'Brien, K., Maxwell, P., Waycott, M., and McKenzie, L. (In Prep-a). Seagrass protection guidleines developed from experimental light response curves.

Collier, C.J., Langlois, L., McKee, K.L., Vlahkis, K., Takahashi, M., McMahon, K.M., and Waycott, M. (In Prep-b). Sub-lethal indicators of light stress in tropical seagrasses: thresholds and effects of nutrient availability.

Collier, C.J., Uthicke, S., and Waycott, M. (2011). Thermal tolerance of two seagrass species at contrasting light levels: implications for future distribution in the Great Barrier Reef. Limnol Oceanogr 56, 2200-2210.

Collier, C.J., and Waycott, M. (2014). Temperature extremes reduce seagrass growth and induce mortality. Mar Pollut Bull 83, 483-490.

Collier, C.J., Waycott, M., and McKenzie, L.J. (2012). Light thresholds derived from seagrass loss in the coastal zone of the northern Great Barrier Reef, Australia. Ecological Indicators 23, 211-219.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R.V., Paruelo, J., et al. (1997). The value of the worlds ecosystem services and natural capital. Nature 387, 253-260.

Cullen-Unsworth, L., and Unsworth, R. (2013). Seagrass Meadows, Ecosystem Services, and Sustainability. Environment: Science and Policy for Sustainable Development 55, 14-28.

De'ath, G., and Fabricius, K. (2010). Water quality as a regional driver of coral biodiversity and macroalgae on the Great Barrier Reef. Ecol Appl 20, 840-850.

Dennison, W.C., Orth, R.J., Moore, K.A., Stevenson, C., Carter, V., Kollar, S., Bergstrom, P.W., and Batiuk, R.A. (1993). Assessing water quality with submersed aquatic vegetation: habitat requirements as barometers of Chesapeake Bay health. Bioscience 43, 86-94.

Devlin, M., Wenger, A., Waterhouse, J., Alvarez-Romero, J., Abbott, B., and Teixeira da Silva, E. (2012a). Reef Rescue Marine Monitoring Program: Flood Plume Monitoring Annual Report 2010-2011 (Townsville: ACTFR (TropWATER) Report Number 12/02).

Devlin, M.J., and Brodie, J. (2005). Terrestrial discharge into the Great Barrier Reef Lagoon: nutrient behavior in coastal waters. Mar Pollut Bull 51, 9-22.

Devlin, M.J., da Silva, E., Petus, C., Wenger, A., Zeh, D., Tracey, D., Álvarez-Romero, J.G., and Brodie, J. (2013). Combining in-situ water quality and remotely sensed data across spatial and temporal scales to measure variability in wet season chlorophyll-a: Great Barrier Reef lagoon (Queensland, Australia). Ecological Processes 2, 31.

Devlin, M.J., McKinna, L.W., Álvarez-Romero, J.G., Petus, C., Abott, B., Harkness, P., and Brodie, J. (2012b). Mapping the pollutants in surface riverine flood plume waters in the Great Barrier Reef, Australia. Mar Pollut Bull 65, 224-235.

Flores, F., Collier, C.J., Mercurio, P., and Negri, A.P. (2013). Phytotoxicity of Four Photosystem II Herbicides to Tropical Seagrasses. PLoS ONE 8, e75798.

Fourqurean, J.W., Duarte, C.M., Kennedy, H., Marba, N., Holmer, M., Mateo, M.A., Apostolaki, E.T., Kendrick, G.A., Krause-Jensen, D., McGlathery, K.J., et al. (2012). Seagrass ecosystems as a globally significant carbon stock. Nature Geosci 5, 505-509.

Fourqurean, J.W., Moore, T.O., Fry, B., and Hollibaugh, J.T. (1997). Spatial and temporal variation in C:N:P Ratios, Delta$^{15}$N and Delta$^{13}$C of eelgrass Zostera marina as indicators of ecosystem processes, Tomales Bay, California, USA. Mar Ecol Prog Ser 157, 147-157.

GBRMPA (2009). Water quality guidelines for the Great Barrier Reef Marine Park (Townsville: Great Barrier Reef Marine Park Authority, The Australian Government).

