Long-Term Seagrass Monitoring in Port Curtis and Rodds Bay:
Quarterly Permanent Transect Monitoring, 2009 to 2016

Final Report

Chartrand KM, Wells JN, & Rasheed MA

Report No. 17/03

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A report for Gladstone Port Corporation Limited

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Prepared by

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KEY FINDINGS

This report summarises the results from the 2009–2016 assessment of seagrass change at permanent transect sites established as part of the Western Basin Dredging and Disposal Project (WBDDP). One of the overarching goals of this program was to assess seagrass condition at the sites in relation to the pre-, during- and post- dredging phases in parallel with recorded in situ environmental data (i.e. light and temperature). This work complements long-term annual seagrass monitoring conducted since 2002 which examines and re-maps the full extent of seagrasses each November at the time of their maximum distribution (see Rasheed et al. 2017b).

- Seagrasses at permanent transects in Port Curtis and Rodds Bay showed distinct seasonal trends as well as significant inter-annual changes in seagrass percent cover, above-ground biomass and species composition.
- Over the duration of the monitoring program (since November 2009), Gladstone has received higher than average rainfall during the majority of wet seasons, punctuated by two of the most extreme flood events on record. This trend is likely part of larger shifts in the Southern Oscillation Index climatic conditions, which correlates with rainfall patterns over longer decadal timescales.
- Significant declines in seagrass abundance occurred at all sites during the monitoring program, particularly following the 2010/2011 flood event. Some sites showed substantial recovery, particularly in the Outer Harbour following the 2010/2011 flood event; however Inner Harbour sites have yet to recover to pre-flood levels.
- The timing of flood-related declines immediately prior to major WBDDP dredging activities makes it difficult to ascertain what additional impact dredging may have had on seagrass condition and recovery; however, *in situ* monitoring at permanent transect sites indicates that *Zostera muelleri* subsp. *capricorni* received enough light to meet their immediate growth requirements during the dredging program.
- Declines in Pelican Banks North and Pelican Banks South seagrass cover over the course of the program were explained best by a positive relationship with mean monthly light levels at the sites and negatively by three month cumulative river flow. Additional factors such as sediment dynamics and grazing pressure which were not monitored in the program are potential drivers of declines beyond river flow and light at these locations.
- In January 2013, the Calliope River discharged at record levels. Substantial declines in seagrass were detected across monitoring sites. Recovery over subsequent growing seasons has varied, with some sites remaining atypically low in seagrass cover; particular concern has been raised for the historically stable Pelican Banks *Z. muelleri* meadow which has substantially declined in the final two years of the permanent transect monitoring program at both monitoring sites in the meadow.
- Gladstone seagrasses were capable of the production of flowers, fruits and seeds. However this varied substantially between meadows and time of year. It is likely that propagule limitation at some sites may be inhibiting seagrass recovery following losses such as at Pelican Banks.
- Sediment seed banks for *Z. muelleri* were detected in the Inner and Outer Harbour and at Rodds Bay at all times since monitoring of seed banks began. A full analysis of this aspect of the program is outlined as part of a separate study and final report on seagrass seed banks and seed viability (Reason et al. 2017).

The primary goal of the seagrass permanent transect sites was to detect significant shifts in seagrass condition in the context of dredging activity and during the dredging period to act as sites to inform the active management of dredge operations. The program spans the pre-, during, and post-dredge phases providing a detailed description of seagrass abundance at both Inner and Outer Harbour locations within

Port Curtis with varying proximity to dredge activity and covers the range of species and meadow types typical throughout the wider Gladstone region. In summary, major rainfall and river flow events prior to, during, and following dredging did impact permanent transect site locations; however any interactive effect of dredging with significant storm and river flow events was unable to be determined, due to its co-incident timing.

Results of seagrass monitoring from pre-, during, and post-dredging phases of the project provide insight into the capacity of seagrass resilience to human activities and parallel environmental perturbations. If low levels of resilience detected at many sites persist, then the tools and thresholds established through the Port Curtis seagrass research programs will be critical in ensuring successful management of their recovery. Currently seagrasses have shown some capacity to recover from impacts in Port Curtis, but as has been seen in other Queensland locations repeated disturbances over multiple years may lead to long-term loss, with recovery trajectories far less certain. The extensive and detailed seagrass monitoring and research efforts in Port Curtis and Rodds Bay means we are well placed to understand these processes and can look to implement measures to reduce the chances of exacerbating natural impacts by human activities.

INTRODUCTION

Background

Seagrass meadows provide important ecosystem services in the coastal environment such as coastal protection, nutrient cycling and particle trapping (Costanza et al. 2014; Hemminga and Duarte 2000).They also provide additional economic value in terms of nursery and feeding habitats for commercial and recreational fisheries species (Blandon and zu Ermgassen 2014; Unsworth and Cullen 2010; Heck et al. 2003; Watson et al. 1993) and are considered to be internationally important due to the food resources they provide for endangered and vulnerable species such as dugong and turtles (Hughes et al. 2009). With globally developing carbon markets, the role that seagrasses play in sequestering carbon is also becoming more widely recognised (Lavery et al. 2013; Fourqurean et al. 2012; Pendleton et al. 2012). Despite this, seagrasses have been declining globally at ever increasing rates due to both natural and anthropogenic causes (Waycott et al. 2009). An assessment by an expert panel on the relative impacts of anthropogenic activities listed industrial and urban run-off, port development, and dredging as the main threats to seagrass ecosystems in the tropical Indo-Pacific region (Grech et al. 2012).

Large areas of seagrass were first identified in the Port Curtis region as part of broad scale state-wide seagrass surveys conducted in 1988 (Coles et al. 1992) and have been the subject of extensive monitoring since 2002 by the James Cook University TropWATER seagrass group (formally the Marine Ecology Group, Fisheries Queensland) in partnership with Gladstone Ports Corporation (GPC). In 2009, proposed developments in the Western Basin including a number of reclamations and a large-scale dredging campaign led to the requirement for more detailed information on inter- and intra-annual seagrass dynamics in the Port Curtis and Rodds Bay area.

The Western Basin Dredging and Disposal Project (WBDDP) posed a high level of environmental risk to marine habitats in the Gladstone area (referred to in this document as Port Curtis and Rodds Bay), particularly to seagrass meadows. The process of marine dredging elevates suspended solids within the water column, reducing light availability to marine habitats and can result in high levels of stress and mortality to benthic primary producers such as seagrasses (Erftemeijer and Lewis 2006). To effectively manage and mitigate the environmental risk from Port Curtis dredging activities, it was critical to understand the background relationships between seagrass and environmental variables on a local scale. Understanding the mechanistic response of seagrass to potential drivers also allows better distinctions between natural climate-induced declines and potential dredge-related impacts (Chartrand et al. 2016).

In 2009, GPC commissioned a suite of research and monitoring including quarterly assessments of seagrass condition at key monitoring sites throughout Port Curtis and in nearby Rodds Bay. The quarterly monitoring program adapts the established Seagrass-Watch protocols which utilise permanent intertidal transects at seagrass meadows to allow for repeated measures to be taken at any future point in time in a relatively simple manner (McKenzie et al. 2007). These permanent transects form the key sensitive receptor sites for assessing seagrass condition before, during and after the WBDDP and during the dredging program were used to inform active dredge management. In addition to the site-based assessments, all seagrasses in the region are re-mapped and assessed annually at the time of their peak distribution and abundance (September-December) as part of the long-term seagrass monitoring program established for the port in 2002 (Rasheed et al. 2017b).

This report details findings of the seagrass permanent transect site program detailing the quarterly and/or monthly monitoring for the Port Curtis and Rodds Bay Western Basin program from 2009 to 2016. Results of this report form part of a broader suite of seagrass investigations that also include quarterly seagrass and light assessments (Bryant et al. 2016b), annual seagrass mapping and long-term monitoring of seagrasses in

Port Curtis and Rodds Bay (Bryant et al. 2014b), a research and management program establishing the light requirements of Port Curtis seagrasses (Chartrand et al. 2016), seed bank studies (Bryant et al. 2016a) and research into establishing sub-lethal indicators of seagrass stress to use in seagrass management (Pernice et al. 2015; Schliep et al. 2015).

Study location

The program was established in October/November 2009, incorporating two existing seagrass assessment sites (Pelican Banks North and Rodds Bay; see Figure 1) monitored as part of the Seagrass-Watch Reef Rescue Marine Monitoring Program (MMP). Four new sites (Pelican Banks South, Facing Island, Fisherman's Landing, and Wiggins Island) were also established in key seagrass meadows (Figure 1). The locations selected encompass the range of representative seagrass meadow types within the area likely to be affected as well as providing a spatial range of sites relative to port developments. Throughout this report, sites are generally grouped into three regions; the Inner Harbour, the Outer Harbour and reference sites outside of the port in nearby Rodds Bay (Figure 1).

In December 2011, an additional permanent transect site was established at Wiggins Island due to the development of the Wiggins Island Coal Terminal near the original site. However, sampling at the new Wiggins Island site was restricted to helicopter surveys only due to access issues, and therefore only provides estimates of above-ground biomass and percent cover. Details are discussed in the next section. An additional site was also established at Facing Island in March 2012 where changes in meadow distribution resulted in the original site being situated at the marginal edge of the meadow. Both the original and new sites at these locations were monitored to determine if there were significant differences in seagrass communities between sites.

During the WBDDP several additional permanent transect sites were assessed for various periods of time to assess specific issues:

- An additional site adjacent to the main shipping channel (Redcliffe) assessed from August 2012 to November 2013;
- Three additional sites to monitor seagrass during dredging occurring as part of the Narrows Pipeline Crossing (Duffy Creek, Black Swan and Grahams Creek) were assessed from August 2012 to November 2013 (Bryant et al. 2014a).

The Black Swan site has continued to be monitored post November 2013 and was added to the program to provide an assessment site in the Narrows.

In May 2014, three new sites (Colosseum Inlet, South Trees and Quoin Island) were also established to monitor areas potentially impacted by the proposed Channel Duplication project. Three offshore monitoring locations were also established to monitor deep-water seagrasses potentially at risk during the project. Monitoring at these sites ceased in August 2015 and the results are available in Davies et al. (2015).

In this report, seagrass permanent transect sites are grouped into Inner and Outer Harbour locations (apart from the Rodds Bay site) in order to evaluate sites in close proximity versus those further removed from port infrastructure and WBDPP activity. In comparison, the Port Curtis Integrated Monitoring Program (PCIMP) and Gladstone Healthy Harbour Partnership (GHHP) breaks down seagrass meadows and other monitoring projects and locations by zone. The Inner Harbour as referred to in this report, incorporates the Narrows, Western Basin and Inner Harbour zones of these complementary programs. The Outer Harbour as discussed herein, equates to the Mid-Harbour in the PCIMP and GHHP zonation. The reason for the discrepancy is also to be consistent with interim reports for the permanent transect site program which have maintained this Inner and Outer Harbour terminology over the course of the 7 year program.

Sampling approach

Seagrass condition was monitored quarterly (typically February, May, August, and November) at each of the original permanent transect locations from October/November 2009. Four metrics were used to determine changes in seagrass meadows across spatial and temporal scales;

- Abundance and community composition (seagrass health)
- Elemental content of plants (seagrass tissue nutrients)
- Meadow reproductive status (seagrass resilience)
- Sexual above-ground productivity and asexual growth (seagrass productivity)

This report describes the results of monitoring conducted between October/November 2009 and November 2016 with a focus on pre-, during and post- dredge phases and the state of seagrass condition in the context of the WBDDP.

Seagrass condition monitoring is assessed in relation to measured water quality parameters at each permanent transect location. Well established as key drivers of seagrass growth (Ralph et al. 2007, Lee et al. 2007), benthic light and temperature were continuously collected at the seagrass canopy in order to assess the relationship in the environment directly within sensitive receptor sites to seagrass condition.

Monitoring seagrass condition against water quality at permanent transect sites was done within the context of the dredging program. Capital dredging started in May 2011 with localised backhoe dredging near Curtis Island and the primary dredging campaign began in earnest in September 2011 and was completed by May 2013.

Throughout the report, seasonal cycles are defined according to the climate-induced pattern of growth and senescence during the year on the east coast of Queensland (Chartrand et al. 2012; McKenzie 1994). Two generalised seasons are distinguished: the growing season, defined as July to January, typifies seagrasses natural increase in biomass and distribution as ideal growing conditions provide a period of opportunistic expansion; and the senescent season, February to June, when seagrasses typically retract and rely more on stores or seeds to get through wet season conditions, including flooding and poor water quality (Chartrand et al. 2012).

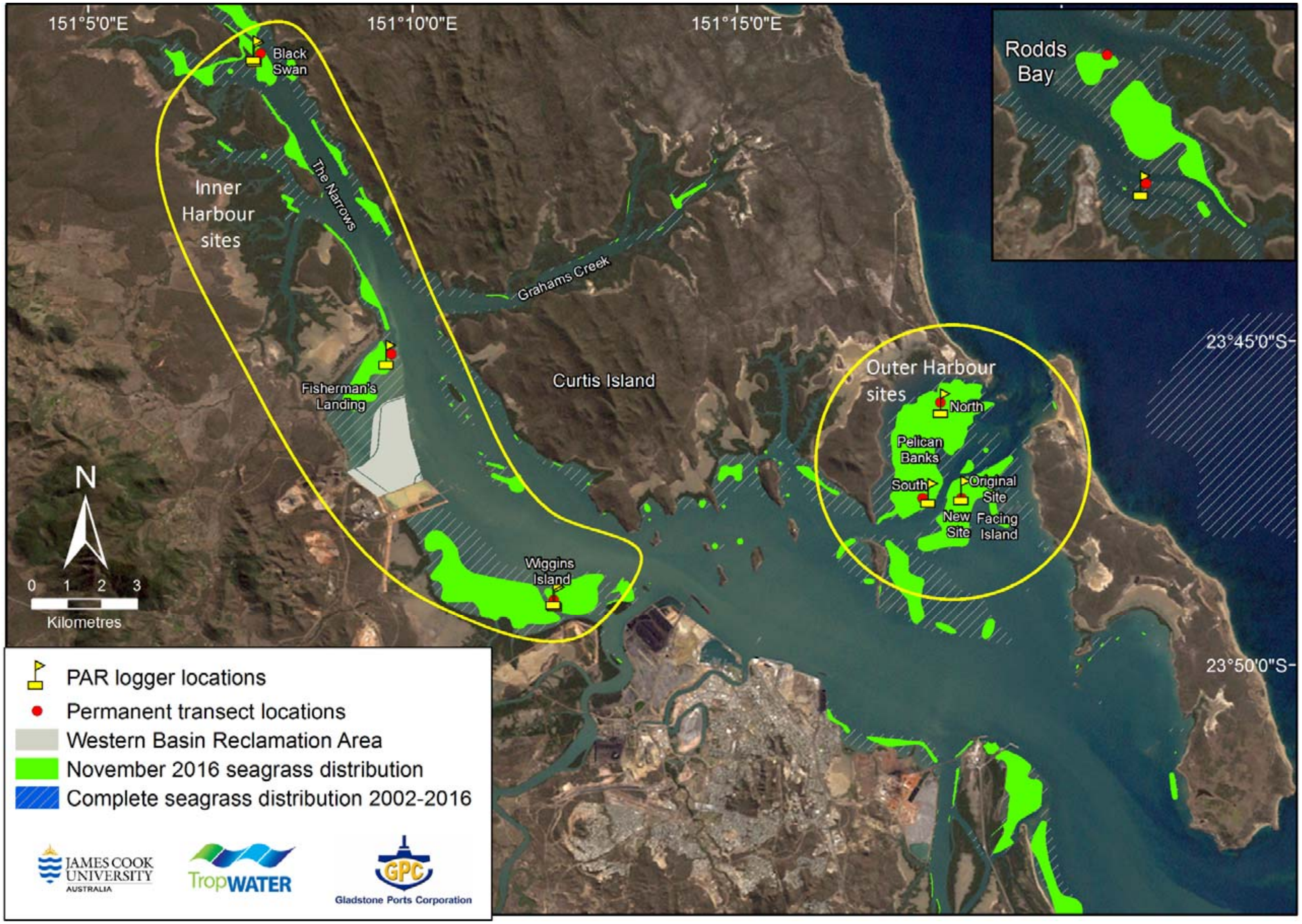


Figure 1: Permanent seagrass monitoring transects and light logger locations in Port Curtis and Rodds Bay, 2016.

1 SEAGRASS HEALTH

1.1 Background and approach

Seasonal changes in seagrass abundance and species composition have been well documented in Port Curtis and Rodds Bay. Bi-annual surveys of seagrass distribution have shown that seagrasses in the region are generally at their peak in distribution and abundance during the late spring/early summer and decline during the winter months (Bryant et al. 2014b; Davies et al. 2013; Chartrand et al. 2012; Rasheed et al. 2012; Chartrand et al. 2011) consistent with seagrasses in other tropical and subtropical areas of Queensland (Rasheed 2004; 1999; McKenzie 1994; Mellors et al. 1993).

While annual and semi-annual monitoring has provided invaluable baseline information on seagrasses at both the peak and lowest point of their distribution, there was little information on the seasonal trends occurring over finer timescales throughout Port Curtis and Rodds Bay. A focused and more frequent assessment of seagrasses was essential in establishing seasonal trends and also to provide a network of permanent transect sites for managing potential impacts of the WBDDP.

1.2 Methods

1.2.1 Quarterly sampling

Each of the monitoring locations was sampled quarterly over the spring low tides around February, May, August and November each year (2009-16). Survey methods followed Seagrass-Watch standard methodology (McKenzie et al. 2007; see also www.seagrasswatch.org) with the addition of seagrass biomass assessments (Mellors 1991).

Each sample location (e.g. Rodds Bay) consisted of one or two sites (e.g. Rodds Bay 1 and Rodds Bay 2), each comprised of 3 x 50 m transects. A site was defined as a 50 m x 50 m area within a relatively homogenous section of the meadow (McKenzie et al. 2000). At each transect, information was recorded on seagrass percent cover, above-ground biomass, species composition, canopy height, macro-algae cover and epiphyte cover. Information was obtained within a 0.25 m^2 sampling quadrat placed at 0 m and then every 5 m along transects (eleven sampling points per transect) (Figure 2).

To assess the above-ground biomass, observers assigned a rank describing the above-ground biomass of seagrass for each quadrat while referencing a set of photographs of seagrass plots for which the above-ground biomass had previously been measured. The observer also ranked a series of 'calibration' quadrats; a set of photographs of seagrass plots where the actual biomass had been determined in the laboratory. A regression of ranks and biomass from these calibration quadrats was generated for the observer and applied to the ranks given during the survey. Biomass ranks were converted into above-ground biomass estimates in grams dry weight per square metre (g DW m^{-2}).

Due to access issues for on-ground sampling at some locations, the Black Swan site and the new site at Wiggins Island were surveyed via helicopter to avoid damaging the sites by sampling on foot (Figure 2). For these sites, only seagrass percent cover and above-ground biomass data were collected during quarterly surveys.

1.2.2 Additional monthly sampling

In September 2011, GPC commissioned additional monthly surveys outside of the regular quarterly monitoring to provide more frequent assessments of seagrass condition during dredging operations. These additional monthly surveys were conducted aerially to avoid damaging sites through repeated on-ground sampling. The surveys occurred over two days each month outside of regular quarterly monitoring surveys from September 2012 until December 2013. Seagrass abundance assessments for the additional monthly surveys were confined to seagrass percent cover estimates.

1.2.3 Quality assurance and quality control

To ensure strict quality assurance and quality control for seagrass percent cover data, a second observer assigned percent cover estimates using a subset of photos of the plots taken in the field to ensure there were no major discrepancies between observers (outside of a margin of 10%). Quality assurance and quality control procedures for biomass estimates are described in section 1.2.1 above.

Figure 2: Monitoring along the transect tape at a permanent transect monitoring site during a quarterly 'on ground' survey (left); and estimating seagrass abundance using aerial survey methods (right).

1.3 Results

1.3.1 Inner Harbour

Fisherman's Landing

Seagrass percent cover at Fisherman's Landing permanent transect sites was light and patchy (< 7%) over the duration of the monitoring program (Figure 3a). Percent cover generally peaked between November and December, declining early in the year as seagrasses entered the senescent season. After a major flood event in early 2011, seagrasses at the site declined to extremely low levels but recovered well by the end of the growing season. In the following years, seagrass cover followed a seasonal trend, however while recovery during the growing season gradually increased from 2012, cover remained well below levels recorded in 2009 and 2011 until November 2016, when seagrass reached the highest percent cover since December 2011 (Figure 3a).

Seagrass above-ground biomass followed a similar seasonal trend to percent cover with extremely low (< 2g DW m^{-2}) levels of biomass recorded when seagrass was present (Figure 3a). These levels are within the range of average biomass found more broadly across the Fisherman's Landing meadow during annual monitoring surveys conducted since 2004, where average biomass was also < 2 g DW m^{-2} for the majority of years sampled (Davies et al. 2016).

