Long-Term Seagrass Monitoring in Port Curtis:

Quarterly Permanent Transect Monitoring Progress Report 2009 to 2015

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A report for Gladstone Port Corporation Limited

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KEY FINDINGS

This report summarises recent (2015) results from the assessment of seagrass change at permanent transect sites established in 2009 as part of the Western Basin Dredging and Disposal Project (WBDDP). 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 Davies et al. 2016).

- 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.
- 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 growth requirements during the dredging program.
- In January 2013, the Calliope River again discharged at record levels and declines in seagrass were detected across monitoring sites. Recovery over subsequent growing seasons has varied with some sites remaining atypically low in seagrass cover.
- 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.
- Sediment seed banks for *Zostera muelleri* subsp. *capricorni* were detected in the inner and outer harbour and at Rodds Bay at each quarterly monitoring event in 2015; however viability decreased across the senescent season (February to May 2015) especially in the outer harbour at Pelican Banks.

Results of seagrass monitoring over the post-dredging phase of the project will continue to provide insight into the capacity of seagrass resilience to human activities. If low levels of resilience detected at many sites persist then the tools and thresholds established through the Gladstone seagrass research programs will be critical in ensuring successful management of their recovery. Currently seagrasses have shown some capacity to recover from impacts in Gladstone, 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 Gladstone 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 Gladstone area.

The Western Basin Dredging and Disposal Project (WBDDP) posed a high level of environmental risk to marine habitats in the Gladstone area, 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. 2012).

In 2009, GPC commissioned a suite of research and monitoring including quarterly assessments of seagrass condition at key monitoring meadows throughout the harbour and in nearby Rodds Bay. The quarterly monitoring program adapts the established Seagrass-Watch protocols which utilise permanent intertidal transects at seagrass beds 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. 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 (Bryant et al. 2014b).

This report details findings of the quarterly seagrass assessment program at permanently established sites for the Gladstone Western Basin program. Results of this report form part of a broader suite of seagrass investigations that also include quarterly seagrass and light assessments (Bryant et al. 2016), annual seagrass mapping of all seagrasses in Port Curtis and Rodds Bay (Bryant et al. 2014b), a research and management program establishing the light requirements of Gladstone seagrasses (Chartrand et al. 2016), seed bank studies (Jarvis et al. 2015) 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 encompassed 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 future 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. 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 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;
- 3 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).

Sampling approach

Seagrass condition was monitored quarterly 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;

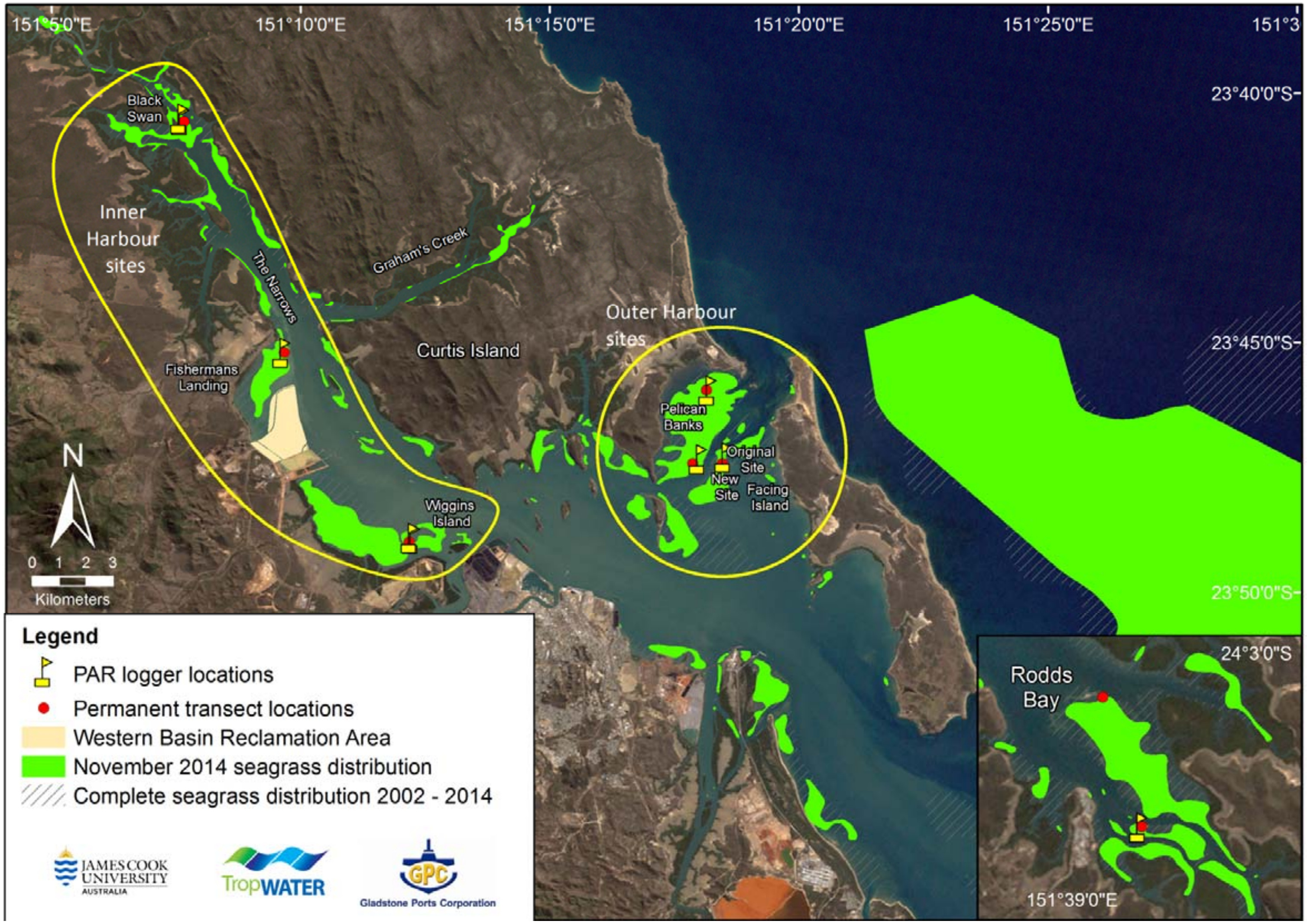
- Abundance and community composition (seagrass health)
- Elemental content of plants (seagrass tissue nutrients)
- Meadow reproductive status (seagrass resilience)
- Sexual above-ground productivity and asexual growth (seagrass productivity)

This report describes the results of monitoring conducted between October/November 2009 and November 2015.

Seagrass condition monitoring is directly linked to water quality assessments at each permanent transect location. Benthic light and temperature have been continuously collected at the seagrass canopy to aid in interpretation of seagrass changes and a suite of water quality parameters were collected as part of the broader Western Basin program.

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 growth 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, 2015.



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 bi-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 the harbour. A focused and more frequent assessment of seagrasses was essential in interpreting potential impacts of the WBDDP and in establishing seasonal trends.

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 (2009-15). Survey methodology followed Seagrass-Watch standard methodology (McKenzie et al. 2007; see also www.seagrasswatch.org) with the addition of seagrass biomass assessments (Mellors 1991). At each location, sampling included three 50m transects nested in each site and one or two sites nested in a location. A site was defined as a 50m x 50m 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 0.25m^2 sampling quadrat placed at 0m and then every 5m along transects (eleven sampling points per transect) (Figure 2).

Biomass assessments were made using photographs taken in the field. Observers assigned a rank describing the above-ground biomass of seagrass for each quadrat. When assigning ranks, the observer referred to a set of photographs of seagrass plots for which the above-ground biomass had previously been measured. The observer additionally 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 also 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 has been characterised by light and patchy seagrass cover (< 7%) over the duration of the monitoring program (Figure 3a). Percent cover has generally peaked between November and December, declining early in the year as seagrasses enter 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. Since that time seagrass cover has followed a seasonal trend, with declines during the senescent season reducing seagrass to extremely low levels. Recovery during the growing season has yet to reach peaks recorded in 2010 and 2011, but seagrass abundance at the height of the growing season has gradually increased since 2012.

Seagrass above-ground biomass has followed a similar seasonal trend as percent cover, with small increases over the growing season and declines early in the year. Biomass has remained very low (< 2g DW m^{-2}) over the duration of the program (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). As with percent cover, above-ground biomass at the height of the growing season has continued to increase since 2012 and the November 2015 survey recorded the highest biomass since November 2011.

Throughout monitoring *Halophila ovalis* has been the dominant species, except for the period from February to April in 2011 following a major flood event. Only a few shoots of *Z. muelleri* subsp. *capricorni* (herein referred to as *Z. muelleri*) remained at the site after the flood, with *H. ovalis* absent for several months (Figure 3b). Since 2012, the opposite has occurred; *H. ovalis* has persisted after the wet season with Z*. muelleri* only appearing late in the growing season (2012-2014) or not at all (2015).

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. 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 (Figure 4a).

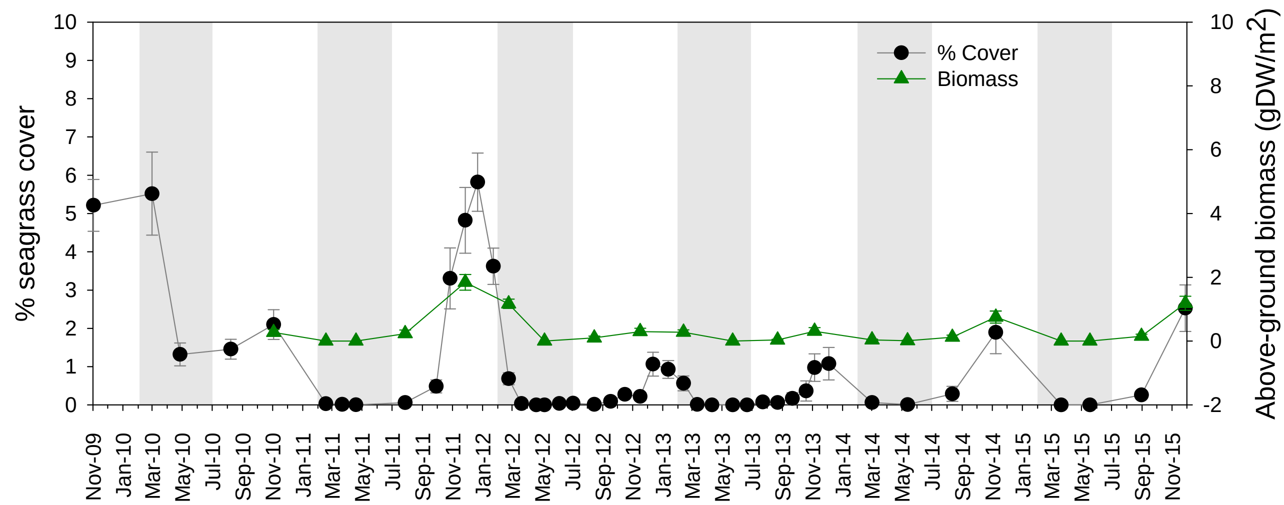
An additional permanent transect site was established on the western side of the bank in December 2011. From December 2011 to July 2012, the new site was higher in seagrass percent cover. However from August 2012 until the original site was decommissioned there was little difference between sites (Figure 4b). Seagrass has followed established seasonal trends with peaks during the growing season at similar levels (<5%) since November 2012. Above-ground biomass has followed a similar trend as percent cover for both sites, remaining below 2 g DW m^{-2} since November 2012 (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 and declined towards the senescent season when *H. ovalis* became dominant (Figure 5a). At the new site the reverse has been recorded; *Z. muelleri* was the dominant species over the senescent season with patches of *H. ovalis* increasing over the growing season (Figure 5b).

Narrows Zone 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 into the 2013 senescent season and has remained extremely low (<3%) (Figure 6). Peaks in November 2014 and August and November 2015 have shown slight improvements (~5% seagrass cover), and are encouraging signs for this site. The site has been dominated by *Z. muelleri* with the exception of February and May 2014 when *Z. muelleri* was absent and *H. ovalis* was the only species present (Figure 7).

a)



b)

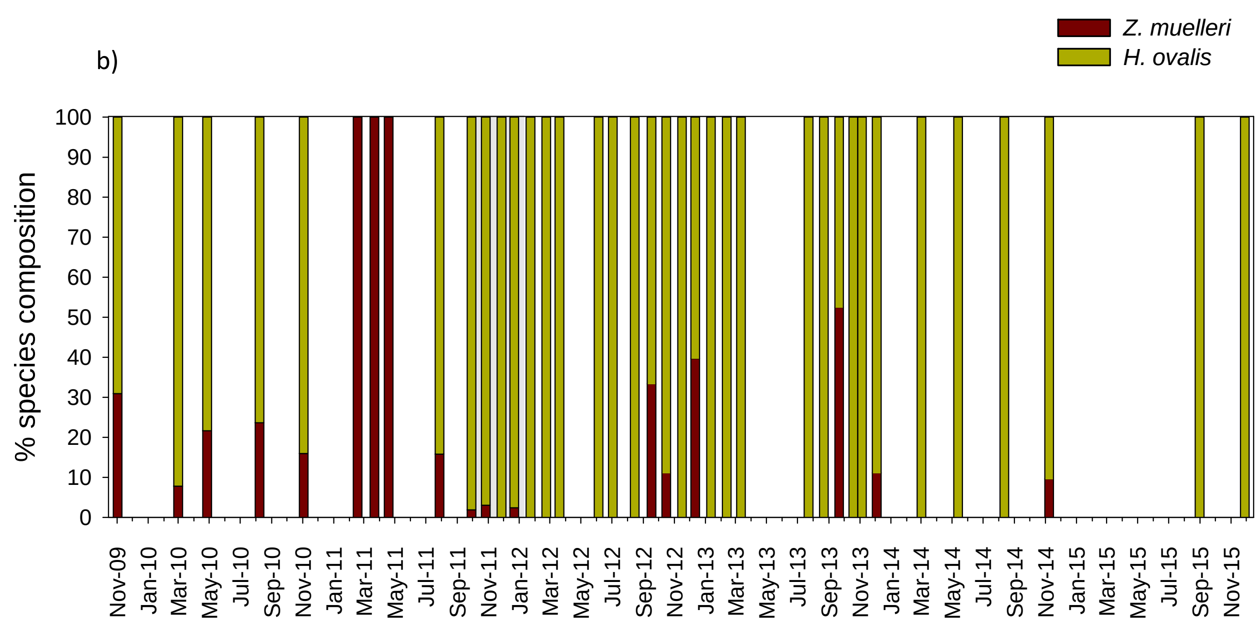
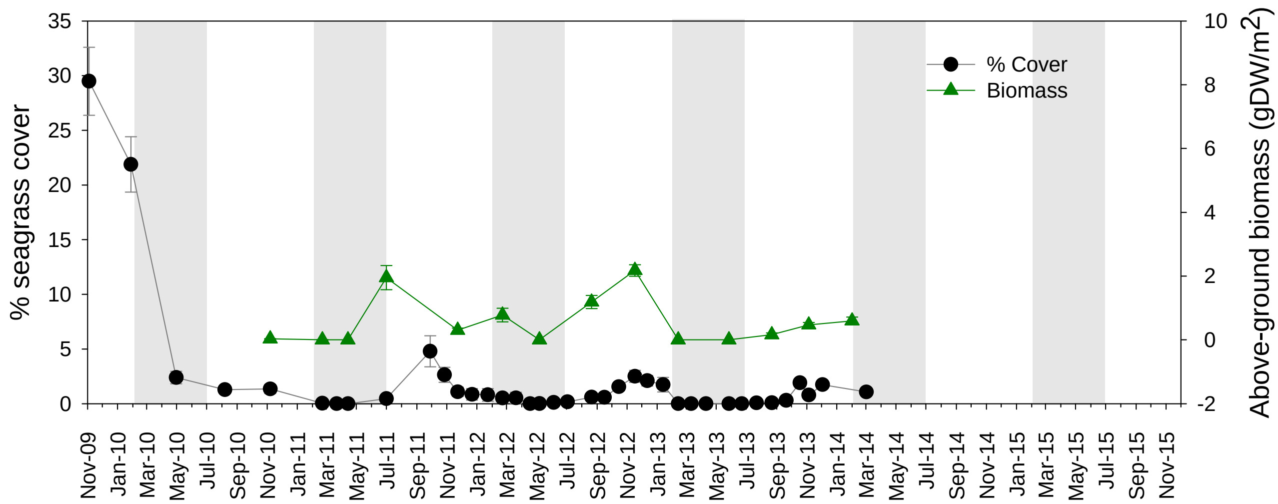


