

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 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 ($[\Delta(A_{ref})]$). 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 ($Sal = 32.44 \pm 4.70$ 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 ($K_d(PAR) = 1.10 \pm 0.97$ 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 ($K_d(PAR) = 0.20 \pm 0.26$ m⁻¹; TSS = 8.37 ± 8.29 mg 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	K _d (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

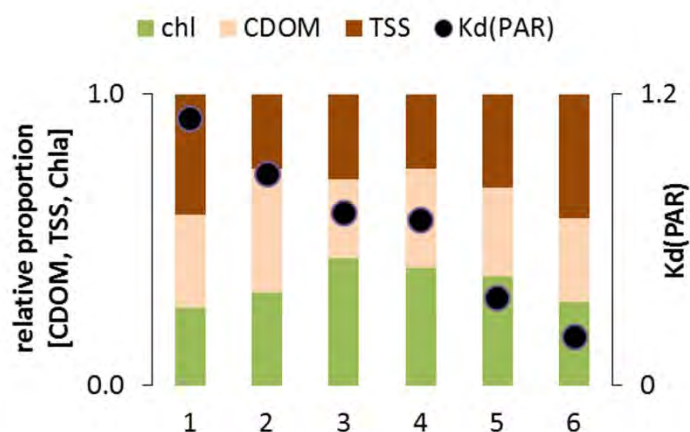


Figure 10. Relative proportion of optically active components [CDOM, Chl-a and TSS] across the 6 colour classes.

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 ($\Delta(A_{ref})$ = -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.