GBRMPA (2014). Reef Rescue Marine Monitoring Program quality assuarnce and quality control manual. 2012/13 (Townsville: Great Barrier Reef Marine Park Authority).

Hughes, A.R., Williams, S.L., Duarte, C.M., Heck, K.L., and Waycott, M. (2008). Associations of concern: declining seagrasses and threatened dependent species. Frontiers in Ecology and the Environment 7, 242-246.

Kendrick, G.A., Holmes, K.W., and Van Niel, K.P. (2008). Multi-scale spatial patterns of three seagrass species with different growth dynamics. Ecography 31, 191-200.

Larkum, W.D., Drew, E.A., and Ralph, P.J. (2006). Photosynthesis and metabolism in seagrasses at the cellular level. In Seagrasses: biology, ecology and conservation, W.D. Larkum, R.J. Orth, and C.M. Duarte, eds. (Dordrecht: Springer).

Longstaff, B.J., and Dennison, W.C. (1999). Seagrass survival during pulsed turbidity events: the effects of light deprivation on the seagrasses Halodule pinifolia and Halophila ovalis. Aquat Bot 65, 101-121.

Marsh, H., O'Shea, T.J., and Reynolds III, J.E. (2011). Ecology and conservation of the sirenia (Cambridge: Cambridge University Press).

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

McKenzie, L., Collier, C.J., and Waycott, M. (2013). Reef Rescue Marine Monitoring Program: Inshore seagrass, annual report for the sampling period 1st September 2011-31st May 2012 (James Cook University), pp. 230pp.

McKenzie, L., Collier, C.J., and Waycott, M. (In Prep). Reef Rescue Marine Monitoring Program: Inshore seagrass annual report for the sampling period 1st September 2012 - 31st May2013 (Townsville: Great Barrier Reef Marine Park Authority).

McMahon, K.M., Collier, C.J., and Lavery, P.S. (2013). Identifying robust bioindicators of light stress in seagrasses: A review. Ecological Indicators 30, 7-15.

Meager, J.J., and Limpus, C.J. (2012). Marine wildlife stranding and mortality database annual report 2011. I. Dugong, pp. 1-30.

Naik, P., D'Sa, E.J., Gomes, H.d.R., Goés, J.I., and Mouw, C.B. (2013). Light absorption properties of southeastern Bering Sea waters: Analysis, parameterization and implications for remote sensing. Remote Sensing of Environment 134, 120-134.

Orth, R.J., Carruthers, T.J.B., Dennison, W.C., Duarte, C.M., Fourqurean, J.W., Heck, K.L.J., Hughes, A.R., Kendrick, G.A., Kenworthy, W.J., Olyarnik, S., et al. (2006). A global crisis for seagrass ecosystems. Bioscience 56, 987-996.

Petrou, K., Jimenez-Denness, I., Chartrand, K., McCormack, C., Rasheed, M., and Ralph, P.J. (2013). Seasonal heterogeneity in the photophysiological response to air exposure in two tropical intertidal seagrass species. Mar Ecol Prog Ser 482, 93-106.

Petus, C., Collier, C.J., Devlin, M., Rasheed, M., and McKenna, S. (2014a). Using MODIS data for understanding changes in seagrass meadow health: a case study in the Great Barrier Reef (Australia). Mar Environ Res.

Petus, C., da Silva, E.T., Devlin, M., Wenger, A.S., and Álvarez-Romero, J.G. (2014b). Using MODIS data for mapping of water types within river plumes in the Great Barrier Reef, Australia: Towards the production of river plume risk maps for reef and seagrass ecosystems. J Environ Manag 137, 163-177.

Pollard, P.C., and Greenway, M. (2013). Seagrasses in tropical Australia, productive and abundant for decades decimated overnight. J Biosci (Bangalore) 38, 1-10.

Preen, A.R., Long, W.J.L., and Coles, R.G. (1995). Flood and cyclone related loss, and partial recovery, of more than 1000 km$^2$ of seagrass in Hervey Bay, Queensland, Australia. Aquat Bot 52, 3-17.