Figure 3: a) Seagrass abundance and b) seagrass species composition at permanent transect sites (pooled) at Fisherman's Landing, November 2009 – November 2015. Shaded area represents the seagrass senescent season.

a)



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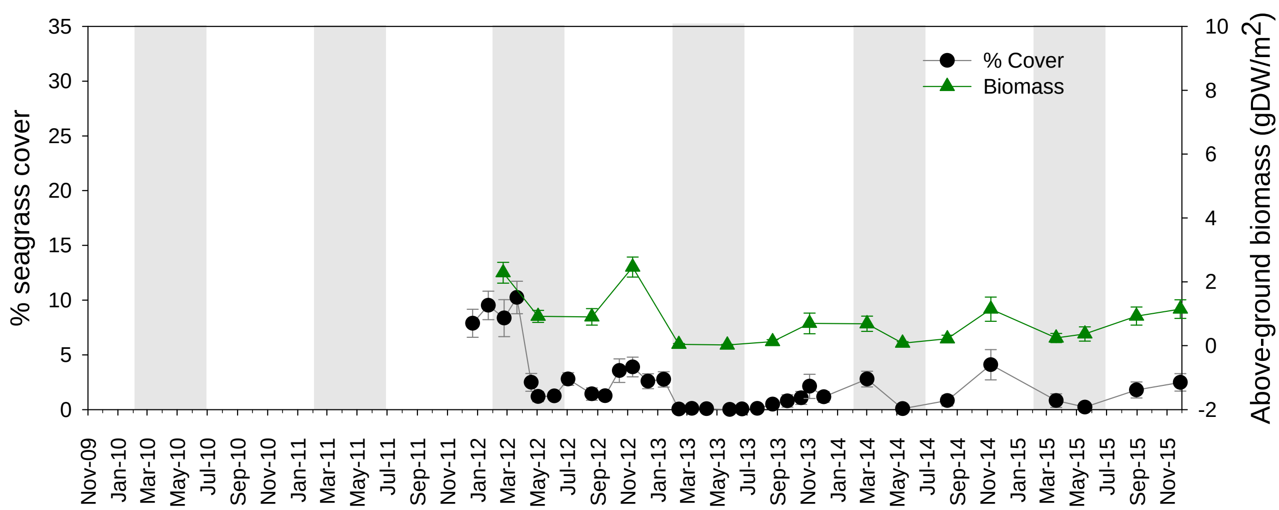
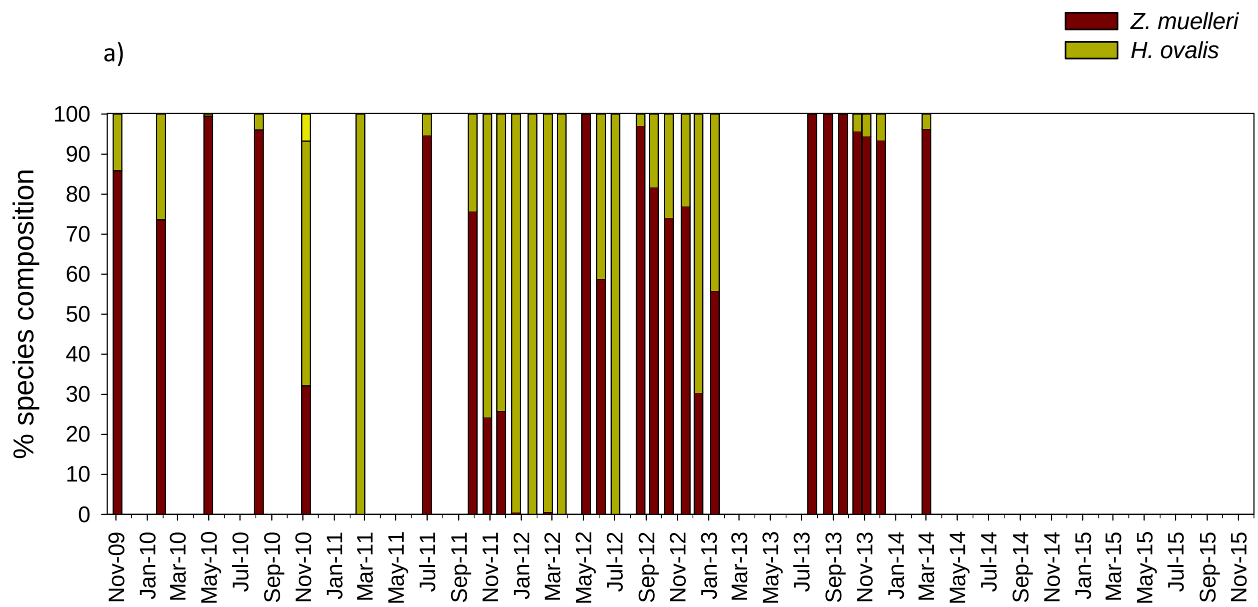


Figure 4: Seagrass abundance at the a) original and b) new permanent transect site at Wiggins Island, November 2009 – November 2015. Shaded area represents the seagrass senescent season.

a)



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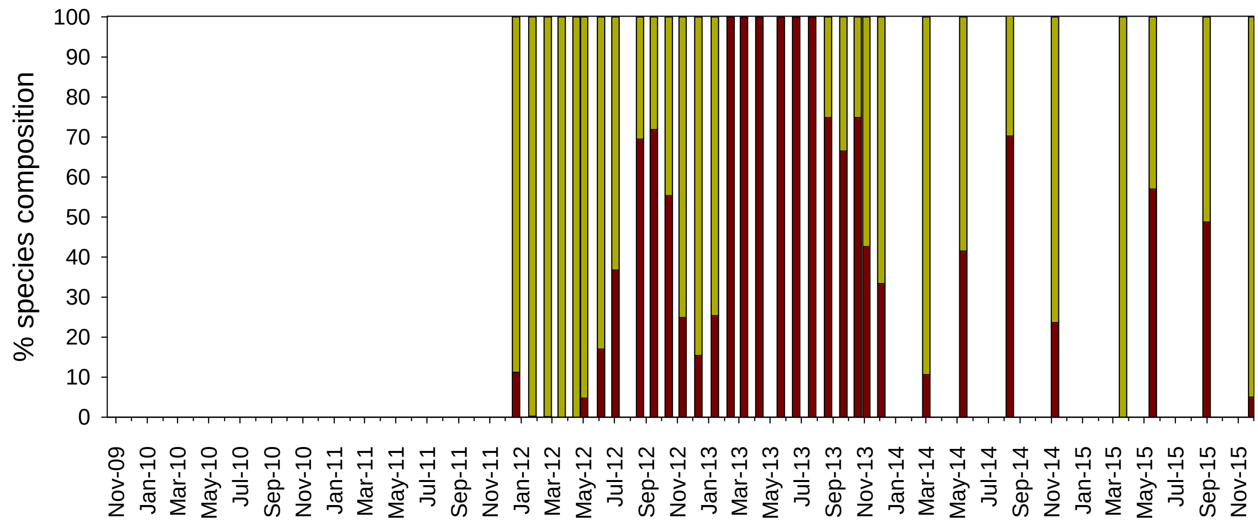


Figure 5: Seagrass species composition at the a) original and b) new permanent transect site at Wiggins Island, November 2009 – November 2015.

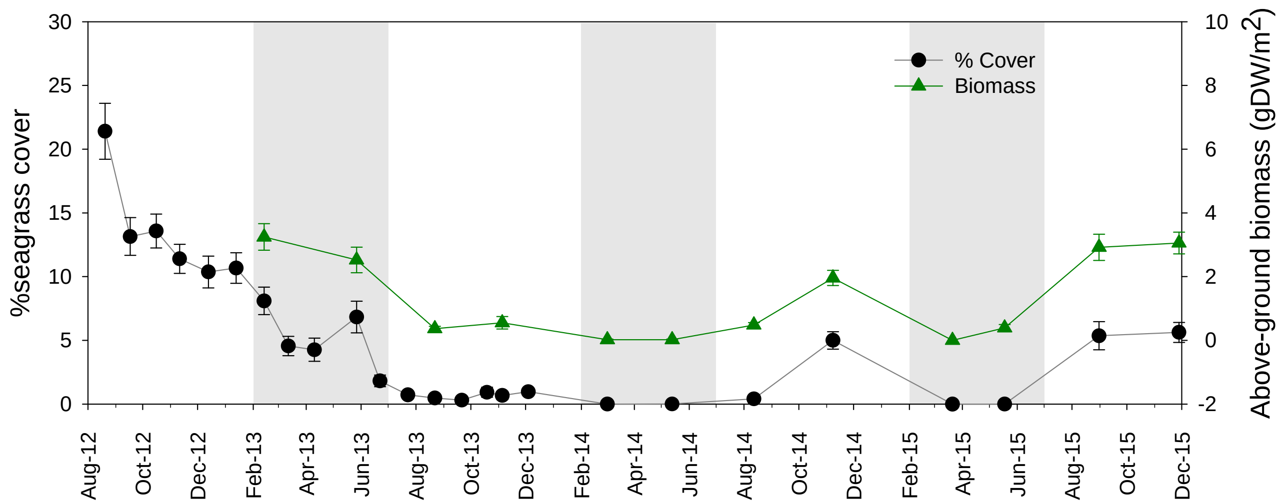


Figure 6: Seagrass abundance (mean ± SE) at Black Swan Island permanent transect site, August 2012 - November 2015.

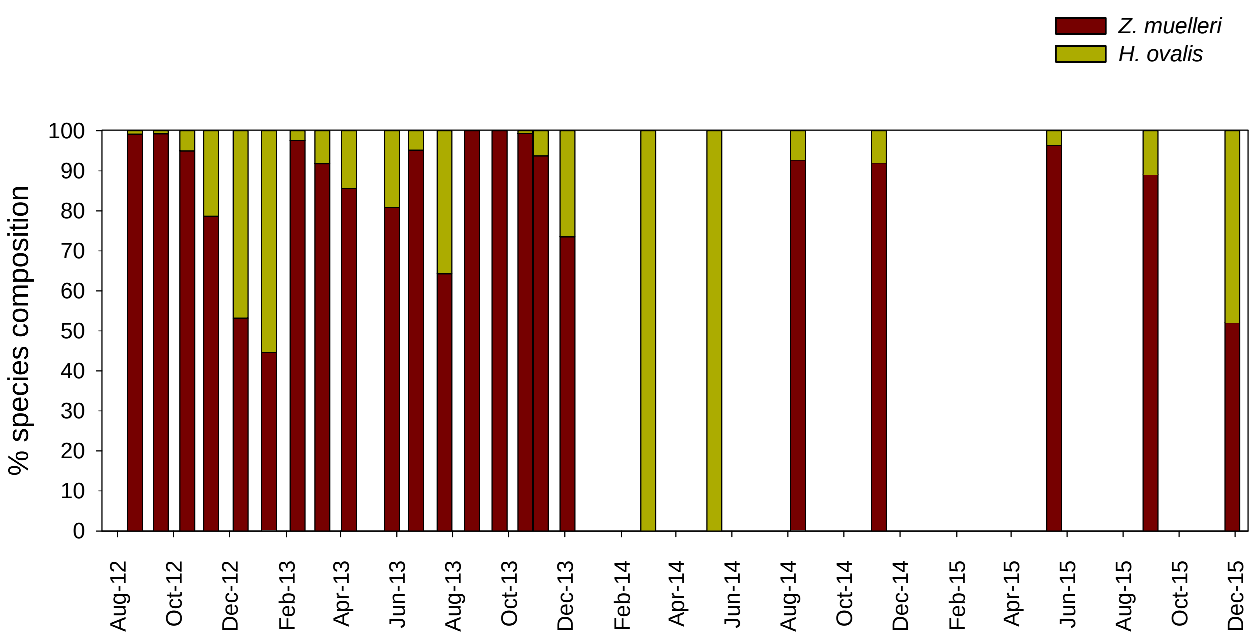


Figure 7: Seagrass species composition at Black Swan Island permanent transect sites, July 2012 - November 2015.

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 Marine Monitoring Program (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 8a). By late 2006 however, seagrasses had already begun to recover and by 2008 had recovered to levels seen when the site was established (Figure 8). Since this time, seagrass percent cover has followed the generally accepted pattern of growth in the Gladstone region with increases over the growing season and sharp declines in percent cover following the wet season (Figure 8a). 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 there have been gradual declines at the site ever since (Figure 8a). A lack of recovery over the 2014 growing season led to the lowest annual peak since 2006 and seasonal declines over the 2015 senescent season brought seagrass cover to the lowest point in almost a decade. Recovery over the 2015 growing season was greater than the previous year; however seagrasses were recovering from such a low base that the peak in November 2015 was the lowest recorded since 2006.

Above-ground biomass was also generally low in 2014 and 2015 compared with the period from 2011 to 2013 (Figure 8a). Annual mapping of the entire bank in 2015 also revealed the lowest average biomass over the course of monitoring (since 2002) (Davies et al. 2016)

Despite the declines in recent years, seagrasses at Pelican Banks North remain the most abundant in terms of percent cover and above-ground biomass in the Gladstone region. The species composition remains almost exclusively *Z. muelleri* with only very minor proportions of *H. ovalis* present at times (Figure 9a).

Pelican Banks South

Seagrass percent cover at Pelican Banks South has followed a similar pattern to Pelican Banks North with significant declines following the 2009/2010 wet season (Figure 8b). Unlike Pelican Banks North, the southern sites failed to recover to initial levels recorded in October 2009 and both seagrass percent cover and above-ground biomass have remained relatively low for the duration of monitoring (Figure 8b). Since late 2011, seagrass percent cover has declined steadily and remains < 1% since May 2014 (Figure 8b). Seagrass biomass followed a similar trend to percent cover and has remained extremely low since the 2014 senescent season. During the growing seasons in 2014 and 2015 there has been minimal recovery at this site.

The species composition at Pelican Banks South from 2009 to 2013 was predominantly *Z. muelleri* with significantly greater proportions of other species present (*Halophila decipiens*, *H. ovalis* and *H. uninervis*) compared with Pelican Banks North (Figure 9b). Coinciding with the sharp declines in biomass and percent cover in 2014 there has been a shift in species composition; a large proportion of *Z. muelleri* has been lost and *H. uninervis* is now the dominant species (Figure 9b).