Historically, *Z. muelleri* subsp. *capricorni* (herein referred to as *Z. muelleri*) has been consistently present and in relatively high proportions in the Fisherman's Landing meadow based on whole-meadow monitoring starting in 2002 (Rasheed et al. 2017b); however *Z. muelleri* presence at the permanent transect site has been significantly lower in comparison since monitoring began in October 2009. The one exception was following the 2010/2011 flood event when only a few shoots of *Z. muelleri* remained at the site and *Halophila ovalis* was absent until detected in August 2011 surveys (Figure 3b). Since 2012, *H. ovalis* has persisted after the wet season with Z. *muelleri* appearing late in the growing season (2012-2014; 2016) or not at all (2015). From 2012 to 2016, the proportion of Z. *muelleri* appearing during the growing season has generally declined, with Z. *muelleri* comprising less than 1% of the species composition in the 2016 growing season (Figure 3b).

a)

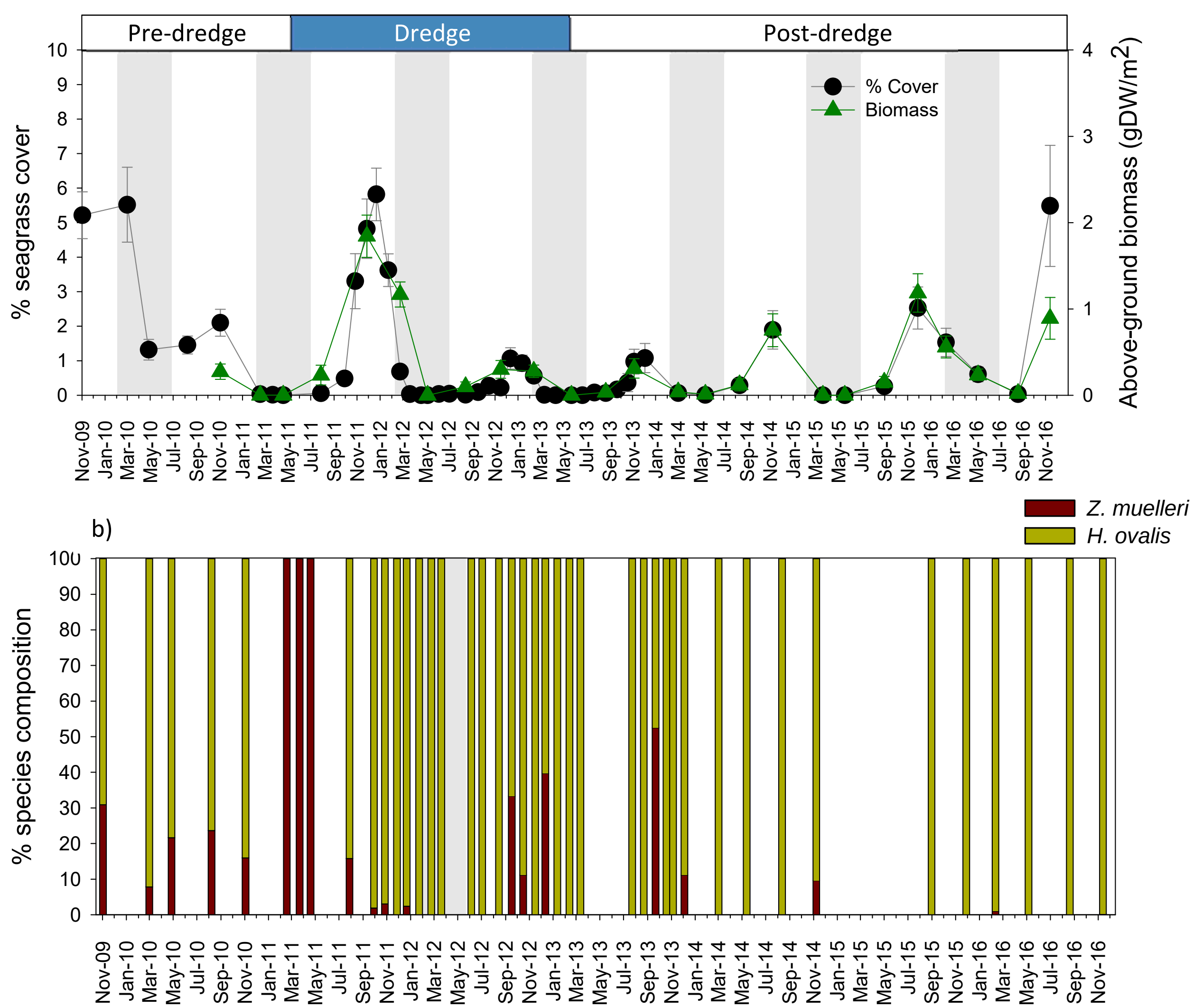


Figure 3: a) Seagrass abundance (mean ± SE) measured as above ground biomass (g DW m^{-2} d^{-1}) and percent cover of shoots and b) seagrass species composition at permanent transect sites (pooled) at

Fisherman's Landing, November 2009 – November 2016. Shaded area represents the seagrass senescent season.

Wiggins Island

When monitoring commenced in October 2009, seagrass percent cover at the original Wiggins Island site near the mouth of the Calliope River was substantially higher (~30%) than in subsequent years (Figure 4a). Seagrass cover declined following substantial rainfall and river discharges in 2010 (see Figure 23 in section 5) and remained at a significantly lower level until this original site was decommissioned in May 2014 due to the development of the Wiggins Island Coal Terminal near the site.

An additional permanent transect site was established on the western side of the bank in December 2011. From December 2011 to July 2012, the new site was higher in seagrass percent cover. However, from August 2012 until the original site was decommissioned there was little difference between sites (Figure 4b). Seagrass followed established seasonal trends with peaks during the growing season generally increasing since 2013 following additional early 2013 flooding from the Calliope River. Above-ground biomass mirrored percent cover for both sites, remaining below 2 g DW m^{-2} since November 2012 inclusive of 2016 (Figure 4b).

The species composition over the first few years of the program shifted seasonally between a *Z. muelleri* and a *H. ovalis* dominated community. *Z. muelleri* was the dominant species over the majority of the growing season before dominance in the senescent season by *H. ovalis.* This occurred as *Z. muelleri* declined during the senescent season and *H. ovalis* maintained or increased in cover at the original site (Figure 5a). At the new site, *Z. muelleri* was also the historically dominant species based on long-term monitoring of the meadow (Rasheed et al. 2017b), however declines from the 2010/2011 floods from the Calliope River impacted the species composition of the permanent transect site from the start of the transect monitoring at this location. *Z. muelleri* did return to the site in mid-2012 but remained a smaller component of the meadow compared to *H. ovalis* at the peak of the growing season each September and was again observed in 2016 (Figure 5b).

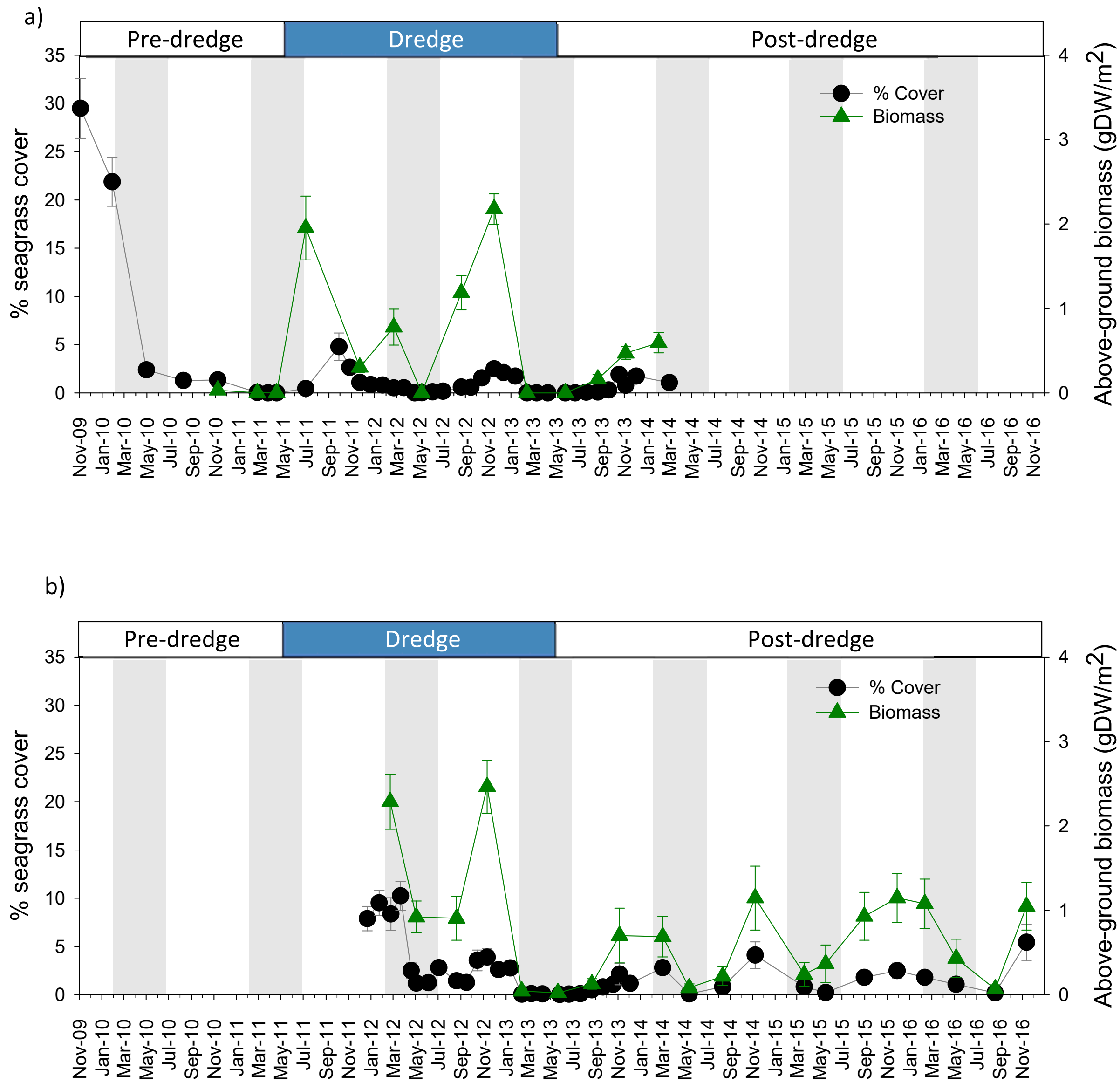


Figure 4: Seagrass abundance (mean ± SE) measured as above ground biomass (g DW m^{-2} d^{-1}) and percent cover of shoots at the a) original and b) new permanent transect site at Wiggins Island, November 2009 – November 2016. Shaded area represents the seagrass senescent season.

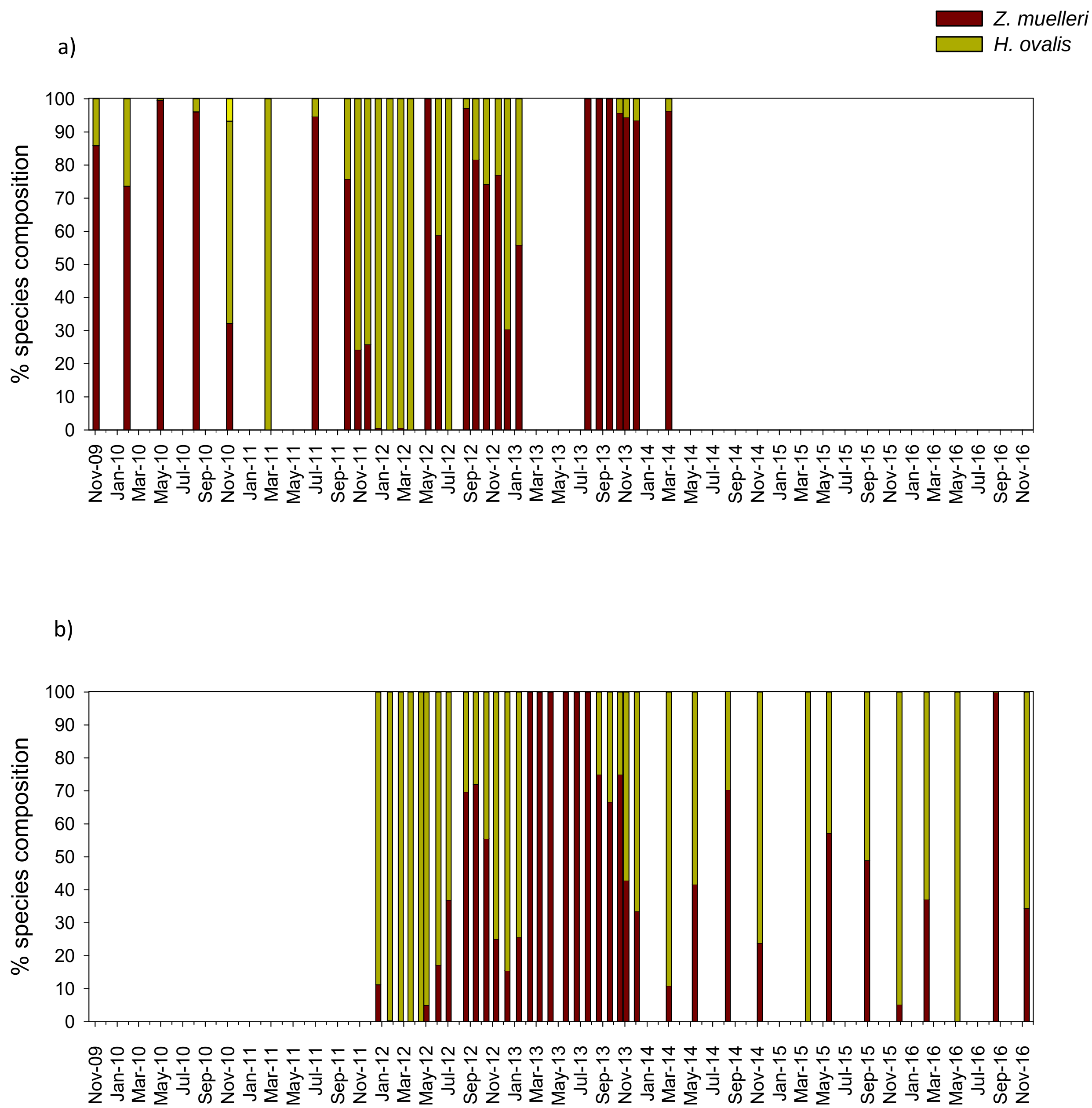


Figure 5: Seagrass species composition at the a) original and b) new permanent transect site at Wiggins Island, November 2009 – November 2016.

Narrows Region additional monitoring sites

In August 2012, three new monitoring sites were established in the Narrows region as part of monitoring for the Narrows pipeline dredging project. Of these new sites, Black Swan Island was the only site to have substantial seagrass cover. Seagrass cover at the Black Swan Island site declined over the latter part of 2012 and was extremely low in the peak of the 2013 growing season (<1%) (Figure 6a). From 2014 to 2016, seagrass cover remained lower compared with the start of monitoring in 2012, but did increase from minute amounts in 2013. Above-ground biomass estimates measured from February 2013 are in agreement with trend in percent cover over the four years of monitoring. The site has been dominated by *Z. muelleri* with *H. ovalis* with a smaller component of *H. ovalis*. *H. decipiens* was also observed but only in February 2016 in very low amounts (Figure 6b). As with other Inner Harbour locations, *Z. muelleri* die-back during the senescent season and growth of *H. ovalis* led to a proportionally greater amount of *H. ovalis* from approximately December to June each year (Figure 6b).

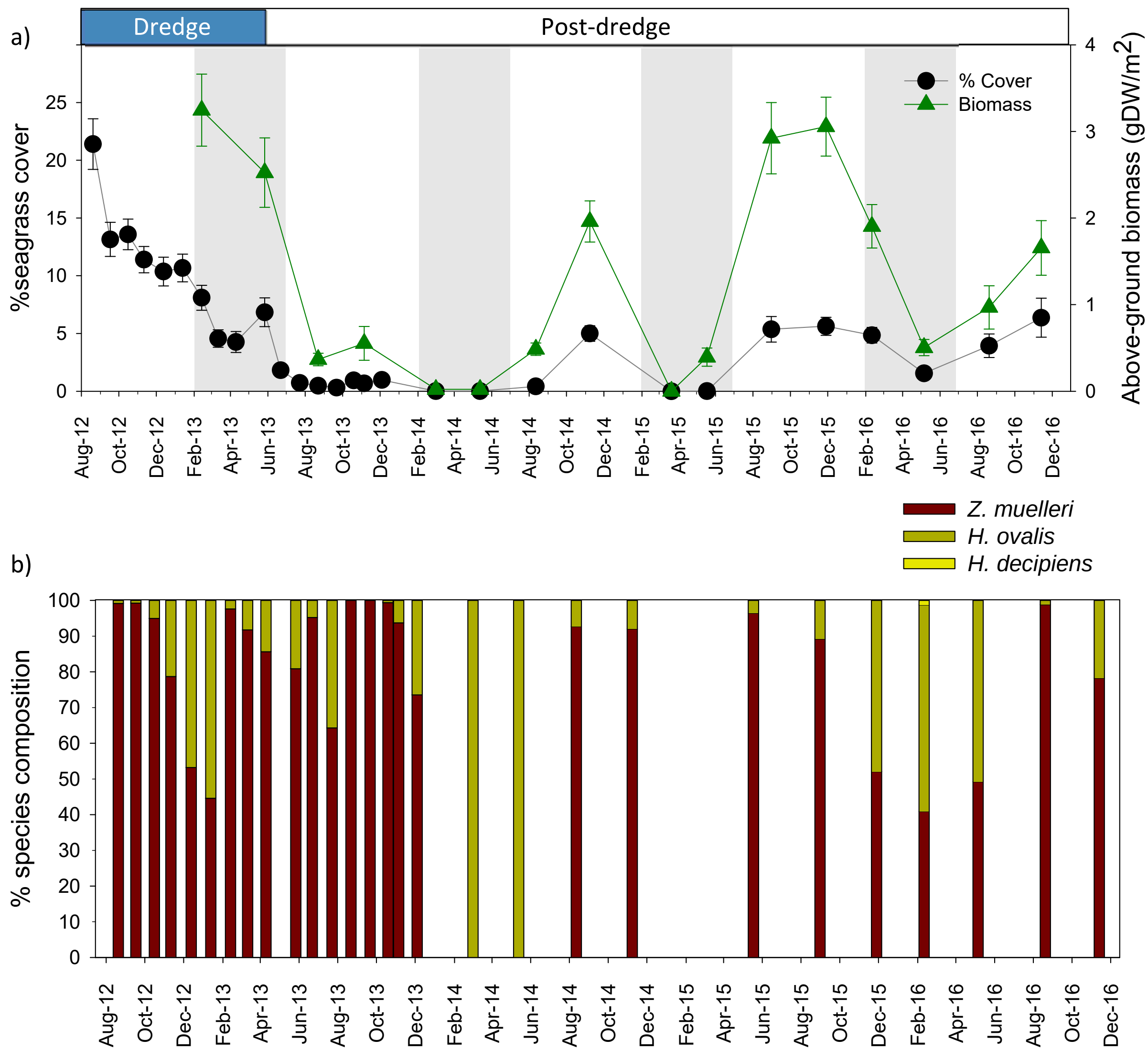


Figure 6: a) Seagrass abundance (mean ± SE) measured as above ground biomass (g DW m^{-2} d^{-1}) and percent cover of shoots and b) seagrass species composition at Black Swan Island permanent transect site, August 2012 - November 2016. Shaded area represents the seagrass senescent season

1.3.2 Outer Harbour

Pelican Banks North

Seagrass abundance data has been collected at permanent transects at Pelican Banks North since August 2005 as part of the Reef Rescue MMP and more recently as part of the Western Basin monitoring program. This long time series (relative to other sites) provides a historical context in which to place trends emerging over recent years of monitoring.

In early 2006, the Reef Rescue MMP reported a significant loss of seagrass at the site, declining in percent cover by ~80% since the previous year (Figure 7a). By late 2006 however, seagrasses had already begun to recover and by 2008 had recovered to levels seen when the site was established (Figure 7a). Since this time, seagrass percent cover has followed the pattern of growth observed throughout the Port Curtis and Rodds Bay region with increases over the growing season and sharp declines in percent cover following the wet season (Figure 7a). The same pattern held for 2016. The declines in percent cover in early 2011 and 2013 following major flood events were the most marked seen since 2006. Seagrass percent cover recovered well over the 2013 growing season; however, seagrass cover in the subsequent 2014 growing season did not recover to typical seasonal peaks. From the 2014 growing season onwards, there has been a gradual decline in both percent cover and biomass at the site (Figure 7a). A reduction in the magnitude of the recovery during the peak growing season from the senescent season, as well as the lower base from which seagrasses begin their recovery, has led to the lowest peak in percent cover since 2006 during the growing season in 2016.

Above-ground biomass has shown a similar trend. In November 2013 biomass peaked, however seagrass made minimal recovery over the 2014 growing season and since that time has remained generally low for this site (< 10 g DW m^{-2}) (Figure 7a). In 2016, there was no recovery observed over the growing season. Annual mapping of the entire bank in 2016 also revealed the lowest average biomass over the course of monitoring (since 2002) (Rasheed et al. 2017b). While Pelican Banks North seagrasses remain the most abundant in terms of percent cover, the substantial decline in biomass over the last few years is alarming and this meadow no longer has the highest biomass of the monitoring meadows regularly assessed.

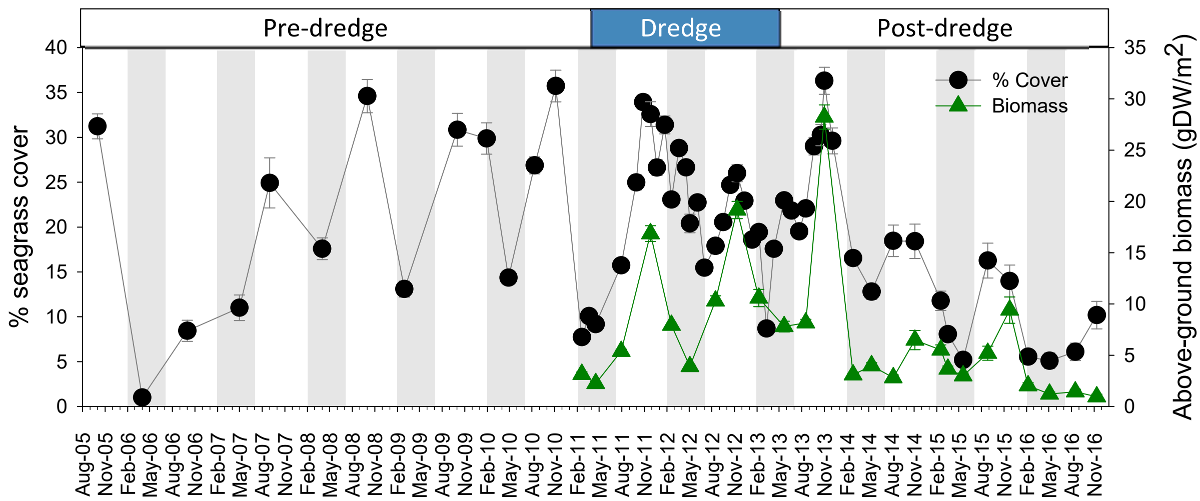
The species composition remains almost exclusively *Z. muelleri*, with only very minor proportions of *H. ovalis* present at times, though since 2014 *H. ovalis* has persisted in in the meadow throughout the year (Figure 8a).

Pelican Banks South

Seagrass percent cover at Pelican Banks South followed a similar pattern to Pelican Banks North with significant declines following the 2010 wet season (Figure 7b). Unlike Pelican Banks North, the southern sites failed to recover to initial levels recorded in October 2009 (Figure 7b). Seagrass percent cover dropped further in 2013 and fell further to below 1% from May 2014 and remained at extremely low levels with < 3% recorded in November 2016 (Figure 7b). Seagrass biomass followed a similar trend to percent cover and remained extremely low since the 2014 senescent season, with minimal seasonal recovery at this site through 2016.

The species composition at Pelican Banks South from 2009 to 2013 was predominantly *Z. muelleri* with substantially greater proportions of other species present (*Halophila decipiens*, *H. ovalis* and *Halodule uninervis*) compared with Pelican Banks North (Figure 8b). Coinciding with the sharp declines in biomass and percent cover in 2014 there was a shift in species composition; *Z. muelleri* was absent as of November 2016 and *H. uninervis* was the dominant species with greater proportions of *Halophila* species also present (Figure 8b). *H. spinulosa* did appear once at this location in March 2015 but has not been found occurring since.

a)



b)

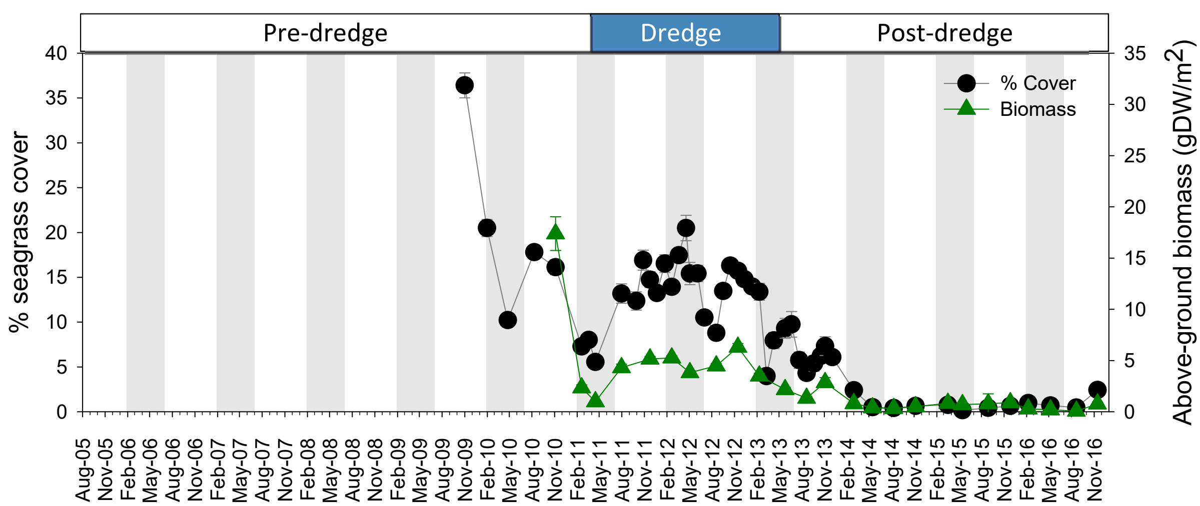


Figure 7: Seagrass abundance (mean ± SE) measured as above ground biomass (g DW m^{-2} d^{-1}) and percent cover of shoots at a) Pelican Banks North and b) Pelican Banks South permanent transect sites (pooled), August 2005 – November 2016. Shaded area represents the seagrass senescent season. Data prior to October 2009 were collected for the Reef Rescue MMP (McKenzie and Unsworth 2009).