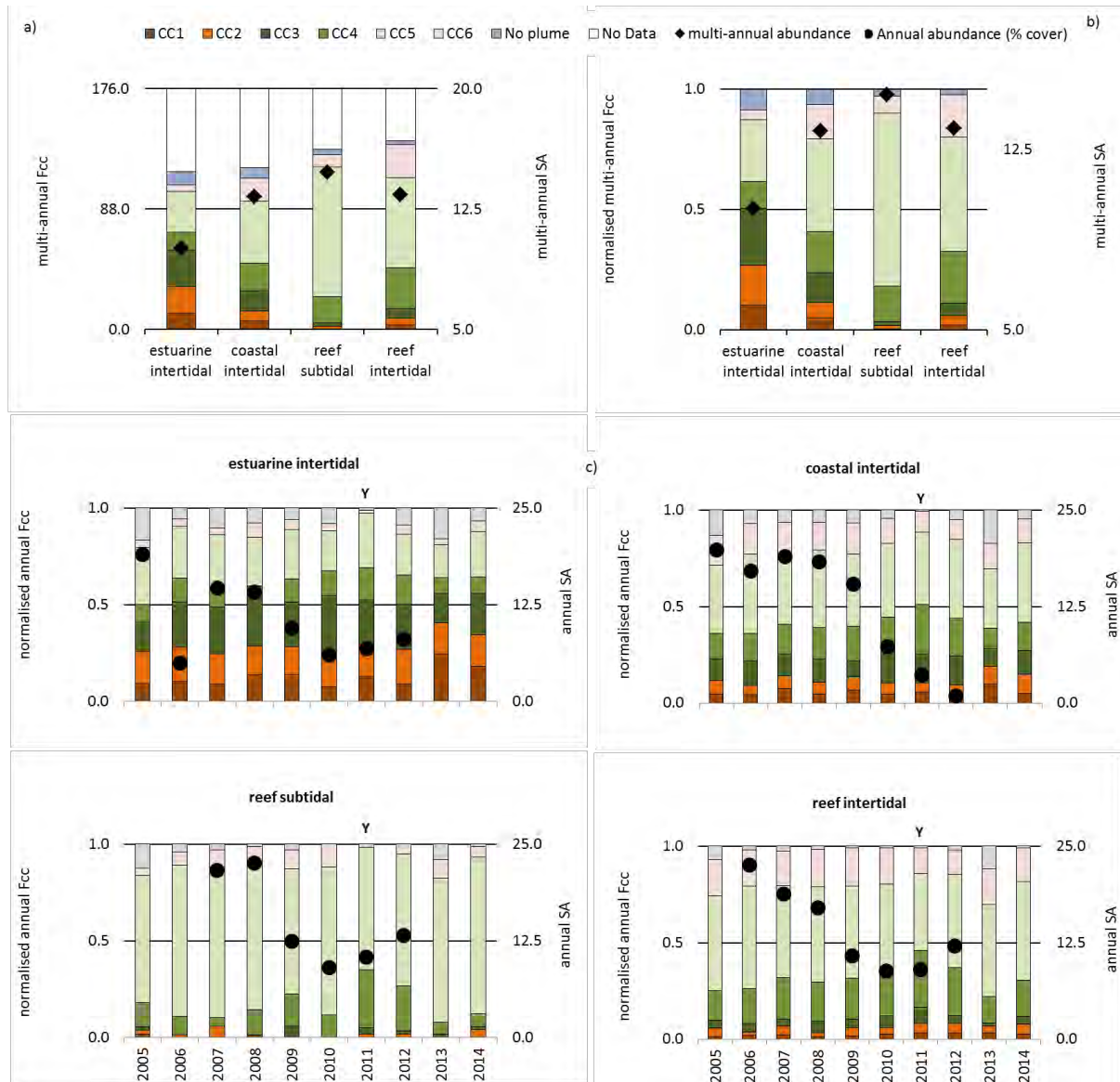


Figure 11. Plume conditions at each different seagrass habitats measured via the frequency of exposure to the plume colour classes (F_{cc}). Multi-annual (2005-12) seagrass abundance (SA, % cover) against (a) multi-annual F_{cc} and b) multi-annual normalised F_{cc} . c) Annual seagrass abundance (SA) 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).

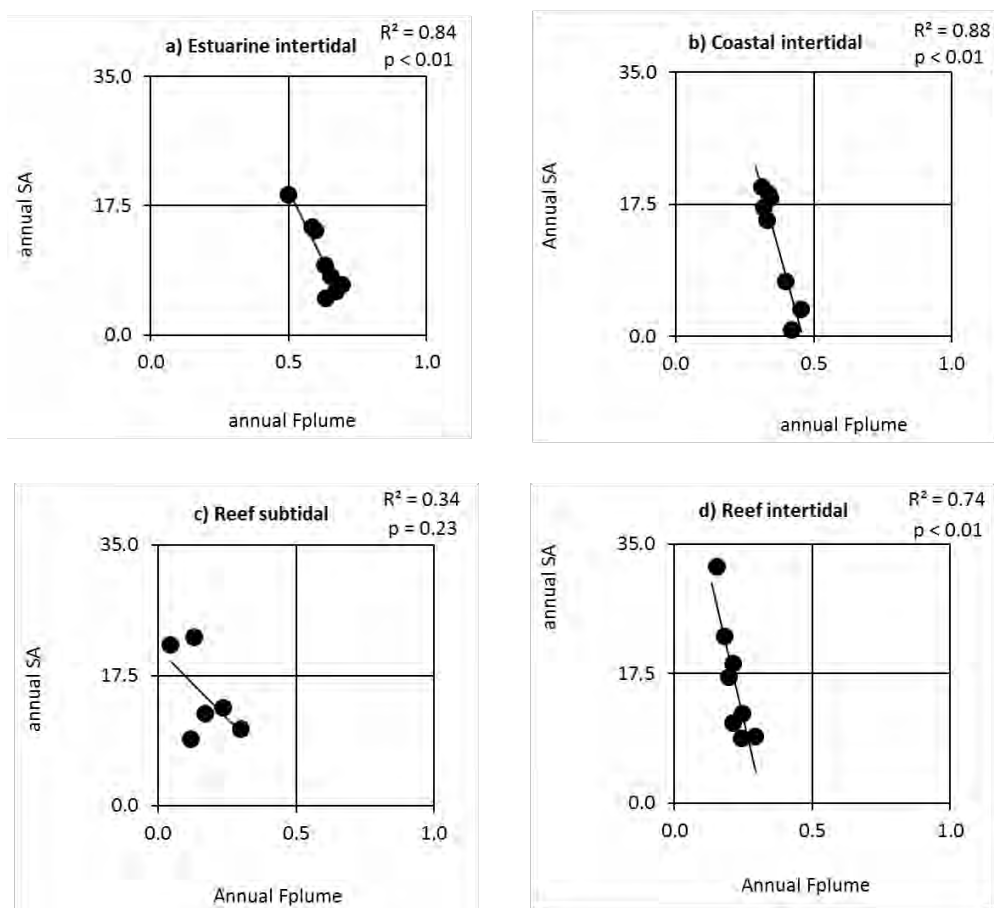


Figure 12. Correlation between the normalised annual F_{plume} and annual SA at the four seagrass habitats. F_{plume} is a proxy measurement that represents a colour class or combination of colour classes that relate directly to the plume condition at each seagrass locations. a) Estuarine intertidal: $F_{\text{plume}} = \text{CC1} + \text{CC2} + \text{CC3} + \text{CC4}$; b) coastal intertidal: $F_{\text{plume}} = \text{CC2} + \text{CC3} + \text{CC4}$ and reef c) subtidal and d) intertidal: $F_{\text{plume}} = \text{CC4}$

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 $F_{\text{(plume)}}$ 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 F_{plume} 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 Teixeira 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.