Facing Island

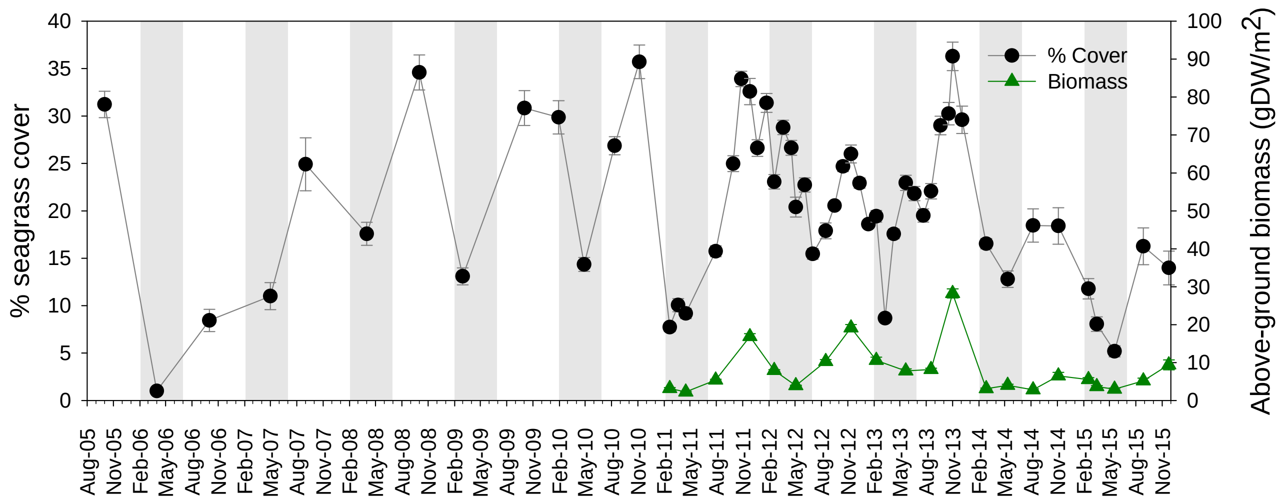
Seagrass abundance at the original Facing Island site has declined significantly from initial levels recorded when the site was established in October 2009 (Figure 10a). By the beginning of 2011, seagrass percent cover had declined by more than 80% and has remained at low levels (<5%) since this time. Peaks in seagrass percent cover have occurred between November and January each year reaching similar levels between 2012 and 2015. Seagrass above-ground biomass fell to low levels in 2013 but has followed an increasing trend since this time. In November 2015 seagrass biomass reached the highest level recorded since 2012 (Figure 10a).

Changes to the shape and distribution of this seagrass meadow identified as part of bi-annual surveys meant that the original site is now 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 10b). Seagrass cover at the new site is consistently slightly higher than the original site but has remained at a lower level since the 2014 decline. Above-ground biomass has been extremely low (< 2 g DW m^{-2}) since February 2013 but has shown an increasing trend since mid-2014.

Species composition at the original site was historically (2009-2013) a mixed species, comprised of a variable mixture of *Z. muelleri*, *H. uninervis* and *Halophila* species. Since 2014 however, the little seagrass remaining at transects is dominated by *H. uninervis* (Figure 11a). Species composition at the new site is comprised of a higher proportion of *Z. muelleri* compared with the original site; however declines in percent cover in 2014 and 2015 have been coupled with a decline in the proportion of *Z. muelleri* present at the site. In May 2015, almost all *Z. muelleri* at the new site was lost but recovered over the growing season and was a similar level as 2014.

a)



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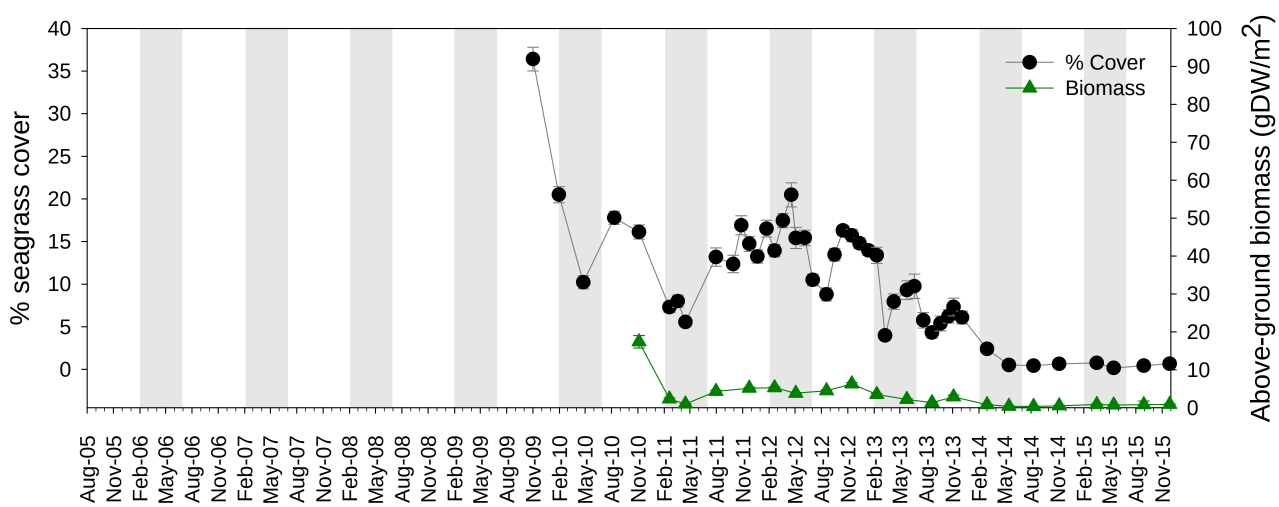


Figure 8: Seagrass abundance at a) Pelican Banks North and b) Pelican Banks South permanent transect sites (pooled), August 2005 – November 2015. Shaded area represents the seagrass senescent season. Data prior to October 2009 were collected for the Reef Rescue Marine Monitoring Program (McKenzie and Unsworth 2009).

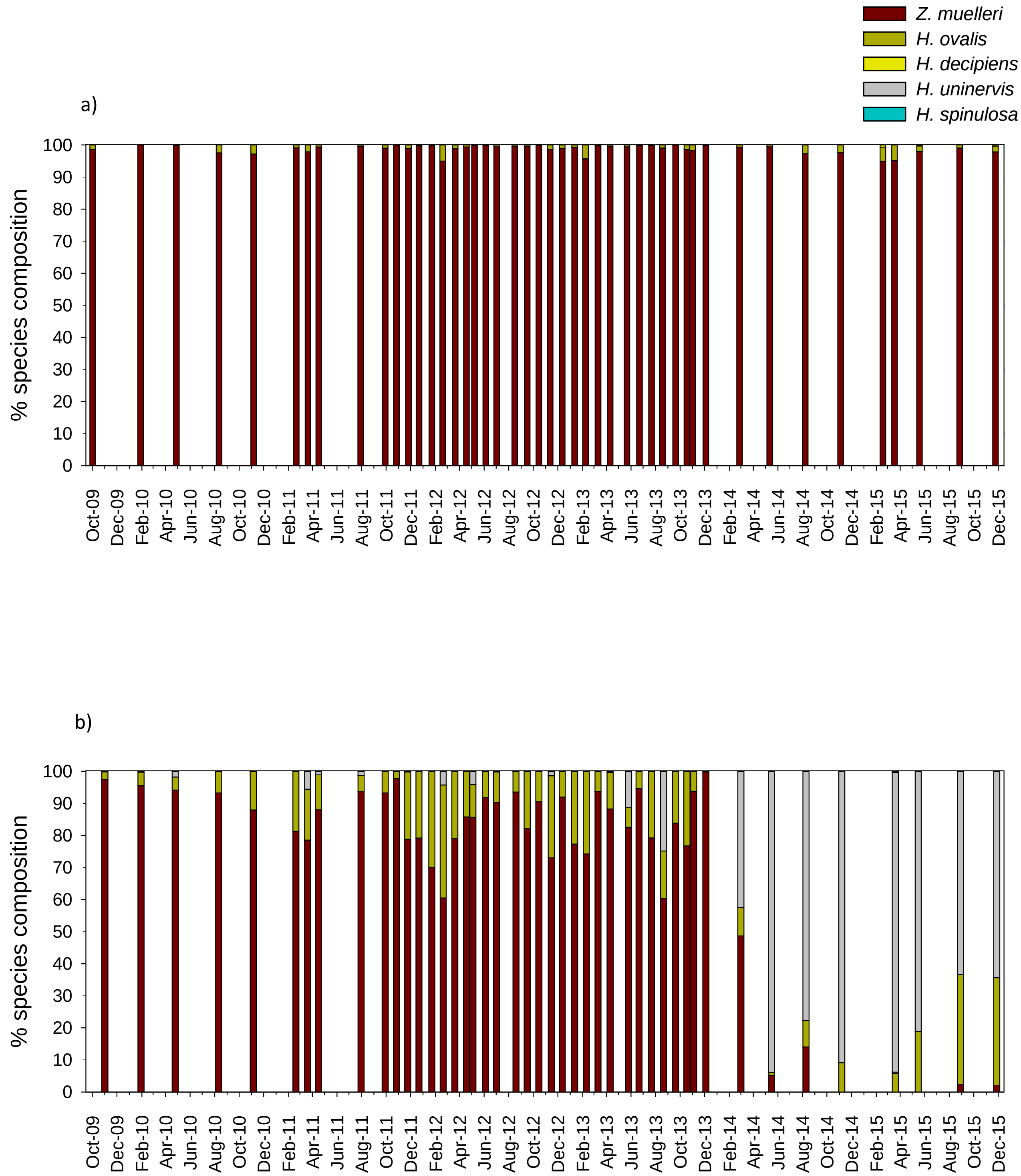
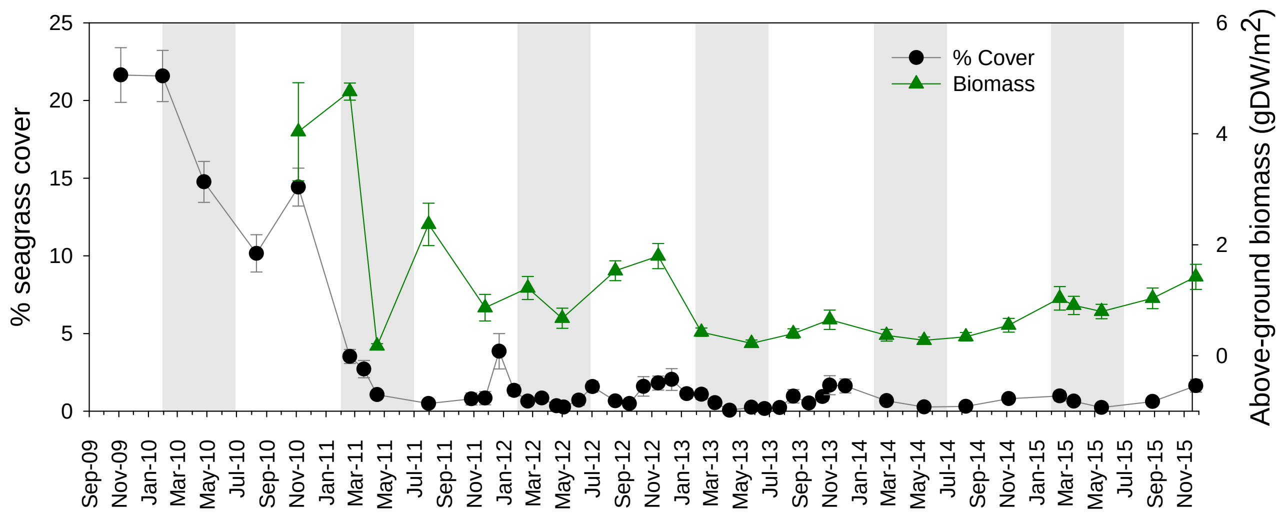


Figure 9: Seagrass species composition at a) Pelican Banks North b) and Pelican Banks South permanent transect sites (pooled), October 2009 - November 2015.

a)



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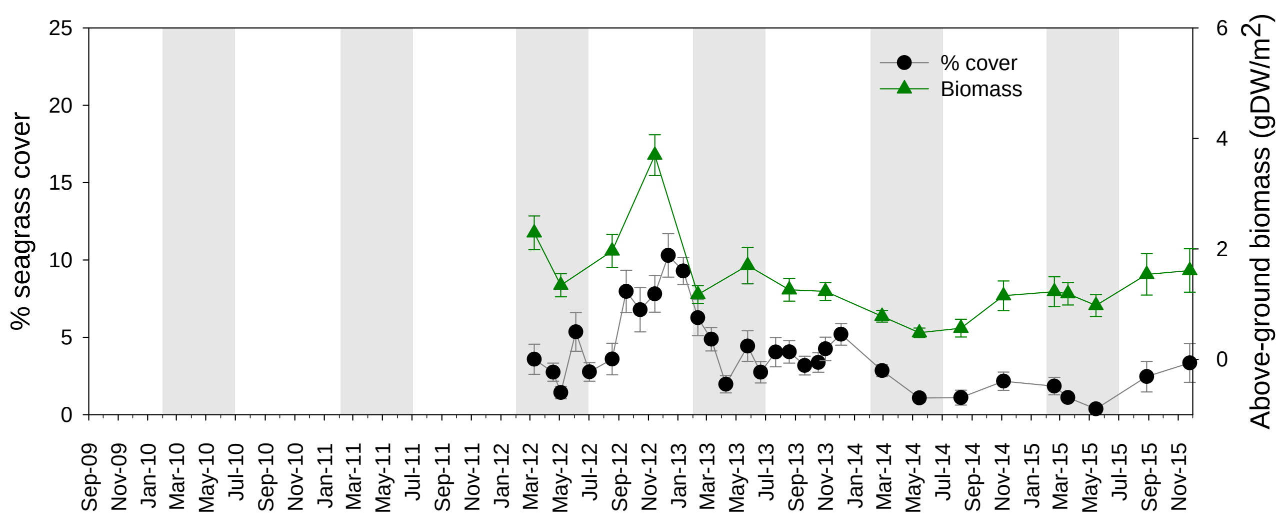


Figure 10: Seagrass abundance at the a) original site and b) new site at Facing Island, November 2009 - November 2015. Shaded area represents the seagrass senescent season.

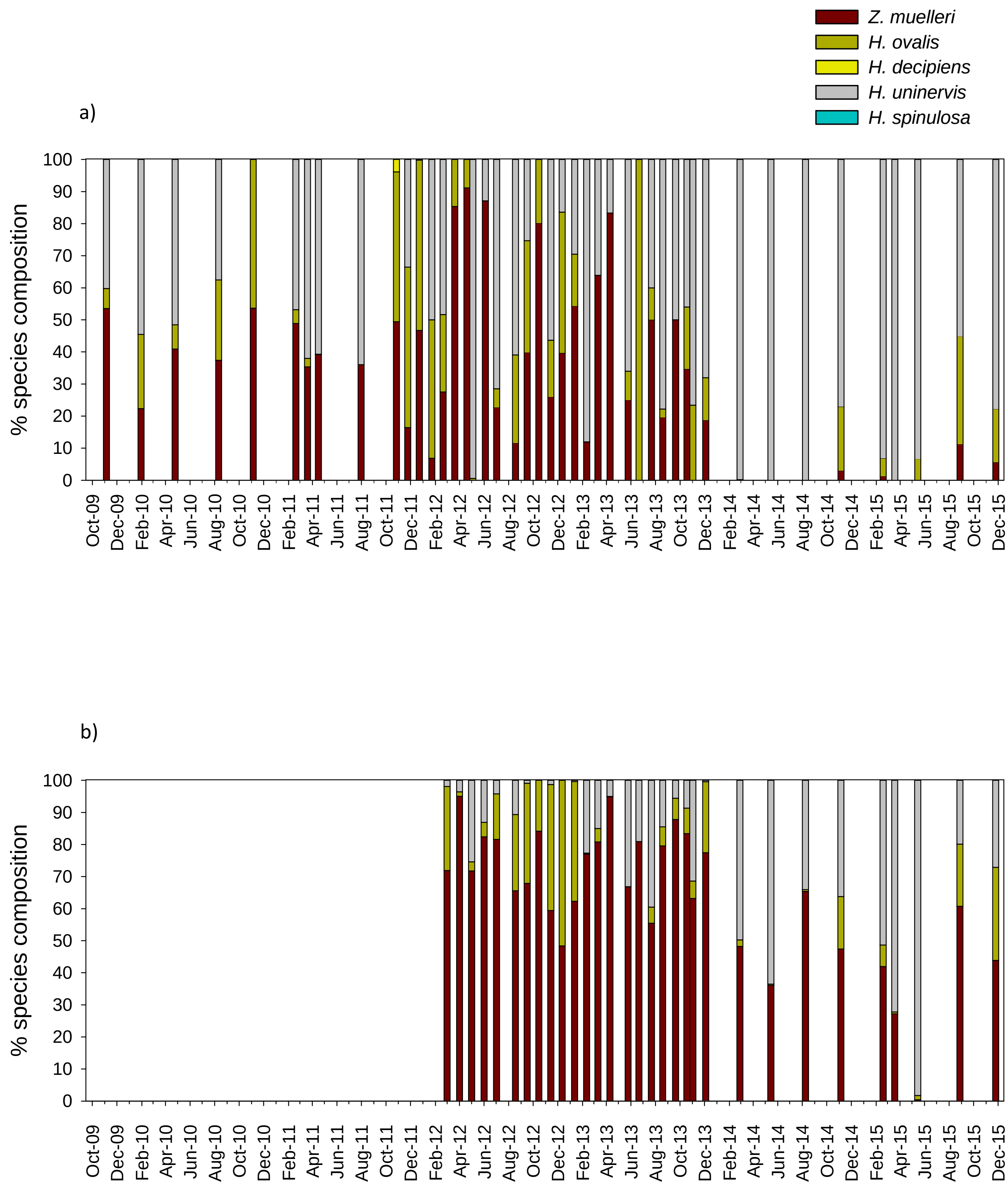


Figure 11: Seagrass species composition at the a) original and b) new permanent transect sites at Facing Island, August 2010 – November 2015.