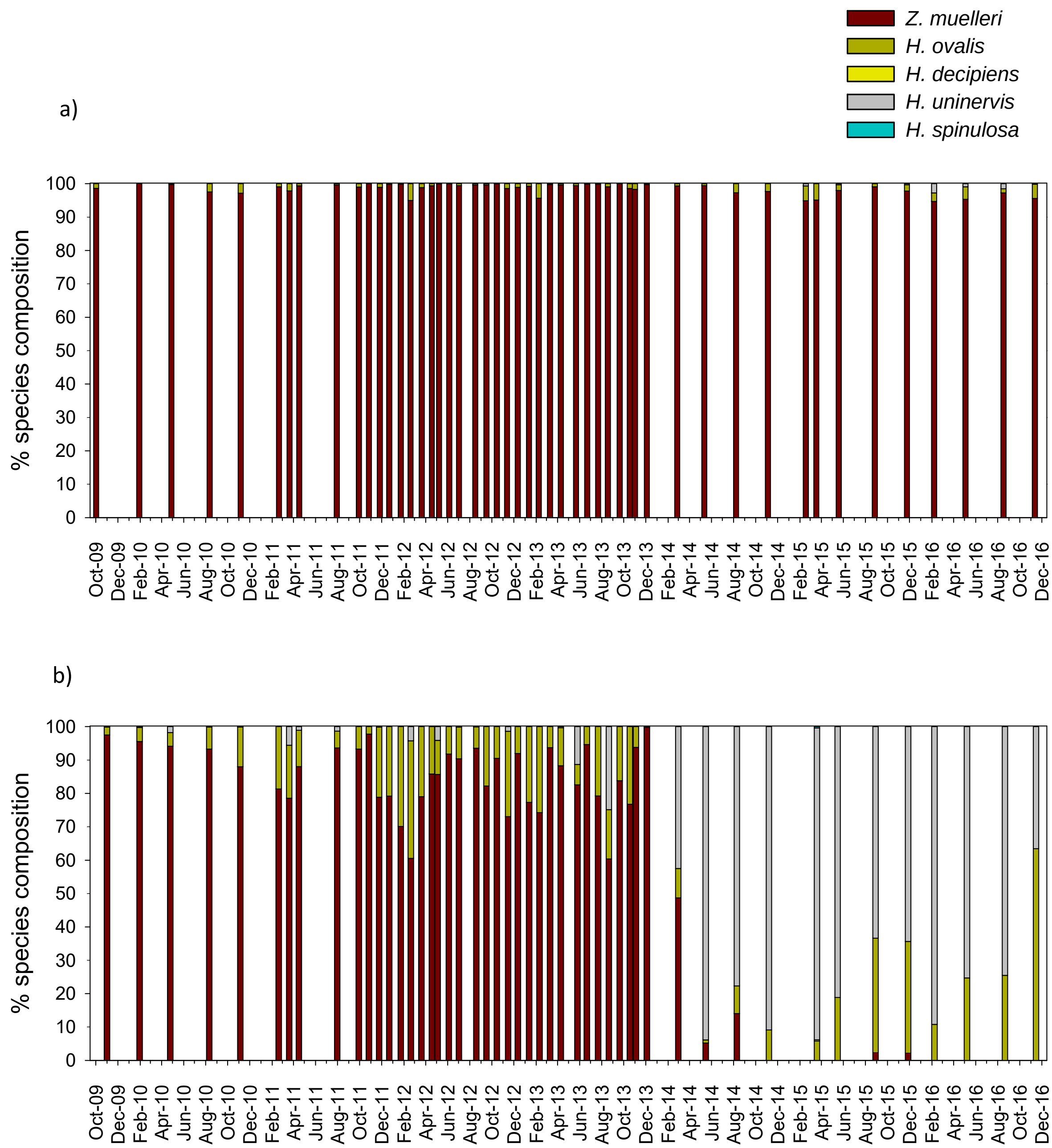


Figure 8: Seagrass species composition at a) Pelican Banks North b) and Pelican Banks South permanent transect sites (pooled), October 2009 – November 2016.

Facing Island

Seagrass abundance at the original Facing Island site declined significantly from initial levels recorded when the site was established in October 2009 (Figure 9a). By the beginning of 2011, seagrass percent cover had declined by more than 80%. While cover remained at low levels (<5%), seagrass at this site slightly increased in the most recent surveys in late 2016. Seagrass above-ground biomass followed the same overall site trend with biomass in November 2016 reaching the highest level recorded since 2011 (Figure 9a).

Changes to the shape and distribution of this seagrass meadow identified as part of semi-annual surveys meant that the original site was located on the marginal edge of the meadow (McCormack and Rasheed 2012). In March 2012, a new site was established adjacent to the original site but further to the centre of the meadow. This enabled a greater understanding of trends in seagrass health within the meadow itself, compared to the meadows marginal edge.

Percent cover at the new site was significantly higher than the original site until a sharp decline in early 2014 (Figure 9b). While some recovery was recorded over the 2015 growing season, seagrass cover at this site remained reduced compared to levels recorded in 2012 and 2013. Above-ground biomass was extremely low (< 2 g DW m^{-2}) since February 2013 but followed a pattern of seasonal recovery with a similar slight increase in seagrass cover in 2016 as observed at the original site (<5%) .

Species composition at both sites followed a similar pattern (Figure 10). Both sites were historically (2009-2013) comprised of a variable mixture of *Z. muelleri*, *H. uninervis* and *Halophila* species, although the new site had a higher proportion of *Z. muelleri* compared with the original site. Since the decline in abundance in 2014 and through 2016, the little seagrass remaining at transects was dominated by *H. uninervis* with some *Halophila* species and *Z. muelleri* appearing over the growing seasons (Figure 10).

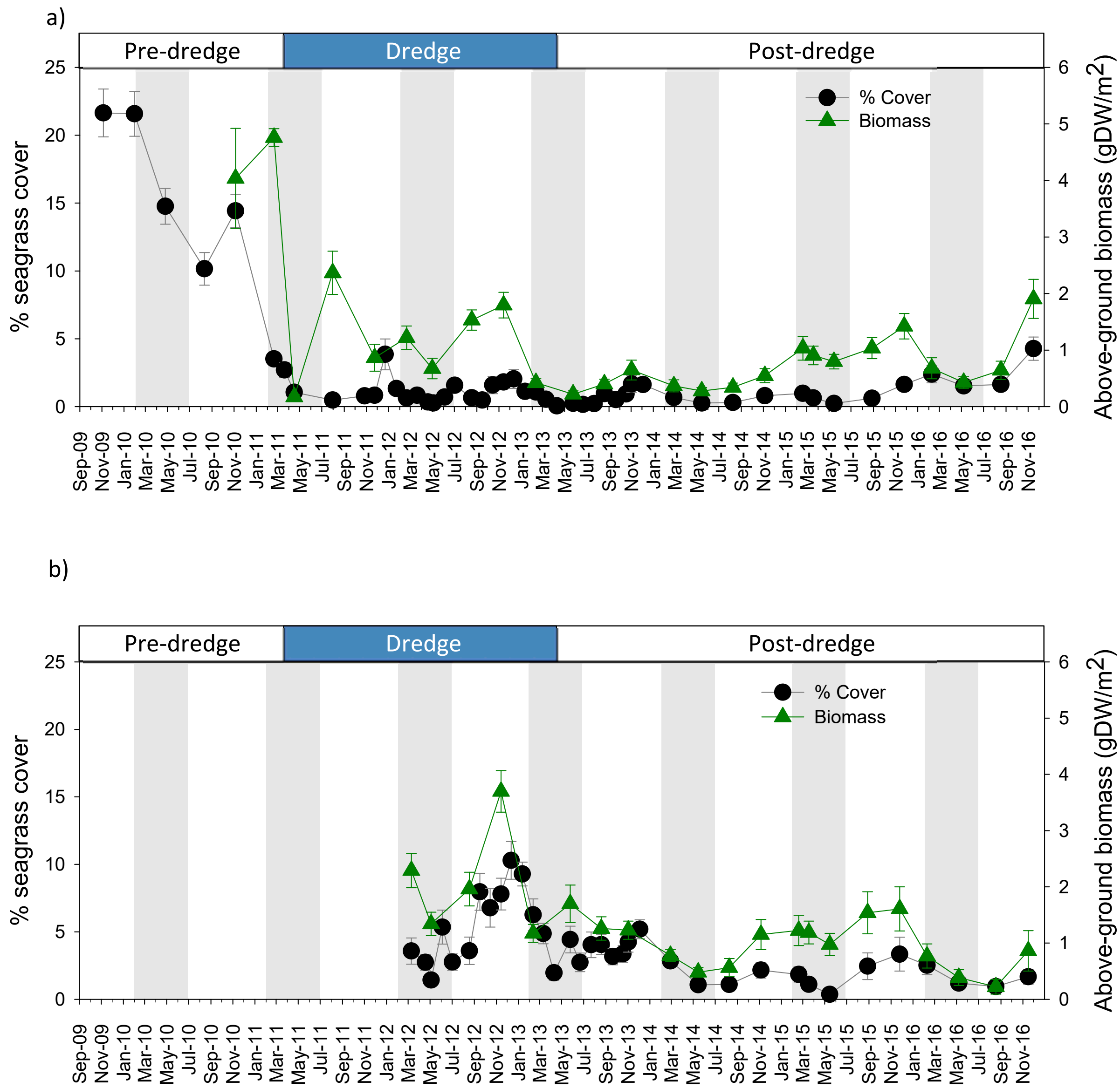


Figure 9: Seagrass abundance (mean ± SE) measured as above ground biomass (g DW m^{-2} d^{-1}) and percent cover of shoots at the a) original site and b) new site at Facing Island, November 2009 – November 2016. Shaded area represents the seagrass senescent season.

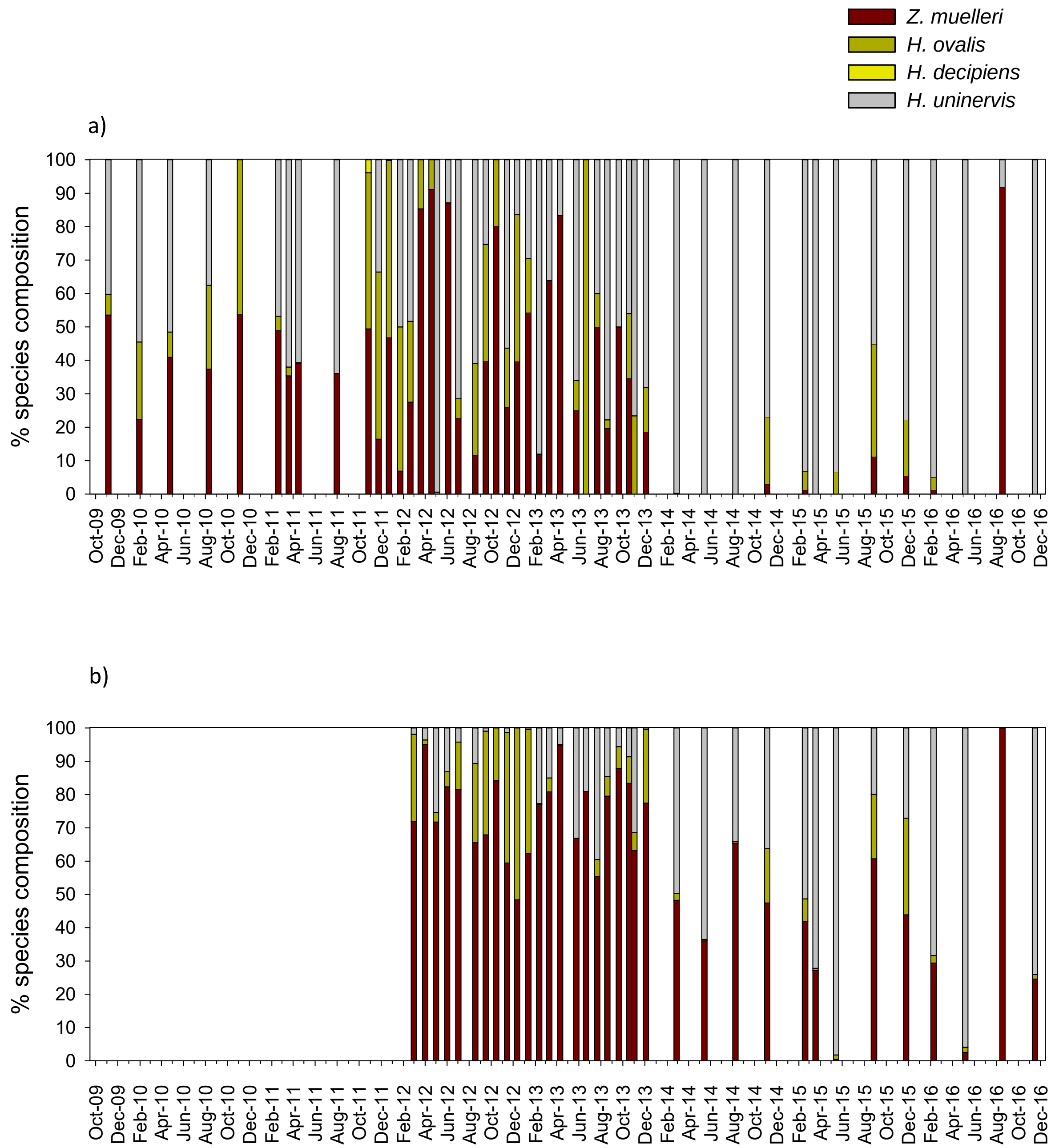


Figure 10: Seagrass species composition at the a) original and b) new permanent transect sites at Facing Island, August 2010 – November 2016.

1.3.3 Out-of-port reference site

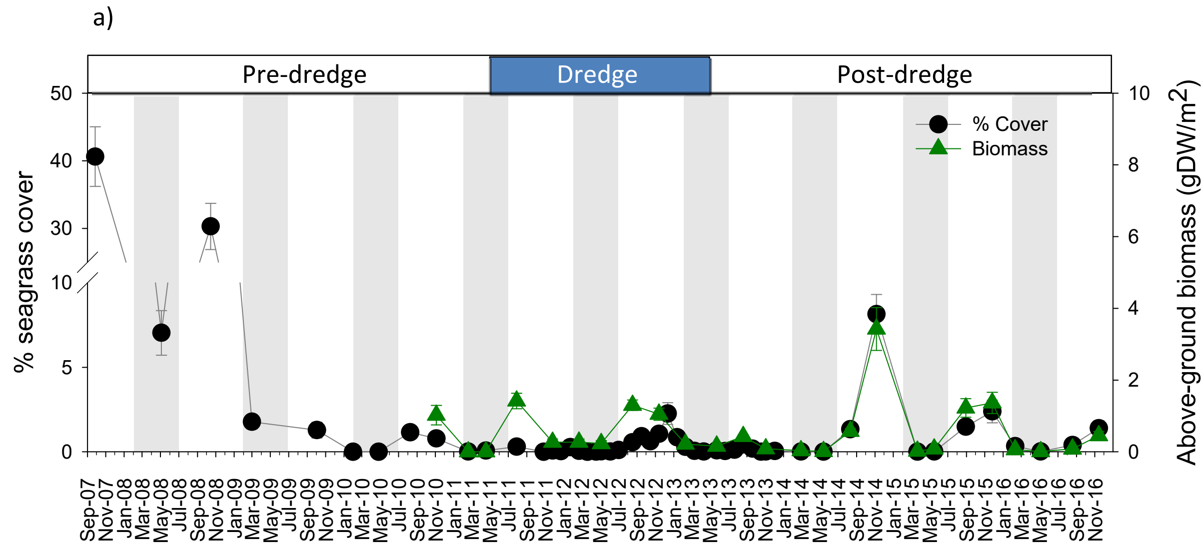
When semi-annual monitoring was established at Rodds Bay in October 2007 as part of the Reef Rescue MMP, seagrass cover was approximately 40% (Figure 11a). In the following years, a seasonal pattern became apparent with greater abundance in the late dry season and lower abundance following the wet season. In October 2009 (coinciding with the beginning of regular quarterly monitoring), seagrass failed to recover to levels seen over the previous two years following the nearby tracking of Severe Tropical Cyclone Hamish in early 2009. By January 2010, seagrass had disappeared from transects entirely. Signs of recovery were observed during the 2010 growing season with new shoots appearing at transect sites, however these patches disappeared again following the 2010/2011 extreme wet season. Patches of seagrass have come and gone seasonally since that time but seagrass cover and biomass have generally remained extremely low with the exception of a strong recovery during the 2014 growing season. In 2016, seagrass at Rodds Bay remained extremely low in cover and equally in above-ground biomass.

Z. muelleri has historically been the dominant species at Rodds Bay, especially over the growing season. However, in August and November 2015 as well as November 2016, *H. ovalis* and a smaller proportion of *H. decipiens* comprised more than 50% of seagrass cover at the site (Figure 11b). Monitoring of the entire seagrass meadow as part of the long-term annual program has shown a larger amount of seagrass growing in areas outside of the permanent transect site on many occasions, including times when seagrass was absent from the permanent transect site (Davies et al. 2016). This is due to the shape of the meadow changing over time, leading to permanent transect sites being outside the boundary of the meadow. Despite these differences between transects and overall meadow estimates, trends in seagrass decline over time are similar in the permanent transect site at Rodds Bay (Rasheed et al. 2017b).

1.4 Discussion

Monitoring of seagrass permanent transect sites over the last seven years at locations from the Narrows to Rodds Bay has shown distinct seasonal changes in the abundance and composition of Port Curtis and Rodds Bay seagrasses, and is consistent with the definitions of growing and senescent seasons in Chartrand et al. (2012).

These seasonal seagrass dynamics varied by location and among years, with changes between the growing and senescent seasons manifesting in different ways throughout Port Curtis and Rodds Bay (Rasheed et al. 2017b). At some Outer Harbour meadows, such as the persistent, high biomass meadow at Pelican Banks North, these seasonal changes are observed as increases and decreases in seagrass cover and the size of seagrass blades throughout the meadow with little change to the species composition within or among monitoring years. In contrast, many Inner Harbour locations with typically light and patchy seagrass cover, such as Fisherman’s Landing, showed seasonal changes through the appearance and disappearance of patches of seagrass. Most of these sites also underwent distinct seasonal trends in species composition, as well as changes to species composition between years following extreme weather events.



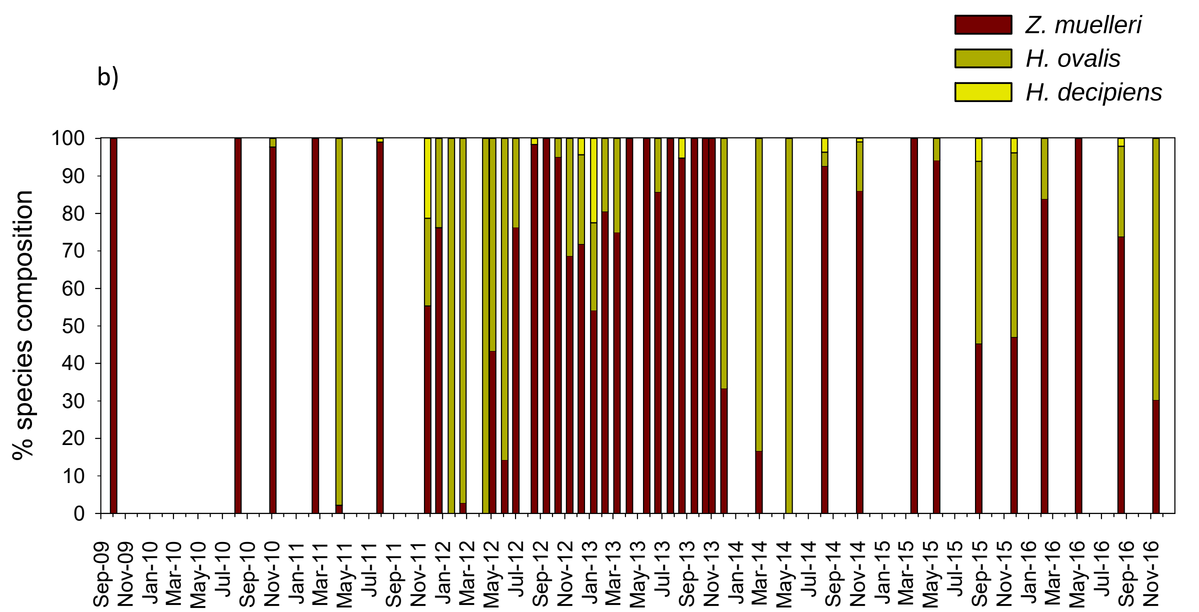


Figure 11: a) Seagrass abundance (mean ± SE) measured as above ground biomass (g DW m^{-2} d^{-1}) and percent cover of shoots and b) seagrass species composition at Rodds Bay, October 2007 – November 2016. Shaded area represents the seagrass senescent season. Data prior to October 2009 were collected for the Reef Rescue MMP (McKenzie and Unsworth 2009).

Long-term monitoring at permanent transect sites has also demonstrated significant inter-annual variability in seagrass distribution, abundance and species composition, which also varied by location. All sites (Wiggins Island, Black Swan, Facing Island, Pelican Banks North and Pelican Banks South) have had major declines in seagrass abundance since sites were established (Davies et al. 2015; Bryant et al. 2014a). While these declines are concerning, a longer time series may be required in order to place these changes in a historical context which goes beyond the 2002 baseline surveys. More notably, both Pelican Banks permanent transect sites have significantly declined since 2014, a multi-year trend which has not been seen since monitoring began in 2002. While this program will not continue monitoring at this site to determine whether recovery from the most recent declines occurs over the next few years, the annual monitoring program will continue to monitor these locations and will continue to record seagrass dynamics at these sites (Rasheed et al. 2017b).