Ralph, P.J., Durako, M.J., Enriquez, S., Collier, C.J., and Doblin, M.A. (2007). Impact of light limitation on seagrasses. J Exp Mar Biol Ecol 350, 176-193.

Rasheed, M.A., McKenna, S.A., Carter, A.B., and Coles, R.G. (2014). Contrasting recovery of shallow and deep water seagrass communities following climate associated losses in tropical north Queensland, Australia. Mar Pollut Bull 83, 491-499.

Rasheed, M.A., and Unsworth, R.K.F. (2011). Long-term climate-associated dynamics of a tropical seagrass meadow: implications for the future. Mar Ecol Prog Ser 422, 93-103.

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

Romero, J., Martinez-Crego, B., Alcoverro, T., and Perez, M. (2007). A multivariate index based on the seagrass Posidonia oceanica (POMI) to assess ecological status of coastal waters under the framework directive (WFD). Marine Pollution Bulletin 54, 631-631.

Schroeder, T., Devlin, M.J., Brando, V.E., Dekker, A.G., Brodie, J.E., Clementson, L.A., and McKinna, L. (2012). Inter-annual variability of wet season freshwater plume extent into the Great Barrier Reef lagoon based on satellite coastal ocean colour observations. Mar Pollut Bull 65, 210-223.

Short, F.T., Polidoro, B., Livingstone, S.R., Carpenter, K.E., Bandeira, S., Bujang, J.S., Calumpong, H.P., Carruthers, T.J.B., Coles, R.G., Dennison, W.C., et al. (2011). Extinction risk assessment of the world's seagrass species. Biol Conserv 144, 1961-1971.

Udy, J.W., Dennison, W.C., Long, W.J.L., and McKenzie, L.J. (1999). Responses of seagrass to nutrients in the Great Barrier Reef, Australia. Mar Ecol-Prog Ser 185, 257-271.

Unsworth, R.K.F., Collier, C.J., Henderson, G.M., and McKenzie, L.J. (2012). Tropical seagrass meadows modify seawater carbon chemistry: implications for coral reefs impacted by ocean acidification. Environmental Research Letters 7, 024026.

Unsworth, R.K.F., Collier, C.J., Waycott, M., McKenzie, L.J., and Cullen-Unsworth, L.C. (Subm). Global change and the resilience of seagras ecosystems.

Van der Heide, T., Smolders, A., Riikens, B., van Nes, E.H., van Katwick, M.M., and Roelofs, J. (2008). Toxicity of reduced nitrogen in eelgrass (Zostera marina) is highly dependent on shoot density and pH. Oecologia 158, 411-419.

Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A., Fourqurean, J.W., Heck, K.L., Hughes, A.R., et al. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proceedings of the National Academy of Sciences 106, 12377-12381.

Waycott, M., Longstaff, B.J., and Mellors, J. (2005). Seagrass population dynamics and water quality in the Great Barrier Reef region: A review and future research directions. Mar Pollut Bull 51, 343-350.

# Attachments

Thresholds information sheet

# Towards water quality thresholds for healthy seagrass habitats in the Great Barrier Reef
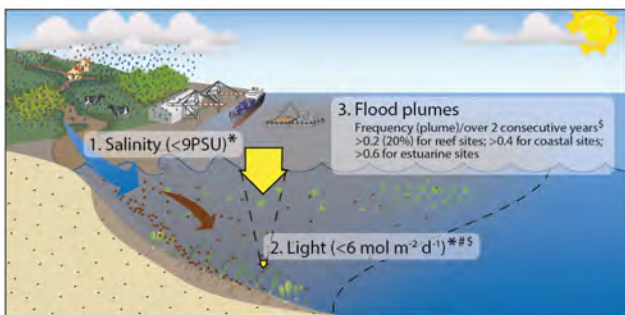


A summary of research from the National Estuarine Research Program Tropical Ecosystem hub  (NERP TE) project 5.3

# Flood plumes affect seagrass health

- Seagrass meadows are exposed to a range of potential impacts including cyclones, physical disturbance, wind-induced resuspension of sediment, dredging operations, sedimentation and flood plumes.