1.3.3 Out of port reference site

When bi-annual monitoring was established at Rodds Bay in October 2007 as part of the Reef Rescue MMP, seagrass cover was approximately 40% (Figure 12). 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 and 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 this time but seagrass cover and biomass have remained extremely low. In November 2014, seagrass abundance was at the highest level recorded since 2008. In 2015 seagrass abundance increased during the growing season but did not reach the peaks recorded in 2014. *Z. muelleri* has historically been the dominant species at Rodds Bay, especially over the growing season. In August and November 2015 however, *H. ovalis* and a smaller proportion of *H. decipiens* comprised more than 50% of seagrass cover at the site (Figure 13). 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 sites on many occasions, including times when seagrass was absent from the transect sites (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.

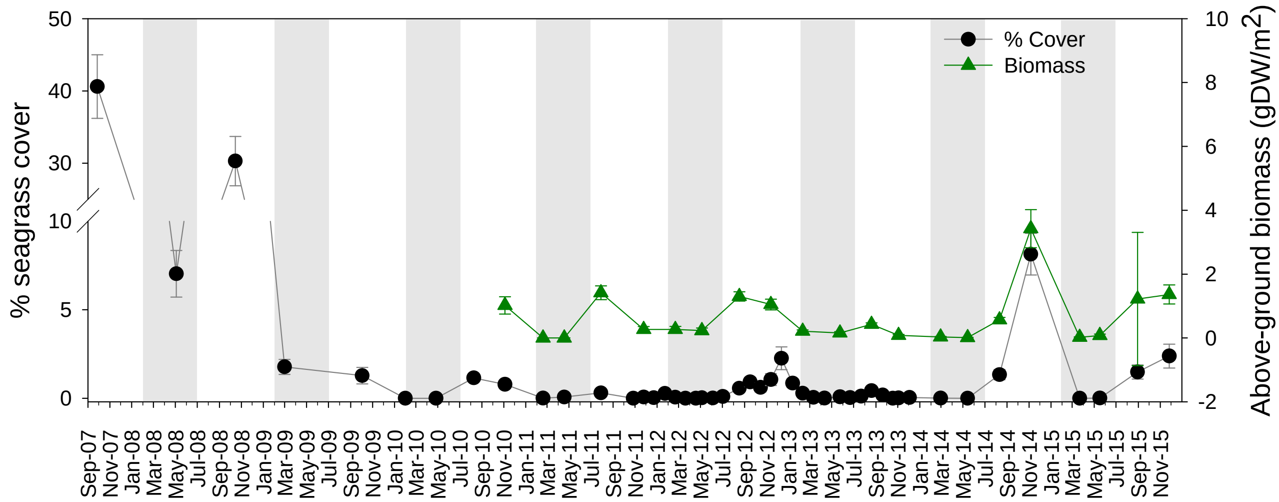


Figure 12: Seagrass abundance at Rodds Bay, October 2007 – November 2015. Shaded area represents the seagrass senescent season. Data prior to October 2009 were collected for the Reef Rescue Marine Monitoring Program (Reef Water Quality Protection Plan 2011).

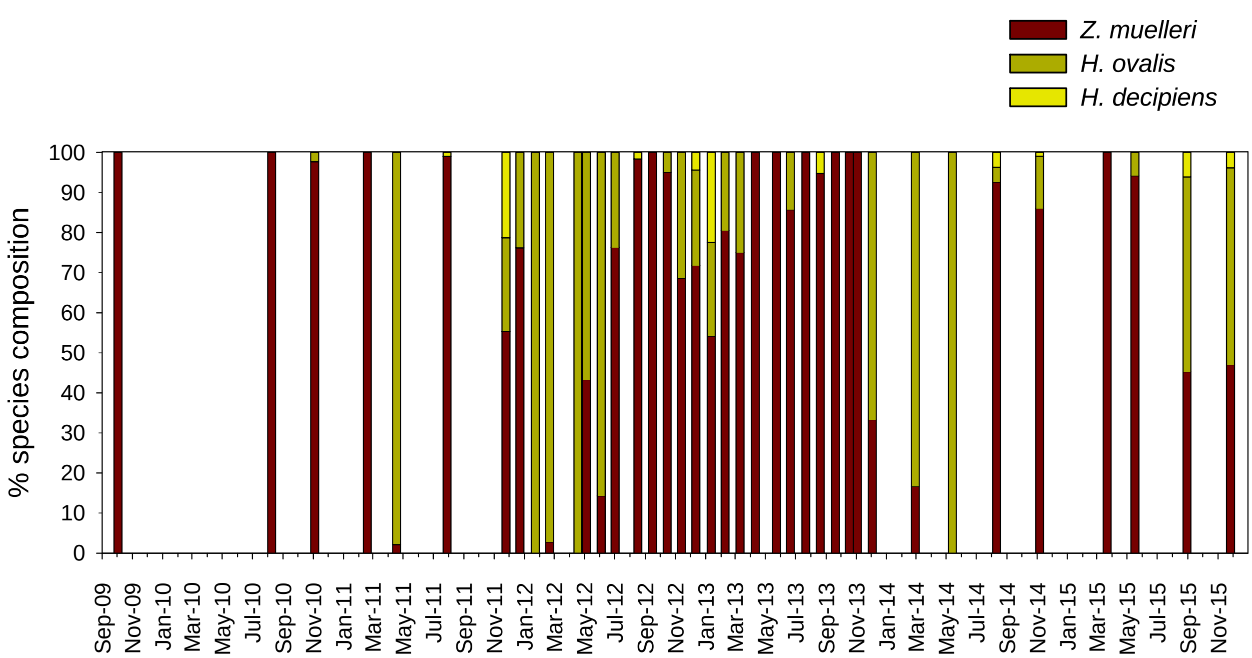


Figure 13: Seagrass species composition at Rodds Bay, October 2009 – November 2015.

1.4 Discussion

Monitoring at permanent transects revealed that seagrasses in the Western Basin and Rodds Bay underwent distinct seasonal changes in abundance and community composition. Seasonality observed throughout this study was consistent with the definitions of growing and senescent seasons in Chartrand et al. (2012). At some outer harbour meadows such as Pelican Banks where seagrasses are abundant, these seasonal changes are observed as increases and decreases in seagrass cover and the size of seagrass blades throughout the meadow. At some inner harbour sites, such as Fisherman's Landing, where seagrass is only ever light and patchy, seasonal changes are expressed in the appearance and disappearance of patches of seagrass and the growth of patch size and blade size over the growing season. Inner harbour sites also underwent seasonal trends in species composition, which were not as pronounced at the more stable outer harbour sites such as Pelican Banks.

As well as demonstrating seasonal trends within years, the monitoring program has highlighted significant inter-annual variability in seagrass distribution, abundance and species composition. At Wiggins Island, Black Swan, Facing Island, and Pelican Banks South there have been major declines in seagrass abundance since sites were established (Davies et al. 2015; Bryant et al. 2014a). When this relatively short time series is considered by itself, the trends are alarming, however a longer time series is necessary in order to place these changes in a historical context. At Pelican Banks North, where there are ten years of data available, monitoring revealed similar declines in 2006, with full recovery of seagrass cover over the following two years. The value in this extended time series 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.

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 impact dredging may have had on seagrass condition at the permanent transect sites. As part of the WBDDP management program, thresholds and triggers have been established that are directly related to the 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 were receiving 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) (see section 4 of this report).

Longer term studies, such as this monitoring program, which examine inter-annual variability in seagrass, remind us that seagrass communities are not static entities and that seasonal changes are often difficult to predict (Erftemeijer and Lewis 2006). Seagrass patches come and go as we have seen at Fisherman's Landing, and also change in position and density (Hemminga and Duarte 2000). 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. In Gladstone, the major driver of change in seagrass abundance appears to be extreme river flow events and the associated increase in nutrient loads and reductions in available irradiance (McCormack et al. 2013a).

The historical data for the program indicates the prospects for recovery at meadows where major declines have been reported are promising. 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 5 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 Gladstone seagrass meadows are discussed in detail in section 3 of this report.

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 indicate seagrass to be 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. 2012a). The median seagrass tissue ratio of C:P is approximately 500 (Atkinson and Smith 1983), therefore deviation from this value is likely to be indicative of 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 and a poor nutrient 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. In the laboratory, leaves were separated from below-ground material and scraped clean of epiphytes. Samples were dried at 60°C to a constant weight 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).

2.3.2 Outer Harbour

In 2015, the C:N ratio peaked in November 2015 and remained below 20 for all other sampling events, similar to the previous two years (Figure 14a). At 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). There has been insufficient *Z. muelleri* at Pelican Banks South and the original Facing Island site over the past two years to assess tissue nutrient ratios. C:N ratios for *H. uninervis* at both Pelican Banks South and Facing Island*,* remained below 20 during 2015, peaking at similar levels as previous years (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 2015, indicating a nutrient replete environment, but fell below 25 in May and August indicating possible Nitrogen limitation (Figure 14c). At Facing Island, N:P ratios for *Z. muelleri* indicate a nutrient replete environment in February, May and August but possible nitrogen limitation in November (Figure 14c). N:P ratios for *H. uninervis* at Pelican Banks South and Facing Island indicate a nutrient replete environment for the majority of 2015 with the exception of Pelican Banks South which showed possible signs of Phosphorus limitation in November (Figure 14d).

Carbon to phosphorous (C:P) ratios for *Z. muelleri* at Pelican Banks North fell below 500 during May and August 2015, indicating nutrient enrichment at the beginning of the growing season, 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* were relatively stable throughout 2015. C:P ratios for *H. uninervis* at Pelican Banks South and Facing Island also suggest a nutrient rich environment in August and nutrient depletion in November 2015 (Figure 14f).

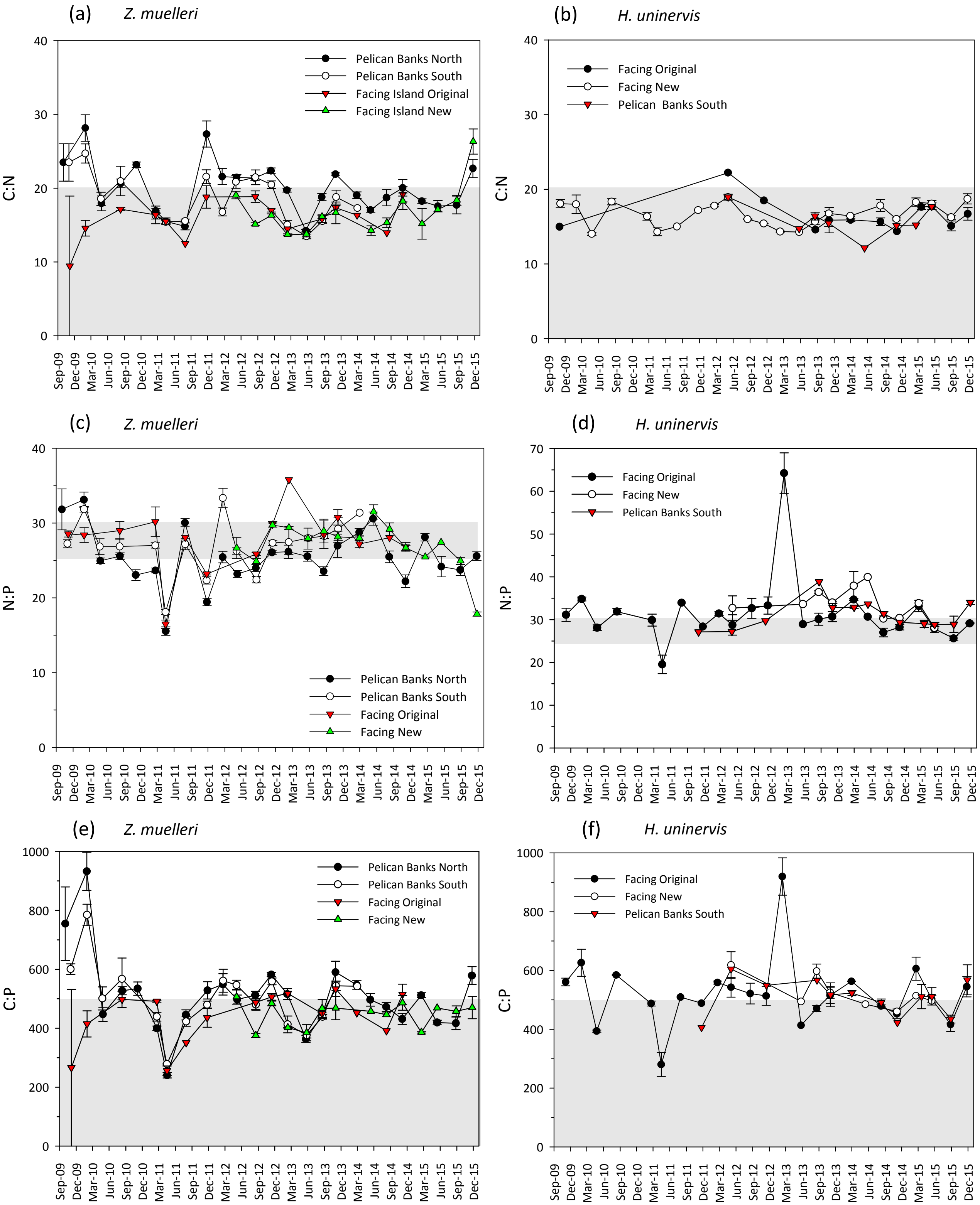


Figure 14: Elemental (atomic) ratios of seagrass leaf tissue for *Z. muelleri* and *H. uninervis* collected at permanent transect locations in the outer harbour (mean ± Standard Error). Shaded areas indicate low light (C:N < 20), nutrient replete (N:P 25-30) and nutrient enriched (C:P < 500) environments.

2.3.3 Out of port reference site

The loss of above-ground biomass at Rodds Bay during the senescent season limited sample collection in 2015 to August and November only. 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).