The value in the extended time series of the annual monitoring program is that we are able to observe cycles of decline and recovery in seagrass meadows over longer (decadal) periods and place them in a historical perspective. Seagrass biomass data for the Wiggins Island meadow are also available as far back as 2002 as part of the longer term annual seagrass monitoring for the port (Chartrand et al. 2010). This relatively long time series (compared to other sites) reveals that average seagrass abundance at the meadow in 2010 (following major declines at the permanent transect site) was actually similar to levels documented in 2004 and 2005. Large scale declines may therefore form part of the natural levels of inter-annual variation at this meadow, located at the mouth of the Calliope River.

While monitoring has revealed the capacity for seagrass abundance to recover from declines, many of the permanent transect sites have revealed shifts in species composition which have yet to revert back to the assemblages first documented at the start of monitoring. Several Inner Harbour sites as well as Pelican Banks South in the Outer Harbour have shifted away from the strong dominance of *Z. muelleri* early in monitoring, to less persistent species such as *Halophila* and *Halodule* species in recent years. At many sites, declines in *Z. muelleri* were concurrent with declines in seagrass abundance following extreme weather events, and while cover and biomass have rebounded at some sites, species composition remains altered.

Data on rates and extent of recovery of seagrass meadows is rare due to a lack of long-term monitoring programs and the time taken for recovery to occur (Erftemeijer and Lewis 2006). Studies that do document recovery report periods of weeks to months for small scale disturbances (Rasheed 2004; 1999) to more than five years (Birch and Birch 1984). Recovery rates are influenced by the intensity, frequency and duration of impacts (Short and Wyllie-Echeverria 1996), as well as the ecology of individual species. Differences in morphological and physiological characteristics as well as the mechanisms for reproduction and rates of productivity will influence the ability of seagrasses to recover after an impact (Macreadie et al. 2014; Potouroglou et al. 2014; Ruesink et al. 2012; Campbell et al. 2007). The resilience of Port Curtis and Rodds Bay seagrass meadows are discussed in detail in section 3 of this report.

The benefit to longer term studies such as this monitoring program is the ability to examine inter-annual variability in seagrass. Seagrass patches come and go as we have seen at Fisherman's Landing, and also change in position and density (Hemminga and Duarte 2000), as seen with meadow movement in and out of permanent transect sites at Facing Island and Rodds Bay. It is important to use caution when basing long-term management decisions on short-term observations (Sofonia and Unsworth 2010; Erftemeijer and Lewis 2006). Intra- and inter-annual changes in seagrass distribution, abundance and species composition are associated with a range of complex interactions of natural and climate related drivers.

2 SEAGRASS TISSUE NUTRIENTS

2.1 Background and approach

Seagrass tissue nutrient ratios (calculated atomic ratios of C:N:P) offer a method for establishing environmental parameters that may be influencing seagrass meadows (Johnson et al. 2006). The magnitude of these ratios and their temporal changes allow for a broad level indication of the physical environment of seagrass meadows (McKenzie and Unsworth 2009).

Carbon to nitrogen (C:N) ratios are related to light levels (Collier et al. 2009; Grice et al. 1996; Abal et al. 1994). Experiments on seagrasses in Queensland have suggested that at an atomic C:N ratio of less than 20 indicates reduced light availability (Grice et al. 1996; Abal et al. 1994).

Nitrogen to phosphorous (N:P) ratios represent the overall nutrient availability to the plant. Tissue N:P ratios of 25-30 suggests seagrass are nutrient replete (Fourqurean and Cai 2001; Fourqurean et al. 1997; Atkinson and Smith 1983), and potentially eutrophic. N:P ratios in excess of 30 are considered indicative of phosphorous limitation, while ratios less than 25 are indicative of nitrogen limitation (McKenzie and Unsworth 2009; Johnson et al. 2006).

Carbon to phosphorous (C:P) ratios are a surrogate for nutrient status in the habitat (McKenzie et al. 2012). The median seagrass tissue ratio of C:P is approximately 500 (Atkinson and Smith 1983), therefore deviation from this value suggests some level of nutrient-enriched (C:P<500) or nutrient-limited (C:P>500) conditions. A nutrient-rich habitat is considered to have a relatively large phosphorous pool while a nutrient-poor habitat would have a small phosphorous pool (McKenzie and Unsworth 2009).

2.2 Methods

During quarterly on-ground sampling events, seagrass was harvested from three independent plots from an area adjacent to each transect monitoring site, and transported frozen to the laboratory. Leaves were then separated from below-ground material and scraped clean of epiphytes. Samples were dried at 60°C and homogenised by milling to a fine powder.

Nitrogen and phosphorus were extracted using a standardised selenium Kjeldahl digest and the concentrations determined with an automatic analyser using standard techniques at Chemcentre in Western Australia (a NATA certified laboratory). Percent carbon was determined by atomic absorption, and C:N:P elemental ratios were calculated on a mole:mole basis using atomic weights (i.e., C=12, N=14, P=31).

Seagrass was not collected at sites where there was minimal or no seagrass available, therefore at some sites and in some years there is no tissue nutrient data.

2.3 Results

2.3.1 Inner Harbour

Low seagrass biomass and access issues prevented the collection of tissue nutrient samples in the Inner Harbour from 2013. For trends prior to 2013 see McCormack et al. (2013a). (McCormack et al. 2013a).

2.3.2 Outer Harbour

In 2016, the C:N ratio at Pelican Banks North remained below 20 for all sampling events, similar to the previous three years (Figure 14a). There has been insufficient *Z. muelleri* at Pelican Banks South and the original Facing Island site over the past three years to assess tissue nutrient ratios.

Facing Island C:N ratios for *Z. muelleri* have historically remained below 20 during quarterly sampling events but peaked in November 2015 (26.32 ± 1.70) indicating high light availability (Figure 14a). C:N ratios for *H. uninervis* at both Pelican Banks South and Facing Island have remained below 20, with only one exception at the original Facing Island site in May 2012 (Figure 14b).

Nitrogen to phosphorous (N:P) ratios for *Z. muelleri* at Pelican Banks North remained within the 25-30 range in February and November 2016, indicating a nutrient replete environment (Figure 14c). At Facing Island, N:P ratios for *Z. muelleri* were generally within the 25 – 30 range as well (Figure 14c). However in November 2015, the N:P ratio was the lowest observed at this site and indicates possible nitrogen limitation (Figure 14c). N:P ratios for *H. uninervis* at Pelican Banks South and Facing Island indicate an oscillation between a nutrient replete environment and phosphorus limitation (Figure 14d). For the original Facing Island site, this phosphorus limitation typically occurs in February while the new site and Pelican Banks South don't appear to follow a seasonal trend and vary between years.

Carbon to phosphorous (C:P) ratios for *Z. muelleri* at Pelican Banks North fell below 500 during February and May 2016, indicating nutrient enrichment following seasonal rains and river flow (see Section 4.3.1), and peaked above 500 during November indicating nutrient limitation towards the end of the growing season (Figure 14e). At Facing Island, C:P ratios for *Z. muelleri* have followed a similar seasonal trend. C:P ratios for *H. uninervis* at Pelican Banks South and Facing Island have also generally suggested a nutrient rich environment during the senescent season and nutrient depletion during the growing season (Figure 14f).

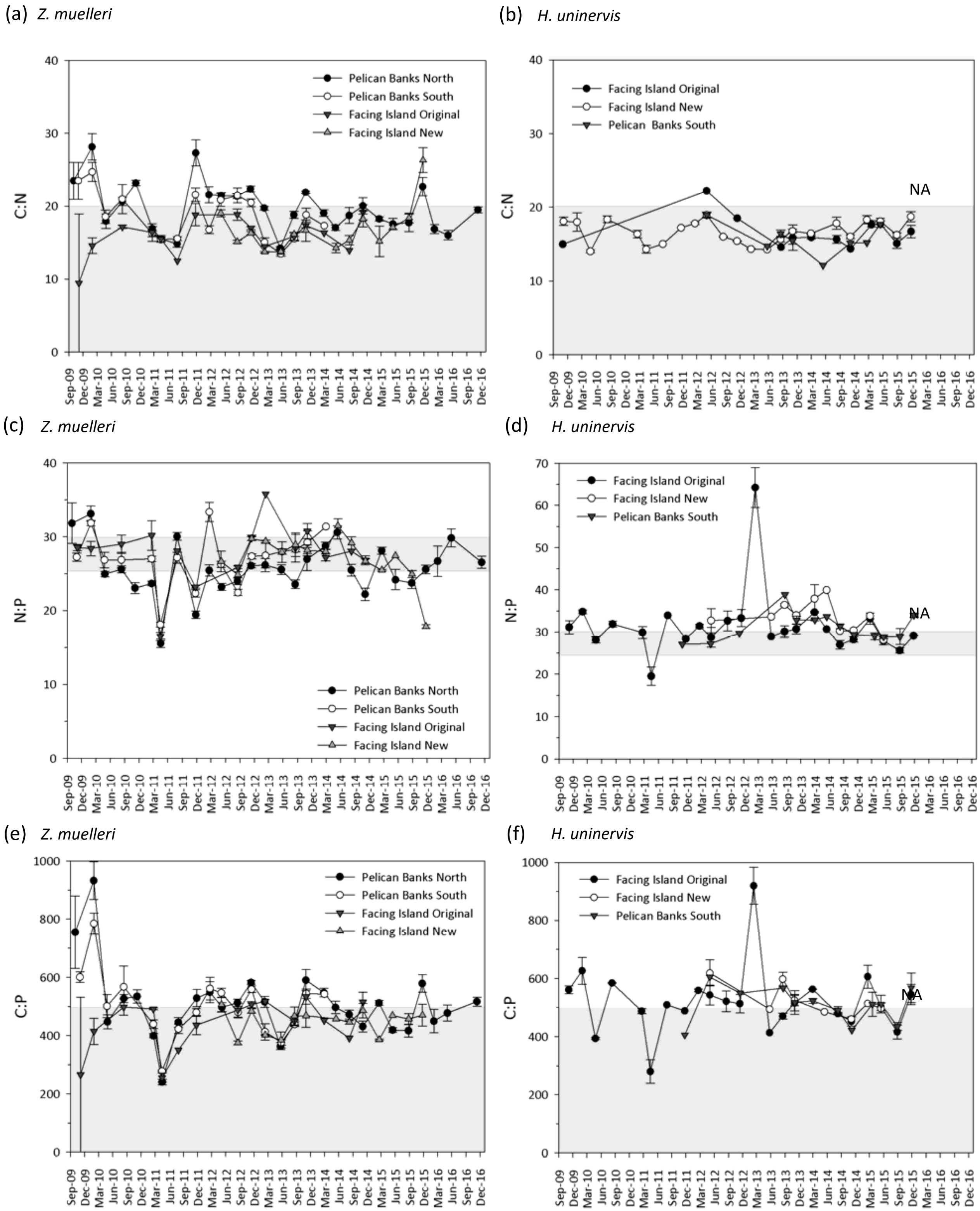


Figure 14: Elemental (atomic) ratios of seagrass leaf tissue for *Z. muelleri* and *H. uninervis* collected at permanent transect sites in the Outer Harbour (mean ± Standard Error). Shaded areas indicate low light (C:N < 20), nutrient replete (N:P 25-30) and nutrient enriched (C:P < 500) environments.

2.3.3 Out of port reference site

Samples were only collected in half of the monitoring trips in 2016, February and November, due to the loss of above-ground biomass at Rodds Bay outside transects in May and August 2016. Carbon to Nitrogen (C:N) ratios for *Z. muelleri* remained below 20 throughout year, indicating persistent lower light conditions (Figure 15a). Nitrogen to Phosphorus (N:P) ratios between 25 and 30 were indicative of nutrient replete conditions (Figure 15b) and C:P ratios below 500, indicate nutrient enrichment (Figure 15c).

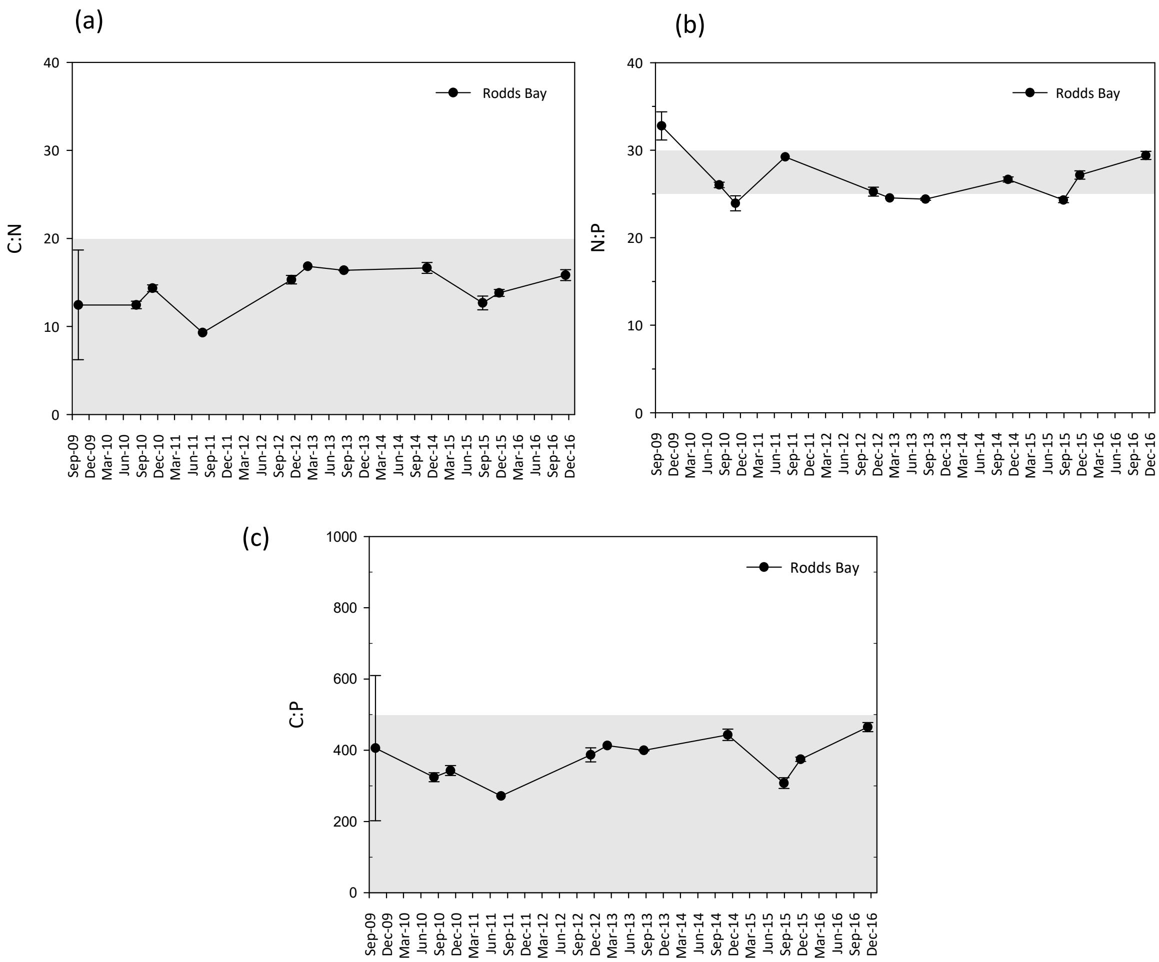


Figure 15: Elemental (atomic) ratios of seagrass leaf tissue for *Z. muelleri* collected at permanent transect sites at the out of port reference monitoring location in Rodds Bay (mean ± Standard Error). Shaded areas indicate low light (C:N < 20), nutrient replete (N:P 25-30) and nutrient enriched (C:P < 500) environments.

2.4 Discussion

The light conditions as indicated by C:N ratios at the Outer Harbour Pelican Banks North and South sites have fluctuated seasonally, with seagrasses generally existing in lower light conditions following the wet season and higher light conditions in the peak growing season. Ratios at Facing Island have indicated that these sites have existed in lower light conditions over the majority of the monitoring program. These results correspond well with seagrass abundance and somewhat well with *in situ* light data collected at these sites, which also shows a higher light environment at the northern Pelican Banks site compared with the southern site or at Facing Island (See Section 4.3). However, tissue nutrient analysis is a course approximation of the light environment, with the light data collected within the seagrass meadow providing finer-scale fluctuations in the light environment. While declines in light and seagrass percent cover coincided with lower C:N ratios following extreme flooding and river flow events in early 2011, 2013, 2014 and 2015, *in situ* data shows the recovery of the light environment while C:N ratios continued to decline. This is likely due to the in-built delay in using plant material as a proxy verses real-time data. Furthermore, C:N ratios indicating reduced light environment were detected at time points where the *in situ* data showed light remained above the locally derived threshold for seagrass growth.

Nutrient enrichment also generally occurs after each wet season with the lowest C:P ratios occurring around April and the highest around November each year. These seasonal patterns are typical of Australian tropical seagrass systems that exhibit shifts in nutrient dynamics corresponding with wet summer and dry winter seasonal shifts. Large rainfall events bring pulses of material and nutrients into the inshore environment which replenishes supplies that may be used up during the previous growing season (Carruthers et al. 2002).

In seagrass ecosystems, nutrients and light are the most common limiting factors that control seagrass abundance (Collier and Waycott 2009). Continuous monitoring of C:N:P ratios can provide a means for the early detection of changes in nutrient regimes for environmentally sensitive seagrasses, However, other seagrass metrics including tracking above-ground biomass, meadow distribution, species composition and even reproductive output trends have shown greater value in Port Curtis and in similar work on tropical seagrasses within the Great Barrier Reef region.

3 SEAGRASS RESILIENCE & RECOVERY

3.1 Background and approach

Physical impacts to seagrasses resulting in loss have occurred due to a range of environmental and anthropogenic events including storms and flooding (Preen et al. 1995; Poiner et al. 1989), grazing (Preen 1995), anchor and boat damage (La Manna et al. 2015; Kininmonth et al. 2014; Hallac et al. 2012), shipping accidents (Kenworthy et al. 1993) and dredging (Erftemeijer and Lewis 2006). The impacts on seagrasses caused by dredging include the physical removal and/or burial of vegetation and effects of increased turbidity and sedimentation (Erftemeijer and Lewis 2006). A key aspect in understanding the resilience of seagrass meadows to stressors is their capacity to recover from impact related to various stressors (Kenworthy 2000).

Seagrass recovery predominantly occurs through the dispersal of sexual propagules (seeds, fruits and flowers), and/or asexual vegetative extension of plants at the periphery and replacement of above-ground structures (Rasheed 2004; 1999; Rollón et al. 1998) or the dispersal of vegetative fragments to new environments (Thomson et al. 2015; McMahon et al. 2014). Some species are also capable of producing relatively long-lived seeds stored in the sediment. This seed bank may be potentially important for the recovery of seagrass meadows following natural or anthropogenic disturbance. Knowledge of the seed bank status will help to determine the resilience of these meadows to damage.

Assessments of the range of sexually reproductive outputs and capacities (flowering, fruiting and seed banks) were examined for key species during regular quarterly monitoring of seagrass condition. Asexual reproduction (clonal growth) was conducted from 2009 to 2014 and results are summarised in Bryant et al. (2014a). Together, this information provides a deeper understanding into the resilience and recovery capacity of Port Curtis seagrasses.

3.2 Methods

3.2.1 Reproductive effort

During quarterly sampling, 15 randomly placed sediment cores (100 mm diameter and 100 mm depth) were collected from an area adjacent to monitoring sites and transported frozen to the laboratory. Cores were thawed and sieved through a 710 µm test sieve to separate seagrass and reproductive material from the sediment. Reproductive structures (male and female flowers, fruits and seeds) were identified and counted. The monoecious *Z. muelleri* bears flowers in spathes on specialised flowering shoots. Each spathe encloses separate male and female flowers (den Hartog 1970) and spathes were counted in lieu of flowers for this species. Reproductive effort was calculated as the average number of reproductive structures (fruits, flowers and spathes) per unit area.

Due to access issues, there were no reproductive core collections at Wiggins Island from 2013. The new Wiggins Island site is only surveyed by helicopter due to the extremely soft sediments making access impossible on foot without significant impact to seagrasses from trampling.

3.2.2 Seed banks

Seagrass seed banks were originally (November 2009 to February 2011) assessed using standard Seagrass-Watch methodology. Sediment cores, measuring 50 mm in diameter and 100 mm in depth, were collected at 0 m, 10 m, 20 m, 30 m, 40 m and 50 m along transects (approximately 2.5 m to the left of the assessed quadrat) and sieved in the field using a 1 mm mesh sieve. The contents of the sieve were inspected for *H. uninervis* and *Z. muelleri* seeds.

In March 2011, following large scale declines in seagrasses in the area, GPC commissioned a more in depth investigation into the status of seed banks in the harbour. Thirty sediment cores were collected at each site at Facing Island, Pelican Banks, Wiggins Island and Fisherman's Landing and transported frozen to the laboratory. Cores were thawed and run through a series of test sieves with fresh water to separate out seagrass seeds from the sediment. For all cores, the 710 µm to 1 mm fraction of the sediment was inspected for *H. uninervis* and *Z. muelleri* seeds. For a subset of the cores, the 250 µm to 710 µm fraction was inspected using a dissecting microscope for *H. ovalis* and *H. decipiens* seeds. This method detected numerous *Z. muelleri* seeds in the 710 µm to 1 mm fraction of the sample. The latter methodology was adopted for subsequent quarterly seed bank assessments. Seed cores were analysed for all on-ground sites in both March and November in 2011 and a subset of sites (Fisherman's Landing, Wiggins Island and Pelican Banks North) were analysed in May 2012 (McCormack et al. 2013a).

Based on results of these initial assessments (McCormack et al. 2013a) we selected a subset of monitoring sites within 3 locations, Pelican Banks North, Wiggins Island and Rodds Bay, to focus further processing efforts. This is in place of the two sites within the Pelican Banks North and Rodds Bay locations surveyed and reported for other metrics. Access issues precluded seed bank sampling at the original Wiggins Island site in 2013 and sampling since 2014 has been undertaken at the new Wiggins Island site. The new Wiggins Island site is sampled by helicopter at low tide for above-ground biomass and percent cover but visited at high tide by boat to collect sediment cores to evaluate seed banks. These meadows provide a geographical spread relative to dredging associated with the WBDDP and have historically been dominated by *Z. muelleri,* the dominant species in the harbour.

3.3 Results

3.3.1 Reproductive effort

The highest density of reproductive structures (*Z. muelleri* spathes and *H. ovalis* fruits and flowers) was generally found in November each year, at the peak of the growing season (Figure 17).

In the Outer Harbour, *Z. muelleri* spathes were found regularly at Pelican Banks during August and November surveys but were noticeably absent from Pelican Banks North in November 2012 and November 2016 and from Pelican Banks South since 2013 (Figure 17a). The density of spathes has been consistently higher at Pelican Banks North than at Pelican Banks South, the maximum density reaching 1417.54 ± 254.04 spathes m^{-2} in November 2010. In November 2015, spathe density at Pelican Banks North (216.45 ± 64.07 m^{-2}) was lower than recent years but within the range detected over the course of the program prior to the absence of spathes in the whole of 2016. Spathes were found for the first time at Facing Island in November 2014 (Figure 17A; 25.46 ± 5.46 m^{-2}) but have not been detected since.

Female and male *H. ovalis* flowers were absent at Pelican Banks North in November 2016 (Figure 18a and b). Flowers were also present at Pelican Banks South with a greater proportion of male flowers than female flowers in November 2016. No fruits were found at either location 2016 (Figure 18a and b). At Facing Island, male flowers only were found in February and November 2016 as well as fruits in November 2016 (Figure 18c).