- Flood plumes generally cover large areas of the inshore GBR and are characterized by low salinity, high nutrient concentrations (triggering "green water"), toxicants, and both dissolved ("coloured water") and particulate matter ("brown water"). Plume waters reduce light penetration, which can reduce seagrass growth and overall health.

- Dredge plumes generally cover relatively small areas of the inshore GBR and can also create brown water with similar characteristics and consequences to seagrass as flood plume waters.

- Repeated exposure to this range of potential impacts resulted in the loss of seagrass meadows in the GBR during 2009-2011 (abundance fell below regional guidelines at 67% of sites). Some signs of recovery are now being observed.

- This research has focused on the impact that salinity, low light and nutrients may have had on the growth and overall health of seagrasses.



**Figure 1**. Salinity (1), light (2) and flood plume (3) thresholds were identified in this research.

- Findings indicate that low light conditions could have been a major contributor to the observed loss of seagrass meadows.

- Salinity, light and flood plume thresholds for these seagrasses have been developed in this research (Figure 1).

## Frequency of exposure to flood plumes

- *F(plume)* is a proxy measurement representing a combination of colour classes that relate directly to the quality of the waters and impact on each seagrass habitat (estuarine, coastal, reef intertidal and subtidal) differently.

- For example, high turbidity waters, represented by a combination of colour class (CC) 1-4, are the main water type influencing the ecological condition of the estuarine seagrass sites. Estuarine seagrass beds exposed to CC 1:4 for >60% of the wet season are predicted to decline >50% in seagrass cover. Annual and multi-annual measurements of ocean colour can be related to broad scale water quality measurements, including annual values of light attenuation over the total seagrass bed.



**Figure 2.** (Left) The combination of ocean colour classes measured (expressed as a normalized frequency) within each seagrass habitat and corresponding multi-annual (2007–2012) seagrass cover. (Right) MODIS image and corresponding colour-class map.
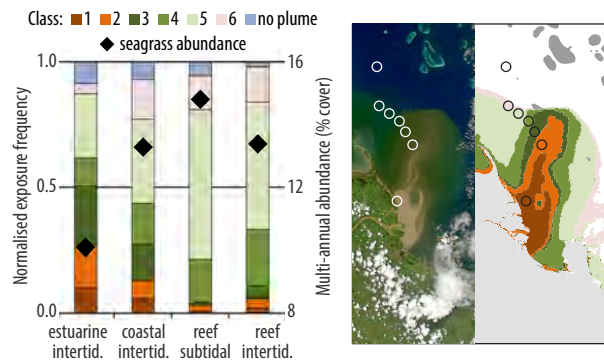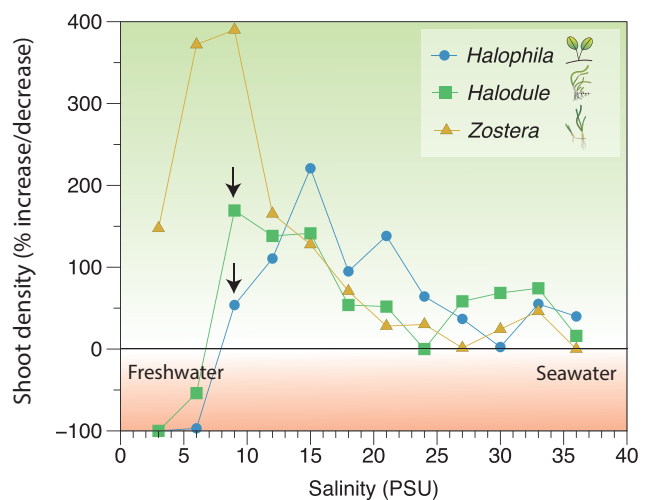
# Salinity thresholds

- Floodwater is low in salinity, but the tolerance limit of seagrasses to low salinity was not known.

- Seagrass (three species) responses to salinity ranging from seawater at 36 practical salinity units (PSU) to 3PSU (almost fresh water) were tested over 10 weeks.

- Seagrasses were quite tolerant of short-term exposure to low salinity.

- They showed what is likely to be a mild stress response with density (number of leafy shoots) actually increasing by 400% at low-moderate salinity (6-9 PSU) for *Zostera muelleri*, and by almost 200% at 9-15 PSU for other species (Figure 3).