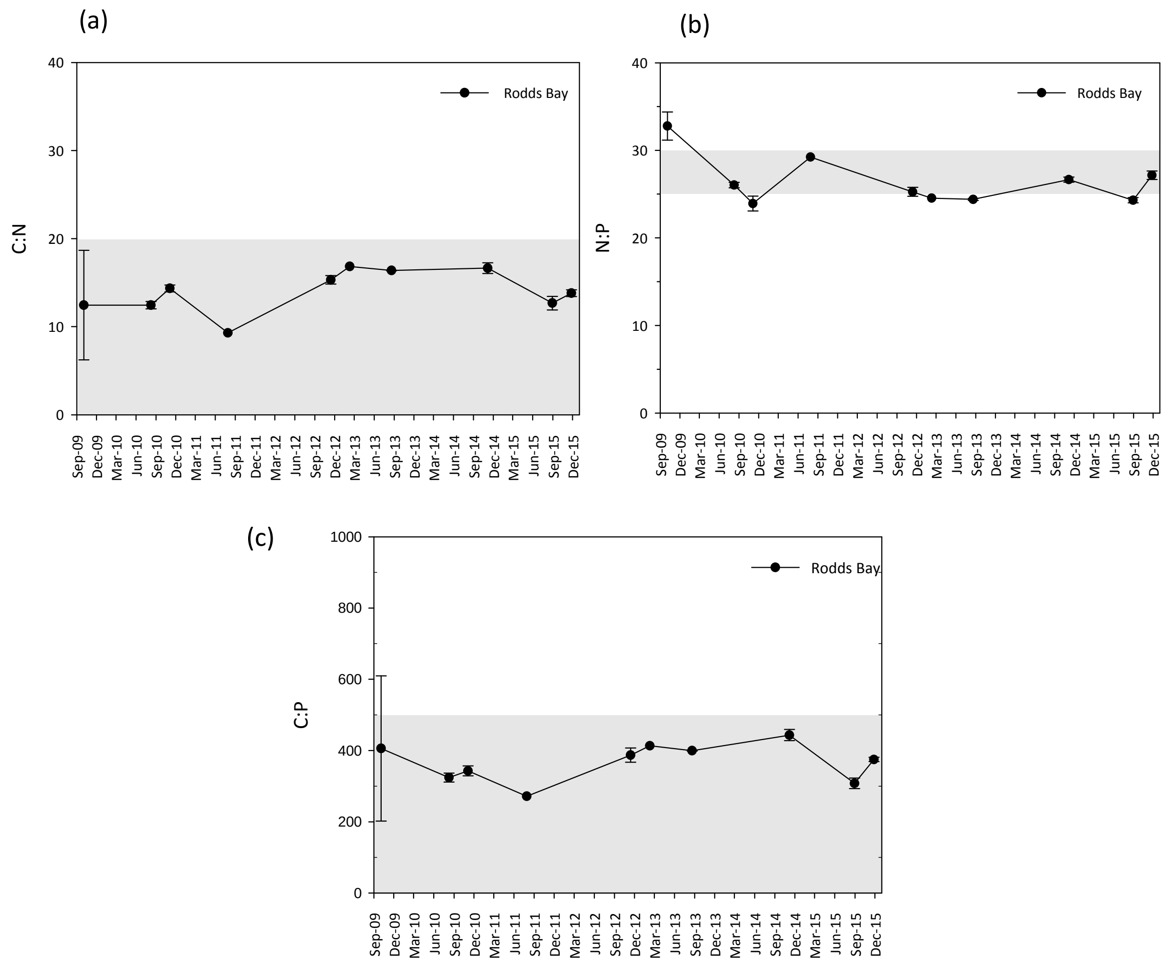


Figure 15: Elemental (atomic) ratios of seagrass leaf tissue for *Z. muelleri* collected at permanent transect locations at the out of port reference sites in Rodds Bay (mean ± Standard Error). Shaded areas indicate low light (C:N < 20), nutrient replete (N:P 25-30) and nutrient enriched (C:P < 500)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

2.4 Discussion

In the outer harbour region, C:N ratios for *Z. muelleri* at Pelican Banks have fluctuated seasonally with seagrasses generally existing in lower light conditions following the wet season and higher light conditions in the peak growing season. A similar trend has occurred at Facing Island although *Z. muelleri* at these sites appear to have existed in lower light conditions over the majority of the monitoring program. These results correspond well with light and seagrass abundance data collected at the respective sites (see section 4). Marked declines in light (below or close to locally determined thresholds) and seagrass percent cover coincided with lower C:N ratios following extreme flooding and peak river flow events in early 2011, 2013, 2014 and 2015. Less extreme declines in light and seagrass percent cover in 2012 coincide with higher C:N ratios following drier conditions.

The peak in *Z. muelleri* C:N ratios at Pelican Banks North and Facing Island in November 2015 indicates a return to more favourable light conditions at those sites. The prevailing lower light conditions at Pelican Banks South correspond with declines in seagrass abundance and the loss of *Z. muelleri* since early 2014. *In situ* data collected at Pelican Banks South confirms that light fell below the minimum requirement to sustain *Z. muelleri* growth in both February 2013 and 2014. Since this time there has been insufficient *Z. muelleri* at the site to perform tissue nutrient analysis. For the majority of the monitoring period however, light has been sustained above minimum requirements and seagrass declines are likely attributed to a combination of factors including propagule limitation (see section 3 below).

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 is advantageous for the early detection of changes in nutrient regimes for environmentally sensitive seagrasses. The capacity of seagrass meadows to naturally recover following disturbance will involve the interaction between a range of factors that is likely to include light availability, nutrient loads and the availability of recruits, to form the foundation of new populations (McKenzie et al. 2012b).

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 (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) is no longer examined as part of the program; results of monitoring from 2009 to 2014 are summarised in (Bryant et al. 2014a).

3.2 Methods

3.2.1 Reproductive effort

During quarterly sampling, 15 randomly placed sediment cores (100mm diameter and 100mm depth) were collected from an area adjacent to monitoring sites (approximately 25 m to the side of the site) 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.

3.2.2 Seed banks

Seagrass seed banks were originally (November 2009 to February 2011) assessed using standard Seagrass-Watch methodology. Sediment cores, measuring 50mm in diameter and 100mm 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 1mm 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 710µm to 1mm 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 1mm 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 (Pelican Banks North (GH1), Wiggins Island (WW1) and Rodds Bay (RD1)) to focus further processing efforts. 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. 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.

In 2014, GPC through the Ecosystem Research and Monitoring Program (ERMP) commissioned additional work to address knowledge gaps in the density and viability of *Z. muelleri* sediment seed banks at current monitoring sites (Pelican Banks North (GH1), Wiggins Island (WW1) and Rodds Bay (RD1)). At each sampling location, 18 sediment cores were collected at 10m intervals and sectioned into depth categories (0-20mm, 20-50mm and 50-100mm). The 710µm to 1mm fraction was inspected for *Z. muelleri* seeds (Figure 16). For samples collected during the pre-growing season (May) and post-growing season (February), any intact seeds detected were also tested for viability (Figure 16). See Jarvis et al. (2015) for a full description of the methods used. A summary of results of the first year of the study are presented here. For further detail of methods including statistical analysis see (Jarvis et al. 2015).

Figure 16: Examples of stained viable and non-viable Z. muelleri seeds using tetrazolium chloride.

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 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. 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 found at Pelican Banks north in November 2015 but were once again absent at Pelican Banks South (Figure 18A and B). Fruits and flowers (female and male) were also detected at Facing Island (Figure 18C). The density of male flowers was the highest recorded at any site over the course of the program (713.01 ± 270.8 m^{-2}).

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 (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}).

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.

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

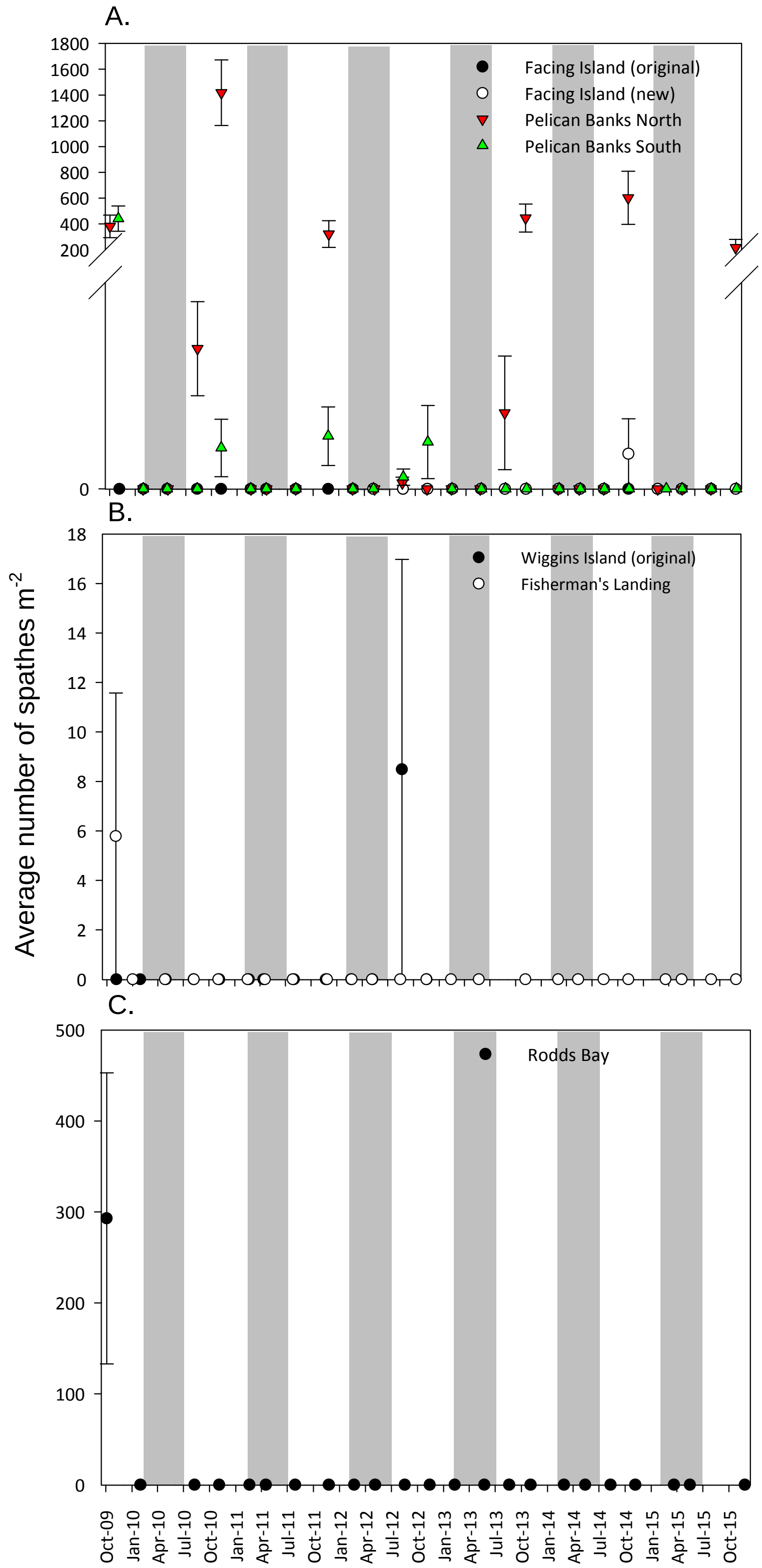


Figure 17: Density of *Z. muelleri* spathes (mean ± Standard Error) at permanent transect locations in the outer harbour (A), inner harbour (B) and Rodds Bay (C) November 2009 to November 2015. Shaded area represents the seagrass senescent season.

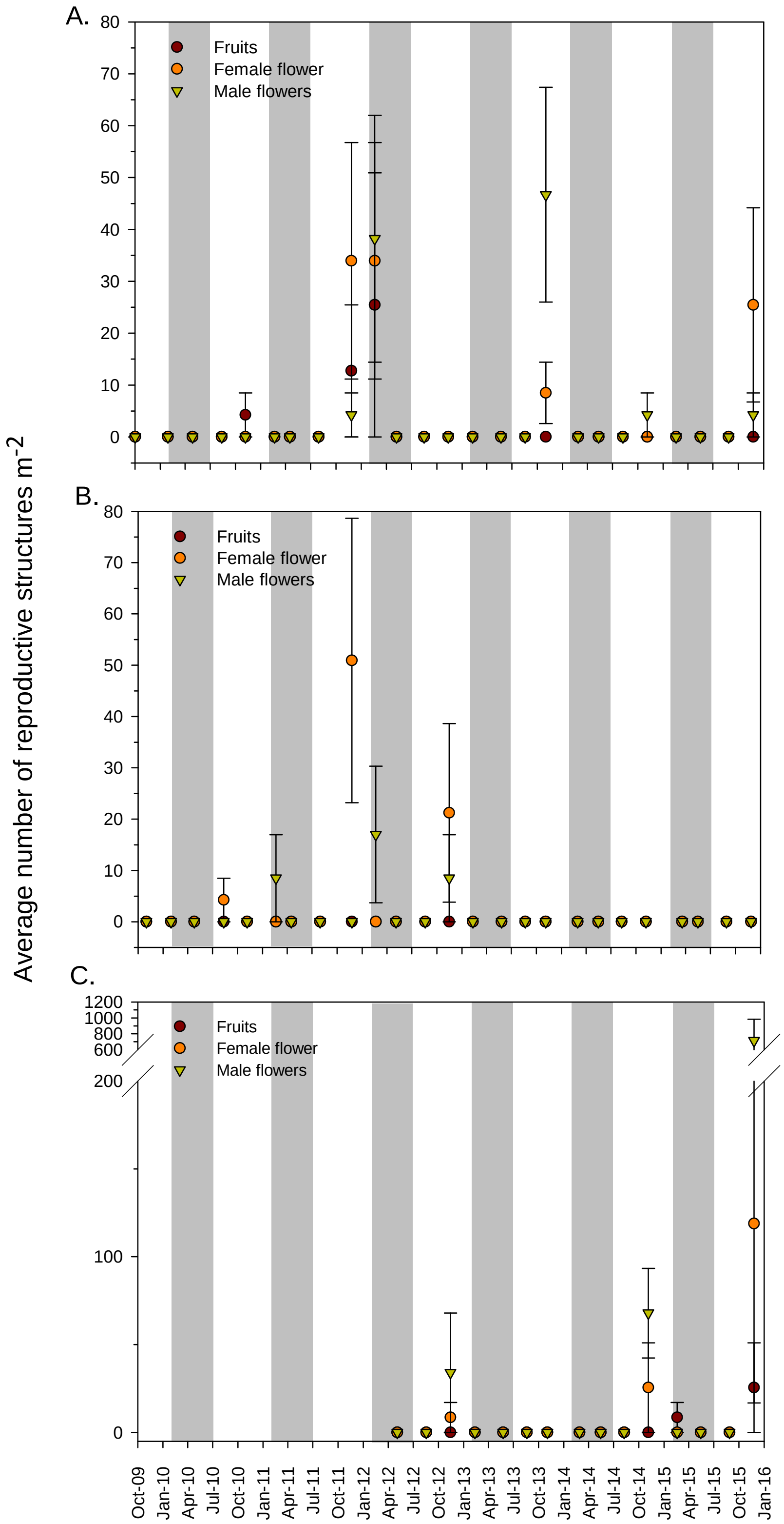


Figure 18: Density of *H. ovalis* fruits and flowers (mean ± Standard Error) at permanent transect locations in the outer harbour (A – Pelican Banks North), (B – Pelican Banks South) and (C – Facing Island) November 2009 to November 2015. Shaded area represents the seagrass senescent season.