In the Inner Harbour, *Z. muelleri* spathes were found at Fisherman's Landing at the beginning of the monitoring program (November 2009) but have not been detected since (Figure 17b). This is not unexpected given the scarcity of *Z. muelleri* plants at the site in recent years. Spathes were also detected at Wiggins

Island in August 2012 before access issues restricted sampling at the site to above-ground biomass and percent cover estimates only (Figure 17b).

Low densities of *H. ovalis* fruits and flowers were found at both Wiggins Island (Figure 19a) and Fisherman's Landing (Figure 19b) earlier in the monitoring program but had not been observed at Inner Harbour sites since November 2010 (Figure 18b). In November 2015, male and female flowers were found at Fisherman's Landing and the density of male flowers was the highest detected at Inner Harbour sites over the course of monitoring (250.40 ± 99.84 m^{-2}). While seed cores were collected at Fisherman's Landing in 2016, the small and patchy shoots led to no above-ground biomass falling in cores from which to measure flowers or fruits (or *Z. muelleri* spathes).

At the out of port reference site in Rodds Bay, *Z. muelleri* spathes have not been detected during surveys since the initial sampling in October 2009 (Figure 17c). *H. ovalis* fruits and flowers (female and male) were detected in November 2015 (Figure 19c) and the density of female flowers was the highest detected at this site over the course of monitoring. No flowers or fruits were detected in November 2016.

The only reproductive structure (excluding seeds) detected for *H. uninervis* throughout the program was a male flower found at Facing Island in November 2009.

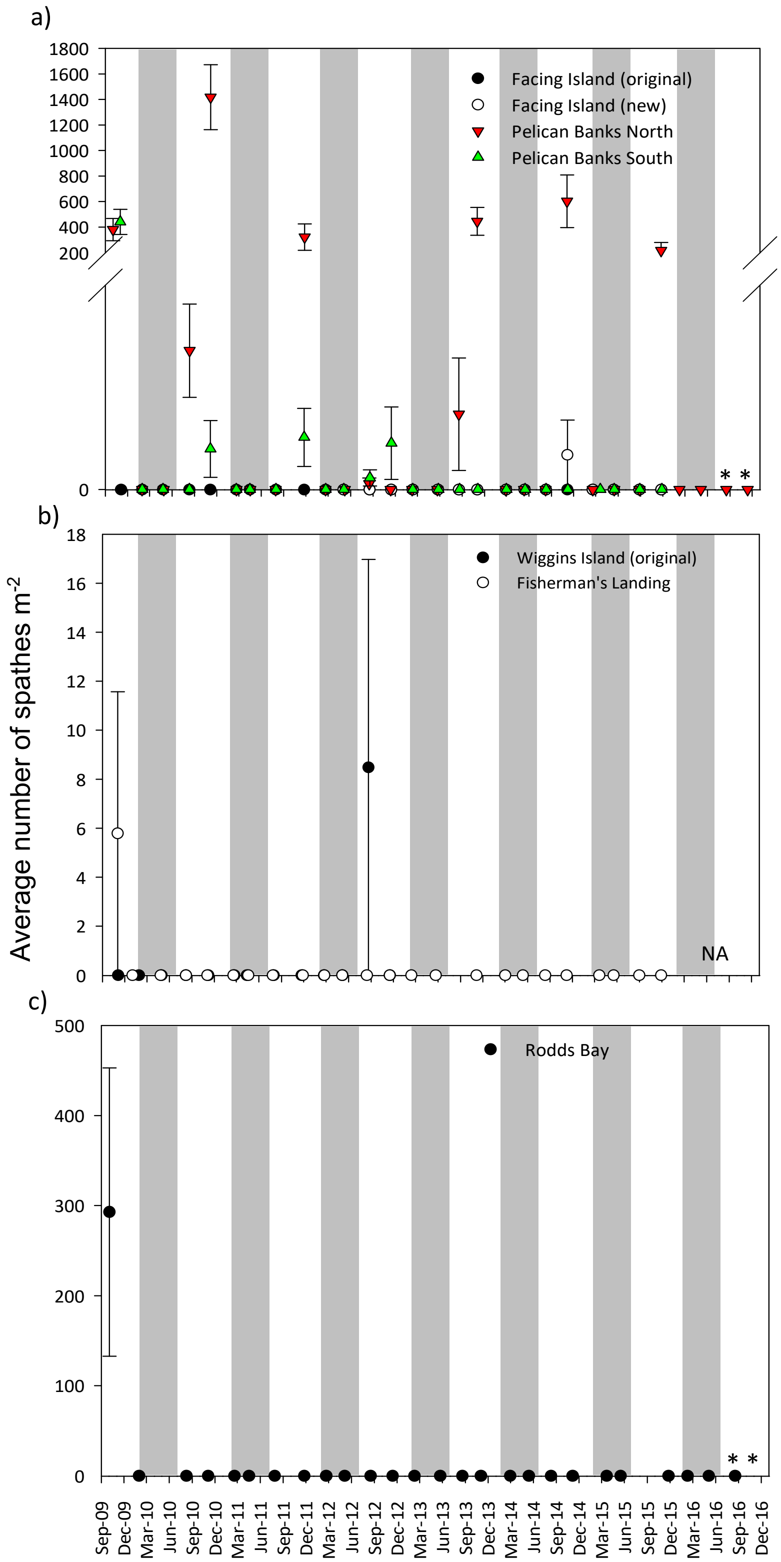


Figure 17: Density of *Z. muelleri* spathes (mean ± Standard Error) at permanent transect locations in the (a) Outer Harbour, (b) Inner Harbour November 2009 to November 2015 and (c) Rodds Bay November 2009 to November 2016. Shaded area represents the seagrass senescent season.* denotes one site at Pelican Banks North and Rodds Bay only sampled at these time points rather than both sites.

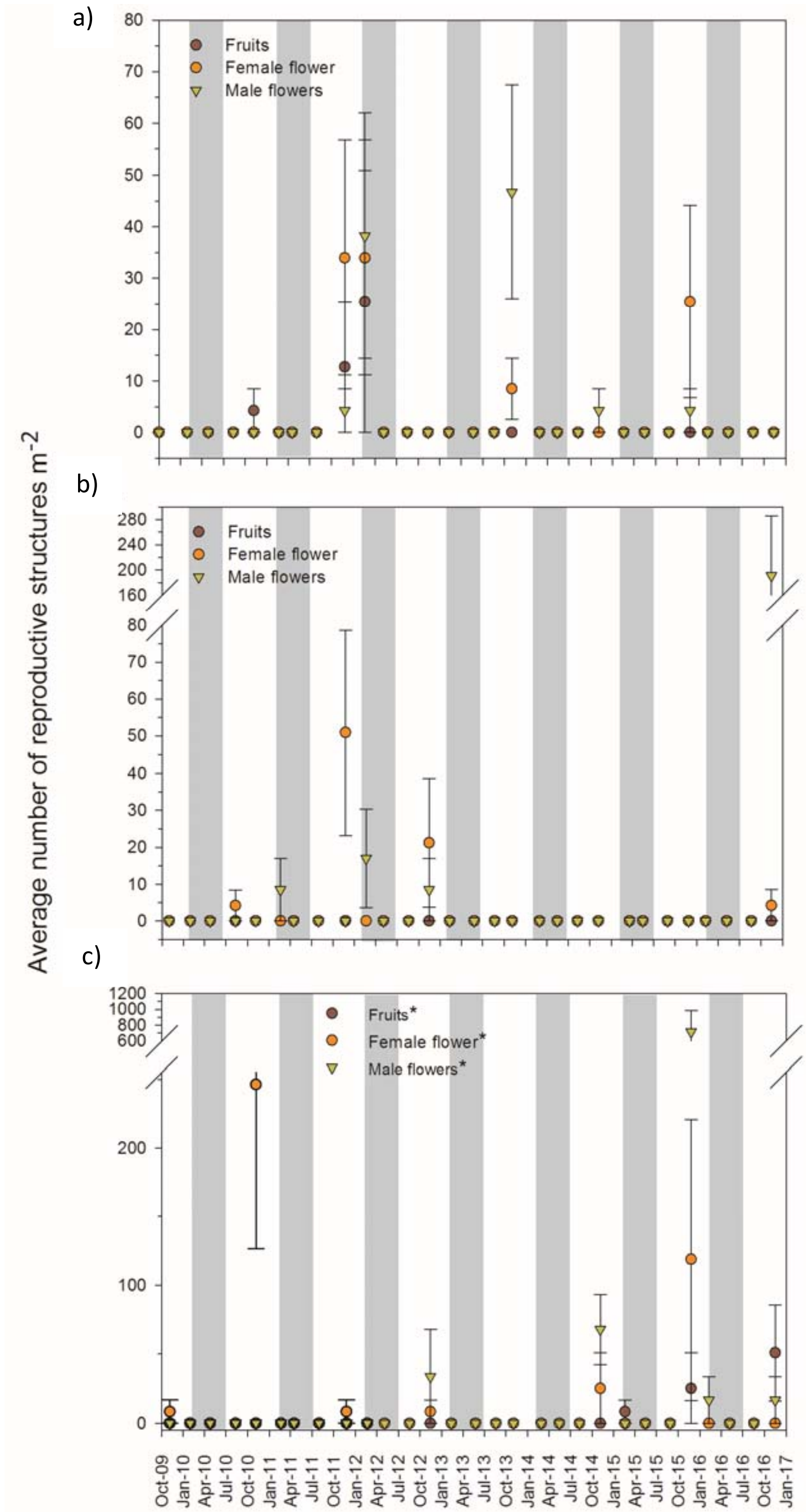


Figure 18: Density of *H. ovalis* fruits and flowers (mean ± SE) at permanent transect locations in the Outer Harbour (a – Pelican Banks North) November 2009 to November 2016, (b – Pelican Banks South) and (c – Facing Island) November 2009 to November 2016. * denotes all reproductive structures prior to January 2013 were found at the original Facing Island site whereas 2013–2016 cores only contained fruits or flowers at the new site. Shaded area represents the seagrass senescent season.

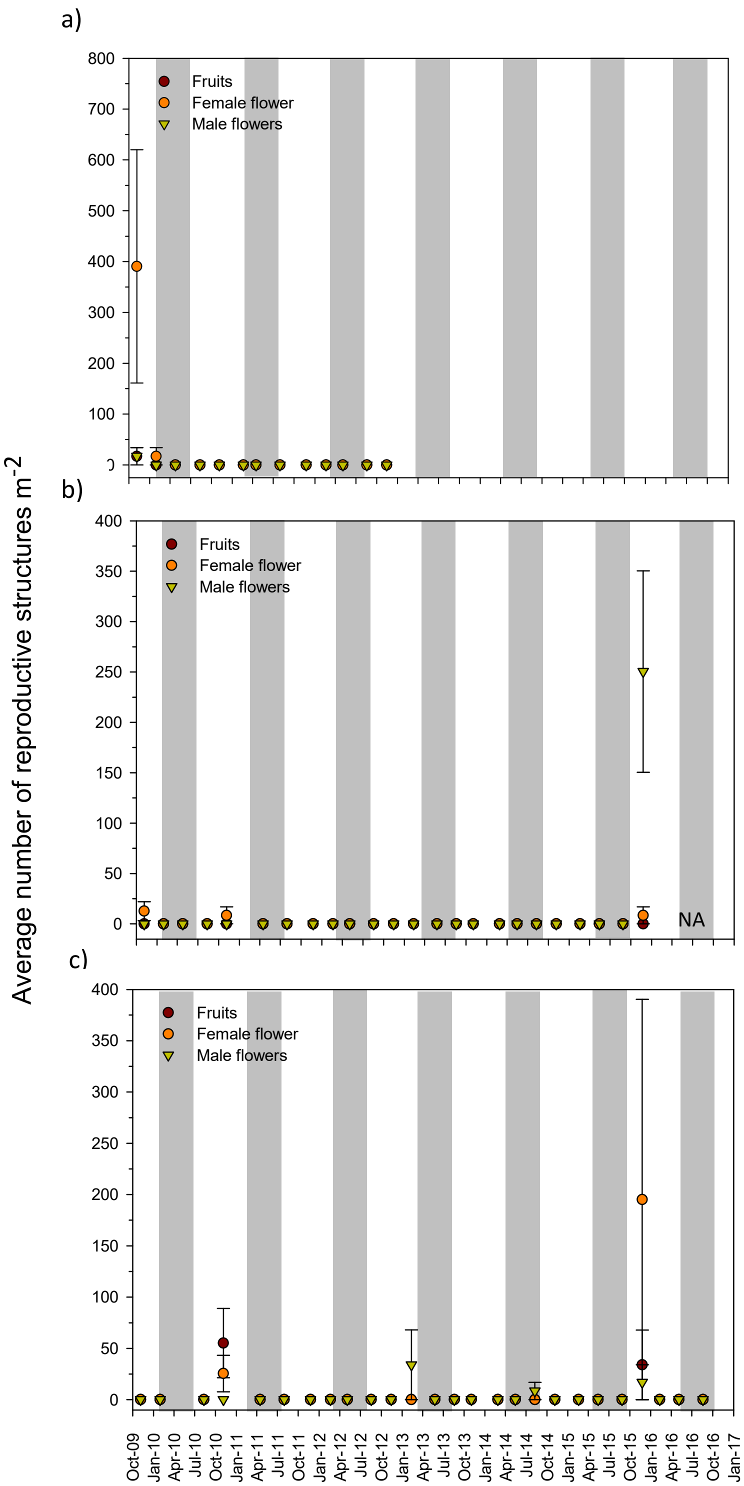


Figure 19: Density of *H. ovalis* fruits and flowers (mean ± Standard Error) at permanent transect locations in the Inner Harbour (a – Wiggins Island Original), (b – Fisherman's Landing) November 2009 to November 2015 and Rodds Bay (c) November 2009 to November 2016. Shaded area represents the seagrass senescent season. NA denotes samples not collected because no seagrass was present for reproductive structure sampling.

3.3.2 Seed banks

In 2016, *Z. muelleri* seeds were found in sediment core samples during all quarterly sampling events at all three monitoring sites (Pelican Banks North, Rodds Bay and Wiggins Island) (Figure 20). Mean seed density was generally highest during February to July surveys and lower during the growing season surveys (August and November).

At Pelican Banks North (GH1), average seed density has varied from a minimum of 56.60 ± 38.83 m^{-2} in February 2016 to a maximum of 1160.36 ± 425.63 m^{-2} in May 2014 (Figure 20). The highest seed densities have historically occurred in November or February samples, coinciding with or just after the November peaks in spathe density (Figure 18). Since 2015, however, there has been a relatively steady decline in average seed densities (Figure 20). In 2016, densities in fact were repeatedly at the lowest levels since seed density estimates were first recorded in 2011. This is in line with lower than usual above-ground biomass estimates for the Pelican Banks meadow in 2016 (Figure 7).

At Rodds Bay (RD1), average seed density has varied from a minimum of 113.23 ± 51.38 m^{-2} in August 2013 to 1613.59 ± 362.07 m^{-2} in July 2011 (Figure 20). Up until mid-2013, average seed densities at Rodds Bay were generally higher than the other sites; however since the 2013 growing season, average seed densities at the site were within the range found elsewhere (Figure 20). With the exception of 2014, there has been a steep decline in seed densities from May to August each year (Figure 20). As with Pelican Banks, average seed densities at Rodds Bay declined across the latter part of the sampling year, with the lowest densities found in November (Figure 20). The highest seed densities are typically in February and this was again true in 2016. Seed densities again fell by May 2016 when seeds were 367.92 ± 362.07 m^{-2} and held at this level through November 2016 at the final sampling event.

At Wiggins Island (WW2), as with other sites, average seed density declined over sampling years with the steepest decline occurring from May to August each year and the lowest seed density occurring in November each year (Figure 20). Seed densities in May 2016 were relatively high for the site's monitoring history, but levels in November 2016 were back down and similar to those in November 2015 (Figure 20).

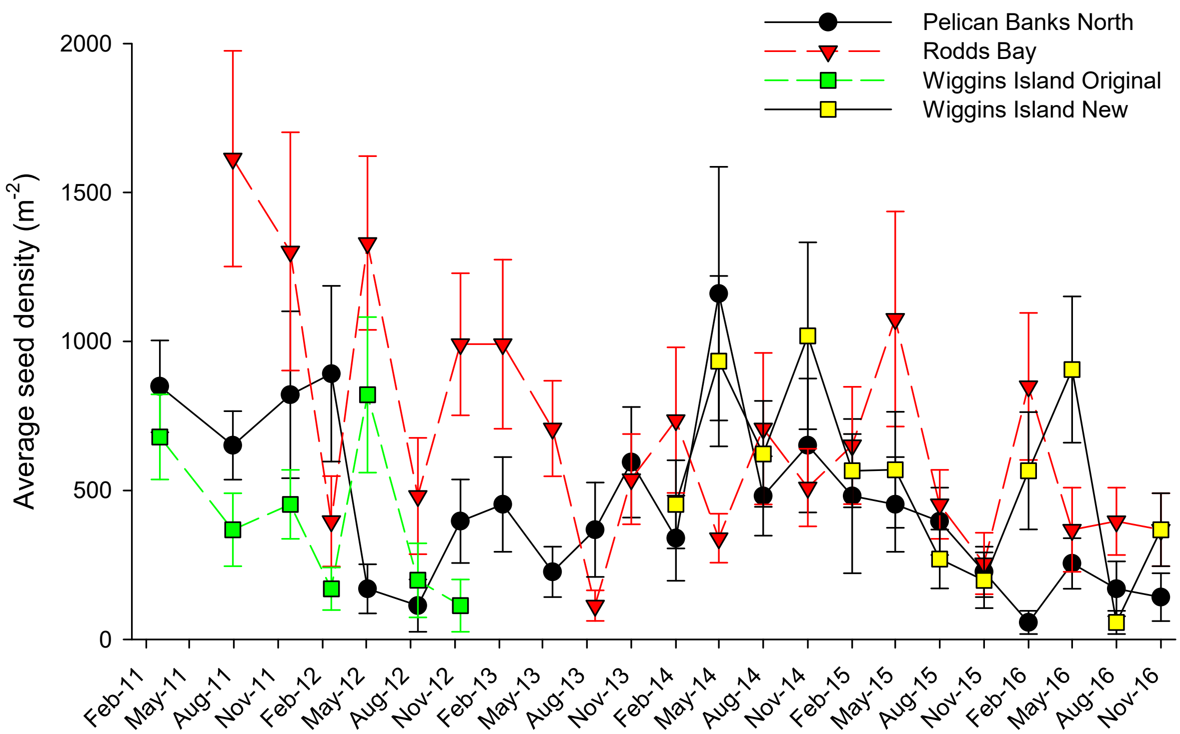


Figure 20: Mean number (± SE) of *Z. muelleri* seeds per square metre at Pelican Banks North, Rodds Bay and Wiggins Island.

3.4 Discussion

Port Curtis seagrasses demonstrated the capacity for sexual reproduction through the production of flowers fruits and seeds, however this varied substantially between locations and time of year.

In the Outer Harbour at Pelican Banks North, we found *Z. muelleri* flowered for at least four months over the growing season (from August to November) but it is possible that flowering extends beyond this period. These findings are consistent with other studies of the species in Queensland (Rasheed 1999; Conacher et al. 1994b). Our findings confirm that *Z. muelleri* in Port Curtis does not appear to flower throughout the year like some *Z. muelleri* meadows in temperate waters (Harris et al. 1980). The density of spathes peaked in November 2010 at substantially higher levels than found in other Queensland studies (Rasheed 1999; Conacher et al. 1994b); however peak spathe densities in other years are comparable to these studies. The cause for atypically low numbers of spathes found in samples in November 2012 is not known. A rise in water temperature has been identified as a cue for flowering in other *Zostera* species such as *Zostera marina* (Jacobs and Pierson 1981). Maximum water temperatures at Pelican Banks North are < 25°C from May to July and > 25°C for the rest of the year (see section 4 below). Spathe production appears to coincide with increases in water temperature in August each year; however reproduction is probably controlled by several environmental factors rather than a single variable (Conacher et al. 1994a).

During the monitoring program the *Z. muelleri* seed bank at Pelican Banks North was generally denser than reported for the same species in a three-year study in Moreton Bay (Conacher et al. 1994b). However in the final sampling period in February 2016 seed bank density had declined to be well below the Moreton Bay study. We found seeds in all quarterly samples, indicating that seeds remain present in the sediment at least five months after production (assuming flowering lasts until December) and possibly longer. This is consistent with Conacher et al. (1994b) who reported seeds in the sediments up to four months after production. Seed densities in 2016 at Pelican Banks North were overall the lowest on record since 2011 when monitoring seed banks began. A continued decline in above-ground biomass and related reproductive structures (i.e. spathes) would likely result in an ongoing lower than average seed bank from which the meadow can rely on for annual growth and recovery.

The lack of spathes detected at Pelican Banks South since 2013 coincides with declines in seagrass percent cover (from February 2013) and a shift in species composition from *Z. muelleri* to *H. uninervis* (from 2014) (see section 1). While the lack of sexual reproduction is unlikely to have been the only driver of declines at the site, propagule limitation is possibly one of the factors inhibiting recovery. At nearby Facing Island, the low density and frequency of *Z. muelleri* or *H. uninervis* flowering events suggests that the Facing Island meadow may rely largely on asexual reproduction and the dispersal of propagules from neighbouring seagrass areas such as Pelican Banks as a mechanism for recovery for those species. The Pelican Banks meadow may act as a donor meadow for seagrass propagules more widely throughout the harbour through dispersal of vegetative fragments, spathes and seeds.

At the out of port reference site in Rodds Bay, there have been no *Z. muelleri* flowering events detected at the site since November 2009 and seagrass abundance has remained at relatively low levels since steep declines the same year. Despite the lack of flowering events in recent years, this site historically contains the densest *Z. muelleri* seed bank of any site tested in the region. Unlike Pelican Banks, seasonal patterns in seed bank density in Rodds Bay do not follow periods of maximum spathe production. Port Curtis *Z. muelleri* populations flower between August and November (Bryant et al. 2014a), with maximum seed release likely to occur soon after. Therefore increases in seed bank density at the Rodds Bay site after meadow senescence (around May 2015) were unexpected. Possible explanations for this discrepancy may be the dispersal of seeds from other areas of the meadow or other meadows in the region, or (secondary dispersal) the movement of seeds within the seed bank after the initial period of incorporation (Kendrick et al. 2012).

At Inner Harbour sites, sampling for reproductive structures has been restricted to Fisherman's Landing because of access issues at the Wiggins Island site associated with port developments. No *Z. muelleri* spathes have been detected in samples at this site since the beginning of the program. This is not surprising given the scarcity of *Z. muelleri* plants at Fisherman's Landing in recent years. Seed bank investigations at the site in 2011 and 2012 (McCormack et al. 2013a) detected extremely low numbers of *Z. muelleri* seeds and there is unlikely to be a viable seed bank in the sediment to assist recovery. Recovery of *Z. muelleri* at the site will be largely dependent on the recruitment of propagules from nearby meadows. On the contrary, the same investigations found relatively high densities of *H. ovalis* seeds at Inner Harbour sites (McCormack et al. 2013b). While the viability of these seeds has not been tested, increases in the abundance of *H. ovalis* at Inner Harbour sites and the presence of fruits and flowers during recent surveys suggests the presence of a functioning seed bank.