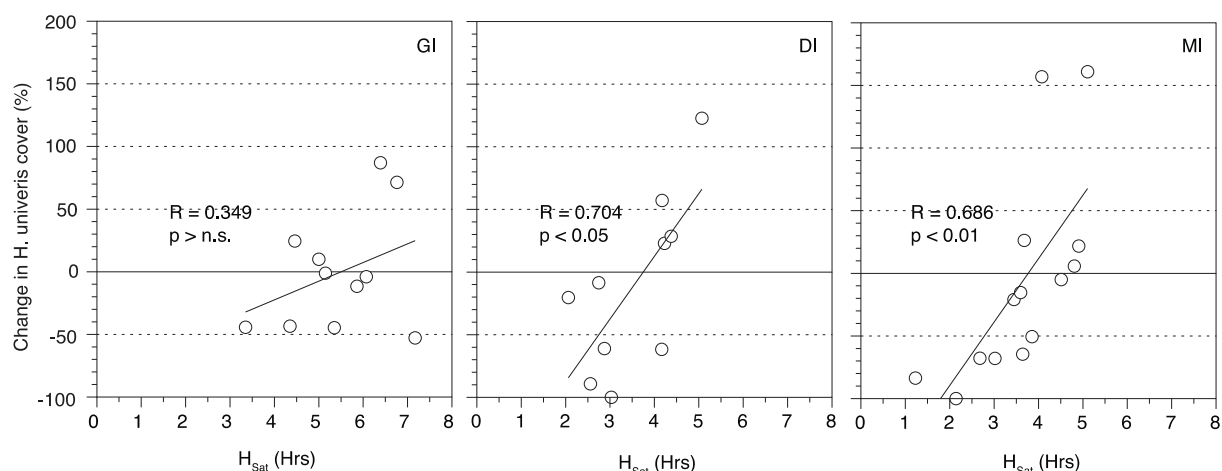


Figure 13. Change in the abundance of the seagrass *Halodule uninervis* and the hours of light saturated photosynthesis at three sites including Green Island (GI, stable meadow), Dunk Island (DI, complete seagrass loss in 2011), and Magnetic Island (MI, complete seagrass loss in 2011) from 2008 to 2011. 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 guidelines 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 <i>in situ</i> seagrass loss/ <i>in situ</i> light loggers	Daily light (mol m ⁻² d ⁻¹) Frequency of low light days Hsat	2.3 – 14.2 mol m ⁻² d ⁻¹ % days <14 mol m ⁻² d ⁻¹ 3.6 – 10.56 hrs	<ul style="list-style-type: none"> • In-situ changes in seagrass abundance (<i>H. uninervis</i>-dominated communities) were correlated with <i>in-situ</i> light • This relationship enabled identification of light thresholds for >50% loss • 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) Percent of surface light (%SI)	2.8 – 11.1 mol m ⁻² d ⁻¹ 10 – 40%	Sub-lethal (early warning) indicators tested (prior to shoot loss) C/N, $\delta^{13}\text{C}$ and rhizome carbohydrates were sensitive to light but:
g. Light x nutrients	Daily light (mol m ⁻² d ⁻¹) Percent of surface light (%SI)	2.2 – 8.7 mol m ⁻² d ⁻¹ 10 – 40%	<ul style="list-style-type: none"> • C/N only sensitive at low-moderate nutrient concentrations • $\delta^{13}\text{C}$ and rhizome carbohydrates were less sensitive to nutrients, and sensitivity was species dependant
d. Light x temperature	Daily light (mol m ⁻² d ⁻¹) Percent of surface light (%SI)	0 – 23 mol m ⁻² d ⁻¹ 0 – 70%	<ul style="list-style-type: none"> • Shoot loss and growth reductions at low light (0 – 10% SI) over 14 weeks • Time to complete seagrass measured: Ho<Zm<Cs<Hu • Light thresholds developed e.g. 3.8 mol m⁻² d⁻¹ for 14 weeks leads to 50% loss of seagrass • Experimental light thresholds were comparable to in-situ thresholds (<i>H. uninervis</i>, <i>Z. muelleri</i>)

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).