- Density declined sharply at the lowest salinity, indicating a salinity threshold of <9PSU.

- Because of their broad salinity tolerance, and the limited scope for management of salinity, establishment of salinity guidelines is a low priority.
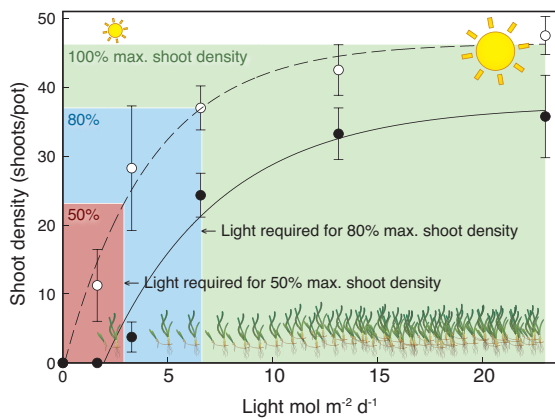


**Figure 3.** Change in seagrass shoot density (leaf pair density for *Halophila*) relative to week zero, after 10 weeks exposure to low salinity ranging from almost freshwater (3 PSU) to marine seawater (36 PSU). Density declined sharply below 9 PSU (↓).
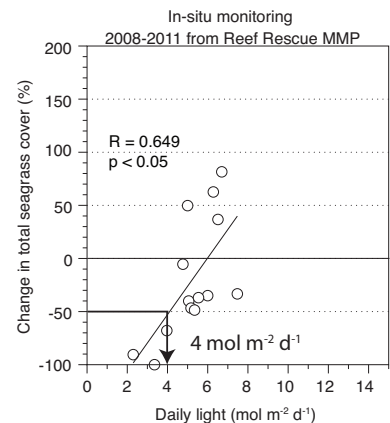
# Light thresholds

- Density and growth responses of seagrass species to low light (shading) was measured for just over 3 months in cool (22ºC) and warm (27ºC) water.

- In very low light (simulating muddy water) density and growth declined (Figure 4). There was more decline as the duration of exposure increased and in warm water.

- In warm water there was complete mortality (no seagrass remaining) of the most sensitive species, *Halophila*, after only 17 days. This was followed closely by *Zostera* (30 d), whereas *Halodule* survived for more than 3 months.

- The light level causing 20% decline (80% saved) ranged from 4 to 10 mol photons $m^{-2} d^{-1}$ over 3 months, and for 50% decline ranged from 3 to 6 mol photons $m^{-2} d^{-1}$ depending on species and water temperature.

- These values are relevant to short-term low light events (up to 3 months), but do not represent long-term minimum light requirements.

- Figure 4 shows aquarium-based threshold calculations for *Zostera*. Using the same method for *Halodule*, 50% loss occurred at 3.8 mol photons $m^{-2} d^{-1}$ in warm water after 3 months. In situ loss of 50% occurred at 4 mol photons $m^{-2} d^{-1}$ after 3 months (Figure 5). These very similar thresholds suggest aquarium results can be used for further threshold development.



**Figure 4.** Shoot density of *Zostera* after almost 3 months at light levels ranging from no sunlight (complete darkness) to 23 mol $m^{-2} d^{-1}$ (70% of full sunlight) in cool (white circles) and warm (black circles) water. From this data we can calculate the light level that caused 20% or 50% decline of seagrass. This calculated light level is a "threshold".

**Figure 5.** Change in the abundance of *Halodule*-dominated meadows at Magnetic Island, with each point representing change over a 3 month period, and average light reaching the seagrass canopy. 50% loss of seagrass was occurred at a light level of 4 mol $m^{-2} d^{-1}$.



