Appendix F
Seagrasses in Port Curtis and Rodds Bay 2014

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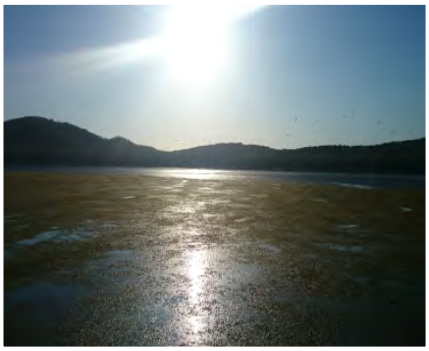

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Annual long-term monitoring, biannual Western Basin, and updated baseline survey

Carter AB, Davies JD, Bryant CV, Jarvis JC, McKenna SA and Rasheed MA

Report No. 15/06

June 2015

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A Report for Gladstone Ports Corporation

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KEY FINDINGS

Seagrass Condition 2014

1. Seagrass meadows in the Port Curtis region and in Rodds Bay generally improved in condition in 2014 compared with 2013, when substantial declines in the condition of many meadows were recorded.

2. Annual monitoring in 2014 found 13 of the 14 monitoring meadows had increased in area. The average condition of monitoring meadows within the Port and in the Rodds bay reference area were classified as 'fair'. Despite increases in area, meadow biomass was below the long-term average in 13 of the 14 monitoring meadows.

3. In addition to the long-term monitoring meadows, an updated baseline survey of all seagrasses in the entire Port Curtis region (The Narrows to Rodds Bay) was conducted in 2014. From November 2013 to November 2014 seagrass habitat had expanded by over 8,300 ha.

4. The majority of this expansion occurred in deepwater areas offshore of the Port limits (> 18,000 ha mapped) which increased by over 6,000 ha from 2013 to 2014.

5. Coastal seagrass meadows also expanded in area, particularly in Rodds Bay and The Narrows regions where much of the 2013 declines had occurred.

6. Despite increases in meadow area in 2014, seagrass meadow area and biomass generally remain below the long-term average following significant declines in 2009/2010. This has likely reduced seagrass resilience to further impacts; it may take some time for meadows to reach historical peak levels documented in 2007 - 2008.

7. Dugong feeding trails continued to be observed in the survey area at Pelican Banks and Quoin Island during the 2014 survey.

8. Environmental monitoring within seagrass meadows indicates that the light environment generally remained favourable for seagrass growth during the 2014 growing season.

9. Broad patterns in seagrass condition in Gladstone have been consistent with other areas of Queensland's east coast where seagrasses are monitored. All of these locations experienced major losses of seagrass in the period around 2010 with these declines associated with climate impacts. These locations have shown variable signs of recovery to pre-flood levels.

IN BRIEF

Seagrass monitoring in Port Curtis and Rodds Bay commenced in 2002 and has been monitored annually from 2004. In November each year, 13 core seagrass meadows (increased to 14 meadows in 2009) representing the range of different seagrass community types in Port Curtis and Rodds Bay (reference meadows) are assessed for changes in biomass, area and species composition (Figure 1). Changes in these metrics are then used to develop a seagrass condition index (see Sections 2.5 and 3.6 of this report for further details). In addition to these core meadows, all seagrasses within the seagrass annual monitoring survey area have been mapped biannually since 2009 (June and November) to determine the total distribution of seagrasses during both the senescent (June) and peak (November) seagrass growing season.

All seagrasses within the greater region from The Narrows to Rodds Bay have been periodically re-mapped. These surveys occurred in 2002, 2009, 2013 and 2014. In 2013 and 2014 the surveys also included adjacent offshore areas in the Great Barrier Reef Marine Park (Figure 1). This report presents the latest findings from these monitoring and mapping surveys. Additional and more frequent (at least quarterly) assessments of seagrass were established at permanent transect locations in The Narrows, Western Basin, Inner Harbour, Outer Harbour, South Trees and Rodds Bay regions to act as sentinel or sensitive receptor assessment sites during the Western Basin Dredging and Disposal Project (WBDDP) and these results are reported separately (Davies et al. 2015a).

In 2014 there were some signs of seagrass recovery from the reductions seen during 2013. Seagrass meadow area from The Narrows to Rodds Bay increased substantially from 2013, including in the Western Basin region (Figure 2). Many of the annual seagrass monitoring meadows had also improved in condition from 2013, driven largely by increases in meadow area (Figures 3 and 4). Despite these improvements, biomass and area for many meadows were often well below peaks recorded between approximately 2006 and 2009.

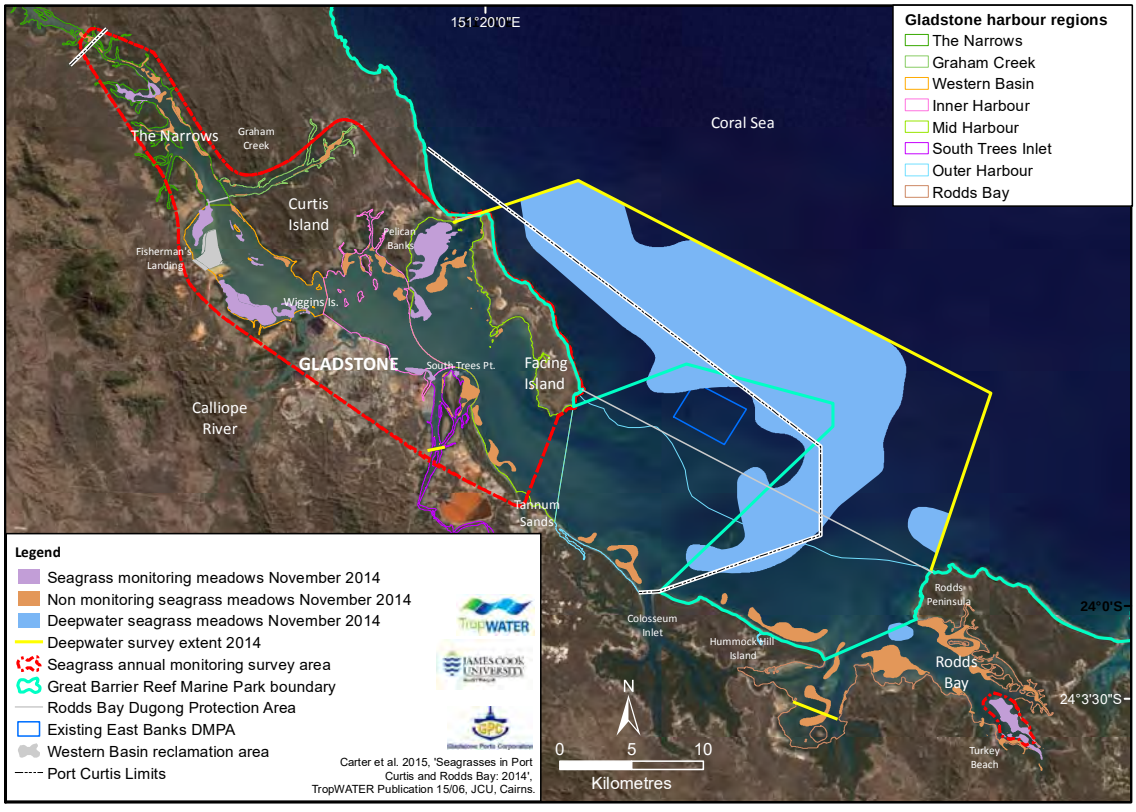

Figure 1. Seagrass distribution in the Port Curtis region, November 2014.

Meadow area increased in all of the monitoring meadows in The Narrows and Inner Harbour regions, with three of the meadows classed as being in "good" condition (Figure 3). Seagrass biomass generally remained below the long-term average following significant declines from peak biomass in 2007 - 2008. Many meadows have experienced long-term shifts from dominance by *Zostera muelleri* subsp. *capricorni** to the colonising *Halophila ovalis.*

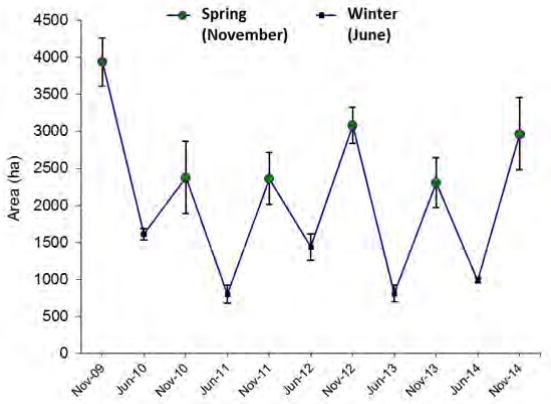

Figure 2. Seasonal changes in total seagrass meadow area in the seagrass annual monitoring survey area, excluding Rodds Bay, 2009 - 2014.

Monitoring meadows in the Inner Harbour, Mid Harbour and South Trees regions were classified as being in "poor" to "fair" condition with biomass remaining below the long-term average (Figure 4). This was despite large increases in meadow area for monitoring meadows near South Trees. At the reference monitoring meadows in Rodds Bay, seagrass had increased from only a few remnant patches of seagrass in 2013 to approach peaks in area observed prior to 2009/2010 in the monitoring program, but as with other locations biomass remained below historical levels (Figure 4).

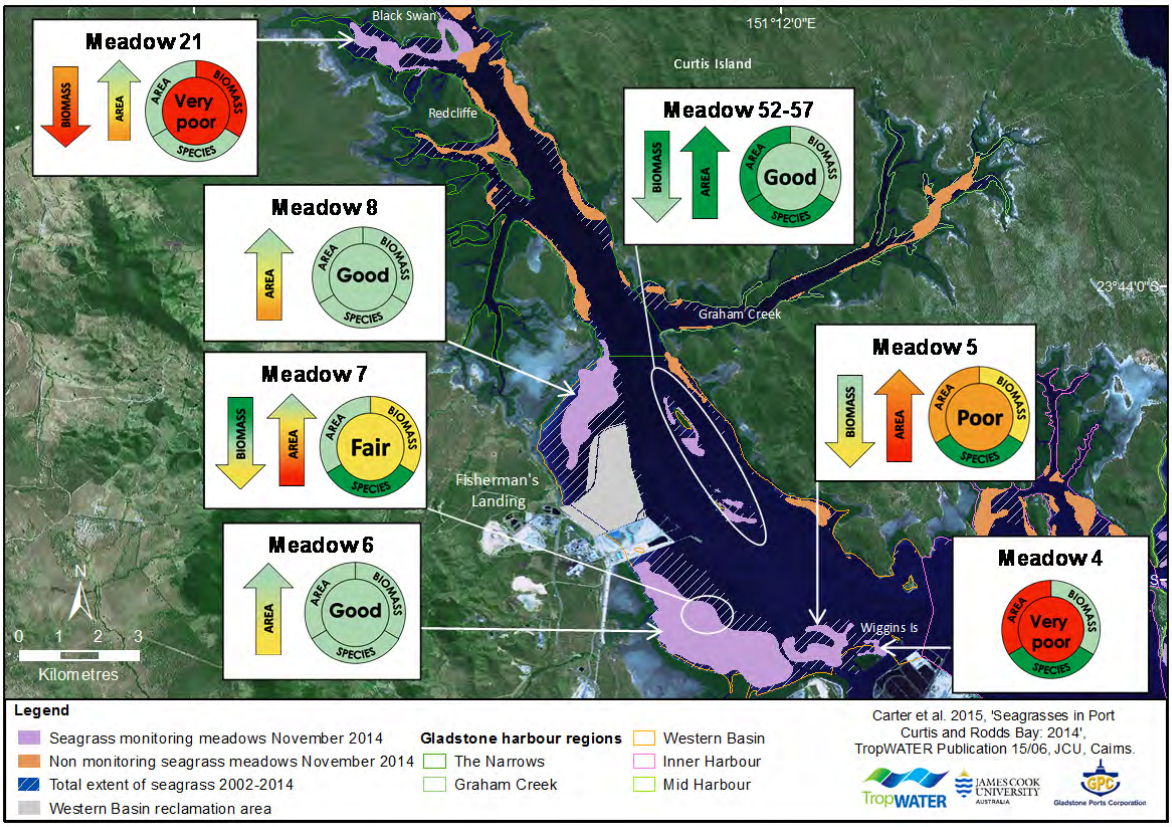

Figure 3. Seagrass distribution in The Narrows and Inner Harbour regions of Port Curtis, November 2014.

*The naming convention used in previous reports has been updated to align with the accepted nomenclature for this species.

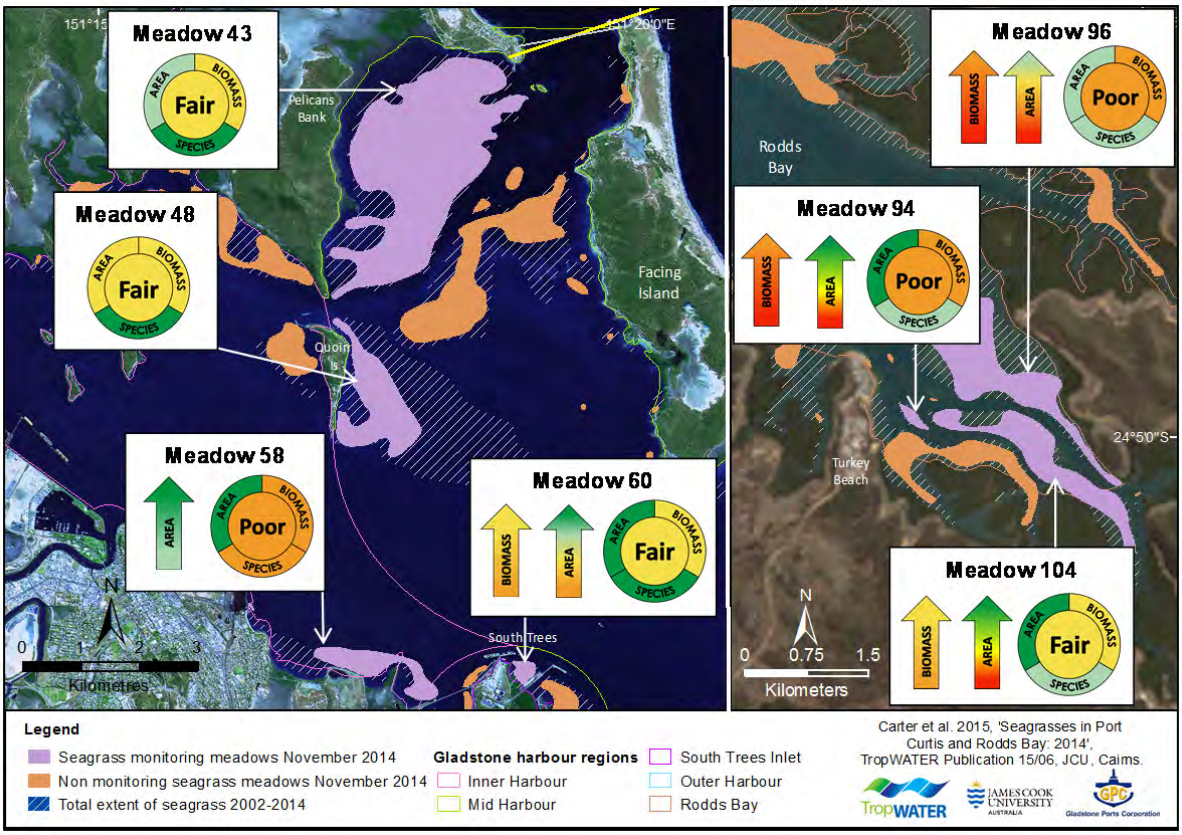

Figure 4. Seagrass distribution in the Inner Harbour, Mid Harbour and South Trees regions of Port Curtis and Rodds Bay, November 2014.

Seagrasses within the broader region (The Narrows to Rodds Bay) were re-mapped in November 2014 and the largest overall area of seagrass ever was recorded. The survey included both coastal and deepwater seagrass with a total of 31,385 ha mapped. Comparing just the survey extent common to all previous surveys, over 12,700 ha was mapped. This is a substantial increase (> 8,000 ha) from 2013, where seagrass area had reduced by 63% compared with the previous 2009 survey of the entire region.

The deepwater areas within the port limits represent a substantial portion of meadow area increases in 2014, and are the areas where approximately half of the reduction in seagrass area occurred in 2013. These deepwater meadows underwent a net reduction of approximately 3,600 ha (~ 75%) from 2009 to 2013 but expanded by over 6,100 ha by November 2014 to the largest seagrass extent detected in the deepwater baseline surveys. Outside of the port and the Dugong Protection Area (DPA), an additional 18,000 ha of deepwater seagrass habitat was mapped in adjacent offshore areas of the Great Barrier Reef Marine Park. These vast meadows were relatively low in average biomass and were comprised of several *Halophila* species typical of deepwater areas in the Great Barrier Reef lagoon.

Seagrass area also expanded in the shallow areas since November 2013 with coastal seagrass area increasing from 3,027 ± 547 ha to 5,227 ± 837 ha. The majority of the loss in coastal seagrass area in 2013 occurred in the inner harbour regions from The Narrows to Wiggins Island and also outside of the port in Rodds Bay. These were also the areas where much of the gains in seagrass area occurred in 2014.

Monitoring of light in the Port Curtis region indicated that during the 2014 growing season light at sites in the Western Basin region remained above the levels required for Gladstone seagrass. The favourable light environment combined with below average rainfall and flow of the Calliope River (Figure 5) may have allowed for increases in meadow area over the 2014 growing season; however the magnitude of declines

over the previous senescent season meant that seagrasses were re-establishing from very low levels. Therefore, as multiple years of climate related impacts are likely to have left a legacy of reduced resilience of seagrasses to impacts, it may take several consecutive years of improved environmental conditions for seagrass meadows in the Port Curtis region to return to 'very good' condition.

The Gladstone seagrass monitoring forms part of James Cook University's Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER) seagrass assessment and research program that examines seagrass condition in the majority of Queensland commercial ports. Seagrasses at other locations along the east coast of Queensland were negatively impacted by a number of climate events between 2010 and 2013, including flooding and cyclones. The trajectory for recovery since then has been varied with the condition of some meadows having substantially improved but many locations remaining in poor condition. For full details of the Queensland ports seagrass monitoring program see www.jcu.edu.au/portseagrassqld.

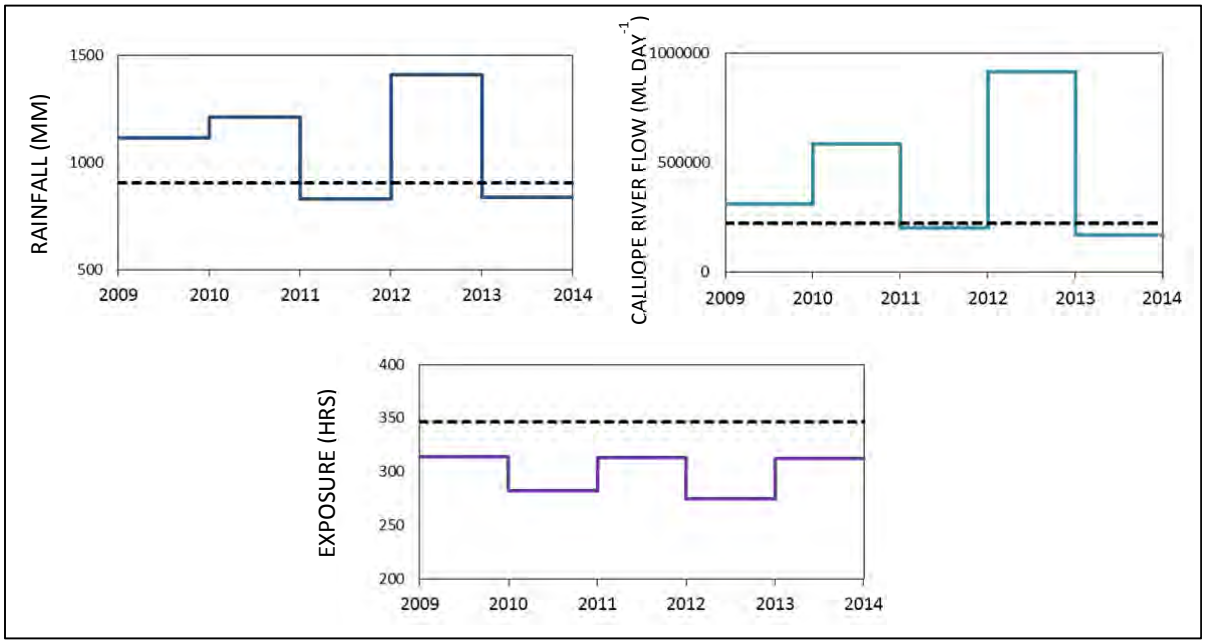

Figure 5. Recent climate trends in Gladstone: Annual average (solid line) compared to the long-term average (dashed line). See Section 3.7 for detailed climate data.

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1 INTRODUCTION

Seagrasses provide a range of critically important and economically valuable ecosystem services including coastal protection, support of fisheries production, nutrient cycling and particle trapping (Costanza et al. 2014; Hemminga and Duarte 2000). Seagrass meadows show measurable responses to changes in water quality, making them ideal sensitive receptors for monitoring the health of marine environments (Orth et al. 2006; Abal and Dennison 1996; Dennison et al. 1993).

1.1 Queensland ports seagrass monitoring program

A long-term seagrass monitoring and assessment program has been established in the majority of Queensland commercial ports. The program was developed by the Seagrass Ecology Group at James Cook University's Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER) in partnership with the various Queensland port authorities. While each location is funded separately, the common methodology and rationale provides a network of seagrass monitoring locations throughout Queensland (Figure 6).

Figure 6. Location of Queensland port seagrass assessment sites (red – long term monitoring; blue – baseline mapping only).

A strategic long-term assessment and monitoring program for seagrasses provides port managers and regulators with the key information to ensure effective management of seagrass resources. It is useful information for planning and implementing port development and maintenance programs so they have a minimal impact on seagrasses. The program also provides an ongoing assessment of many of the most vulnerable seagrass communities in the state.

The program not only delivers key information for the management of port activities to minimise impacts on seagrasses but has also resulted in significant advances in the science and knowledge of tropical seagrass ecology. It has been instrumental in developing tools, indicators and thresholds for the protection and management of seagrasses and an understanding of the drivers of tropical seagrass change. It provides local information for individual ports as well as feeding into regional assessments of the status of seagrasses.

For more information on the program and reports from the other monitoring locations see www.jcu.edu.au/portseagrassqld

1.2 Gladstone seagrass monitoring program

Seagrass surveys in the Gladstone region indicate the harbour contains diverse and productive seagrass meadows and macro-benthic fauna (McKenna et al. 2014; Rasheed et al. 2003; Lee Long et al. 1992). Seagrasses in the Gladstone region are of particular value as a food source to dugong, recognised by the declaration of the Rodds Bay Dugong Protection Area (DPA). The collection of baseline information on seagrass meadows in port areas, and subsequent monitoring of a subset of representative meadows, enables port managers and regulators to make informed decisions regarding planning and development of port infrastructure.

Gladstone Ports Corporation (GPC) recognised that seagrasses are an important and sensitive component of the marine habitats within the port limits and are committed to maintaining the health of these habitats. In 2002, GPC commissioned TropWATER (formerly Fisheries Queensland) to conduct a fine-scale baseline survey of seagrass resources within the port limits and nearby Rodds Bay (Rasheed et al. 2003). The 2002 baseline survey identified large areas of seagrass within the port limits including 7,246 ± 421 ha of coastal seagrass habitat. The complete baseline survey area was resurveyed in November 2009 (Thomas et al. 2010) and November 2013 (Bryant and Rasheed 2013). The 2009 survey mapped 7,150 ± 509 ha of seagrass habitat within Port Curtis and the Rodds Bay DPA, excluding the deepwater meadows. Meadows in 2009 covered the greatest seagrass distribution since surveys began in 2002. The updated baseline survey in November 2013 mapped 3,027 ± 547 ha of coastal seagrass habitat within Port Curtis and the Rodds Bay DPA.

The annual seagrass monitoring program commenced in 2004 in response to a whole of port review (SKM 2004) and following recommendations from the Port Curtis Integrated Monitoring Program (PCIMP). Initially 10 seagrass meadows representative of the range of seagrass communities within the port were selected for monitoring, and included meadows considered in 2004 most likely to be impacted by port facilities and developments. Monitoring locations include intertidal and subtidal seagrass meadows, meadows preferred by dugong, and those likely to support high fisheries productivity. Three meadows in Rodds Bay (outside port limits) were also selected as reference sites for monitoring to provide information on seagrasses unlikely to be impacted by port activity, and to assist in identifying port-related versus regional causes of seagrass change. From 2009 an additional two monitoring meadows were added to the long-term monitoring program to reflect the shift in new port activity to the Curtis Island area as part of the Western Basin developments, and these meadows are in the vicinity of the development (Meadows 21 and 52-57). Due to the expansion of the reclamation area at Fisherman's Landing, Meadow 9 is no longer monitored as part of this program.

In previous reports, annual monitoring surveys have been referred to as the Western Basin surveys. The naming convention for the areas surveyed has been updated in this report to align with the PCIMP water quality zones (Figure 1), and the survey area is now referred to as the seagrass annual monitoring survey area. The Western Basin area referred to in previous reports is now referred to as The Narrows, Graham Creek, Western Basin, Inner Harbour, South Trees and Mid Harbour regions (Figures 3 and 4). These changes are reflected throughout this report.

Monitoring since 2002 has documented considerable inter-annual variability in seagrass meadow biomass and area. Variation in seagrass meadows is most likely a response to regional and local climatic conditions (Chartrand et al. 2009). Climate induced inter-annual variability is common throughout tropical seagrass meadows of the Indo-Pacific (Agawin et al. 2001). Seagrasses in Gladstone also are highly seasonal. Two broad seasons for seagrass growth have been identified in Gladstone; the growing season (July – January) where seagrasses typically increase in biomass and distribution in response to favourable conditions for growth; and the senescent season (February – June) when seagrasses typically retract and rely on stores or seeds to endure wet season conditions such as flooding, poor water quality and light reductions (Chartrand et al. 2012b). The peak of the growing season occurs between October - November (corresponding to the time of annual surveys) and seagrass biomass and area is at its lowest around June.

1.3 Additions to the annual seagrass monitoring program

In 2009 proposed infrastructure developments within the port area, including a number of reclamation and dredging projects, required an expansion of the established long-term seagrass monitoring program.

Additions to the monitoring program included:

1. Re-map all seagrass meadows within Port Curtis and Rodds Bay in 2009, 2013 and 2014 (Thomas et al. 2010) and reported here);
2. Establishment of a network of sensitive receptor seagrass sites where information on seagrass change is collected at least quarterly (and monthly during dredging) and linked to water quality monitoring and assessments of seagrass resilience (Davies et al. 2015a);
3. A research program to establish the light requirements of Gladstone seagrasses and investigate sub-lethal indicators of seagrass stress (Schliep et al. 2014; Chartrand et al. 2012b; Bostrom et al. 2006). Biannual mapping and assessment of all seagrasses within the seagrass annual monitoring survey area (2009 – 2014) (reported here).

The biannual surveys are conducted at the peak of each growing (November) and senescent season (June) and survey all seagrass habitat in the seagrass annual monitoring survey area including the previously established PCIMP annual long-term monitoring meadows. These biannual surveys document the spatial extent and biomass of intertidal and shallow subtidal seagrass meadows from The Narrows to the mouth of the Boyne River and assess seasonal dynamics prior to, during and after capital dredging associated with the WBDDP. The annual (November) long-term seagrass monitoring also includes Rodds Bay which is not directly affected by port activities. These surveys record changes in the distribution, biomass and species composition of seagrass monitoring meadows within Port Curtis and Rodds Bay. Due to the scale of developments and dredging operations in Gladstone, it was considered important to regularly determine the condition of seagrass meadows to complement the information gathered at the sensitive receptor site network.

New development proposals, including the proposed Port of Gladstone Gatcombe and Golding Cutting Channel Duplication Project, led to the requirement for a complete re-mapping of all of the seagrass meadows within Port Curtis and Rodds Bay in 2013. The updated survey included intertidal and shallow subtidal areas from The Narrows to Rodds Bay as well as deepwater areas (>6 m) within Port Curtis and the Rodds Bay DPA and adjacent offshore areas. These areas were surveyed again in 2014 to build a more comprehensive baseline understanding of these habitats leading up to any proposed works.

Results from the 2014 biannual seagrass surveys are discussed in relation to light and temperature data recorded in the seagrass meadows as part of the sensitive receptor seagrass transect monitoring program established for the Western Basin program. These are discussed in more detail in the report for that monitoring program (Davies et al. 2015a).

2 METHODS

2.1 Sampling approach and methods

Survey and monitoring methods followed the established techniques for the TropWATER Queensland-wide ports seagrass monitoring program (see Unsworth et al. 2012; Rasheed and Unsworth 2011; Taylor and Rasheed 2011; Lee Long et al. 1996). Detailed methods for its application in Gladstone can be found in the 2002 baseline and 2004 monitoring program reports (Rasheed et al. 2005; Rasheed et al. 2003).

Seagrass surveys were conducted 10-15 June 2014 (senescent season survey) and 4-10 November 2014 (growing season survey). The growing season survey involved mapping and assessing:

- All intertidal and shallow subtidal seagrasses within the seagrass annual monitoring survey area of Port Curtis and Rodds Bay;
- Established PCIMP meadows within Port Curtis and at Rodds Bay (annual long-term monitoring survey);
- All intertidal and shallow subtidal seagrasses within Port Curtis and the Rodds Bay DPA (updated coastal baseline survey);
- Deepwater areas within Port Curtis and the Rodds Bay DPA as well as adjacent offshore areas (updated deepwater baseline survey).

The growing season survey was conducted during November 2014 as seagrasses in the region are likely to be at their maximum biomass and distribution in late spring. This also allows appropriate comparisons with baseline (2009 and 2013) and PCIMP surveys conducted every October/November since 2004. The senescent season survey was conducted in June 2014 as seagrasses in the region were likely to be at their lowest biomass and area. This also allowed direct comparison with senescent season surveys conducted since June 2010.

At each survey site, seagrass characteristics including seagrass percent cover, species composition, above-ground biomass, percent algal cover, depth below mean sea level (MSL) (for subtidal meadows), sediment type, time and position (latitude and longitude) were recorded.

Three sampling techniques were used (Figure 7):

1. Intertidal areas: helicopter survey;
2. Shallow subtidal areas: boat-based free diving/grab survey;
3. Deepwater areas: boat-based CCTV camera sled tows.

Figure 7. Seagrass monitoring conducted using (A) helicopter aerial surveillance; (B) boat based free divers; and C) boat based CCTV camera sled tows.

2.1.1 Intertidal and shallow subtidal areas

Intertidal meadows were sampled at low tide using a helicopter. GPS was used to record the position of meadow boundaries. Seagrass characteristics were recorded at sites scattered within the seagrass meadow as the helicopter hovered two metres above the seagrass. Power analysis techniques were used to determine the appropriate number of sampling sites for each meadow in order to detect seagrass meadow change (Rasheed et al. 2003).

Shallow subtidal meadows were sampled from a small boat using free divers. Seagrass characteristics were recorded at sites located along transects perpendicular to the shoreline at ~100 - 500 m intervals, or where major changes in bottom topography occurred. Transects extended to the offshore edge of seagrass meadows with random sites used to measure continuity of habitat between transects.

Seagrass above-ground biomass was determined using a "visual estimates of biomass" technique (see Mellors 1991; Kirkman 1978). A 0.25 m^2 quadrat was placed randomly three times at each site. For each quadrat, an observer assigned a biomass rank made in reference to a series of quadrat photographs of similar seagrass habitats for which the above-ground biomass had previously been measured. Two separate ranges were used - low biomass and high biomass. The relative proportion of the above-ground biomass (i.e. percentage) of each seagrass species within each quadrat was also recorded. At the completion of ranking, the observer also ranked a series of photographs of calibration quadrats that represented the range of seagrass observed during the survey. These calibration quadrats had previously been harvested and the actual biomass determined in the laboratory. A separate regression of ranks and biomass from the calibration quadrats was generated for each observer and applied to the biomass ranks given in the field. Field biomass ranks were converted into above-ground biomass estimates in grams dry weight per square metre (g DW m^{-2}).

2.1.2 Deepwater areas

At each offshore sampling site the underwater camera system was towed for approximately 100 m while footage was observed on a TV monitor and recorded. Benthos on the seafloor was captured in the net and used to confirm seagrass species observed on the monitor. The technique ensures that a large area of seafloor was sampled at each site so that patchily distributed seagrass typically found in deepwater habitats was detected.

Seagrass species composition was measured from the sled net sample and from the video screen when several species according to Kuo and McComb (1989). Seagrass biomass estimates were made from video images using a calibrated visual estimates technique. Algae percent cover was estimated from a video grab and algae type was identified according to Cribb (1996).

2.2 Habitat mapping and Geographic Information System

Spatial data from the 2014 survey were entered into the Port Curtis Geographic Information System (GIS). Three seagrass GIS layers were created in ArcGIS® - site information, seagrass meadow characteristics and seagrass landscape category:

1. **Site information** - Includes seagrass percent cover, seagrass above-ground biomass for each species, depth below mean sea level (dbMSL), sediment type, algal cover, time, date, latitude, longitude, sampling method, and any comments;
2. **Seagrass meadow characteristics** - Area data for individual seagrass meadows. Seagrass meadows were assigned a meadow identification number which was used to compare individual meadows between annual monitoring surveys. Monitoring meadows are referred to by these identification numbers throughout this report. Seagrass community types were determined according to overall species composition (the mean species composition of all sites within the meadow boundary) from

nomenclature developed for seagrass meadows of Queensland (Table 1). Density categories (light, moderate, dense) were assigned to community types according to above-ground biomass of the dominant species (Table 2);

3. **Seagrass meadow landscape category** - Area data showing the seagrass landscape category determined for each meadow (Figure 8).

Table 1. Nomenclature for community types in Port Curtis and Rodds Bay.

Community type	Species composition
Species A	Species A is 90-100% of composition
Species A with mixed species	Species A is 50-90% of composition
Species A/Species B	Species A is 40-60% of composition

Table 2. Density categories and mean above-ground biomass ranges for each species used in determining seagrass community density in Port Curtis and Rodds Bay.

Density	Mean above-ground biomass (g DW m^{-2})				
	H. uninervis **(thin)**	***H. ovalis*** ***H. decipiens***	***H. uninervis*** **(wide)**	***H. spinulosa***	***Z. muelleri*** **subsp.** ***capricorni***
Light	< 1	< 1	< 5	< 15	< 20
Moderate	1 - 4	1 - 5	5 - 25	15 - 35	20 - 60
Dense	> 4	> 5	> 25	> 35	> 60

Meadows were also assigned a mapping precision estimate (in metres) based on mapping methods used for that meadow (Table 3). The mapping precision for coastal seagrass meadows ranged from ±5 m for isolated seagrass patches to ±750 m for offshore deepwater meadows. The mapping precision estimate was used to calculate a range around each meadow area estimate and is expressed as a meadow reliability estimate (R) in hectares.

Isolated seagrass patches
The majority of area within the meadows consisted of unvegetated sediment interspersed with isolated patches of seagrass

Aggregated seagrass patches
Meadows are comprised of numerous seagrass patches but still feature substantial gaps of unvegetated sediment within the meadow boundaries

Continuous seagrass cover
The majority of area within the meadows comprised of continuous seagrass cover interspersed with a few gaps of unvegetated sediment

Figure 8. Seagrass meadow landscape categories: (A) Isolated seagrass patches, (B) Aggregated seagrass patches, (C) Continuous seagrass cover.

Table 3. Methods used to determine mapping precision estimates for each seagrass meadow.

Mapping precision	Mapping method
≤5 m	Meadow boundaries mapped in detail by GPS from helicopter, Intertidal meadows completely exposed or visible at low tide, Relatively high density of mapping and survey sites, Recent aerial photography aided in mapping.
10-20 m	Meadow boundaries determined from helicopter and diver/grab surveys, Intertidal boundaries interpreted from helicopter sites, Subtidal boundaries interpreted from survey sites and aerial photography, Moderately high density of mapping and survey sites.
20-50 m	Meadow boundaries determined from helicopter and diver/grab surveys, Intertidal boundaries interpreted from helicopter sites, Subtidal boundaries interpreted from diver/grab survey sites, Lower density of survey sites for some sections of boundary.
>50 m	Subtidal meadow boundaries interpreted from survey sites

2.3 Environmental data

Light and water temperature data was collected by Vision Environment Queensland throughout Gladstone Harbour as part of the Western Basin water quality monitoring program until December 2013. Light and water temperature data for 2014 was collected as part of seagrass monitoring at the sensitive receptor sites (Davies et al. 2015a).

Tidal exposure to air for Gladstone intertidal meadows was calculated by summing the total daylight hours that tidal height was < 1 m (the tidal height at which seagrass meadows expose at this location) for the 12 months preceding each November survey. Tidal data was provided by Maritime Safety Queensland (© The State of Queensland (Department of Transport and Main Roads) 2014, Tidal Data) for Gladstone at Auckland Point (MSQ station # 052027A; www.msq.qld.gov.au). Total daily rainfall (mm) was obtained for the nearest weather station (Gladstone Airport, station # 039123) from the Australian Bureau of Meteorology website (http://www.bom.gov.au/climate/data/) for the 12 months preceding each November survey. Calliope River water flow data (total monthly megalitres (ML)) was obtained from the Department of Natural Resources and Mines (station # 132001A; www.watermonitoring.derm.qld.gov.au).

2.4 Statistical analysis

Prior to analysis, data exploration protocols were conducted as defined by Zuur et al. (2010) including checks for zero inflation of biomass data. Following data exploration, all biomass data were square-root transformed to increase homoscedasticity. One-way analysis of variance (ANOVA) was then conducted on monitoring meadows with non-zero inflated data (meadows 5, 6, 21, 43, 48, 58, 60, 94, 96 and 104) to test for differences in mean biomass between years. Following analysis all residuals were inspected visually and pairwise comparisons were made using a Tukey's Honest Significant Differences (HSD) test in the multcomp package(Hothorn et al. 2008). Analysis was conducted using R (R Development Core Team 2013). Detailed results are presented in the Appendix 2.

Biomass data for meadows 4, 7 and 8 was zero-inflated, so a binomial-Gamma hurdle model was used to quantify the effects of year (Zuur et al. 2012). Linear regressions using a Gamma distribution are appropriate for continuous data that are > 0 (Zuur et al. 2007). However, the Gamma distribution does not allow for zero values. Therefore, the hurdle model approach models the zero data separately from the non-zero data by first quantifying the effect of year on the presence or absence of seagrass biomass with a binomial regression, and then analysing the non-zero biomass data with a linear regression using a Gamma distribution. When visual inspection of residual plots did not reveal any obvious deviations from homoscedasticity or normality, p-values were obtained by maximum likelihood ratio tests of the model with the effect in question against the null model. Analysis was conducted using the lme4 package (Bates et al. 2013). Following analysis with the binomial-Gamma hurdle model, all residuals were inspected visually and pairwise comparisons were made using a holm adjusted Tukey's HSD test as described previously.

2.5 Seagrass meadow condition index

This is the second year of applying a condition index to the seagrass monitoring meadows. The condition index is based on changes in mean above-ground biomass, total meadow area and species composition of each seagrass meadow. The classifications of the initial index applied across ports in 2013/2014 have been modified slightly. The previous three classifications of seagrass meadow condition: "good", "moderate" or "poor" were expanded to five classifications: "very good", "good", "fair", "poor" and "very poor" to more closely match other report card programs in Queensland that apply condition indices (i.e. Reef Rescue Marine Monitoring Program, Gladstone Healthy Harbour Program). The index allows for comparisons of current meadow condition against the long-term average.

Two different threshold ranges for biomass and three different threshold ranges for area were developed to recognise that biomass and area of some seagrass meadows are historically stable while others are relatively variable and fluctuate substantially from year to year. These differences reflect growth characteristics of species that comprise different meadows, as well as whether meadows are subtidal or intertidal. This resulted in four classes of monitoring meadow for reporting purposes.

- **Class 1 Meadows –** stable biomass, stable area
- **Class 2 Meadows –** variable biomass, stable area
- **Class 3 Meadows –** variable biomass, variable area (intertidal)
- **Class 4 Meadows –** variable biomass, variable area (subtidal)

For biomass and area the current value for each meadow was compared against the average for the meadow from the first 10 years of the program (2002, 2004-2012) and categorised according to one of the five condition categories (Table 4). Condition index ranges were selected based on the class of each meadow (see above). For meadows monitored since 2009, the long-term average of biomass and area was calculated from 2009 – 2013 data. When 10 years of data have been collected for these meadows the long-term average will remain fixed to maintain consistency in reporting.

Species composition was assessed qualitatively as “very good” when the species composition remained stable; “good” when there had been minor loss of the climax species; “fair” when there had been a substantial shift in species toward colonising species that indicates disturbance or stress; “poor” when the meadow had shifted to become dominated by colonising species; and “very poor” when there was a complete loss of the climax species. Species shifts are relative and determined on a meadow by meadow basis taking into account both the current years’ species composition and historical trends.

It is important to note that tropical seagrass communities vary in condition naturally due to a number of factors including climate. Some monitoring meadows classified as being in “poor” condition can reflect the natural range of expected conditions and not necessarily indicate anthropogenic impacts. The index provides a means of comparing current meadow condition and likely resilience to impacts against the known long-term average.

The final condition of each monitoring meadow was determined by looking at all three condition indicators (biomass, area and species composition), with the lowest of any of the three indicators determining the overall condition of each meadow. The average condition for all of the monitoring meadows combined was used to determine the overall seagrass condition for the study area. A trend arrow indicates where there has been an increase or decrease over a threshold boundary for any of the three indicators (biomass, area, species composition) from the previous year (Table 4).

Table 4. Seagrass condition index for Port Curtis and Rodds Bay seagrass monitoring meadows.

Seagrass indicators/class		Very Good	Good	Fair	Poor	Very Poor
Biomass	Stable	More than 20% above the baseline	Within 20% of the baseline (above or below)	Between 20% and 50% below the baseline	Between 50% and 80% below the baseline	More than 80% below the baseline
	Highly variable	More than 40% above the baseline	Within 40% of the baseline (above or below)	Between 40% and 70% below the baseline	Between 70% and 90% below the baseline	More than 90% below the baseline
Area	Stable	More than 10% above the baseline	Within 10% of the baseline (above or below)	Between 10% and 30% below the baseline	Between 30% and 50% below the baseline	More than 50% below the baseline
	Highly variable intertidal	More than 20% above the baseline	Within 20% of the baseline (above or below)	Between 20% and 50% below the baseline	Between 50% and 80% below the baseline	More than 80% below the baseline
	Highly variable subtidal	More than 40% above the baseline	Within 40% of the baseline (above or below)	Between 40% and 70% below the baseline	Between 70% and 90% below the baseline	More than 90% below the baseline
Species composition		Composition remains stable	Some loss of climax species	Shift towards colonising species	Colonising species dominant	Complete loss of climax species
Trend Indicators		Substantial increase from previous year		Substantial decrease from previous year		

3 RESULTS

3.1 Seagrasses in the Port Curtis and Rodds Bay region

At total of 2,789 habitat characterisation sites were surveyed in the Port Curtis and Rodds Bay region in November 2014 (Figure 9). Seagrass was present at 37% of sites.

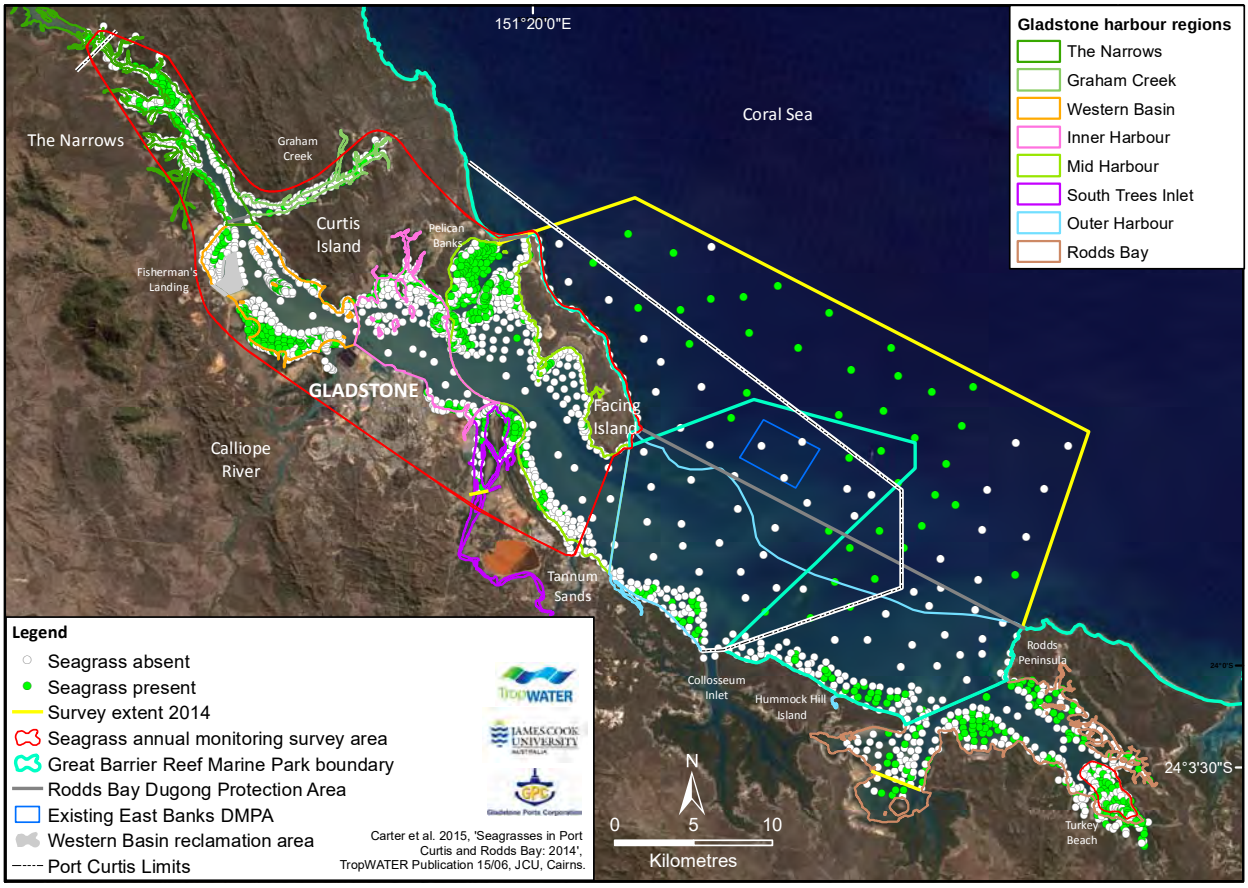


Figure 9. Habitat characterization sites in Port Curtis, Rodds Bay Dugong Protection Area and adjacent offshore areas, November 2014.

Five seagrass species (from two families) were observed during the November 2014 survey (Figure 10).

Family HYDROCHARITACEAE Jussieu:

Halophila decipiens
Halophila ovalis
Halophila spinulosa
Halodule uninervis (wide and thin leaf morphology)

Family ZOSTERACEAE Drummortier:

Zostera muelleri subsp. *capricorni*

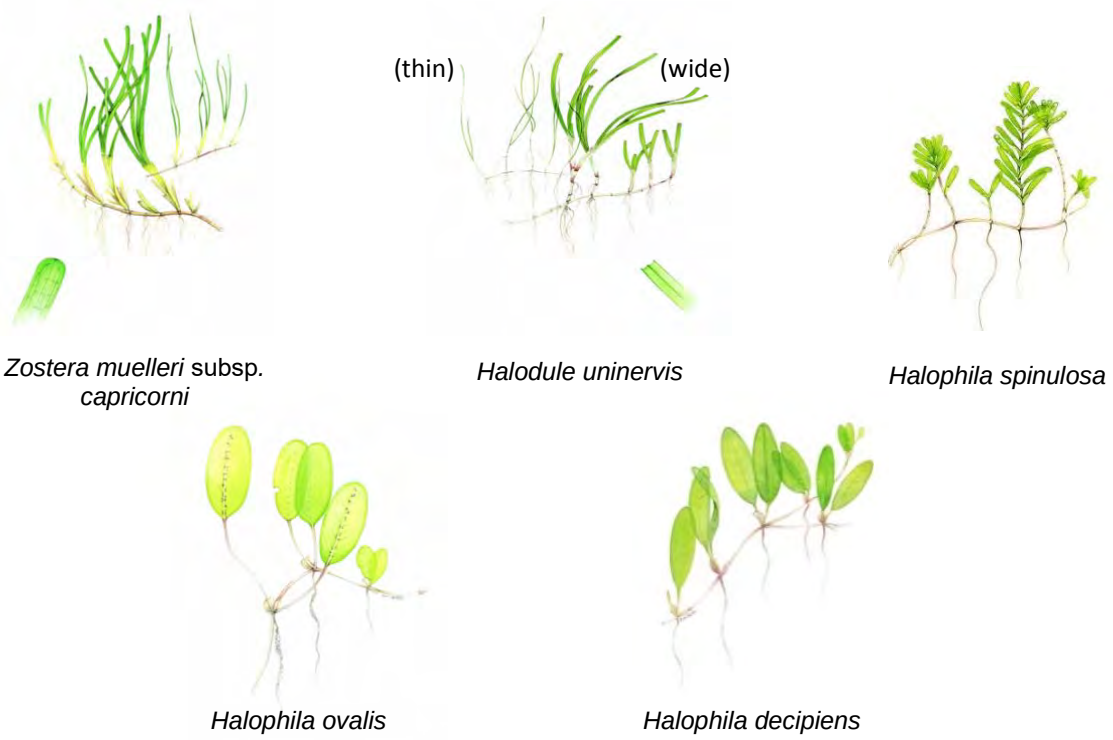


Figure 10. Seagrass species present in Port Curtis, Rodds Bay Dugong Protection Area and adjacent offshore areas, November 2014.

3.2 Coastal seagrasses

A total of 2,668 coastal habitat characterisation sites were surveyed in November 2014, with seagrass present at 37% of sites (Figure 9). The coastal survey incorporated intertidal and shallow (<6 m dbMSL) subtidal areas within Port Curtis and the Rodds Bay DPA. A total of 5,227 ± 838 ha of coastal seagrass habitat was mapped in 89 meadows (Figures 11 - 17; Tables 5 - 11). The area covered by seagrass meadows ranged from approximately 0.1 ha for several isolated patches located throughout the survey area and up to 595 ± 33 ha for the largest intertidal meadow located at Pelican Banks (Meadow 43; Figure 14; Table 7). Average above-ground seagrass biomass ranged from 0.01 g DW m^{-2} for three *Halophila ovalis* meadows in the Inner and Mid Harbour (Figures 14 – 15; Tables 8 - 9), to 15.05 g DW m^{-2} for a small intertidal *Zostera muelleri* subsp. *capricorni* meadow in Rodds Bay (Figure 17; Table 11). Twenty-two different seagrass community types were identified based on species composition and density (Figures 11 - 17; Tables 5 - 11).

In The Narrows, Grahams Creek and Western Basin regions the majority of seagrass meadows were comprised of aggregated patches (Figures 11 - 13; Tables 5 - 7). Meadows comprised of isolated patches included *Halophila decipiens* meadows in the middle sections of The Narrows and Graham Creek and *H. ovalis* meadows near the Channel Islands (Figures 11 - 13; Tables 5 - 7). The dominant species in intertidal meadows were *Z. muelleri* subsp. *capricorni* and *H. ovalis*, and *H. decipiens* in subtidal meadows.

In the Inner Harbour region, in the bays opposite the Clinton Coal Wharves between Wiggins Island and the western side of Quoin Island, seagrass meadows were equally comprised of isolated and aggregated patches (Figure 14; Table 8). Meadows in this region were dominated by either light *Z. muelleri* subsp. *capricorni* or light *H. ovalis*. Meadow 50, on the western side of Quoin Island, was comprised of all three *Halophila* spp. found in Gladstone (Figure 14; Table 8).

The Mid Harbour region, including the eastern side of Quoin Island, Pelican Banks, South Trees and Tannum Sands, contained a mix of seagrass landscape categories. The only meadow in the entire Port Curtis and Rodds Bay survey classed as having continuous seagrass cover was the *Z. muelleri* subsp. *capricorni* meadow 43 at Pelican Banks (Figure 14; Table 9). This Pelican Banks meadow consistently has the highest seagrass biomass meadow recorded in the port (12.31 ± 1.36 g DW m^{-2}). The large *Halodule uninervis* (thin) with mixed species meadows 48 (on the eastern side of Quoin Island) and 66 (Facing Island) were comprised of aggregated patches (Figure 14; Table 9). Meadow 66 was also one of the most diverse meadows surveyed with four species present. *Halophila decipiens* was present in meadows 48 and 66 in 2013 but was not recorded in these meadows in 2014 (Figure 14; Table 9). Meadow 159 is a large subtidal meadow which was observed and mapped for the first time in November 2013. In 2014 the meadow had substantially decreased in area by over 40 ha. The small meadows along the western side of Facing Island were mostly isolated patches and dominated either by light *H. ovalis*, *H. decipiens* or *Z. muelleri* subsp. *capricorni* (Figure 14; Table 9).

In the Mid Harbour region near South Trees, seagrass meadows consisted of isolated or aggregated patches, light cover, and dominated primarily by *H. ovalis* or *Z. muelleri* subsp. *capricorni* (Figure 15; Table 10). Aggregated patches of light and moderate *H. uninervis* (thin) dominated the sandier sediments along the coast from Boyne Island to Hummock Hill Island (Figures 15 - 16; Tables 10 - 11). In the Mid Harbour region near Tannum Sands the majority of meadows consisted of aggregated patches and were dominated by light *Z. muelleri* subsp. *capricorni* (Figure 16; Table 11).

Rodds Bay contained predominantly *Z. muelleri* subsp. *capricorni* dominated meadows; the larger meadows tended to contain aggregated patches while small meadows contained isolated patches (Figure 17; Table 12). Two moderate *H. ovalis* dominated meadows were recorded near Turkey Beach (Figure 17; Table 12). The small *Z. muelleri* subsp. *capricorni* meadow off Turkey Beach supported the highest above-ground biomass recorded in November 2014 with 15.05 g DW m^{-2} (Meadow 93; Figure 17; Table 12).

Dugong feeding trails (DFTs) have historically been recorded at The Narrows (Meadow 23; Figure 11), Fisherman's Landing (Meadow 6; Figure 13), Wiggins Island (Meadows 4 and 5; Figure 13), Pelican Banks (Meadow 43; Figure 14), Facing Island (Meadow 66; Figure 14) and South Trees (Meadow 60 and 76; Figure 15). In November 2014, DFTs were observed in the Western Basin at Wiggins Island (Meadows 4 and 5; Figure 13), in the Mid Harbour at Pelican Banks and Facing Island (Meadows 43 and 66; Figure 14) and South Trees (Meadow 76; Figure 15), and at Rodds Bay (Meadow 119; Figure 17). Dugongs were sighted in the channel and southern end of the Pelican Banks meadow (Meadow 43; Figure 14).

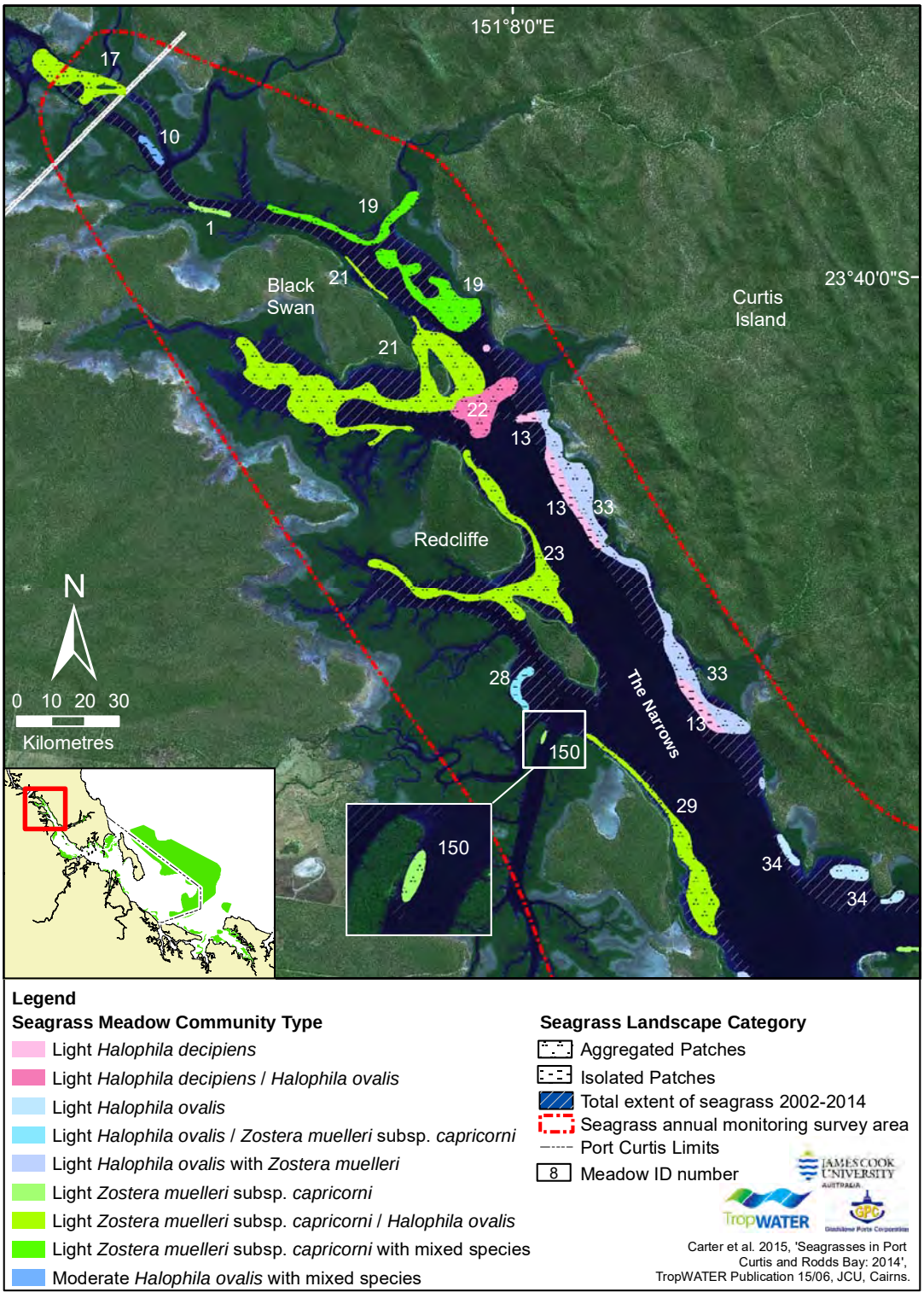


Figure 11. Seagrass distribution and community types in The Narrows region, November 2014.