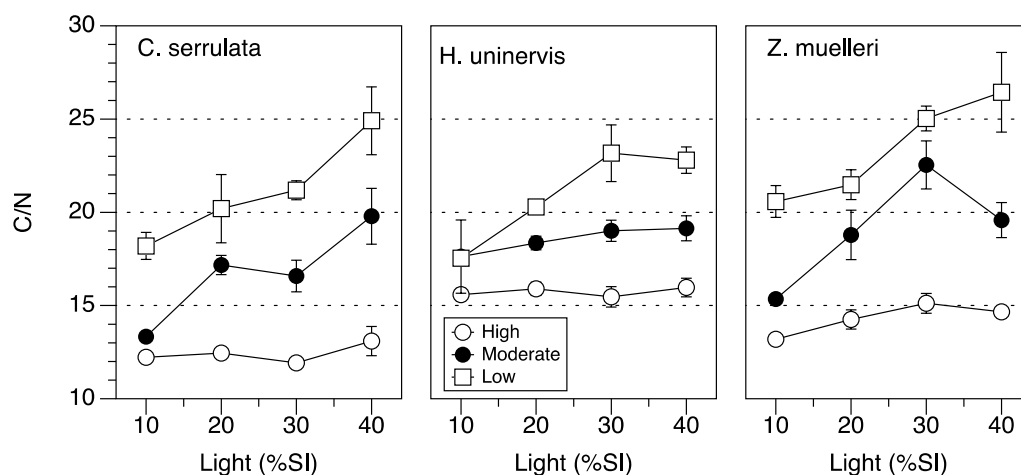


Figure 14. Tissue nutrient ratio of carbon (C) and nitrogen (N) at light levels ranging from 0-40% surface light at low (white square), moderate (black circle) and high nutrient (white circle) concentrations after 7 weeks in three seagrass species. C/N was highly sensitive to nutrient concentration and 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.

Thresholds and indicators of declining water quality as tools for tropical seagrass management

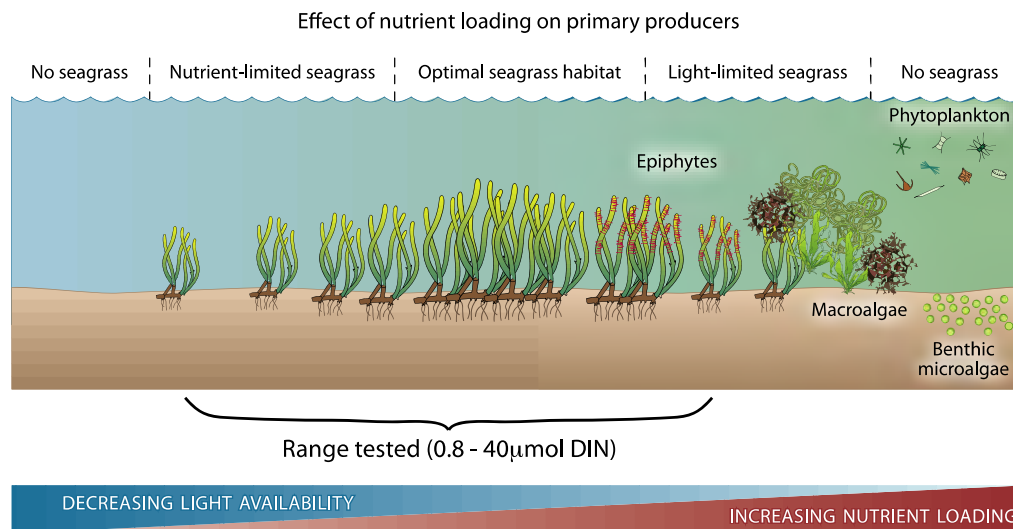


Figure 15. Nutrient response model for benthic primary producers showing the range in DIN ($\text{NH}_4^{++} \text{NO}_3^-$) tested in this study over the short-term (8 weeks). Plants were likely to be nutrient limited in the low (0.5 μmol) and nutrient replete in the high nutrient treatment (40 μmol).

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](https://doi.org/10.1371/journal.pone.0094014)