# How can this information be applied?

## Monitoring and reporting

The Reef Rescue Marine Monitoring Program (MMP) measures and reports annually, on water quality and seagrass health. These seagrass salinity and light tolerance thresholds can be used to explain some of the observed changes in seagrass meadow health. There are additional environmental factors that can affect seagrass health, and these are also considered when explaining changes in seagrass health.
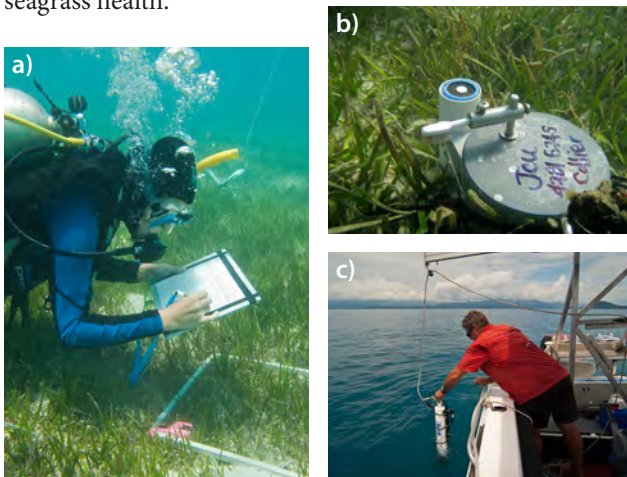
## Development of water quality guidelines

Researchers have proposed that results of this research be used to develop short-term guidelines for these seagrasses. These guidelines could be an important consideration in determining and managing potential environmental impacts on seagrass meadows associated with dredging operations. For these thresholds to become operational, information on what reduces the available light is also required. Management actions that can be taken will primarily relate to catchment management actions that target the reduction of sediment and/or organic matter input.



© Xanthe Rivett / CAFNEC / WWF

**Figure 6.** Long-term seagrass monitoring for the Reef Rescue MMP (a-c) includes ongoing measurement of light received in the seagrass canopy (b) as well as water quality (c). The development of water quality guidelines for seagrass will help managers reduce the associated impacts of dredging operations (d).

# RELATED PROJECTS
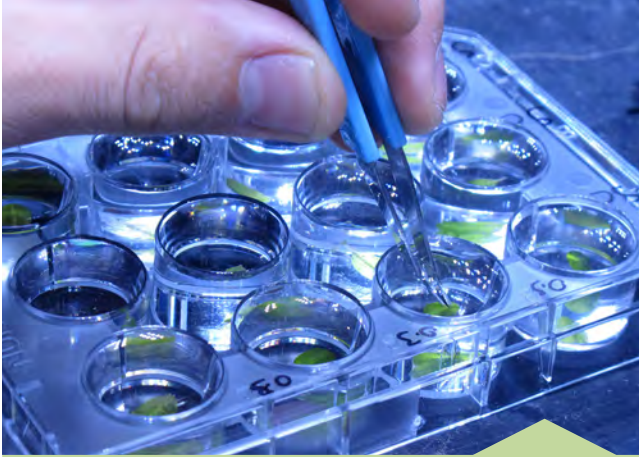


## Herbicides

In a related NERP TE project led by Dr Andrew Negri (AIMS), the effect of herbicides on seagrasses were tested. Threshold levels of herbicide exposure that reduce seagrass photosynthetic efficiency and affect seagrass energetic balances were identified.

**Figure 7 (above).** A bioassay for rapidly screening seagrass exposure to herbicides has been developed. This is just one of the techniques being used to identify seagrass herbicide thresholds.

## Port Curtis seagrass light requirements

Developing a light-based seagrass management strategy using locally derived light thresholds for *Zostera muelleri* was implemented as part of a dredge management plan in Port Curtis. The work used in situ shading studies, long term light and seagrass monitoring and lab based manipulative experiments to derive locally relevant light thresholds. These were adapted into traditional turbidity-based monitoring programs. Sub-lethal indicators of light stress and the effect of spectral quality of light were also invested in this project.

## Dynamics of deep-water seagrasses

The TropWATER seagrass group also has research programs established to determine the light requirements of deep-water Halophila species (>15m). These studies include a range of field and laboratory manipulative studies focusing on light requirements and their interactions with temperature, season and spectral shift, as well as establishing sub-lethal indicators for management. Study sites are at Green Island, Lizard Island, Abbot Point and Mackay.