Table 5. Seagrass community type, seagrass landscape category, and species present in The Narrows region, November 2014 (see Figure 11; all values mean ± standard error).

Meadow ID	Biomass (g DW m^{-2})	Area ± R (ha)	Community Type	Landscape Category	Species Present
1	1.61 ± 1.58	3 ± 1	Light *Z. muelleri* subsp. *capricorni*	Aggregated patches	*Z. muelleri* subsp. *capricorni*
10	1.14 ± 0.36	4 ± 1	Moderate *H. ovalis* with *Z. muelleri* subsp. *capricorni*	Aggregated patches	*H. ovalis* with *Z. muelleri* subsp. *capricorni*
13	n.a.	27 ± 13	Light *H. decipiens*	Isolated patches	*H. decipiens*
17	2.85 ± 1.02	27 ± 4	Light *Z. muelleri* subsp. *capricorni*/*H. ovalis*	Aggregated patches	*Z. muelleri* subsp. *capricorni, H. ovalis*
19	0.6 ± 0.2	56 ± 9	Light *Z. muelleri* subsp. *capricorni* with *H. ovalis*	Aggregated patches	*Z. muelleri* subsp. *capricorni, H. ovalis*
21	0.5 ± 0.1	131 ± 15	Light *Z. muelleri* subsp. *capricorni* with mixed species	Aggregated patches	*Z. muelleri* subsp. *capricorni, H. ovalis, H. decipiens*
22	0.44 ± 0.31	27 ± 8	Light *H. decipiens* with mixed species	Aggregated patches	*H. decipiens, Z. muelleri* subsp. *capricorni, H. ovalis*
23	0.23 ± 0.12	61 ± 21	Light *Z. muelleri* subsp. *capricorni* with mixed species	Aggregated patches	*Z. muelleri* subsp. *capricorni, H. ovalis, H. decipiens*
28	0.52 ± 0.26	7 ± 2	Light *H. ovalis/ Z. muelleri* subsp. *capricorni*	Isolated patches	*H. ovalis, Z. muelleri* subsp. *capricorni*
29	0.84 ± 0.33	40 ± 6	Light *Z. muelleri* subsp. *capricorni* with mixed species	Aggregated patches	*Z. muelleri* subsp. *capricorni, H. ovalis, H. decipiens*
33	0.59 ± 0.19	66 ± 22	Light *H. ovalis* with *Z. muelleri* subsp. *capricorni*	Aggregated patches	*H. ovalis. Z. muelleri* subsp. *capricorni*
34	0.02 ± 0.01	14 ± 3	Light *H. ovalis*	Aggregated patches	*H. ovalis*
150	0.50	1 ± 0.4	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni*

n.a. - no biomass calculated.

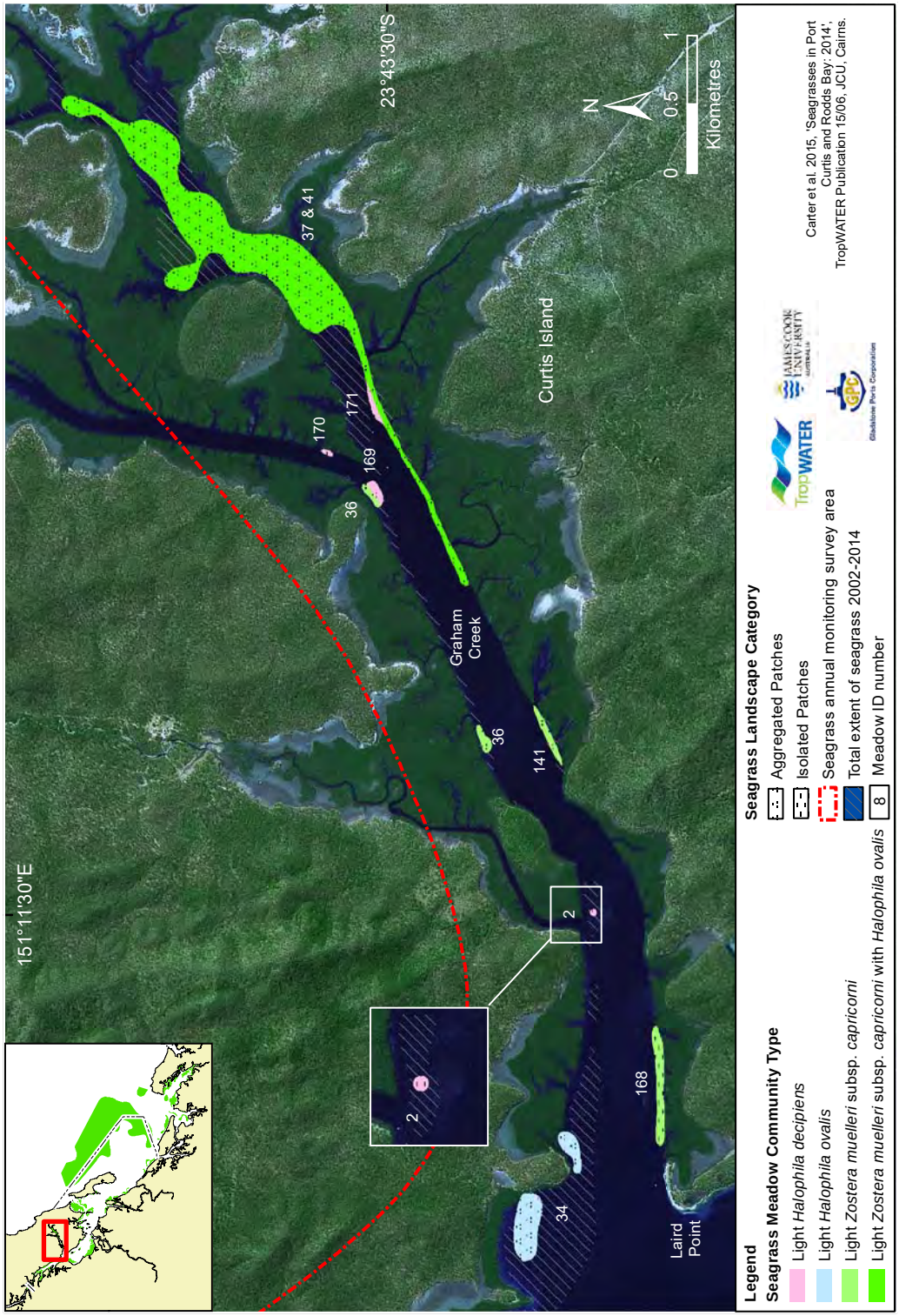


Figure 12. Seagrass distribution and community types in Graham Creek region, November 2014.

Table 6. Seagrass community type, seagrass landscape category, and species present in Graham region, November 2014 (see Figure 12; all values mean ± standard error).

Meadow ID	Biomass (g DW m^{-2})	Area ± R (ha)	Community Type	Landscape Category	Species Present
2	n.a.	0.2 ± 0.1	Light *H. decipiens*	Isolated patches	*H. decipiens*
36	0.80 ± 0.66	2 ± 1	Light *Z. muelleri* subsp. *capricorni*	Aggregated patches	*Z. muelleri* subsp. *capricorni*
37 and 41	0.56 ± 0.19	75 ± 21	Light *Z. muelleri* subsp. *capricorni* with *H. ovalis*	Aggregated patches	*Z. muelleri* subsp. *capricorni*, *H. ovalis*
141	0.37 ± 0.34	2 ± 1	Light *Z. muelleri* subsp. *capricorni*	Aggregated patches	*Z. muelleri* subsp. *capricorni*
168	0.02	1 ± 0.5	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni*
169	n.a.	1 ± 0.5	Light *H. decipiens*	Aggregated patches	*H. decipiens*
170	n.a.	0.30 ± 0.25	Light *H. decipiens*	Isolated patches	*H. decipiens*
171	n.a.	1 ± 0.8	Light *H. decipiens*	Isolated patches	*H. decipiens*

n.a. - no biomass calculated.

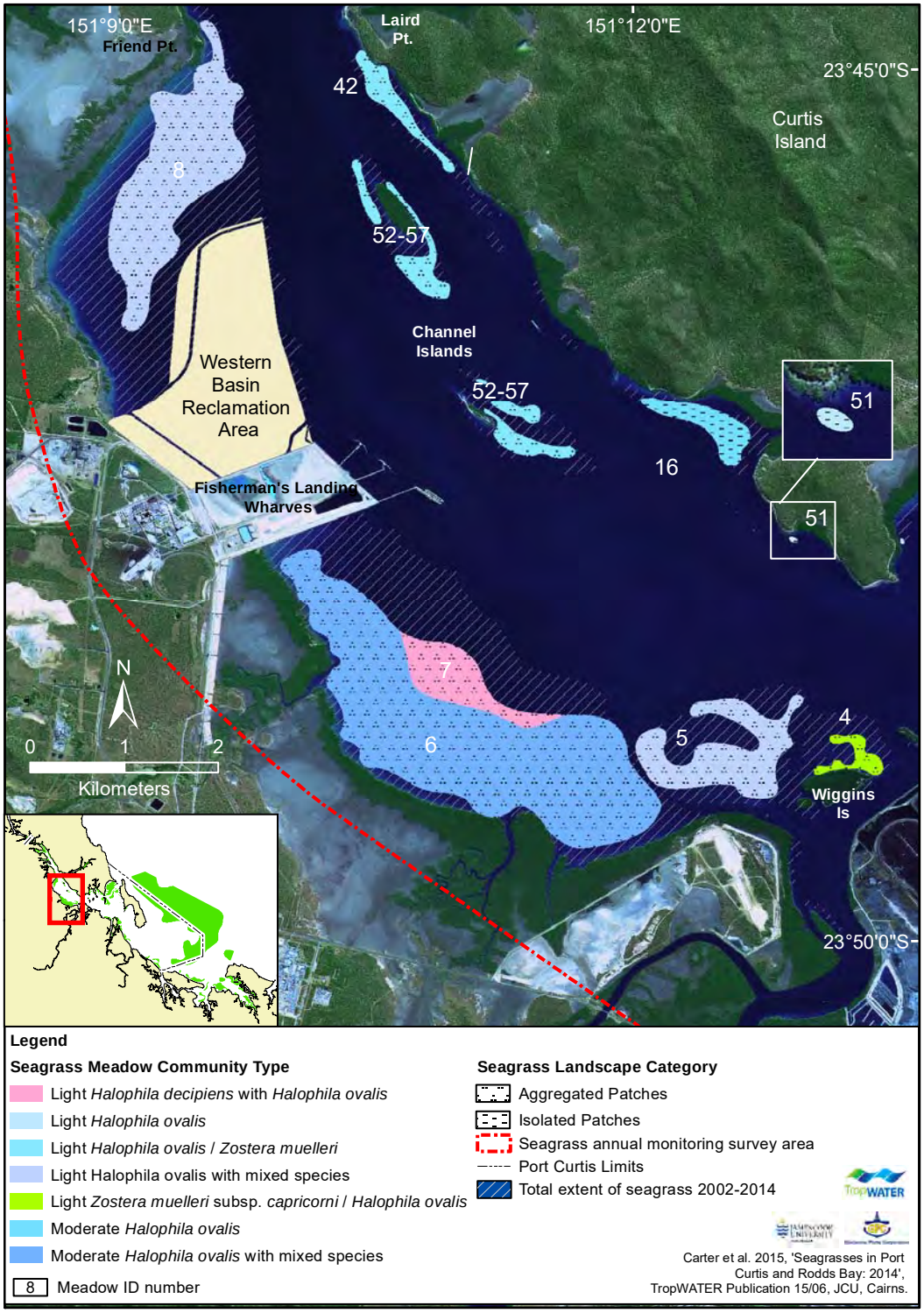


Figure 13. Seagrass distribution and community types at Fisherman's Landing (Western Basin region), November 2014.

Table 7. Seagrass community type, seagrass landscape category, and species present at Fisherman's Landing (Western Basin region), November 2014 (see Figure 13; all values mean ± standard error).

Meadow ID	Biomass (g DW m^{-2})	Area ± R (ha)	Community Type	Landscape Category	Species Present
4	1.82 ± 0.49	12 ± 3	Light *Z. muelleri* subsp. *capricorni* /*H.* ovalis	Aggregated patches	*Z. muelleri* subsp. *capricorni, H. ovalis*
5	0.98 ± 0.24	85 ± 14	Light *H. ovalis* with mixed species	Aggregated patches	*H. ovalis, Z. muelleri* subsp. *capricorni*, *H. uninervis* (thin)
6	1.22 ± 0.14	402 ± 24	Moderate *H. ovalis* with *Z. muelleri* subsp. *capricorni*	Aggregated patches	*H. ovalis, Z. muelleri* subsp. *capricorni*, *H. decipiens*
7	0.52	52 ± 13	Light *H. decipiens* with *H. ovalis*	Aggregated patches	*H. decipiens, H. ovalis*
8	0.71 ± 0.14	180 ± 17	Light *H. ovalis* with mixed species	Aggregated patches	*H. ovalis, Z. muelleri* subsp. *capricorni*, *H. decipiens*
16	0.60 ± 0.8	27 ± 3	Moderate *H. ovalis*	Isolated patches	*H. ovalis*
42	0.79 ± 0.63	21 ± 4	Light *H. ovalis/ Z. muelleri* subsp. *capricorni*	Aggregated patches	*H. ovalis, Z. muelleri* subsp. *capricorni*
51	0.29	0.4 ± 0.3	Light *H. ovalis*	Isolated patches	*H. ovalis*
52-57	0.38 ± 0.09	44 ± 9	Light *H. ovalis/ Z. muelleri* subsp. *capricorni*	Aggregated patches	*Z. muelleri* subsp. *capricorni*, *H. ovalis, H. decipiens*

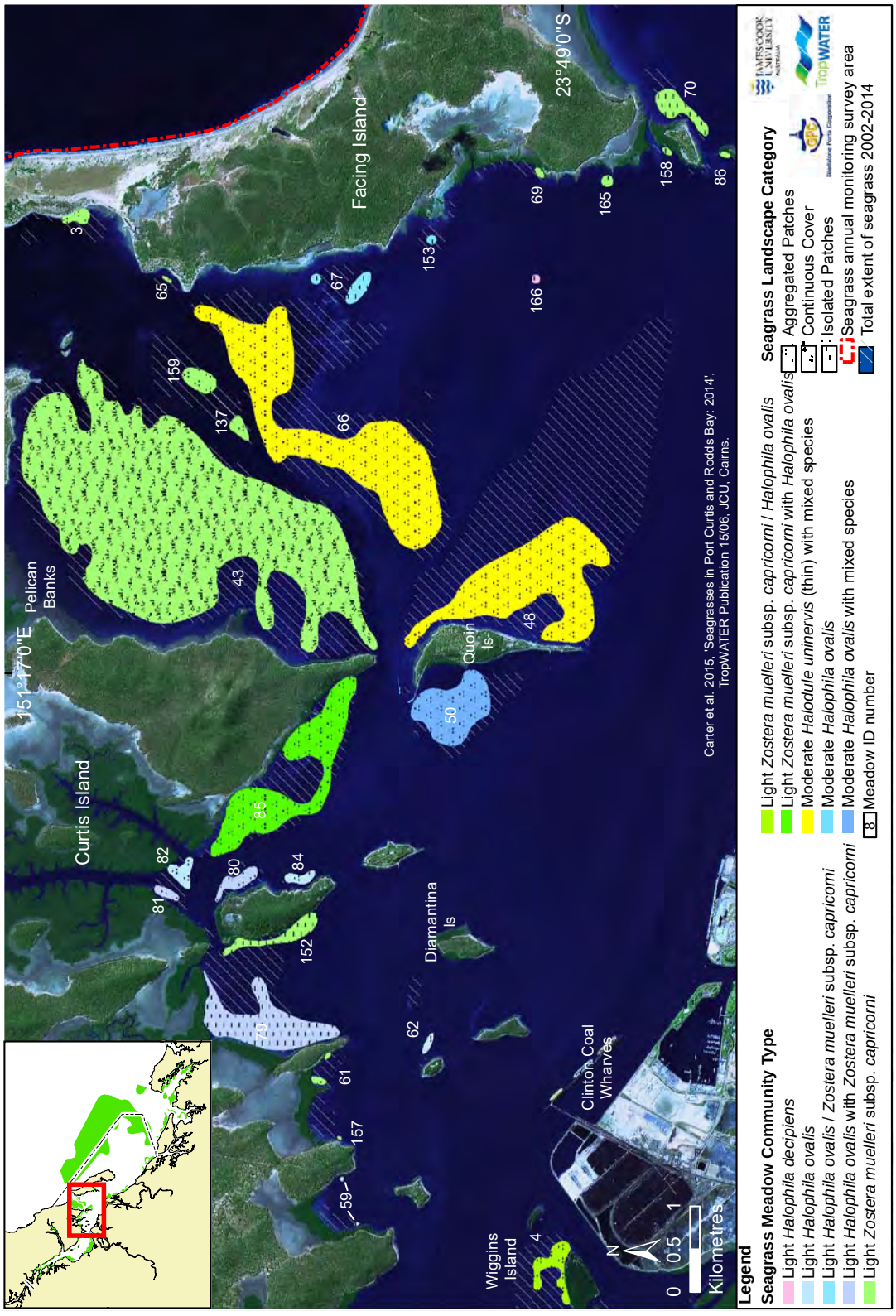


Figure 14. Seagrass distribution and community types in the Inner and Middle Harbour regions, November 2014.

Table 8. Seagrass community type, seagrass landscape category, and species present in the Inner Harbour region (west of Quoin Island), November 2014 (see Figure 14; all values mean ± standard error).

Meadow ID	Biomass (g DW m^{-2})	Area ± R (ha)	Community Type	Landscape Category	Species Present
50	1.73 ± 0.44	45 ± 6	Moderate *H. ovalis* with mixed species	Aggregated patches	*H. ovalis, H. decipiens, H. spinulosa*
59	0.3 ± 0.2	0.4 ± 0.2	Light *H. ovalis*	Isolated patches	*H. ovalis*
61	1.30 ± 1.27	2 ± 1	Light *Z. muelleri* subsp. *capricorni*	Aggregated patches	*Z. muelleri* subsp. *capricorni, H. ovalis*
62	0.79 ± 0.76	2 ± 1	Light *H. ovalis*	Aggregated patches	*H. ovalis*
79	0.10 ± 0.05	32 ± 10	Light *H. ovalis* with *Z. muelleri* subsp. *capricorni*	Isolated patches	*H. ovalis, Z. muelleri* subsp. *capricorni*
80	0.05 ± 0.02	6 ± 3	Light *H. ovalis* with *Z. muelleri* subsp. *capricorni*	Isolated patches	*H. ovalis, Z. muelleri* subsp. *capricorni*
81	0.01	3 ± 2	Light *H. ovalis /Z. muelleri* subsp. *capricorni*	Isolated patches	*H. ovalis, Z. muelleri* subsp. *capricorni*
82	0.02 ± 0.01	6 ± 2	Light *H. ovalis*	Aggregated patches	*H. ovalis*
84	0.22	3 ± 2	Light *H. ovalis*	Aggregated patches	*H. ovalis*
85	0.66 ± 0.42	86 ± 14	Light *Z. muelleri* subsp. *capricorni* with *H. ovalis*	Aggregated patches	*Z. muelleri* subsp. *capricorni*, *H. ovalis*
152	0.07 ± 0.06	12 ± 5	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni*
157	n.a.	0.2 ± 0.1	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni*

n.a. - no biomass calculated.

Table 9. Seagrass community type, seagrass landscape category, and species present in the Mid Harbour region (east of Quoin Island), November 2014 (see Figure 14; all values mean ± standard error).

Meadow ID	Biomass (g DW m^{-2})	Area ± R (ha)	Community Type	Landscape Category	Species Present
3	0.35 ± 0.16	4 ± 1	Light *Z. muelleri* subsp. *capricorni*	Aggregated patches	*H. ovalis*, *Z. muelleri* subsp. *capricorni*
43	12.31 ± 1.36	595 ± 33	Light *Z. muelleri* subsp. *capricorni*	Continuous cover	*Z. muelleri* subsp. *capricorni*, *H. ovalis*
48	1.21 ± 0.26	135 ± 39	Moderate *H. uninervis* (thin) with mixed species	Aggregated patches	*H. uninervis* (wide and thin), *Z. muelleri* subsp. *capricorni*, *H. ovalis*
65	n.a.	0.4 ± 0.3	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni*
66	2.08 ± 0.35	200 ± 52	Moderate *H. uninervis* (thin) with mixed species	Aggregated patches	*H. uninervis* (wide and thin), *Z. muelleri* subsp. *capricorni*, *H. ovalis*, *H. spinulosa*
67	0.88 ± 0.44	7 ± 3	Light *H. ovalis/Z. muelleri* subsp. *capricorni*	Isolated patches	*H. ovalis*, *Z. muelleri* subsp. *capricorni*
69	1.02	1 ± 0.6	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni*
70	1.24 ± 0.38	11 ± 4	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni*
86	2.78	1 ± 0.4	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni*
137	8.80 ± 4.99	4 ± 2	Light *Z. muelleri* subsp. *capricorni*	Aggregated patches	*Z. muelleri* subsp. *capricorni*, *H. decipiens*
153	2.83 ± 0	1 ± 0.6	Moderate *H. ovalis*	Isolated patches	*H. ovalis*
158	1.53	1 ± 0.6	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni*
159	7.20 ± 6.72	8 ± 2	Light *Z. muelleri* subsp. *capricorni*	Aggregated patches	*Z. muelleri* subsp. *capricorni*, *H. decipiens*
165	n.a.	1.3 ± 1	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni*
166	0.23 ± 0.15	0.8 ± 0.6	Light *H. decipiens*	Isolated patches	*H. decipiens*

n.a. - no biomass calculated.

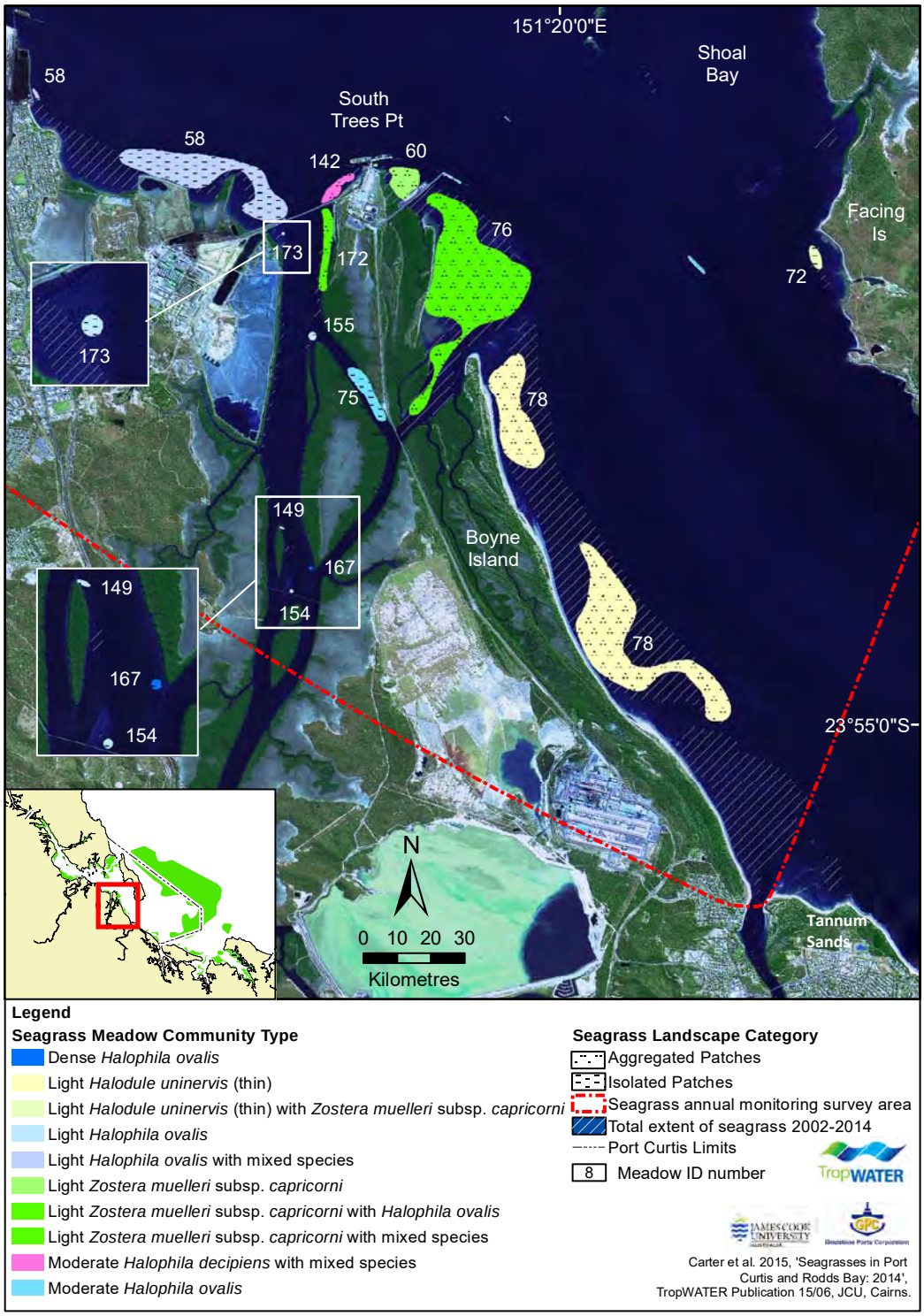


Figure 15. Seagrass distribution and community types at South Trees (Inner and Mid Harbour region), November 2014.

Table 10. Seagrass community type, seagrass landscape category, and species present at South Trees (Inner and Mid Harbour region), November 2014 (see Figure 15; all values mean ± standard error).

Meadow ID	Biomass (g DW m^{-2})	Area ± R (ha)	Community Type	Landscape Category	Species Present
58	0.47 ± 0.14	64 ± 12	Light *H. ovalis* with mixed species	Isolated patches	*H. ovalis*, *H. unin*ervis (thin), *Z. muelleri* subsp. *capricorni*
60	1.28 ± 0.35	11 ± 2	Light *Z. muelleri* subsp. *capricorni*	Aggregated patches	*Z. muelleri* subsp. *capricorni*
70	1.24 ± 0.38	11 ± 4	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni*
72	0.03 ± 0.01	3 ± 2	Light *H. uninervis* (thin) with *Z. muelleri* subsp. *capricorni*	Isolated patches	*H. uninervis* (thin), *Z. muelleri* subsp. *capricorni*
75	1.12 ± 0.11	10 ± 2	Moderate *H. ovalis*	Aggregated patches	*H. ovalis*
76	1.84 ± 0.46	138 ± 8	Light *Z. muelleri* subsp. *capricorni* with mixed species	Aggregated patches	*Z. muelleri* subsp. *capricorni*, *H. uninervis* (thin), *H. ovalis*
78	0.98 ± 0.32	191 ± 59	Light *H. uninervis* (thin)	Aggregated patches	*H. uninervis* (thin)
142	1.17 ± 0.66	5 ± 1	Moderate *H. decipiens* with mixed species	Aggregated patches	*H. decipiens*, *Z. muelleri* subsp. *capricorni*, *H. ovalis*
149	0.01	0.14 ± 0.10	Light *H. ovalis*	Isolated patches	*H. ovalis*
154	1.0	0.2 ± 0.1	Light *H. ovalis*	Isolated patches	*H. ovalis*
155	0.01	0.8 ± 0.4	Light *H. ovalis*	Isolated patches	*H. ovalis*
167	5.87 ± 0.67	0.2 ± 0.1	Dense *H. ovalis*	Isolated patches	*H. ovalis*
173	0.57 ± 0.40	0.2 ± 0.1	Light *H. ovalis*	Isolated patches	*H. ovalis*, *Z. muelleri* subsp. *capricorni*

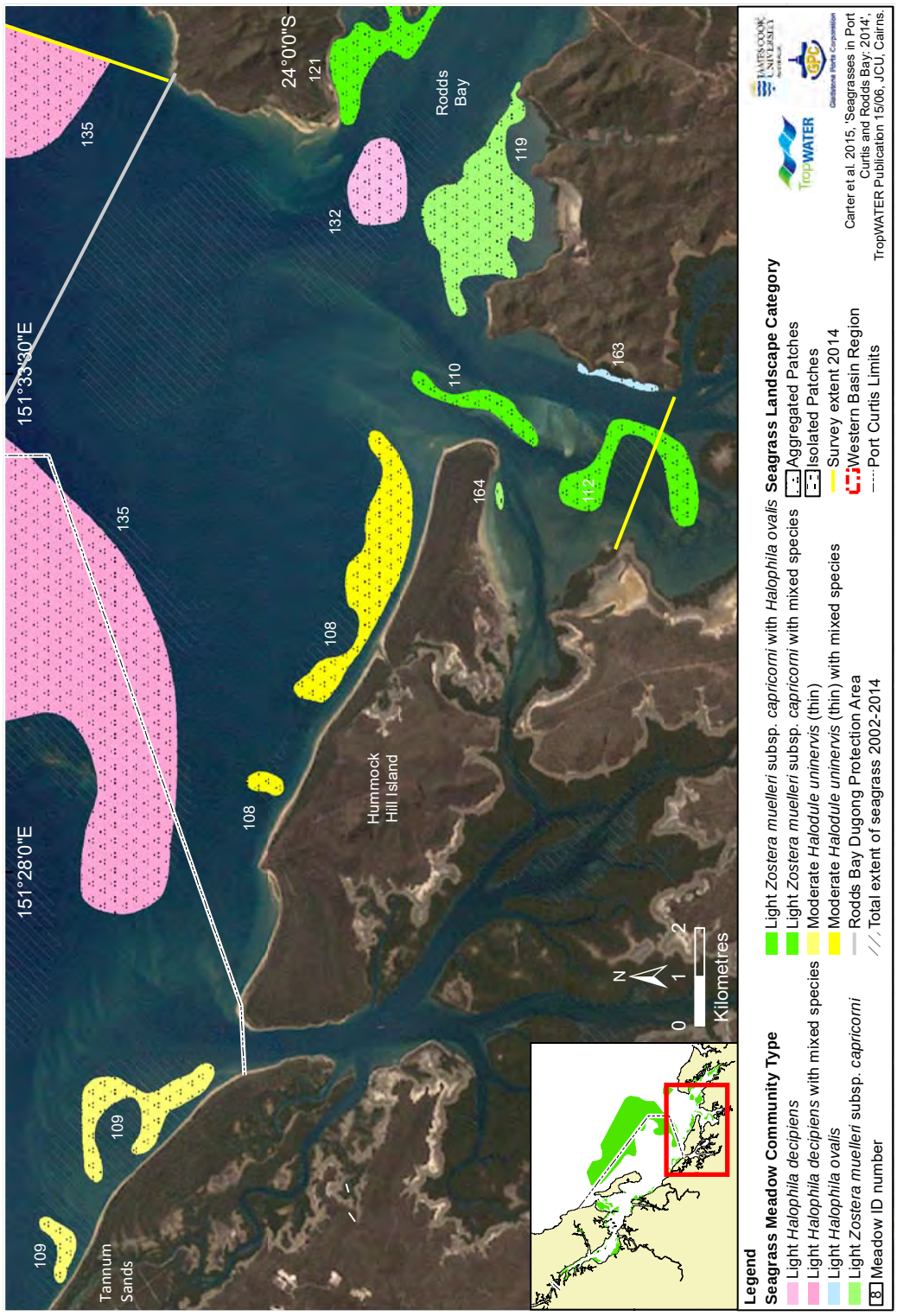


Figure 16. Seagrass distribution and community types at Tannum Sands (Outer Harbour region), November 2014.

Table 11. Seagrass community type, seagrass landscape category, and species present at Tannum Sands (Outer Harbour region), November 2014 (see Figure 16; all values mean ± standard error).

Meadow ID	Biomass (g DW m^{-2})	Area ± R (ha)	Community Type	Landscape Category	Species Present
108	2.87 ± 0.53	380 ± 77	Moderate *H. uninervis* (thin) with mixed species	Aggregated patches	*H. uninervis* (thin & wide), *Z. muelleri* subsp. *capricorni, H. decipiens*
109	3.30 ± 0.57	240 ± 73	Moderate *H. uninervis* (thin)	Aggregated patches	*H. uninervis* (thin), *Z. muelleri* subsp. *capricorni*
110	2.01 ± 1.29	87 ± 13	Light *Z. muelleri* subsp. *capricorni* with mixed species	Aggregated patches	*Z. muelleri* subsp. *capricorni, H. uninervis* (thin), *H. decipiens, H. ovalis*
112	1.17 ± 0.38	205 ± 21	Light *Z. muelleri* subsp. *capricorni* with *H. ovalis*	Aggregated patches	*Z. muelleri* subsp. *capricorni, H. ovalis*
163	0.75 ± 0.52	463 ± 56	Light *H. ovalis*	Aggregated patches	*H. ovalis, Z. muelleri* subsp. *capricorni*
164	0.01	6 ± 1	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni*

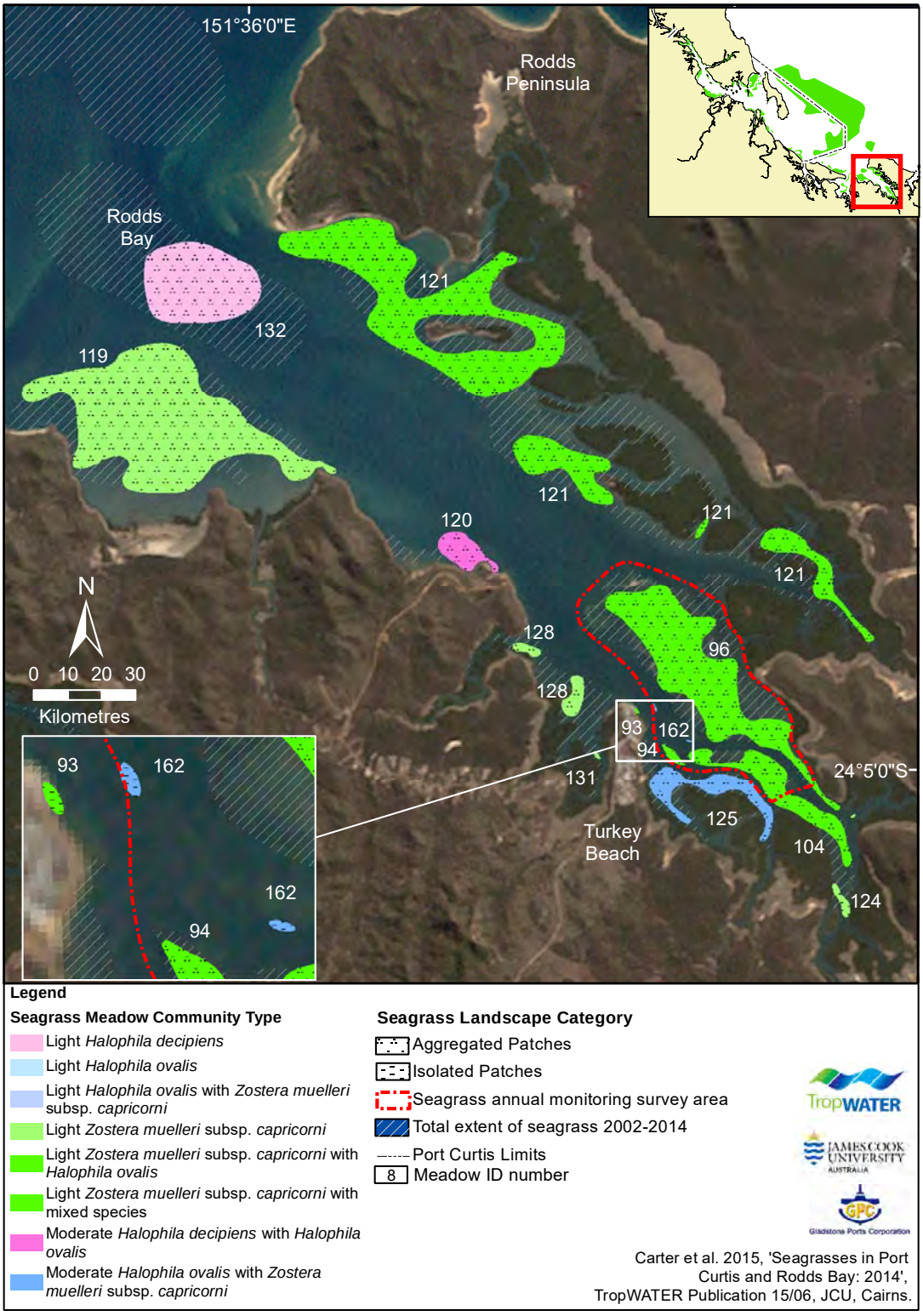


Figure 17. Seagrass distribution and community types in the Rodds Bay region, November 2014.

Table 12. Seagrass community type, seagrass landscape category, and species present in the Rodds Bay region, November 2014 (see Figure 17; all values mean ± standard error).

Meadow ID	Biomass (g DW m^{-2})	Area ± R (ha)	Community Type	Landscape Category	Species Present
93	15.05	0.4 ± 0.3	Light *Z. muelleri* subsp. *capricorni* with *H. ovalis*	Isolated patches	*Z. muelleri* subsp. *capricorni, H. ovalis*
94	2.13 ± 0.93	4 ± 1	Light *Z. muelleri* subsp. *capricorni* with *H. ovalis*	Aggregated patches	*Z. muelleri* subsp. *capricorni, H. ovalis*
96	1.55 ± 0.46	235 ± 23	Light *Z. muelleri* subsp. *capricorni/ H. ovalis*	Aggregated patches	*Z. muelleri* subsp. *capricorni, H. ovalis*
104	3.24 ± 0.91	70 ± 14	Light *Z. muelleri* subsp. *capricorni* with mixed species	Aggregated patches	*Z. muelleri* subsp. *capricorni, H. ovalis, H. decipiens*
119	5.88 ± 1.44	450 ± 38	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni, H. ovalis, H. decipiens*
120	1.91 ± 1.07	27 ± 5	Moderate *H. decipiens* with *H. ovalis*	Aggregated patches	*H. ovalis, H. decipiens*
121	2.71 ± 1.44	463 ± 56	Light *Z. muelleri* subsp. *capricorni* with mixed species	Aggregated patches	*Z. muelleri* subsp. *capricorni, H. decipiens, H. ovalis*
125	1.31 ± 0.96	51 ± 13	Moderate *H. ovalis* with *Z. muelleri* subsp. *capricorni*	Isolated patches	*H. ovalis, Z. muelleri* subsp. *capricorni*
128	0.18	17 ± 3	Light *Z. muelleri* subsp. *capricorni*	Aggregated patches	*Z. muelleri* subsp. *capricorni, H. ovalis*
124	4.23 ± 3.65	5 ± 3	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni, H. ovalis*
131	1.37	0.6 ± 0.4	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni*
161	0.45	0.2 ± 0.1	Light *Z. muelleri* subsp. *capricorni*	Isolated patches	*Z. muelleri* subsp. *capricorni*
162	1.69	0.7 ± 0.6	Moderate *H. ovalis* with *Z. muelleri* subsp. *capricorni*	Isolated patches	*H. ovalis, Z. muelleri* subsp. *capricorni*

3.3 Deepwater seagrasses

A total of 121 habitat characterisation sites were surveyed in November 2014 in the deepwater areas (>6 m dbMSL) of Port Curtis and the Rodds Bay DPA (Figure 18). Seagrass was present at 31% of sites and a total of 26,158 ± 8,759 ha of deepwater seagrass meadows were mapped across two meadows (Figure 18; Table 13).

Seagrasses occurred offshore from Facing Island and extended over a large portion of the survey area adjacent to the port and the Rodds Bay DPA. Deepwater seagrasses were also identified around Seal Rocks near the port limits (Figure 18). No seagrass was found in deep areas (> 6 m dbMSL) within the Mid Harbour region (on the inshore side of Facing Island).

The large offshore deepwater meadow (Meadow 135; Figure 18) contained aggregated patches of seagrass. *Halophila decipiens* was the dominant species in meadow above-ground biomass *(*present at 88% of sites) (Table 13). Above-ground biomass at individual sites ranged from 0 to 3.2 g DW m^{-2}. Seagrasses were recorded up to 26 m dbMSL. Seagrass is likely to extend beyond the survey boundary.

The Rodds Bay deepwater meadow (Meadow 132) contained aggregated patches of *H. decipiens* with very low above-ground biomass of 0.02 g DW m^{-2} (Table 13). This meadow was recorded at 11 m dbMSL.

Table 13. Seagrass community type, seagrass landscape category, and species present in deepwater areas of Port Curtis and the Rodds Bay DPA and adjacent offshore areas, November 2014 (see Figure 18; all values mean ± standard error).

Meadow ID	Biomass (g DW m^{-2})	Area ± R (ha)	Community Type	Landscape Category	Species Present
132	0.02	164 ± 51	Light *H. decipiens*	Aggregated patches	*H. decipiens*
135	0.37 ± 0.11	25,995 ± 8,708	Light *H. decipiens* with mixed species	Aggregated patches	*H. decipiens, H. ovalis, H. spinulosa*

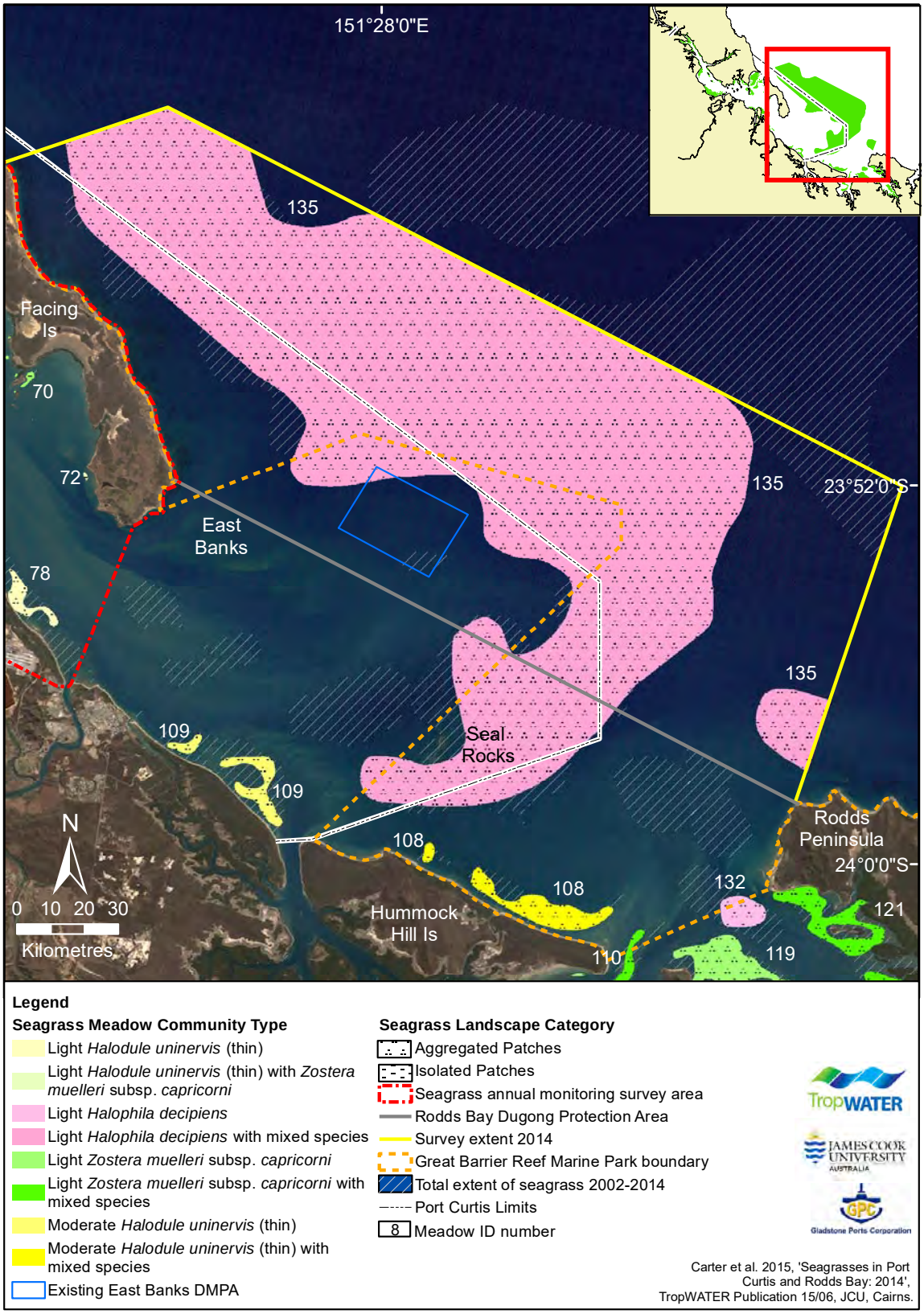


Figure 18. Seagrass distribution and community types for deepwater seagrass communities in Port Curtis, the Rodds Bay DPA and adjacent offshore areas, November 2014.

3.4 Comparison with previous baseline assessments

The baseline surveys in November 2013 and 2014 covered a much larger area than the 2009 assessment, with extended offshore areas outside of the port limits surveyed and large seagrass meadows detected (Figure 20). Although this extended offshore survey area has been mapped, only the area common among the three surveys will be used to draw comparisons.

A total seagrass area of 12,700 ha of seagrass habitat was mapped during the baseline survey in 2014 (Figures 19 - 21). This included coastal and deepwater seagrass within the port, Rodds Bay DPA and offshore areas. Total meadow area had increased by over 8,000 ha since 2013. This follows a 63% (~7,700 ha) reduction in seagrass area from 2009 to 2013 (Figures 19 - 21).

The dramatic changes in meadow area within the port limits were largely driven by changes in the deepwater meadows. Approximately half of the reduction in seagrass area that occurred in 2013 was due to a reduction in deepwater meadow area (Figure 20). Deepwater meadows underwent a net loss of ~3,600 ha (~75%) from 2009 to 2013, then expanded by over 6,100 ha between 2013 and 2014 (Figure 20). Increases in total meadow area between 2013 and 2014 were, to a lesser extent, due to increases in meadow area of coastal intertidal and shallow subtidal meadows. In Port Curtis, significant increases in meadow area were recorded in The Narrows and Graham Creek, and Fisherman's Landing (Western Basin region) (Figure 19).

In the Outer Harbour region the distribution of seagrass meadows in 2014 was similar to 2009, with coastal meadows less fragmented than in 2013. Seagrass meadows expanded off Wild Cattle Island and Hummock Hill Island and at the mouth of Rodds Bay, although not to the peak meadow area recorded in 2009 (Figure 20).

In Rodds Bay a dramatic reduction in seagrass meadow area from over 1,100 ha in 2009, to just 150 ha by 2013, had left only a few remnant patches of seagrass (Figure 21). In November 2014 seagrass area had increased to over 870 ha since 2013, and included expanded and new meadows along both sides of Rodds Bay and adjacent to Turkey Beach (Figure 21).

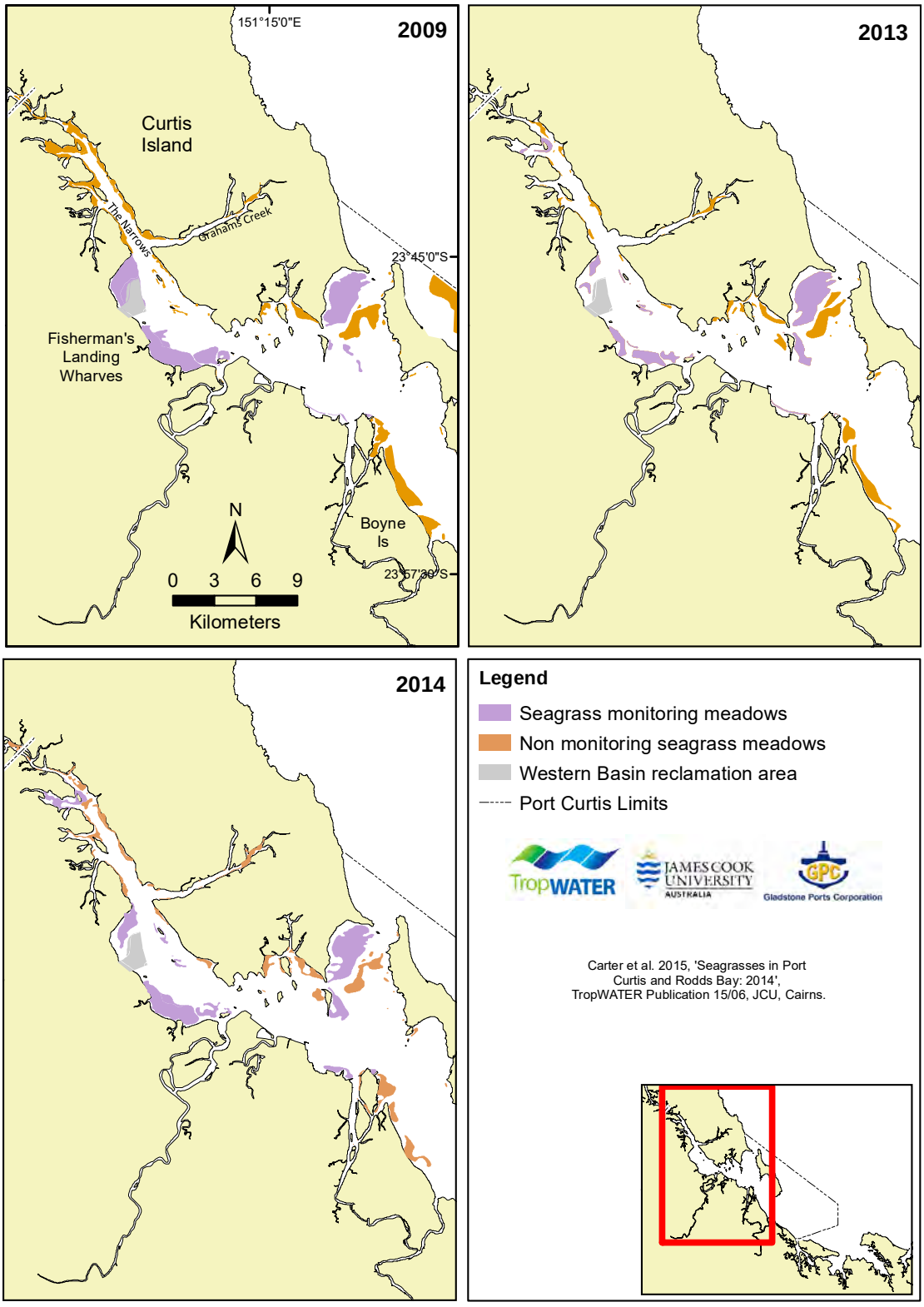


Figure 19. Seagrass distribution in The Narrows, Grahams Creek, Western Basin, Inner Harbour and Mid Harbour regions of Port Curtis, 2009, 2013 and 2014.

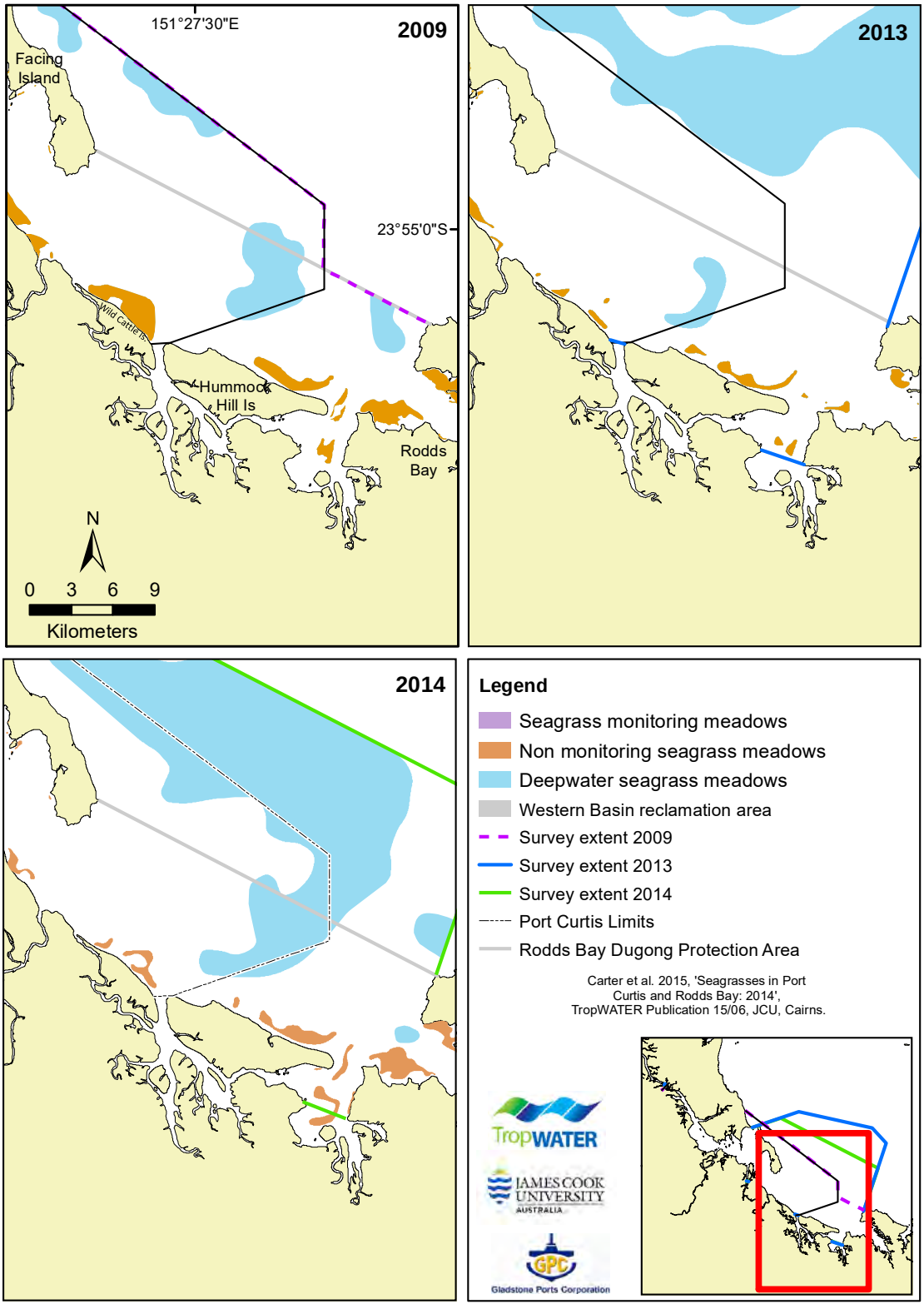


Figure 20. Seagrass distribution in the Outer Harbour region of Port Curtis, 2009, 2013 and 2014.

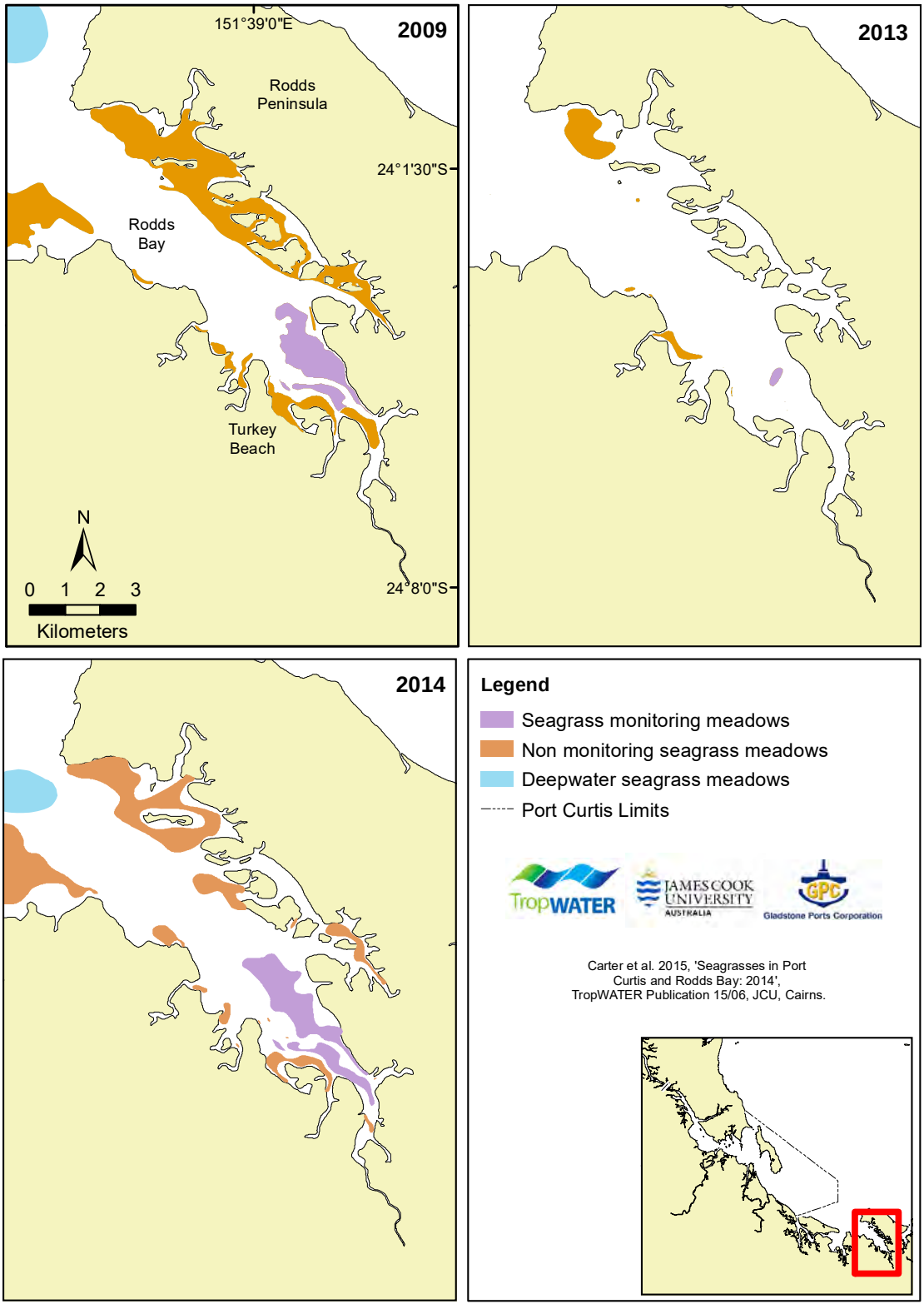


Figure 21. Seagrass distribution in Rodds Bay, 2009, 2013 and 2014.