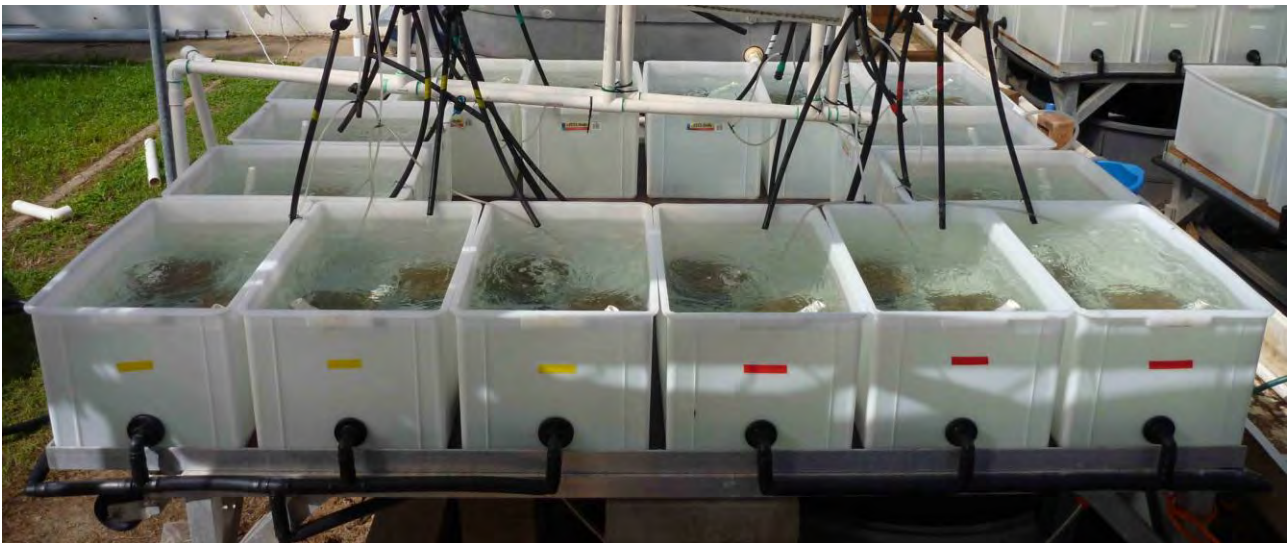


Figure 16. Experimental set-up for testing responses of seagrass to seagrass at a fine-scale resolution with salinity ranging from 3 PSU (almost freshwater) to 36 PSU (seawater) at 3 PSU increments. This was run for 10 weeks and three species were tested. n=4.

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 $\sim 4 \text{ mol m}^{-2} \text{ d}^{-1}$) 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).

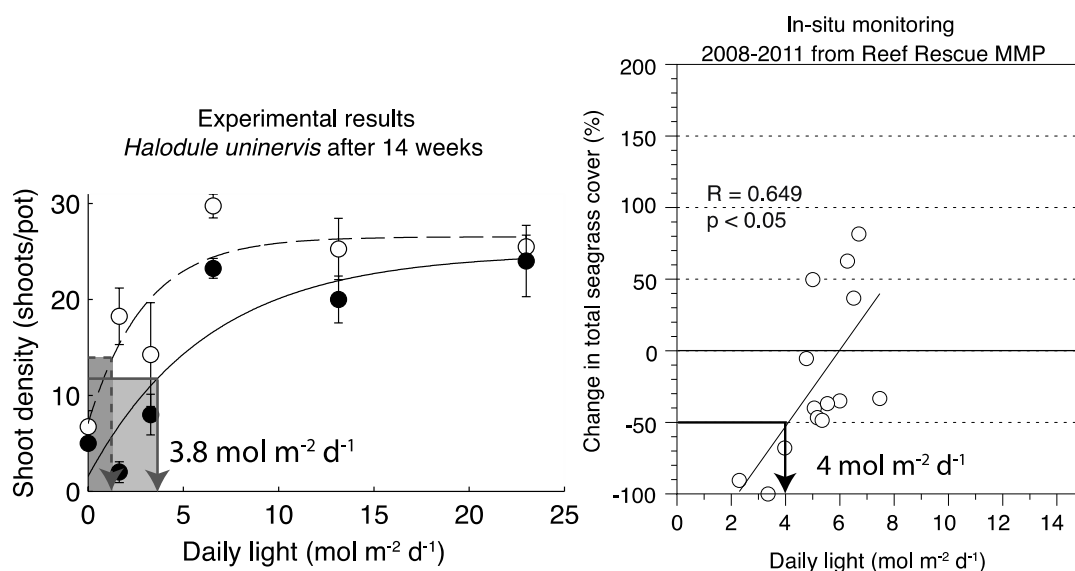


Figure 17. *Halodule uninervis* shoot density after 14 weeks in light levels ranging from 0 to $23 \text{ mol m}^{-2} \text{ d}^{-1}$ (70% sunlight) in warm ($\sim 27^\circ\text{C}$ black circles) and cool ($\sim 22^\circ\text{C}$ white circles) water. After 14 weeks, 50% loss of shoots was related to $3.8 \text{ mol m}^{-2} \text{ d}^{-1}$ (left). Change in seagrass cover (*Halodule uninervis*-dominated meadows) at Magnetic Island from 2008 to 2011 and Daily light ($I_d \text{ mol m}^{-2} \text{ d}^{-1}$). 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 $4 \text{ mol m}^{-2} \text{ d}^{-1}$ 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 $5.7 \text{ mol m}^{-2} \text{ d}^{-1}$, while *H. uninervis* $3.8 \text{ mol m}^{-2} \text{ d}^{-1}$ 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.