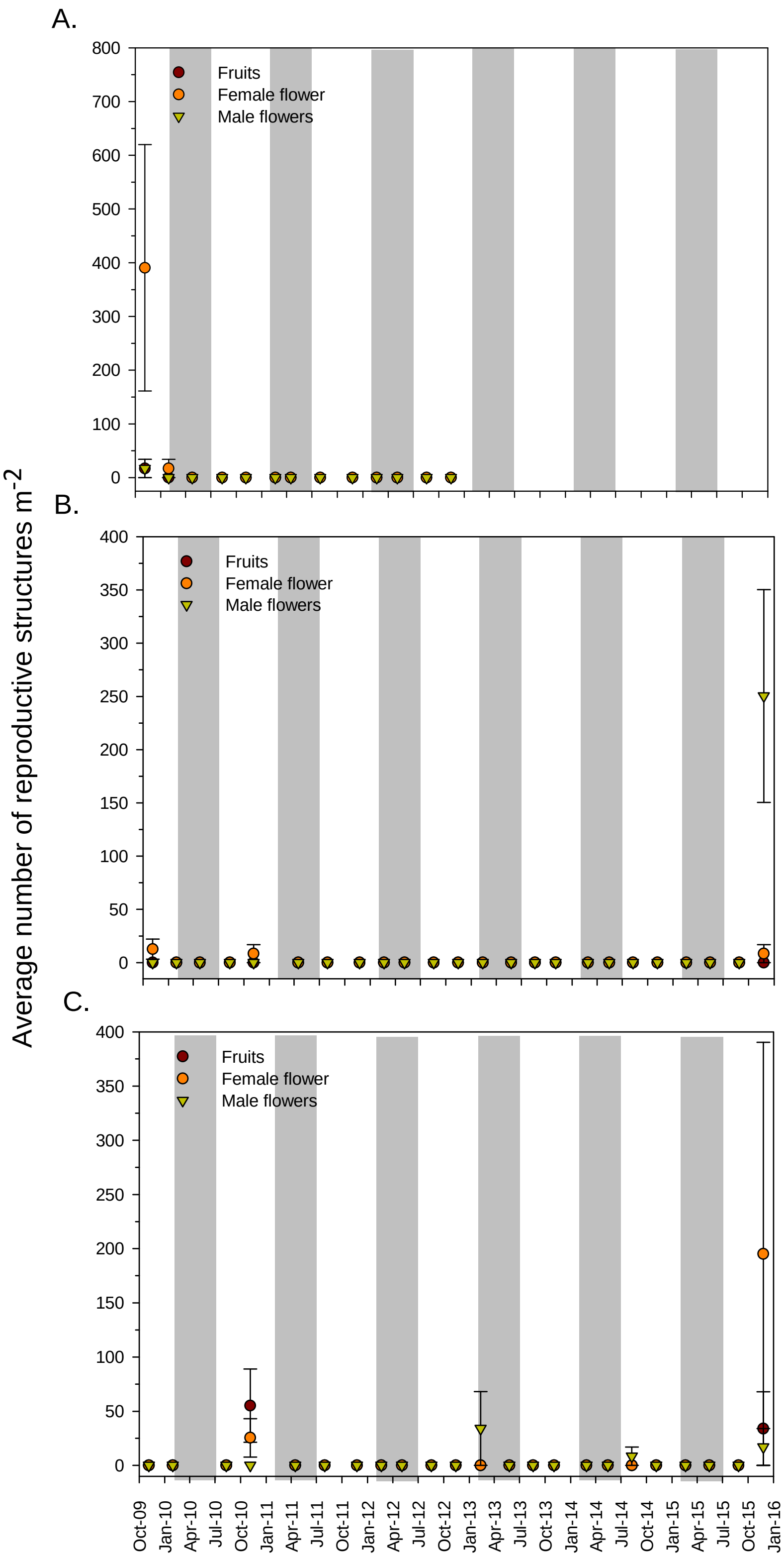


Figure 19: Density of *H. ovalis* fruits and flowers (mean ± Standard Error) at permanent transect locations in the inner harbour (A – Wiggins Island Original), (B – Fisherman's Landing) and Rodds Bay (C) November 2009 to November 2015. Shaded area represents the seagrass senescent season.

3.3.2 Seed banks

In 2015 *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 and 21). Mean seed density was highest during the senescent season (February and May) and lower during the growing season (August and November). November samples produced the lowest seed densities at all sites (Figure 21).

At Pelican Banks North (GH1), average seed density has varied from a minimum of 113.23 ± 87.93 m^{-2} in August 2012 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). In 2015 however, there was little change in average seed densities until November, when seed density declined slightly in line with seasonal increases in percent cover (Figure 21). The proportion of seeds found at the deepest zone (50 to 100mm) has gradually increased since November 2013 when seed burial depth was first investigated (Figure 22A).

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 21). The proportion of seeds found at each depth was within the range detected in previous years. The 50-100mm depth zone continues to contain the highest seed densities at the site (Figure 22B).

At Wiggins Island (WW2), as with other sites, average seed density declined across the sampling year with the steepest decline occurring from May to August and the lowest seed density occurring in November (Figure 21). Seed densities in August and November 2015 were at a similar low level as found at the original site in 2012 (Figure 20.) This is markedly different from 2014, when seed density peaked in November (Figure 20). As with other sites, the vast majority of seeds were found in the 50-100mm depth zone (Figure 22C).

Seed bank viability decreased at all sites between February and May 2015 (Table 1). Pelican Banks had the greatest decrease in viability from 54 ± 19% to 0 %. Rodds Bay and Wiggins Island also decreased but to a lesser and non-significant extent. As with total seed bank density, the majority of viable seeds were found at sediment depths > 50mm at all sites (Figure 23) (Jarvis et al. 2015).

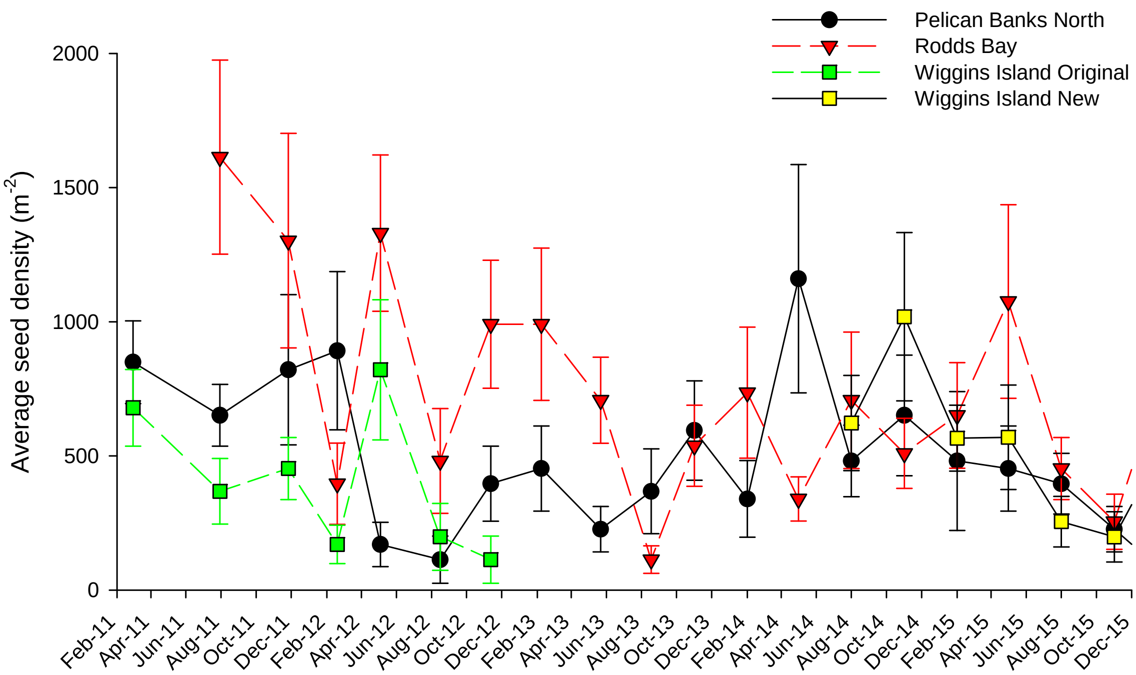


Figure 20: Mean number (± SE) of *Z. muelleri* seeds per square metre at Pelican Banks North, Rodds Bay and Wiggins Island.

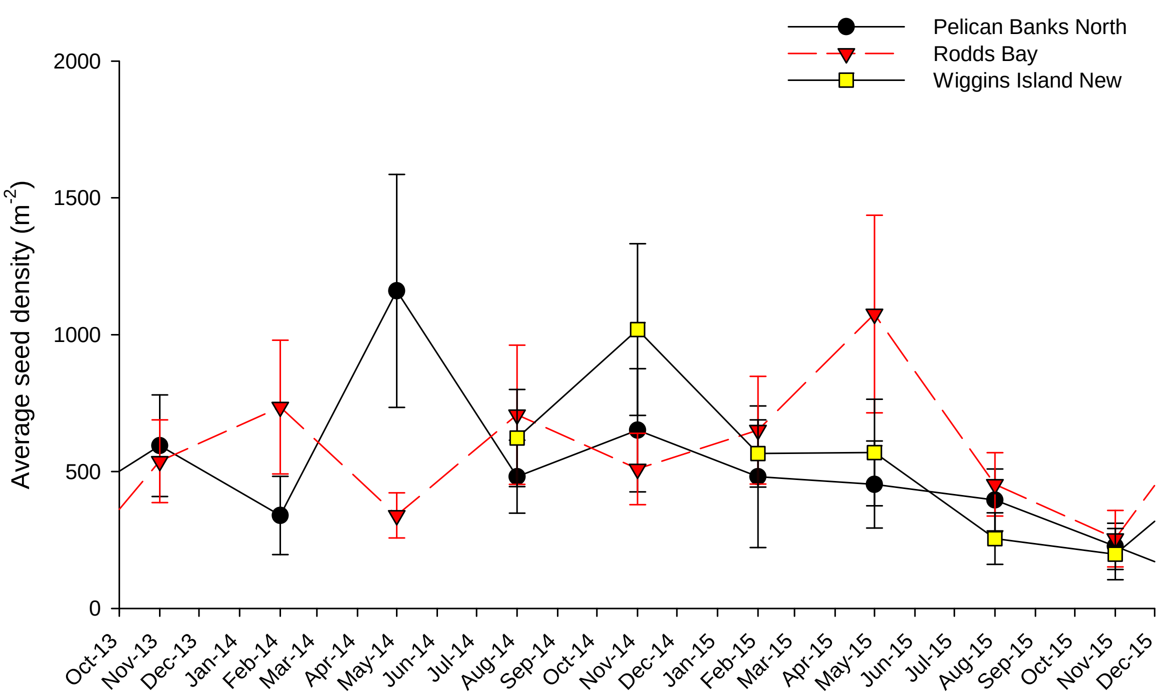


Figure 21: Mean number (± SE) of *Z. muelleri* seeds per square metre at Pelican Banks North, Rodds Bay and Wiggins Island.

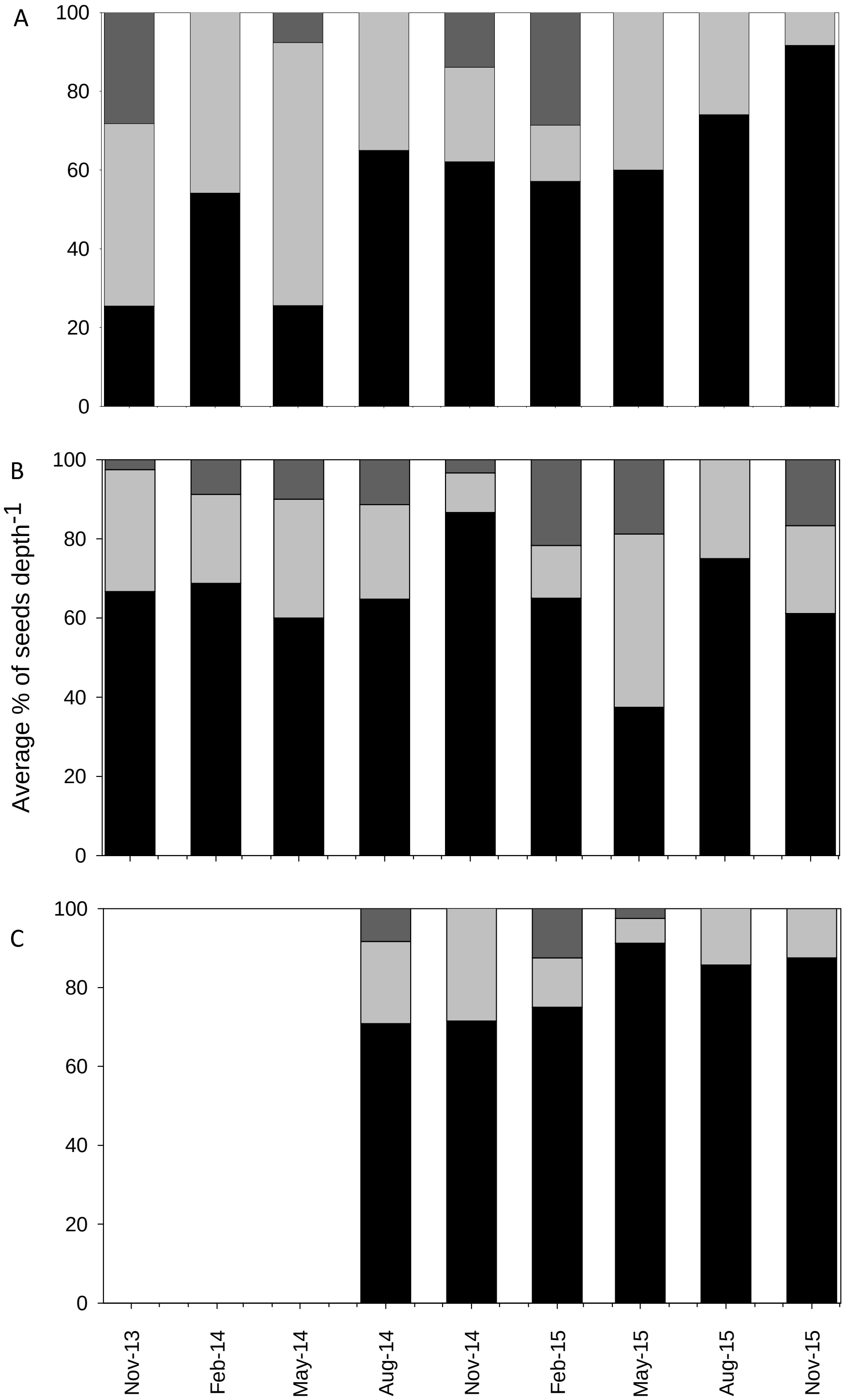


Figure 22: Average proportion of total (non-viable + viable) *Z. muelleri* seeds at 0-20mm (dark grey), 20-50mm (light grey) and 50-100mm (black) depths for (A) Pelican Banks North, (B) Rodds Bay and (C) Wiggins Island.

Table 1. Seed bank density and percentage of viable seeds for *Z. muelleri* at selected monitoring sites in 2015. Values are reported as mean ± S.E. Adapted from Jarvis et al. (2015).