Assessments of Port Curtis and Rodds Bay seed banks have been valuable in determining the presence or absence of seeds at most monitoring sites; and for a subset of these sites there is also good information on changes in density and more recently viability over time. Given the importance of seeds for the initial recovery of seagrass beds after large scale declines (Jarvis and Moore 2010; Lee et al. 2007a), quantifying both the density and viability of the sediment seed bank is necessary to determine the potential of seagrass meadows for recovery and their resilience to future impacts. Comprehensive results of the seed density and viability studies undertaken by JCU TropWATER in 2015 and 2016 are available in a separate report (Reason et al. 2017).

4 ENVIRONMENTAL FACTORS & RELATIONSHIP TO SEAGRASS CHANGE

4.1 Background and approach

It is widely recognised that seagrass abundance and distribution is influenced by a range of environmental variables such as river flow, temperature (Rasheed and Unsworth 2011), tidal exposure, solar radiation (Unsworth et al. 2012) and light availability (Chartrand et al. 2016; McCormack et al. 2013b; Ralph et al. 2007). To effectively manage and mitigate the environmental risk from dredging activities, it is critical to understand the background relationships that exist between seagrass and environmental variables on a local scale. Understanding the mechanistic response of seagrass to potential climatic drivers will allow better distinctions between natural climate-induced declines and dredge-related impacts (Chartrand et al. 2012).

The availability and quality of light is one of the primary environmental drivers of seagrass distribution, abundance and productivity (Duarte et al. 1997; Vermaat et al. 1997). Though some seagrasses grow in turbid water and are subject to naturally variable light conditions, anthropogenic activities such as dredging and disposal events and the associated turbidity plumes can have direct (e.g. burial) and indirect effects, such as elevating suspended particles in the water, reducing the quality and availability of light and increasing stress to the plants (Grech et al. 2013). When the light environment begins to deteriorate, an imbalance in the plant's carbon budget is created where a greater amount of carbon is used for respiration than is being fixed through photosynthesis. If sustained this can eventually lead to the loss of seagrass biomass and cover (York and Smith 2013; Ralph et al. 2007; Erftemeijer and Lewis 2006; Fourqurean et al. 2003).

Seagrasses have a range of strategies for responding to reductions in light. These responses may be physiological, such as changes in productivity and turnover (Collier et al. 2012b; Collier et al. 2009; Williams and Dennison 1990), adjustments of light harvesting capacity (Abal et al. 1994) or morphological such as changes to biomass, numbers of shoots or canopy height (Collier et al. 2012b; Ralph et al. 2007; Longstaff et al. 1999). Physiological responses typically precede morphological changes, and provide temporary relief from low light stress (Longstaff and Dennison 1999). However, if low light conditions persist or the quality of light remains deteriorated, seagrasses are unlikely able to cope for sustained periods of time and meadow-wide loss may occur (Kirkman 1978). The type of response and magnitude of impact to the seagrass meadow will depend on a number of factors, including site-specific environmental conditions and species composition (Ralph et al. 2007) as well as the intensity and duration of the reduced light conditions (Collier et al. 2012a).

Chartrand et al. (2016) determined that *Z. muelleri* in Port Curtis requires at least 5 mol photons m^{-2} d^{-1} during the growing season over a minimum period of two weeks to survive. Trends in irradiance and seagrass condition at permanent transect sites showed that, in general *Z. muelleri* consistently received greater than 6 mol photons m^{-2} d^{-1} over a two week rolling average at most monitoring locations during the growing season (when seagrass remained stable or increased in abundance). This value has been successfully used as a trigger for management actions as part of a light based approach to managing possible impacts of dredging to seagrasses during the WBDDP (Chartrand et al. 2016).

While light is generally accepted as the primary environmental factor limiting the growth of tropical seagrasses, several studies on *Zostera* species have indicated that that the interaction between temperature and irradiance may play a key role in the seasonality and survival of seagrasses in inshore turbid environments (Chartrand et al. 2016; York et al. 2013; Collier et al. 2011; Olesen and Sand-Jensen 1994).

In this section we examine the critical environmental variables measured at the permanent transect sites (benthic light and temperature) as well as major climate related drivers operating across the area (rainfall and river flow) and their association with seagrass change. The analysis includes an examination of patterns of change and these potential drivers through the pre-, during, and post dredging phases of monitoring with the now complete dataset for this final report.

4.2 Methods

4.2.1 Environmental parameter monitoring

Local climate/weather conditions

Environmental data on rainfall (mm) and river discharge (Mega Litres) are publicly available from the Australian Bureau of Meteorology website (http://www.bom.gov.au/climate/data/). Data for the nearest weather station at Gladstone Airport (station # 039123) were used. Tidal data was provided by Maritime Safety Queensland (© The State of Queensland (Department of Transport and Main Roads) 2015, Tidal Data) for Gladstone at Auckland Point (MSQ station # 052027A; www.msq.qld.gov.au).

Light and Temperature at seagrass meadows

At the beginning of the monitoring program, benthic irradiance and temperature loggers, deployed and maintained by Vision Environment, were positioned adjacent to each permanent transect location to compare light with known seagrass condition (McCormack et al. 2013a; Vision Environment QLD 2012).

In December 2013, benthic irradiance and temperature loggers were deployed by JCU TropWATER at the seven locations identified for Western Basin post dredge monitoring. Equipment was deployed at the existing benthic light compliance sites where possible to maintain consistency with the historical dataset (Figure 1). At Fisherman's Landing, loggers were deployed adjacent to the permanent seagrass monitoring site to improve our ability to interpret changes in seagrass in response to changes in light. Reference loggers were also deployed outdoors at the Gladstone Marina to assess trends in surface irradiance and assist QAQC procedures.

See (Bryant et al. 2016b) for a full description of the methods, data processing and QAQC procedures.

4.2.2 Statistical Analysis

Generalised additive mixed models (GAMMs) were used to examine the effects of environmental variables on seagrass percent cover using the "mgcv" package for R (Wood 2014). GAMMs fit a non-parametric model to the data where the functional form is not specified *a priori*, but instead additive non-parametric functions are estimated using smoothing splines (*s*) to model covariates (Zuur et al. 2014). GAMMs were used because data exploration indicated a non-linear response of percent cover to light (i.e. photosynthetic active radiation; PAR), sampling date and rainfall. Figures 30-31 were created using the rgl package (Adler and Murdoch 2017) in R (R Core Team 2016). Data were only fully modelled for Pelican Banks North and South due to gaps in datasets for PAR and high proportion of zeroes from other locations making statistical modelling output limited for other locations. The period modelled includes all time points from the program unless fields were missing (i.e. logger failure) for a specific sampling date at these two locations.

Prior to fitting models, environmental covariates were tested for collinearity using variance inflation factors (VIFs). VIFs were calculated for environmental data collected one and three months prior to measuring seagrass percent cover as well as for sampling date versus dredge period. Collinearity was high between environmental variables for both data sets. Air temperature and water temperature were excluded from the analysis (negative relationship with exposure), as was rainfall (positive relationship with river flow, see Appendix I). Dredge period was also excluded from analysis due to the high collinearity with sampling date (i.e. both variables assessed change over time, see Appendix I). The VIFs of PAR, river flow, and tidal exposure were <1.1 for the one and three month data sets, indicating that collinearity was within reasonable limits and would not substantially inflate the standard errors of the model's parameter estimates (Zuur et al. 2009). The response variable percent cover was averaged across 11 quadrats per transect before analysis. Transect means were analysed to reduce zero-inflation inherent in the quadrat data set which led to unstable models, with zero counts reduced from 55% of quadrats in the total data set to 8% of transects.

Global models for were run to determine optimal models for final analysis with either one or three month data sets. The global models were:

$$C = s(P) + s(R) + s(D) + E + \beta transect + \varepsilon$$

Where *C* is percent cover, *P* is PAR, *R* is river flow, *E* is tidal exposure, *D* is sampling date, $\beta_{transect}$ is the random effect of transect, and *ε* is the random error term. Sub-model sets of the global model were generated using the dredge function in the MuMIn package (Bartoń 2013). The best-fit model was considered to be the simplest model with the lowest Akaike's Information Criterion corrected for small sample sizes (AICc) that fell within two of the lowest AICc (Burnham and Anderson 2002). Normalised residuals were inspected for the best-fit final model using residual plots and qq-plots for violations of the assumptions of homogeneity of variance and normality (Zuur et al. 2014). Graphical residual analysis of the response variable (*C*) and explanatory variables from the best-fit model also provided model validation in choosing a model that created no residual patterns.

4.3 Results

4.3.1 Local climate conditions

The local climate in the Port Curtis region over the course of monitoring has been characterised by above average rainfall, punctuated by severe flood events. The period from 2008 to 2016 was considerably wetter than the period from 2005 to 2007, with total monthly rainfall during the summer wet season frequently exceeding the long-term average during this time (Figure 24). The peak in total rainfall and river flow during January 2013 was the highest on record with discharges of over 600 000 Mega litres of water causing severe flooding in the Port Curtis/Gladstone region (Figures 24 and 25). Rainfall during the 2016 senescent season was higher than the comparable time period the two previous years.

In February 2015, TC Marcia crossed the coast just north of Port Curtis following above average rainfall and river flow in January (Figures 24 and 25). Flow from the Calliope River peaked well above the long-term average (since 1970) but below the peaks recorded in previous years (Figure 25).

4.3.2 Light and temperature at permanent transect locations

Inner Harbour

The rolling average total daily light at Fisherman's Landing and Wiggins Island in the Inner Harbour frequently fell below the minimum light requirement for *Z. muelleri* (6 mol m^{-2} day^{-2}) over both growing and senescent seasons earlier in the program (late 2009 to early 2011) (Figure 26a). Since mid-2011 however, the light environment has improved; at Fisherman's Landing, dips below 6 mol m^{-2} day^{-2} primarily occurred during the senescent season (Figure 26a) and light at Wiggins Island has remained above this threshold since early 2012 (Figure 26b). Light levels at both Inner Harbour sites have fluctuated between similar ranges from 2014 to 2016 and are generally lower than the period from 2012 to 2013 (Figure 26a and b).

At Black Swan, the rolling average total daily light has remained above 6 mol m^{-2} day^{-2} during the growing season over the duration of monitoring with the exception of a brief period in August 2014, January 2015 and July 2016 (Figure 27). Light levels shifted to a lower level from 2014-2016 compared with 2012-2013; however this may reflect the change of equipment in November 2014. Whilst every attempt was made to ensure that loggers were placed in the same location, it is possible that the new equipment was located at a slightly deeper section of the meadow.

Temperature at the seagrass canopy followed a similar trend over the course of the program, increasing over the growing season and decreasing over the senescent season (Figure 26a and b). Maximum daily temperatures at Fisherman's Landing and Wiggins Island reached higher levels over longer periods of time from 2012 to 2016 compared with earlier in the program. Temperatures frequently exceeded 35°C during this period, which coincides with lower levels of seagrass percent cover at Inner Harbour sites (Figure 25a and b). At Black Swan, maximum daily temperature also exceeded 35°C during the late growing season and early senescent season in 2013/14, 2014/15, and 2015/16 (Figure 27).

Outer Harbour

At Pelican Banks North, the rolling average total daily light remained above 6 mol photons m^{-2} d^{-1} during the growing season since 2011 (Figure 28a). Deviations below or close to this threshold generally correspond with peak river flow events and were closely followed by declines in seagrass percent cover (Figure 28a). From 2014 to 2016, the rolling average total daily light increased across the growing season but remained lower than levels detected over the growing season in 2012 and 2013 (Figure 28a). Temperature followed established seasonal trends with peaks during the summer growing season. It is not uncommon for temperatures to reach extreme temperatures towards the end of the growing season; however these events rarely last for sustained periods of time. The exception was from November 2014 to March 2015, when temperature repeatedly exceeded 35°C for sustained periods of time (Figure 28a).

At Pelican Banks South, the rolling average total daily light remained above 6 mol photons m^{-2} d^{-1} during the growing season, with the exception of January 2014 (Figure 28b). As with Pelican Banks North, deviations below this threshold correspond with peak river flow events in early 2013, 2014 and 2015 (Figure 28b). The exception was in June 2015, when the rolling average total daily light fell briefly below 6 mol photons m^{-2} d^{-1}. Light levels at the Pelican Banks South follow a similar trend but are generally lower than Pelican Banks North (Figure 28a and b). As with Pelican Banks North, from 2014 to 2016, the rolling average total daily light increased across the growing season but remained lower than levels detected over the growing seasons in 2012 and 2013 (Figure 28b). Changes in light levels detected since 2014 coincide with declines in seagrass percent cover at the site (Figure 28b). Temperature during this time appears to follow seasonal trends observed elsewhere. While maximum daily temperature did not commonly reach the sustained extreme high temperatures observed at Pelican Banks North, temperatures repeatedly exceeded 33°C over the same period (Figure 28b).

At nearby Facing Island, the light environment has been more variable than other Outer Harbour sites (Figure 29a). Over the first few years of the study, the rolling average light fell below 6 mol photons m^{-2} d^{-1} almost as much as it remained above (Figure 29a). Since 2013 however, light levels have remained above minimum light requirements for *Z. muelleri* during the growing season with the exception of January 2014 following an extreme rainfall and river flow event (Figure 29a). In 2015, the rolling average light briefly fell below 6 mol photons m^{-2} d^{-1} in June (Figure 29a) but remained above this threshold over the growing season when seagrasses at both the original and new sites increased in abundance. Light levels over the 2015 growing season failed to reach levels detected in previous years (Figure 29a). Maximum daily temperatures at Facing Island reached higher levels over longer periods of time from 2012 to 2016 compared with earlier in the program. Temperatures frequently exceeded 33°C during this period, with maximum daily temperatures in excess of 35°C at the peak of each growing season since 2014 (Figure 29a).

Out of port reference site

At the out of port reference site at Rodds Bay, the rolling average total daily light has remained above 6 mol photons m^{-2} d^{-1} during the growing season over the majority of the study, with the exception of a short period in early 2012 when the logger was temporarily shifted to a deeper location (Figure 29b) and for a brief period in August 2014 (Figure 29b). Light also fell below this threshold at the beginning of the senescent season in 2013 following an extreme rainfall and river flow event (Figure 29b). In 2016, the rolling average total daily light remained well above 6 mol m^{-2} day^{-1} in both the senescent and growing seasons. Peaks in light over the growing season reached a similar level as the previous year and coincided with increases in seagrass percent cover at the sites (Figure 29b).

Maximum daily temperature at the site followed seasonal trends with increases in over the growing season and decreases in the senescent season (Figure 29b). Extreme temperatures have been more common at this site than at Western Basin transects. Maximum daily temperatures exceeded 35°C for prolonged periods in the spring/summer of 2009/10, 2010/11, 2014/15 and again in 2015/16 where 40°C was reached in late October (Figure 29b).

4.3.3 Modelling potential drivers of seagrass change

Results from generalised additive mixed modelling using the range of available explanatory variables, showed a significant temporal effect on Pelican Banks North and Pelican Banks South seagrass percent cover (Figures 30-31; Table 1). This was to be expected based on plotted seagrass abundance at both locations showing a downward trend from the beginning of the permanent transect site program until late 2016 when seagrass cover had declined to record lows since monitoring in Port Curtis and Rodds Bay began in 2002 (Figure 7; Rasheed et al. 2017b). While left out of the final models, cover varied by dredge period (pre-, during, and post-) with a similar downward trend as seen with sampling date (i.e. lowest cover measured in the post-dredge monitoring period overall). In addition to modelled declines over time, the best models for each location included one month mean total daily PAR (mol photons m^{-2} d^{-1}) and three-month cumulative river flow prior to sampling (Figures 30-31; Table 1). The full models did a relatively good job at explaining the proportion of variance measured at both locations with 62% and 63% at Pelican Banks North and Pelican Banks South respectively (based on the adjusted R squared statistic; Table 1). Percent cover increased with higher light at both locations; however there is a greater positive slope for Pelican Banks North than with Pelican Banks South due to the higher starting cover at the former site early in monitoring (Figures 30-31).

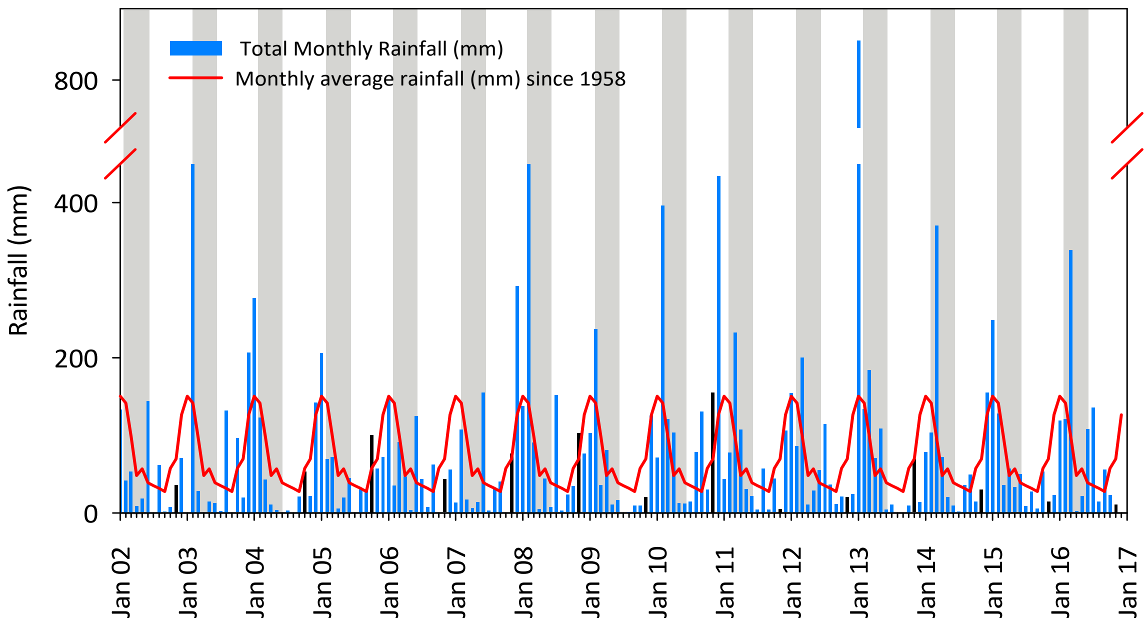


Figure 24: Total monthly rainfall (mm) for Port Curtis and Rodds Bay, from January 2002 to December 2016. Data taken from station number 039123 (Gladstone Airport); from Bureau of Meteorology http://www.bom.gov.au/climate/data/). Black bars represent survey months.

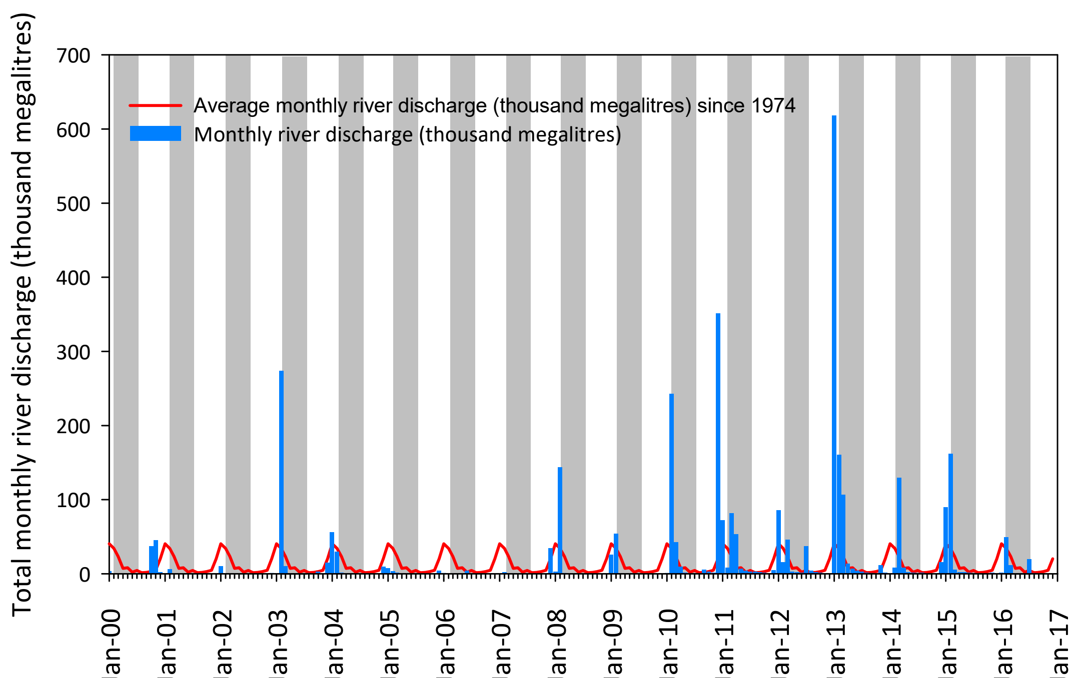


Figure 25: Mean monthly river discharge (volume Mega Litres) for Calliope River at Castlehope, Gladstone, from January 2000 to December 2016. Data taken from Calliope Basin, site 132001A; from DERM Water Monitoring (http://watermonitoring.derm.qld.gov.au/host.htm).