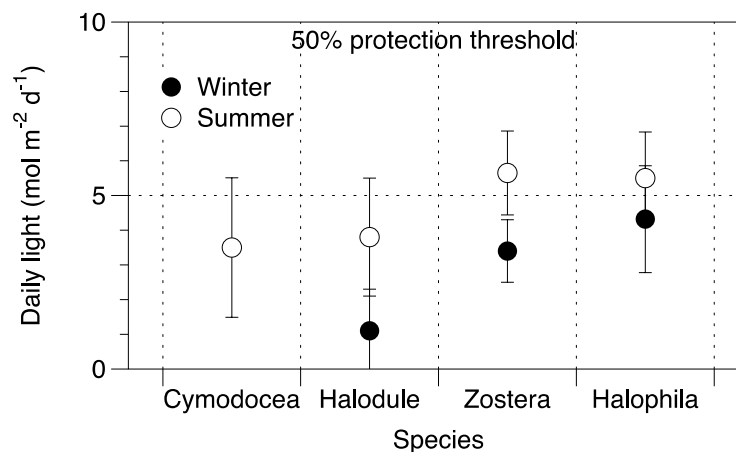


Figure 18. 50% decline threshold calculated from experimental light response curves (see 36). All species were exposed to a range of light treatments, and the light level causing 50% loss of shoots was calculated. This plot is the maximum threshold (generally 3 months) calculated over the experimental 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](https://doi.org/10.1371/journal.pone.0094014)

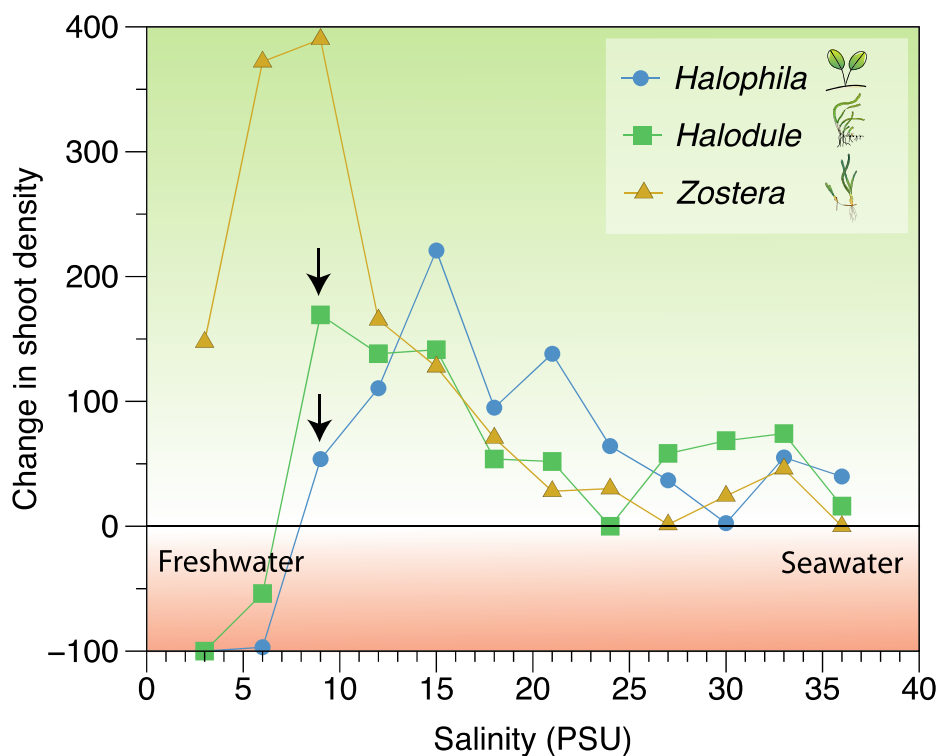


Figure 19. Response of seagrass shoot density to reduced salinity ranging from almost freshwater (3PSU) to seawater (36PSU) after 10 weeks exposure. At mid- to low-salinity (6-15) there were increases in shoot density, by up to 400%, which indicates mild hypo-salinity stress. Below hypo-salinity thresholds (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. $\delta^{13}\text{C}$ 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 $\delta^{13}\text{C}$ 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.