Site	Date	Total Seeds m^{-2}	% Viable
Pelican Banks	Feb-15	481 ± 259	54 ± 19
	May-15	453 ± 159	0 ± 0
Rodds Bay	Feb-15	651 ± 197	35 ± 11
	May-15	1075 ± 361	27 ± 12
Wiggins Island	Feb-15	566 ± 123	21 ± 11
	May-15	569 ± 195	19 ± 12

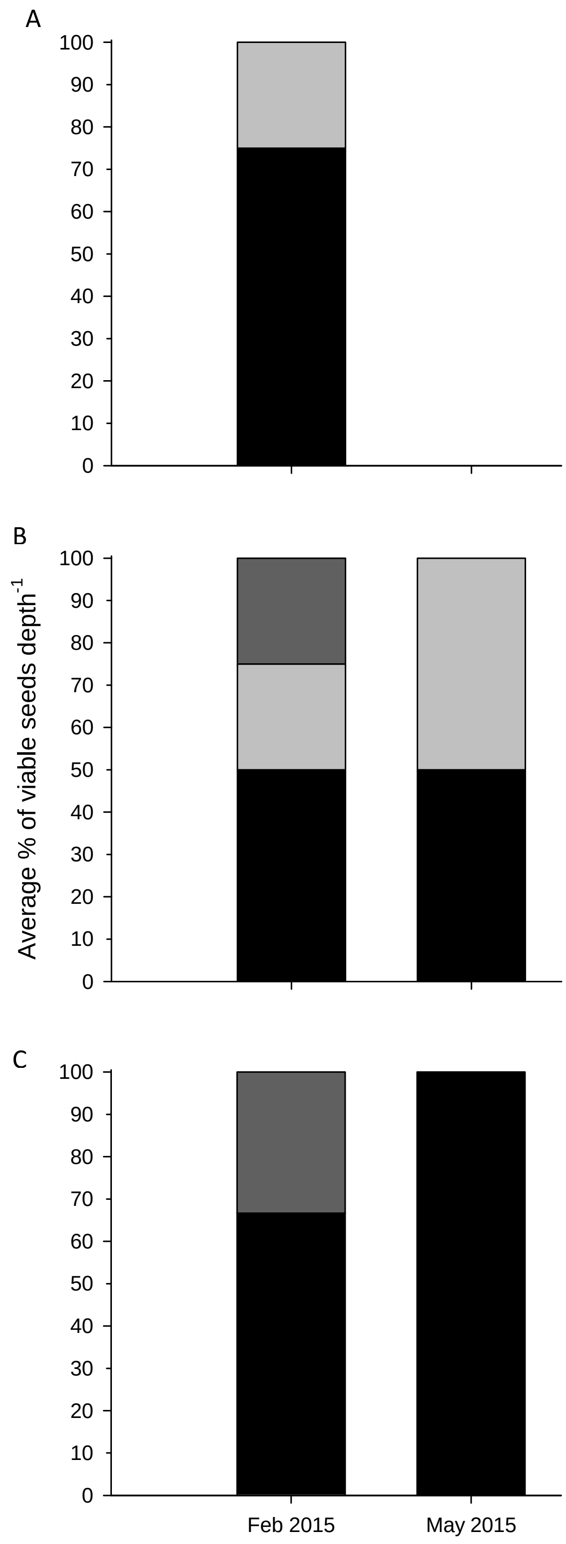


Figure 23. Mean proportion of viable *Z. muelleri* seeds at 0-20mm (dark grey), 20-50mm (light grey) and 50-100mm (black) for (A) Pelicans Bank, (B) Rodds Bay and (C) Wiggins Island. Adapted from Jarvis et al. (2015).

3.4 Discussion

Gladstone 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).

The *Z. muelleri* seed bank detected at Pelican Banks North was also denser than reported for the same species in Moreton Bay (Conacher et al. 1994b). 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 at Pelican Banks North were similar at both the beginning (February 2015) and near the end (May 2015) of the senescent season; however the proportion of viable seeds declined from ~ 50% to 0% over the same period. Loss of viability in seagrass seed banks is not well understood (Jarvis et al. 2015), although decreases in viability have been attributed to germination, predation and mortality (Sumoski and Orth 2012; Fishman and Orth 1996). The loss of viability at Pelican Banks is unlikely to be attributed to germination given the timing of the study (i.e. during the senescent period) or predation given the stability of seed densities over the same period.

Burial depth of *Z. muelleri* seeds at Pelican Banks North has increased over the duration of the study (since November 2013) – the proportion of seeds buried at >50mm increased from ~30% to almost 90% over a two year period. Seeds that are buried too deep are removed from the viable seed bank, incapable of germinating and producing seedlings once buried beyond a species-specific depth (Jarvis et al. 2015; Granger et al. 2000). While the effects of burial depth on the viability and germination of *Z. muelleri* are unknown, germination of the similar species *Z. marina* is limited when seeds are buried below 50mm (Jarvis and Moore 2015). The trend in burial depth at Pelican Banks North coincides with gradual declines in seagrass abundance over the same period. Although we cannot attribute these declines to seed bank function alone, we suggest that burial depth may be producing a bottleneck in the germination of viable seeds at Pelican Banks North and other Port Curtis meadows (Jarvis et al. 2015). Despite these concerns, the relatively high density of plants at Pelican Banks North capable of both sexual and asexual reproduction, as well as the presence of a substantial seed bank provides several mechanisms for recovery and the meadow is likely more resilient than other meadows in the Western Basin.

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).

Surveys undertaken in Rodds Bay in May and February 2015 found little change in the viable proportion of the seed bank despite the increase in total seed numbers over the senescent period. The presence of a viable seed bank and adequate light environment at the site suggests that other factors are driving the lack of germination or seedling success for *Z. muelleri*. While further investigation is required, as with Pelican Banks, it is possible that seed burial depth may be one factor inhibiting any substantial recovery (Jarvis et al. 2015).

At inner harbour sites, sampling for reproductive structures in 2015 was restricted to Fisherman’s Landing because of access issues at the Wiggins Island site. 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 Gladstone seed banks to date 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 (Bryant et al. In prep; Jarvis et al. 2015). Additional studies would enhance the understanding of seagrass resilience in Port Curtis including a) manipulative field studies examining the ‘persistence’ of the seed bank (viability of *Z. muelleri* seeds over time) and b) an examination of seed dispersal and movement within the region to assess likely sink and source meadows.

4 CLIMATE & LIGHT DRIVERS OF 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; Erftemeijer and Lewis III 2006). 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; 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) recently determined that *Z. muelleri* in Port Curtis requires between at least 5 mol quanta 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 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).

4.2 Methods

4.2.1 Environmental parameter monitoring

Local climate conditions

Environmental data on rainfall (mm), wind speed (km/h) and river discharge (Mega Litres) data 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 were provided by Maritime Safety Queensland. The number of daytime hours that seagrasses at each of the sites were exposed was calculated by summing the total daylight hours that the tidal height was below known exposure datum depths.

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 2013).

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. 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. 2016) for a full description of the methods, data processing and QAQC procedures.

4.3 Results

4.3.1 Local climate conditions

The local climate in Gladstone over the course of monitoring has been characterised by above average rainfall, punctuated by severe flood events. The period from 2008 to 2013 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 is the highest on record with discharges of over 600 000 Mega litres of water causing severe flooding in the Gladstone region (Figures 24 and 25).

In February 2015, TC Marcia crossed the coast just north of Gladstone 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). In 2015 light levels at both inner harbour sites were similar to the previous year and 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 and January 2015 (Figure 27). Light levels appear to have shifted to a lower level in 2014/2015 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 is located at a slightly deeper section of the meadow.

Temperature at the seagrass canopy has 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 2015 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 and 2014/15 (Figure 27).

Outer harbour

At Pelican Banks North, the rolling average total daily light has remained above 6 mol m^{-2} day^{-2} during the growing season since 2011 (Figure 28A). Deviations below or close to this threshold generally correspond with peak river flow events and are closely followed by declines in seagrass percent cover (Figure 28A). In both 2014 and 2015, 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 has remained above 6 mol m^{-2} day^{-2} 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 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 m^{-2} day^{-2}. 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, in both 2014 and 2015, 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 m^{-2} day^{-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 m^{-2} day^{-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 2015 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 the 2014/15 growing season (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 m^{-2} day^{-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 2015 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 and again in 2014/15 and 2015/16 (Figure 29B).

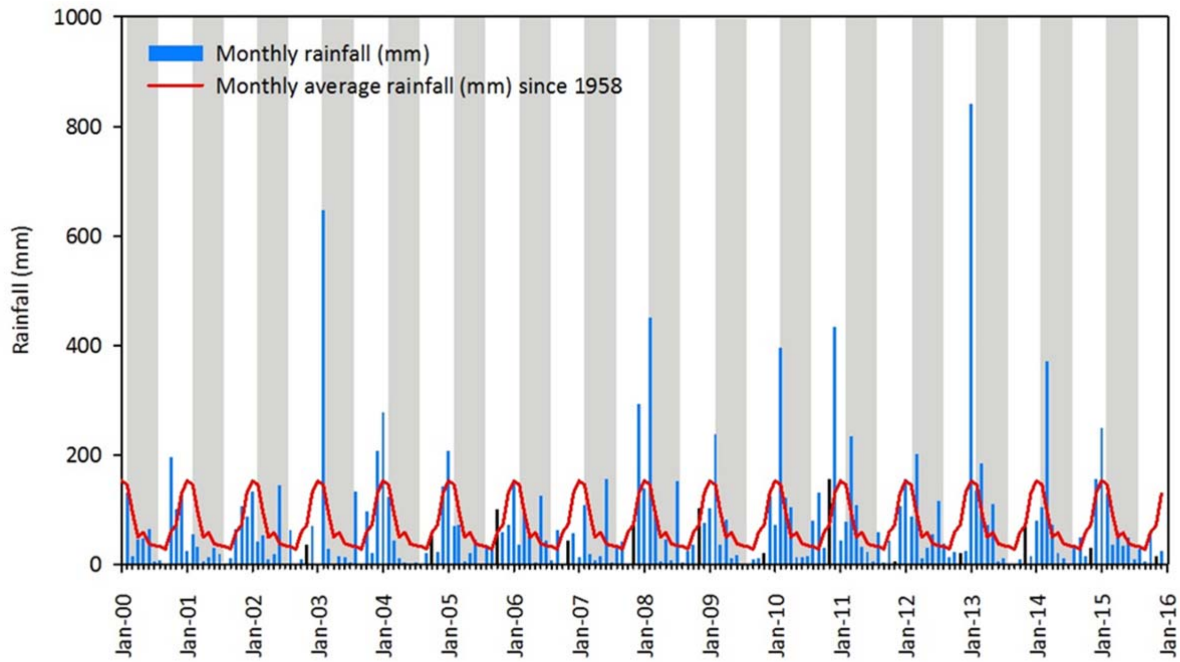


Figure 24: Total monthly rainfall (mm) for Gladstone, from January 2000 to December 2015. Data taken from station number 039123 (Gladstone Airport); from Bureau of Meteorology (http://www.bom.gov.au/climate/data/).

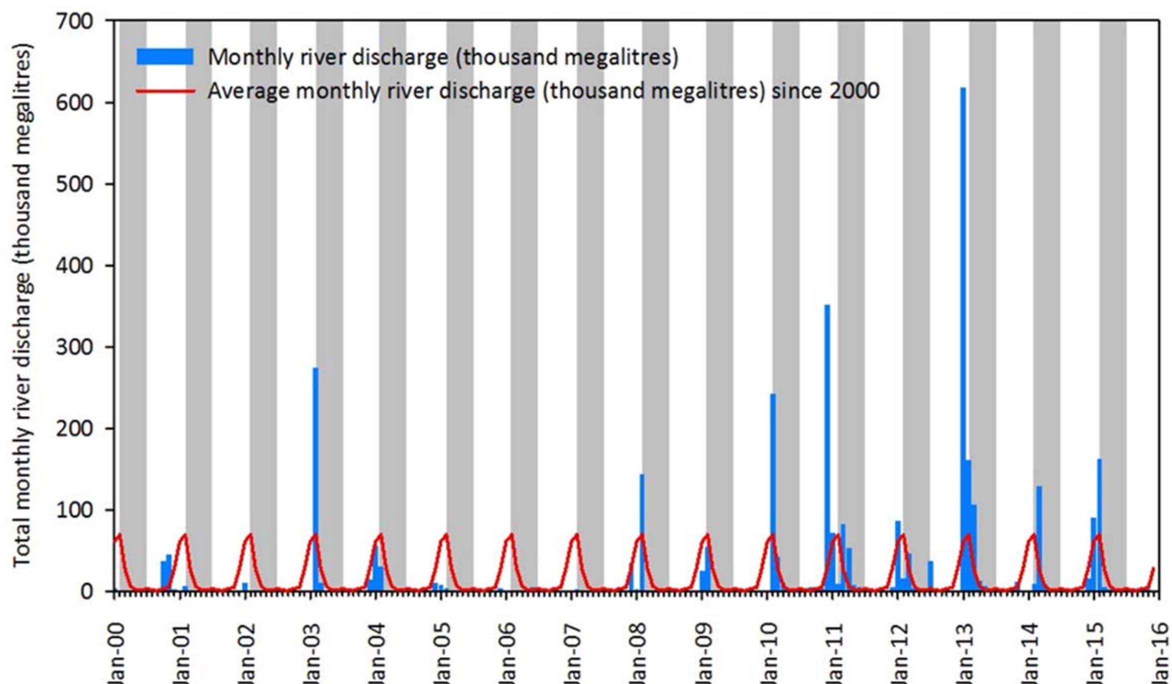


Figure 25: Mean monthly river discharge (volume Mega Litres) for Calliope River at Castlehope, Gladstone, from January 2000 to December 2015. Data taken from Calliope Basin, site 132001A; from DERM Water Monitoring (http://watermonitoring.derm.qld.gov.au/host.htm).

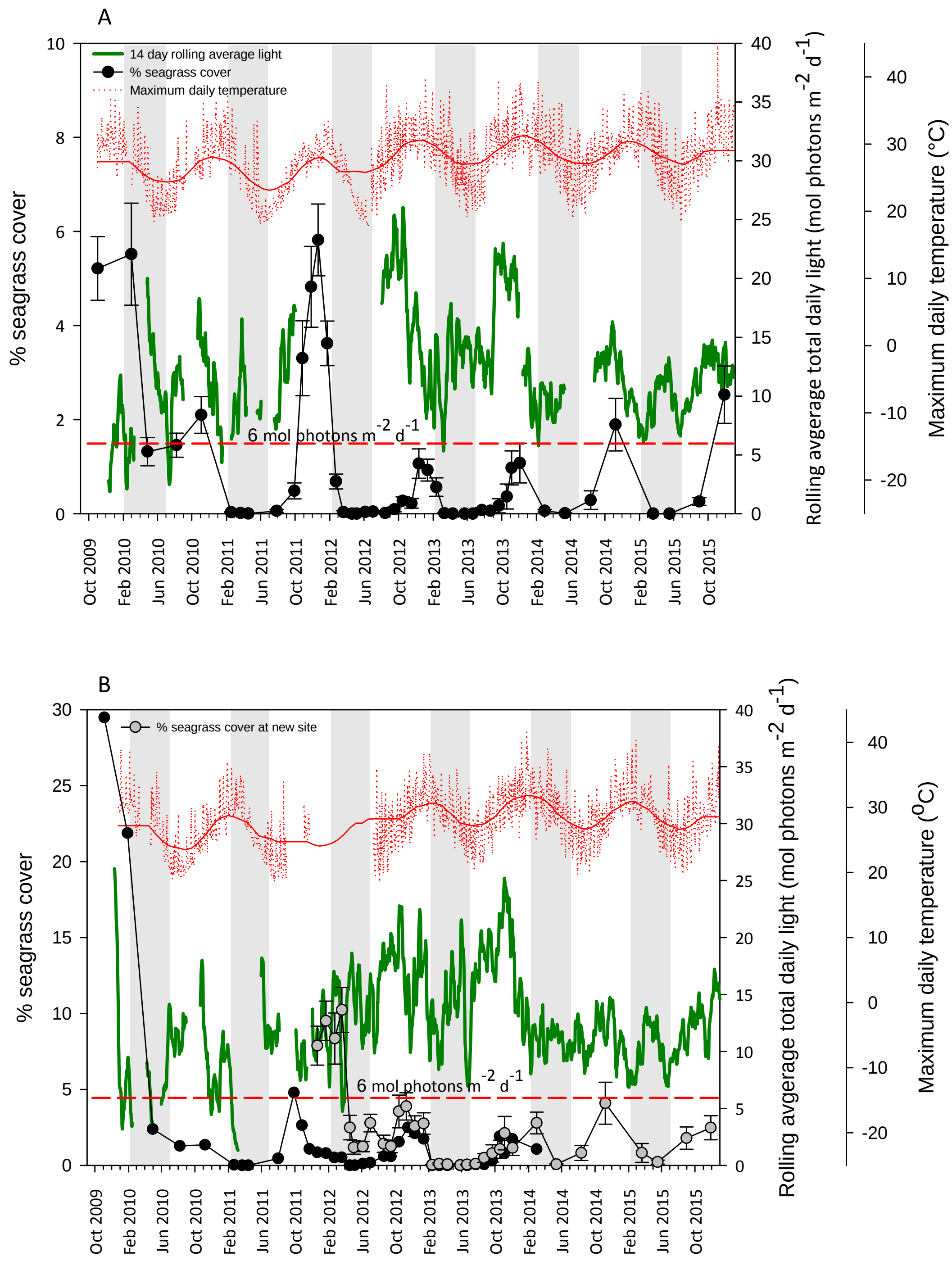


Figure 26: Rolling 14 day average total daily light (mol photons m⁻² d⁻²) and seagrass percent cover at Fisherman's Landing (A) and Wiggins Island (B) in the inner harbour over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Gladstone. Light and temperature data (to November 2013) sourced from Vision Environment QLD (2013).