Further details can be found at: http://research.jcu.edu.au/research/tropwater/research-programs/seagrass-ecology-1/seagrass-ecology

## Publications arising from this work:

Collier, C.J., Villacorta-Rath, C., van Dijk, K.-j., Takahashi, M., and Waycott, M. (2014). Seagrass Proliferation Precedes Mortality during Hypo-Salinity Events: A Stress-Induced Morphometric Response. *PLoS ONE* 9, e94014.

Collier, C.J., Langlois, L., McKenzie, L., and Waycott, M. (In Prep). Seagrass protection guidleines developed from experimental light response curves.

Collier, C.J., Waycott, M., and McKenzie, L.J. (2012). Light thresholds derived from seagrass loss in the coastal zone of the northern Great Barrier Reef, Australia. *Ecological Indicators* 23, 211-219.

Petus, Caroline, Catherine Collier, Michelle Devlin, Michael Rasheed, and Skye McKenna. "Using MODIS data for understanding changes in seagrass meadow health: A case study in the Great Barrier Reef (Australia)." *Marine environmental research* 98 (2014): 68-85.

Devlin, Michelle J., Eduardo Teixeira da Silva, Caroline Petus, Amelia Wenger, Daniel Zeh, Dieter Tracey, Jorge G. Álvarez-Romero, and Jon Brodie. "Combining in-situ water quality and remotely sensed data across spatial and temporal scales to measure variability in wet season chlorophyll-a: Great Barrier Reef lagoon (Queensland, Australia)." *Ecological Processes 2*, no. 1 (2013): 1-22.

## Contributors:

Catherine Collier, Michelle Devlin, Cecilia Villacorta-Rath, Lucas Langlois, Matthew Adams, Kate O'Brien, Len McKenzie, Carol Honchin, Katherine Martin, Caroline Petus, Eduardo da Silva, John Statton, Kathryn McMahon, Michelle Waycott.

# 3 Monitoring large scale water quality impacts on seagrass communities in the Great Barrier Reef utilising MODIS imagery and long term monitoring data

## Abstract

We explored the environmental components of wet season conditions, high flow and their effects on seagrass meadow health. This included an assessment of remote sensing-derived ocean colour analysis and identified water quality thresholds associated with loss of seagrass (2007 – 2014) over annual and multi-annual ocean colour measurements derived from remote sensing.  Plumes of floodwaters with high concentrations of TSS, chl-a and CDOM were detected using satellite imagery and in-situ data. Decline of seagrass was exacerbated by the consecutive above wet season flow conditions from 2008 to 2011 in the catchments of the Great Barrier Reef leading to widespread flooding. High flow condition and the extent and distribution of flood plumes can help track low salinity conditions, high nutrient concentrations (triggering blooms of phytoplankton) and both dissolved and particulate sediment that drive reductions in light. We found that seagrass meadows in coastal and estuarine regions of the GBR were exposed to flood plumes of high turbidity water for ~20% and green water for ~50% of the wet season months (Nov-April). Declines in habitat-consolidated seagrass % cover were compared with their frequency of exposure to River plume colour classes; each 6CC class being associated with different concentrations and proportion of land-sourced contaminants and light availability. The correlations between colour class, water quality and seagrass health confirmed that MODIS data can be used to explain changes in seagrass health at the seagrass habitat scale and indicated that declines in seagrass areas and biomasses over the monitored period were linked to cumulative exposure to plume waters.

Intermittent exposure to reduced water quality can result in relatively high biomass meadows but slight change in water quality can shift the balance in these seagrass communities. Large scale water quality mapping can help define the type of seagrass communities and identify the main water types which shape and drive seagrass response. Thus long term water quality data, both in-situ and through remote sensing can provide measures of risk relative to the seagrass community health, including measures of seagrass biomass, cover and species.