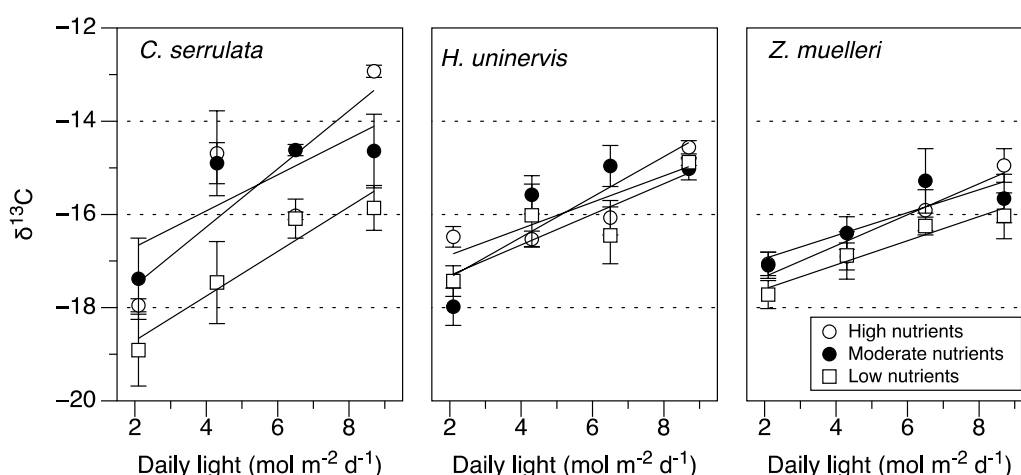































Figure 20. $\delta^{13}\text{C}$ at light levels ranging from 0-40% surface light at low (white square), moderate (black circle) and high nutrient (white circle) concentrations after 7 weeks in three seagrass species. $\delta^{13}\text{C}$ reduced in low light conditions in all nutrient treatments. Adapted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

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Attachments

Thresholds information sheet



National Environmental
Research Program

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.

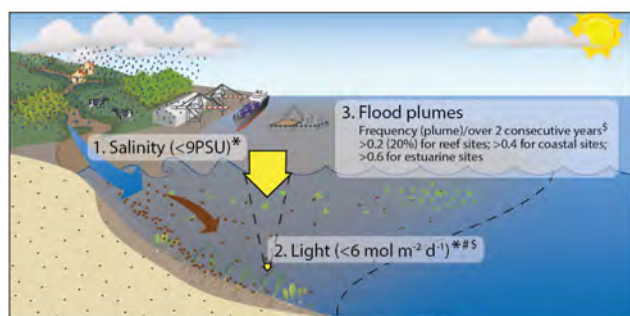


Figure 1. Salinity (1), light (2) and flood plume (3) thresholds were identified in this research.

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.

- 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

- $F(\text{plume})$ 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.

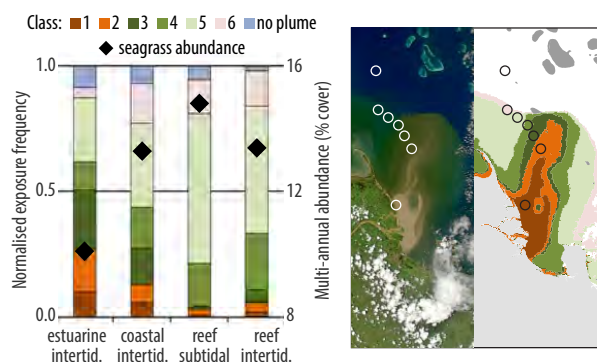


Figure 2. (Left) The combination of ocean colour classes measured (expressed as a normalized frequency) within each seagrass habitat and corresponding multi-annual (2007-2012) seagrass cover. (Right) MODIS image and corresponding colour-class map.

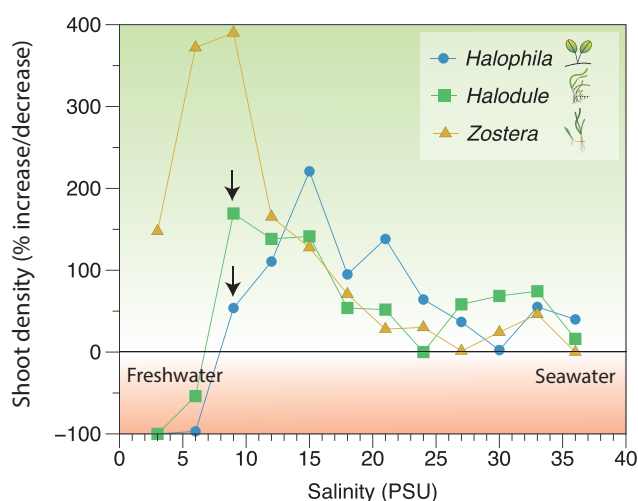


Figure 3. Change in seagrass shoot density (leaf pair density for *Halophila*) relative to week zero, after 10 weeks exposure to low salinity ranging from almost freshwater (3 PSU) to marine seawater (36 PSU). Density declined sharply below 9 PSU (↓).