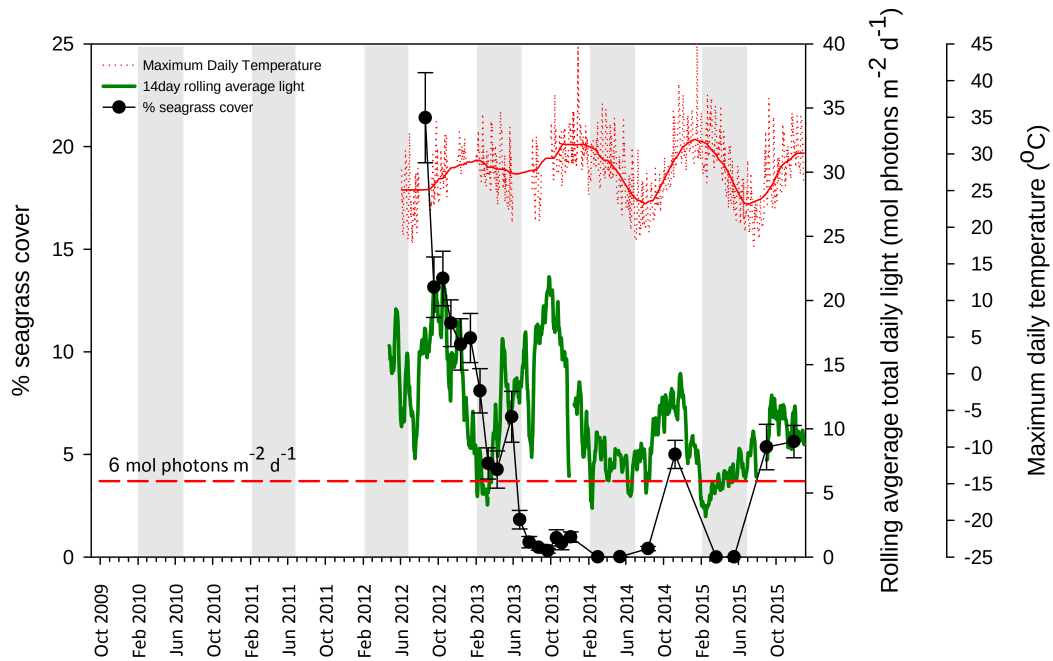


Figure 27: Rolling 14 day average total daily light (mol photons m^{-2} d^{-2}) and seagrass percent cover at Black Swan in the inner harbour over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Gladstone. Light and temperature data (to November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

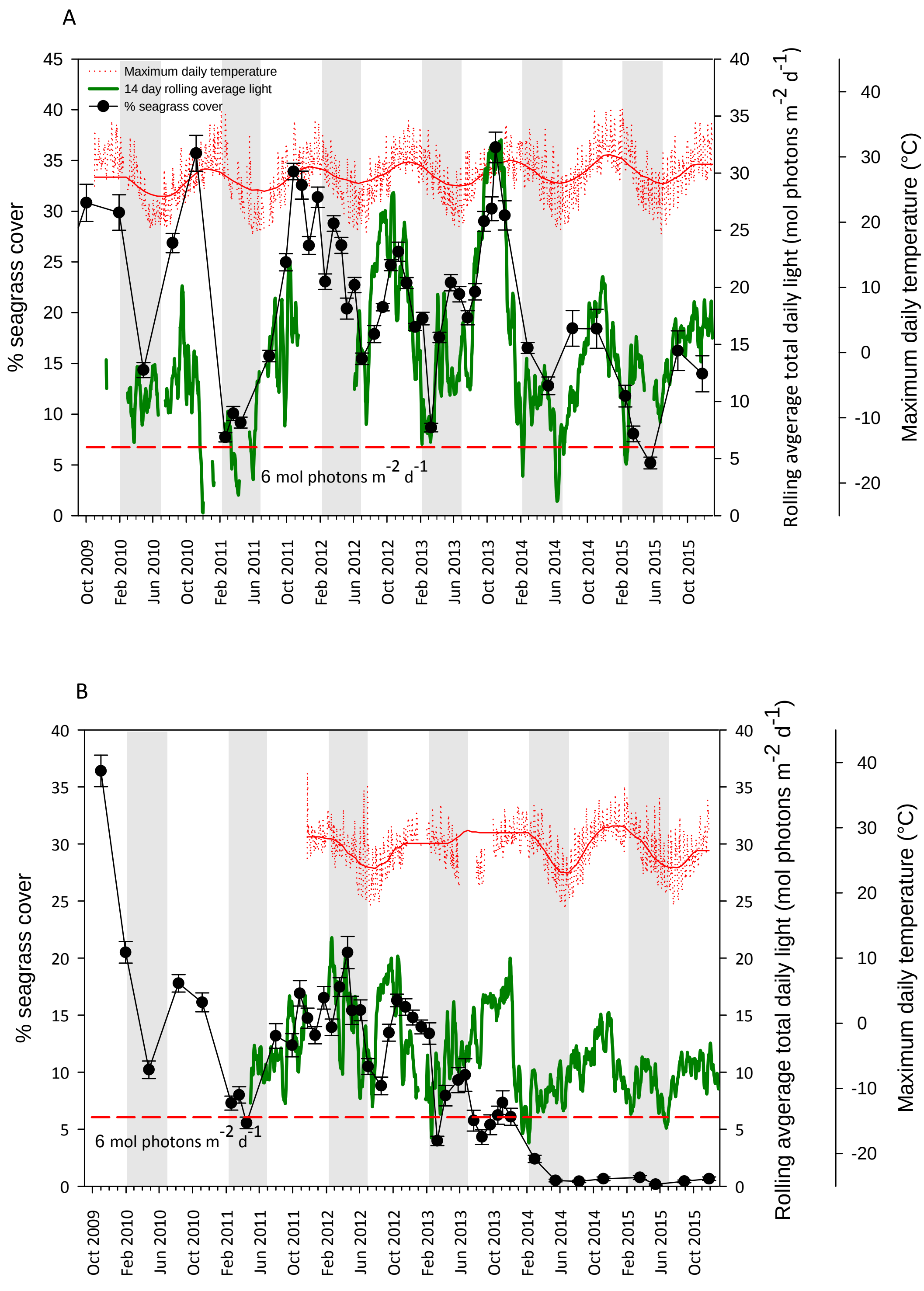
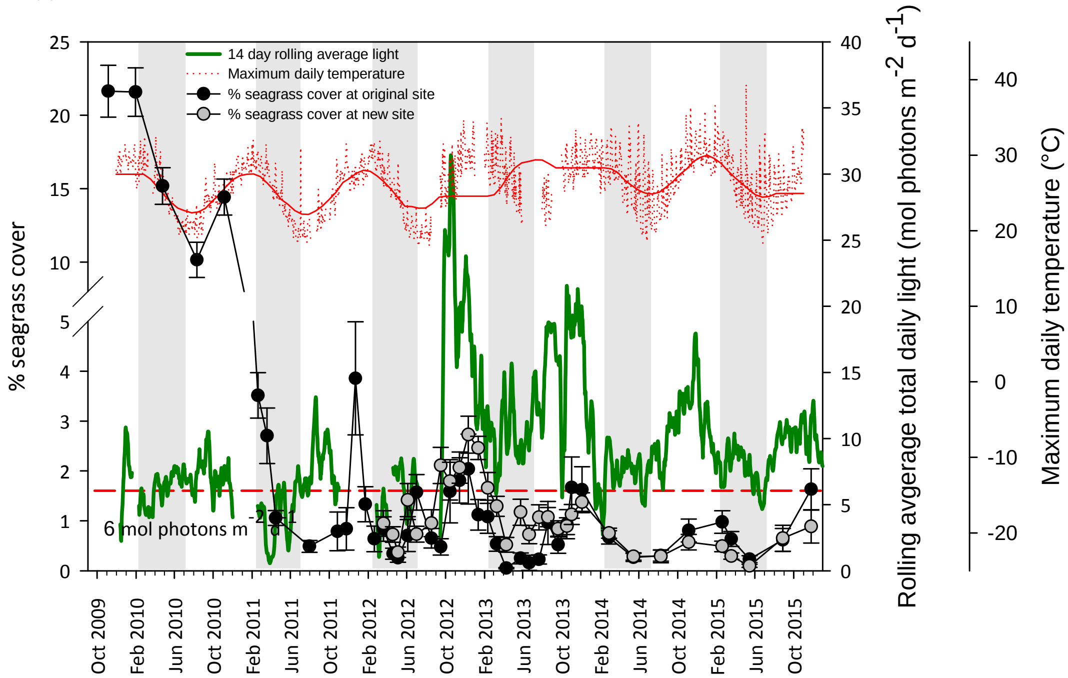


Figure 28: Rolling 14 day average total daily light (mol photons m^{-2} d^{-2}) and seagrass percent cover at Pelican Banks North (A) and Pelican Banks South (B) in the outer harbour over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Gladstone.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

A



B

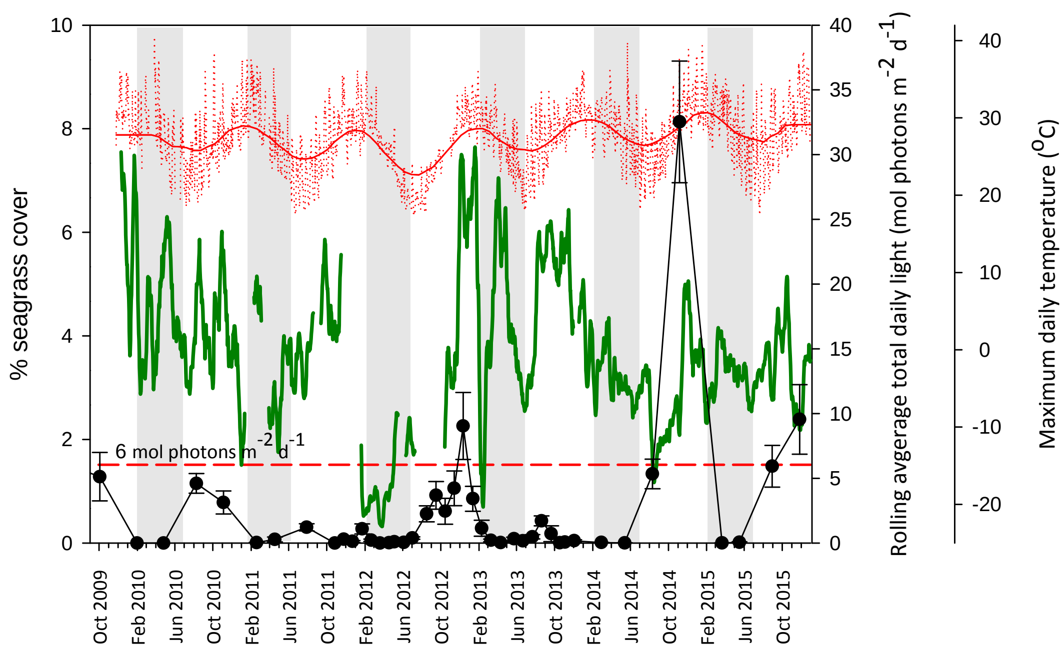


Figure 29: Rolling 14 day average total daily light (mol photons m^{-2} d^{-2}) and seagrass percent cover at Facing Island (A) and Rodds Bay (B) over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Gladstone. Light and temperature (to November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

4.4 Discussion

In seagrass ecosystems, nutrients and light are the most common limiting factors that control seagrass abundance (Waycott and McKenzie 2010; 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 Gladstone Harbour, 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), Gladstone 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. 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 Gladstone 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.

In January 2013, the Calliope River discharged at record levels and significant declines in seagrass percent cover were again detected at sites throughout the harbour. Since the event, monitoring at transect sites has 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. 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) but the recent trajectory for seagrasses has been markedly different between sites.

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 meeting minimum requirements for *Z. muelleri* growth (Chartrand et al. 2016), the species is almost absent from the site. On the contrary, at Pelican Banks North, trends in seagrass abundance continue to closely match trends in the light history. Declines in light below growth requirements are closely followed by declines in abundance and seasonal peaks in seagrass correspond with peak light levels. This demonstrates that while light is an important factor in determining seagrass growth there are a range of other variables at play. Environmental factors such as temperature and nutrients as well as propagule supply and grazing pressure are also likely to be key influences on seagrass change.

Recent research in Gladstone (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 Gladstone 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.

5 CONCLUSIONS

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 climate induced patterns of inter-annual variation. However, given the frequency and intensity of recent climatic 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 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 light requirements of local seagrass species (Chartrand et al. In review; 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 a major driver of seasonal changes in seagrass abundance in Gladstone (Chartrand et al. 2016). Additional studies to determine the impact of temperatures on seagrass light requirements in Gladstone would allow further refinement of the existing management tools.

In January 2013, the Calliope River discharged at unprecedented levels causing declines in seagrass across permanent transects sites. At inner harbour sites, there have been promising signs of recovery, with gradual increases in abundance each growing season. At most inner harbour sites, seagrass percent cover at the peak of the 2015 growing season had almost reached pre flood levels (November 2012). However, seagrass abundance in the inner harbour remains well below levels detected at the onset of 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 harbour.

At outer harbour sites, recovery since the 2013 flood event were initially promising; however, monitoring over the 2014 and 2015 growing seasons has shown seagrass abundance to be at the lowest levels recorded over the course of the program. Levels of resilience at Pelican Banks North are likely still quite high due to the multiple mechanisms for reproduction and the presence of a relatively 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 is well equipped to recover from impacts. Conversely, at the southern end of Pelican Banks, the low density of plants, prolonged nutrient enrichment and a lack of propagules may be hampering recovery. The loss of the dominant species *Z. muelleri* from the site in 2015, 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 Port Curtis in November 2015 (Davies et al. 2016) were generally in line with transect monitoring results. Above-ground biomass at the Pelican Banks meadow was the lowest recorded over the history of monitoring (since 2002) and seagrasses at most monitoring meadows remained at low levels following significant declines in 2009/10 (Davies et al. 2016). Davies et al. (2016) attributed the low biomass in November 2015 to a combination of 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. The annual monitoring program also accounts for changes in total meadow area. Despite biomass declines, meadow area increased in the majority of monitoring meadows, particularly in the inner harbour region (Davies et al. 2016). The long-term and spatially expansive monitoring undertaken during annual re-mapping surveys seems to better capture overall trends. There are some differences in results between the programs however, due to the spatial variability in location of some meadows. 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 last year of the post dredging phase of the project (2016) will continue to provide insight into the capacity of seagrass resilience to human activities. If low levels of resilience detected at many monitoring sites persist then the tools and thresholds established through major research programs in Gladstone (Schliep et al. 2014; Chartrand et al. 2012) will be critical in ensuring successful management of their recovery. Currently seagrasses have shown some capacity to recover from impacts in Gladstone, 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 (McKenna et al. 2015; York et al. 2015; Rasheed et al. 2014; Pollard and Greenway 2013). The extensive and detailed seagrass monitoring and research efforts in Gladstone means we are well placed to understand these processes and look to implement measures to reduce the chances of exacerbating natural impacts by human activities.

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