3.5 Seasonal seagrass distribution in the seagrass annual monitoring survey area, excluding Rodds Bay, 2009 - 2014

Seagrass distribution in the annual monitoring survey area demonstrated strong seasonal trends in meadow area. In 2014 the distribution of coastal seagrass, excluding Rodds Bay, during both the senescent (June) and growing season (November) surveys was greater than in 2013, and similar to peak total meadow area recorded during the November 2012 growing season (Figure 22). Total meadow area in November 2014 was approximately 900 ha below the peak of 4 000 ha mapped in November 2009 (Figure 22).

Considerable reductions in seagrass cover and biomass occur during each senescent season. The seasonal reduction of 57% in seagrass area between November 2013 and June 2014 was one of the smallest yet (compared with 74% in 2013). This reduced seasonal decline may have placed these seagrass meadows in a more resilient state for 2015. Seasonal fluctuations in seagrass meadow area occur to a greater extent in areas such as The Narrows and Fisherman's Landing (The Narrows region), Wiggins Island (Western Basin region), and at Boyne Island (Mid Harbour region) (Figure 23). The Pelican Banks meadow (meadow 43) is generally more stable, with biomass and meadow area remaining relatively stable throughout between seasons (Figure 23). Increases in meadow area during the 2014 growing season were greater than most years, with an increase of 1,976 ha (~67%) of seagrass area from June to November.

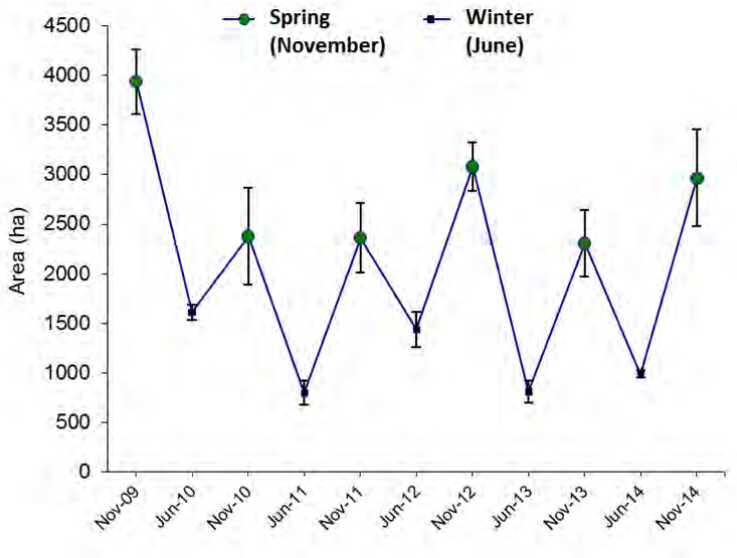


Figure 22. Seagrass meadow area (hectares) in the Gladstone seagrass annual monitoring survey area during November (spring growing season) and June (winter senescent season), 2009 to 2014.

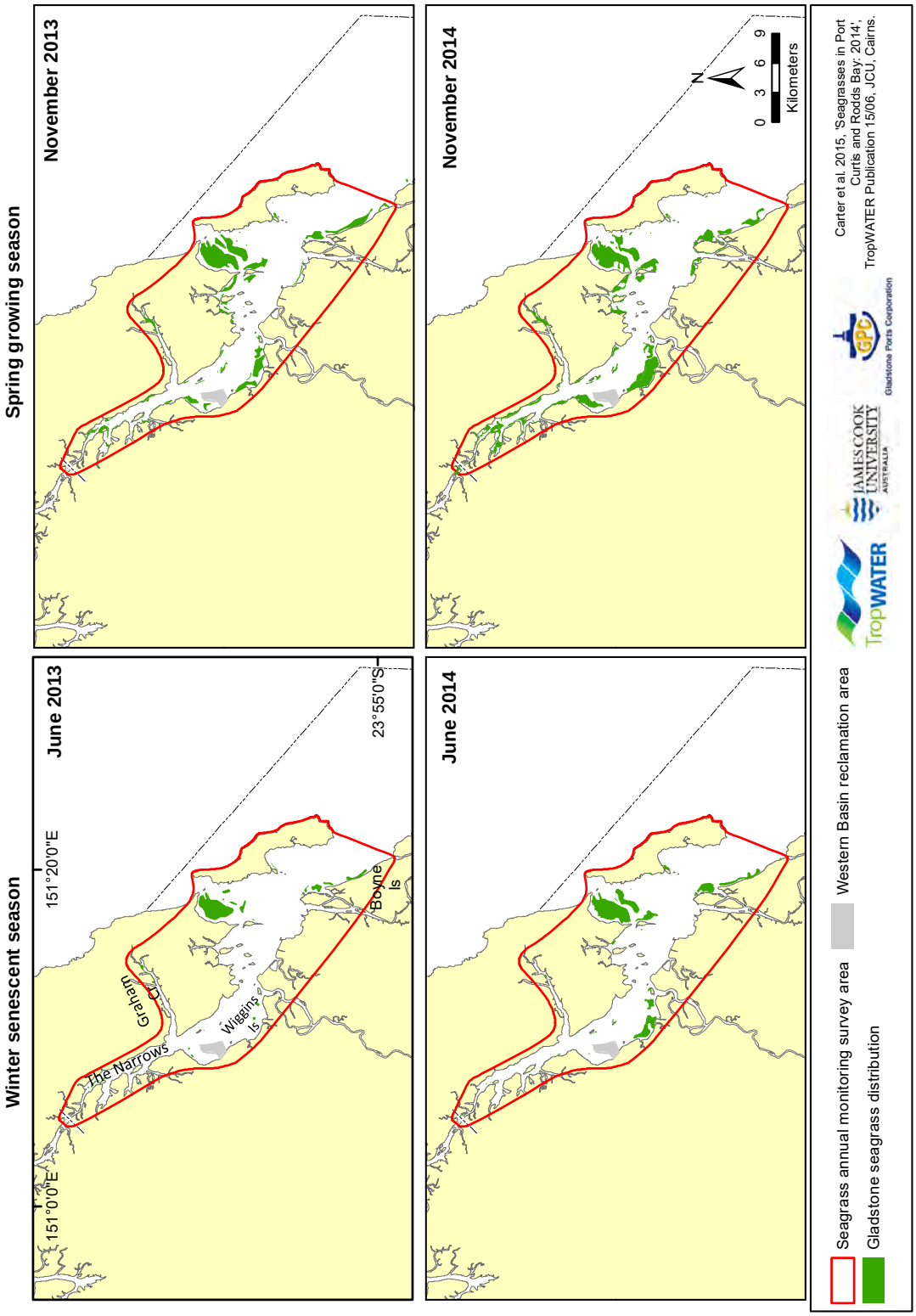


Figure 23. Seagrass meadow distribution in November (spring growing season) and June (senescent winter season) in the seagrass annual monitoring survey area, 2013 - 2014.

3.6 Comparison at annual monitoring meadows

For the long-term monitoring meadows that have been assessed since 2002, the November 2014 survey detected improvements in the condition of seagrass meadows in the Mid Harbour and Rodds Bay, and varied conditions in The Narrows, Western Basin and Inner Harbour meadows. Thirteen of the fourteen monitoring meadows expanded in area, but biomass was more variable. Within The Narrows and Western Basin regions seagrass condition improved in two monitoring meadows due to area increases, one meadow remained in "very poor" condition due to low biomass and in four meadows the seagrass was in worse condition than in 2013 due to biomass decreases. In the Mid Harbour region two monitoring meadows remained in "fair" condition, while at South Trees in the Inner Harbour one meadow increased from "good" to "very good" due to area increases, and one meadow increased from "poor" to "fair" due to biomass increases. In Rodds Bay two monitoring meadows moved from "very poor" to "poor" condition, and one moved from "poor" to "fair" condition, due to biomass increases (Figure 4).

3.6.1 The Narrows, Western Basin and Inner Harbour monitoring meadows

Most monitoring meadows in The Narrows (meadow 21), Western Basin (meadows 4-8, 52-57) and Inner Harbour (meadow 58) regions had low levels of biomass during the earlier years of the program, particularly around 2004 and 2005. Most meadows then experienced peak seagrass biomass between 2006 and 2008, followed by declines in 2009 or 2010 that brought seagrass biomass well below the long-term average (Figures 24- 30). Biomass has varied significantly among years for every meadow surveyed ($p < 0.001$ for all meadows; Appendix 2).

Peak levels of biomass recorded in 2006 – 2008 have not occurred again in the Western Basin meadows over the last six years. Biomass has remained similar in the intertidal *H. ovalis* dominated meadows 5, 6 and 8 since significant declines in 2009 ($p < 0.001$), 2010 ($p < 0.001$) and 2008 ($p < 0.01$), respectively (Figures 25, 26 and 28). Small declines in meadow 5 pushed this meadow beyond the threshold from "good" to "fair" condition in 2014 (Figure 25). Biomass in the intertidal *H. ovalis* dominated meadow 52-57 at the Channel Islands increased significantly between 2010 and 2012 ($p < 0.05$) and again in 2013 ($p < 0.001$) following an almost complete loss of seagrass in 2010 – 2011, but decreased significantly between 2013 and 2014 ($p < 0.001$). Biomass declines in meadow 52-57 in 2014 breached a condition threshold and the meadow went from being classed as being in "very good" to "good" condition (Figure 30). A similar pattern was seen in the subtidal *H. decipiens* meadow 7 at South Fisherman's Landing, although the biomass declines in this meadow meant the meadow went from being classed as in "very good" condition in 2013 where biomass was the highest recorded since 2007, to "fair" condition following declines in 2014 (Figure 27). Meadows 7 and 8 are characterised as patchy meadows, with significant variation in the number of habitat characterisation sites with seagrass present among years (Appendix 2). The mean probability of observing seagrass at a site in meadow 7 was 0.54 (Chi-Square < 0.001; 95% confidence interval 0.46 to 0.61) and in meadow 8 was 0.65 (Chi-Square < 0.001; 95% confidence interval 0.60 to 0.70).

Biomass had very different patterns in the intertidal *Z. muelleri* subsp. *capricorni* dominated meadows. Meadow 21 at Black Swan (The Narrows) has experienced successive years of often significant biomass declines since monitoring began at that meadow in 2009, with this meadow now classed as being in "very poor" condition (Figure 29). In contrast, biomass in the intertidal *Z. muelleri* subsp. *capricorni* dominated meadow 4 at Wiggins Island (Western Basin region) has been at record highs since 2012, marking a dramatic increase since the almost complete collapse of this meadow biomass in 2010 (Figure 24). However, meadow 4 is also characteristically patchy, with a mean probability of 0.42 of observing seagrass at a site (Chi-Square < 0.001; 95% confidence interval 0.36 to 0.49).

Meadow area increased in all monitoring meadows in these regions between 2013 and 2014. Monitoring meadows had all previously experienced dramatic declines in meadow area in 2010 – 2011, with losses of 22% - 98% (Figures 24 – 37). Since these losses, meadow area for most monitoring meadows has remained

below the long-term average. Between 2010 and 2011, intertidal meadows in these regions reached their lowest recorded meadow area since monitoring began in 2002. By 2012 the majority of intertidal meadows had shown signs of recovery. Meadow area increased in the intertidal *H. ovalis* dominated meadows 5, 6, 8 and 52-57. In many cases the increases in meadow area breached two condition thresholds (Figures 25 - 26, 28, 30). Meadow area had very different patterns in the intertidal *Z. muelleri* subsp. *capricorni* dominated meadows. Despite record high biomass in meadow 4, meadow area has remained in "very poor" condition for four out of the last five years of monitoring (Figure 24). In contrast, the area of meadow 21 increased dramatically between 2013 and 2014 despite record low biomass (Figure 29). The South Fisherman's Landing subtidal *H. decipiens* dominated meadow 7 contained a few relatively dense seagrass patches along the waters' edge in 2013; by November 2014 meadow area had increased from 7 ha to 52 ha (Figure 27).

Species composition of several of intertidal monitoring meadows in these regions has shifted in recent years from *Z. muelleri* subsp. *capricorni* dominated communities to colonising *H. ovalis* being the dominant species (meadows 5, 6 and 8; Figures 25, 26 and 28). The shift in the dominant species for these meadows began during the 2008 – 2009 growing season and has continued slowly since. A similar increase in *H. ovalis* at the expense of *Z. muelleri* subsp. *capricorni* has occurred in meadow 21 at Black Swan. In 2014 this meadow remained a *Z. muelleri* subsp. *capricorni* dominated meadow, but this species has decreased from close to 100% of species composition in 2009 to just over 50% in 2014 (Figure 29).

On the mainland coast, biomass in the intertidal *H. ovalis* dominated meadow declined significantly in 2010 when most of the previously dominant *Z. muelleri* subsp. *capricorni* was lost ($p < 0.05$; Meadow 58 (South Trees Inlet); Figure 31). Biomass has fluctuated between 2010 and 2014 but these changes have not been significant. Meadow area also declined to extremely low levels in 2010, leaving the meadow in "very poor" condition. Dramatic changes in meadow area have followed, with an increase of almost 60 ha from 2011 to 2012, a 50% reduction in area between 2012 and 2013, and an increase between 2013 and 2014 to the largest area recorded for this meadow (64 ha) since monitoring began in 2002 (Figure 31). As with several meadows in The Narrows and Western Basin regions, species composition has shifted from a *Z. muelleri* subsp. *capricorni* dominated to a *H. ovalis* dominated meadow. Despite meadow area being in "very good" condition, below average biomass and the shift in species composition to pioneering species has resulted in the overall "poor" condition of this meadow.

3.6.2 Mid Harbour and South Trees Inlet monitoring meadows

The condition of Mid Harbour (meadows 43 and 48) and South Trees Inlet seagrass monitoring meadows were relatively stable compared with other regions, with no significant changes in meadow biomass between 2013 and 2014 ($p < 0.001$ for all meadows; Appendix 2). As with The Narrows, Western Basin and Inner Harbour meadows, the Mid Harbour meadows were characterised by low levels of seagrass biomass during the 2004 – 2005 period, peak biomass from approximately 2006 - 2008, and declines in 2009 or 2010 that brought seagrass biomass well below the long-term average through 2014 (Figures 32 - 34).

The large intertidal *Z. muelleri* subsp. *capricorni* dominated meadow at Pelican Banks (Mid Harbour region) remained in a "fair" condition due to average biomass remaining almost 40% below the long-term average (Meadow 43; Figure 32). Relatively low but stable biomass has characterised this meadow from 2011 – 2014 with significant declines between 2008 and 2012 ($p < 0.05$). Meadow area has remained relatively stable over the course of monitoring with small fluctuations around the long-term average, and has always been classed as being in "good" condition. Species composition is consistently >95% *Z. muelleri* subsp. *capricorni*.

The condition of the subtidal *H. uninervis* dominated meadow at Quoin Island (Mid Harbour region) remained "fair", with biomass and meadow area stable but greater than 40% below the long-term average (Meadow 48; Figure 33). Seagrass biomass declined significantly between 2006 and 2008 ($p < 0.001$) and again between 2008 and 2009 ($p < 0.01$); between 2010 and 2014 biomass has fluctuated but these changes have not been significant. Meadow area also declined to extremely low levels in 2009, leaving the meadow in "very poor" condition; since 2009 meadow area has generally increased (with the exception of 2011 –

2012) and meadow area is not classed as being in "fair" condition. Species composition has remained relatively stable.

Similar patterns in meadow biomass occurred in the intertidal *Z. muelleri* subsp. *capricorni* dominated meadow at South Trees Inlet, although the significant declines occurred one year earlier in 2009 (Meadow 60; Figure 34). Biomass has not changed significantly since 2009, but a small increase over a condition threshold in 2014 meant the meadow biomass moved from being classed as "poor" to "fair" condition. Dramatic increases in area were also recorded in 2014, from approximately 2 ha to 11 ha; the meadow has been this size only twice previously – in 2002 and in 2008. Species composition shifted dramatically in 2014 from a *H. uninervis* to a *Z. muelleri* subsp. *capricorni* dominated meadow; this marked a return to the species composition recorded from 2002 – 2009 (Figure 34).

3.6.3 Rodds Bay monitoring meadows

The condition of the three intertidal *Z. muelleri* subsp. *capricorni* monitoring meadows in the out of port reference region of Rodds Bay showed similar patterns to the monitoring meadows discussed above. Rodds Bay meadows were characterised by low levels of seagrass biomass around 2004, peak biomass from approximately 2007 - 2008, and significant declines in 2009 – 2010 that brought seagrass biomass well below the long-term average (all p-values <0.05; Figures 35 – 37). There were no significant changes in biomass between 2010 and 2014, although small increases in 2014 above a threshold level meant meadows 94 and 96 moved from "very poor" to "poor" condition, while meadow 104 moved from "poor" to "fair" condition (Figures 35 – 37). Meadow area in Rodds Bay declined dramatically in 2011 and since then has been highly variable, with complete crashes in meadow area in 2011 and 2013 followed by dramatic increases in 2012 and 2014. The most recent increase meant that in 2014 area in the three meadows was classed as either "very good" or "good". *Z. muelleri* subsp. *capricorni* has remained the dominant species in these meadows.

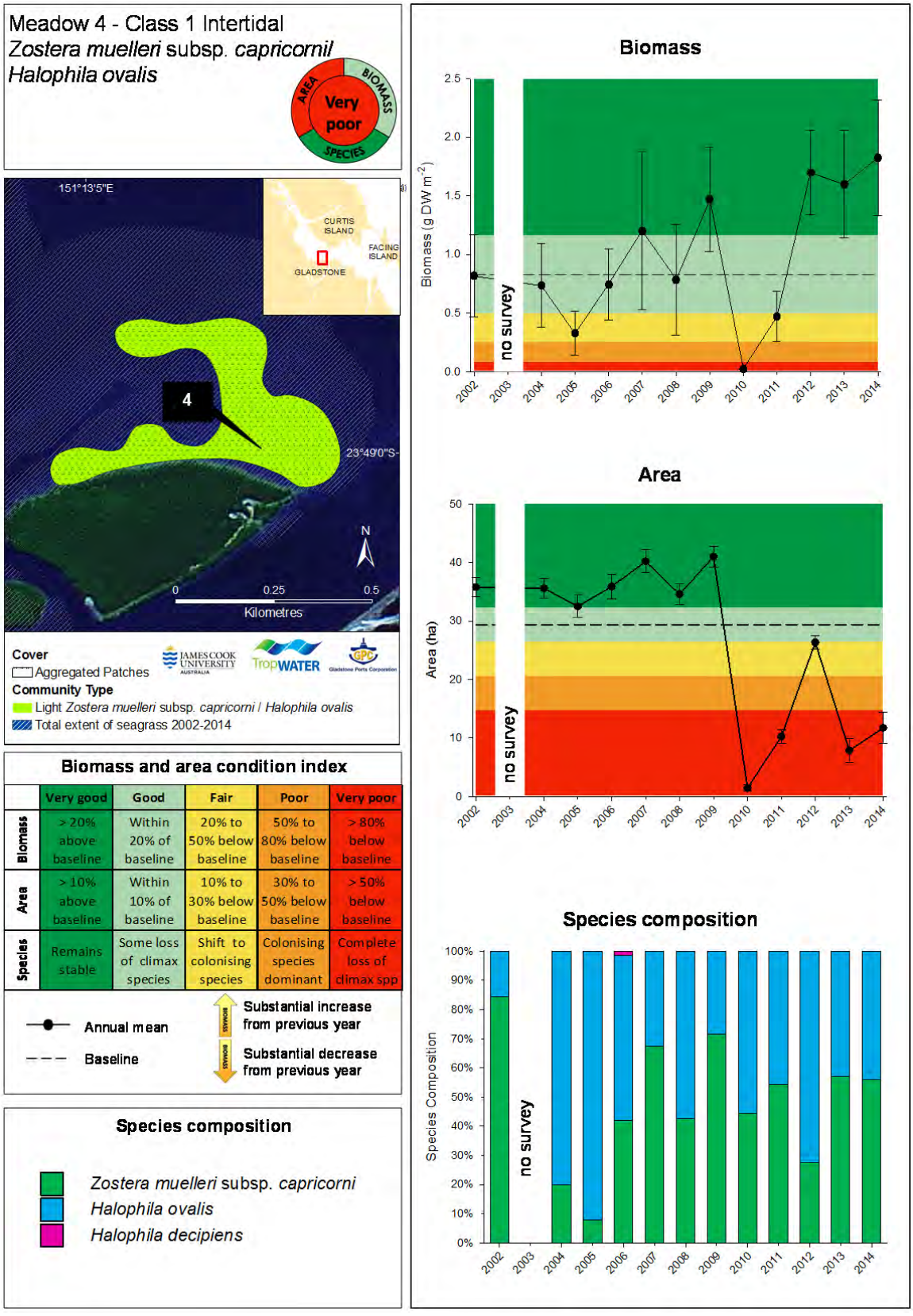


Figure 24. Changes in meadow area, biomass and species composition for seagrass at Meadow 4, Wiggins Island (Western Basin region), November 2002 - 2014 (biomass error bars = SE; area error bars = "R" reliability estimate).

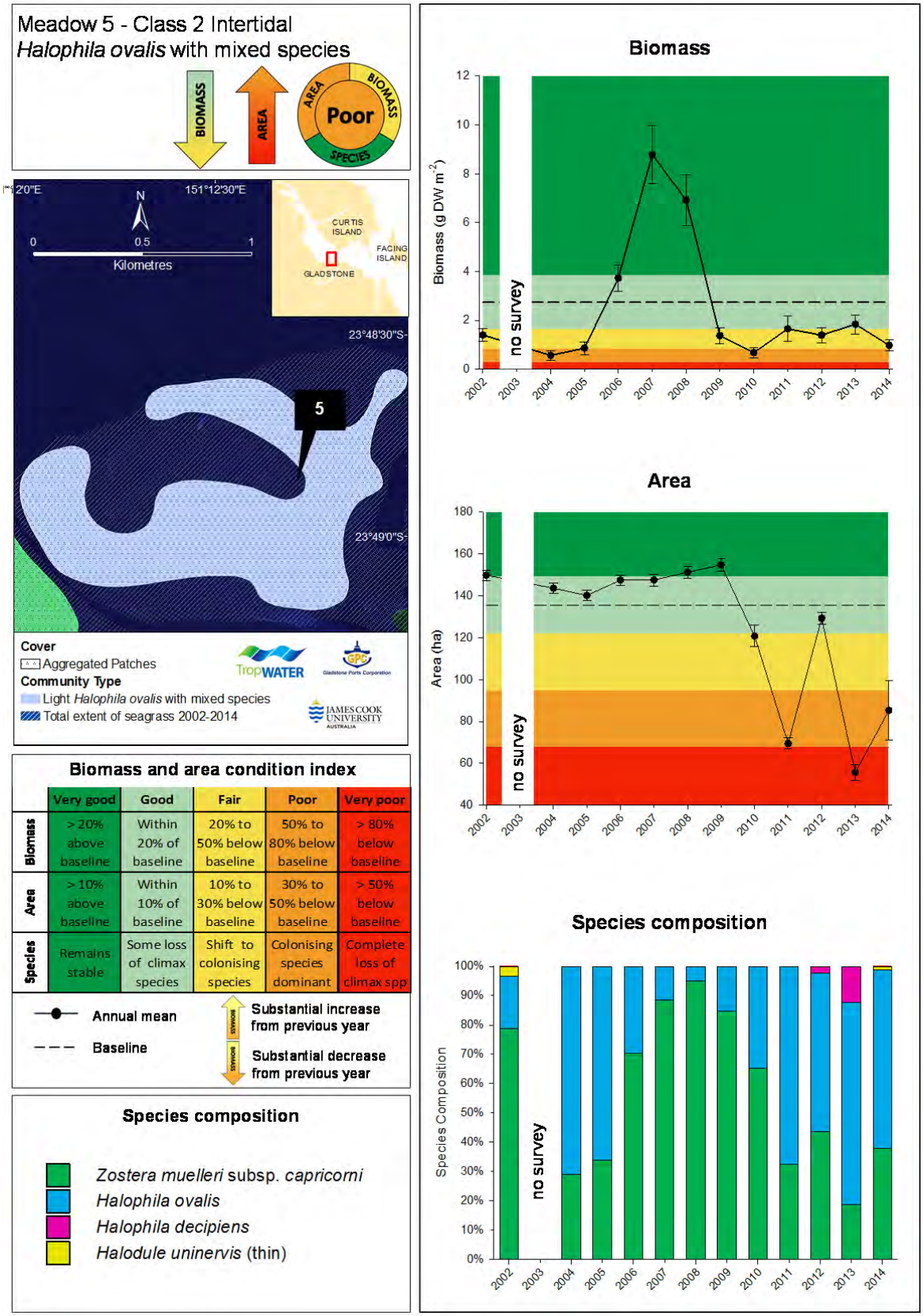


Figure 25. Changes in meadow area, biomass and species composition for seagrass at Meadow 5, Wiggins Island (Western Basin region), November 2002 - 2014 (biomass error bars = SE; area error bars = "R" reliability estimate).

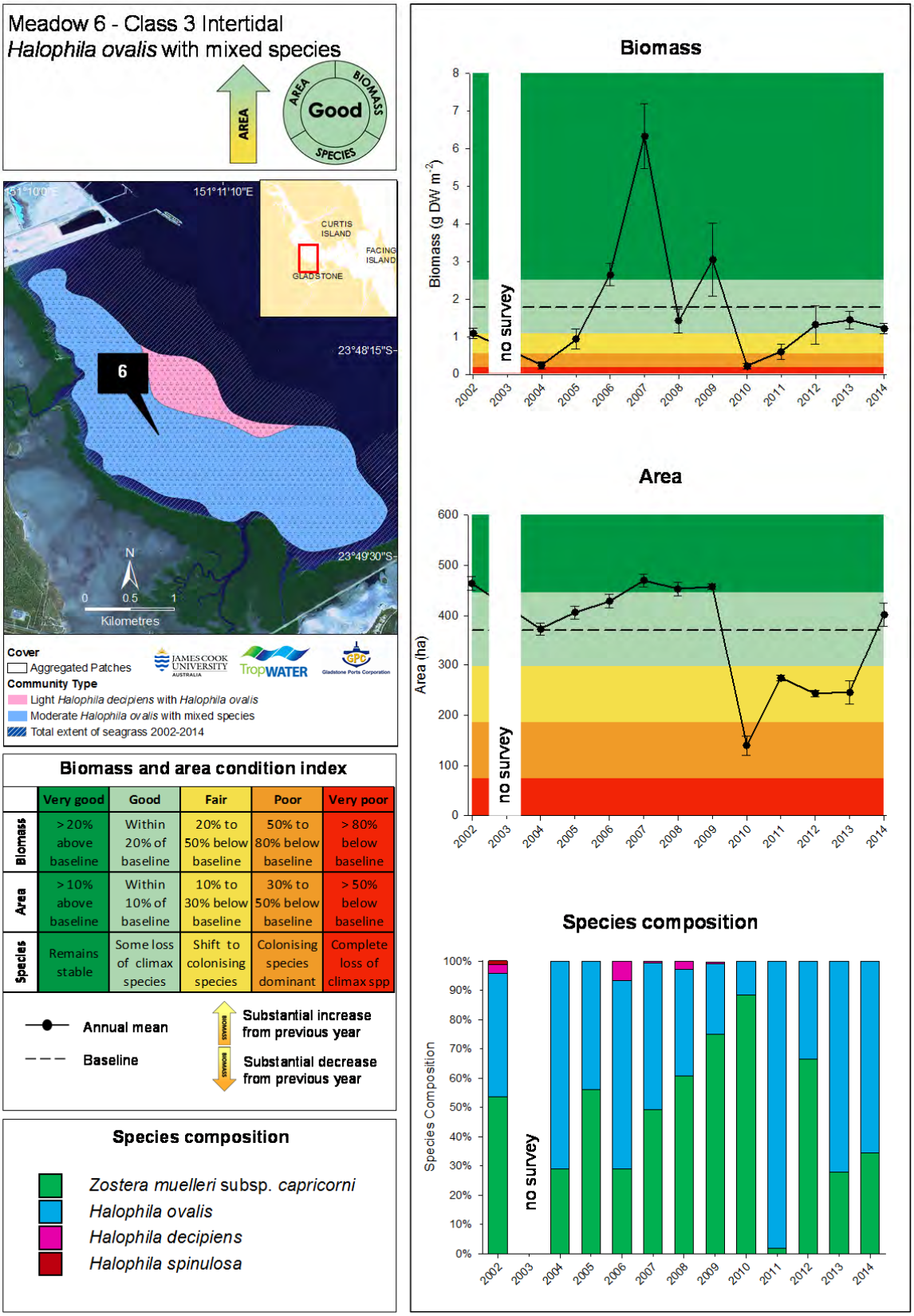


Figure 26. Changes in meadow area, biomass and species composition for seagrass at Meadow 6, South Fisherman's Landing (Western Basin region), November 2002 - 2014 (biomass error bars = SE; area error bars = "R" reliability estimate).

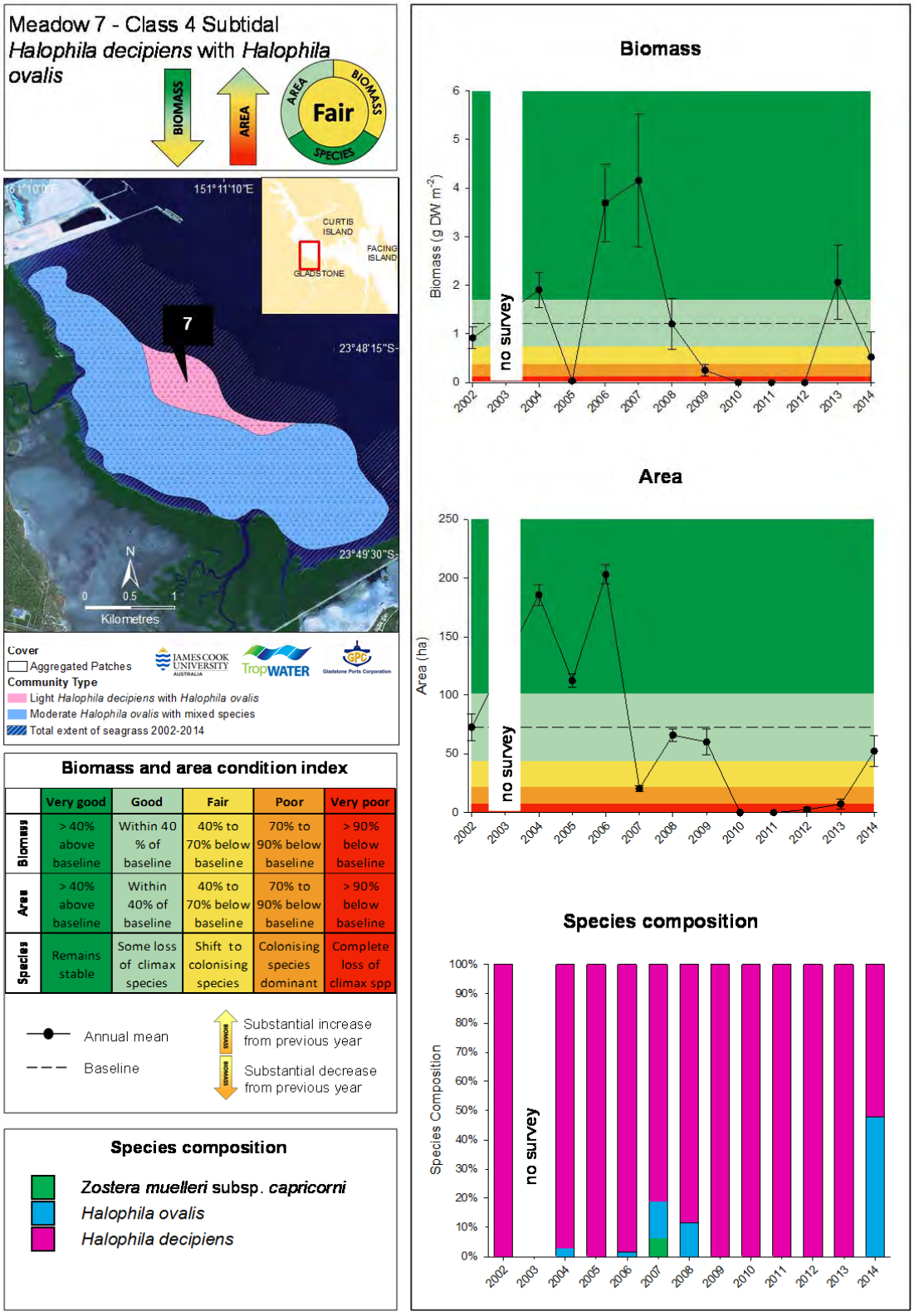


Figure 27. Changes in meadow area, biomass and species composition for seagrass at Meadow 7, South Fisherman's Landing (Western Basin region), November 2002 - 2013 (biomass error bars = SE; area error bars = "R" reliability estimate).

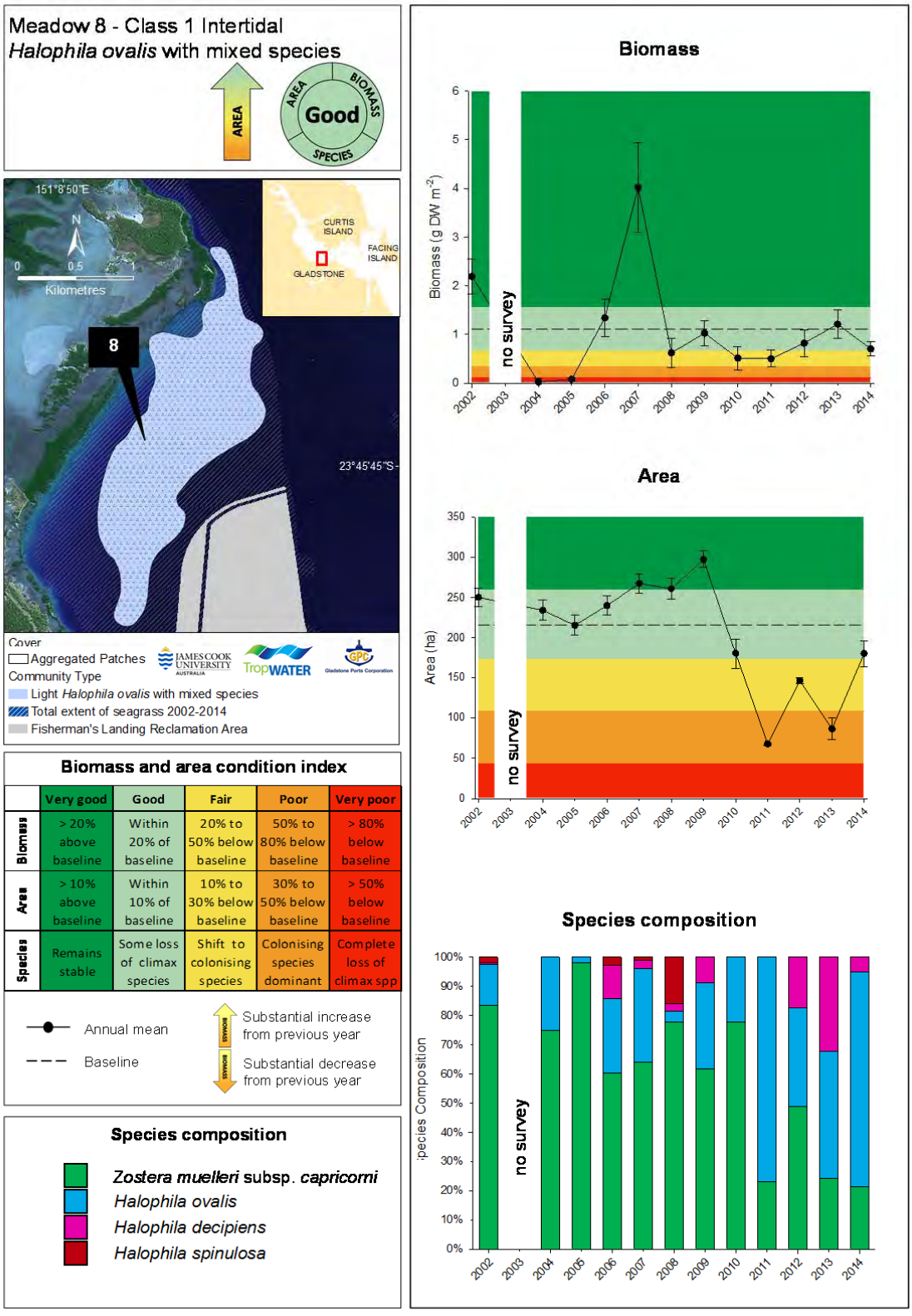


Figure 28. Changes in meadow area, biomass and species composition for seagrass at Meadow 8, North Fisherman's Landing (Western Basin region), November 2002 - 2013 (biomass error bars = SE; area error bars = "R" reliability estimate).

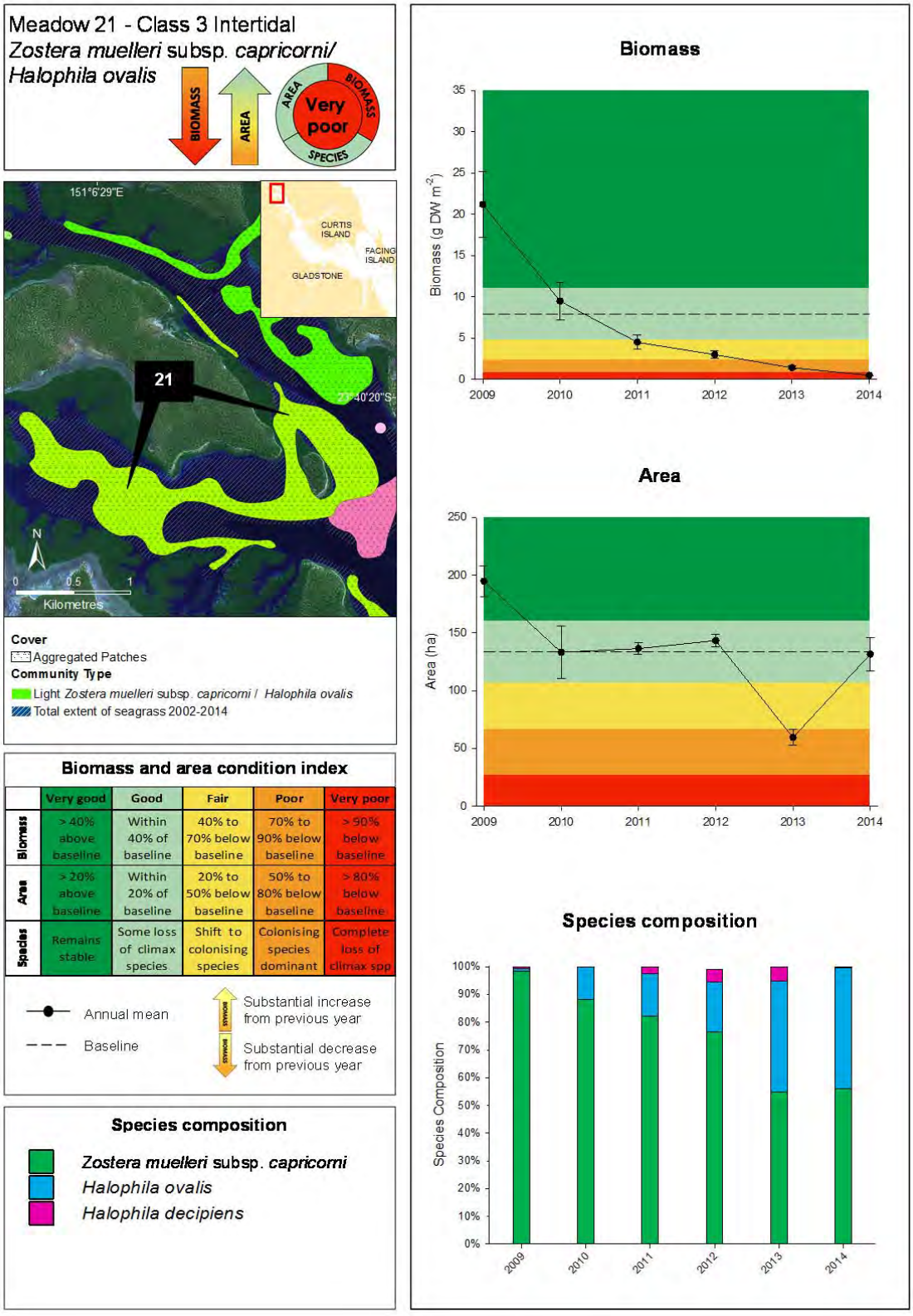


Figure 29. Changes in meadow area, biomass and species composition for seagrass at Meadow 21, Black Swan (The Narrows region), November 2002 - 2014 (biomass error bars = SE; area error bars = "R" reliability estimate).

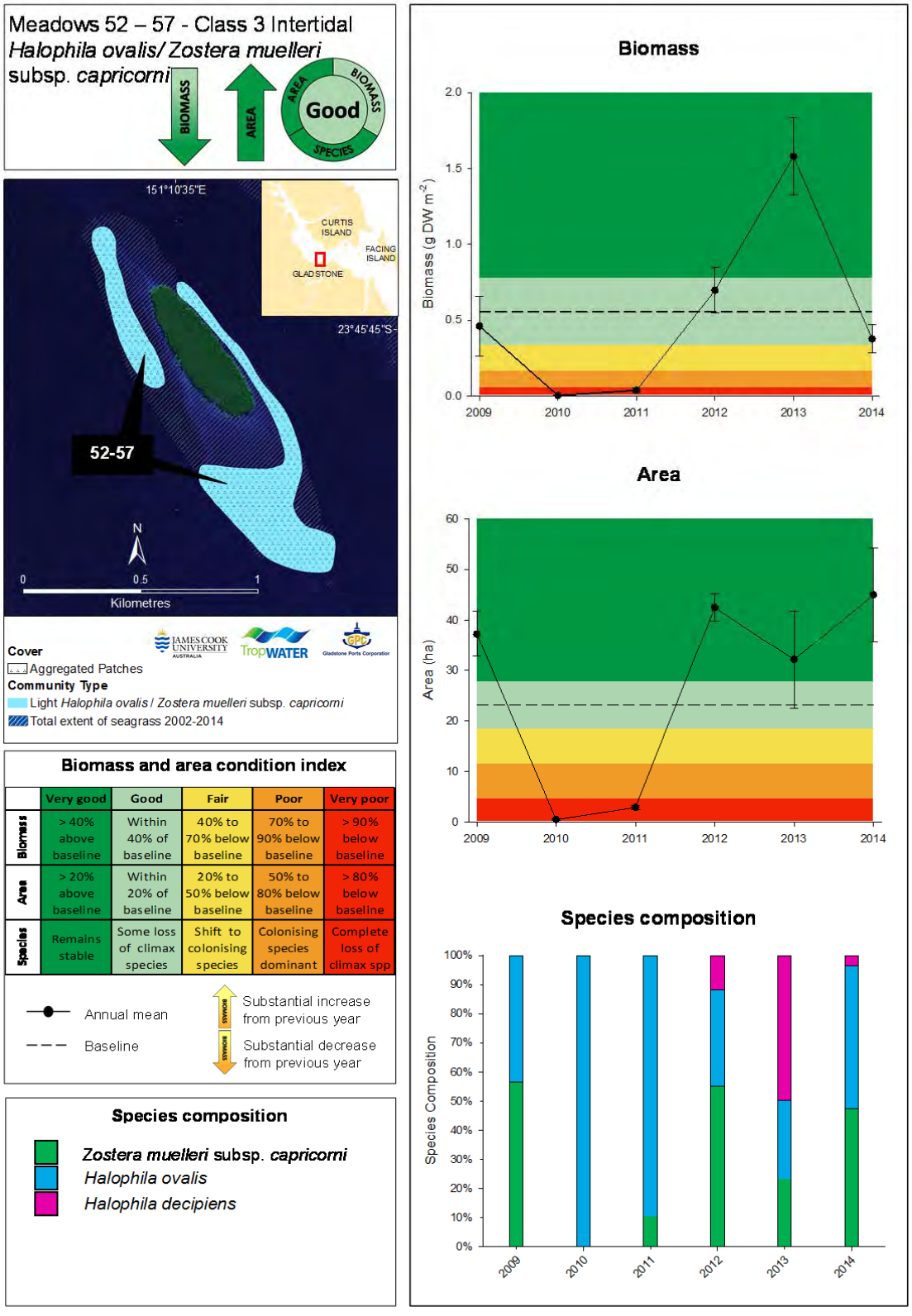


Figure 30. Changes in meadow area, biomass and species composition for seagrass at Meadows 52-57, Channel Islands (Western Basin region), November 2002 - 2014 (biomass error bars = SE; area error bars = "R" reliability estimate).

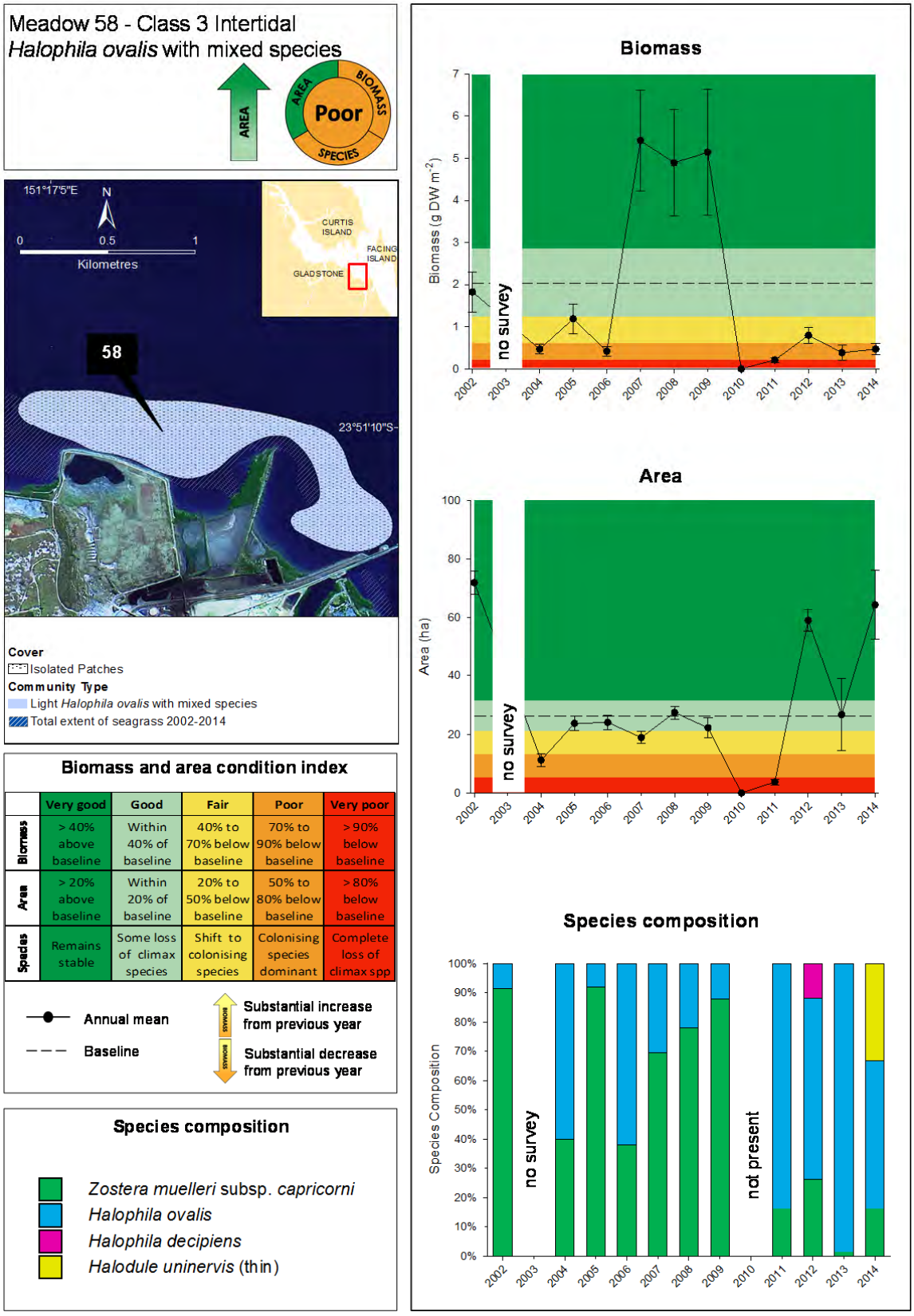
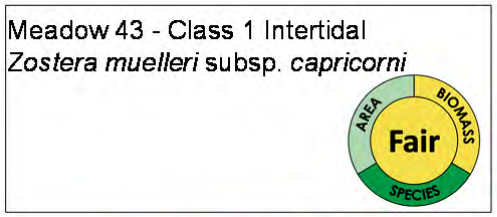
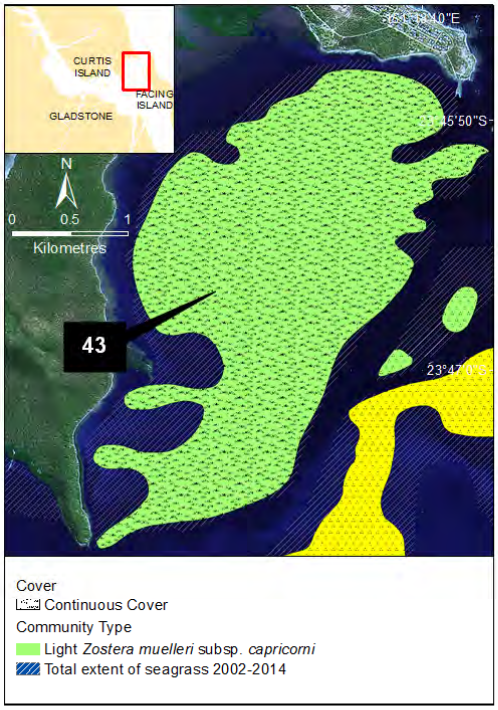


Figure 31. Changes in meadow area, biomass and species composition for seagrass at Meadow 58, South Trees Inlet region, November 2002 - 2014 (biomass error bars = SE; area error bars = "R" reliability estimate).

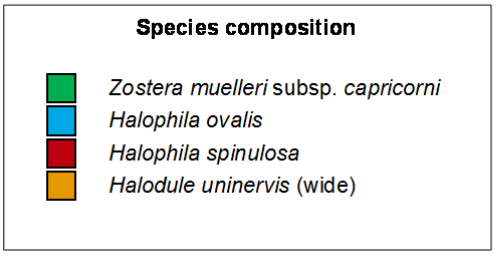




Biomass and area condition index

	Very good	Good	Fair	Poor	Very poor
Biomass	> 20% above baseline	Within 20% of baseline	20% to 50% below baseline	50% to 80% below baseline	> 80% below baseline
Area	> 10% above baseline	Within 10% of baseline	10% to 30% below baseline	30% to 50% below baseline	> 50% below baseline
Species	Remains stable	Some loss of climax species	Shift to colonising species	Colonising species dominant	Complete loss of climax spp

Annual mean
Baseline
Substantial increase from previous year
Substantial decrease from previous year



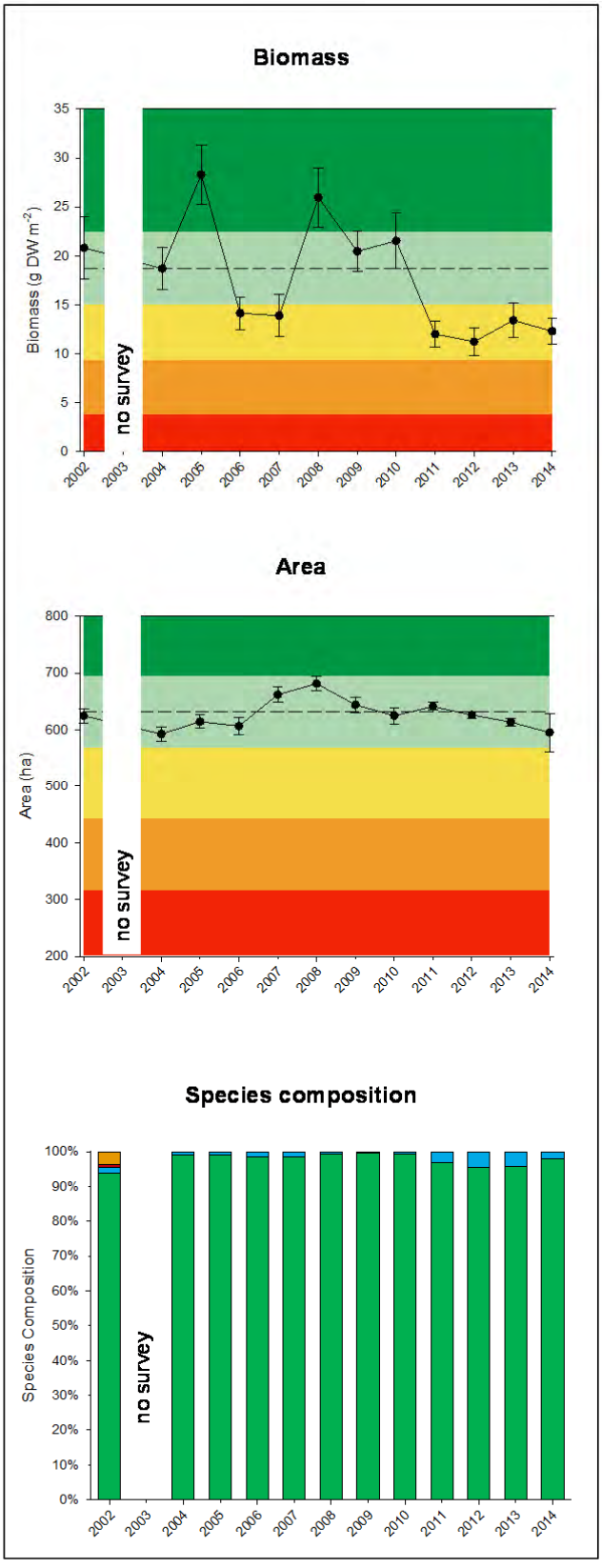


Figure 32. Changes in meadow area, biomass and species composition for seagrass at Meadow 43, Pelican Banks (Mid Harbour region), November 2002 - 2014 (biomass error bars = SE; area error bars = "R" reliability estimate).

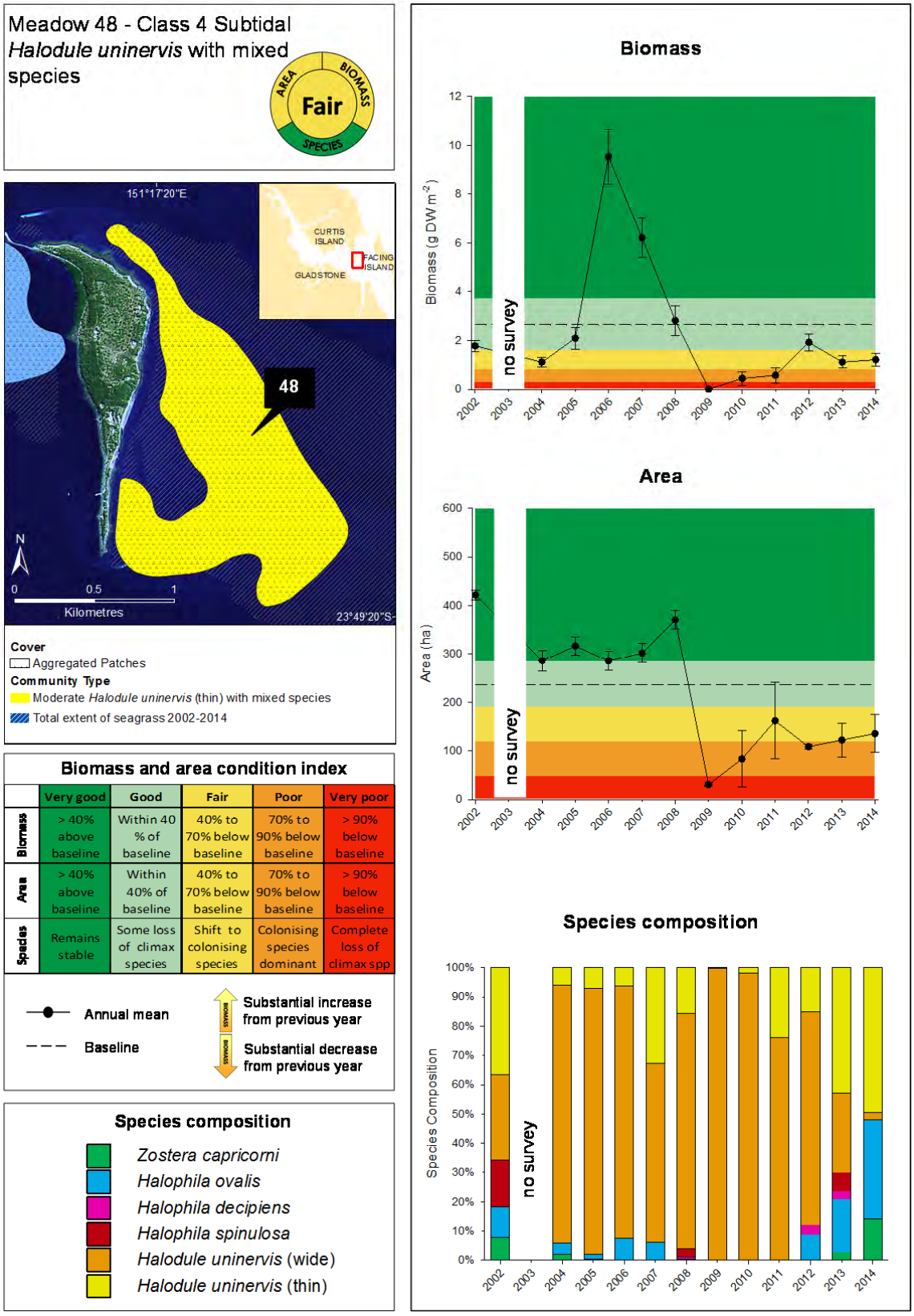


Figure 33. Changes in meadow area, biomass and species composition for seagrass at Meadow 48, Quoin Island (Mid Harbour region), November 2002 - 2014 (biomass error bars = SE; area error bars = "R" reliability estimate).