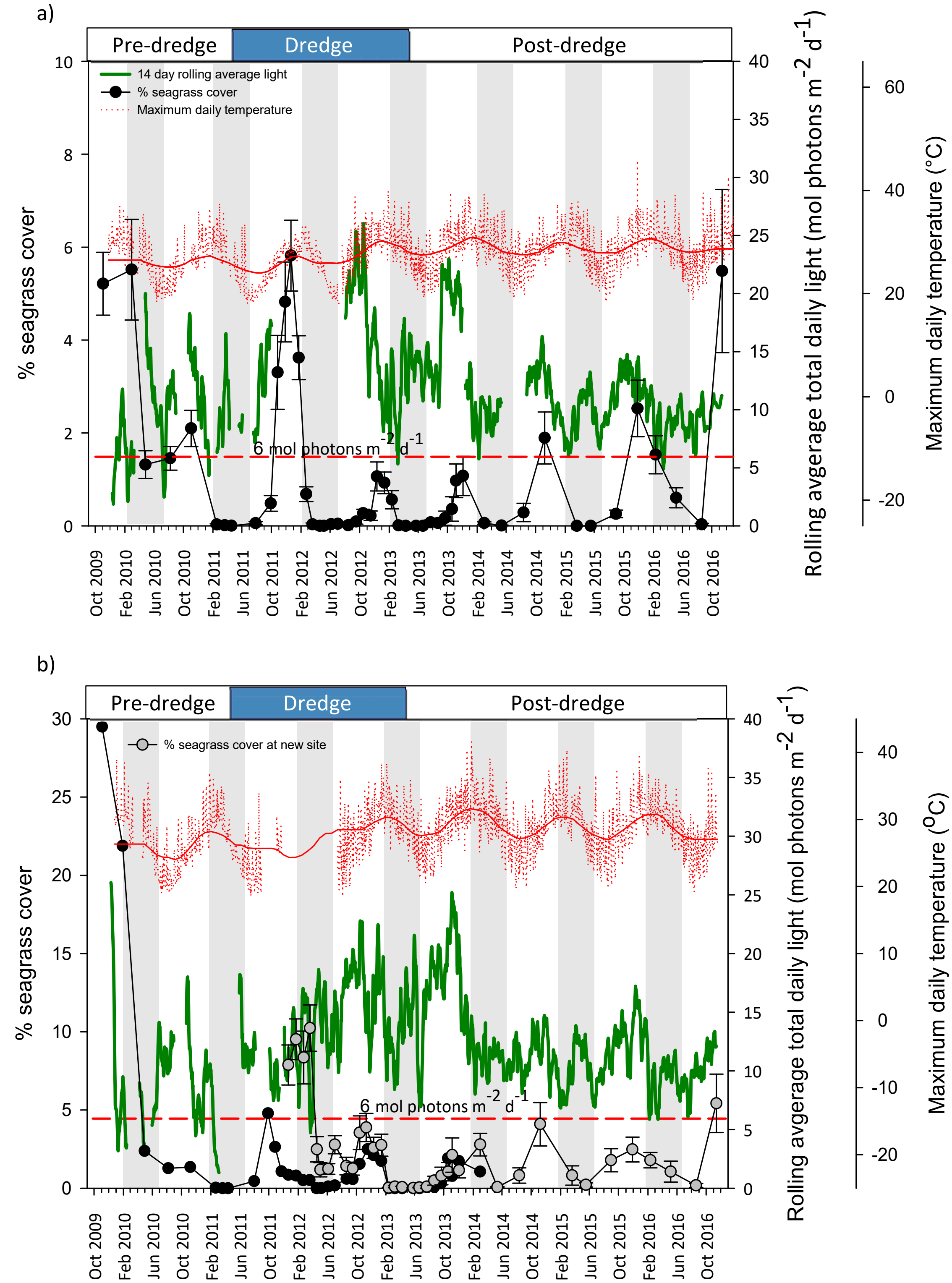


Figure 26: Rolling 14 day average total daily light (mol photons m^{-2} d^{-1}) and seagrass percent cover at Fisherman's Landing (a) and Wiggins Island (b) in the Inner Harbour over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Port Curtis and Rodds Bay. Light and temperature data (to November 2013) sourced from Vision Environment QLD (2013).

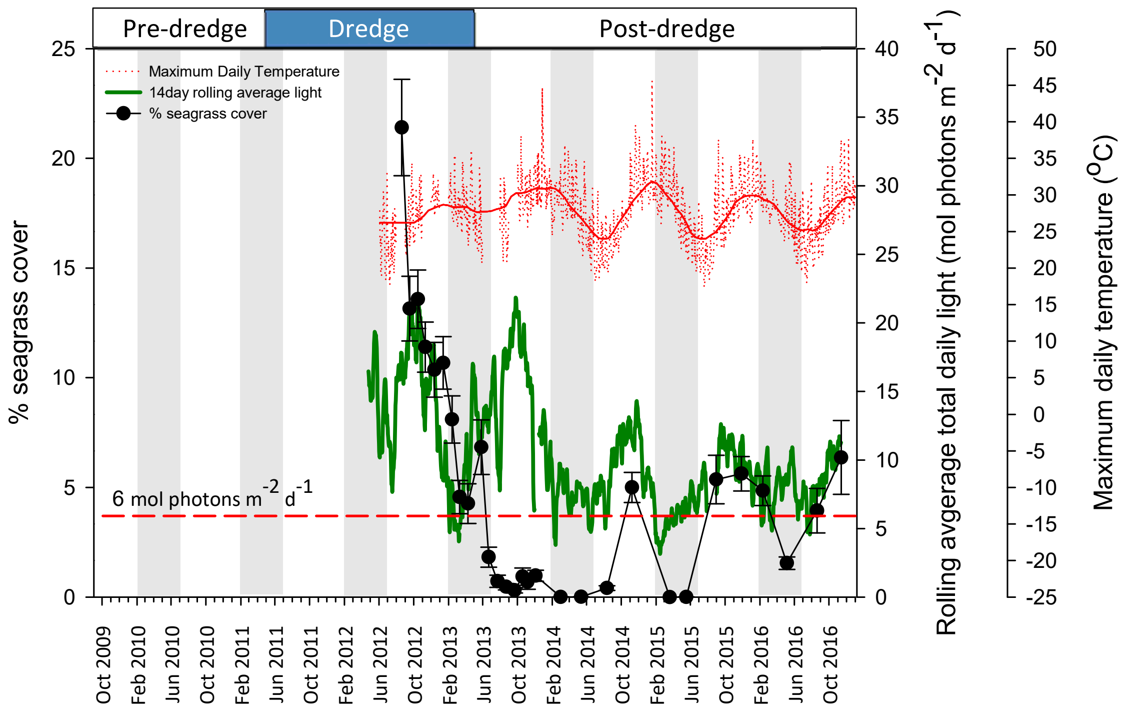
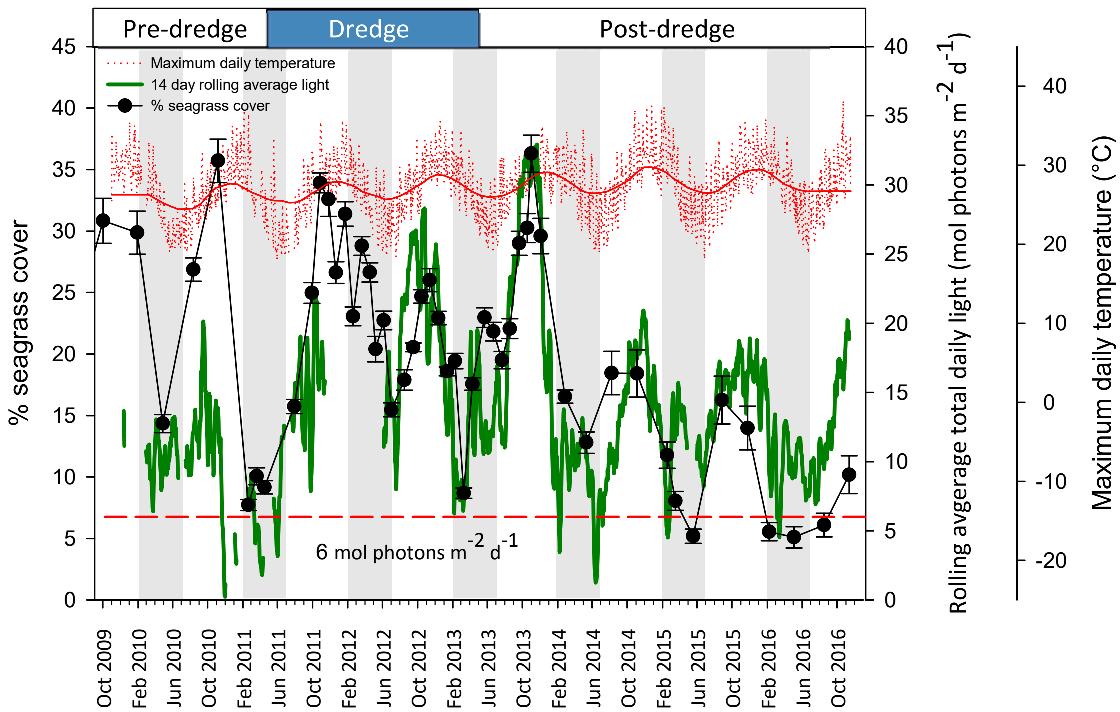


Figure 27: Rolling 14 day average total daily light (mol photons m^{-2} d^{-1}) and seagrass percent cover at Black Swan in the Inner Harbour over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Port Curtis and Rodds Bay. Light and temperature data (to November 2013) sourced from Vision Environment QLD (2013).

a)



b)

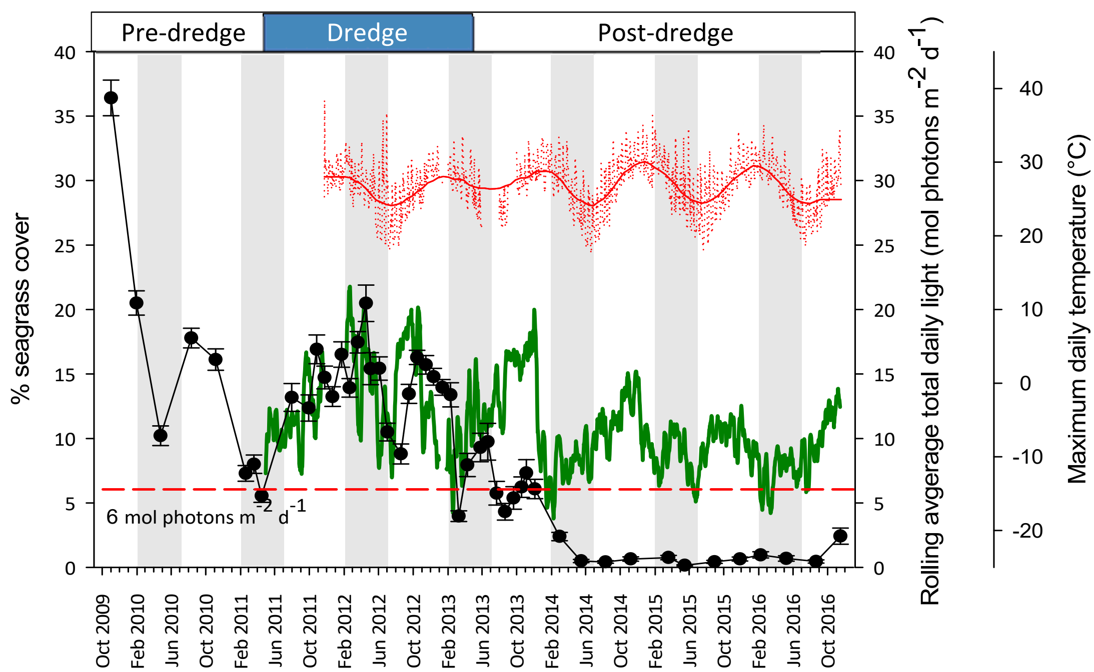
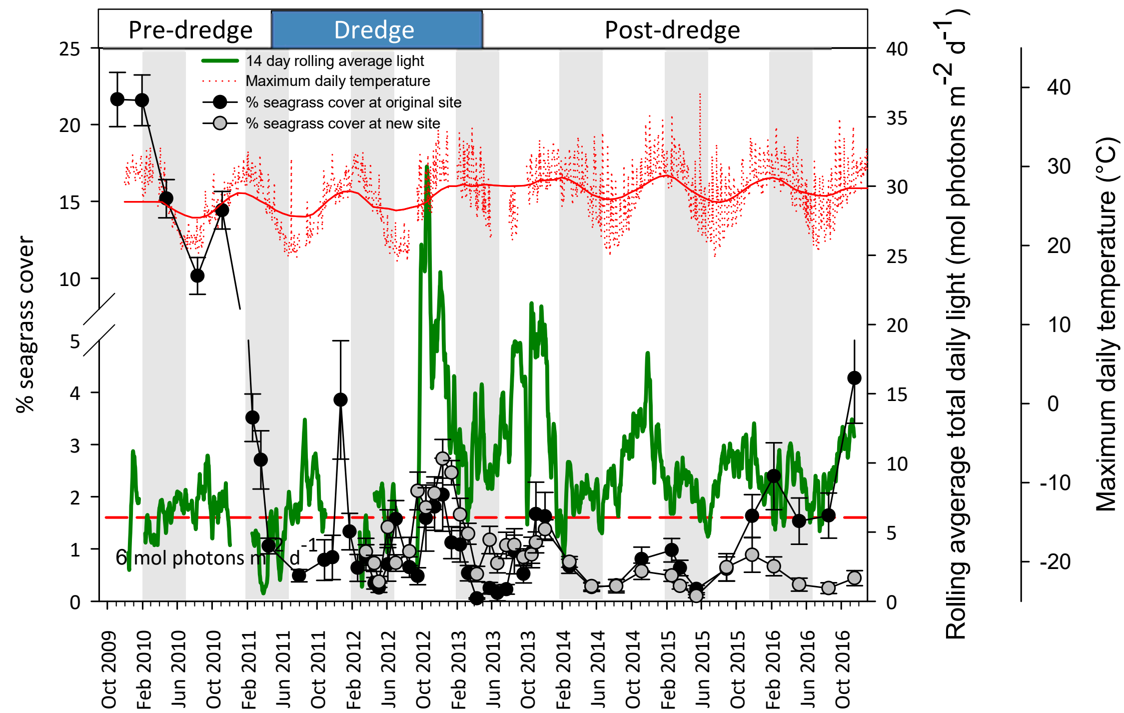


Figure 28: Rolling 14 day average total daily light (mol photons m^{-2} d^{-1}) and seagrass percent cover at Pelican Banks North (a) and Pelican Banks South (b) in the Outer Harbour over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Port Curtis and Rodds Bay. Light and temperature data (to November 2013) sourced from Vision Environment QLD (2013).

a)



b)

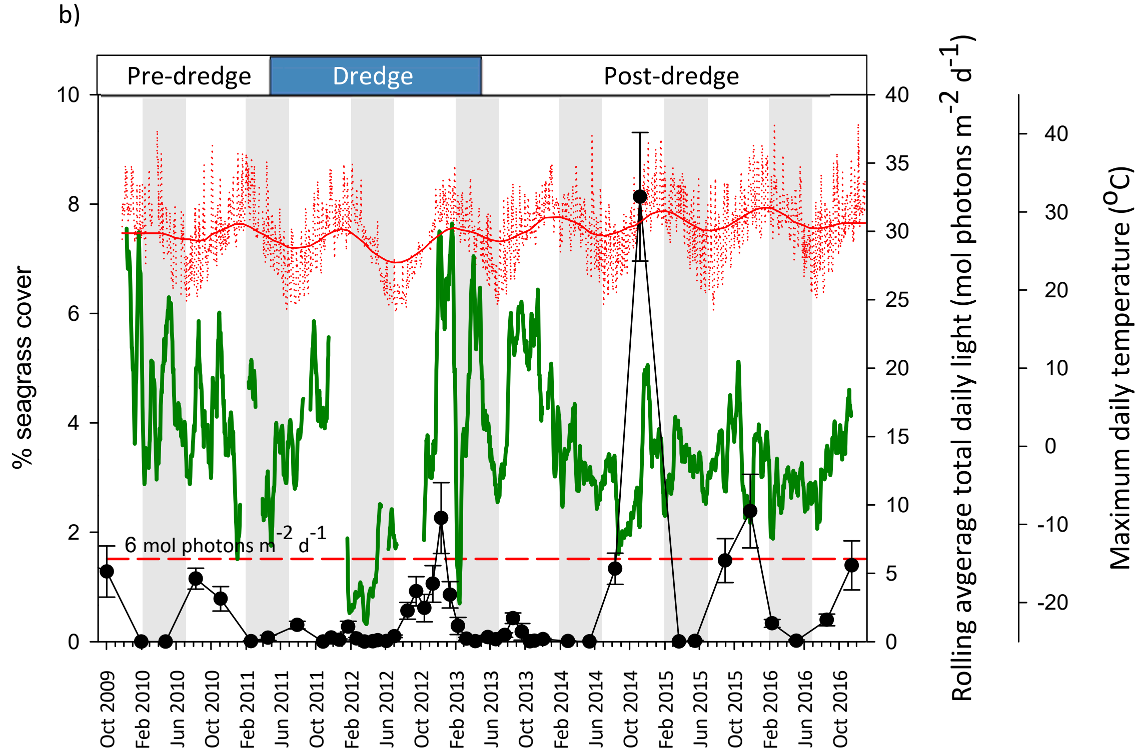


Figure 29: Rolling 14 day average total daily light (mol photons m^{-2} d^{-1}) and seagrass percent cover at Facing Island (a) and Rodds Bay (b) over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Port Curtis and Rodds Bay. Light and temperature (to November 2013) data sourced from Vision Environment QLD (2013).

a)

b)

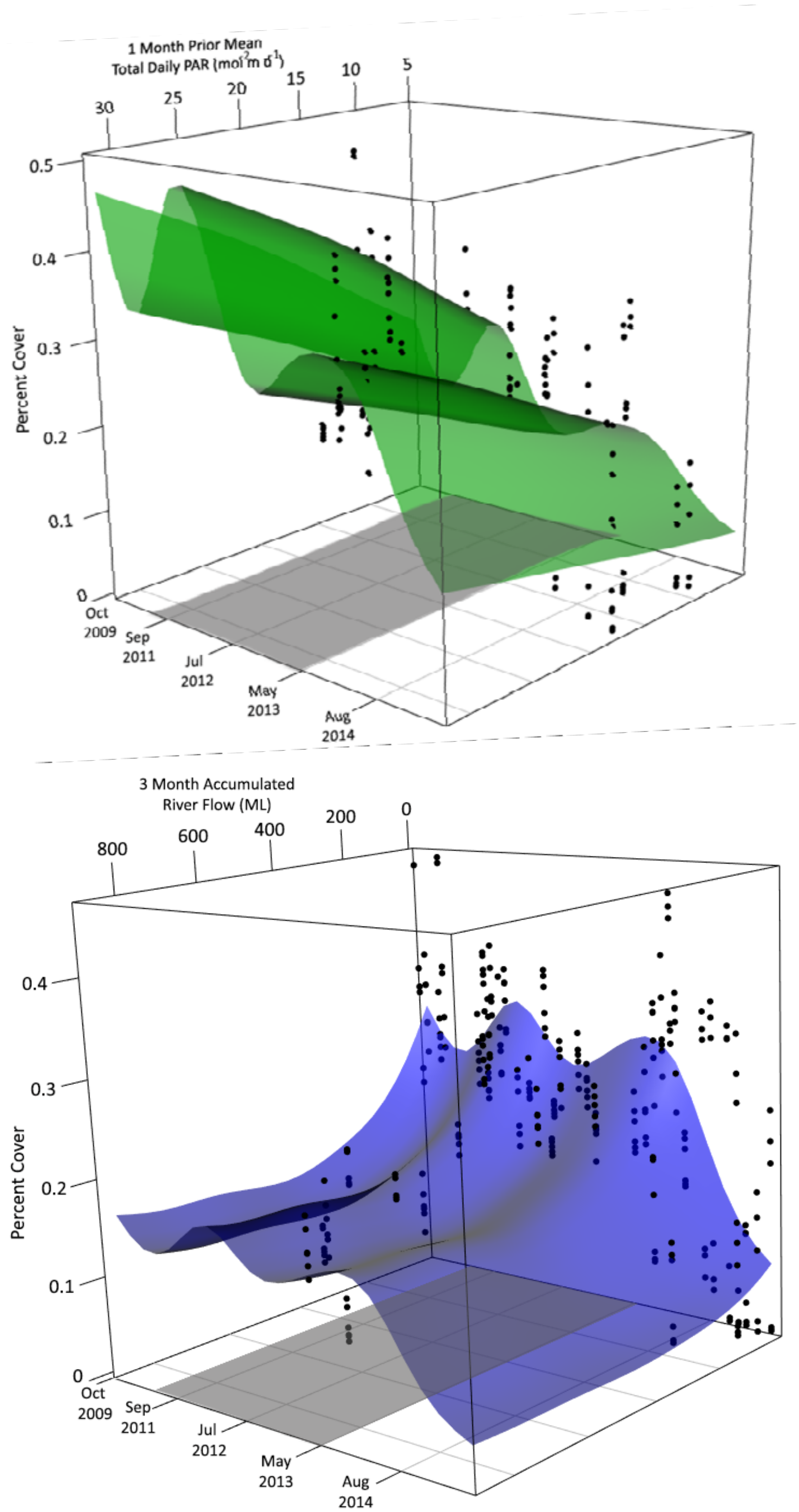
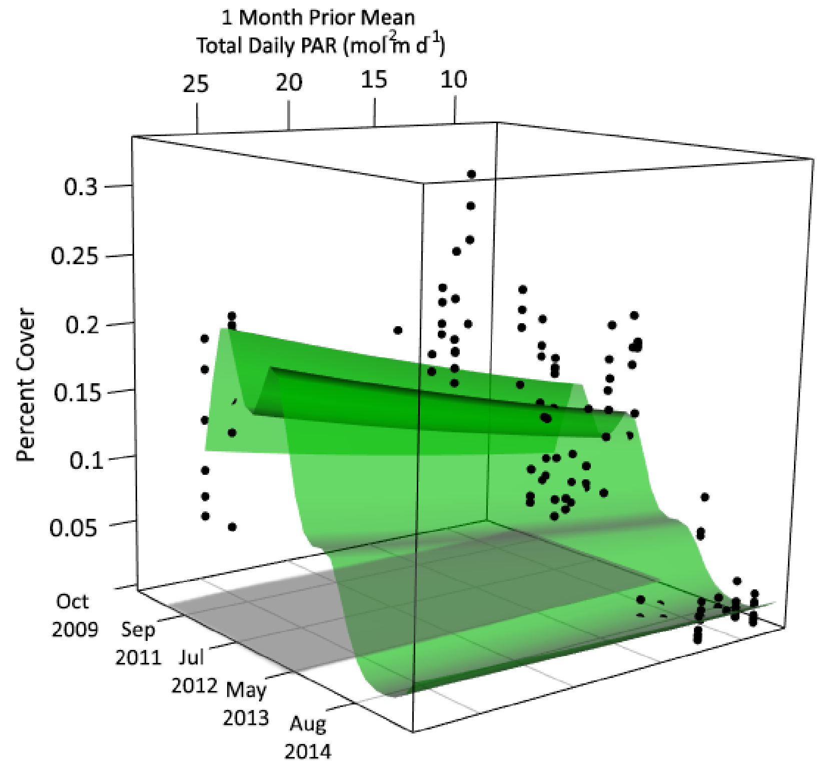


Figure 30: Predicted percent cover values based on final model output for Pelican Banks North seagrass percent cover by a) date and PAR (mol photons m^{-2} d^{-1}) and b) by sampling date and three month accumulated total river flow (x 1,000 ML) over the duration of the monitoring program. Greyed out region represents sampling events during dredging.

a)



b)

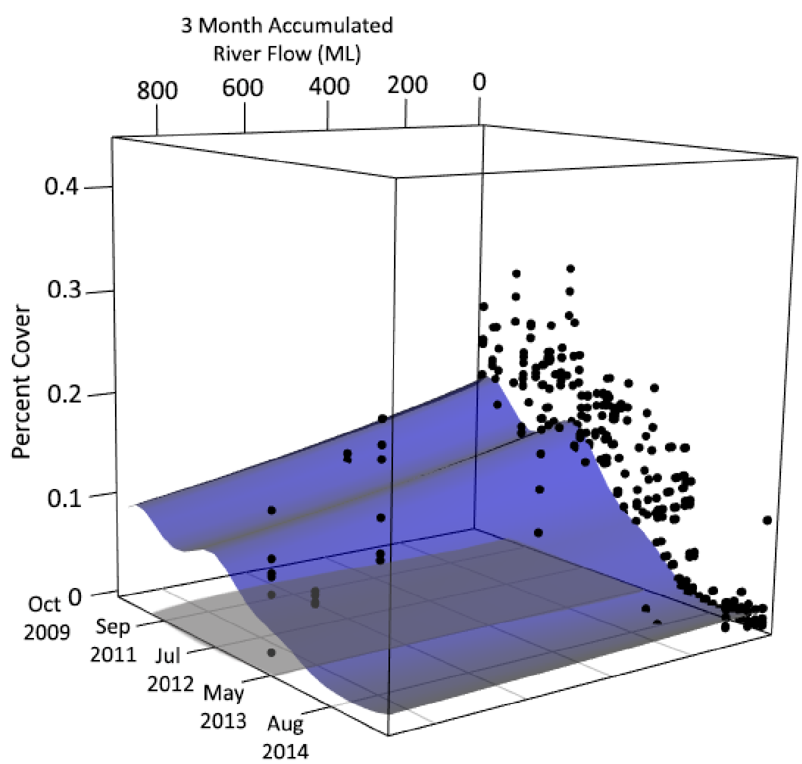


Figure 31: Predicted percent cover values based on final model output for Pelican Banks South seagrass percent cover by a) date and PAR (mol photons m^{-2} d^{-1}) and b) seagrass percent cover by sampling date and three month accumulated river flow (x 1,000 ML) over the duration of the monitoring program. Greyed out region represents sampling events during dredging.