## Introduction

Remote Sensing (RS) data combined with in situ sampling of river plumes has provided an essential source of data related to the movement and composition and frequency of river plumes and land-sourced pollutants in GBR waters (Bainbridge et al., 2012; Brodie et al., 2010; Devlin et al., 2012b; Petus et al., 2014a; Petus et al., 2014b; Schroeder et al., 2012). GBR Plume waters have been described as different river plume water masses from the coast to the offshore boundary of the plume; each being characterized by varying water quality characteristics, light availability, salinity and colours (Devlin et al., 2012b; Petus et al., 2014b). RS data identify areas which may experience acute or chronic high exposure to the river plumes water masses and thus enhance our ability to map the risk to GBR seagrasses from exposure to reduced water quality. Working on the Cleveland bay subset study area, Petus et al. (2014b) have thus shown that strong correlation existed between the changes of biomass and area of individual meadows and exposure of seagrass ecosystems to turbid plume water masses mapped through MODIS images.

Mapping seagrass meadows and associated ecological measurements through remote sensing has been challenging to investigate for both GBR and world-wide systems, due to the water

quality type typically found around seagrass meadows. Many seagrass meadows are found in shallow, coastal waters, which are typically Type 2 /optically complex waters where suspended sediment and coloured dissolved organic matter co-occur with phytoplankton. The standard and global bio-optical algorithms used in clear or "oceanic" waters are mostly inaccurate when applied to these coastal waters, although regional parameterization of these algorithms can help increase their accuracies (Brando et al., 2012; Naik et al., 2013). However, utilization of remote sensing imagery provides data across large spatial and temporal scales that would not be possible with traditional in-situ monitoring. To avoid issues with the extraction of Level 2 data, such as chlorophyll and total suspended sediments, we have utilised only MODIS true color images (Level 1b products), which represent water colour as a proxy for the water quality condition.

Multi-scale (temporal and spatial) studies of seagrass species distributions is often the starting point for examining environmental drivers and interpreting responses of seagrass meadows to climate change and decreased water quality (Kendrick et al., 2008; Petus et al., 2014b). The main objective of this study was to test if relationships can be established between the frequency of exposure to river plume water masses and changes in seagrass health in the GBR at different spatial and temporal scales. We focused on a 2-year period (2005–2007) of below median rainfall followed by a five-year period (2008–2012) of above-median rainfall and flooding to test seagrass health responses to river plume exposure in the GBR. River plume water masses exposure was assessed through the satellite mapping of GBR plume colour classes; each class being associated with different concentrations and proportion of land-sourced contaminants. Seagrass health was defined in this study by the seagrass percentage cover.

# Methods

The frequency and spatial extent of flood plumes is mainly driven by the size and intensity of flow (Devlin et al., 2012b). Flow data was sourced from the Department of Environment and Resource Management (Queensland, http://watermonitoring.derm.qld.gov.au/host.htm). River discharge data from 1975 to 2011 were obtained for the rivers that have the greatest influence over the study sites (Devlin et al., 2013).

Flow is calculated as an annual median for the whole GBR. Timing of sampling against total river flow is described, taking into account; all 35 rivers distributed throughout the GBR and also for Tully and Fitzroy rivers, separately. Two descriptive statistics derived from daily river flow data were used to describe the river flow regimes considering all 35 together: (i) the total annual flow into the GBR lagoon, and (ii) the long-term annual median flow calculated for the period 1970 to 2001.

## Satellite mapping of GBR river plumes

Three distinct plume water types have been described within GBR river plumes (from the inshore to the offshore boundary of river plumes) characterized by varying salinity levels, colour, spectral properties and WQ concentrations (Table 6). Flood plumes were mapped in this work using the method presented in Álvarez-Romero et al. (2013). In this method, daily MODIS Level-0 data acquired from the NASA Ocean Colour website (http://oceancolour.gsfc.nasa.gov) are converted into true colour images with a spatial resolution of about 500×500 m using SeaWiFS Data Analysis System (SeaDAS; Baith et al., 2001). True colour images are then spectrally enhanced (from RGB to HSI colour system) and classified into six river plume colour classes (CC1 to CC6) corresponding to six distinct inshore-to-offshore plume water masses through a supervised classification using spectral signature from river plume waters in the GBR.

Numerous recent studies used the method presented in Álvarez-Romero et al. (2013) to describe GBR plume waters and ecosystems exposure to land sourced pollutants (e.g., Devlin et al., 2013; Petus et al., 2014b). All were based on GBR river plume colour classes reclassified into 3 plume