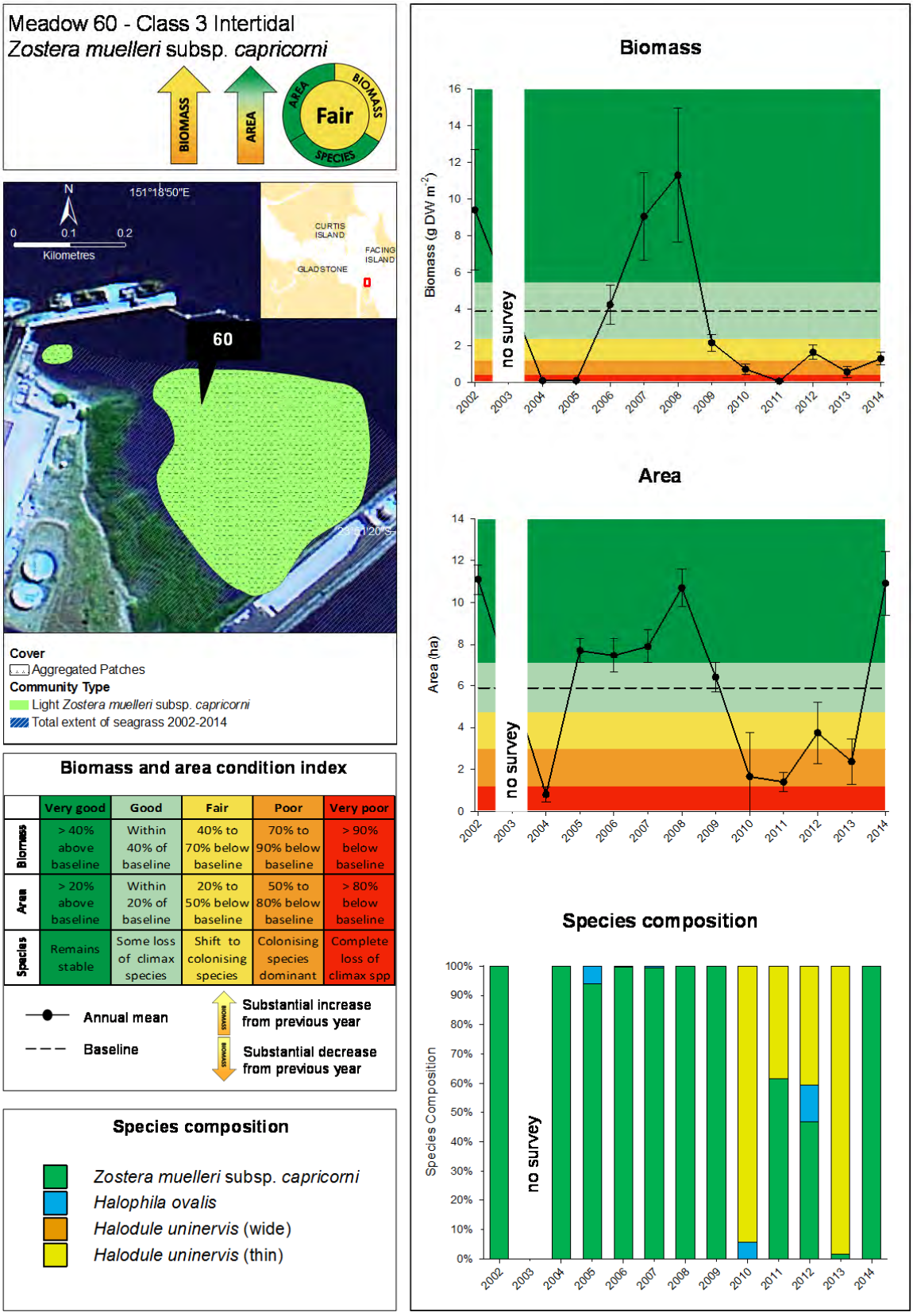


Figure 34. Changes in meadow area, biomass and species composition for seagrass at Meadow 60, South Trees Inlet region, November 2002 - 2014 (biomass error bars = SE; area error bars = "R" reliability estimate).

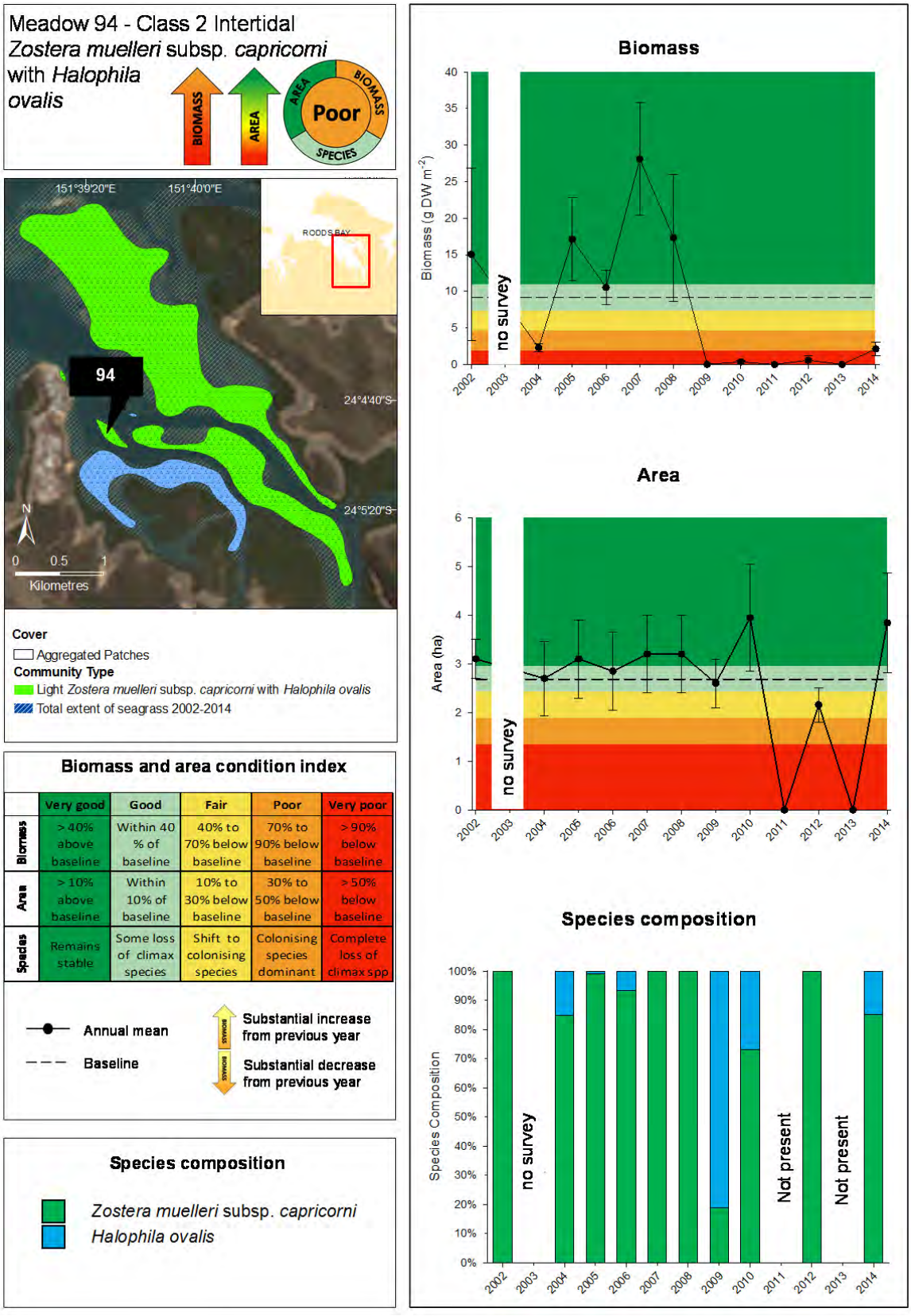


Figure 35. Changes in meadow area, biomass and species composition for seagrass at Meadow 94, Rodds Bay region, November 2002 - 2014 (biomass error bars = SE; area error bars = "R" reliability estimate).

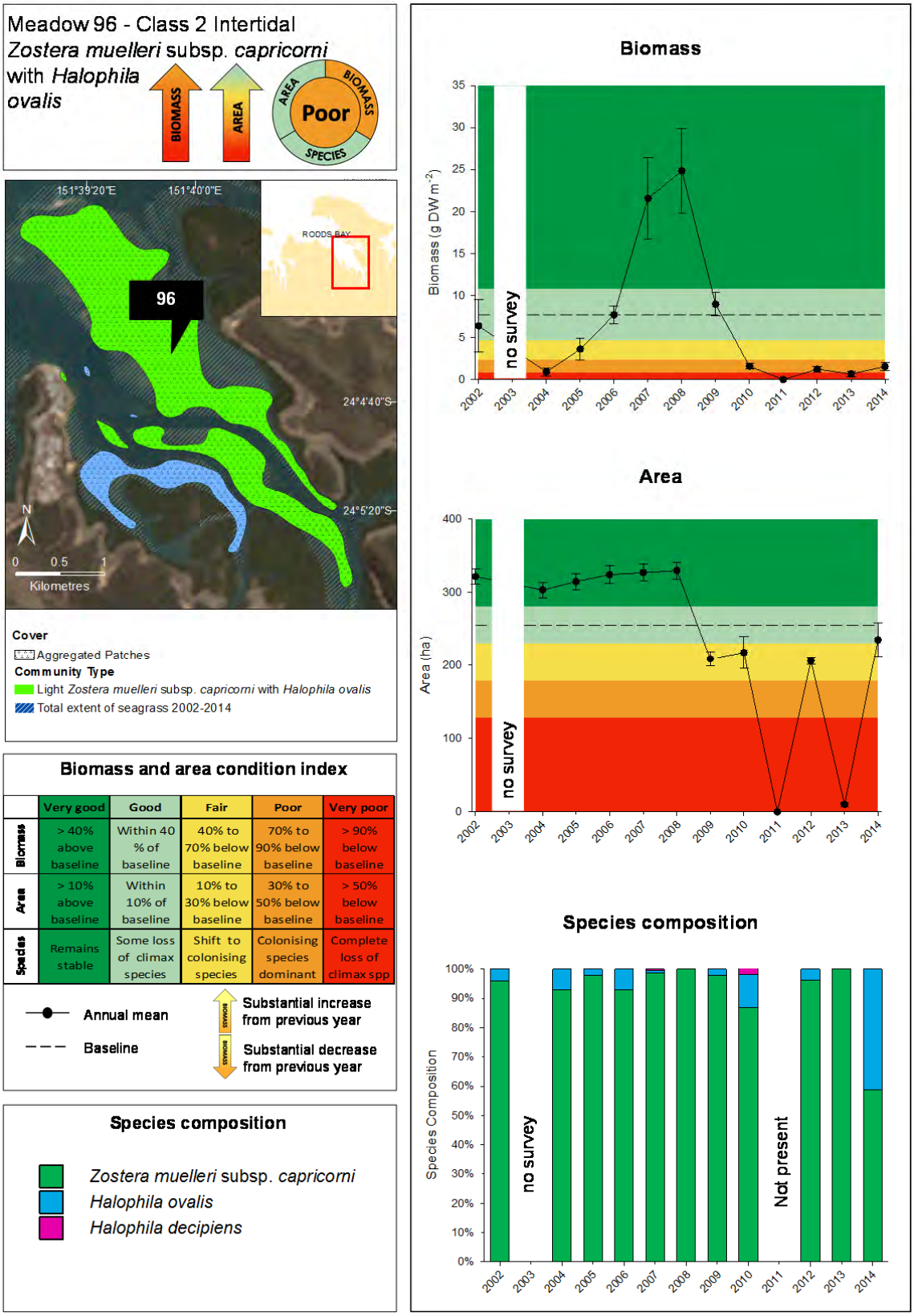
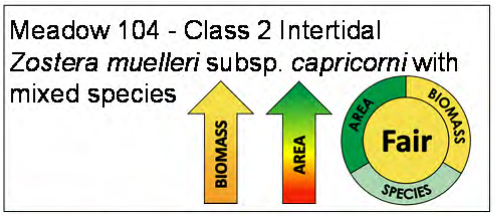
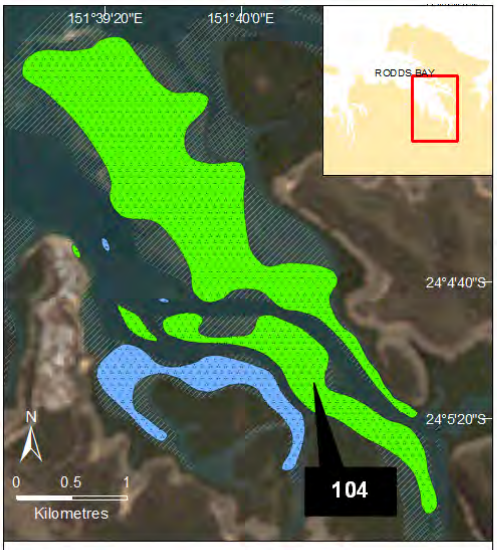


Figure 36. Changes in meadow area, biomass and species composition for seagrass at Meadow 96, Rodds Bay region, November 2002 - 2014 (biomass error bars = SE; area error bars = "R" reliability estimate).





Biomass and area condition index

	Very good	Good	Fair	Poor	Very poor
Biomass	> 40% above baseline	Within 40 % of baseline	40% to 70% below baseline	70% to 90% below baseline	> 90% below baseline
Area	> 10% above baseline	Within 10% of baseline	10% to 30% below baseline	30% to 50% below baseline	> 50% below baseline
Species	Remains stable	Some loss of climax species	Shift to colonising species	Colonising species dominant	Complete loss of climax spp

Annual mean
Baseline
Substantial increase from previous year
Substantial decrease from previous year

Species composition

Zostera muelleri subsp. *capricorni*
Halophila ovalis
Halophila decipiens
Halodule uninervis (thin)

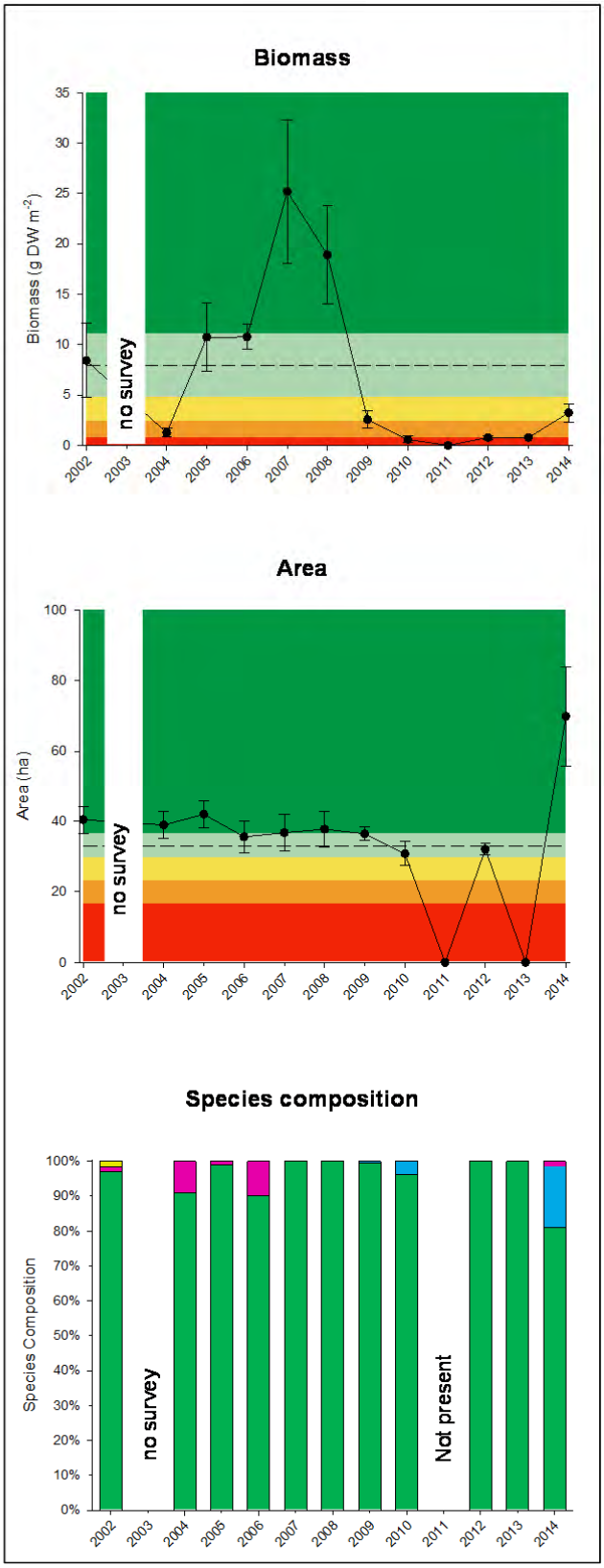


Figure 37. Changes in meadow area, biomass and species composition for seagrass at Meadow 104, Rodds Bay region, November 2002 - 2014 (biomass error bars = SE; area error bars = "R" reliability estimate).

3.7 Gladstone environmental parameters

Environmental conditions experienced by Gladstone seagrass meadows in 2014 were characterised by below average daytime tidal exposure, below average rainfall, and relatively low river flow compared with 2010, 2011 and 2013. Water temperature was generally above average and seagrasses were growing in an adequate light environment.

Total daytime tidal exposure of seagrass meadows in the year preceding the November 2014 survey was 313 hours. This was the seventh consecutive year daytime tidal exposure was below the long-term average (2000 – 2014) of 344 hours (Figure 38). Tidal exposure of seagrass meadows is naturally variable and is dependant on the tidal regime of that year. Total annual rainfall in the year preceding the survey was below the long-term annual average (since 2000), following the highest recorded rainfall over the course of the monitoring program in 2013 (Figure 39). In 2014 total monthly rainfall from March to April was above the long-term average (since 1958), and rainfall was below average in May to July (Figure 40). River flow from the Calliope River also peaked in March 2014 above the long-term average (since 1970) then remained below the long-term average leading up to November sampling (Figure 41).

Light and water temperature data collected *in situ* at monitoring meadows in 2014 generally followed established seasonal trends; water temperature generally peaked in summer and light generally peaked in spring (Figures 42 and 43). Light levels in 2012 and 2013 were generally higher than in 2010, 2011 and 2014. A threshold of an average of six mol m^{-2} day^{-1} over a two week period had been successfully used as a management threshold in Gladstone Harbour (Chartrand et al. 2012b). At this level there is sufficient light to maintain growth of *Z. muelleri* subsp. *capricorni*. Light levels in the Western Basin region at Fisherman's Landing (Meadow 8) and Wiggins Island (Meadow 5) often fell below six mol m^{-2} day^{-1} between late 2009 and early 2011, but have not done since that time (Figure 42). At Wiggins Island light levels steadily declined from November 2013 with no increase during the growing season (Figure 42). The area of both meadows increased substantially between 2013 and 2014, while mean above-ground biomass declined (Figure 43).

In the Mid Harbour Pelican Banks meadow (Meadow 43) light levels dropped below six mol m^{-2} day^{-1} several times during the senescent season for the first time since early 2011. Light levels during the growing season did not reach the same peaks as the 2012 and 2013 growing seasons; however, mean seagrass biomass at the meadow remained comparable with these years (Figure 43). At the out of port reference site at Rodds Bay (Meadow 104) light levels had declined since November 2013, similar to the trend at Wiggins Island, and did not increase during the growing season (Figure 43). With the exception of a very brief period in August 2014; however, light remained above six mol m^{-2} day^{-1}.

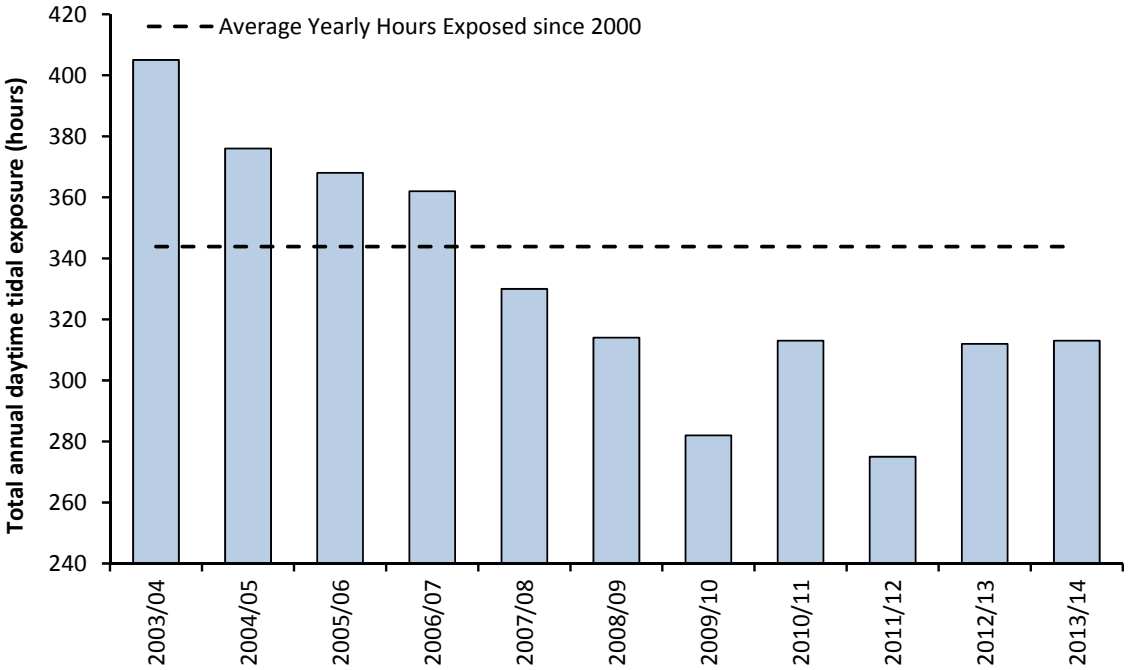


Figure 38. Total annual hours of daytime tidal exposure (0600 – 1800; <1.0 m below mean sea level) at Gladstone Harbour, 2003 - 2014.

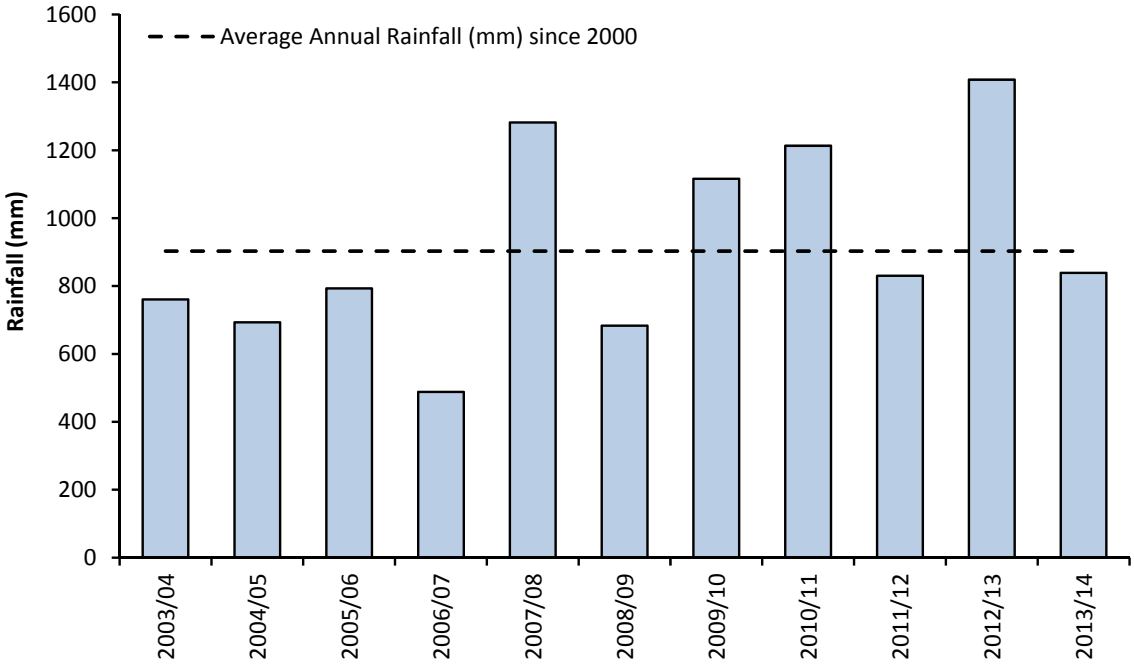


Figure 39. Total annual rainfall (mm) at Gladstone Harbour, 2003 - 2014.

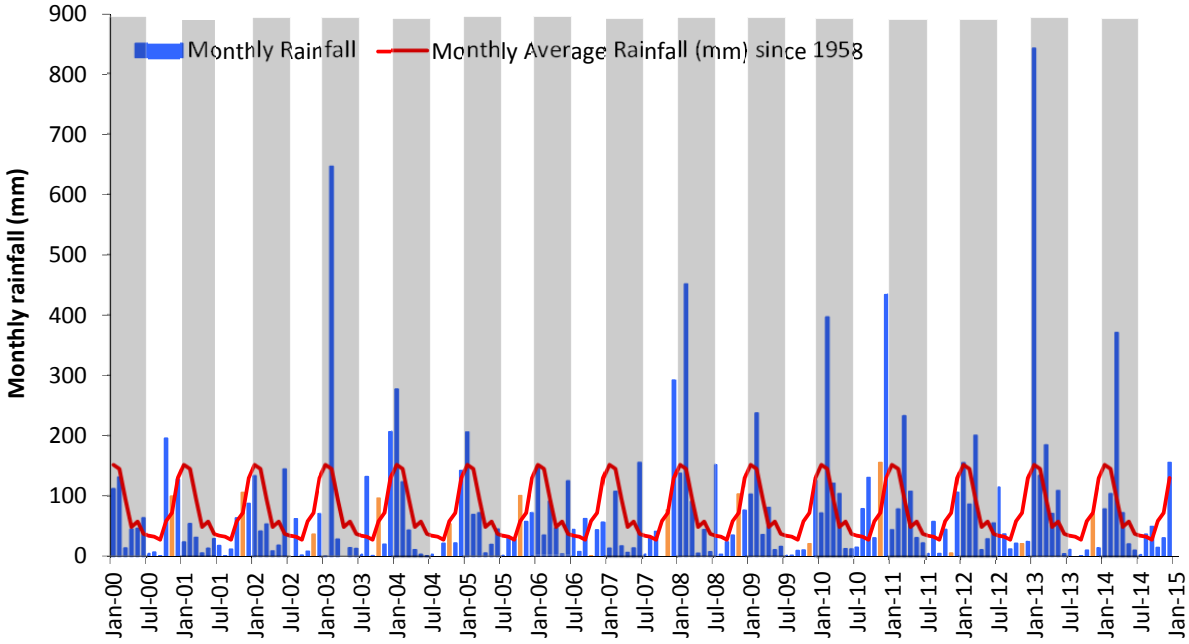


Figure 40. Total monthly rainfall (mm), January 2000 – January 2015. Orange bars indicate November rainfall when seagrass was sampled (spring peak growth period). Shaded areas represent the seagrass senescent season.

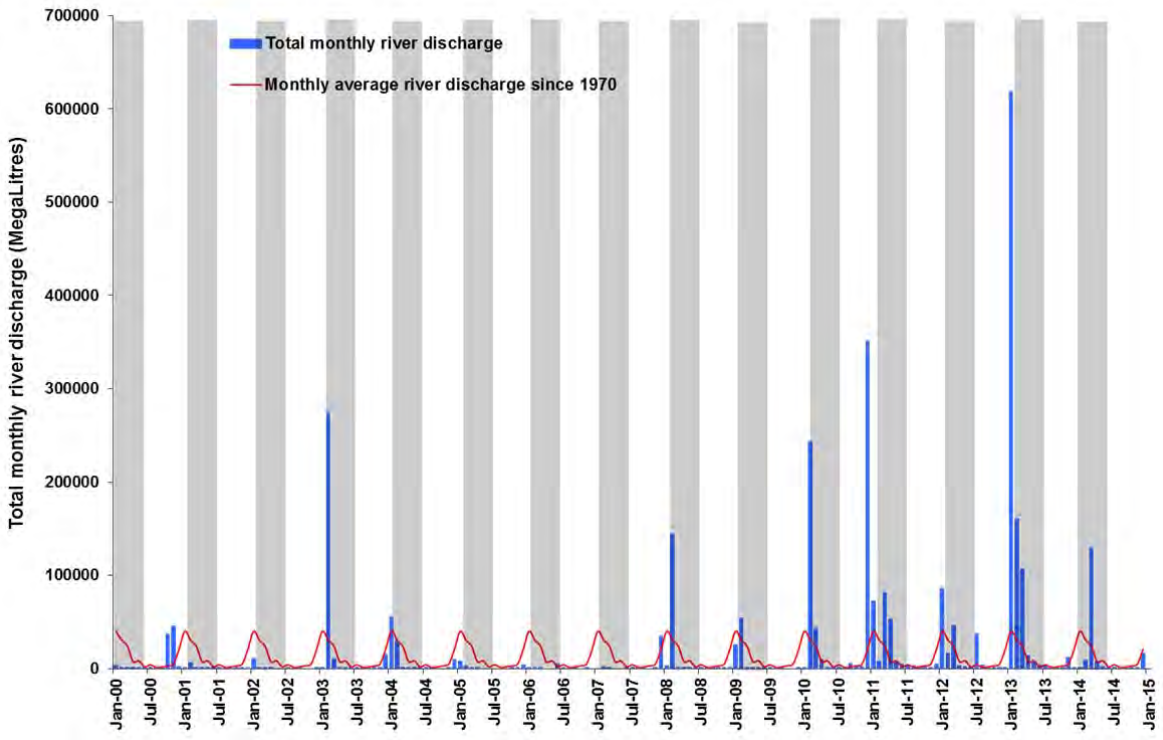


Figure 41. Monthly total river flow for the Calliope River (megalitres (ML)), January 2000 – January 2015. Shaded areas represent the seagrass senescent season.

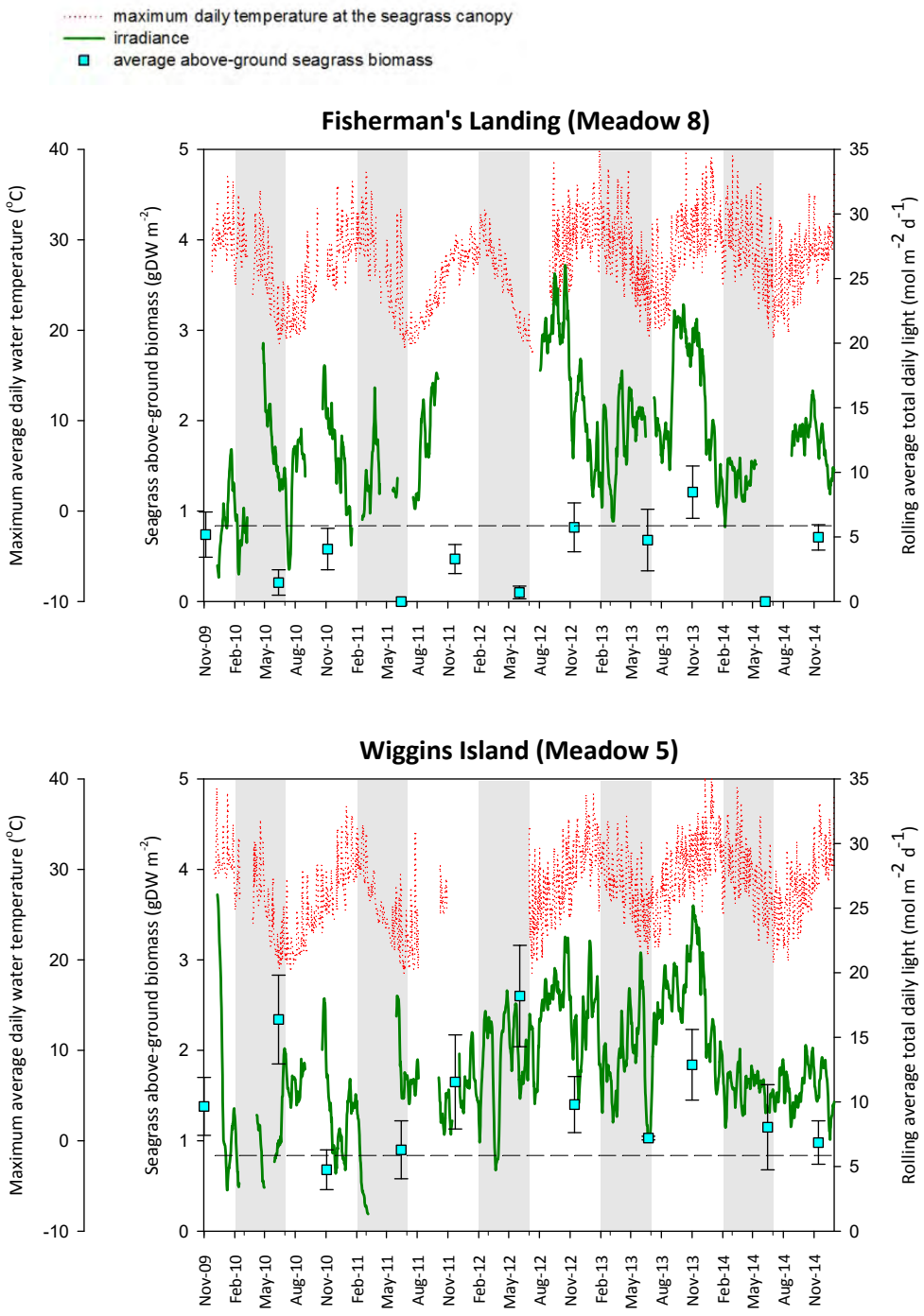


Figure 42. Mean (± standard error seagrass) above-ground biomass (blue squares) at (A) Fisherman's Landing (Meadow 8) and (B) Wiggins Island (Meadow 5), Western Basin region, plotted against the fortnightly rolling average total daily *in situ* light (mol m^{-2} day^{-1}) and maximum average daily water temperature (°C). The black dashed line indicates 6 mol m^{-2} day^{-1} which was the management trigger value for monitoring light during Western Basin dredging operations (Chartrand et al. 2012).

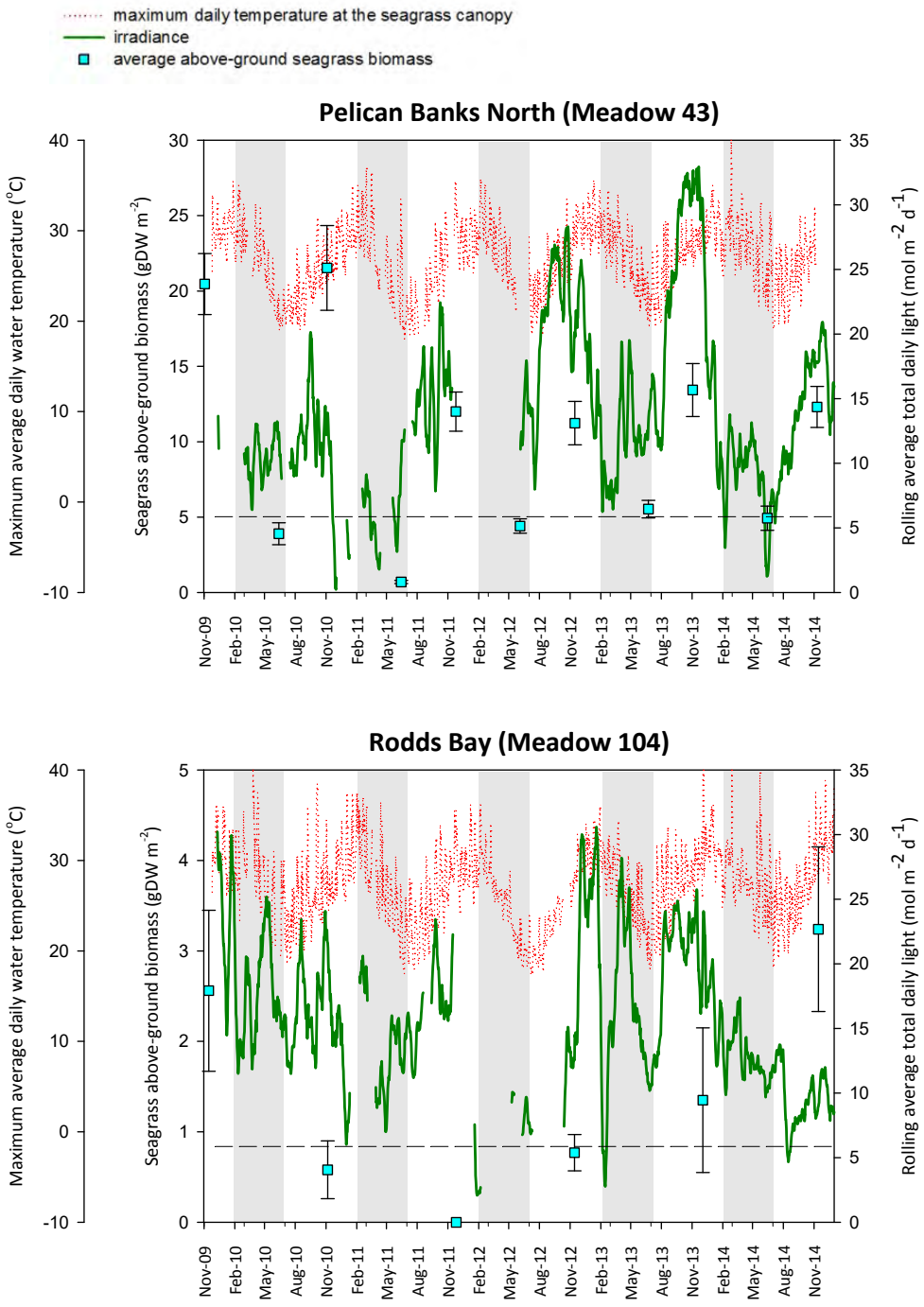


Figure 43. Mean (± standard error seagrass) above-ground biomass (blue squares) at Pelican Banks (Meadow 34, Mid Harbour region) and Rodds Bay (Meadow 104), plotted against the fortnightly rolling average total daily *in situ* light (mol m^{-2} day^{-1}) and maximum average daily water temperature (°C). The black dashed line indicates 6 mol m^{-2} day^{-1} which was the management trigger value for monitoring light during Western Basin dredging operations (Chartrand et al. 2012).

4 DISCUSSION

During 2014 there were substantial gains in seagrass cover in the Port Curtis region from the reductions that occurred during 2013. Many of the annual seagrass monitoring meadows also had improved in condition from 2013, driven largely by increases in meadow area. Despite these improvements seagrass condition remained well below historical highs recorded between 2006 and 2008, and biomass for many monitoring meadows remained below the long-term average. In 2014 any seagrass recovery has had to occur from a low base following the declines in 2013, and seagrasses likely remained in a state of reduced resilience to natural or anthropogenic impacts.

There appears to be a good link between seagrass condition in Gladstone and major climate events, especially high rainfall and flow events of the Calliope River (McCormack et al. 2013). Above average rainfall and flow from the Calliope River was recorded in 2010, 2011 and 2013, often coinciding with tropical cyclones in the region (http://www.bom.gov.au/climate/data/). For seagrasses in the Gladstone region, these years were characterised by significant declines in seagrass biomass and meadow area and a shift in species composition from *Z. capricorni* subsp. *muelleri* to the colonising *H. ovalis* in many of the monitoring meadows. Increases in seagrass biomass and meadow area during 2012 were reversed in 2013, likely due to high rainfall and the greatest river flow from the Calliope River in over a decade in early 2013. This led to the majority of monitoring meadows classified as as being in "poor" condition in 2013. Environmental conditions in 2014 were more favorable for seagrass growth, with below average rainfall and river flow. Resultant increases in meadow area at the majority of monitoring meadows meant that the overall condition of seagrass in 2014 had improved to "fair" in both the Port Curtis region and at Rodds Bay.

The updated survey of the entire Port Curtis region in November 2014 found the meadow area increases seen in monitoring meadows were reflected in in the broader region from The Narrows to Rodds Bay. From November 2013 to November 2014 deepwater seagrass habitat expanded by over 6,100 ha, and by over 8,300 ha for coastal seagrass. This followed reductions of around 75% in deepwater seagrass meadow area between November 2009 and 2013, and reductions of more than 50% for coastal seagrasses. These reductions in seagrass area also correspond with several years (2010, 2011 and 2013) of above average rainfall and peak river flow events, and the WBDDP in the years between baseline surveys (Figures 40 – 41).

The large changes seen in deepwater seagrass reflects the high level of environmentally-driven, inter-annual variability expected in these types of seagrass meadows (Rasheed et al. 2014a). Deepwater meadows and species are naturally highly variable, and large changes between years are not unexpected. Surveys of deepwater seagrasses in Gladstone and in other areas of Queensland (e.g. Townsville, Abbot Point and Mackay) have shown high levels of inter-annual variability in this type of seagrass habitat (Davies et al. 2014b; Rasheed et al. 2014a; Thomas and Rasheed 2011). While not the only factor to effect seagrass growth, changing light levels are thought to be one of the principal factors driving deepwater seagrass changes. In low light conditions the *Halophila* spp. that dominate deepwater meadows show rapid declines (Durako et al. 2003; Kenworthy et al. 1989), but may respond quickly with fast growth and recovery when conditions improve (Rasheed et al. 2014a; Hammerstrom et al. 2006). Large areas of deepwater seagrasses were identified further offshore from the port limits in November 2013. In November 2014 the deepwater seagrass meadow in this region had substantially increased and seagrass was observed in areas where no meadows had been mapped previously (Figure 20).

In The Narrows, Grahams Creek, Western Basin and Inner Harbour regions there was an increase in seagrass meadow area between November 2013 and November 2014, but seagrass is yet to reach the peaks recorded during the earlier years of the monitoring program (2002 – 2009). Reductions in coastal seagrasses in 2013 were most concentrated in The Narrows and Western Basin regions; these meadows are closest to the source of episodic flooding from the Calliope River and potential impacts from Western Basin dredging operations. Light availability is considered the key environmental determinant of the distribution, abundance and species composition of seagrass assemblages (Duarte et al. 1997; Vermaat et al. 1997) and

flood plumes and dredging have been linked to seagrass declines at other sites due to a reduction in available light (Erftemeijer and Lewis III 2006; Campbell and McKenzie 2004). The species composition of several meadows in The Narrows and Western Basin regions had also changed from *Z. muelleri* subsp. *capricorni* dominated communities to those increasingly dominated by colonising *H. ovalis* between 2009 and 2013. In November 2014 the proportion of *Z. muelleri* subsp. *capricorni* had increased in several meadows (e.g. meadow 52-57 at the Channel Islands), indicating an availability of propagules for successful recruitment. Viable *Z. muelleri* subsp. *capricorni* seed banks have been detected in these regions over the course of monitoring; however there have been no local flowering events recorded in these meadows in recent years to replenish seeds (McCormack et al. 2013). As a consequence, the age and viability of remaining seeds will be critical in understanding the ongoing ability of these meadows to regenerate. Seeds and viable fragments may well be able to disperse to these sites from more robust meadows in the Mid and Outer Harbours such as at Pelican Banks. Investigations of the viability of seed banks at these sites is currently being undertaken by TropWATER as part of research funded throught the Ecosystem Research and Monitoring Program (ERMP) and results will be available later in 2015.

Severe weather events including tropical cyclones and high rainfall and river flows from the Port Curtis catchments between 2010 and 2013 are likely to be the main drivers of seagrass declines observed in Port Curtis and Rodds Bay. Declines in seagrass biomass and area generally occcurred before the onset of the Western Basin Project capital dredging activities, with declines also occurring at the out of port reference sites in Rodds Bay, and more broadly on the east coast of Queensland between Cairns and Hervey Bay during the same period (see Section 4.1). In comparison, 2012 was a relatively dry year with below average rainfall and river flow with improved water quality conditions allowing an increase in seagrass meadow area, and biomass to a lesser extent. As discussed earlier, those gains were reversed during 2013, with timing of losses seen at sites where quaterly assessments are performed suggesting they were linked with the flooding events of the Calliope River (Davies et al. 2015a). An analysis of the relationship between a broad range of environmental variables and seagrass change in Port Curtis and Rodds Bay was performed at permanent transect monitoring sites between 2009 and 2012 (McCormack et al. 2013) and found significant relationships between river discharges into Port Curtis and seagrass abundance, with the strongest relationships for those meadows closest to the mouth of the Calliope River (Wiggins Island and Pelican Banks South). The patterns in seagrass change and river flow since that analysis remain consistent with major declines during 2013 following record flows of the Calliope River into Port Curtis, followed by increases in seagrass cover in 2014 with a return to below average rainfall and river flow conditions.

Light levels were generally favourable for seagrasses in the twelve months preceding the 2014 survey, remaining above the locally derived light requirement threshold for *Z. muelleri* subsp. *capricorni* for the majority of the year at monitoring sites (6 mol m^{-2} day^{-1} over a two week period; Chartrand et al. 2012). The favourable light conditions likely allowed for some of the seagrass increases recorded during 2014, including a greater presence of higher light requiring species in The Narrows and Western Basin regions. However, the range of seagrass responses at the different monitoring areas emphasises that it is not light alone that determines the fate of seagrass change and recovery. It is likely that light in combination with other environmental parameters such as water temperature and daytime tidal exposure provides a more complete understanding of the seasonal drivers of seagrass productivity and expansion (Carter et al. 2014b), although the influence of different environmental factors are likely to be site-specific. In addition, factors such as propagule supply the status of seed banks and success of seagrass seed germination will also likely play a critical role (Rasheed et al. 2014a).

The timing of flood-related declines in seagrass during 2010 and 2011 immediately prior to the onset of the major Western Basin Project capital dredging activities makes it difficult to ascertain what additional impact dredging and dredge material placement may have had on seagrass condition or the influence it played on the subsequent rate of recovery. A comprehensive water quality monitoring program during the WBDDP has shown that light levels were maintained above locally derived light requirements at seagrass meadows outside of the immediate dredging locations during the dredging campaign. Regardless, repeated pulse rainfall and flood events in recent years have likely reduced the resilience and capacity for recovery of

seagrasses in the region. Increases in area for monitoring meadows in 2012 and in 2014 indicate that seagrass meadows in the port and Rodds Bay have maintained a capacity to recover and with continued mild climate conditions further gains to seagrass cover and abundance are expected.

4.1 Comparisons with Queensland-wide monitoring program and seagrass resilience

Patterns of seagrass change observed in Gladstone have been generally consistent with similar changes to seagrasses recorded on the east coast of Queensland between Cairns and Gladstone. Large scale declines in seagrass distribution and meadow biomass occurred in 2009/2010 throughout the north-eastern coast of Queensland, including in Cairns (Jarvis et al. 2014), Mourilyan (York et al. 2014), Townsville (Davies et al. 2014a), and Abbot Point (McKenna and Rasheed 2014; Rasheed et al. 2014b). These declines coincided with above average rainfall and river flow (McKenna et al. 2015) often associated with tropical cyclones (TC) that have impacted the Cairns to Gladstone region such as TC Hamish (March 2009), TC Ului (March 2010), TC Anthony (January 2011), TC Yasi (March 2011) and TC Oswald (January 2013). There was a reprieve from cyclones in the region in 2012, which was reflected by below-average rainfall and river flow in these locations.

The trajectory of recovery since the declines has varied somewhat between locations, likely as a response to local differences in climate since, as well as the severity of the initial declines. For some sites the loss of seagrass meadows was total (e.g. Mourilyan) but in others such as Townsville remnant poulations of adult plants remained. Reductions in meadow area and biomass during years of extreme weather events reduce not only the adult plant population but also limit the resources available for that meadow to initiate recovery. With limited or no adult plants remaining, recovery will depend upon seed banks in the sediment or sexual propagules sourced from nearby locations locations (Jarvis and Moore 2010; Duarte and Sand-Jensen 1990; Phillips and Lewis 1983). Under these circumstances the rate of recovery is likely to be much slower, particularly where no local or nearby sources of propagules exist.

Tropical seagrasses in Queensland have generally demonstrated an ability to recover from previous impacts locations (Davies et al. 2013; Rasheed and Unsworth 2011; Rasheed 2004). Coastal seagrass meadows in monitoring locations such as Townsville (Davies et al. 2015b) have returned to conditions similar to the pre-2009 levels, and at other locations such as Abbot Point foundation species have returned and signs of ongoing recovery are positive. Other monitoring locations such as Cairns (Jarvis et al. 2015) and Mourilyan (York et al. 2015) did not show similar signs of improvement in 2014, although recent information from 2015 suggests recovery may have begun in Cairns (Jarvis pers. comm). In Mourilyan there seems little prospect of the foundation seagrass species returning to the site without some form of restoration. In this context meadows in Gladstone have shown resilience and capacity for recovery.

On the northern Cape York, Torres Strait and Gulf of Carpentaria seagrasses have fared much better over recent times. Seagrass meadows at monitoring locations in Thursday Island (Carter et al. 2014a), Weipa (Taylor et al. 2015) and Karumba (Sozou et al. 2015) have not followed the same patterns of decline and were in a “good” condition in 2014, with meadows remaining relatively stable in biomass and meadow area. These regions generally experienced a lower frequency or severity of the extreme weather events, rainfall and flooding than that of the east coast south of Cooktown.

4.2 Implications for port management

Gains in meadow area during 2014 indicate seagrasses in Port Curtis and Rodds Bay have been resilient to recent impacts, however natural recovery from large declines can take up to five years (Preen et al. 1995) or potentially longer, with recovery delayed if additional stressors such as high turbidity, poor water quality or low light levels are present.

Continued increases in seagrass meadow area and biomass within Port Curtis and Rodds Bay over the next twelve months are likely in the absence of another severe weather event as occurred in January 2013. Seagrasses in Gladstone have shown a capacity to recover from impacts and substantial seed banks detected at several sites are a good indication of continued resilience at some sites. However, as in other Queensland locations, multiple years of high rainfall, river flow and cyclone activity in the region may have reduced resilience and capacity for recovery (Rasheed et al. 2014b; Pollard and Greenway 2013).

The seagrass management tools and thresholds established through major research programs in Gladstone (Schliep et al. 2014; Chartrand et al. 2012a) provide a good basis to assist in management to protect seagrasses. Additional work funded through the ERMP to look in more detail at seed banks and their viability and assessments of dugong habitat use will also assist in assessing the ongoing capacity of Gladstone's seagrass to recover and remain resillient. These measures combined with the extensive seagrass monitoring efforts means port managers and regulators are ideally placed to understand the natural ecological processes occuring and identify measures that may be introduced to reduce the chances of exacerbating natural impacts with human activities.

APPENDICES

Appendix 1. Area and mean above-ground biomass at monitoring meadows

Table A1.1. Area (ha) ± reliability estimate for monitoring meadows in Port Curtis and Rodds Bay, 2002 - 2014.

Meadow ID	See Map	Location	Meadow Depth	Meadow area (ha)											
				2002	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
4	8	Wiggins Island	intertidal	35.8 ± 1.7	35.6 ± 1.7	32.5 ± 1.9	35.9 ± 2.1	40.2 ± 2.0	34.6 ± 1.8	41.0 ± 1.8	1.5 ± 0.5	10.27 ± 1.2	26.36 ± 1.17	7.9 ± 2.03	11.75 ± 2.61
5	8	Wiggins Island	intertidal	149.8 ± 2.5	143.6 ± 2.5	140.11 ± 2.5	147.4 ± 2.5	147.5 ± 2.9	151.2 ± 2.7	154.7 ± 2.9	120.7 ± 5.1	69.49 ± 2.69	129.24 ± 2.7	55.68 ± 3.91	85.31 ± 14.36
6	8	South Fisherman's	intertidal	464.0 ± 12.9	373.5 ± 11.9	406.4 ± 12.7	428.8 ± 13.0	470.1 ± 12.9	453.1 ± 13.2	457.3 ± 6.0	139.7 ± 18.7	274.40 ± 5.48	243.1 ± 6.01	245.98 ± 23.36	401.91 ± 23.54
7	8	South Fisherman's	subtidal	72.6 ± 11.4	185.6 ± 8.7	112.1 ± 12.3	203.1 ± 8.2	20.6 ± 2.4	65.9 ± 5.1	60.0 ± 10.8	n.p.	n.p.	2.58 ± 1.57	7.33 ± 4.09	52.22 ± 13.22
8	8	North Fisherman's	intertidal	269.1 ± 11.3	268.3 ± 12.5	231.1 ± 12.3	275.2 ± 12.0	309.9 ± 12.0	294.9 ± 12.6	311.8 ± 10.4	180.3 ± 18.4	67.71 ± 1.73	146.2 ± 3.37	86.38 ± 13.18	179.74 ± 16.68
21	6	Redcliffe	intertidal							194.7 ± 13.1	133.15 ± 22.9	136.44 ± 4.97	143.38 ± 5.71	59.48 ± 6.88	131.46 ± 14.78
43	9	Pelican Banks	intertidal	624.8 ± 12.3	592.8 ± 12.4	614.6 ± 11.9	606.8 ± 14.5	662.0 ± 13.2	681.4 ± 12.8	644.1 ± 13.3	625.0 ± 14.2	641.27 ± 6.65	626.18 ± 6.47	613.74 ± 6.75	595.4 ± 33.48
48	9	Quoin Island	intertidal/ subtidal	421.4 ± 10.2	285.8 ± 21.8	316.6 ± 18.7	285.1 ± 19.9	301.7 ± 19.8	370.9 ± 19.4	62.9 ± 6.8	83.3 ± 58.1	161.72 ± 78.53	108.36 ± 5.54	122.13 ± 34.74	135.44 ± 38.91
52-57	8	Channel Islands	intertidal							137.20 ± 4.5	0.4 ± 0.2	2.84 ± 0.44	42.46 ± 2.73	1.58 ± 0.26	44 ± 9
58	10	South Trees	intertidal	71.9 ± 3.9	11.2 ± 2.3	23.7 ± 2.4	24.0 ± 2.4	18.9 ± 2.1	27.4 ± 2.2	22.2 ± 3.5	n.p.	3.73 ± 0.92	59.07 ± 3.62	26.65 ± 12.3	64.31 ± 11.7
60	10	South Trees	intertidal	11.1 ± 0.7	0.8 ± 0.4	7.7 ± 0.6	7.5 ± 0.8	7.9 ± 0.8	10.7 ± 0.9	6.4 ± 0.7	1.66 ± 2.1	1.39 ± 0.44	3.74 ± 1.48	2.38 ± 1.09	10.91 ± 1.51
94	12	Rodds Bay	intertidal	3.1 ± 0.4	2.7 ± 0.8	3.1 ± 0.8	2.9 ± 0.8	3.2 ± 0.8	3.2 ± 0.8	2.6 ± 0.5	3.9 ± 1.1	n.p.	2.16 ± 0.35	n.p.	3.84 ± 1.02
96	12	Rodds Bay	intertidal	321.9 ± 10.6	303.5 ± 10.3	314.8 ± 10.6	324.4 ± 11.9	327.0 ± 11.5	329.9 ± 11.6	209.1 ± 10.0	217.9 ± 22.2	n.p.	205.95 ± 4.94	10.14 ± 1.32	235.29 ± 23.15
104	12	Rodds Bay	intertidal	47.7 ± 4.3	38.9 ± 3.8	41.94 ± 3.8	35.6 ± 4.5	36.7 ± 5.2	37.7 ± 5.0	36.4 ± 1.9	30.8 ± 3.4	n.p.	32.01 ± 1.61	0.01 ± 0.01	69.88 ± 14.11

n.p.; no seagrass present during the survey
n.a.; no biomass calculated

Table A1.2. Mean above-ground biomass (g DW m^{-2}) + standard error for monitoring meadows in Port Curtis and Rodds Bay, 2002 – 2014.