Table 1: Significant values are in bold. Overall fit of selected best models (quasibinomial generalised additive mixed models (GAMM)) of Percent Cover (*C*) for (1) Pelican Banks North and (2) Pelican Banks South, including the proportion of variance explained by the model (R^2), estimated degrees of freedom (e*df*), *F*-statistic and *p*-values. Covariates are PAR (*P*), sampling date (*D*) and cumulative three month river flow (*R*).

Model	R^2	Model terms		*edf*	*F*	*p*-value
Pelican Banks North	0.62	Smoother terms	*PAR x sampling date*	20.57	10.99	< 0.001
s(*D* X *P*) + s(*R*)			*River flow*	2.23	16.18	< 0. 001
Pelican Banks South	0.63	Smoother terms	*sampling date*	8.03	52.085	< 0. 001
s(*D*) + s(*P*) + s(*R*)			*PAR*	1.00	7.593	< 0. 01
			River flow	2.07	8.23	< 0. 001

4.4 Discussion

In seagrass ecosystems, nutrients and light are the most common limiting factors that control seagrass abundance (Lee et al. 2007b). Major flood events and associated river flows deliver large volumes of highly turbid water to shallow coastal areas often resulting in increased concentrations of silt, organic matter and nutrients (Kamp-Nielsen et al. 2002) and reducing levels of light available to seagrasses. River flow has a significant negative effect on seagrass abundance in Port Curtis, explaining up to 73 percent of the variation in abundance depending on the location of the monitoring site. The strongest relationships found were at Wiggins Island and Pelican Banks South which are also the closest sites to the mouth of the Calliope River (McCormack et al. 2013b).

Over the course of monitoring (since November 2009), Port Curtis has experienced some of the most intense rainfall and river flow events on record. Flow from the Calliope River peaked above the long-term average during each and every summer wet season over the monitoring period, compared with only two events in the decade preceding the monitoring program. The most extreme rainfall and river flow events occurred very early in the program (February 2010 and December through March 2011) then again in January 2013. Significant declines in seagrass abundance occurred across all Inner and Outer Harbour permanent transect sites from early 2010 to early 2011 following extreme rainfall and river flow events (McCormack et al. 2013a). The light record during this time shows that the rolling average daily light fell below the minimum requirement for *Z. muelleri* growth (Chartrand et al. 2016) for extended periods. From mid-2011 through to the end of 2012, the Port Curtis region experienced a reprieve from extreme weather events and the light environment improved at most Western Basin sites. Seagrass abundance showed significant recovery over the 2011 growing season; however some sites (e.g. Wiggins Island and Pelican Banks South) failed to recover to pre-flood levels (McCormack et al. 2013a).

In January 2013, the Calliope River discharged at record levels and significant declines in seagrass percent cover were again detected at sites throughout Port Curtis. Since the event, monitoring at transect sites revealed different rates of recovery. The most noticeable difference was between Pelican Banks North and Pelican Banks South. At Pelican Banks North, light conditions remained favourable for seagrass growth throughout 2013 and seagrasses recovered well, increasing in abundance throughout the year and peaking in November at a similar level as previous years (Bryant et al. 2014a). At Pelican Banks South however, initial recovery was short lived and seagrass abundance was atypically low over the 2013 growing season. An additional peak river flow event in March 2014 likely exacerbated seasonal declines at both sites (Davies et al. 2015). Pelican Banks North continued to follow a downward trajectory for seagrasses from typical growing season peaks.

Following seasonal declines in 2014, seagrass abundance at Pelicans South reached the lowest point recorded over the history of the monitoring program. There has been very little recovery at the site in subsequent growing seasons, and despite light consistently exceeding minimum requirements for *Z. muelleri* growth (Chartrand et al. 2016), the species is almost absent from the site. At Pelican Banks North, trends in seagrass abundance continued to closely match trends in the light history; however the pattern shows a falling peak in seagrass cover almost consecutively since 2013. The modelled results for both Pelican Banks North and South sites shows a strong relationship (R^2 = 0.62 and 0.63 respectively) between the trend in seagrass cover over time and the three-month cumulative river flow and one-month mean total daily light since WBDPP permanent transect site monitoring began. While the models demonstrate light and river flow are important factors in determining seagrass growth there are a range of other variables not measured in the program that can affect monitored sites. Environmental factors such as temperature and nutrients as well as propagule supply are also likely to be key influences on seagrass change.

Recent research in Port Curtis (Chartrand et al. 2016) and elsewhere in Queensland (Collier et al. 2011) indicates that temperature may be a key factor affecting tropical seagrass seasonality and survival. Chartrand et al. (2016) documented a correlation between water temperature and seagrass biomass up until temperature exceeded 30°C, after which seagrass declined, despite high light intensity. According to Chartrand et al. (2016), warmer conditions bring with them a high metabolic demand. In the presence of high light, photosynthetic processes will keep up with rising seasonal temperatures up until a point, after which respiration continues to increase but photosynthesis does not (Lee et al. 2007b; Bulthuis 1987). Monitoring at permanent transects has shown that annual seagrass senescence begins at approximately the start of the year when temperatures consistently reach >30°C at the seagrass canopy and rain and flooding cause reductions in light (Chartrand et al. 2016). The precise thermal tolerances of Port Curtis and Rodds Bay seagrasses is not well understood but in a similar population, *Z. muelleri* carbon fixation and above-ground biomass have been shown to decline significantly under saturating light levels in conjunction temperatures of >33°C (Collier et al. 2011). Temperatures at several of our permanent transect sites have commonly reached levels of >33°C and up to 40°C over the spring and summer months especially in the past two years. Sustained high temperatures during the 2014 growing season and exposure-related stress caused by high total daytime tidal exposure at the beginning of the 2015 growing season may play a part in biomass declines detected at Pelican Banks in November 2015 (Davies et al. 2016). Further studies would greatly assist our understanding of the degree to which temperature affects light thresholds for *Z. muelleri* in Port Curtis.

Other factors in addition to rainfall and river flow may also be contributing to dynamics at Pelican Banks North, the progressive decline measured at Pelican Banks South, and declines in the wider meadow measured through the annual monitoring program (Rasheed et al. 2017b). First, observations and studies into dugong and turtle feeding activity at Pelican Banks have highlighted the strong grazing pressure from resident and/or migrant populations on this meadow (Rasheed et al. 2017a). With no data on feeding pressure prior to the 2015-2016 study, it is unknown whether feeding activity was stable or had increased in the meadow relative to previous years. If feeding activity in fact has increased, the recent declines in above-ground biomass and cover may be an indication it is being grazed more than in years' past rather than anthropogenic and/or climate-driven declines.

A second factor that was not measured directly through either the permanent transect site monitoring program nor in the annual monitoring program was sedimentation rates in local seagrass meadows. Research has shown that changes to sedimentary dynamics, whether due to high catchment loading or from modifications to coastal landscapes, can lead to seagrass loss. Benham et al. (2016) measured sedimentation rates at Pelican Banks in 2014 and 2015 which was used to experimentally test the effects of burial together with reduced light on *Z. muelleri* growth rates (Benham et al. 2016). Results showed significant impact to shoot density and rhizome growth rates when plants were shaded and placed under burial conditions concomitant with measured sedimentation rates at Pelican Banks. This study suggests burial could play a role in Pelican

Banks *Z. muelleri* declines over the last few years; however the source of sediments, if increasing in the meadow, is uncertain between catchment sources versus local port sediment movements and/or a mixture of both. It is also unclear if net sedimentation has occurred at Pelican Banks, as deposition may well be balanced by events and conditions that cause sediments to move off the banks.

On a longer time-scale than measured as part of the seagrass permanent transect site program, decadal scale climate patterns should be considered in order to understand broader patterns in environmental drivers in the Port Curtis region. For example, the Southern Oscillation Index (SOI) is an indicator of El Niño and La Niña conditions which are correlated with below- and above-average rainfall years respectively (Figure 32; bom.gov.au). Values between about +7 and −7 generally indicate neutral conditions, while sustained values above +7 are typically considered La Niña and sustained values below -7 are typically El Niño (www.bom.gov.au/climate/). When baseline mapping of Port Curtis and Rodds Bay seagrasses occurred in 2002 and for the subsequent six years, the SOI was in an El Niño or neutral pattern which correlates with the recorded low rainfall and river flow in the Port Curtis and Rodds Bay region including the Calliope River. The 2010 – 2012 period shows strong La Niña patterns which coincides with the large rainfall and flooding events in 2010/2011 wet season and drove large seagrass declines across the greater central Queensland coast (McKenzie et al. 2014; Rasheed et al. 2014). While individual storm and river flow events can occur outside of these decadal scale patterns, longer-term cycles in rainfall and river flow may be a factor for patterns in Port Curtis seagrasses. It is unclear what state seagrasses may have been in during previous prolonged La Nina cycles such as in the late 1990s (Figure 32).

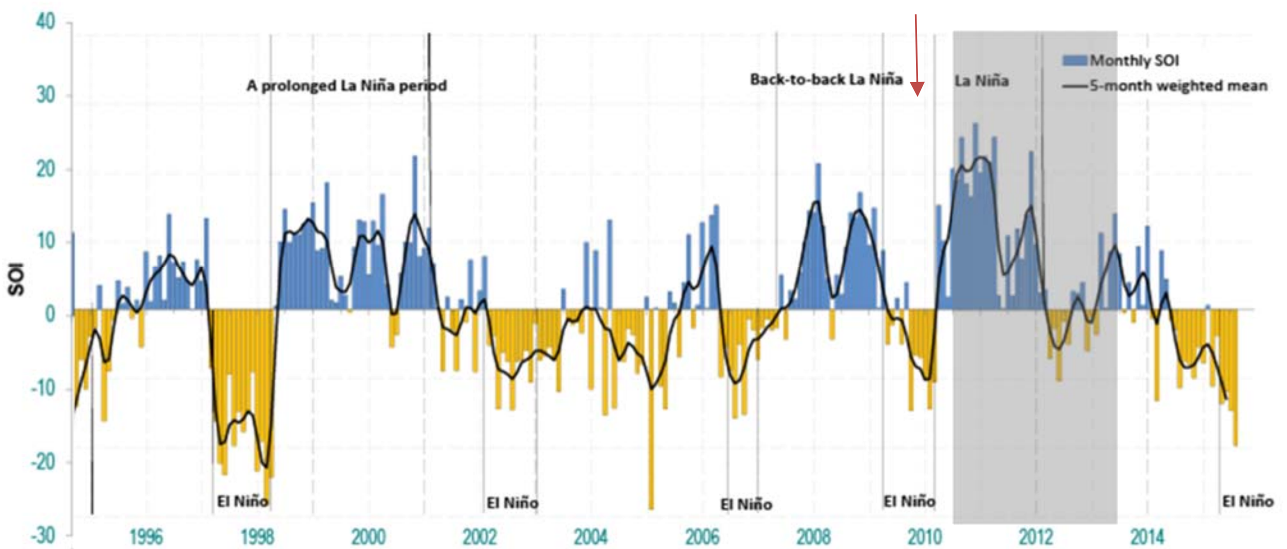


Figure 32: Timeline of monthly Southern Oscillation Index (SOI) values from 1996 – 2014. Sustained negative values (bottom/yellow) of the SOI below −7 may indicate El Niño, while sustained positive values above +7 may indicate La Niña. La Niña and El Niño events since 1900 are indicated on the graph. Red arrow indicates start of permanent transect site monitoring program and greyed out region is during the WBDPP active dredging campaign. Data and plot adapted from Bureau of Meteorology (http://www.bom.gov.au/climate/).

5 CONCLUSIONS

The seagrass permanent transect site monitoring program was established to detect significant shifts in seagrass condition in the context of dredging activity and during dredging, act as the sentinel assessment sites to inform active dredge management. The program spans the pre-, during, and post-dredge phases providing a detailed description of seagrass abundance at both Inner and Outer Harbour locations with varying proximity to dredge activity and covering the range of species and meadow types typical throughout the wider Port Curtis and Rodds Bay region. The significant declines in seagrass at many sites including Wiggins Island (original site), Facing Island (original site), Pelican Banks South, and Rodds Bay occurred prior to dredging and were likely related to region-wide flood and storm events. Declines at other locations such as Black Swan, Wiggins Island (new site), Facing Island (new site), and Pelican Banks North are more difficult to pinpoint definitive drivers. Major rainfall and river flow events during dredging likely impacted these locations; however any interactive effect of dredging with significant storm and river flow events was not possible to determine due to the coincidence of timing of these events.

Long-term monitoring of seagrass condition at permanent transect sites in Port Curtis and at Rodds Bay has revealed significant intra- and inter-annual variability in seagrass abundance, species composition, tissue nutrient characteristics and reproductive output across sites. Within years, trends in seagrass abundance have generally followed a seasonal pattern; increasing over the growing season from around July or August, peaking in abundance around November, and declining in summer with increasing temperatures and the onset of heavy rainfall and associated changes in light availability. Over the course of monitoring (since November 2009) several intense rainfall and flood events have led to declines in seagrass abundance and/or shifts in species composition at most sites. Longer term historical data available at both Pelican Banks North and Wiggins Island suggest that the declines documented over the course of the current monitoring program may be part of the natural weather/climate induced patterns of inter-annual variation. However, given the frequency and intensity of recent weather events including two of the largest flood events on record, the resilience of some seagrass meadows may be reduced making them more vulnerable to further impacts, be they natural or anthropogenic in nature.

The timing of flood related declines in seagrass abundance immediately prior to the onset of the major dredging activities makes it difficult to ascertain what additional impact, if any, dredging may have had on seagrass condition and rates of recovery at the permanent transect sites. As part of the Western Basin dredge management plan, thresholds and triggers have been established that are directly related to the acute light requirements of local seagrass species (Chartrand et al. 2016; Chartrand et al. 2012). The WBDDP dredging activities were actively managed to ensure that seagrasses received enough light for their growing requirements at these seagrass sites. The program was largely successful in achieving this and generally seagrass growth during the growing season has not been limited by light falling below requirements (outside of major flood events). Recent research suggests that temperature is also likely to be a major driver of seasonal changes in seagrass abundance in Port Curtis (Chartrand et al. 2016). Additional studies to determine the impact of temperatures on seagrass light requirements in Port Curtis would allow further refinement of the existing management tools as well as further studies looking at the interactive effects of reduced light and sediment loading on local seagrasses.

In January 2013, the Calliope River discharged at unprecedented levels causing declines in seagrass across permanent transects sites. At Inner Harbour sites, there has been some signs of recovery, with gradual increases in abundance each growing season at Fisherman's Landing. Unfortunately, due to the relocation of Wiggins Island site and the late addition of the Black Swan site, detailed tracking of pre- to post- dredge phases in not possible solely at permanent transect sites but is more broadly assessed through the larger scale annual meadow monitoring programs. However, seagrass abundance in the Inner Harbour remains well below levels detected at the onset of quarterly transect monitoring (November 2009) and recovery to these initial levels is likely to take several years. The gradual loss of the dominant species *Z. muelleri* at some Inner Harbour sites means that the sediment seed bank is unlikely to be locally replenished and recovery will rely on the transport of propagules (seeds or fragments) from other areas of the Port Curtis.

At Outer Harbour sites, recovery since the 2013 flood event was initially promising; however, monitoring during the growing seasons since 2014 has shown seagrass abundance to be at the lowest levels recorded over the course of the program in consecutive years. Levels of resilience at Pelican Banks North are particularly concerning with little indication reproductive effort has been maintained through flowering and fruit production and a concurrent shrinking of a once dense seed bank. Significant increases at the meadow after declines in 2006 and following major rainfall and flood events in 2010/11 and 2013 suggest that the meadow has been able to recover in the past if conditions are favourable. At the southern end of Pelican Banks, the low density of plants, prolonged nutrient enrichment and a lack of propagules also appears to be hampering recovery. The loss of the dominant species *Z. muelleri* from the site in 2015 and 2016, despite sufficient light over the growing season confirms that these other factors are also important drivers of seagrass abundance.

Annual surveys of complete seagrass distribution and abundance in the Western Basin region of Port Curtis in November 2016 (Rasheed et al. 2017b) were generally in line with transect monitoring results showing a mixed result among meadows and no explicit relationship between distance from port activity and meadow condition. Above-ground biomass at the Pelican Banks meadow was the lowest recorded over the history of monitoring (since 2002) and seagrass condition as graded for the whole of the annual assessment was deemed poor (Rasheed et al. 2017b). Rasheed et al. (2017b) could not definitely attribute the severe declines in biomass and area at Pelican Banks in November 2016 to any one strong driver but rather identify a number of potential contributing factors such as herbivory, sediment movement, and cumulative impacts from floods, cyclones and anthropogenic pressures likely at play. While similar in overall findings the long-term and spatially expansive monitoring undertaken during annual re-mapping surveys seems to better capture overall trends than the transect-based monitoring. Differences in results occur between the programs, due to the spatial variability in location of some meadows and the shifting mosaic of seagrass density within meadow boundaries. At locations like Rodds Bay for example this at times results in the permanent transect sites being outside of the meadow footprint and consequently underestimating seagrass presence in the region.

Results of seagrass monitoring over the entirety of the dredging program have provided some insight into the capacity of seagrass resilience to human activities and environmental perturbations. A number of seagrass meadows in Port Curtis and Rodds Bay (e.g. Pelican Banks) have decreased resilience as measured at the permanent transect sites making the tools and thresholds established through major research programs in Port Curtis (Schliep et al. 2014; Chartrand et al. 2012) highly valuable for understanding how to best manage these sensitive communities until they recover from recent losses. Currently seagrasses have shown some capacity to recover from impacts in Port Curtis, but as has been seen in other Queensland locations repeated disturbances over multiple years may lead to longer-term loss, with recovery trajectories far less certain (McKenna et al. 2015; York et al. 2015; Rasheed et al. 2014; Pollard and Greenway 2013).

Given the likely low resilience of seagrasses in 2016, maintaining an understanding of the state of seed banks and the light environment as part of the seagrass monitoring program as the permanent transect site program has finished would assist in informing appropriate management actions to address causes of decline if they should continue, as well as understanding the changing state of resilience of seagrasses. The extensive and detailed seagrass monitoring and research efforts in Port Curtis and Rodds Bay means we are well placed to understand these processes, implement additional investigative studies to supplement current research and monitoring efforts, and look to implement measures to reduce the chances of exacerbating natural impacts by human activities.

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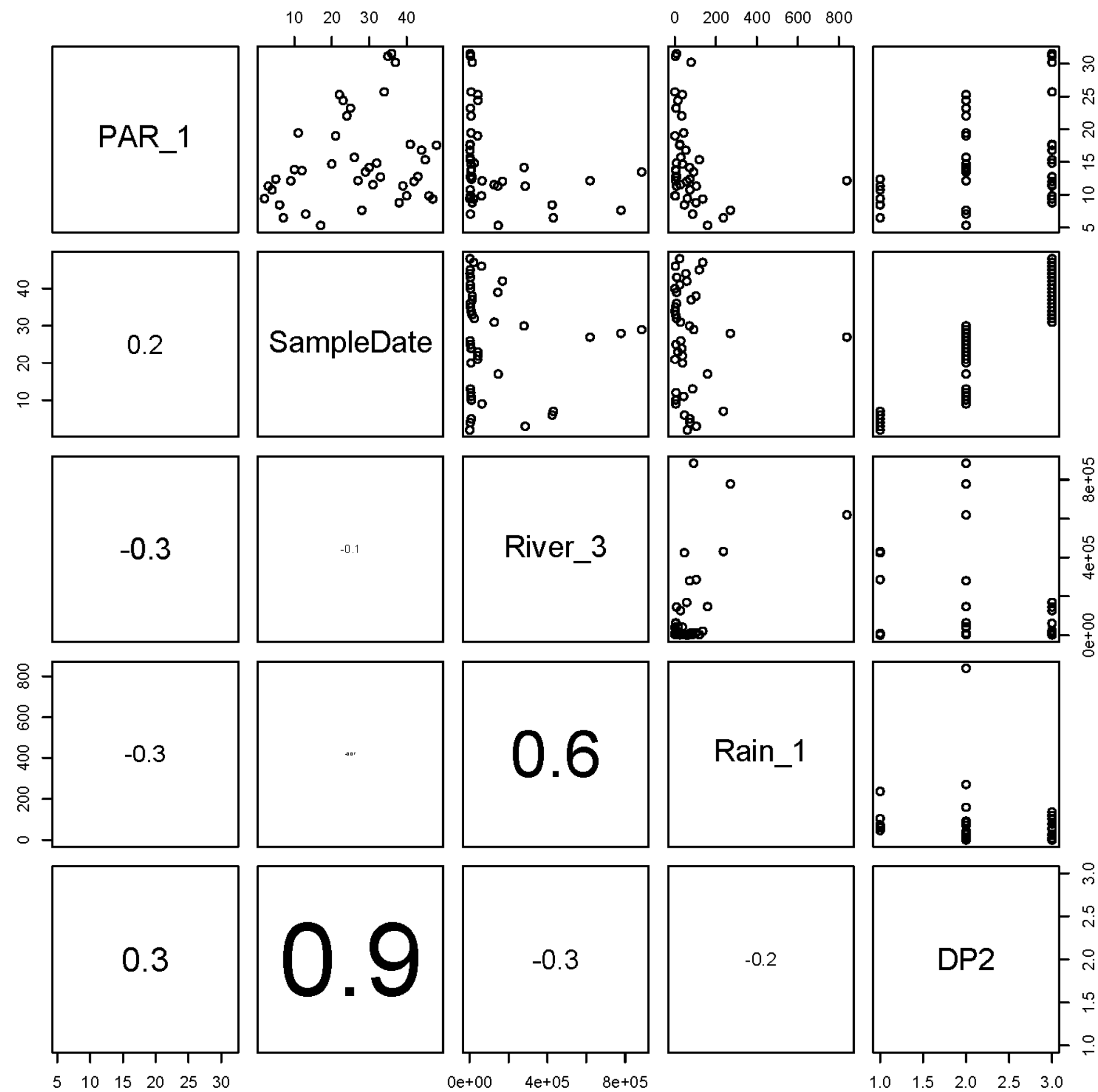
APPENDIX

Matrix of correlations for seagrass analysis

Scatterplot matrix of correlations between environmental variables prior to seagrass sampling and sample date and dredge period (DP2) at Port Curtis permanent transect sites. Values in bottom left portion of figure are Pearson correlation coefficients.

Pelican Banks North



Pelican Banks South