Light thresholds

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

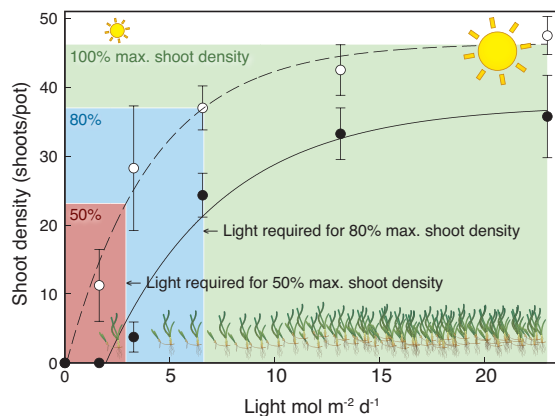
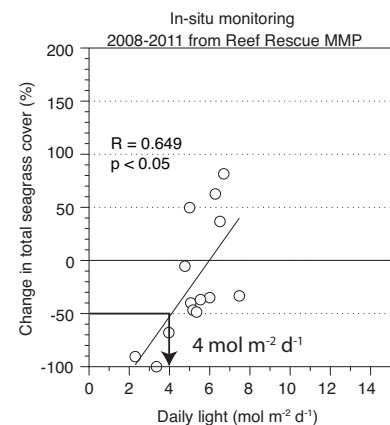


Figure 4. Shoot density of *Zostera* after almost 3 months at light levels ranging from no sunlight (complete darkness) to 23 mol m⁻² d⁻¹ (70% of full sunlight) in cool (white circles) and warm (black circles) water. From this data we can calculate the light level that caused 20% or 50% decline of seagrass.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

- 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 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.

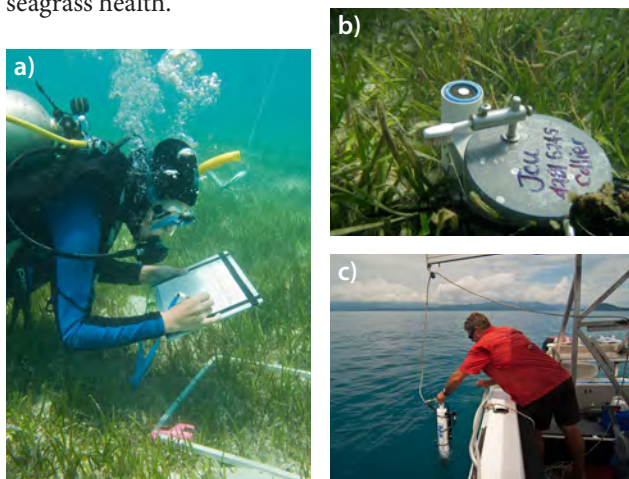


Figure 6. Long-term seagrass monitoring for the Reef Rescue MMP (a-c) includes ongoing measurement of light received in the seagrass canopy (b) as well as water quality (c). The development of water quality guidelines for seagrass will help managers reduce the associated impacts of dredging operations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

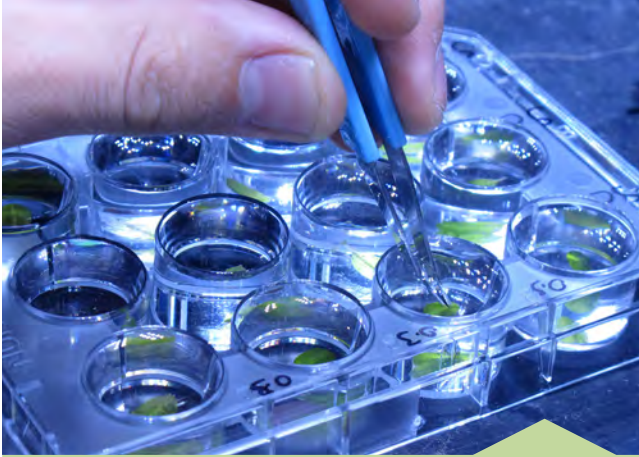
Development of water quality guidelines

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.



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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 investigated 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 guidelines 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.

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