Meadow ID	See Map	Location	Meadow Depth	Mean biomass (g DW m^{-2})											
				2002	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
4	8	Wiggins Island	intertidal	0.8 ± 0.4	0.74 ± 0.36	0.33 ± 0.15	0.74 ± 0.26	1.20 ± 0.67	0.78 ± 0.47	1.47 ± 0.45	0.024 ± 0.003	0.47 ± 0.21	1.70 ± 0.36	1.6 ± 0.46	1.8 ± 0.49
5	8	Wiggins Island	intertidal	1.4 ± 0.3	0.57 ± 0.19	0.86 ± 0.5	3.73 ± 0.77	8.78 ± 1.17	6.92 ± 1.03	1.38 ± 0.32	0.68 ± 0.22	1.65 ± 0.52	1.4 ± 0.31	1.84 ± 0.39	0.98 ± 0.24
6	8	South Fisherman's	intertidal	1.1 ± 0.1	0.24 ± 0.09	0.94 ± 0.61	2.65 ± 0.66	6.32 ± 0.86	1.42 ± 0.32	3.05 ± 0.97	0.22 ± 0.08	0.60 ± 0.20	1.32 ± 0.51	1.44 ± 0.23	1.22 ± 0.14
7	8	South Fisherman's	subtidal	0.9 ± 0.2	1.91 ± 0.36	0.03 ± 0.02	3.7 ± 0.95	4.16 ± 1.36	1.20 ± 0.53	0.26 ± 0.12	n.p.	n.p.	n.a.	2.07 ± 0.76	0.52 ± 0
8	8	North Fisherman's	intertidal	2.1 ± 0.3	0.14 ± 0.08	0.06 ± 0.04	1.28 ± 0.49	3.89 ± 0.77	0.69 ± 0.25	0.74 ± 0.25	0.58 ± 0.23	0.47 ± 0.16	0.82 ± 0.27	1.21 ± 0.29	0.71 ± 0.14
21	6	Redcliffe	intertidal							21.18 ± 3.96	9.46 ± 2.23	4.48 ± 0.88	2.98 ± 0.43	1.45 ± 0.22	0.48 ± 0.11
43	9	Pelican Banks	intertidal	20.8 ± 3.1	18.71 ± 2.13	28.3 ± 3.3	14.17 ± 1.07	13.87 ± 2.24	25.96 ± 3.03	20.46 ± 2.02	21.53 ± 2.81	12.0 ± 1.30	11.23 ± 1.44	13.43 ± 1.76	12.31 ± 1.36
48	9	Quoin Island	intertidal/ subtidal	1.8 ± 0.2	1.11 ± 0.2	2.12 ± 1.01	9.52 ± 1.85	6.21 ± 0.80	2.81 ± 0.60	0.13 ± 0.04	0.44 ± 0.28	0.57 ± 0.30	1.92 ± 0.35	1.11 ± 0.24	1.21 ± 0.26
52 -57	8	Channel Islands	intertidal							0.45 ± 0.13	0.005	0.04 ± 0.01	2.77 ± 1.57	1.58 ± 0.26	0.38 ± 0.09
58	10	South Trees	intertidal	1.8 ± 0.5	0.47 ± 0.12	1.19 ± 0.35	0.44 ± 0.34	5.60 ± 1.24	4.89 ± 1.26	5.14 ± 1.50	n.p.	0.21 ± 0.06.	0.80 ± 0.19	0.38 ± 0.18	0.47 ± 0.14
60	10	South Trees	intertidal	9.4 ± 3.3	0.08 ± 0.01	0.09 ± 0.03	4.23 ± 0.58	9.04 ± 2.40	11.29 ± 3.65	2.16 ± 0.46	0.71 ± 0.28	0.05 ± 0.03	1.63 ± 0.39	0.56 ± 0.3	1.28 ± 0.35
94	12	Rodds Bay	intertidal	15.1 ± 11.8	2.3 ± 0.51	17.11 ± 3.02	10.54 ± 1.38	28.11 ± 7.67	17.33 ± 8.70	0.01 ± 0.01	0.34 ± 0.19	n.p.	0.58 ± 0.58	n.a.	2.13 ± 0.93
96	12	Rodds Bay	intertidal	6.4 ± 3.1	0.9 ± 0.5	3.62 ± 1.41	7.7 ± 1.58	21.56 ± 4.88	24.84 ± 5.01	8.96 ± 1.38	1.57 ± 0.34	n.p.	1.23 ± 0.3	0.67 ± 0.34	1.55 ± 0.46
104	12	Rodds Bay	intertidal	8.4 ± 3.7	1.26 ± 0.43	10.73 ± 2.62	10.76 ± 1.81	25.20 ± 7.09	18.89 ± 4.84	2.56 ± 0.89	0.58 ± 0.32	n.p.	0.77 ± 0.2	0.77	3.24 ± 0.91

n.p.; no seagrass present during the survey

n.a.; no biomass calculated

Appendix 2. Statistical analyses

Table A2.1. One-way analysis of variance (ANOVA) tables comparing mean biomass among years (2002, 2004 - 2014). All biomass data were square-root-transformed prior to analyses. Significant p-values (p > 0.05) in **bold**.

Meadow 5	**DF**	**SS**	**MS**	**F**	**P**
Year	11	169.2	15.38	18.23	**<0.001**
Residuals	354	298.6	0.84		
Meadow 6	**DF**	**SS**	**MS**	**F**	**P**
Year	11	113.8	10.35	15.05	**<0.001**
Residuals	473	325.3	0.69		
Meadow 21	**DF**	**SS**	**MS**	**F**	**P**
Year	5	275.1	55.01	23.82	**<0.001**
Residuals	229	528.9	2.31		
Meadow 52-57	**DF**	**SS**	**MS**	**F**	**P**
Year	5	13.10	2.62	9.22	**<0.001**
Residuals	146	41.51	0.28		

Table A2.2. Analysis of deviance tables from binomial-Gamma hurdle model with zero-inflated biomass data. (A) Binomial regression of presence/absence data among years compared with null model; and (B) generalised linear model of biomass-only sites among years (2002, 2004 - 2014) compared against the null model. All biomass data were square-root-transformed prior to analyses. Significant p-values (p > 0.05) in **bold**.

	Meadow 4	**DF**	**Deviance**	**Resid. DF**	**Resid. Deviance**	**Pr(>Chi)**
(A)	NULL			222	303.63	
	Year	11	51.56	211	252.07	**<0.001**
(B)	NULL			93	35.72	
	Year	10	7.17	83	28.55	**<0.001**
	Meadow 7	**DF**	**Deviance**	**Resid. DF**	**Resid. Deviance**	**Pr(>Chi)**
(A)	NULL			161	223.69	
	Year	11	109.14	150	114.55	**<0.001**
(B)	NULL			86	77.90	
	Year	8	40.18	48	37.72	**<0.001**
	Meadow 8	**DF**	**Deviance**	**Resid. DF**	**Resid. Deviance**	**Pr(>Chi)**
(A)	NULL			246	181.72	
	Year	11	62.58	235	119.14	**<0.001**
(B)	NULL			246	181.72	
	Year	11	62.58	235	119.14	**<0.001**

Table A2.3. One-way analysis of variance (ANOVA) comparing mean biomass among years (2002, 2004 - 2014). All biomass data were square-root-transformed prior to analyses. Significant p-values (p > 0.05) in **bold**.

Meadow 43	**DF**	**SS**	**MS**	**F**	**P**
Year	11	263	23.88	4.49	**<0.001**
Residuals	745	3964	5.32		
Meadow 48	**DF**	**SS**	**MS**	**F**	**P**
Year	11	155.2	14.11	17.84	**<0.001**
Residuals	336	265.7	0.79		
Meadow 58	**DF**	**SS**	**MS**	**F**	**P**
Year	11	72.8	6.62	5.34	**<0.001**
Residuals	263	326.4	1.24		
Meadow 60	**DF**	**SS**	**MS**	**F**	**P**
Year	11	108.4	9.86	9.79	**<0.001**
Residuals	140	141.0	1.01		

Table A2.4. One-way analysis of variance (ANOVA) comparing mean biomass among years (2002, 2004 - 2014) for Rodds Bay meadows. All biomass data were square-root-transformed prior to analyses. Significant p-values (p > 0.05) in **bold**.

Meadow 94	**DF**	**SS**	**MS**	**F**	**P**
Year	11	340.6	30.96	15.01	**<0.001**
Residuals	127	262.0	2.06		
Meadow 96	**DF**	**SS**	**MS**	**F**	**P**
Year	11	451.3	41.03	12.23	**<0.001**
Residuals	336	1127.4	3.36		
Meadow 104	**DF**	**SS**	**MS**	**F**	**P**
Year	11	354.2	32.20	12.65	**<0.001**
Residuals	223	567.6	2.55		

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Appendix G
Western Basin and Channel Duplication Seagrass and Light Monitoring

TropWATER
Centre for Tropical Water and Aquatic Ecosystem Research

JAMES COOK UNIVERSITY
AUSTRALIA

Update for Gatcombe and Golding Cutting Channel Duplication Project EIS

Port Curtis, Rodds Bay and Channel Duplication Seagrass and Light Monitoring: January 2013 to May 2015

Davies JN, McKenna SA and Rasheed MA

Report No. 15/20

July 2015

Update for Gatcombe and Golding Cutting Channel Duplication Project EIS

Port Curtis, Rodds Bay and Channel Duplication Seagrass and Light Monitoring: January 2013 to May 2015

A report for Gladstone Ports Corporation

Report No. 15/20

July 2015

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ACRONYMS AND ABBREVIATIONS

MMP	Marine Monitoring Program
BPAR	Benthic Photosynthetically Active Radiation
WBDDP	Western Basin Dredging and Disposal Project
EIS	Environmental Impact Statement
HSD	Honest Significant Difference
ZOIB	Zero/one Inflated Binomial
WICET	Wiggins Island Coal Export Terminal

KEY FINDINGS

This report details findings from the assessment of seagrass change at a range of permanent transect and deepwater monitoring sites that were established as part of the Western Basin Dredging and Disposal Project (WBDDP) and the Gatcombe and Golding Cutting Channel Duplication Project (Channel Duplication Project). The report provides the latest findings for inclusion in the Channel Duplication Project Environmental Impact Statement (EIS). These sites are assessed quarterly throughout the year and this report includes results up to 19 May 2015, updating the most recent annual report for the program which included data up to November 2014. This work is in addition to the long term annual seagrass monitoring that has been conducted since 2002 which examines and re-maps the full extent of seagrasses each November at the time of their maximum distribution. The results of that program are presented in a separate report (see Carter et al. 2015).

- Monitoring at permanent transects revealed that seagrasses in Port Curtis and Rodds Bay underwent distinct seasonal changes as well as significant inter-annual changes in seagrass distribution, abundance and species composition.
- In January 2013, the Calliope River again discharged storm water runoff at record levels and declines in seagrass were detected across most of the monitoring sites. Recovery over the 2013 and 2014 growing seasons varied with some sites (e.g. Pelican Banks South) remaining atypically low in seagrass cover, while the out of Port reference sites at Rodds Bay had substantial recovery. In the first half of 2015, seagrasses at most sites had declined in line with seasonal trends.
- Additional coastal sites established in May 2014 as part of the Channel Duplication Project followed seasonal trends in biomass and seagrass precent cover, with peaks during the growing season in November 2014. These sites generally had higher seagrass cover at their peak (5-20%) than other coastal sites within the Port.
- Three deep water monitoring sites were also established in May 2014. These sites were dominated by *Halophila ovalis* with a very small proportion of *H. decipiens* and *H. spinulosa*. All three sites had extremely low biomass (< 1 g DW m^{-2}) throughout the monitoring period, with peaks in above-ground biomass in either September or November 2014. By March 2015 seagrass was absent from all three sites, as expected for meadows dominated by highly ephemeral species.
- Development of initial light ranges suitable for the maintenance and growth of seagrass species within Port Curtis was undertaken by analysis of literature values, previous studies and the light and seagrass condition information collected as part of this program.
- Based on these assessments, it was found that seagrasses in Port Curtis were maintained or increased during the growing season when the light environment was within the following thresholds:
    - *Zostera muelleri* subsp. *capricorni** requires **6** mol photons m^{-2} day^{-1} over a rolling **14 day average.**
    - *Halodule uninervis* requires **5** mol photons m^{-2} day^{-1} over a rolling **14 day average**.
    - *Halophila* species require **1-2** mol photons m^{-2} day^{-1} over a rolling **7 day average**.

*The naming convention used in previous reports has been updated to align with the accepted nomenclature for this species.

With the exception of *Zostera muelleri* subsp. *capricorni,* the light thresholds presented in this report are considered preliminary. The ability to further refine local light thresholds for other species was not possible due to the 12 month timeframe that paired light and seagrass assessments had been conducted at the additional Channel Duplication Project monitoring sites. While these threshold values present a useful guide for the planning and EIS stage of the Channel Duplication Project, further refinement would be required to test the relationships between light and seagrass abundance. An additional 12 months of seagrass monitoring and assessment are recommended at the three coastal seagrass sites and three deep water seagrass sites established for the Channel Duplication Project prior to dredging works commencing to strengthen locally derived seagrass light thresholds. Additional research activities including field shading studies to test observed relationships would also assist in refining these levels.

Currently seagrasses have shown a capacity to recover from natural and anthropogenic impacts in Port Curtis, but as has been seen in other Queensland locations repeated disturbances over multiple years may lead to long term loss, with recovery trajectories far less certain. The extensive and detailed seagrass monitoring and research efforts in Port Curtis means an improved understanding of the processes and opportunities 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 (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 (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). A recent assessment by an expert panel on the relative impacts of anthropogenic activities listed industrial and urban runoff, port development, and dredging as the main threats to seagrass ecosystems in the tropical Indo-Pacific region (Grech et al. 2012). Severe storms and heavy rainfall and riverflow events can also negatively impact seagrasses through increased turbidity, reduced light quality and availability and direct burial.

Large areas of seagrass were first identified in the Port Curtis region (from the Narrows to Rodds Bay) as part of broad scale state-wide seagrass surveys conducted in 1988 (Coles et al. 1992). A detailed annual seagrass monitoring program was established following a baseline survey of all seagrasses in the port in 2002 (see Rasheed et al 2003; Carter et al. 2015). In 2009, Gladstone Ports Corporation (GPC) commissioned a suite of additional research and monitoring, including quarterly assessments of seagrass condition at key monitoring meadows throughout Port Curtis and in nearby Rodds Bay. The quarterly monitoring program adapts the established Seagrass-Watch protocols which utilise permanent intertidal transects at seagrass meadows (McKenzie et al. 2000). These permanent transects form the key sensitive receptor sites for assessing seagrass condition before, during and after the WBDDP which concluded at the end of 2013. For detailed information about the establishment of this program, see McCormack et al. (2013).

As part of the EIS process for the proposed dredging and dredged material placement activities associated with the Channel Duplication Project, additional permanent sites were established in May 2014 to complement the pre-existing network of seagrass sites and improve the spatial coverage of coastal and offshore deepwater seagrass meadows most at risk. Seagrass has been monitored at three deepwater sites (deepwater sites 1 – 3) and three coastal sites (Colosseum, South Trees Inlet and Quoin Island) in conjunction with established quarterly monitoring of permanent transect sites established for the WBDDP.

This report details findings of the quarterly seagrass assessment program at permanently established sites within Port Curtis from January 2013 through May 2015 as an update to the previously published report that summarised results through to the end of 2014 (see Davies et al. 2015) to provide the latest data and analysis for incorporating into the Channel Duplication Project EIS. Results of this report form part of a broader suite of seagrass investigations that also include annual seagrass mapping of all seagrasses in the Western Basin region (Carter et al. 2015), a research and management program establishing the light requirements of Port Curtis seagrasses (Chartrand et al. In review; Chartrand et al. 2012), investigations of seed bank viability (Jarvis et al. 2015) as well as ongoing research into establishing sub-lethal indicators of seagrass stress to use in seagrass management (Pernice et al. 2015).

Study location

Seven locations (Pelican Banks North, Pelican Banks South, Wiggins Island, Facing Island, Fisherman's Landing, Black Swan, and Rodds Bay) have been established for quarterly monitoring of seagrass and benthic light for the WBDDP and continued for the post-dredging phase of the WBDDP (Figure 1). Throughout this report these monitoring locations are referred to as the WBDDP post-dredging monitoring meadows.

In May 2014, three new coastal sites (Colosseum Inlet, South Trees and Quoin Island) and three deepwater monitoring sites (sites 1-3) were established to monitor additional areas potentially impacted by the proposed Channel Duplication Project (Figure 1).

Throughout this report, sites are generally grouped into three sections; the WBDDP post-dredging monitoring sites, including the reference sites outside of the Port in nearby Rodds Bay, the Channel Duplication coastal monitoring sites and deepwater monitoring sites. For more information on the establishment and monitoring history of permanent transect sites in Port Curtis as well as trends in seagrass prior to 2013, see McCormack et al. (2013).

For the results of other seagrass monitoring initiatives such as the long term annual monitoring program and remapping of all seagrass areas see Carter et al. (2015)

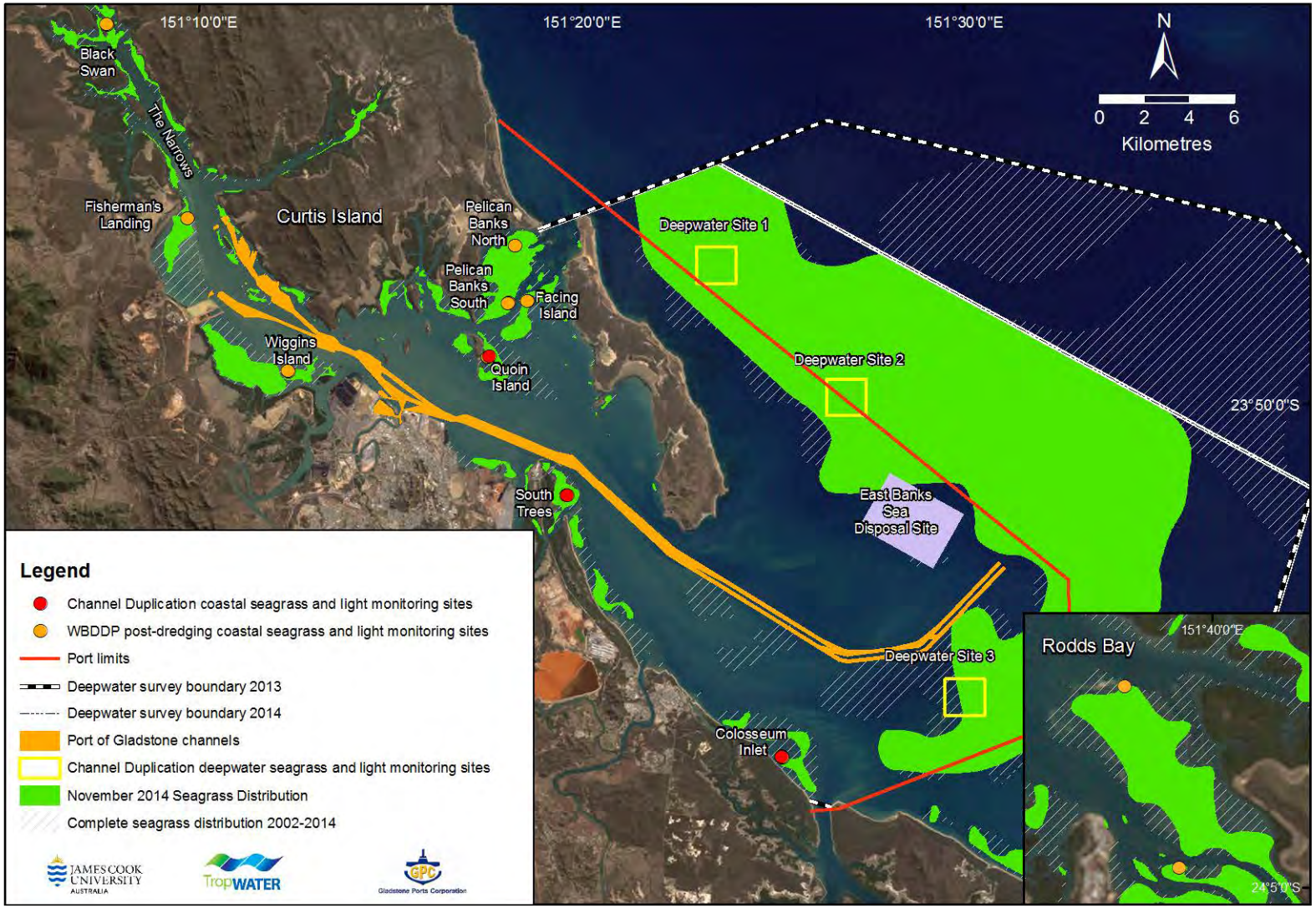
Sampling approach

Seagrass condition was monitored quarterly at each of the permanent transect locations based on the Seagrass-Watch Reef Rescue Marine Monitoring Program protocols (McKenzie et al. 2007). Changes in seagrass meadows across spatial and temporal scales were determined by monitoring abundance and community composition, elemental content of plants and meadow reproductive status. For a discussion of the latest results of monitoring at permanent transect sites between October/November 2009 and December 2014, as well as an assessment of these results in the context of climatic events occurring in Gladstone, over the duration of the program, see Davies et al. (2015).

Seagrass condition monitoring is directly linked to water quality assessments at each permanent transect location. Benthic light (Benthic Photosynthetically Active Radiation, or BPAR) 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. Light at deep water sites was collected by Vision Environment.

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 the Port Curtis and Rodds Bay.



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. Semi-annual surveys of seagrass distribution have shown that seagrasses in Port Curtis and Rodds Bay are generally at their peak in distribution and abundance during the late spring/early summer and decline during the winter months (Bryant et al. 2014; 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; 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 Port of Gladstone. A focused and more frequent assessment of seagrasses was essential in interpreting potential impacts of the WBDDP and Channel Duplication Project, and in establishing seasonal trends.

1.2 Methods

1.2.1 Coastal quarterly sampling

Each of the WBDDP post-dredging monitoring locations were sampled quarterly over the spring low tides in February, May, August and November each year (2013 - 2015), and the new Channel Duplication sites sampled quarterly from May 2014 to May 2015. 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 coastal location, sampling included three 50 m transects nested in each site and one or two sites nested in a location. A site was defined as a 50 m x 50 m area within a relatively homogenous section of the meadow (McKenzie et al. 2000). At each transect, seagrass percent cover, above-ground biomass, species composition, canopy height, macro-algae cover and epiphyte cover were recorded for a 0.25 m^2 sampling quadrat placed at 0 m and then every 5 m along the transect (11 sampling points per transect) (Figure 2a).

Those sites which were monitored by helicopter due to access issues included the Channel Duplication sites South Trees Inlet and Quoin Island, and the WBDDP post-dredging monitoring site at Black Swan (Figure 2b). Colosseum Inlet is monitoring by divers as the site does not expose completely. At these sites only seagrass percent cover, above-ground biomass, and species composition were recorded.

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}).

1.2.1 Deepwater quarterly sampling

Deepwater sites were sampled quarterly from May 2014 to May 2015. Each deepwater site consisted of three random 100m transects nested within three blocks, for a total of nine transects monitored at each

deepwater site (Figure 1). At each transect, an underwater camera was towed for approximately 100 m while footage was observed on a monitor and recorded (Figures 2c and d). Benthos on the seafloor was captured in the net and used to confirm seagrass species observed on the monitor. This technique ensured that a large area of seafloor was sampled at each site so that patchily distributed seagrass typically found in deepwater habitats was detected.

Seagrass species composition was measured from the sled net sample and from the video screen according to Kuo and McComb (1989). Seagrass biomass was estimated by an observer ranking seagrass at 10 random time frames allocated within the 100m of footage for each site. The video was paused at each of the ten time frames then advanced to the nearest point on the tape where the bottom was visible and the sled was stable on the bottom. From this frame an observer recorded an estimated rank of seagrass biomass and species composition. A 0.25 m^2 quadrat, scaled to the video camera lens used in the field, was superimposed on the screen to standardise biomass estimates and biomass determined using the same methodology as described in 1.2.1 above. Percent cover information was not collected at these deepwater sites due to the difference in data collection methodology required to assess these sites (oblique video transect recordings) compared to the shallow sites.

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 photographs 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.

1.2.4 Statistical analysis

Seagrass abundance data (percent cover and biomass) for the Wiggins Island and Facing Island sites were analysed separately to determine if any significant differences existed between original and newly established monitoring sites at these locations. All other abundance data for sites at each location were pooled.

Seagrass percent cover

Prior to analysis, data exploration protocols defined by Zuur et al. (2010) including checks for collinearity and zero inflation of percent cover data were conducted. When zero-inflation was not present, the cover data was logit transformed when appropriate to reduce the spread of the data (Warton and Hui, 2011). We then used R (R Core Team 2014)and lme4 (Bates et al. 2014) to perform a linear mixed effects analysis of the relationship between month and year and seagrass percent cover. As fixed effects, the month and year (with an interaction term) were entered into the model. Quadrat and area were entered as random effect intercepts when appropriate. When visual inspection of residual plots did not reveal any obvious deviations from homoscedasticity or normality, p-values were obtained by maximum likelihood ratio tests of the model with the effect in question against the null model. Multcomp (Hothorn et al. 2008) was then used to run pairwise comparisons for linear mixed effects models using a Tukey's HSD test with a holm adjustment for p-values.

For zero-inflated data the effects of time on percent cover was quantified using a zero/one inflated beta binomial (ZOIB) regression with the gamlss package in R (Rigby and Stasinopoulos 2005). ZOIB regression models are used to model a response variable, including percent cover data, that is bound between or equal to 0 or 1. Data were compared against the same fixed and random effects, and residual analyses were identical to non-zero inflated analysis.

Seagrass above-ground biomass

Similar to the analysis of percent cover data, following data exploration we used R and lme4 (Bates et al. 2014) to perform a linear mixed effects analysis of the relationship between month and year (with an interaction term) and non-zero inflated measures of seagrass biomass. As fixed effects, the month and year (with an interaction term) were entered into the model. As a random effect, quadrat and area were used. When visual inspection of residual plots did not reveal any obvious deviations from homoscedasticity or normality, p-values were obtained by maximum likelihood ratio tests of the model with the effect in question against the null model. Multcomp (Hothorn et al. 2008) was then used to run pairwise comparisons for linear mixed effects models using a Tukey's Honest Significant Difference (HSD) test with a holm adjustment for p-values.

For zero-inflated biomass data, a binomial-Gamma hurdle model was used to quantify the effects of month and year (with an interaction term) (Zuur et al. 2012). Linear regressions using a gamma distribution are appropriate for continuous data that are > 0 (Zuur et al. 2007). However, the Gamma distribution does not allow for zero values. Therefore, the hurdle model approach models the zero data separately from the non-zero data by first quantifying the effects of the covariates on the presence or absence of seagrass biomass with a binomial regression and then analysing the non-zero biomass data with a Gamma distribution. Following analysis with the binomial-Gamma hurdle model, all residuals were inspected visually and pairwise comparisons were made using a holm adjusted Tukey's HSD test as described previously.

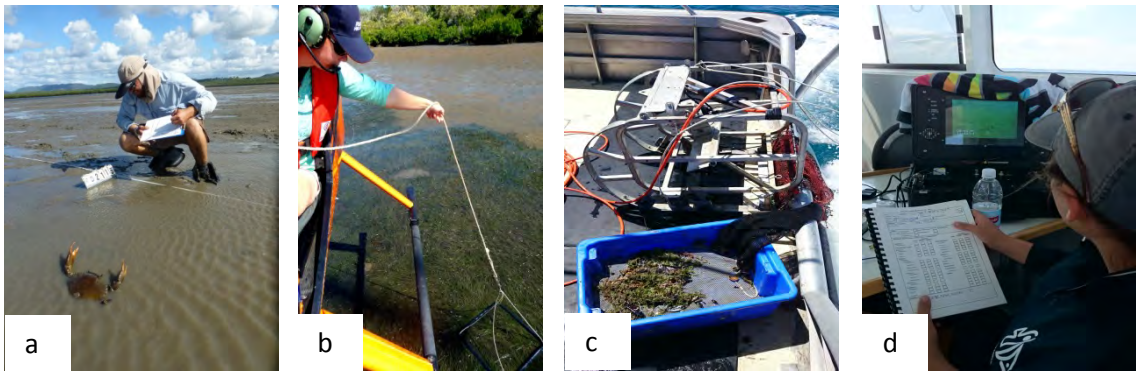


Figure 2: a) Monitoring along the transect tape at a permanent transect monitoring site during a quarterly 'on ground' survey, b) estimating seagrass abundance using aerial survey methods , c) underwater camera and sled tow equipment, and d) observing the benthos from a screen while being towed.

1.3 Results

1.3.1 WBDDP post-dredging monitoring sites

Fisherman's Landing

Seagrass percent cover at Fisherman's Landing permanent transect sites has been characterised by light and patchy seagrass cover (< 7%) since the beginning of the monitoring program in November 2009 (Davies et al. 2015). Percent cover has generally peaked between November and December, declining early in the year as seagrasses enter the senescent season. In 2014, seagrass was present throughout the entire year but cover remained very low (< 1%) for the majority of the year. Seagrass cover was greater

during the 2014 growing season (November) than 2013 (Figure 3a). Percent cover had undergone the typical seasonal decline by March 2015 (Figure 3a).

Seagrass above-ground biomass has remained very low (< 2g DW m^{-2}) over the duration of the program and followed a similar trend as percent cover, with small increases over the growing season and declines early in the year. Seagrass biomass during the growing season in November 2014 was higher than in 2013 (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 (Bryant et al. 2014).

Throughout monitoring *Halophila ovalis* has been the dominant species in terms of percent cover. *H. ovalis* has persisted after the wet season and *Z. muelleri* subsp. *capricorni* was absent until well into the growing season.

Wiggins Island

The seagrass meadow at Wiggins Island near the mouth of the Calliope River is characterised by a dynamic landscape with observations of periodical formations of deep gullies at the site and deposits of sand following major flood events (Amies et al. 2013). Seagrass percent cover at the original site was significantly higher (~30%, $p < 0.01$) when monitoring commenced than in subsequent years when seagrass cover declined markedly following substantial rainfall and river discharges in 2010 (Davies et al. 2015). In 2013, seagrasses were absent from the site over the senescent season with minimal recovery (~3% cover) over the growing season, remaining well below historical levels.

Due to the development of the Wiggins Island Coal Export Terminal (WICET) near the original site, an additional permanent transect site was established on the western side of the bank in December 2011 and the original site was decommissioned in May 2014. Seagrass biomass and percent cover at the new site are low but have remained stable since 2013 (Figure 4b).

The species composition over the first few years of the program shifted seasonally between a *Z. muelleri* subsp. *capricorni* and a *H. ovalis* dominated community. At the new site in 2013 *Z. muelleri* subsp. *capricorni* was the dominant species over the senescent season with patches of *H. ovalis* increasing over the growing season. In 2014, *H. ovalis* remained in greater proportions than seen previously and for the first time in March 2015 the site was exclusively dominated by *H. ovalis* (Figure 5b). By May 2015, *Z. muelleri* subsp. *capricorni* had returned as the dominant species.

Black Swan Island

Seagrass cover at the Black Swan Island site declined over the latter part of 2012 and into the 2013 senescent season, remaining extremely low (<3%) in 2014 (Figure 6). The site has been dominated by *Z. muelleri* subsp. *capricorni*; however this trend reversed in February and May 2014 with the complete replacement of *Z. muelleri* subsp. *capricorni* at the site by *H. ovalis* (Davies et al. 2015). In August 2014, *Z. muelleri* subsp. *capricorni* returned as the dominant species and remained through to May 2015 (Figure 7).

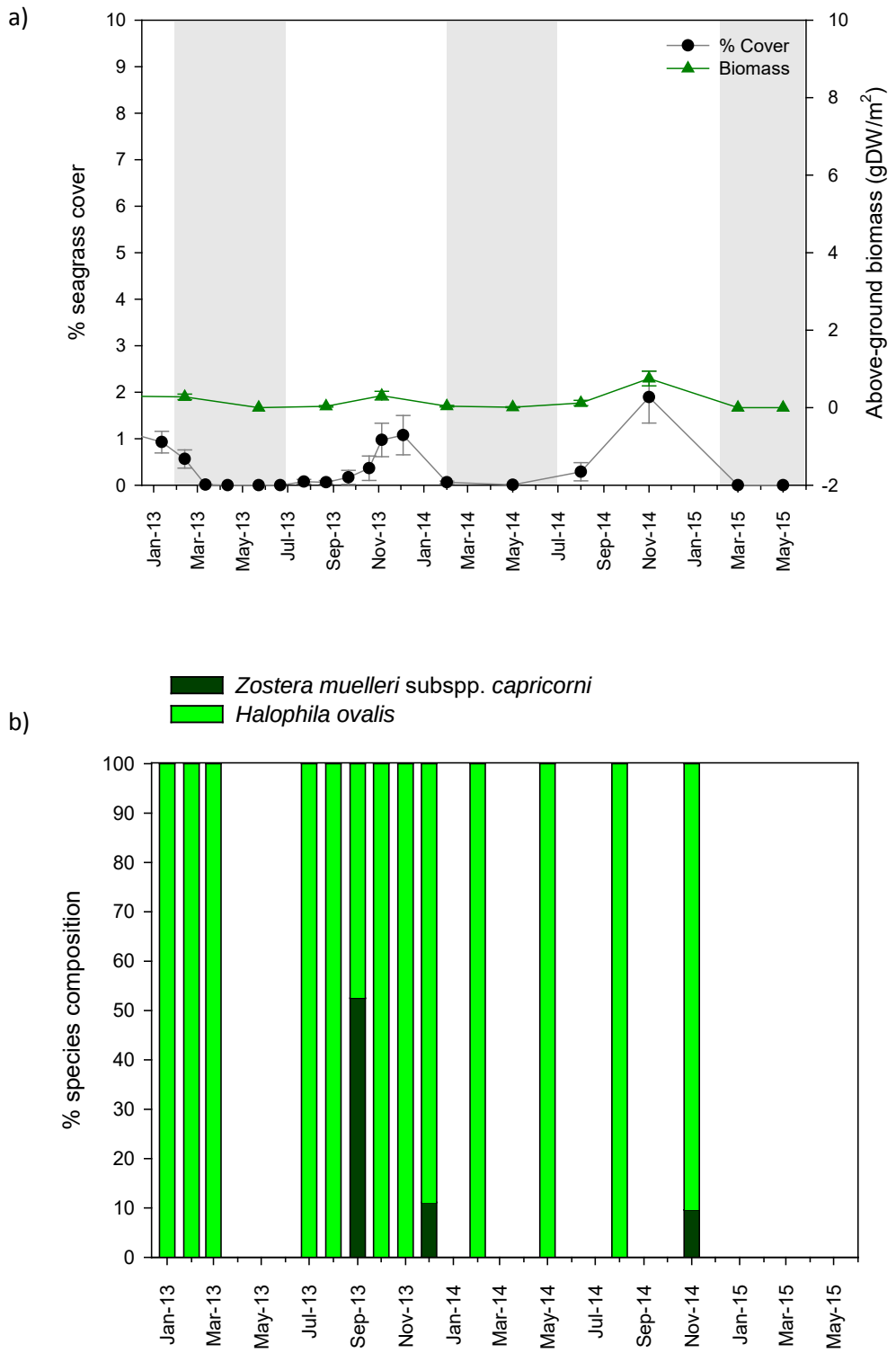
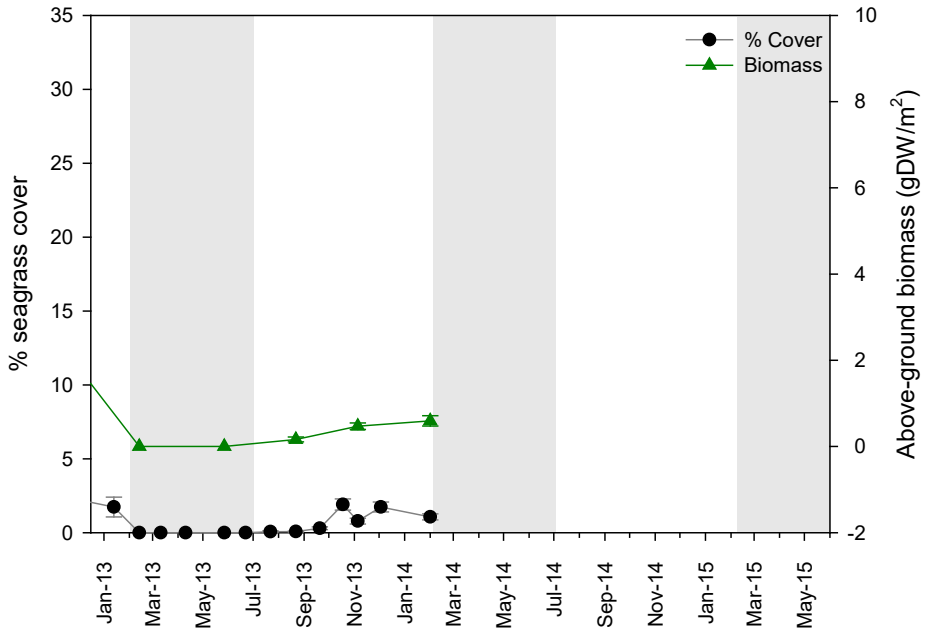


Figure 3: a) Seagrass abundance and b) seagrass species composition at permanent transect sites (pooled) at Fisherman's Landing, January 2013 – May 2015. Shaded area represents the seagrass senescent season.

a)



b)

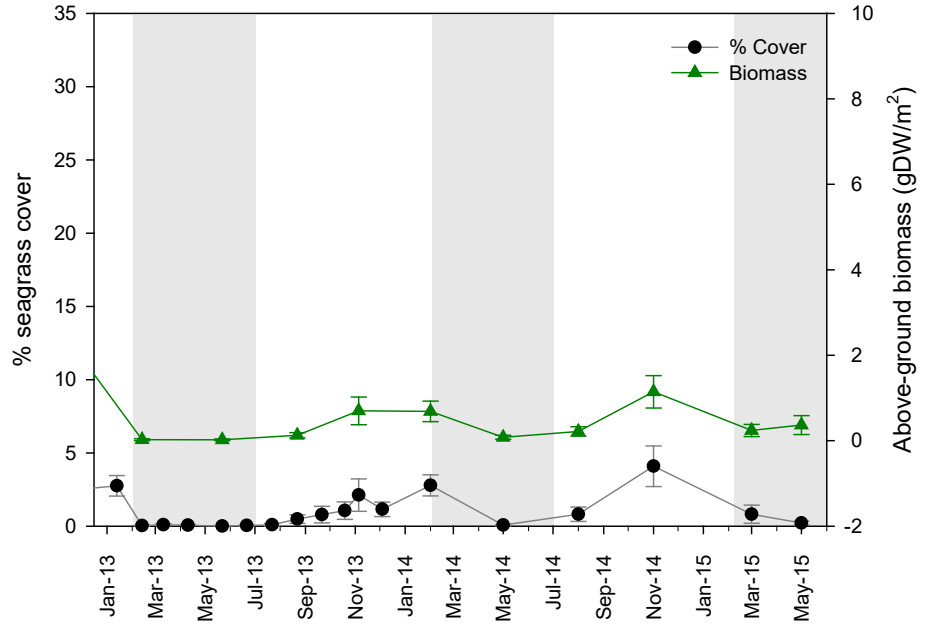


Figure 4: Seagrass abundance at the a) original and b) new permanent transect site at Wiggins Island, January 2013 – May 2015. Shaded area represents the seagrass senescent season.

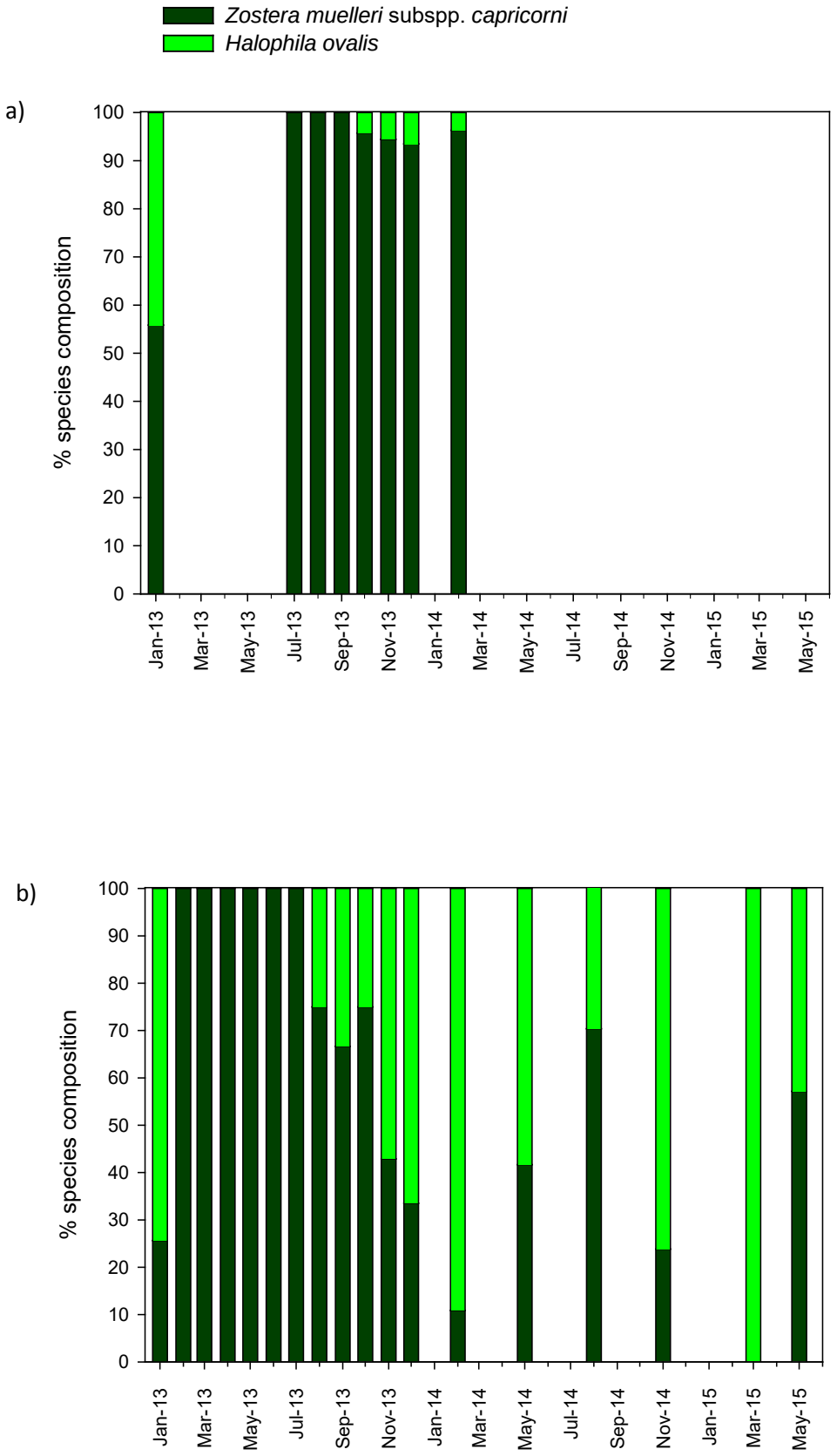


Figure 5: Seagrass species composition at the a) original and b) new permanent transect site at Wiggins Island, January 2013 – May 2015. The original site was discontinued following the February 2014 survey.

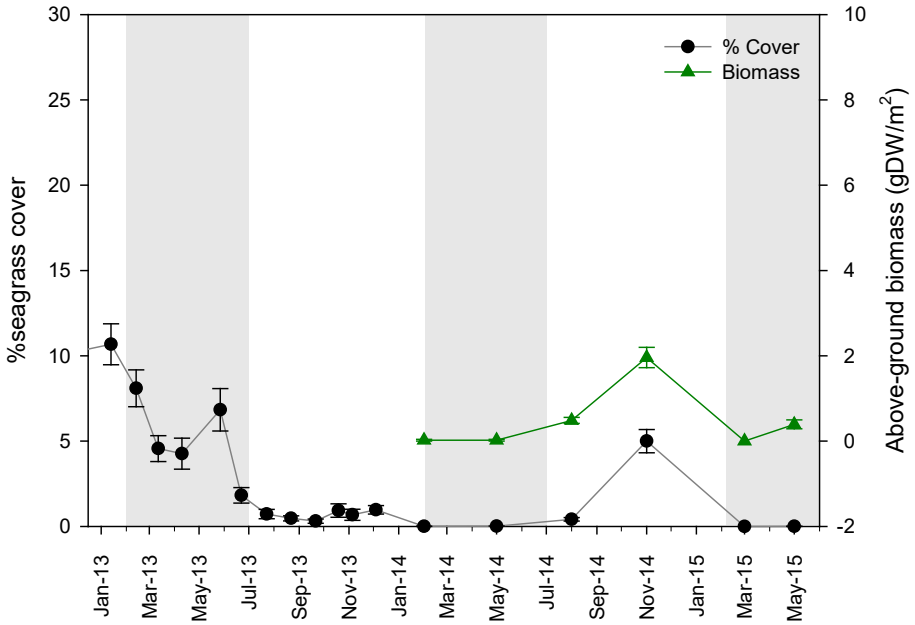


Figure 6: Seagrass abundance (mean ± SE) at Black Swan Island permanent transect site, January 2013 – May 2015.

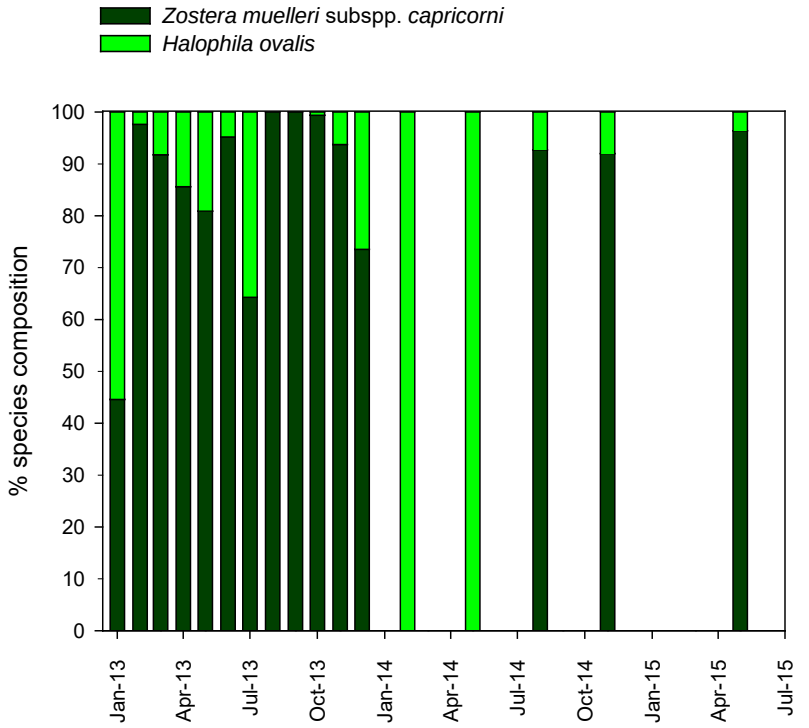


Figure 7: Seagrass species composition at Black Swan Island permanent transect sites, January 2013 – May 2015.

Pelican Banks North

Declines in early 2013 followed major flood events. In 2014, percent cover did not decline as sharply as in previous years following the wet season, however during the growing season seagrass did not recover to the same peaks observed in previous years resulting in significantly lower percent cover in November 2014 than in 2013 ($p < 0.01$). Above-ground biomass was also generally low in 2014 with less recovery than recorded in November of previous years and significantly lower than peaks observed in November 2013 ($p < 0.01$) (Figure 8a). Lower cover during the 2014 growing season meant that when the seagrasses started the seasonal decline in early 2015, they were declining from a much lower standing crop, and percent cover was significantly lower by May 2015 than the last two years ($p < 0.01$).

Despite this seagrasses at Pelican Banks North generally remain the most abundant in terms of percent cover and above-ground biomass in Port Curtis and Rodds Bay. The species composition at Pelican Banks North remains almost exclusively *Z. muelleri* subsp. c*apricorni* with only very minor proportions of *H. ovalis* and *H. uninervis* present at times (Figure 9a).

Pelican Banks South

Seagrass percent cover at Pelican Banks South has followed a similar pattern to Pelican Banks North, however unlike Pelican Banks North, the southern sites failed to recover during the 2013 growing season and both seagrass percent cover and above-ground biomass have remained relatively low (Figure 8). Peak cover over the 2013 and 2014 growing seasons was significantly lower than previous years ($p < 0.01$ for 2010 – 2014) (Figure 8b). Seagrass biomass followed a similar trend to percent cover and throughout 2014 and the beginning of 2015 was very low. Seagrass biomass recovered during the 2013 growing season but remained significantly below peaks observed in previous years ($p < 0.01$ for all years) (Davies et al. 2015).

The species composition at Pelican Banks South is predominantly *Z. muelleri* subsp. *capricorni* but has greater proportions of other species present (*H. ovalis*, *H. decipiens, H. spinulosa* and *H. uninervis*) than at Pelican Banks North (Figure 9). In 2014, the proportion of *H. uninervis* greatly increased, becoming the most dominant species for the first time in May and remaining throughout the growing season and into early 2015 (Figure 9).

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 (Davies et al. 2015). Peaks in seagrass percent cover have typically occurred between November and January each year. Seagrass above-ground biomass over the 2014 growing season was similar to 2013 ($p = 1.00$) (Figure 10a).

Seagrass abundance at the new site remains higher than the original site (Figure 10a). However, while there were no significant differences in above-ground biomass at the new Facing Island site between 2013 and 2014 with the exception of May 2013 and May 2014 ($p = 0.02$), percent cover was significantly lower in 2014 than in 2013 ($p < 0.01$).

The species composition at the new site is predominantly *Z. muelleri* subsp. *capricorni,* while the original site is comprised of larger proportions of the smaller *H. ovalis* and *H. uninervis* species (Figure 11). The new site had greater proportions of *Halophila* spp. during 2014 (Figure 11b). For the first time since monitoring began, the original site was entirely composed of *H. uninervis* from February to August 2014 with the return of *Halophila* spp. during the growing season (Figure 11a).

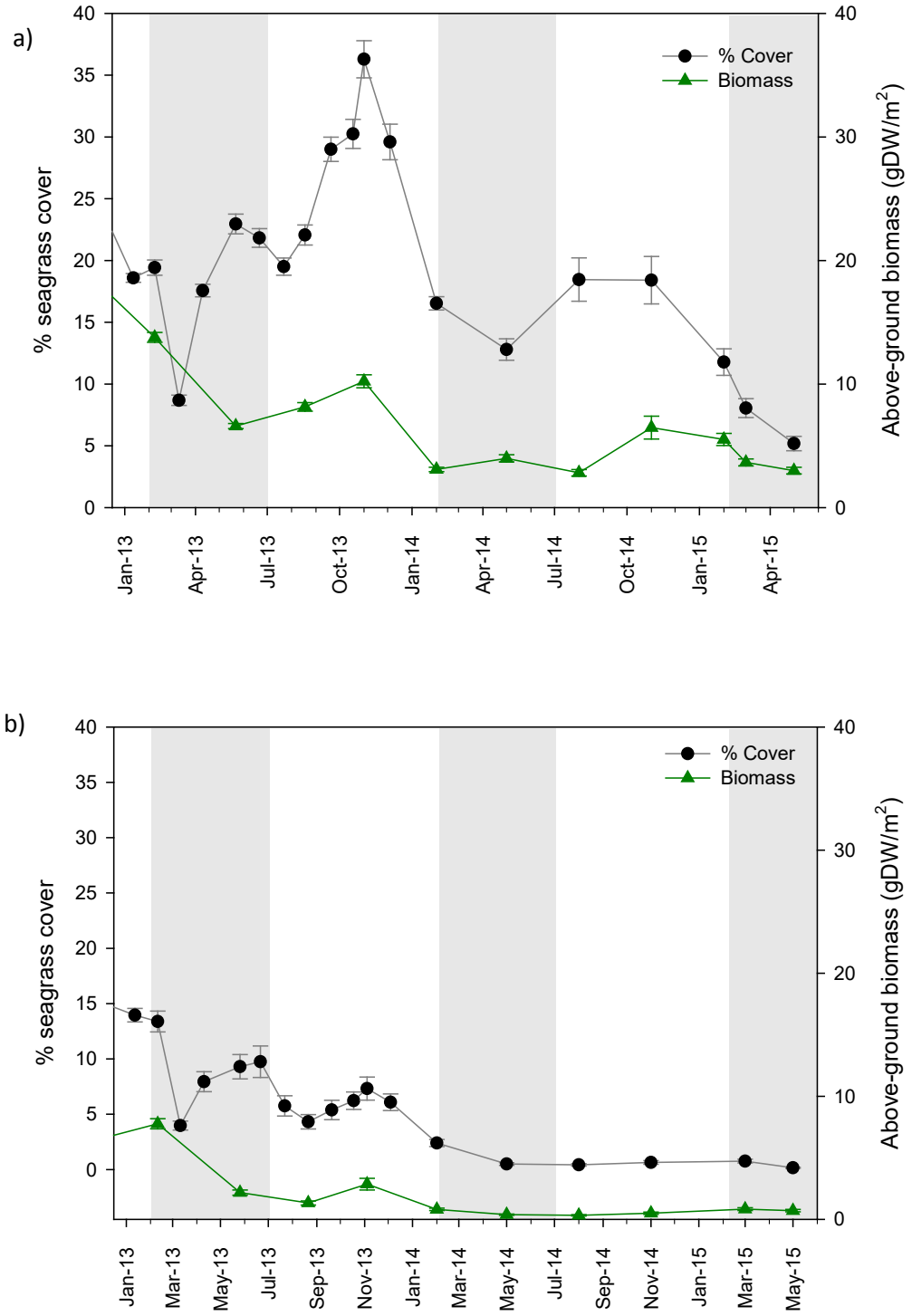


Figure 8: Seagrass abundance at a) Pelican Banks North and b) Pelican Banks South permanent transect sites (pooled), January 2013 – May 2015. Shaded area represents the seagrass senescent season.

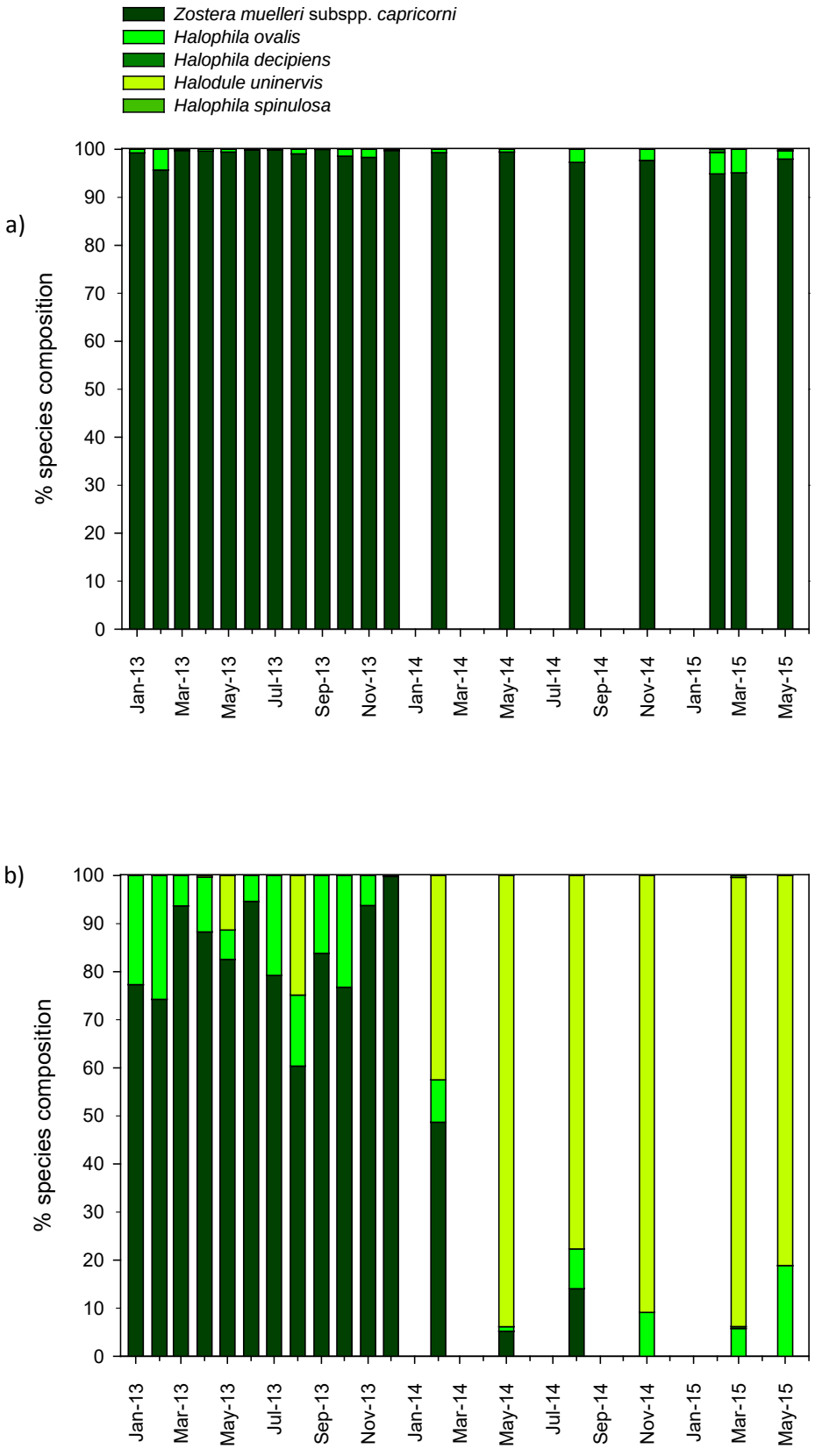


Figure 9: Seagrass species composition at a) Pelican Banks North b) and Pelican Banks South permanent transect sites (pooled), January 2013 – May 2015.

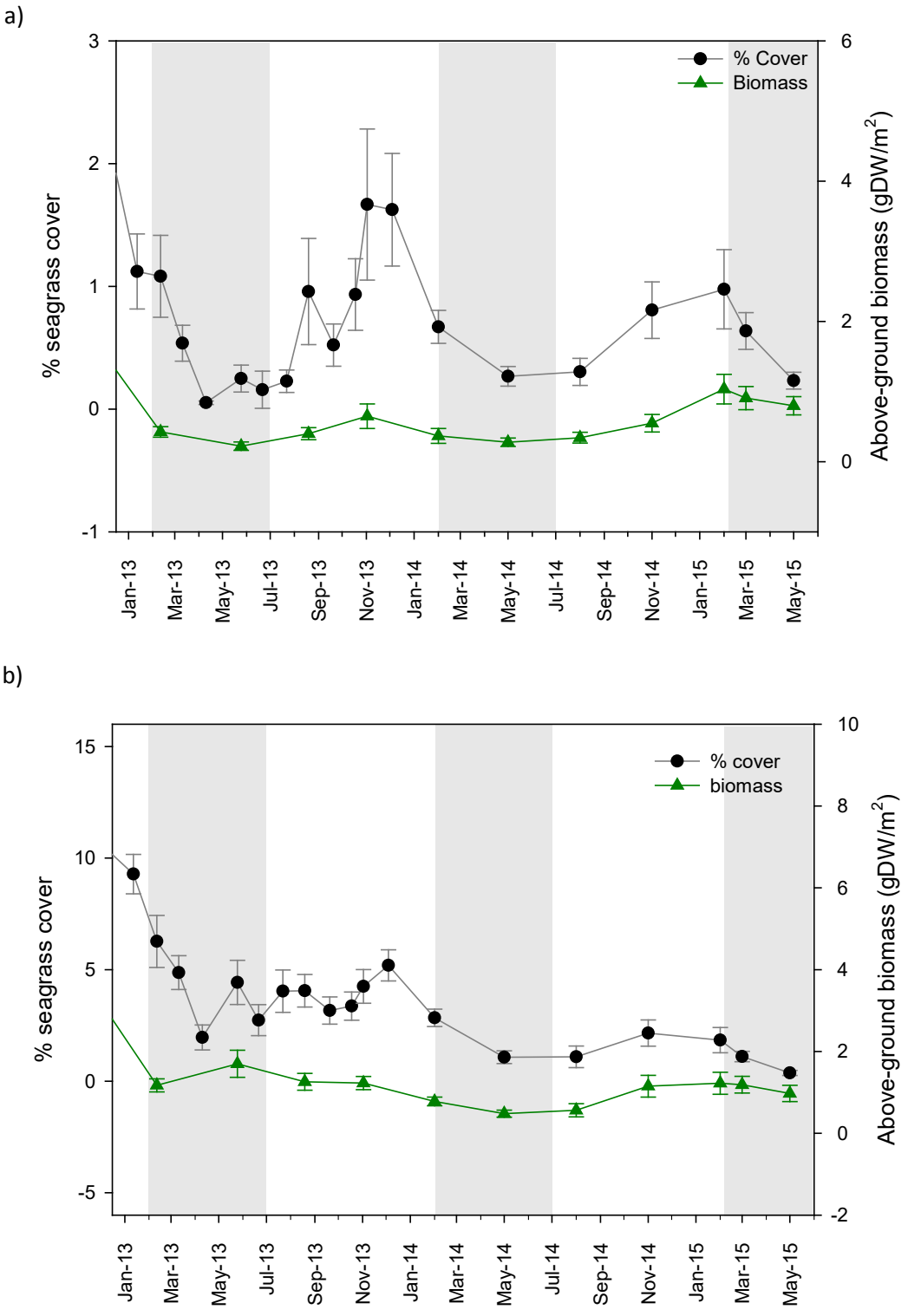


Figure 10: Seagrass abundance at the a) original site and b) new site at Facing Island, January 2013 – May 2015. Shaded area represents the seagrass senescent season.

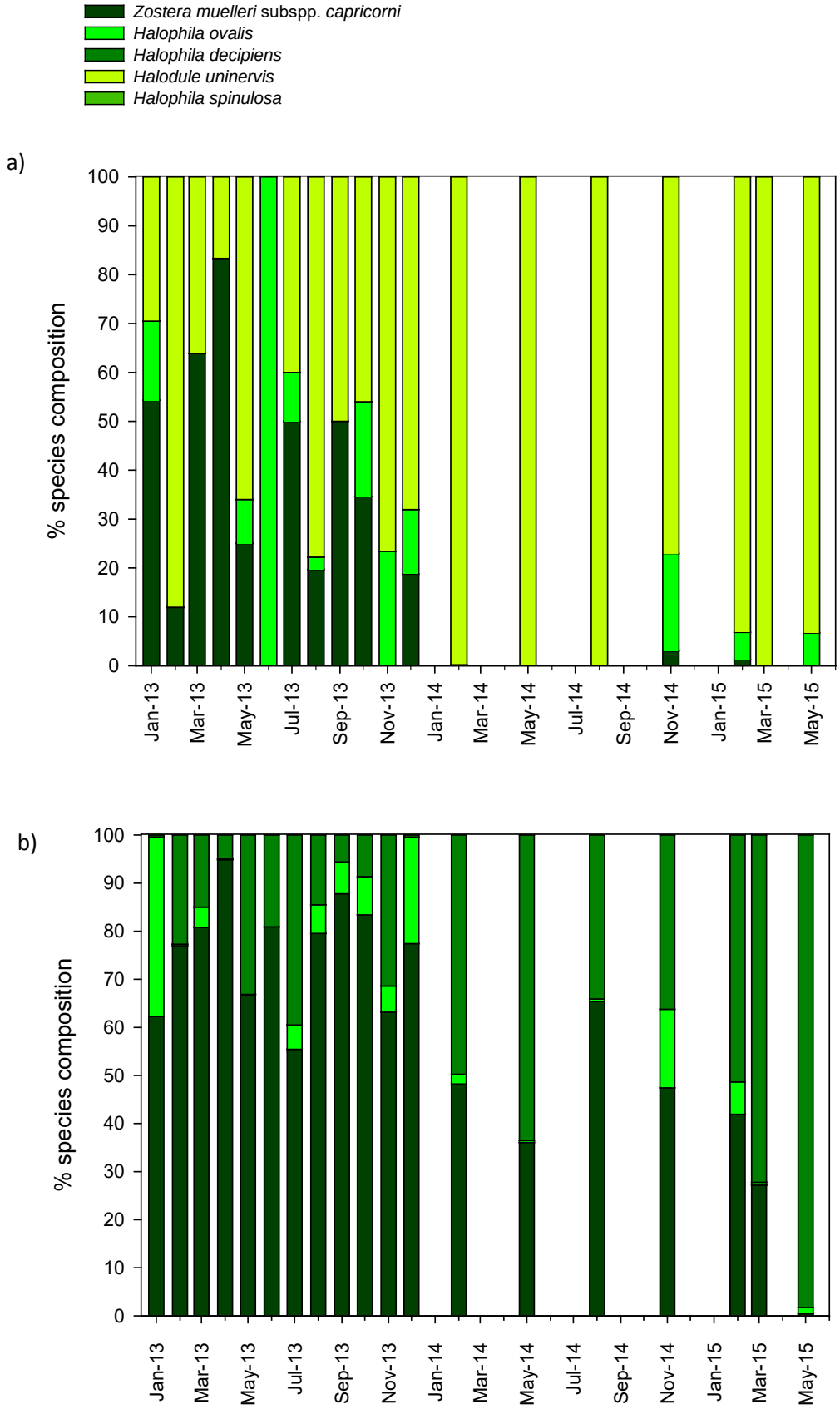


Figure 11: Seagrass species composition at the a) original and b) new permanent transect sites at Facing Island, January 2013 – May 2015.

1.3.2 Out of Port reference site

Rodds Bay follows a seasonal pattern with greater abundance in the late dry season and lower abundance following the onset of the wet season. Seagrass cover has remained low following the 2010/2011 extreme wet season (Figure 12). Patches of seagrass have come and gone seasonally since this time but seagrass cover and biomass have remained extremely low. In 2013, some shoots (mostly *Z. muelleri* subsp. *capricorni* remained at transects throughout the year (Figure 13a). Both seagrass cover and above-ground biomass were the highest during the 2014 growing season since peaks observed in 2007 (Davies et al. 2015) and had entered the senescent season by March 2015 (Figure 13a).

1.3.3 Channel Duplication coastal monitoring sites

Quoin Island

Quoin Island has had moderate seagrass cover (<10 %) since monitoring began in May 2014 (Figure 14a), and shows a seasonal trend with peaks in percent cover and biomass between November and March. Biomass has followed a similar trend, although the peak in percent cover (November) preceded the peak in biomass (March). The site is dominated by *H. uninervis* with some *H. ovalis,* and a very small proportion of *H. spinulosa* (Figure 14b).

South Trees Inlet

Seagrass percent cover and biomass at the South Trees Inlet permanent transect site have had an increasing trend in cover and biomass during 2014, peaking in March 2015 (Figure 15a) similar to biomass at Quoin Island. While both cover and biomass declined in May 2015, percent cover fell by over 7 %, while biomass only showed a slight decline. The species composition has remained nearly exclusively *Z. muelleri* subspp. *capricorni* since monitoring began in May 2014, with the inclusion of a very small proportion of *H. spinulosa* (Figure 15b).

Colosseum Inlet

Seagrass percent cover and biomass at Colosseum Inlet also followed a seasonal trend, peaking in November 2014 (Figure 16a). Both percent cover and biomass in May 2015 were lower than May 2014 (Figure 16a). The site has been exclusively dominated by *H. uninervis* since monitoring began (Figure 16b).

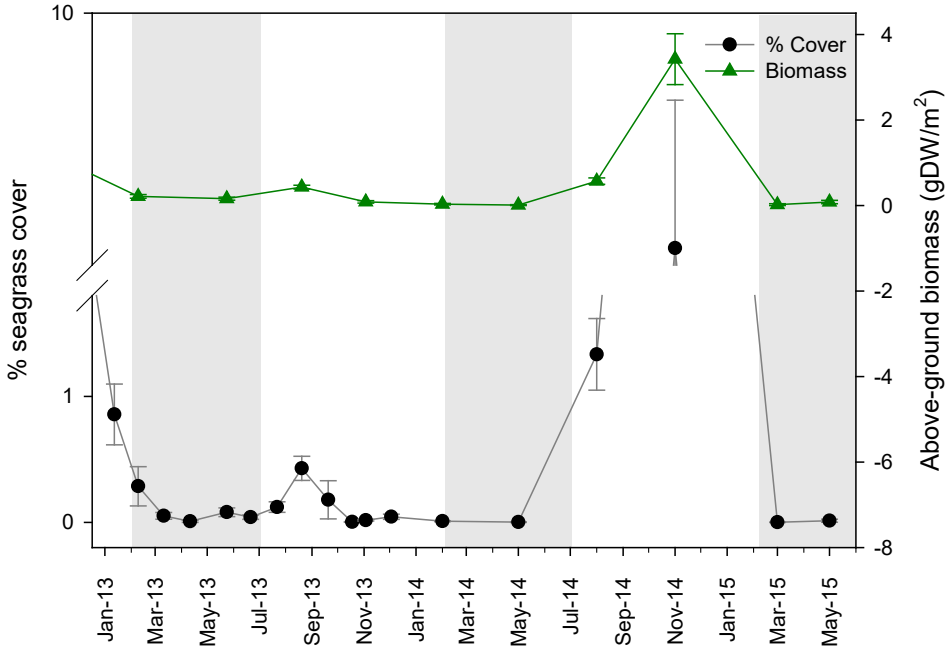


Figure 12: Seagrass abundance at Rodds Bay, January 2013 – May 2015. Shaded area represents the seagrass senescent season.

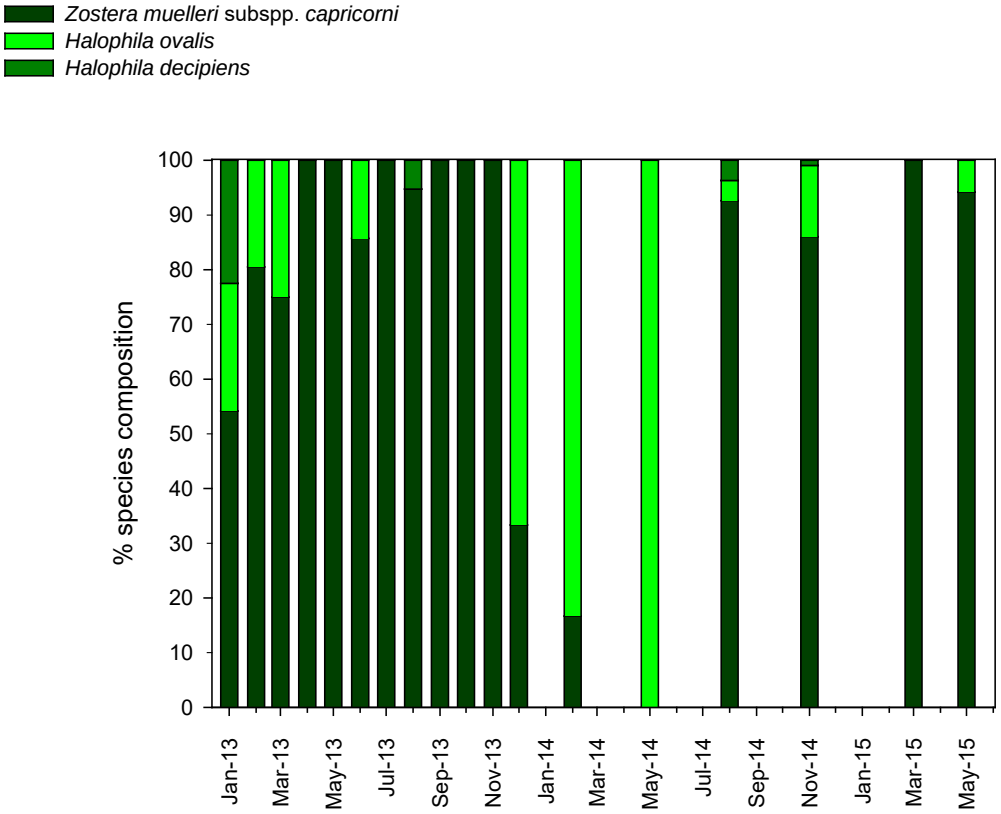


Figure 13: Seagrass species composition at Rodds Bay, January 2013 – May 2015.

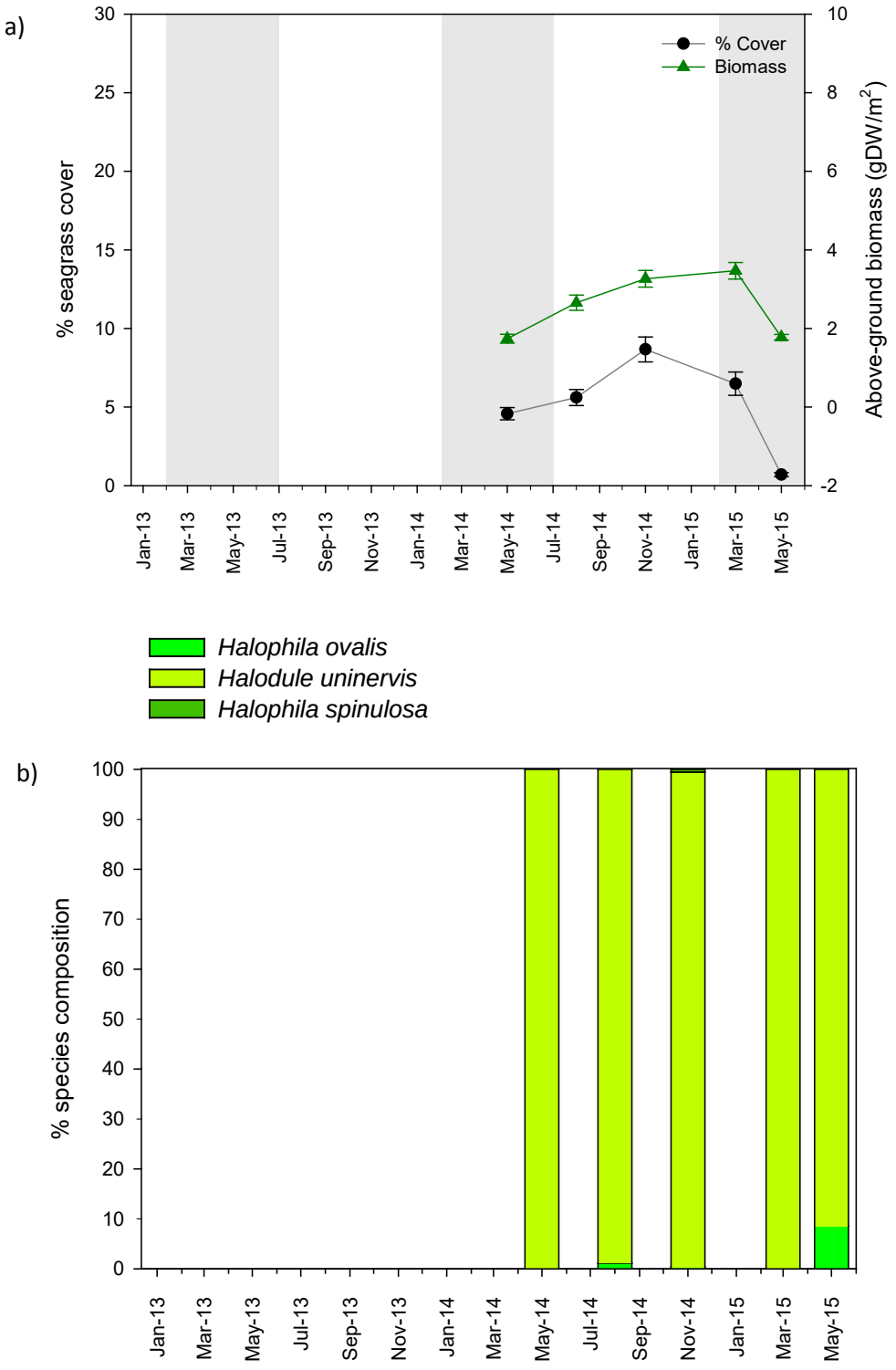


Figure 14: Seagrass a) abundance and b) species composition at permanent transect sites at Quoin Island, May 2014 – May 2015.

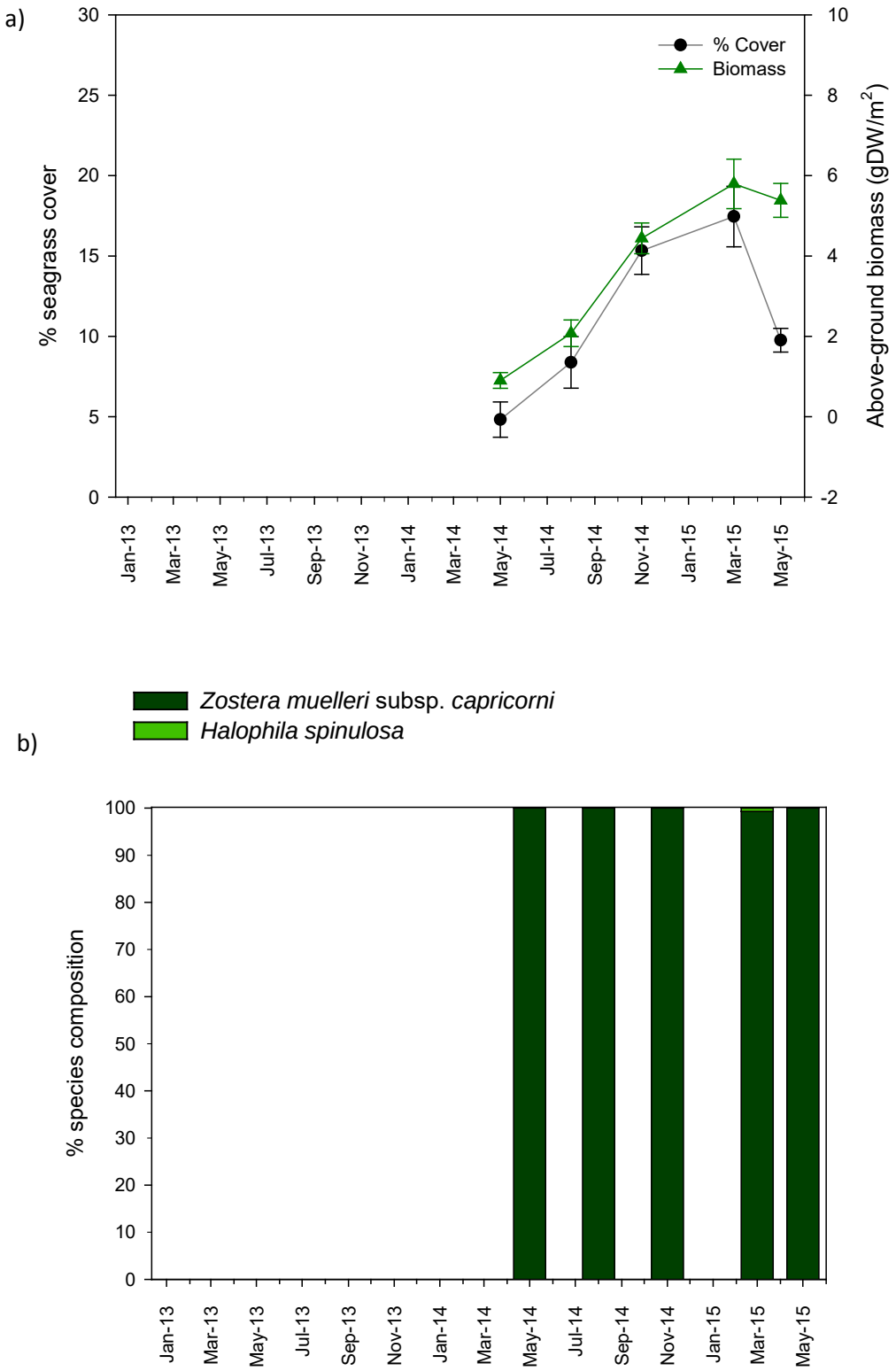


Figure 15: Seagrass a) abundance and b) species composition at permanent transect sites at South Trees, May 2014 – May 2015.

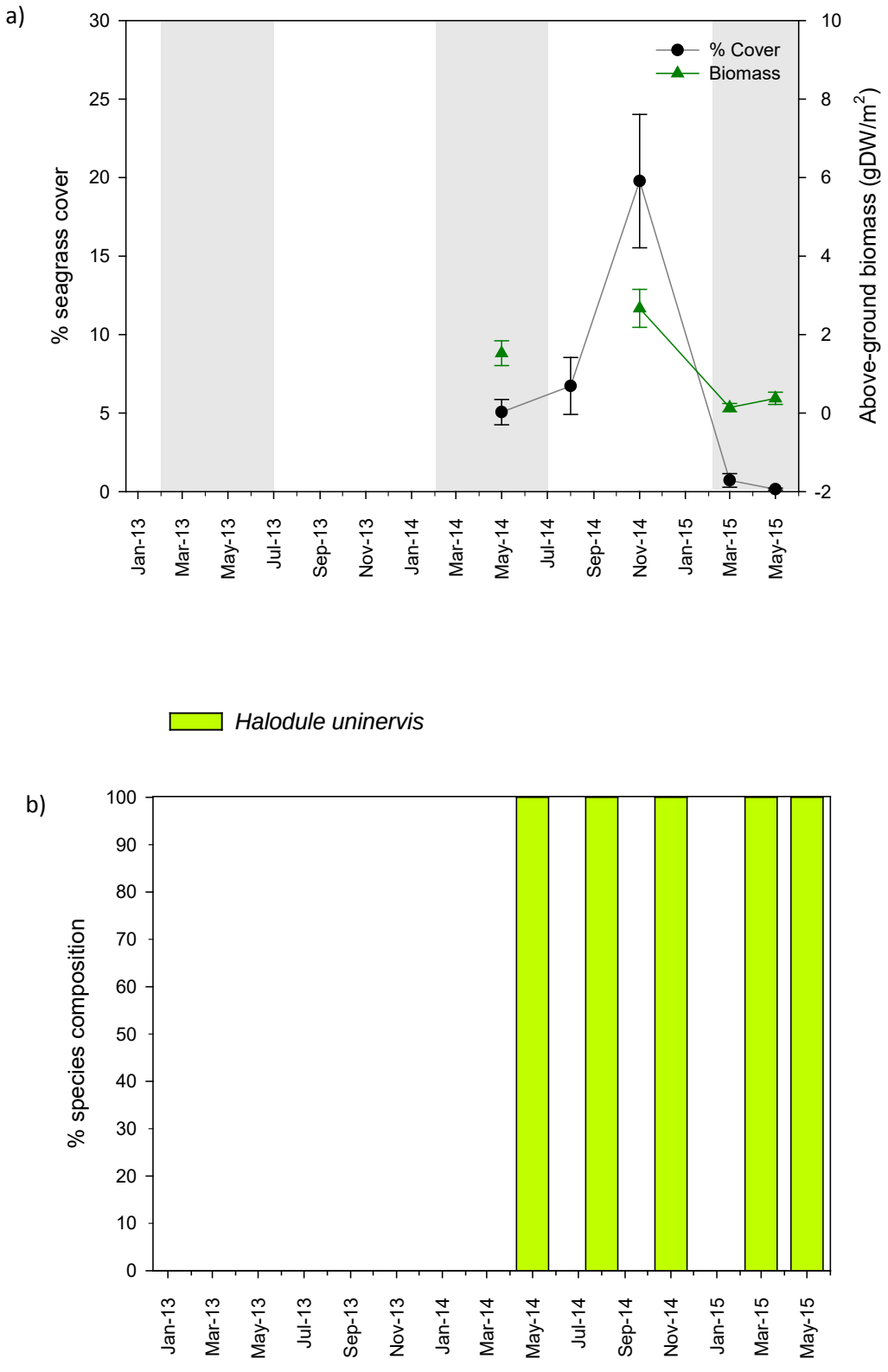


Figure 16: Seagrass a) abundance and b) species composition at permanent transect sites at Colosseum, May 2014 – May 2015.

1.3.4 Channel Duplication deepwater monitoring sites

Deepwater site 1

Seagrass biomass at deepwater site 1 has been extremely low (<0.3 g DW m^{-2}) since monitoring began; above-ground biomass peaked in September 2014 and had declined by November 2014 (Figure 17a). Biomass at this site appears to peak early and had already begun to decline by November when the majority of coastal sites reach their maximum production. Seagrass was absent from the site in March 2015 and May 2015 (Figure 17a). Species composition at the site is dominated by *H. ovalis*, with the appearance of *H. decipiens* confined to the first sampling event in June 2014 (Figure 17b).

Deepwater site 2

Seagrass biomass at deepwater site 2 is much higher than at site 1, peaking at 0.78 g DW m^{-2} in November 2014 (Figure 18a). Seagrass at this site had also been completely lost by March 2015 (Figure 18a). The species composition has been dominated by *H. ovalis* with some *H. spinulosa* found in November 2014 (Figure 18b).

Deepwater site 3

Seagrass biomass at deepwater site 3 is the lowest of the three deepwater sites, with only a few shoots found in September and November 2014 (Figure 19a). The site is exclusively dominated by *H. ovalis* (Figure 19b).

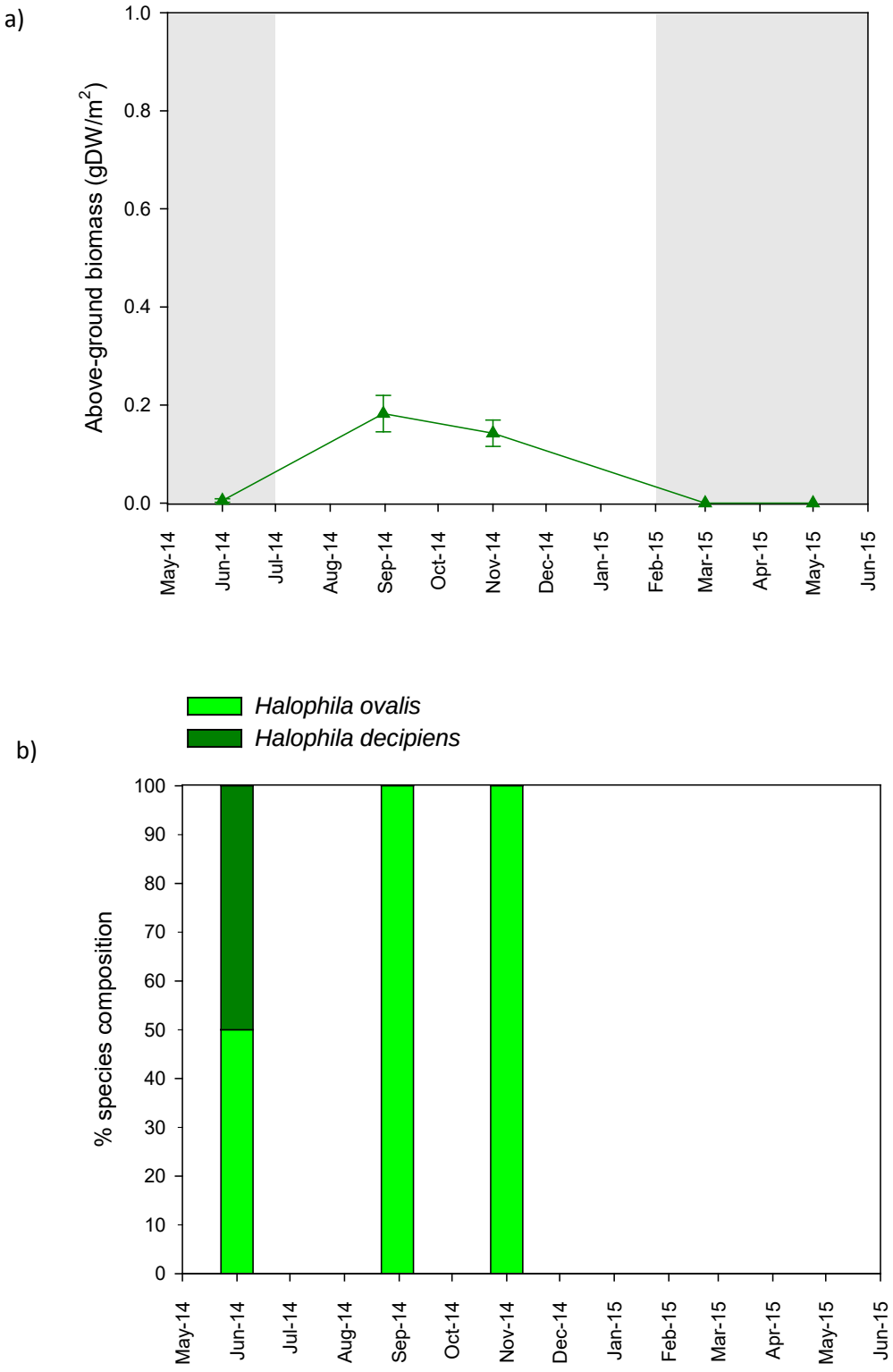


Figure 17: Seagrass a) abundance and b) species composition at deepwater site 1, June 2013 – May 2015.

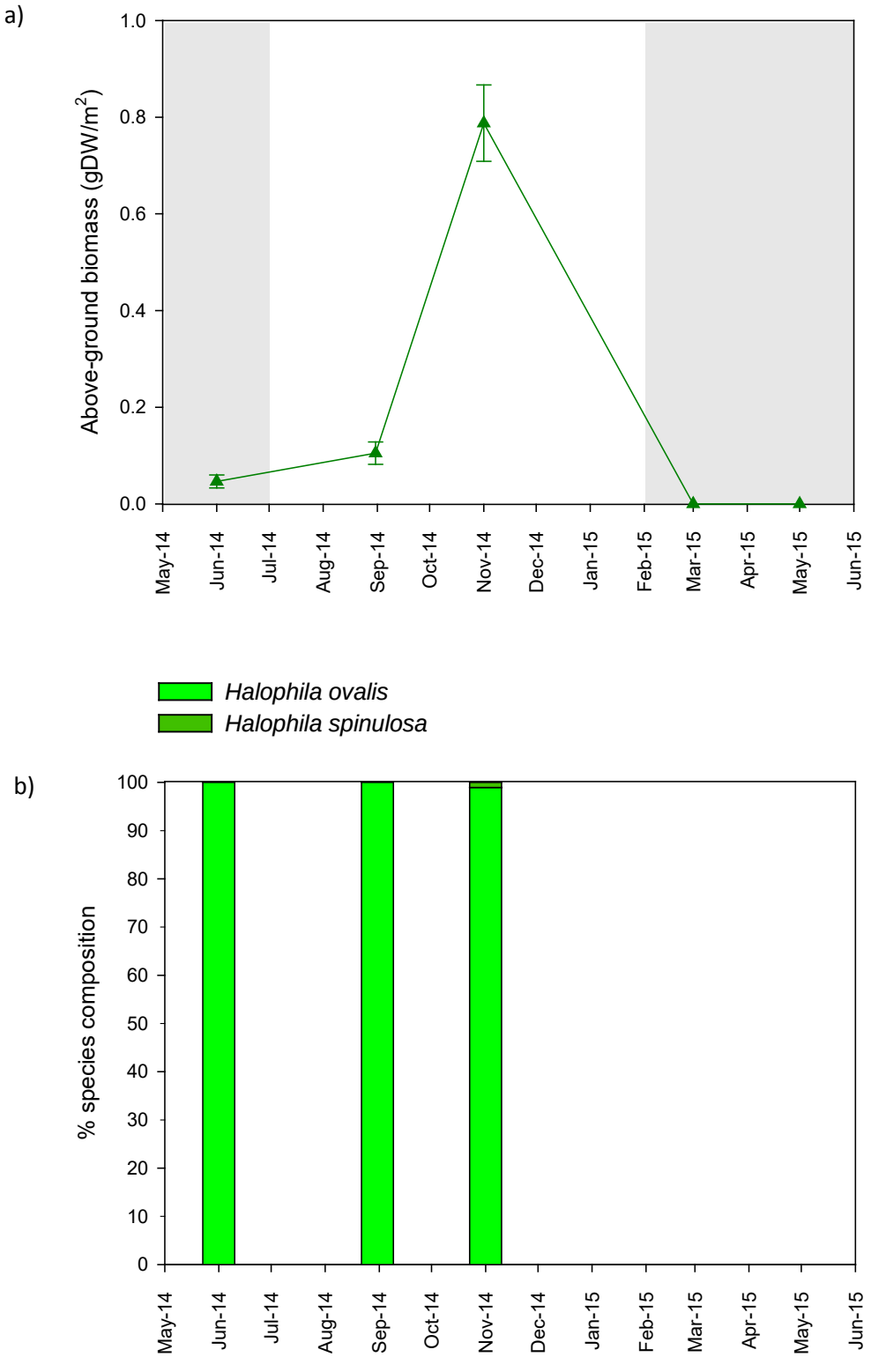


Figure 18: Seagrass a) abundance and b) species composition at deepwater site 2, June 2013 – May 2015.

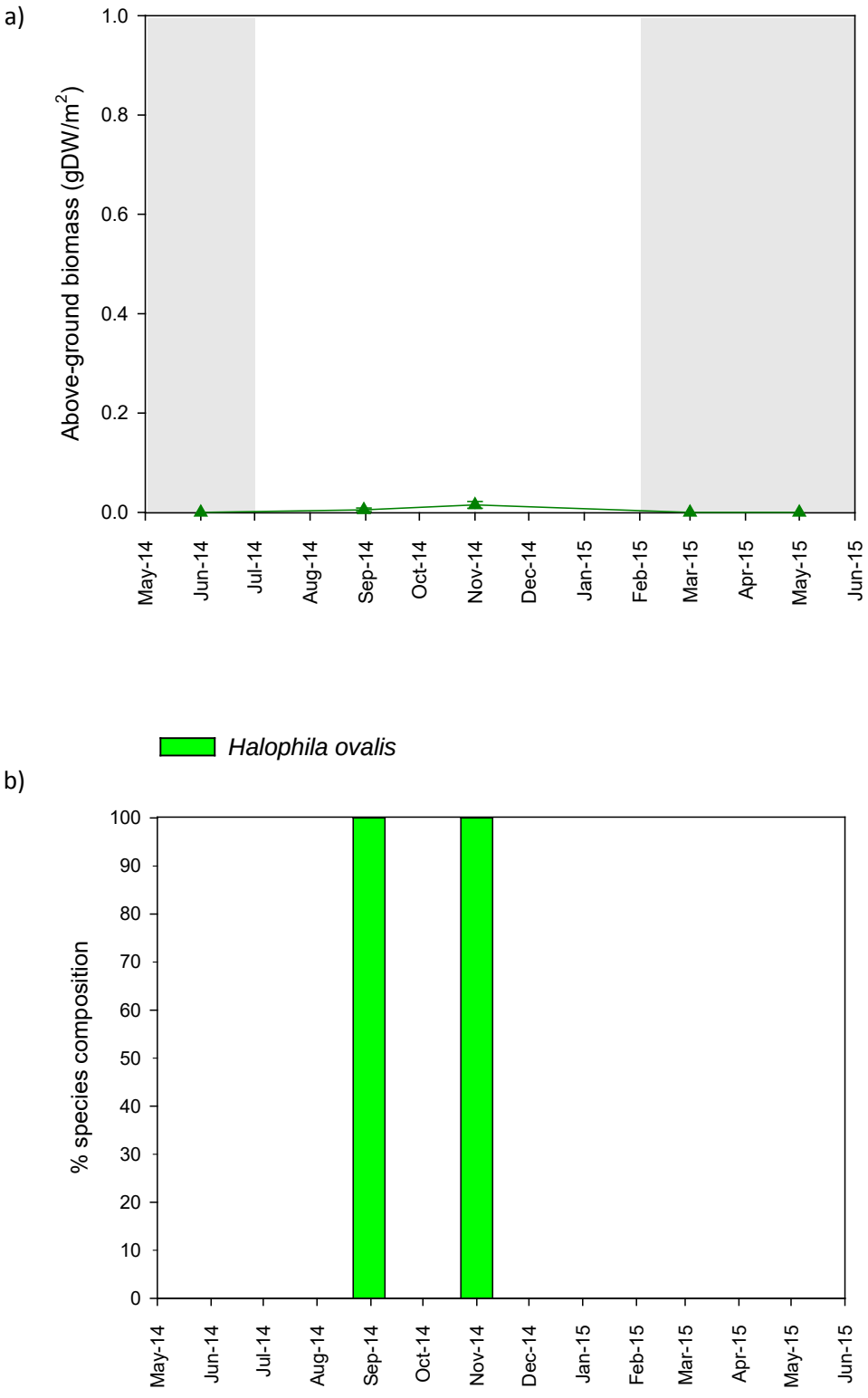


Figure 19: Seagrass a) abundance and b) species composition at deepwater site 3, June 2013 – May 2015.

1.4 Discussion

Monitoring at coastal permanent transects and deepwater sties revealed that seagrasses 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 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. Sites located within the inner harbour also underwent seasonal trends in species composition, which were not as pronounced at the more stable sites such as Pelican Banks. At the deepwater sites, all three sites are dominated by *Halophila* spp. with extremely low biomass and show a seasonal trend, with peaks in biomass in September or November and generally a complete loss of all above-ground plant components during the senescent season.

Halophila species typically found in deep water meadows are naturally variable in their abundance (Carter et al. 2015; York et al. 2015a; Rasheed et al. 2014). Surveys of deepwater meadows in other Queensland locations, including Gladstone, Townsville and Abbot Point have shown this type of seagrass habitat has a high level of inter-annual variability (Carter et al. 2015; Davies et al. 2014; Rasheed et al. 2014). Due to the dominance of these deepwater meadows by highly variable species, these meadows are highly ephemeral and in many areas are only present during the peak growing season, with complete loss of seagrass or extremely low biomass for the remainder of the year (McKenna et al. 2015a). *Halophila* spp. are known to decline rapidly in low light conditions (Durako et al. 2003; Kenworthy et al. 1989) but when environmental conditions improve, may respond quickly with fast growth and recovery (Rasheed et al. 2014; Hammerstrom et al. 2006).

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, Facing Island, and Pelican Banks South there have been major declines in seagrass abundance since sites were established in 2009 (Davies et al. 2015). The timing of flood related declines in seagrass abundance immediately prior to the capital dredging activities associated with the WBDDP, which occurred between May 2011 and September 2013, makes it difficult to ascertain what impact dredging may have had on seagrass condition at the permanent transect sites.

As part of the WBDDP dredging management program (i.e. Dredge Management Plan, Water Quality Management Plan and seagrass monitoring), thresholds and triggers were 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 were receiving enough light for their growing requirements at these seagrass sites. The dredging management program was largely successful in achieving this and generally seagrass growth during the growing season was not limited by light falling below requirements (outside of major flood events) (refer 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 shown 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 Port Curtis, 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 light (McCormack et al. 2013)

The results described in the above sections indicate that while many seagrass meadows in Port Curtis have undergone reduced percent cover and mean biomass compared to observations from earlier in the monitoring program, many meadows are either showing signs of recovery in cover, biomass and species composition or are in a good position to do so if conditions remain favourable for seagrass growth. 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). 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 and productivity of Port Curtis seagrass meadows are discussed in detail in Sections 3 and 4 of this report.

2 ENVIRONMENTAL DRIVERS OF SEAGRASS CHANGE

2.1 Background and approach

Seagrass abundance and distribution are 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. In review; McCormack et al. 2013; 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 dredging 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). Seagrasses grow in a broad range of conditions from shallow coastal waters and areas of high turbidity, to deepwater areas where the quantity and quality of light reaching the seagrass bed may be highly variable. The effect of changes in the availability and quality of light reaching seagrasses differs among species and seasons (Lee et al. 2007b; Ralph et al. 2007; Erftemeijer and Lewis 2006; Hemminga and Duarte 2000; Bjork et al. 1999).

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 elevate 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 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, which if sustained can eventually lead to the loss of seagrass biomass and cover (York and Smith 2013).

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. 1994a) or morphological such as changes to biomass, numbers of shoots or canopy height (Collier et al. 2012a; 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. 2012b).

2.2 Methods

2.2.1 Monitoring of environmental conditions

Local climate conditions

Environmental data on rainfall (mm) and river discharge (Megalitres (ML)) 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 within seagrass meadows

At the beginning of the WBDPP monitoring program benthic light loggers, deployed and maintained by Vision Environment, were positioned adjacent to each permanent transect location to compare light with known seagrass condition (see Bryant et al. 2013; Vision Environment QLD). In December 2013, light monitoring loggers were deployed by JCU TropWATER at the seven locations identified for WBDDP post-dredging monitoring and at an additional three sites (Colosseum, South Trees and Quoin Island) in May 2014 to address areas of potential impact from the Channel Duplication Project. Light data is also monitored by Vision Environment at offshore water quality monitoring sites adjacent to the seagrass deepwater monitoring sites. Temperature was monitored in the seagrass meadows concurrently with benthic irradiance by a temperature logger (Tinytag Aquatic 2, Gemini Data Loggers (UK) Ltd) attached to the wiper unit associated with each light logger.

After data deconfounding (Vision Environment QLD), a rolling mean total daily irradiance at each location was calculated over a two week period to incorporate both spring and neap tide conditions as well as the variation in tide height and the degree of exposure that affects the light conditions reaching the seagrass (Chartrand et al. 2012). The total daily irradiance for a given day and the previous 13 days were averaged and the calculation rolled forward each day respectively. The rolling mean irradiance values were plotted against seagrass percent cover recorded at each of the monitoring locations.

2.2.2 Light threshold ranges for seagrasses in Port Curtis

As part of the baseline data collection for the Channel Duplication Project EIS, initial seagrass light threshold ranges are provided for inclusion into the EIS impact assessment and management plans to address the potential seagrass impacts within the predicted impact zones. The light requirement threshold ranges are defined as the point at which a change in external conditions will likely cause a significant negative change in seagrass condition (measured as above-ground biomass) and have been based on the information available at the time of this report.

In order to develop these light threshold ranges, a literature review was conducted as well as an investigation into current research conducted on light requirements for tropical seagrasses. Once a range of biologically relevant light values were established from the literature review and ongoing studies, these values were used as a starting point to examine light and seagrass biomass trends at key sensitive receptor sites monitored as part of this program.

The seagrass and light assessment sites were chosen based on the dominant species and the abundance of seagrass that would most likely allow for analysis of the emerging trends in light and seagrass data. The sites chosen for *Z. muelleri* subspp. *capricorni* were Pelican Banks North (post-dredging monitoring) and South Trees Inlet (Channel Duplication). The sites chosen for *H. uninervis* were Facing Island (post-dredging monitoring) and Quoin Island (Channel Duplication). The sites chosen for *H. ovalis* were the offshore deepwater monitoring sites 1 and 2, as these sites consistently had the greatest biomass of this species.

For coastal sites dominated by *Z. muelleri* subspp. *capricorni* and *H. uninervis,* light data was integrated as a rolling 14 day mean of the total daily irradiance to provide a relevant timeframe for measuring impacts to the plant. Under low light stress conditions, physiological adjustments first occur over a matter of days whereas plant-scale changes take place after a number of weeks (McMahon et al. 2013). An assessment of integrated light over a two week period is therefore in line with both tidally-driven fluxes in light (e.g. re-suspension of bottom sediments due to shear stress) as well as a period of time preceding apparent

morphological change for these larger growing species. Trends in light levels were compared against changes in seagrass between consecutive quarterly sampling events.

At deepwater sites, light was integrated over a seven day period instead of 14 days. A shorter interval was used due to the fast response time of *Halophila* spp. to light and the comparative lack of below ground energy stores for these species to resist low light conditions compared with the larger shallow species. There are little to no carbohydrate reserves to rely on when light is limiting and physiological acclimation is limited prior to seeing apparent morphological changes or shoot loss in *Halophila*. While it is unclear how quickly changes may occur in some populations, a seven day average was used to remove the extreme variability from the dataset and provide an opportunity to discern if clear trends between light and seagrass abundance exist at the deepwater sites.

2.3 Results

2.3.1 Local climate conditions

Over the duration of this study, the local climate in Gladstone has been characterised by above average rainfall, punctuated by severe flood events. River flow peaked in January 2013 well above the long-term monthly average. The peak in January 2013 is the highest on record with discharges of over 600 000 ML of water causing severe flooding in the Gladstone region (Figures 20 and 21).

2.3.2 Irradiance at post-dredging permanent transect monitoring sites

The rolling mean total daily irradiance at Fisherman's Landing and Wiggins Island in the inner harbour frequently fell to low light levels over both growing and senescent seasons earlier in the program (late 2009 to early 2011) (McCormack et al. 2013). Since mid-2011 however, the light environment appears to have improved. Light conditions at Fisherman's Landing and Wiggins Island have remained favourable for seagrass growth over the majority of 2013 and 2014 (Figure 22).

Trends in seagrass cover and *in situ* light at Pelican Banks North are closely aligned (Figure 23). The rolling mean total daily irradiance at Pelican Banks North has remained generally favourable during the growing season since the end of 2010, when there was a major flood event and peaks in river flow (Davies et al. 2015). During the growing season in 2013, mean total daily irradiance at Pelican Banks North reached the highest levels recorded over the duration of the program (>30 mol photons m^{-2} day^{-1}; Figure 23). During 2014 irradiance declined during February and June/July and increased during the August and November surveys, corresponding to peaks in percent cover. Seagrass percent cover continued to decline in March and May 2015 despite light levels remaining adequate, and is likely a consequence of other environmental factors, such as high temperatures, cuing the start of the senescent season.

At Pelican Banks South, mean total daily irradiance has also remained favourable for seagrass growth for the duration of the program, with the exception of a period in February 2013 following the extreme rainfall and river flow event and again in January/February 2014. Total daily irradiance at the southern end of Pelican Banks generally align with light conditions at the northern end, however seasonal peaks in light levels were not as pronounced at Pelican Banks South during the 2013 growing season as in the northern section (Figure 23). Irradiance was generally lower in 2014 and 2015 at this site compared with 2013, and although light levels increased during the growing season percent cover did not recover (Figure 23), highlighting the need for a clearer understanding of other possible factors affecting seagrass senescence, such as temperature.

At nearby Facing Island, the light environment has been more variable. Over the first few years of the study when the site was more heavily dominated by *Z. muelleri* subspp. *capricorni*, the rolling average

irradiance at Facing Island fell to low levels (below 6 mol photons m^{-2} day^{-2} over a 14 day rolling average) almost as much as it remained at higher levels, with light conditions at Facing Island appearing to be far poorer than at the turbid inner harbour sites (McCormack et al. 2013). In 2013, light levels at the site improved and mean total daily irradiance remained favourable for seagrass growth for the majority of the year, with the exception of a few days in February 2013 following the extreme rainfall and river flow event and longer periods in January, February and June 2014 (Figures 20, 21 and 24). Since early 2014 the species composition has shifted to a *H. uninervis* dominated meadow, which may not require as much light as *Z. muelleri* subspp. *capricorni)* (see Section 2.3.6).

Out of Port reference site

At the out of Port reference site at Rodds Bay, the rolling average total daily irradiance has remained above 6 mol photons m^{-2} day^{-1} over the majority of the study, with the exception of a short duration of time when the logger was shifted to a deeper location (Figure 24). In 2013, however the rolling average irradiance fell at the same time as declines in the inner and outer harbour following the extreme rainfall and river flow event (Figures 20 and 21). In 2014 and 2015, irradiance was generally high and conditions for seagrass growth were favourable, with a substantial peak in seagrass percent cover detected in November 2014 (Figure 24).

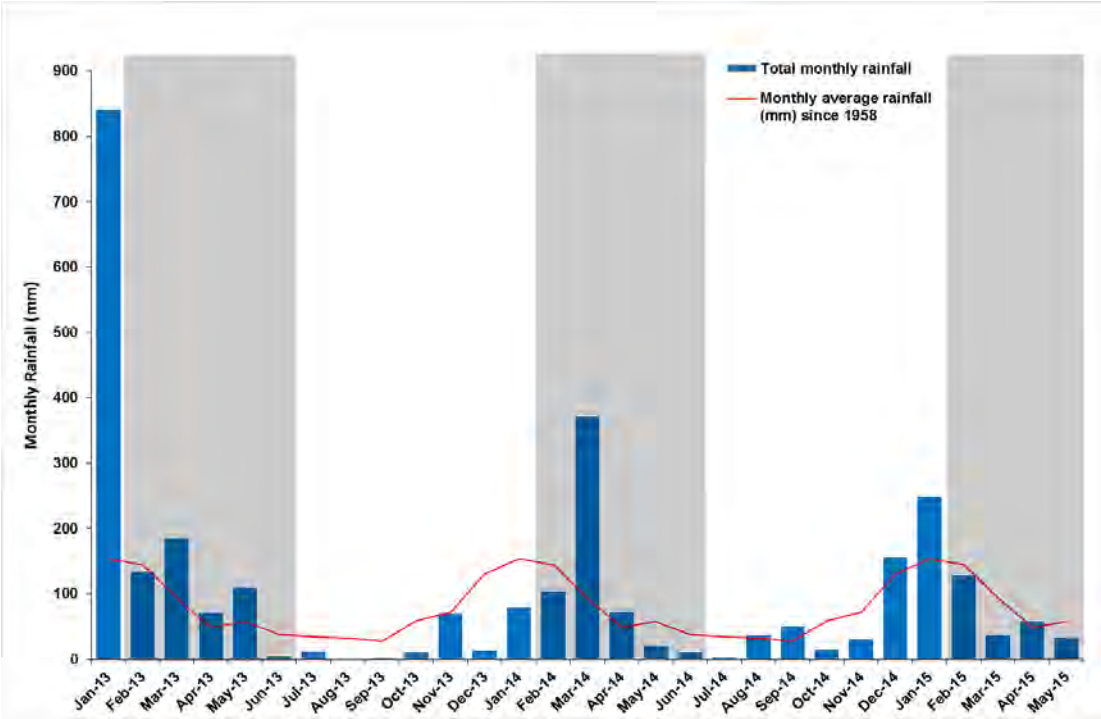


Figure 20: Total monthly rainfall (mm) for Gladstone, from January 2013 to May 2015. Data taken from station number 039123 (Gladstone Airport); from Bureau of Meteorology (http://www.bom.gov.au/climate/data/).

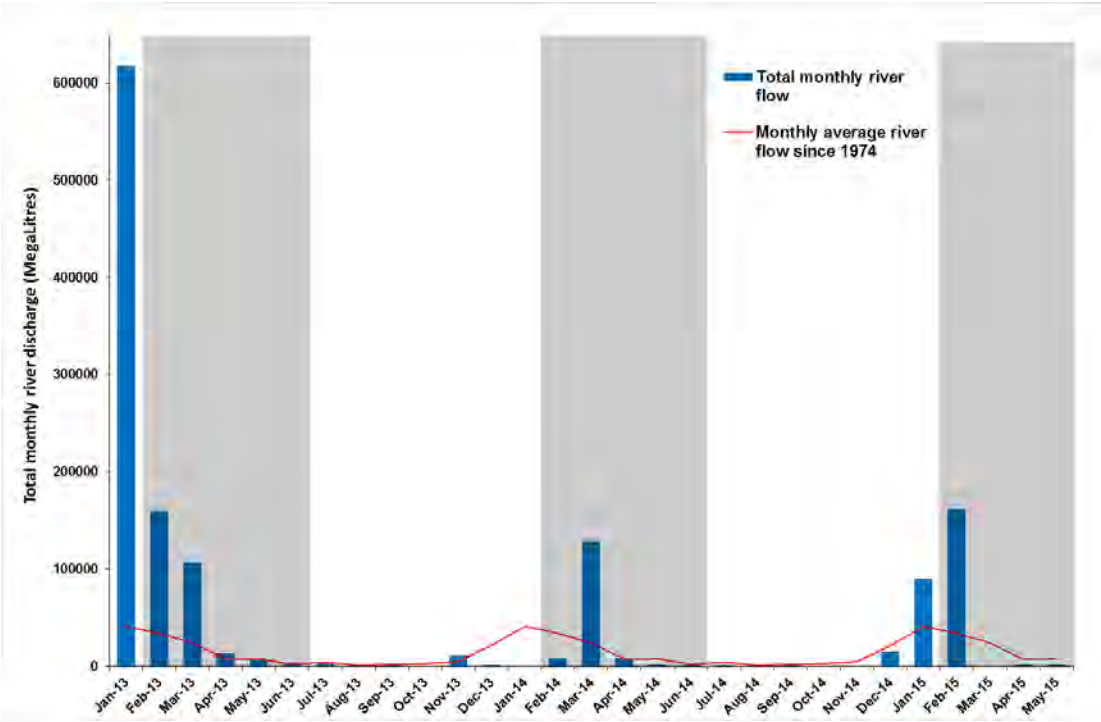


Figure 21: Total monthly river discharge (volume ML) for Calliope River at Castlehope, Gladstone, from January 2013 to May 2015. Data taken from Calliope Basin, site 132001A; from DERM Water Monitoring (http://watermonitoring.derm.qld.gov.au/host.htm).

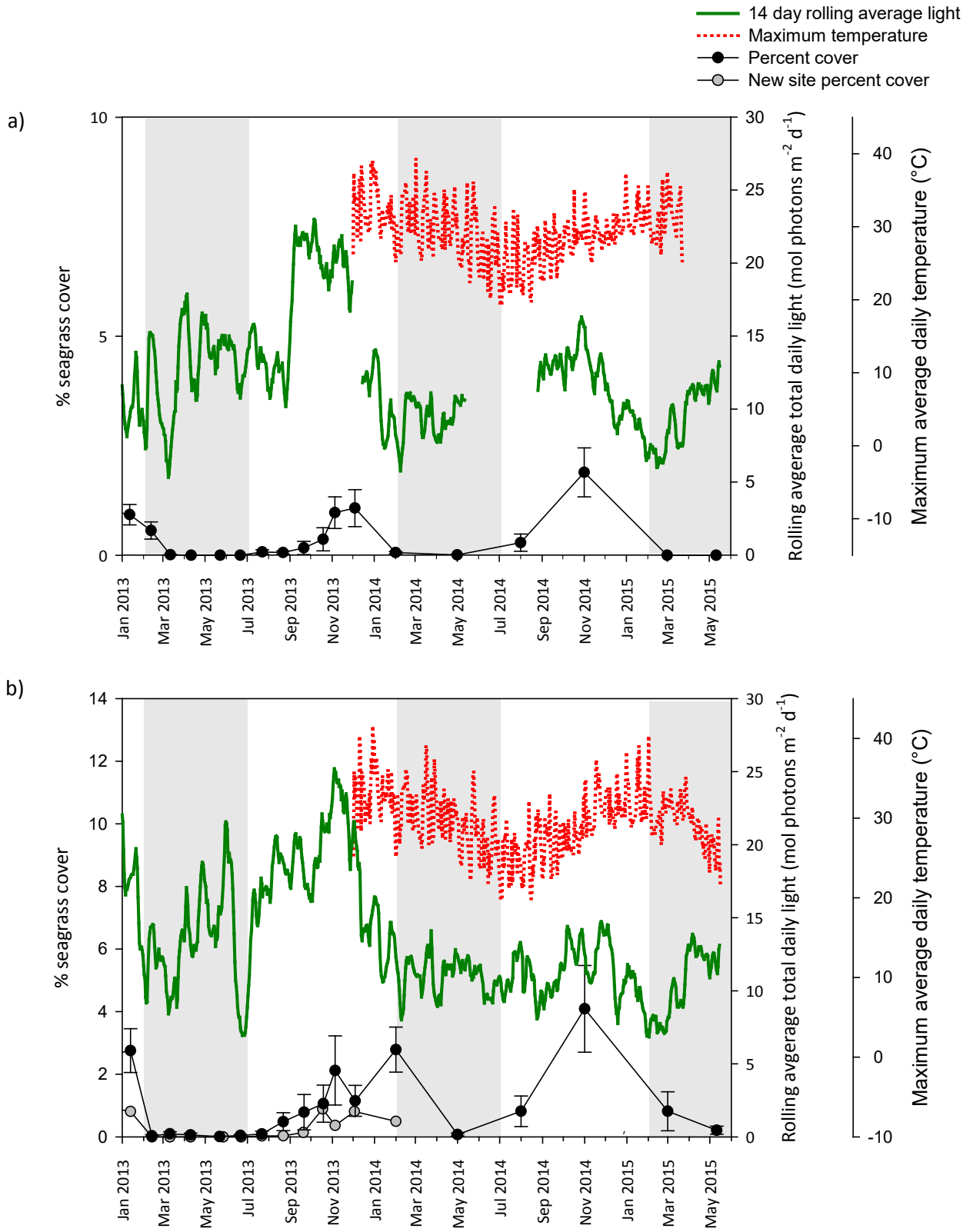


Figure 22: Rolling 14 day average total daily irradiance (mol photons m^{-2} d^{-2}) and seagrass percent cover at a) Fisherman's Landing and b) Wiggins Island over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Port Curtis.

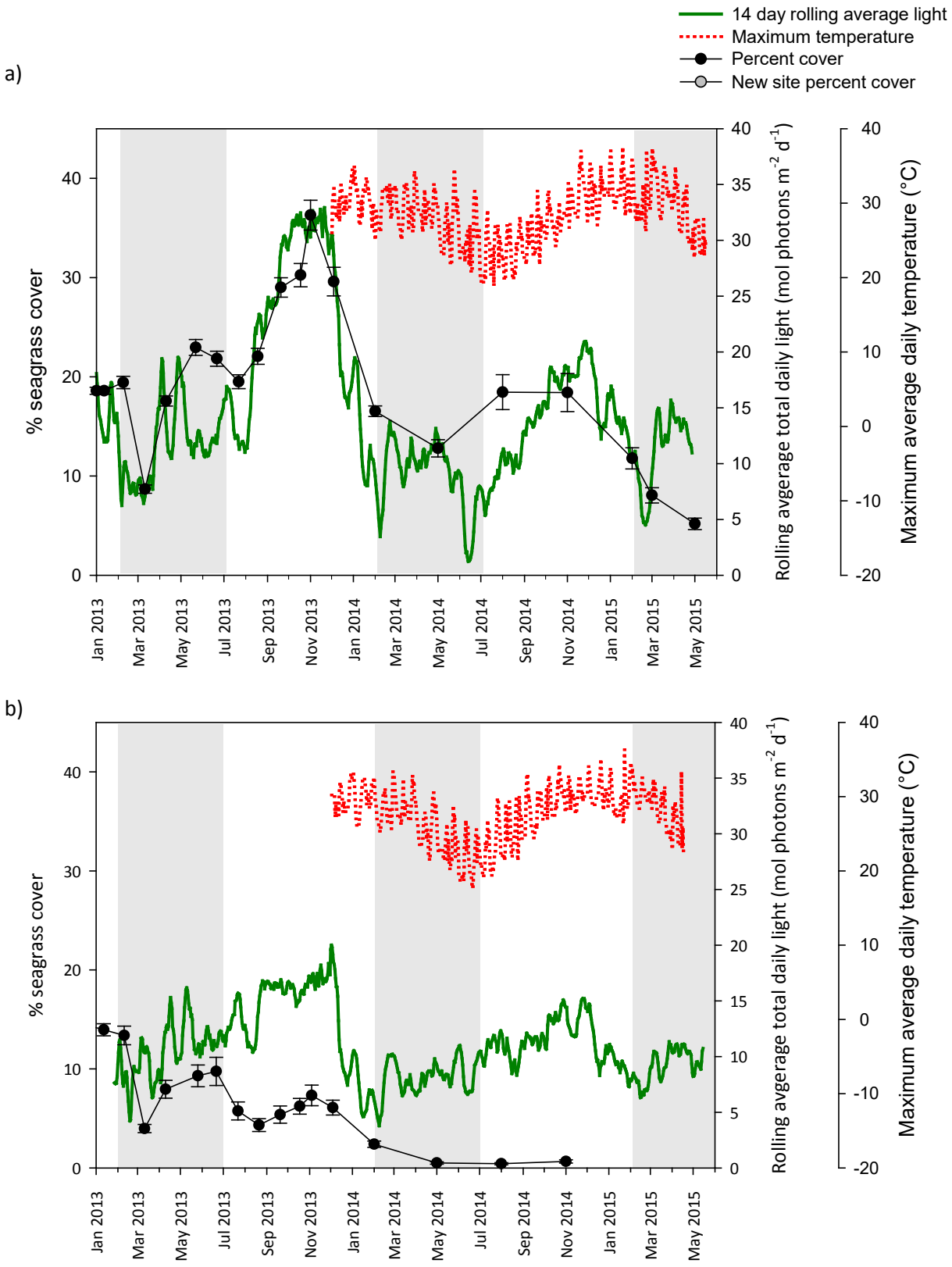


Figure 23: Rolling 14 day average total daily irradiance (mol photons m^{-2} day^{-1}) and seagrass percent cover at a) Pelican Banks North and b) Pelican Banks South over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Port Curtis.

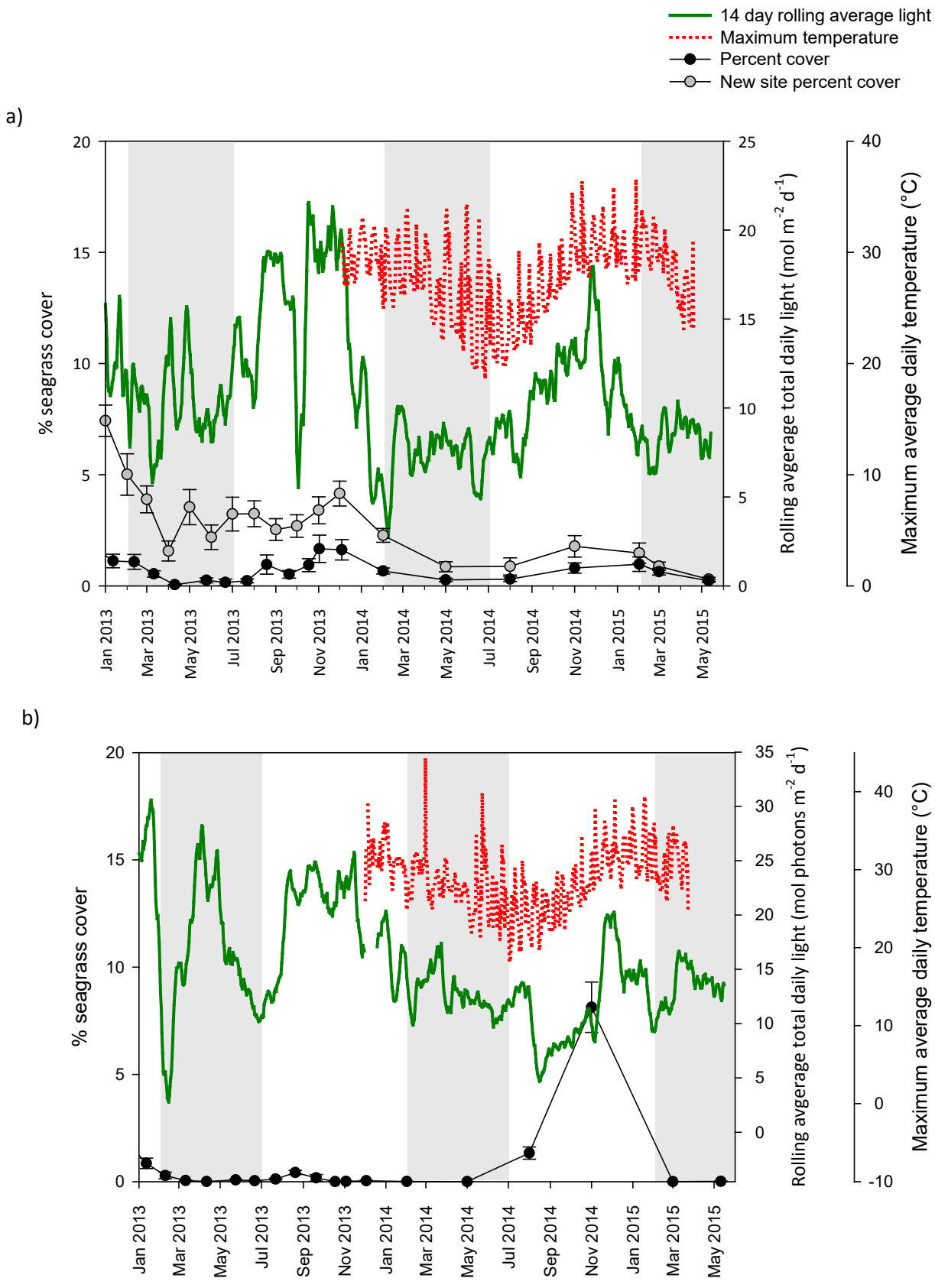


Figure 24: Rolling 14 day average total daily irradiance (mol photons m^{-2} day^{-1}) and seagrass percent cover at a) Facing Island and b) Rodds Bay over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Port Curtis.

2.3.3 Irradiance at Channel Duplication permanent transect monitoring sites

Examination of the *in situ* light recorded at the subtidal site at Colosseum Inlet revealed a much lower light environment than other coastal sites which expose at low tide (Figure 25). Increases in seagrass percent cover corresponded with increased light during the 2014 growing season, and declined to low levels during the 2015 senescent season (Figure 25).

Mean total daily irradiance at Quoin Island follows a similar seasonal trend yet is much higher than at Colosseum Inlet, with light levels ranging between 5 to 15 mol photons m^{-2} day^{-1} over a two week period. The seagrass cover peak in November 2014 corresponded with peaks in light; however the light levels did not decline to the same degree as many of the other sites during the senescent season (Figure 25). Temperature was similar between the sites (Figure 25).

Mean total daily irradiance at the South Trees Inlet meadow generally stayed well above 10 mol photons m^{-2} day^{-1} for the growing season in 2014 and into the senescent season in 2015 (Figure 26). This was one of the few sites to have the peak in cover occur in March, rather than at the height of the growing season. Light levels for both periods of time were above 15 mol photons m^{-2} day^{-1}, suggesting light is not limiting at this site and the delay in peak abundance relates to a different variable.

2.3.4 Irradiance at deepwater monitoring sites

Mean total daily irradiance at deepwater site 1 has been extremely low (< 4 mol photons m^{-2} day^{-1} over a 7 day period) since monitoring began in June 2014, as expected from a site at this depth. The light environment at site 1 was generally higher during the growing season, and fell below a rolling average of 1 mol photons m^{-2} day^{-1} over a 7 day period in January 2015 (Figure 27). Above-ground biomass was also extremely low (< 0.2 g DW m^{-2}) but showed a distinct seasonal trend, with biomass peaking during the growing season in August and November 2014 (Figure 27).

Mean total daily irradiance at deepwater site 2 was similar to site 1, generally remaining below 5 mol photons m^{-2} day^{-1} over a 7 day period since monitoring began in June 2014. The light environment at site 2 was generally higher during the growing season, and fell below a rolling average of 2 mol photons m^{-2} day^{-1} over a 7 day period in January 2015 as the senescent season began (Figure 28). Above-ground biomass at site 2 was the greatest of the three deepwater sites, peaking at 0.8 g DW m^{-2} in November 2014 (Figure 28).

Deepwater site 3 had the highest light environment during both the growing season and into the senescent season, reaching a peak of nearly 10 mol photons m^{-2} day^{-1} over a 7 day rolling average period at the end of September 2014 (Figure 28). This site also showed a strong seasonal trend in light, however the light level at the start of the senescent season (< 2 mol photons m^{-2} day^{-1} over a 7 day period) was more similar to the range of light experienced during the growing season at sites 1 and 2 (Figures 27 and 28). Despite this, site 3 had the lowest biomass, with seagrass present only in August and November 2014 at extremely low levels (0.02 g DW m^{-2}) (Figure 28).

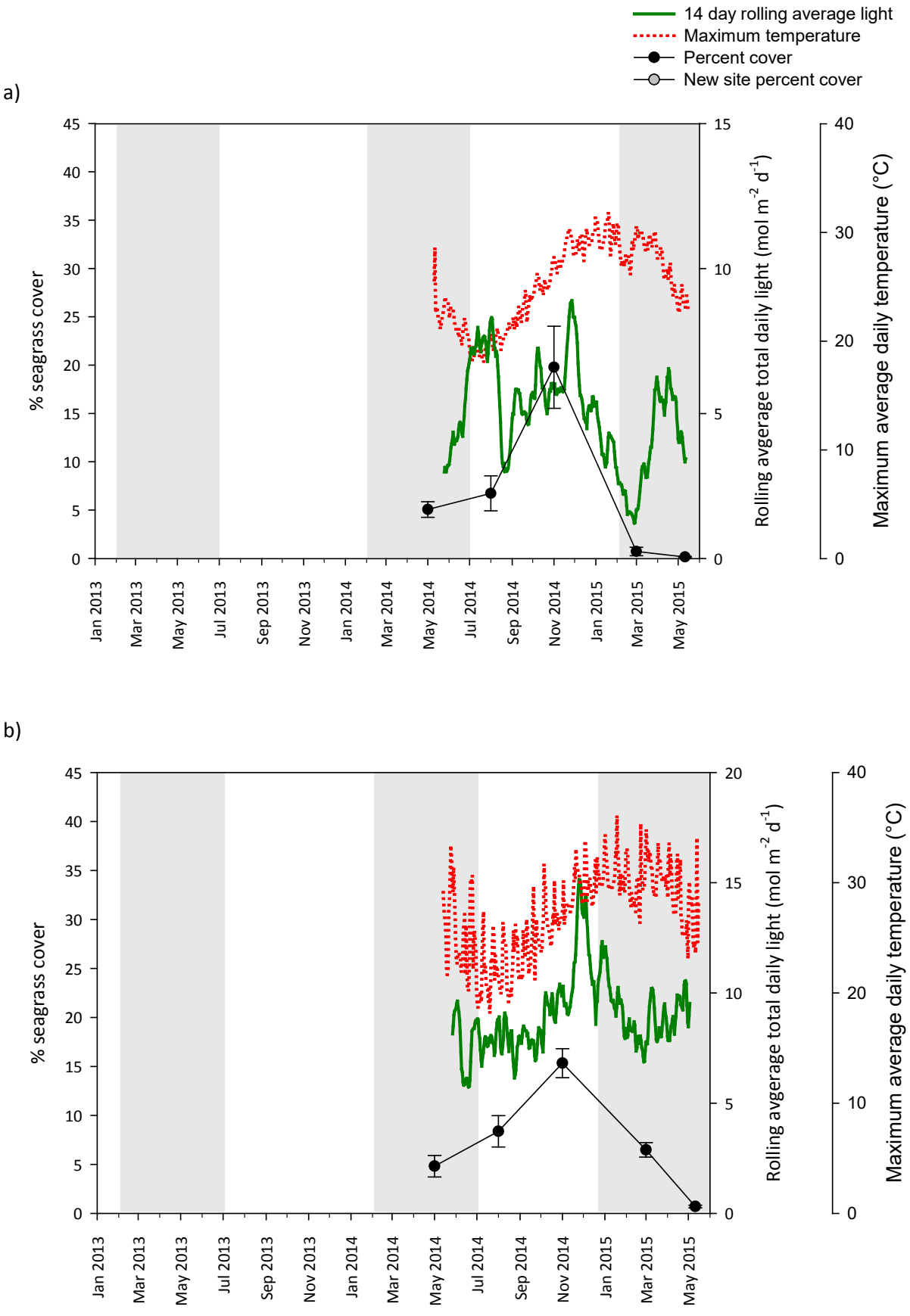


Figure 25: Rolling 14 day average total daily irradiance (mol photons m^{-2} day^{-1} and seagrass percent cover at a) Colosseum Inlet and b) Quoin Island over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Port Curtis.

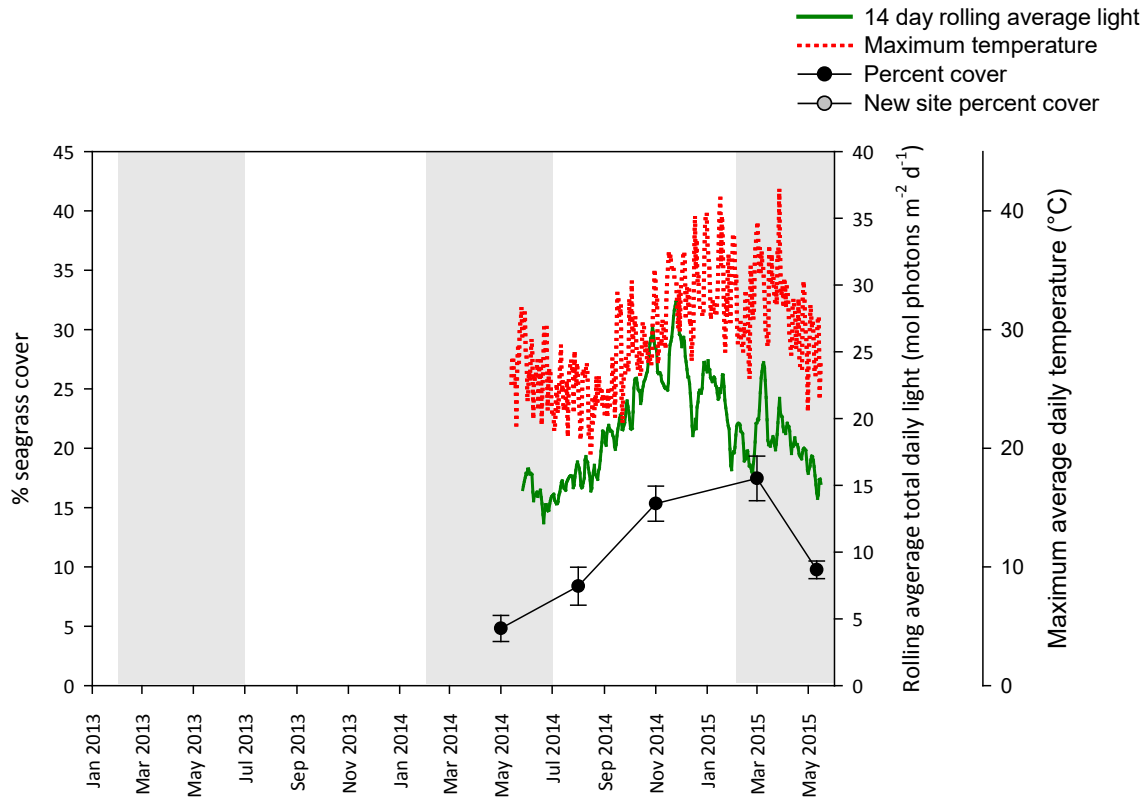


Figure 26: Rolling 14 day average total daily irradiance (mol photons m^{-2} day^{-1}) and seagrass percent cover at South Trees Inlet over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Gladstone.

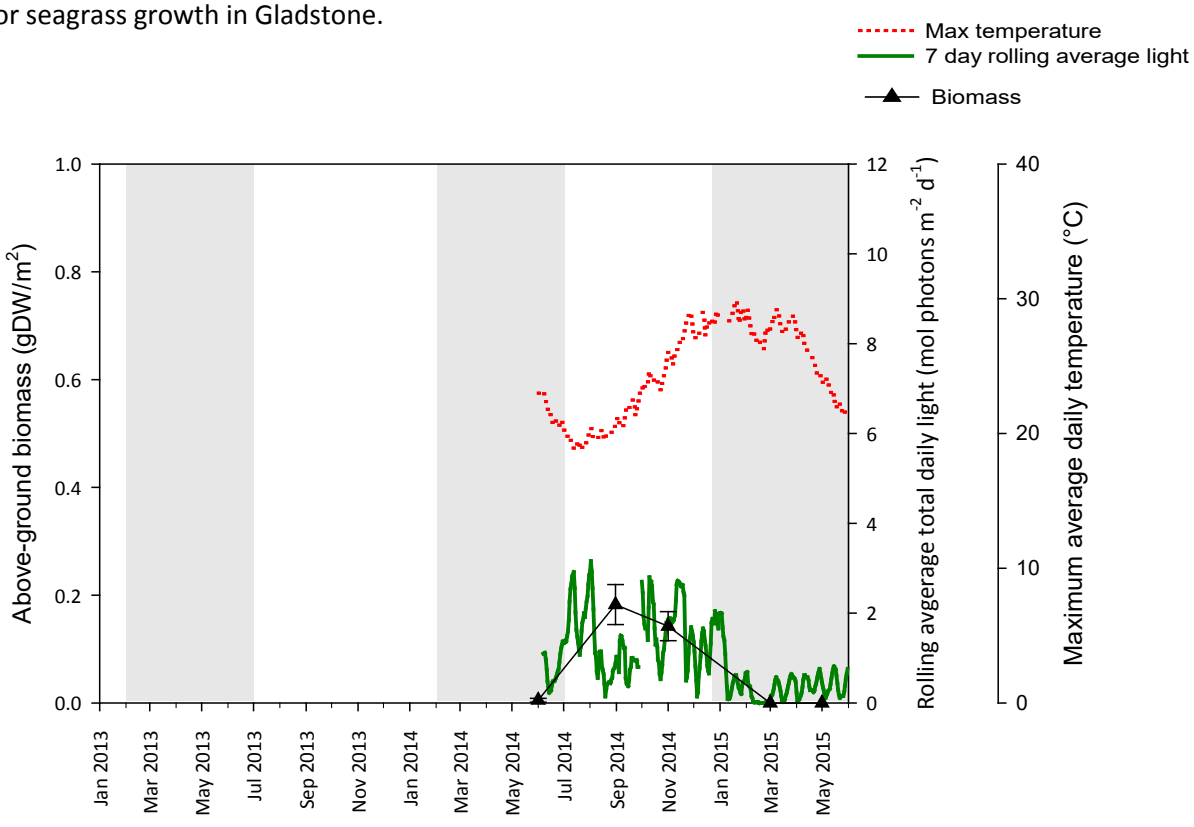


Figure 27: Rolling 7 day average total daily irradiance (mol photons m^{-2} day^{-1}) and seagrass biomass at deepwater site 1 over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Gladstone. Light and temperature data are provided by Vision Environment QLD.

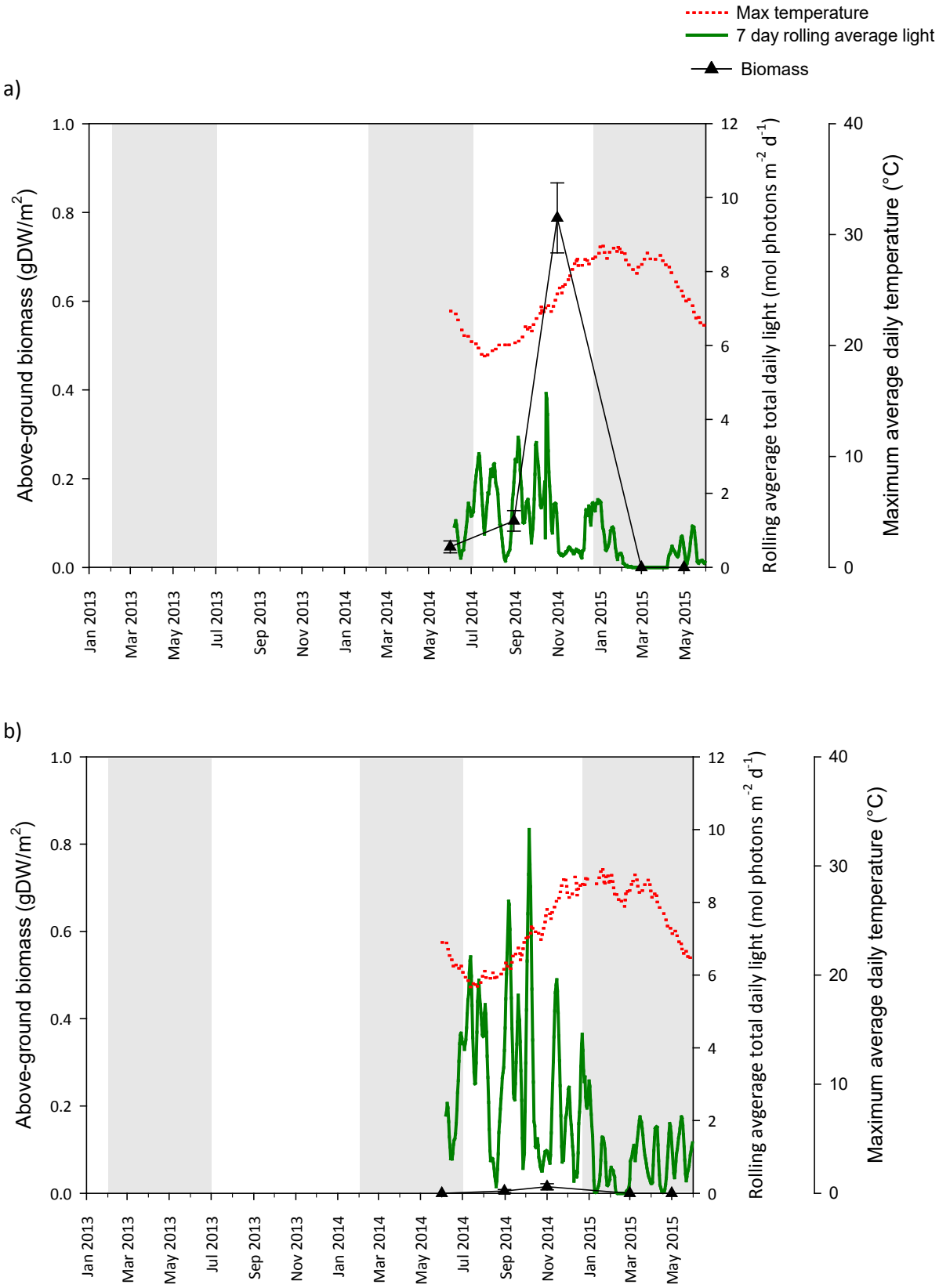


Figure 28: Rolling 7 day average total daily irradiance (mol photons m^{-2} day^{-1}) and seagrass biomass at a) deepwater site 2 and b) site 3 over the duration of the monitoring program. Shaded areas represent the senescent season for seagrass growth in Port Curtis. Light and temperature data is provided by Vision Environment QLD.

2.3.5 Literature values and other studies

A review of seagrass and light literature was used as a guide for the following light ranges developed for the species found in Port Curtis and Rodds Bay. Previous work conducted in Port Curtis by Chartrand et al. (2012) found seagrasses were not affected by reductions in light during the senescent season, and therefore examination of the above-ground biomass data in relation to light is restricted to the growing season only.

Deepwater Halophila species

Chartrand et al. (2014) collected *H. decipiens* from Green Island (offshore from Cairns) and *H. spinulosa* from Abbot Point and subjected the samples to different light and temperature regimes in mesocosm experiments. The key findings of the studies so far show that deepwater *H. decipiens* and *H. spinulosa* shoot density was negatively affected by light treatments of 1 mol photons m^{-2} day^{-1} compared to typical ambient growing season conditions and control light treatments of 3.2 mol photons m^{-2} day^{-1}. *H. decipiens* shoot density significantly declined after 2 weeks under low light treatments whereas *H. spinulosa* was not affected until 4 weeks. A critical light threshold in the range of 1-2 mol photons m^{-2} day^{-1} was identified from the study and the long-term field monitoring program, and is currently being examined further as part of ongoing field monitoring and research for its validity.

Based on this study and other relevant literature (Appendix 1), a range of between 1 and 2 mol photons m^{-2} day^{-1} was used to examine light and seagrass data for deepwater *Halophila* species.

Halodule uninervis

Halodule uninervis generally requires a higher light environment than *Halophila* species (Bach et al. 1998; Longstaff and Dennison 1999). Recent laboratory experiments by Collier et al. (2015 subm) tested the response of *H. uninervis* and *H. ovalis* shoot density and growth rates to six light levels ranging from 0 to 23 mol photons m^{-2} day^{-1} in cool (~23 °C) and warm (~28 °C) temperatures over 14 weeks. The impact of light deprivation on shoot densities and growth rates was faster in *H. ovalis* than in *H. uninervis*. This study found that low light levels (≤3.3 mol photons m^{-2} day^{-1} treatments) had a significant negative effect on shoot density in cool temperatures between 10 – 14 weeks and in warm temperatures between 4 – 7 weeks.

Previous work at Magnetic Island off Townsville (Collier et al. 2012a) determined light thresholds required for the long-term survival of tropical seagrass meadows dominated by *H. uninervis* identified during 'real-time' loss of seagrass. The thresholds in this study were determined during natural *in situ* reductions in light and concurrent seagrass loss. Natural variability of the light environment was captured and incorporated changes in light colour, diurnal fluctuations in intensity, tides and seasonality (Collier et al. 2012a). The study found that a significant loss (>50%) in seagrass occurred at <4 mol photons m^{-2} d^{-1}, while minimum light levels associated with seagrass gain occurred when light was >5 mol photons m^{-2} d^{-1}. In a separate study, Collier et al. (2012b) also found *H. uninervis* shoot density declined in the lab after 8.7 weeks when held under 4.4 mol photons m^{-2} day^{-1}.

Based on these studies and other relevant literature (Appendix 1) a likely range of between 3 and 5 mol photons m^{-2} day^{-1} was used to examine light and seagrass change data set for *H. uninervis*.

Zostera muelleri subsp*. capricorni*

Chartrand et al. (2012) determined that *Z. muelleri* subsp. *capricorni* in the Port Curtis requires between 4.5 and 12 mol photons m^{-2} day^{-1} during the growing season over a minimum period of two weeks to survive. Trends in irradiance and seagrass condition at permanent transect sites in Port Curtis and Rodds Bay have shown, in general, *Z. muelleri* subsp. *capricorni* consistently received greater than 6 mol photons m^{-2} day^{-1} over a two week rolling average at most locations in the harbour during the growing season (when seagrass remained stable or increased in abundance) (Davies et al. 2015). Most laboratory shading studies have found reductions in biomass, leaf density or shoot density when the light intensity has dropped below 2 mol photons m^{-2} day^{-1}, while studies where light intensity was above 4.6 to 4.8 mol photons m^{-2} day^{-1} found no significant impact to seagrass (Appendix 1).

Work by Collier et al. (2012b) has shown that below 4.4 mol photons m^{-2} day^{-1}, *Z. muelleri* subsp. *capricorni* shows significant changes in leaf morphology, shoot density and biomass and was the fastest to negatively respond to declines in light compared with three other tropical species (including *H. uninervis*). This fast response is attributed to smaller carbohydrate reserves in the below-ground tissues compared to other species. Small reserves may get used rapidly in low light conditions, requiring the morphological changes and reduced growth rates to occur more quickly for *Z. muelleri* subsp. *capricorni* than other species which have greater carbohydrate reserves, delaying the need to make such changes (Collier et al. 2012b).

Based on these studies and other relevant literature (Appendix 1) a range of between 4.8 and 12 mol photons m^{-2} day^{-1} was used to examine light and seagrass data for *Z. muelleri* subsp. *capricorni*. Chartrand e*t al.* (2012) adopted a management trigger based on 6 mol photons m^{-2} day^{-1} over a two week rolling average for seagrasses in the Western Basin area, and we further examine this trigger for future application in the potential Channel Duplication area of interest in 2.3.6 below.

2.3.6 Port Curtis light history and seagrass biomass change

Coastal sites – Zostera muelleri subsp. capricorni dominated meadows

Examination of the light history and biomass trends at these sites with the values derived from previous studies conducted in Port Curtis as well as other literature derived values supports a light threshold of a rolling average of 6 mol photons m^{-2} day^{-1} over 14 days (Figure 29). During the growing seasons in 2013 and 2014, seagrass was maintained or increased when light was at least 6 mol photons m^{-2} day^{-1} over a two week rolling average, consistent with the findings from Chartrand et al. (2012).

The light data collected at the *Z. muelleri* subsp. *capricorni* dominated Pelican Banks North and South Trees Inlet sites indicates these meadows are not growing in a light-limited environment. Mean total daily irradiance during growing seasons in 2013 and 2014 was maintained well above the suggested light threshold of 6 mol photons m^{-2} day^{-1} over a 14 day period, with the exception of a few days in early July 2014 (Chartrand et al. 2012). As light increased following the senescent season in 2014 over 6 mol photons m^{-2} day^{-1}, biomass increased as well (Figure 29). South Trees Inlet in particular experiences a high light environment, with recorded light levels never falling below 12 mol photons m^{-2} day^{-1} (Figure 29).

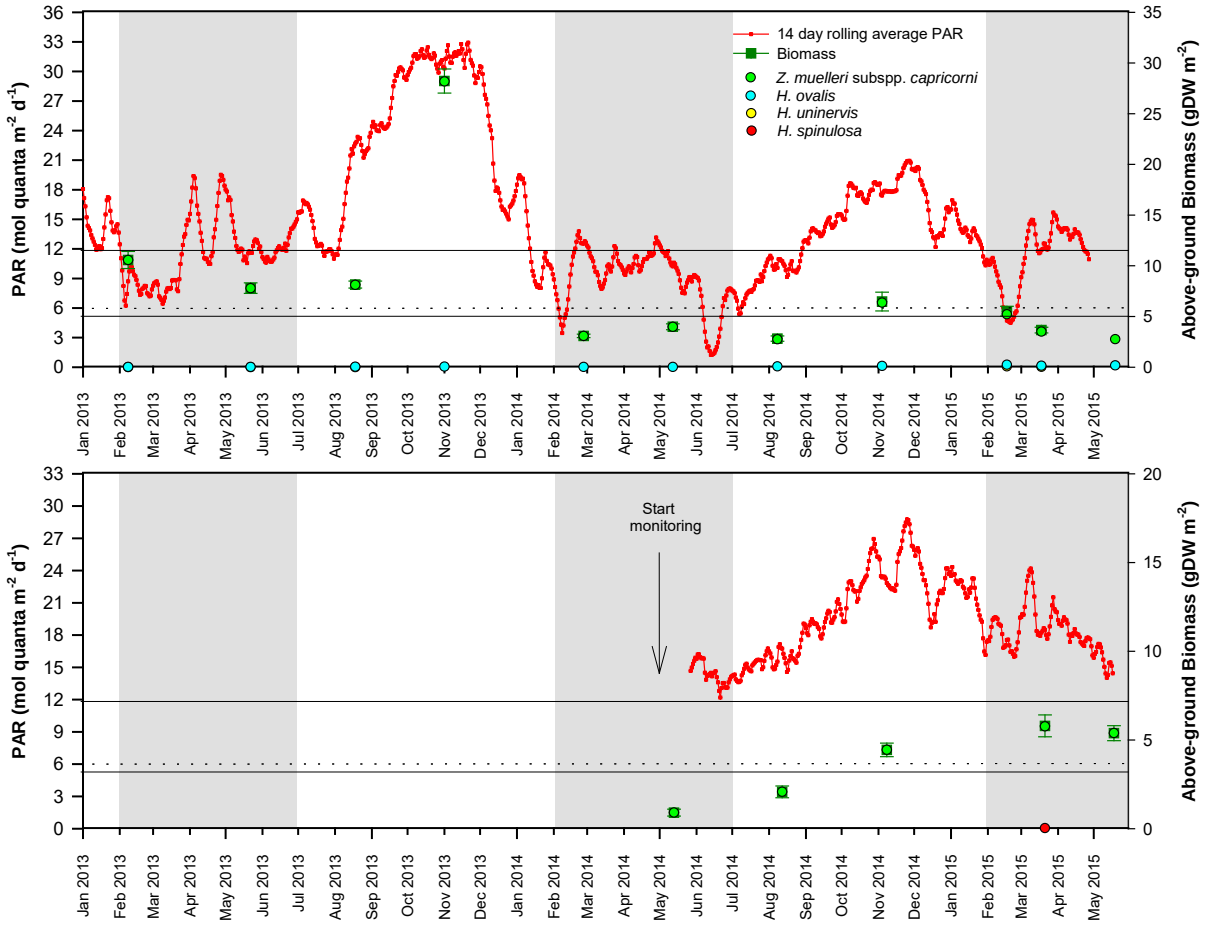


Figure 29: Rolling 14 day average of light (mol photons m^{-2} day^{-1}) for coastal *Z. muelleri* subspp. *capricorni* dominated meadows at (top) Pelican Banks North and (bottom) South Trees Inlet from January 2013 to May 2015 and mean (±SE) above-ground biomass at quarterly surveys. Coloured circles represent a species breakdown of the total biomass. Shaded areas represent the senescent season in Gladstone.

Coastal sites - Halodule uninervis dominated meadows

Examination of the light history and biomass trends at these sites with the literature derived range supported using a light threshold of approximately of 5 mol photons m^{-2} day^{-1} over a rolling average of 14 days (Figure 30). Seagrass abundance, though very low, was maintained at the Facing Island site for the entirety of the study and, for the majority of time, light levels exceeded the maximum threshold range derived from literature and other studies (Figure 30). Mean total daily irradiance during the majority of the growing seasons in 2013 and 2014 was maintained well above the suggested light threshold of 5 mol photons m^{-2} day^{-1} and therefore light was unlikely to be a limiting factor over the period of monitoring. The lack of sustained low light periods at these sites (below the likely threshold for the species) meant that the paired light history and seagrass change data was not able to provide any refinement to the range obtained from existing literature values, so we have suggested at this stage to adopt the upper end of that range as an initial threshold value.

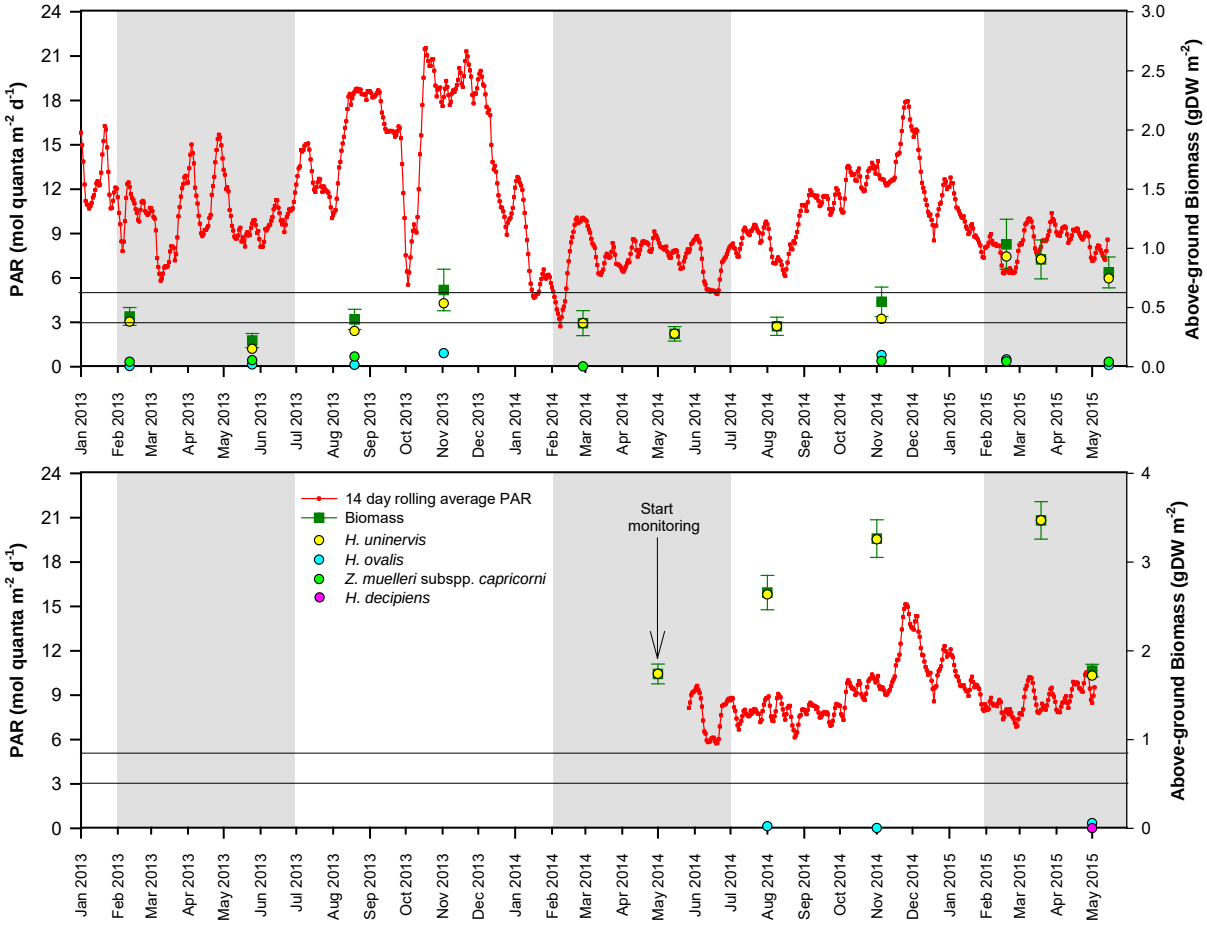


Figure 30: Rolling 14 day average of light (mol photons m^{-2} day^{-1}) for coastal *H. uninervis* dominated meadows at (top) Facing Island (original site) and (bottom) Quoin Island from January 2013 to May 2015 and mean (±SE) above-ground biomass at quarterly surveys. Coloured circles represent a species breakdown of the total biomass. Shaded areas represent the senescent season in Port Curtis.

Deepwater sites - Halophila spp. *dominated meadows*

Examination of the light history and biomass trends at deepwater sites with literature derived range found the light threshold range of 1 to 2 mol photons m^{-2} day^{-1} describes the environment where seagrass abundance was generally maintained or had increased (Figure 31).

Due to the three months sampling frequency for seagrass abundance, developing firm correlations with continuously collected light history is difficult. The light environment for the month leading up to a survey often fell below the 1 mol photons m^{-2} day^{-1} threshold, for example leading up to the September 2014 survey at both sites 1 and 2 (Figure 31). However it is possible that seagrass biomass peaked at a higher level (during the 3 months between sampling events) and then declined due to a low light environment. Despite this there is a clear trend at both sites 1 and 2 following June 2015 when the 7 day rolling average of light fell below 1 mol photons m^{-2} day^{-1} (Figure 31). The sites were unable to be sampled in February 2015 due to a cyclone, and by the March 2015 when the 7 day rolling average of light had remained below 1 mol photons m^{-2} day^{-1} for over a month, seagrass was completely absent from both sites (Figure 31). These data suggest that the conservative light threshold of 1-2 mol photons m^{-2} day^{-1} developed from other Queensland studies was likely to apply in Port Curtis.

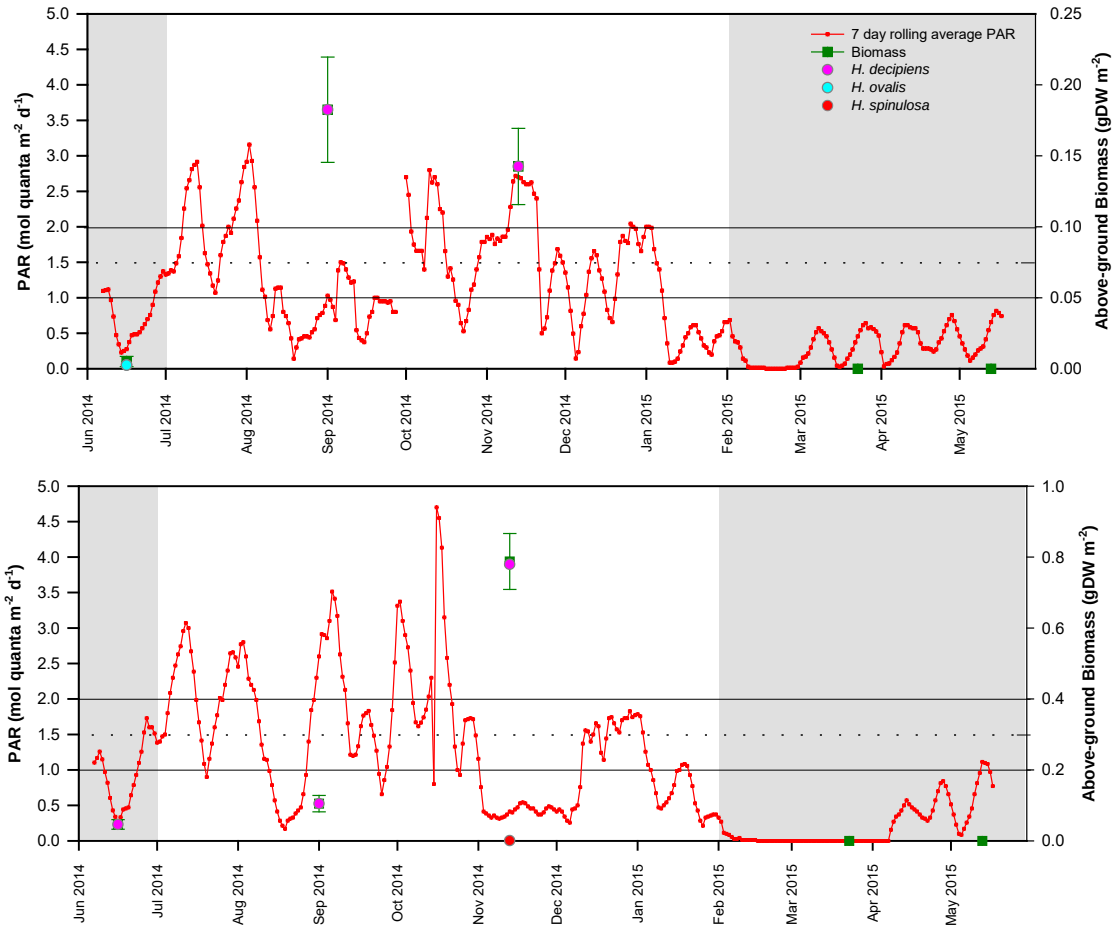


Figure 31: Rolling 7 day average of light (mol photons m^{-2} day^{-1}) for deepwater *H. ovalis* dominated meadows at (top) site 1 and (bottom) site 2 from January 2013 to May 2015 and mean (±SE) above-ground biomass at quarterly surveys. Coloured circles represent a species breakdown of the total biomass. Shaded areas represent the senescent season in Port Curtis.

2.4 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). Recent work in Port Curtis by Chartrand et al. (In review) indicates temperature may also be a key factor affecting tropical seagrass seasonality and survival. 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. McCormack et al. (2013) showed that river flow has a significant negative effect on seagrass abundance in Port Curtis, explaining up to 73 percent of the variation in abundance depending on the location of the monitoring site. The strongest relationships found were at Wiggins Island and Pelican Banks South which are also the closest assessed sites to the mouth of the Calliope River (McCormack et al. 2013).

In January 2013, the Calliope River discharged at record levels and significant declines in seagrass percent cover were detected at sites throughout the Port of Gladstone. Monthly monitoring of seagrasses at transect sites over the course of 2013 has revealed different rates of recovery following the extreme river flow event. The most noticeable difference was between Pelican Banks North and Pelican Banks South where seagrasses at both sites declined markedly in March 2013. 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. Over 2014 and into 2015, seagrass percent cover at Pelican Banks North was lower than in previous years but remains within the range observed at this site.

At Pelican Banks South however, the initial recovery detected at the site following the extreme weather events in 2013 was short lived and seagrass abundance was atypically low over the growing season. Though light levels were generally favourable based on *in situ* measurements, seagrass percent cover had severely declined during 2014 and no recovery over the growing season was detected. There was a further flow event of the Calliope River in February 2014 and a corresponding drop in light levels was recorded at Pelican Banks South. These declines in light during January and February resulted in substantial periods of time below the light requirement for seagrasses (Chartrand et al. 2012) which likely contributed to this decline initially. However for the remainder of the year, including the growing season, light conditions were favourable and seagrasses were not able to recover at this site. Elemental ratios of seagrass leaf tissue at Pelican Banks South (see Section 3) indicate that seagrasses continued to exist in lower light and nutrient enriched conditions for the majority of 2013 and 2014. Analysis of *H. uninervis* elemental ratios detected phosphorous limitation at the site for the majority of the year in 2014. These sustained conditions for the past two years following the January 2013 flood event may have contributed to declines in seagrass at the site.

While seagrass abundance closely matches trends in irradiance at the Pelican Banks North site, the link is less clear for most of the other locations. At other sites, including Pelican Banks South, seagrass abundance has often remained at low levels despite light improving significantly over the latter part of the program. This demonstrates that while light is an important factor in determining seagrass growth there are a range of other variables that influence seagrass growth. 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.

Conclusions and recommendations for light ranges of Port Curtis seagrasses

Following a literature review of light requirements for the species found in Port Curtis, relevant studies and analysis of the *in situ* light environment and above-ground biomass change of seagrass, a range of initial light thresholds for *H. uninervis* and deepwater *Halophila* spp. were recommended for Port Curtis

seagrasses as well as a re-examination of the established Port Curtis threshold for *Z. muelleri* subsp. *capricorni* for future application in the Channel Duplication project.

Light history recorded at sensitive receptor sites in Port Curtis in conjunction with measurements of seagrass biomass recorded quarterly provided site-specific data with which to compare the light threshold range from most appropriate studies and published literature for these species. Seagrass abundance at the sites monitored in Port Curtis was generally maintained or increased when at or above the recommended light thresholds during the growing season. However the generally high light levels maintained at these sites over the course of the study meant that for some species the literature derived ranges were unable to be further refined for a local value, particularly for *Halodule uninervis*. Despite this for most sites, trends in seagrass above-ground biomass and the light environment experienced at the seagrass canopy were closely related with increased light generally preceding increases in seagrass abundance.

For deep water seagrass sites dominated by *Halophila* spp., seagrass was completely lost when light fell below 1 mol photons m^{-2} day^{-1} for periods of time greater than 4 weeks. It is likely that seagrass above-ground biomass decline would have been detected earlier than 4 weeks had there been a seagrass sampling event prior to the quaterly sampling in March. *Halophila* spp. are physically very small and are not as structurally replete below-ground compared with other large and longer-lived species, which are able to rely on these below-ground stores to temporarily alleviate stresses incurred by reductions in light and water quality (Ralph et al. 2007). Due to this we have suggested a rolling average period of 7 days for the trigger rather than the longer 14 days used for the larger growing species.

While light is generally accepted as the primary environmental factor limiting the growth of tropical seagrasses, several studies on *Zostera* spp. have indicated 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. In review; York et al. 2013; Collier et al. 2011; Olesen and Sand-Jensen 1994). Collier et al. (2011) found that high temperatures (>33 °C) combined with either light limiting or saturating conditions will lead to declines in plant growth.

3 SEAGRASS TISSUE NUTRIENTS

3.1 Background and approach

Seagrass tissue nutrient ratios (calculated atomic ratios of C:N:P) can provide an indication of environmental forcing parameters in seagrass meadows (Johnson et al. 2006). The magnitude of these ratios and their temporal changes allow for a broad level understanding 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; Cabaço and Santos 2007; Grice et al. 1996; Abal et al. 1994b). 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. 1994b).

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. 2012). 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).

3.2 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 years there is no tissue nutrient data.

3.3 Results

3.3.1 Inner Harbour

Extremely low seagrass biomass and access issues prevented the collection of tissue nutrient samples in the inner harbour in 2013 and 2014. For trends prior to 2013 see McCormack et al. (2013)

3.3.2 Outer Harbour

Carbon and nitrogen ratios (C:N) for *Zostera muelleri* subsp. *capricorni* fluctuate seasonally at Pelican Banks, with higher levels in November during the growing season and lower levels during the wet season indicating a reduction in light available to seagrasses (Figure 32a). The C:N ratio peaked in November 2014 at 20.03 however remained below 20 for all other sampling events in 2014, similar to 2013 (Figure 32a). Facing Island C:N ratios remained below 20 for both *Z. muelleri* subsp. *capricorni* and *H. uninervis,* indicating a lower light environment compared with Pelican Banks (Figure 32a and b).

In 2014, N:P ratios for *Z. muelleri* subsp. *capricorni* remained within the 25-30 range in February, August and November, indicating a nutrient replete environment. Pelican Banks North was an exception, falling below 25 in November indicating nitrogen limitation (Figure 32c). All sites in May 2014 had N:P ratios above 30 for *Z. muelleri* subsp. *capricorni* indicating phosphorous limited environments at this time of year. Both Pelican Banks South and Facing Island New had N:P ratios for *H. uninervis* above 30 for all sampling events except November 2014 (Figure 32d). These results indicated sites were phosphorus limited for the majority of 2014.

Z. muelleri subsp. *capricorni* C:P ratios were near or under 500 for the majority of sites throughout 2014 with the exception of Pelican Banks South which did not drop below this threshold (Figure 32e). For *H. uninervis* all sites remained above 500 for February and May, and fell below the threshold of 500 for August and November, indicating a nutrient rich environment during the growing season (Figure 32f).

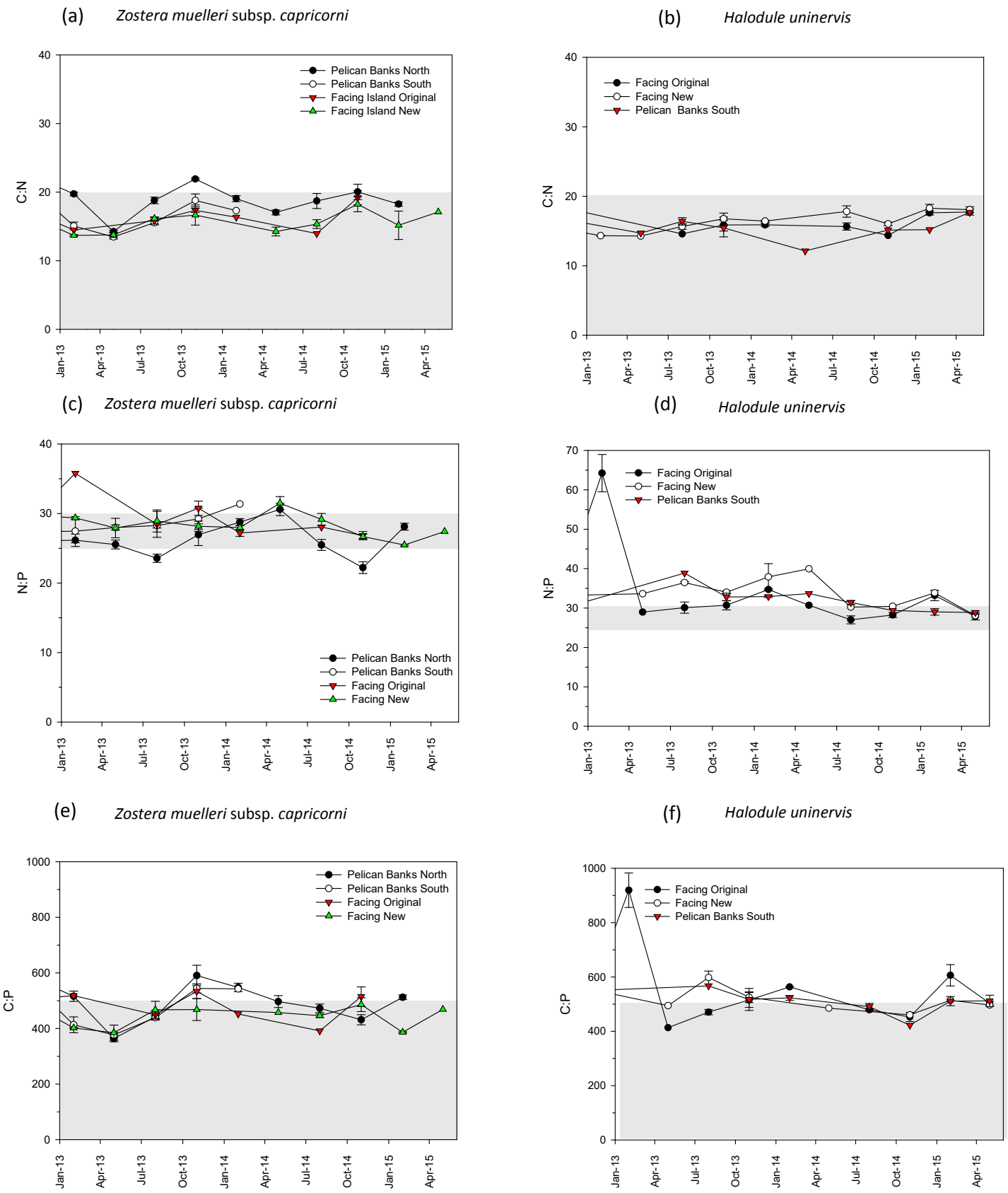


Figure 32: Elemental (atomic) ratios of seagrass leaf tissue for *Zostera muelleri* subsp. *capricorni* and *Halodule uninervis* collected at permanent transect locations in the outer harbour (mean ± SE). Shaded areas indicate low light (C:N < 20), nutrient replete (N:P 25-30) and nutrient enriched (C:P < 500) environments.

3.3.3 Out of Port reference site

The loss of above-ground biomass at Rodds Bay for most of 2014 limited sample collection to November only. Carbon and nitrogen (C:N) ratios for *Z. muelleri* subsp. *capricorni* remained below 20 throughout year, indicating persistent low light conditions (Figure 33a). N:P ratios for November remained between 25 and 30, (Figure 33b). C:P ratios have remained below 500, indicating seagrasses at this site existed in a nutrient rich environment (Figure 33c). C:P and C:N ratios could not be determined for the May 2014 survey as carbon could not be assessed due to insufficient samples.

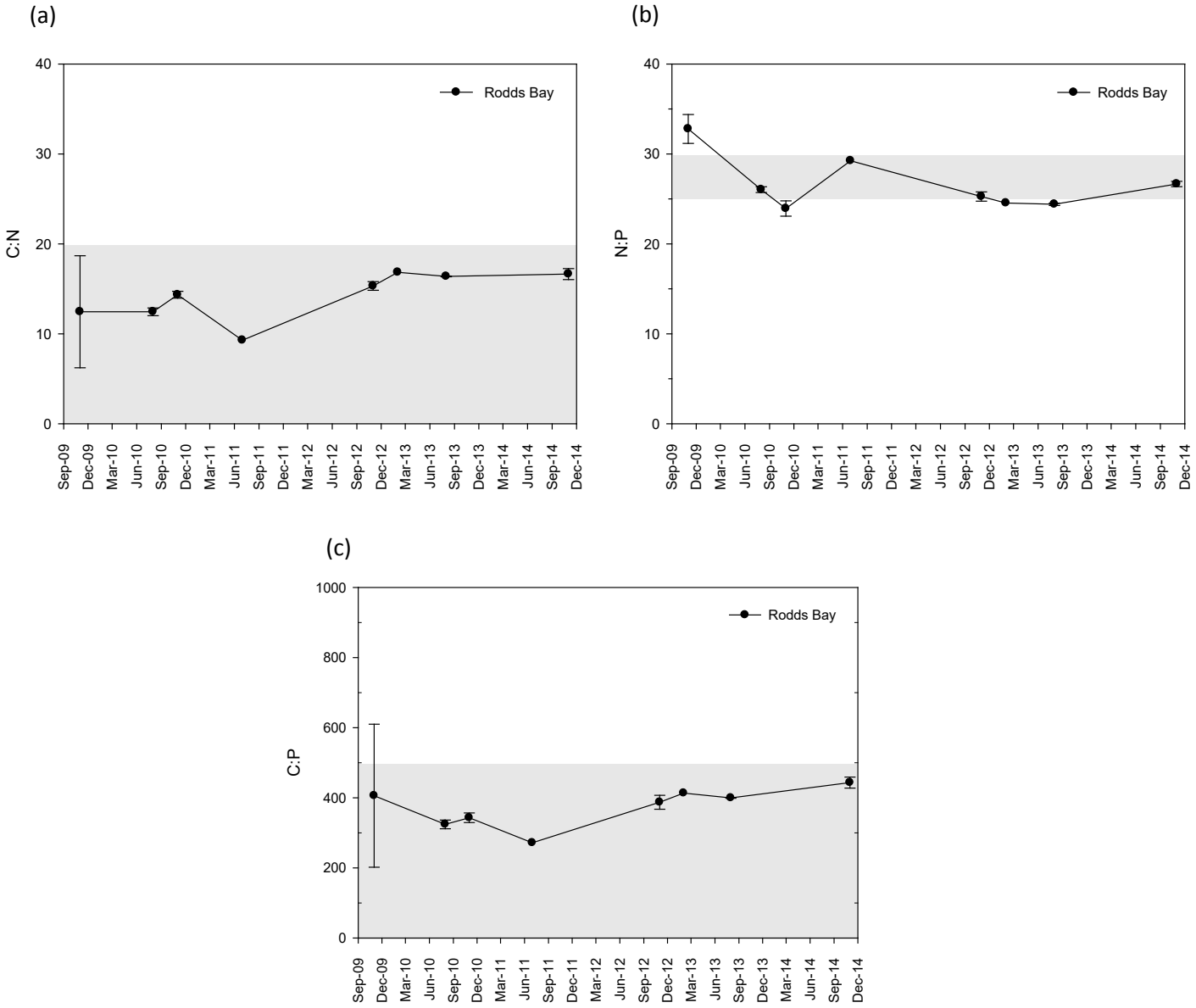


Figure 33: Elemental (atomic) ratios of seagrass leaf tissue for *Z. muelleri* subsp. *capricorni* collected at permanent transect locations at the out of Port reference sites in Rodds Bay (mean ± SE). Shaded areas indicate low light (C:N < 20), nutrient replete (N:P 25-30) and nutrient enriched (C:P < 500) environments.

3.4 Discussion

Seagrass tissue nutrient analysis at the outer harbour sites in 2014 showed the seasonal fluctuations in nutrient enrichment typically seen previously in Port Curtis, with low light conditions following the wet season and higher light conditions in the dry season. 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. Seagrass habitat at Facing Island sites was characterised as having reduced light levels for the majority of sampling events, and Pelican Banks was the only site to approach the threshold indicating favourable light conditions. Nitrogen limitation in November at Pelican Banks north may be a possible explanation for lower than average biomass in 2014 compared to previous years (Kelaher et al. 2013).

During 2011, *Z. muelleri* subsp. *capricorni* and *H. uninervis* C:P, N:P and C:N ratios declined, indicating that seagrasses were existing in nitrogen limited, low light and nutrient enriched conditions. This was likely associated with very high rainfall and river flow events in early 2011 bringing large volumes of highly turbid water and increased nutrients to the system. Drier conditions in 2012 contributed to an increase in C:N ratios (higher light availability) which persisted for the majority of the year however in 2013 C:N ratios declined, particularly at the southern end of Pelican Banks and at Facing Island. These outer harbour sites were likely impacted by very high rainfall and subsequent flooding of the Calliope River in January 2013. In 2014, declines in C:N ratios corresponded with similar trends in irradiance, with both declining in August 2014 (see Section 2).

Trends in light and seagrass abundance at these outer harbour sites generally reflected trends in tissue nutrient analysis. Marked declines in light (below locally determined thresholds) and seagrass percent cover coincided with lower C:N and C:P ratios following extreme river flow events in early 2011 and 2013. Less extreme declines in seagrass percent cover in 2012 coincide with higher C:N and C:P ratios following drier conditions.

4 SEAGRASS RESILIENCE AND RECOVERY

4.1 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 may 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 and knowledge of the seed bank status will assist in determining the resilience of these meadows to damage.

Assessments of the range of reproductive outputs and capacities (flowering, fruiting, clonal growth rates and seed banks) were examined for key species during regular quarterly monitoring of seagrass condition.

4.2 Methods

4.2.1 Reproductive effort

During quarterly sampling, 15 randomly placed sediment cores (100 mm diameter and 100 mm depth) were collected from an area adjacent to monitoring sites (approximately 25 m to the side of the site). Cores were 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* subsp. *capricorni* bears flowers in spathes on specialised flowering shoots (Figure 34b). 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.

4.2.2 Seed banks

Sediment cores, measuring 50 mm in diameter and 100 mm in depth, were collected at 0 m, 10 m, 20 m, 30 m, 40 m and 50 m along transects (approximately 2.5 m to the left of the assessed quadrat). Based on the results of initial assessments commissioned by GPC (McCormack et al. 2013), a subset of monitoring sites (Pelican Banks North (GH1), Wiggins Island (WW1) and Rodds Bay (RD1)) was selected to focus further processing efforts. Access issues precluded seed bank sampling at the original Wiggins Island site in 2013 and sampling during 2014 was 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* subsp. *capricorni,* the dominant species in Port Curtis.

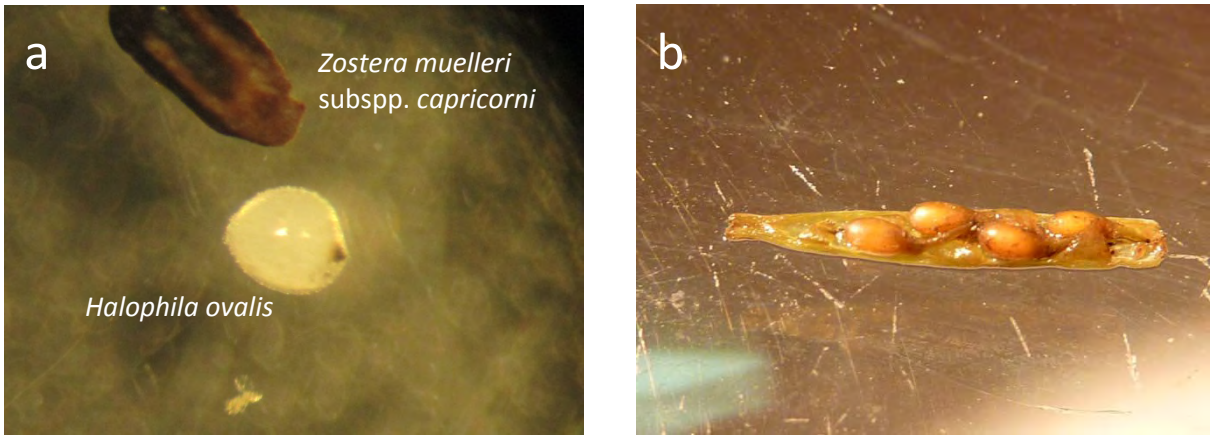


Figure 34: (a) *Zostera muelleri* subsp. *capricorni* and *Halophila ovalis* seeds and (b) *Z. muelleri* subsp. *capricorni* spathe found in sediment samples collected at permanent transects.

4.3 Results

4.3.1 Reproductive effort

The highest density of reproductive structures (total number of flowers, fruits and spathes) was found in November each year when the production of *Z. muelleri* subsp. *capricorni* spathes was at its peak (Figure 35).

In November 2014, *Z. muelleri* subspp. *capricorni* spathe density reached 602.67 ± 205.75 m^{-2} at the Pelican Banks North site. No *Z. muelleri* subspp. *capricorni* spathes were found at Pelican Banks South in 2013 or 2014. Spathes were found for the first time at Facing Island in November 2014 (25.46 ± 5.46 m^{-2}) (Figure 35 and Appendix 1).

Z. muelleri subspp. *capricorni* spathes were also found at Rodds Bay and Fisherman's Landing at the beginning of the monitoring program (November 2009) but have not been found in samples since this time. This is not unexpected given the scarcity of *Z. muelleri* subspp. *capricorni* plants at both sites in recent years.

There were no *H. ovalis* fruits/flowers found at Pelican Banks South in 2014 and a low density of male flowers found at Pelican Banks North in November (4.24 ± 4.24 m^{-2})(Figure 36). This was generally consistent with previous years, however in 2013, *H. ovalis* flowers were found throughout the year at Pelican Banks North (Appendix 1).

In August 2014, no reproductive structures were found at any of the sites except for male *H. ovalis* flowers in Rodds Bay (16.98 ± 16.98 m^{-2}) (Figure 36; Appendix 1). The only reproductive structure (excluding seeds) detected for *H. uninervis* throughout the program was a male flower found at Facing Island in November 2009.

No *Z. muelleri* subspp. *capricorni* spathes or *H. ovalis* fruits/flowers were detected in 2015 except for male *H. ovalis* flowers at Facing Island in February (8.49 ± 8.49 m^{-2}) (Appendix 1).

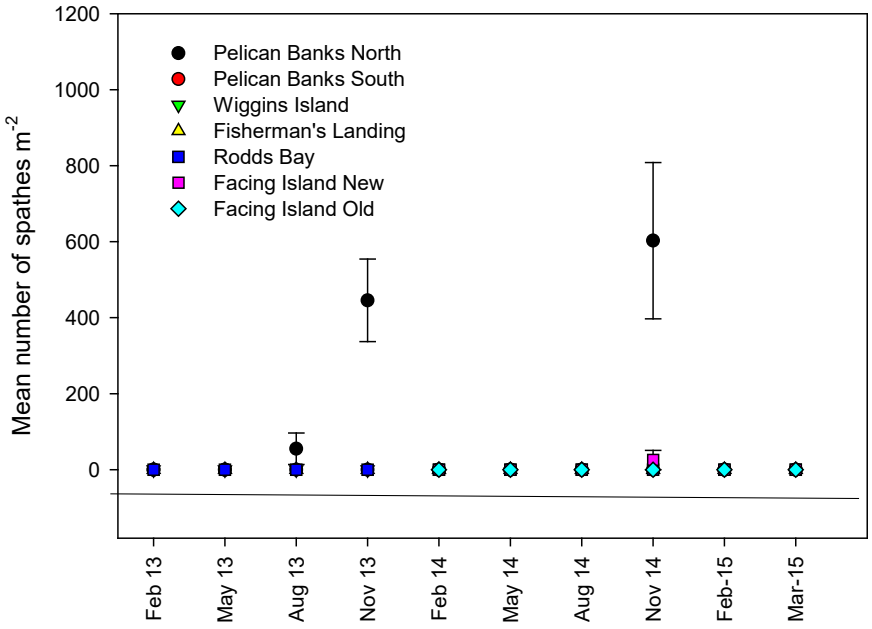


Figure 35: Density of *Z. muelleri* subspp. *capricorni* spathes (mean number of spathes m^{-2} ± SE) found in core samples taken quarterly at permanent transect locations from February 2013 - March 2015.

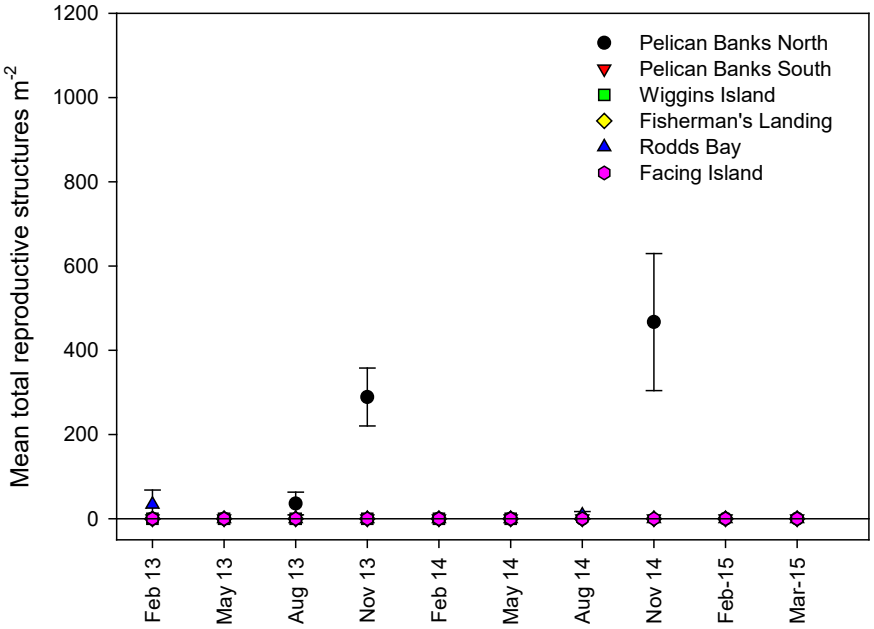


Figure 36: Total reproductive effort (mean number of reproductive structures m^{-2} ± SE) at permanent transect locations, February 2013 – March 2015 (pooled for all species); includes fruits, flowers and spathes.

4.3.2 Seed banks

In 2014, substantial seed banks were found at Pelican Banks North, Rodds Bay and Wiggins Island (Figure 37). Mean seed density at Pelican Banks North generally peaks around November and February each year following peaks in spathe production (Appendix 1); however, in 2014 the greatest densities occurred in May and November (Figure 37). Seed density was low in February 2014 despite the peak in spathes observed in November 2013 (Figure 35 and Appendix 1). In previous years, seed density peaked following low levels of spathe production (as detected in reproductive samples above) but overall density generally increased throughout 2014-15 (Figure 37). The new Wiggins Island site had the highest mean density of *Z. muelleri* subsp. *capricorni* seeds in May and November 2014 (Figure 37).

At the reference site in Rodds Bay, mean *Z. muelleri* subsp. *capricorni* seed density generally increased in 2014 following a gradual downward trend in seed density leading up to 2013 (Figure 37). Average seed density declined noticeably leading up to 2013 following small increases in seagrass abundance at the site (Davies et al. 2015). There was an increase in seed density in November 2013 despite the absence of spathes in reproductive samples (Figure 37 and Appendix 1). Rodds Bay showed the opposite pattern to the harbour sites in 2014, with peaks in February and August and the lowest seed densities in May and November (Figure 37). Mean *Z. muelleri* subsp. *capricorni* seed density peaked in March 2015 though no reproductive structures were detected at Rodds Bay for the past two years (Appendix 1).

Because of the difficulty in differentiating between *H. ovalis* and *H. decipiens* seeds, the results for these species are pooled (Figure 38). In 2013, a substantial *Halophila* spp. seed bank was detected at Rodds Bay, coinciding with an increase in the number of flowers detected in reproductive samples at the site (Figure 38 and Appendix 1). This peak in mean seed density in August 2014 followed two months of the complete absence of *Halophila* spp. reproductive structures at the site (Appendix 1). The low density of seeds in November 2014 follows the lowest density of flowers detected in the reproductive samples for this site (Figure 36). In March 2015, there was another peak in mean *Halophila* spp. seed density, similar to 2013 (Figure 38).

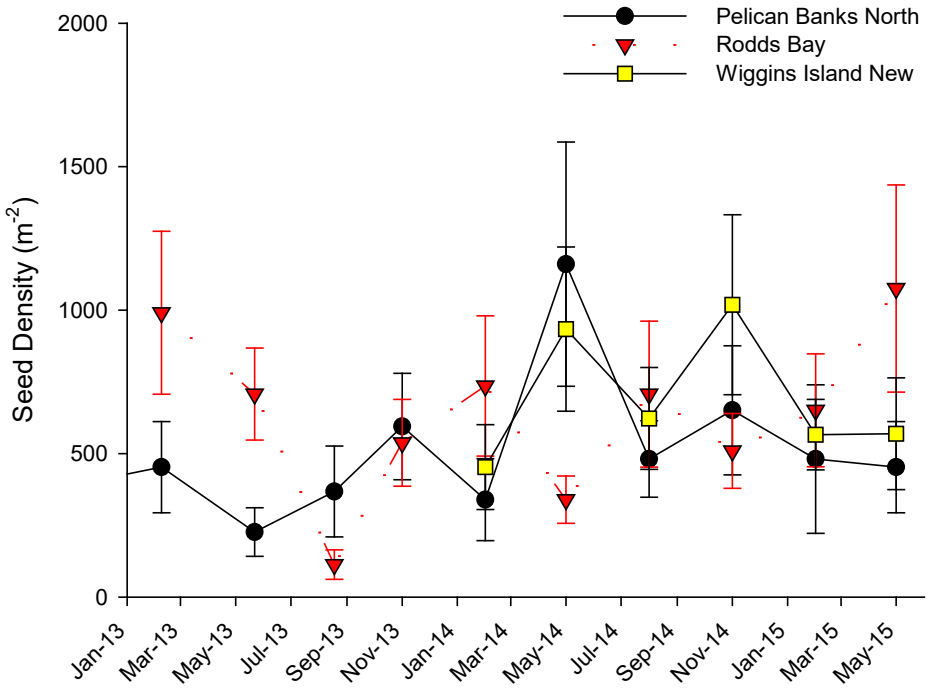


Figure 37: Mean number (± SE) of *Z. muelleri* subsp. *capricorni* seeds per square metre at Pelican Banks North, Rodds Bay and Wiggins Island New.

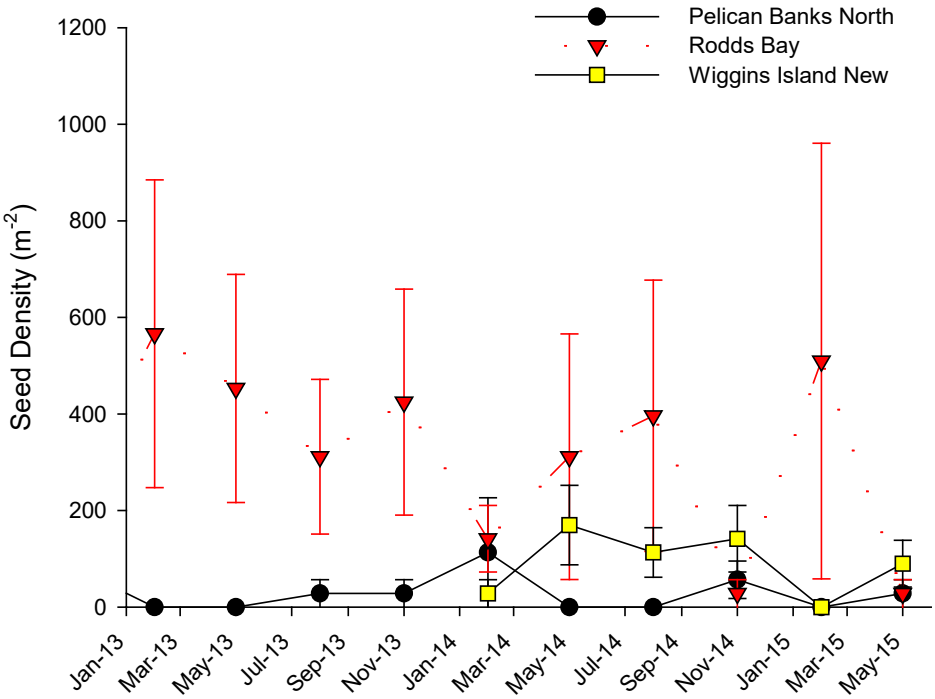


Figure 38: Mean number (± SE) of *Halophila spp.* seeds per square metre at Pelican Banks North, Rodds Bay and Wiggins Island New.

4.4 Discussion

Port Curtis seagrasses demonstrated the capacity for sexual reproduction through the production of flowers, fruits and seeds, however this varied substantially between locations and time of year. At Pelican Banks North, *Z. muelleri* subsp. *capricorni* flowered for at least four months over the growing season (from August to November) but it is possible that flowering extends beyond this period (McCormack et al. 2013). These findings are consistent with Rasheed (1999) who found flowering of *Z. muelleri* subsp. *capricorni* occurred between July and November in Cairns, and Conacher et al. (1994b), who reported an eight month flowering season in Moreton Bay in South East Queensland with shorter flowering periods in sparser areas of the meadow. The study findings confirm that *Z. muelleri* subsp. *capricorni* in Port Curtis does not appear to flower throughout the year like some *Z. muelleri* subsp. *capricorni* meadows in temperate waters (Harris et al. 1980). The relatively high density of plants at Pelican Banks capable of both sexual and asexual reproduction as well as the presence of a substantial seed bank provides several mechanisms for recovery and the Pelican Banks meadow is likely more resilient than other meadows in the inner harbour.

The peak spathe densities found at Pelican Banks North (602.67 ± 205.75 m^{-2}) are comparable to levels found in other Queensland studies, for example Cairns (110 m^2) and Moreton Bay (around 550 m^{-2}) (Rasheed 1999; Conacher et al. 1994b). A rise in water temperature has been identified as a cue for flowering in other *Zostera* species such as *Z. 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 2). Spathe production at Pelican Banks 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).

At nearby Facing Island, a low density of *Z. muelleri* subsp. *capricorni* spathes was found for the first time in November 2014 and only one *H. uninervis* flower has been identified over the duration of the program. *Halophila ovalis* flowers were also detected for the first time in May 2014 in low densities at the new site and again in February 2015. Despite reproductive structures present at different times throughout the year, the low density and frequency of these 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. The Pelican Banks meadow may act as a donor meadow for seagrass propagules more widely throughout the Port of Gladstone through dispersal of vegetative fragments, spathes and seeds.

The *Z. muelleri* subsp. *capricorni* seed bank detected at both Pelican Banks North and Rodds Bay (over 1000 m^{-2} for each site) were denser than reported for the same species in Moreton Bay (Conacher et al. 1994b). Seeds have been found 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 (Bryant et al. 2014). This is consistent with Conacher et al. (1994b) who reported seeds in the sediments up to four months after production. The high densities of seeds found at Rodds Bay despite the lack of seagrass and flowering events in recent years means the remaining seed bank is likely much older. Until very recently, the viability of this seed bank was largely unknown; however surveys undertaken in Rodds Bay in November 2014 detected significant increases in seagrass abundance in areas where seagrasses have been absent for several years. This indicates that either some of the seed bank at Rodds Bay likely remained viable to initiate recovery or propagules had recruited from other sources (McMahon et al. 2014).

Halophila species seeds at Rodds Bay also peaked in May 2015 following a lack of seagrass and flowering events at the site in the preceding surveys. There was a high abundance of *Halophila* seeds despite the complete loss of *Halophila* seagrass at the sites in March 2015 and only small proportions of *Halophila* present since August 2014. There may have been a flowering event following the peak in biomass in

November 2014 which went undetected between the November 2014 and March 2015 surveys, as the site was not able to be sampled in February 2015 due to a cyclone.

Sampling for reproductive structures within the inner harbour in 2013 was restricted to Fisherman's Landing because of access issues at the Wiggins Island site due to the construction of the Wiggins Island Coal Terminal. *Zostera muelleri* subsp. *capricorni* plants have been scarce at Fisherman's Landing in recent years, and no *Z. muelleri* subsp. *capricorni* flowers have been detected in samples at this site since the beginning of the program. Relatively high densities of *H. ovalis* flowers were detected at the site over the 2013 growing season, and it is likely that there is still a viable seed bank present (McCormack et al. 2013).

Assessments of Port Curtis 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 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 at the end of 2014 and beginning of 2015 will be available as part of a project funded under GPC's Ecosystem Research and Monitoring Program.

5 CONCLUSIONS

Seagrass abundance and health at permanent transect sites

Long-term monitoring of seagrass condition at permanent transect sites in the Port Curtis and Rodds Bay has revealed significant intra and inter-annual variability in seagrass abundance, species composition, tissue nutrient characteristics, reproductive output and productivity across sites. Within years, trends in seagrass abundance at seagrass monitoring sites have generally followed a seasonal pattern of expansion over the growing season from around July or August each year, peaking in abundance around November, and declining with the onset of the wet season and a reduction in light availability.

Longer term historical data available at 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.

There has been some recovery following the 2013 floods at sites such as Pelican Banks North in 2013 and Wiggins Island in 2014. Levels of resilience at Pelican Banks North are naturally quite high due to the multiple mechanisms for reproduction and the presence of a relatively dense seed bank. Significant increases at the meadow following declines and following major rainfall and flood events suggest that the meadow is well equipped to recover from potential impacts. At the southern end of Pelican Banks, recovery following the January 2013 flood event was short lived and, despite the rapid improvement in light levels at the site, seagrasses remained low in abundance throughout 2014 following additional low light events early in the year. Abundance surveys, tissue nutrient analysis and monitoring of seed banks suggest that the low density of plants as well as prolonged nutrient enrichment and a lack of propagules may be hampering recovery at the southern end of the bank. In the inner harbour seagrass meadows have continued to recruit each growing season, however for the majority of the inner harbour sites, the seagrass percent cover and biomass remain low and have continued to decline during 2014.

This pattern of seagrass changes was not confined to Port Curtis, with many other tropical coastal and estuarine seagrasses in Queensland exhibiting similar mixed recovery success from the impacts associated with recent floods and cyclones (Jarvis et al. 2014; Rasheed et al. 2014; York et al. 2014). The lack of recovery in 2013 was exacerbated by another extreme flood event in January of that year, when the Calliope River discharged at unprecedented levels causing declines in seagrass across monitoring sites. Encouragingly, annual surveys of complete seagrass distribution and abundance in Port Curtis in November 2014 revealed significant recovery in several regions especially The Narrows, South Trees and Rodds Bay (Carter et al. 2015). While several permanent transect sites in The Narrows and inner harbour showed increased percent cover, large-scale recovery may not be captured at the scale of monitoring conducted at the permanent transect sites. More spatially expansive monitoring that remaps seagrass meadows more broadly in the region seems to better capture overall trends. Several monitoring meadows assessed during the annual Port Curtis survey expanded in area although trends in biomass were mixed. In the absence of another extreme wet season in 2014/2015, it is anticipated that further increases in seagrass abundance will occur across other sites in Port Curtis.

Currently, seagrasses in Port Curtis have shown a capacity to recover from natural and anthropogenic impacts, however 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. 2015b; York et al. 2015b; Rasheed et al. 2014; Pollard and Greenway 2013). The extensive and detailed seagrass monitoring and research efforts in Port Curtis enhances the ability to understand potential anthropogenic impacts and implement measures to reduce the chances of exacerbating natural impacts by human activities.

Recommendations for light requirements for seagrass species in Port Curtis

As part of the WBDDP dredging management program, thresholds and triggers were established that are directly related to the light requirements of the local seagrass species *Z. muelleri* subsp. *capricorni* (Chartrand et al. In review; 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 was not limited by light falling below requirements (outside of major flood events).

Development of initial light ranges for additional seagrass species found in Port Curtis was undertaken by analysis of literature values, previous studies and the light and seagrass condition information collected as part of this program. These thresholds were developed to aid in the protection of seagrasses from potentially deteriorating light conditions associated with the proposed dredging and dredged material placement works based upon the local background light variability within Port Curtis meadows as well as the local light requirements for these species. Based on these assessments, the seagrasses in Port Curtis were maintained or increased during the growing season when the light environment was within the following thresholds:

- *Zostera muelleri* subsp. c*apricorni*: requires **6** mol photons m^{-2} day^{-1} over a rolling **14 day average.**
- *Halodule uninervi*s: requires **5** mol photons m^{-2} day^{-1} over a rolling **14 day average**.
- *Halophila* species: require **1-2** mol photons m^{-2} day^{-1} over a rolling **7 day average**.

With the exception of *Z. muelleri* subsp. *capricorni,* the light thresholds presented in this report are considered preliminary. Refinement of locally relevant light threshold for *H. uninervis* beyond literature values and other studies was not possible as during the 12 month timeframe of this study in Port Curtis (May 2014 - May 2015) and seagrasses were generally not growing in conditions approaching these light thresholds. The seagrass site at Facing Island has been monitored since November 2009 however from that time until 2013, the meadow was dominated by *Z. muelleri* subsp. *capricorni* with *H. ovalis*. Conclusions regarding light threshold ranges for *H. uninervis* are therefore not possible at this site prior to the shift in species dominance in 2013. As such the light threshold for *H. uninervis* is unable to be refined beyond the maximum of the range derived from previous studies at the time of this report.

While these threshold values present an appropriate guide for the planning and EIS stage of the Channel Duplication Project, further refinement would be required to test the relationships between light and seagrass abundance. An additional 12 months of seagrass monitoring and assessment are recommended at the three coastal seagrass sites and three deepwater seagrass sites established for the Channel Duplication Project prior to dredging works commencing to strengthen locally derived seagrass light thresholds as well as additional research activities which led to the light trigger development for *Z. muelleri* subsp. *capricorni* in Gladstone (Chartrand et al 2012).

6 APPENDIX 1

Mean (± Standard Error) number of reproductive structures for *Z. muelleri* subsp. *capricorni* at monitoring locations from February 2013 to March 2015.

Location	Wiggins Old	Wiggins New	Fisherman's Landing	Rodds Bay
Date	*Z. muelleri* subsp. *capricorni*	*Z. muelleri* subsp. *capricorni*	*Z. muelleri* subsp. *capricorni*	*Z. muelleri* subsp. *capricorni*
	Spathes m^{-2}	Spathes m^{-2}	Spathes m^{-2}	Spathes m^{-2}
February 2013	NS	NS	0	0
May 2013	NS	NS	0	0
August 2013	NS	NS	0	0
November 2013	NS	NS	0	0
February 2014	NS	NS	0	0
May 2014	NS	NS	0	0
August 2014	NS	NS	0	0
November 2014	NS	NS	0	0
February 2015	NS	NS	NS	NS
March 2015	NS	NS	0	0

NS not sampled

Mean (± Standard Error) number of reproductive structures for *Z. muelleri* subsp. *capricorni* at monitoring locations from February 2013 to March 2015.

Location	Pelican Banks North	Pelican Banks South	Facing Old	Facing New
Date	*Z. muelleri* subsp. *capricorni*	*Z. muelleri* subsp. *capricorni*	*Z. muelleri* subsp. *capricorni*	*Z. muelleri* subsp. *capricorni*
	Spathes m^{-2}	Spathes m^{-2}	Spathes m^{-2}	Spathes m^{-2}
February 2013	0	0	0	0
May 2013	0	0	0	0
August 2013	**55.17 ± 41.26**	0	0	0
November 2013	**445.63 ± 108.65**	0	0	0
February/March 2014	0	0	0	0
May 2014	0	0	0	0
August 2014	0	0	0	0
November 2014	**602.67 ± 205.75**	0	0	**25.46 ± 25.46**
February 2015	0	NS	0	0
March 2015	0	0	0	0

Mean (± Standard Error) number of reproductive structures for *Halophila ovalis* at monitoring locations from February 2013 to March 2015.

Location	Wiggins Old			Wiggins New			Fisherman's Landing			Rodds Bay		
Date	*Halophila ovalis*			*Halophila ovalis*			*Halophila ovalis*			*Halophila ovalis*		
	Fruits m^{-2}	Female Flowers m^{-2}	Male flowers m^{-2}	Fruits m^{-2}	Female Flowers m^{-2}	Male flowers m^{-2}	Fruits m^{-2}	Female Flowers m^{-2}	Male flowers m^{-2}	Fruits m^{-2}	Female Flowers m^{-2}	Male flowers m^{-2}
February 2013	NS	NS	NS	NS	NS	NS	0	0	**267.4 ± 96.6**	0	**34.0 ± 34.0**	**135.8 ± 81.9**
May 2013	NS	NS	NS	NS	NS	NS	0	0	0	0	0	**21.2 ± 21.2**
August 2013	NS	NS	NS	NS	NS	NS	0	0	0	0	0	**93.4 ± 33.4**
November 2013	NS	NS	NS	NS	NS	NS	0	0	**67.9 ± 67.9**	0	0	**109.8 ± 53.9**
February 2014	NS	NS	NS	NS	NS	NS	0	0	0	0	0	0
May 2014	NS	NS	NS	NS	NS	NS	0	0	0	0	0	0
August 2014	NS	NS	NS	NS	NS	NS	0	0	0	0	0	**17.0 ± 17.0**
November 2014	NS	NS	NS	NS	NS	NS	0	0	0	0	0	0
February 2015	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
March 2015	NS	NS	NS	NS	NS	NS	0	0	0	0	0	0

NS not sampled

Mean (± Standard Error) number of reproductive structures for *Halophila ovalis* at monitoring locations from February 2013 to March 2015.

Location	Pelican Banks North			Pelican Banks South			Facing Island- Old			Facing Island- New		
Date	*Halophila ovalis*			*Halophila ovalis*			*Halophila ovalis*			*Halophila ovalis*		
	Fruits m^{-2}	Female Flowers m^{-2}	Male flowers m^{-2}	Fruits m^{-2}	Female Flowers m^{-2}	Male flowers m^{-2}	Fruits m^{-2}	Female Flowers m^{-2}	Male flowers m^{-2}	Fruits m^{-2}	Female Flowers m^{-2}	Male flowers m^{-2}
February 2013	0	0	**250.4 ± 68.1**	0	0	**46.7 ± 28.3**	0	0	**34.0 ± 34.0**	0	0	0
May 2013	0	0	**199.5 ± 73.2**	0	0	0	0	0	0	0	0	0
August 2013	0	0	**335.3 ± 87.9**	0	0	**182.5 ± 62.2**	0	0	0	0	0	0
November 2013	**8.5 ± 5.9**	**46.7 ± 20.7**	**1218.1 ± 226.2**	0	0	**377.7 ± 121.7**	0	0	0	0	0	0
February 2014	0	0	0	0	0	0	0	0	0	0	0	0
May 2014	0	0	0	0	0	0	0	0	0	0	**12.7 ± 12.7**	**34.0 ± 29.9**
August 2014	0	0	0	0	0	0	0	0	0	0	0	0
November 2014	0	0	**4.2 ± 4.2**	0	0	0	0	0	0	0	0	0
February 2015	0	0	0	NS	NS	NS	0	0	0	0	0	**8.49± 8.49**
March 2015	0	0	0	0	0	0	0	0	0	0	0	0

7 APPENDIX 2

Light thresholds (expressed as % of surface irradiance (%SI) and total daily mol photons m^{-2} d^{-1}), impact and time (days) to impact of the seagrass species.

Species	Location	Impact	Light intensity	Time to Impact (days)	Zone	Study Location	Notes	Reference
Zostera muelleri* subsp. *capricorni	Moreton Bay	Max depth limit of meadow	4.6 mol photons m^{-2} d^{-1}		Sub-tropical	Intertidal		Longstaff 2003
	Port Hacking	**No impact found** - Biomass	7.2 mol photons m^{-2} d^{-1}	86	Sub-tropical	Lab	Shading exp- not measured prior to 86 days	Fyfe 2004
	Magnetic Island	Leaves per shoot Shoot density	4.4 mol photons m^{-2} d^{-1} 4.4 mol photons m^{-2} d^{-1}	46 61	Tropical	Lab		Collier et al. 2012
	Magnetic Island	Above-ground biomass	1.7 mol photons m^{-2} d^{-1}	≤34	Tropical	Lab	Biomass only measured at time 3 (days 28-34).	Collier et al. 2011
	Lake Macquarie	Above-ground biomass	2 mol photons m^{-2} d^{-1}	≤84	Sub-tropical	Lab	Shading exp- not measured prior to 86 days	York et al. 2013
	Lake Macquarie	**No impact found** – Above-ground biomass	4.8 mol photons m^{-2} d^{-1} 6.9 mol photons m^{-2} d^{-1}	≤84	Sub-tropical	Lab	Shading exp- not measured prior to 86 days	York et al. 2013
	Gladstone	Above-ground biomass Percent cover	≤5 mol photons m^{-2} d^{-1} (14 day mean)	30-40	Sub-tropical	Intertidal	In situ shading exp; season has a strong effect with light; Interaction between light and temp likely a major driver.	Chartrand et al. 2015 in prep
Halophila ovalis	Magnetic Island	Shoot density	0, 1.6 & 3.3 mol photons m^{-2} d^{-1}	7-28	Tropical	Lab	Shading exp – measured at time 0 then at 4, 7, 10 and 14 weeks. Declines under warm temperature (27.7˚C) treatment.	Collier et al. 2015 subm.
Halophila decipiens	Green Island	Shoot density	1 mol photons m^{-2} d^{-1}	14	Tropical	Lab	Shading exp- measured weekly. Declines under 26 ˚C and 30 ˚C treatments	Chartrand et al. 2014
Halophila spinulosa	Bowen	Shoot density	1 mol photons m^{-2} d^{-1}	28	Tropical	Lab	Shading exp- measured weekly. Declines under 26 and 30 ˚C treatments	Chartrand et al. 2014
	Bowen	**No impact found** – Shoot density	3.2 mol photons m^{-2} d^{-1}	28	Tropical	Lab	Shading exp- measured weekly.	Chartrand et al. 2014

Halodule uninervis	Moreton Bay	Biomass	≤50% SI	≤30	Sub-tropical	Lab	Shading exp- not measured prior to 30 days.	Grice et al. 1996
	Magnetic Island	**No impact found** – Photosynthesis	1.7 mol photons m^{-2} d^{-1}	≤34	Tropical	Lab	No impact at 27 or 33 ˚C.	Collier et al. 2011
	Magnetic Island	Above-ground biomass	1.7 mol photons m^{-2} d^{-1}	≤34	Tropical	Lab	Biomass only measured at time 3 (days 28-34).	Collier et al. 2011
	Magnetic Island	Percent cover loss	<4 mol photons m^{-2} d^{-1}	≤84	Tropical	Subtidal	Seagrass cover loss relates to the entire meadow which includes other spp.	Collier et al. 2012a
	Magnetic Island	Percent cover gain	>5 mol photons m^{-2} d^{-1}	≤84	Tropical	Subtidal	Seagrass cover loss relates to the entire meadow which includes other spp.	Collier et al. 2012a
	Gulf of Carpentaria	Biomass Shoot density Canopy height	≤1% SI	>38	Tropical	Intertidal		Longstaff and Dennison 1999
	Magnetic Island	Shoot density	4.4 mol photons m^{-2} d^{-1}	60	Tropical	Lab		Collier et al. 2012b
	Magnetic Island	Shoot density	0, 1.6 & 3.3 mol photons m^{-2} d^{-1}	28-49	Tropical	Lab	Shading exp – measured at time 0 then at 4, 7, 10 and 14 weeks. Declines under warm temperature (27.7˚C) treatment.	Collier et al. 2015 subm.

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Appendix H
Gladstone Western Basin and Channel Duplication BPAR Monitoring

TropWATER
Centre for Tropical Water and Aquatic Ecosystem Research

JAMES COOK UNIVERSITY
AUSTRALIA

GLADSTONE WESTERN BASIN & CHANNEL DUPLICATION BPAR MONITORING May - August 2015

Davies JN, Scott EL, Bryant CV & Rasheed MA

Report No. 15/51

September 2015

GLADSTONE WESTERN BASIN & CHANNEL DUPLICATION BPAR MONITORING May – August 2015

A Report for Gladstone Ports Corporation (GPC)

Report No. 15/51

September 2015

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INTRODUCTION

The dredging projects associated with port expansion in the Western Basin were identified as potentially posing a high level of environmental risk to the marine habitats of the area, particularly to seagrass meadows. A detailed seagrass and light assessment program has been established for the Western Basin project to assist in management of the project during and after dredging. This program includes a requirement for continued assessment of seagrasses and Benthic Photosynthetically Active Radiation (BPAR) or benthic light following the end of dredging activities to assess seagrass recovery. These post-dredge seagrass and light monitoring requirements are outlined in several conditions and plans connected with the Western Basin Dredging and Disposal Project (WBDDP) and a strategy to meet them was developed and endorsed by the Dredge Technical Reference Panel.

In May 2014, benthic light monitoring equipment was also established at additional sites (Colosseum Inlet, South Trees and Quoin Island) to expand the seagrass and light assessment program to areas most likely to be impacted by dredging associated with the proposed Channel Duplication project. In order to put any changes to seagrasses that occur during the project into perspective, it is critical to collect pre-dredge background information on the natural seasonal and inter annual changes in seagrasses in relation to benthic irradiance at the intended sensitive receptor monitoring sites for as long a period as possible.

METHODS

Benthic light monitoring equipment was initially deployed in December 2013 at the seven locations identified for Western Basin post-dredge monitoring (Map 1). 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. In May 2014 benthic light monitoring equipment was deployed at the three new locations identified for baseline monitoring for the Channel Duplication Environmental Impact Statement (Map 1). These sites were decommissioned during the August 2015 survey.

Submersible Odyssey™ photosynthetic irradiance autonomous loggers (light loggers) were deployed with automatic wiper units which clean the optical surface of the sensor every 15 minutes to prevent marine organisms fouling the sensors (Figure 1A & B). At each location, two independent light loggers and wiper units were deployed on the seabed approximately 5 to 10m apart and affixed to a metal stake. The presence of dual loggers reduces the risk of data loss through unit malfunction, fouling or disturbance during the three months between equipment exchange and download. During the initial three months of the program, light loggers were replaced and downloaded every month to provide confidence in the equipment and to ensure no loss of data. From March 2013, equipment has been changed out and downloaded approximately every three months.

At Black Swan and Wiggins Island, where the sediment is extremely soft, repeated visits on ground may compromise the site. At this location, two independent light loggers and wiper units were deployed on two separate deployment cradles connected to a surface float (Figure 1C & D) to allow loggers to be exchanged at high tide. At Colosseum Inlet, seagrasses do not expose tidally and light loggers were also deployed on cradles to allow exchanges at high tide.

A reference logger and wiper unit was also deployed outdoors at the Queensland Boating and Fisheries Patrol office at the Gladstone Marina to assess trends in surface irradiance and assist QA/QC procedures. In September 2014 a second reference logger and wiper unit was deployed outdoors at AB Marine at the Gladstone Marina to further aid in the assessment of surface irradiance trends. Base data for part of the monitoring period has been excluded in the following figures due to suspected shading. Base loggers have been relocated to rectify shading issues. Base data for the period from 14th June to 31st July sourced from Vision Environment Queensland (2014).

Odyssey loggers log a cumulative reading at 15 minute intervals, which is calibrated and summed to gain total daily benthic light (mol m^{-2} day^{-1}). The raw data captured by the loggers requires calibrating to a known light value. Sensors were calibrated against a certified reference irradiance sensor (LI-CORTM LI-192SB Underwater Quantum Sensor) in controlled laboratory conditions. A custom built calibration device held the light loggers a consistent distance from the source light and excluded all external light. To allow for the differences in light absorption properties between air (calibration medium) and water (deployment media), a multiplication factor of 1.33 was applied to data during submersion periods based on the tide record (supplied by MSQ) and the depth of each logger.

Total daily rainfall is sourced from the Bureau of Meteorology (station 039123).

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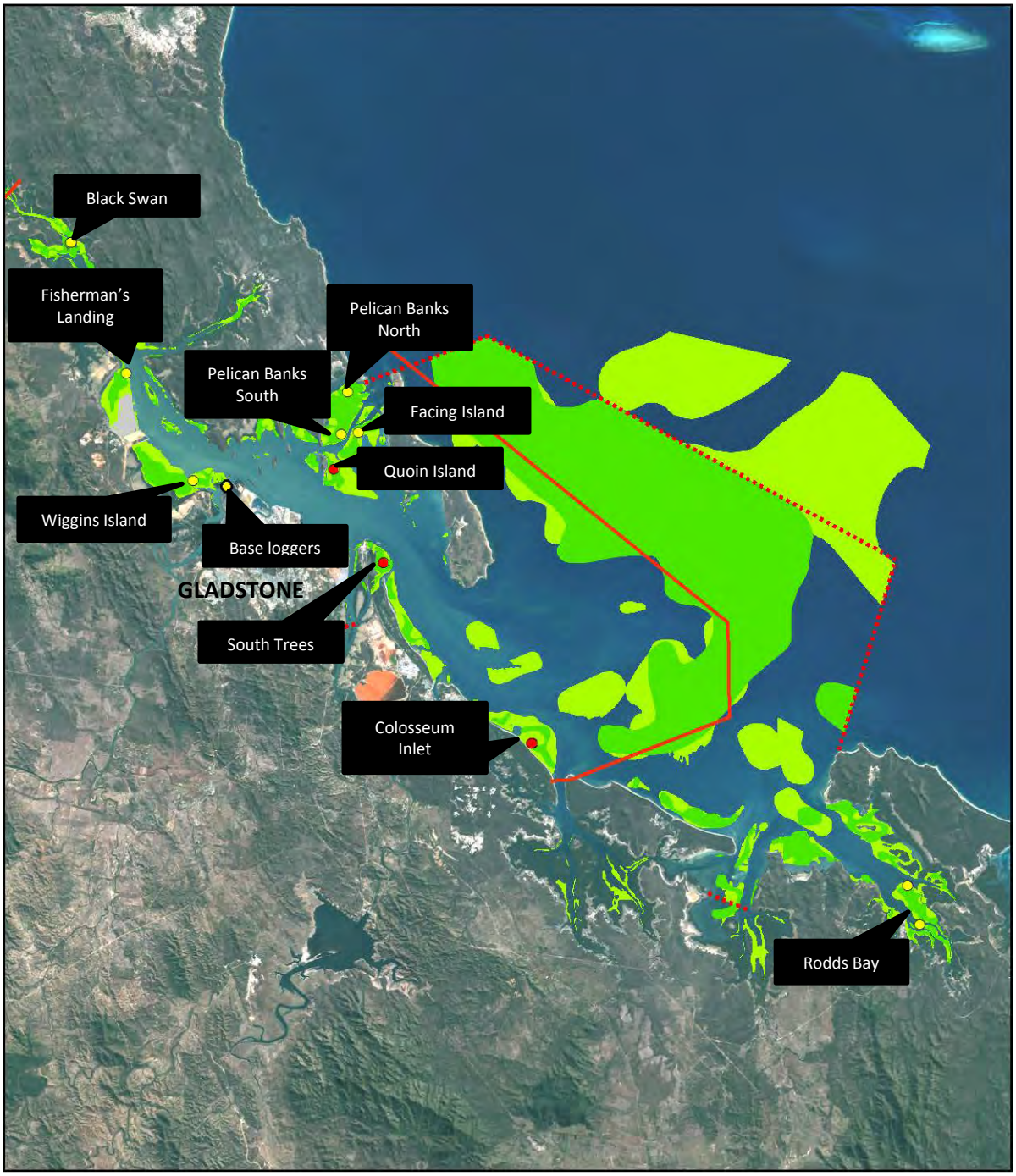
Map 1. Seagrass Permanent Transect and BPAR monitoring locations



Davies et al. 2015. 'Gladstone Western Basin & Channel Duplication BPAR Monitoring, May - August 2015', TropWATER Publication 15/51, JCU, Cairns.

Figure 1. Odyssey BPAR loggers and wiper units deployed on stakes in the sediment (A & B) and on deployment cradles connected to surface floats (C & D).

DATA PROCESSING AND QA/QC PROCEDURES

Benthic light monitoring equipment utilised for this work is not set up for reactive management of a dredge operation where real-time data is required to ensure day-to-day compliance. Rather, it is intended to track the relationship between light and seagrass change and to interpret seasonal dynamics and/or extreme changes to the light climate which may affect seagrass success during this post-dredge term.

When data loggers are retrieved from the field, a report on their condition and appearance is made to identify any fouling of the sensor or issues with the wiping apparatus. After downloading, data is adjusted using the most recent calibration factor for each logger and adjusted for periods of submersion (see methods above). The adjusted data is used to calculate a total daily benthic light value for each logger.

At each site and for the control bases at the Gladstone Marina, loggers A & B are compared to determine if there are any significant differences (>20%) in total daily light value. This comparison was initially undertaken for 15 minute increments however, for some sites the distance between loggers meant that differences in excess of 20% were common. Even a distance of 10m can cause loggers to differ substantially during exposure and submersion events. It was therefore considered more accurate to consider the total daily light when discerning differences between loggers.

Obvious failures of one unit mean that data is removed from the record (e.g. where a logger failed to record data or where there were exceptionally high readings outside of possible light levels). Data is also removed from the record when the condition report indicated that fouling had occurred. Where there is no obvious cause for the difference between units, the total daily light value for that site is compared to the data at the other nearby sites and to the reference site to assist in ascertaining which is the errant logger.

If both loggers are deemed to be recording accurately and loggers agree (difference <20%), a mean total daily light value is calculated. Where there are no obvious malfunctions and units disagreed by more than 20% in one day, the data for the unit recording the higher value is used as it is more likely to be reading within the 5% error margin for that logger. The lower of the two is considered more likely to have been affected by physical disturbances such as shading by floating debris.

During this phase of the post-dredge monitoring program where a continuous dataset is not as critical – if there is no compelling evidence as to which of the two loggers is functioning correctly, then the data may be quarantined from use in any future compliance situations.

RESULTS

Interpretation of any changes in the condition of seagrasses in relation to benthic light at permanent monitoring sites is provided in an annual report detailing results of seagrass and benthic light monitoring at sensitive receptor sites (Davies et al. 2015). Figures 2 to 12 represent a 14 day rolling average of the total daily benthic light value at each of the monitoring sites since monitoring began in December 2013 (Western Basin sites) and May 2014 (Channel Duplication sites).

Studies conducted in Gladstone by Chartrand et al. (2012) found that the dominant species *Zostera capricorni* subsp. *muelleri* requires between 4.5 and 12 mol quanta m^{-2} day^{-1} over a minimum period of two weeks to survive during the growing season, typically from July to January. From these studies, a working light threshold value of 6 mol quanta m^{-2} day^{-1} over a two week period had been successfully used as a management threshold in Gladstone Harbour during GPC operations (Chartrand et al. 2012). At this level there is sufficient light to maintain growth of *Z. muelleri* subsp. *capricorni*. In the following figures, light levels in *Z. muelleri* subsp. *capricorni* dominated meadows fall below this threshold value periodically as expected with the natural variation in the light environment, with the majority occuring during the senescent season. Periods of time below this threshold value are part of expected background conditions, particularly during the seagrass senescent season (Chartrand et al. 2012).

Light levels can vary substantially between sites, as light is recorded at the seagrass canopy level for this study rather than at a standard depth. This is because the goal of the light monitoring was to determine associations between seagrass changes and changes in light that the various monitoring meadows experience.

This report accompanies data sets provided to Gladstone Ports Corporation and Aurecon. The data presented here updates data presented in previous reports where more accurate tidal exposure information has been obtained.

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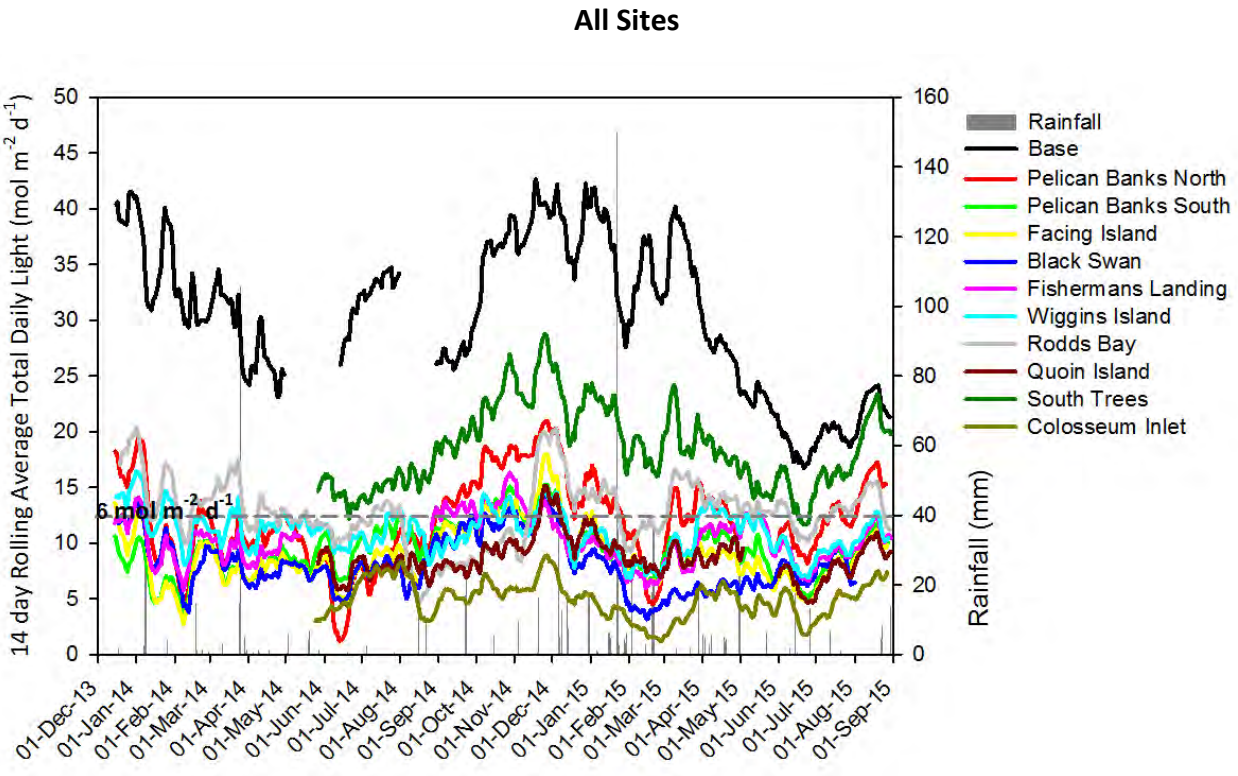

Figure 2. Average total daily benthic light (presented as a 14 day rolling average) at seagrass permanent transect sites in Gladstone Harbour and Rodds Bay, and daily surface light from Base loggers.

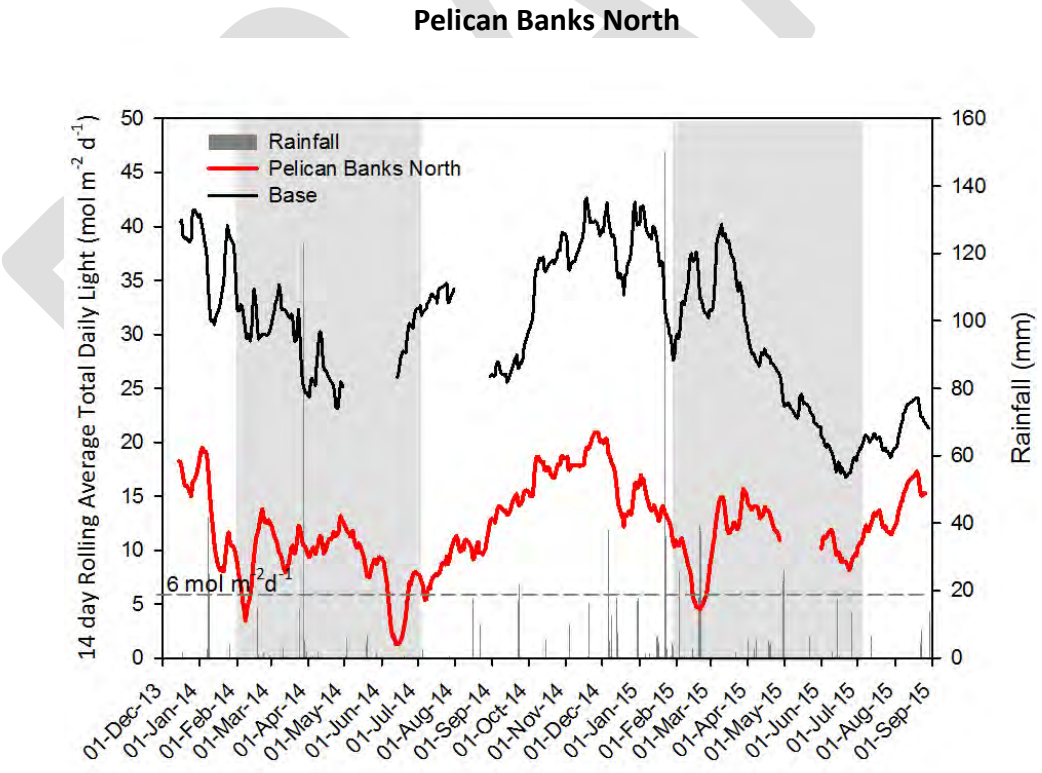

Figure 3. Average total daily benthic light (presented as a 14 day rolling average) at the Pelican Banks North seagrass monitoring site and daily surface light from Base loggers. Shading represents the seagrass senescent season in Gladstone (February through June).

Pelican Banks South

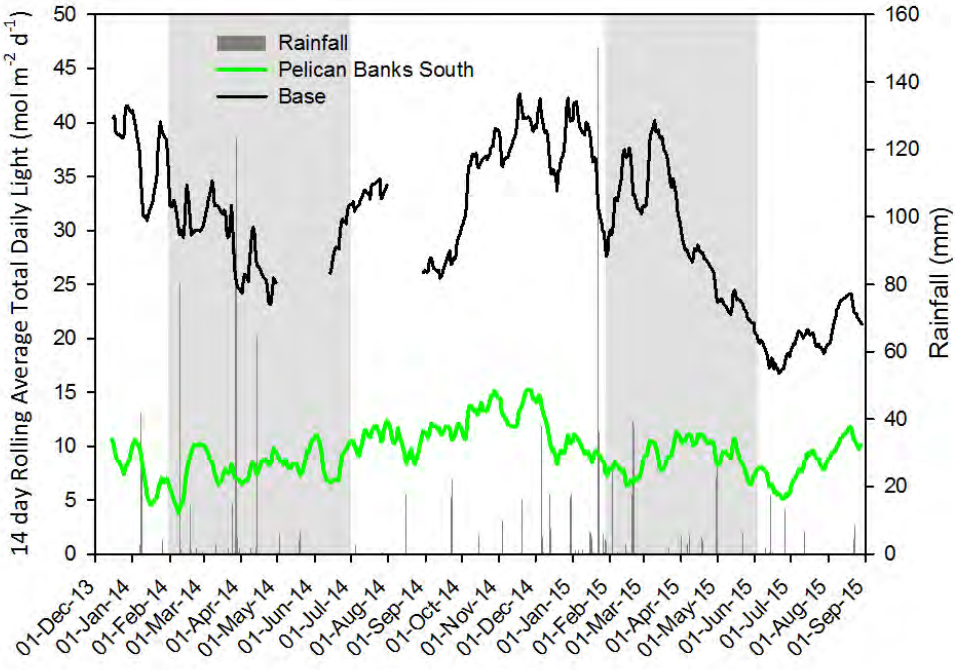

Figure 4. Average total daily benthic light (presented as a 14 day rolling average) at the Pelican Banks South seagrass monitoring site and daily surface light from Base loggers. Shading represents the seagrass senescent season in Gladstone (February through June).

Facing Island

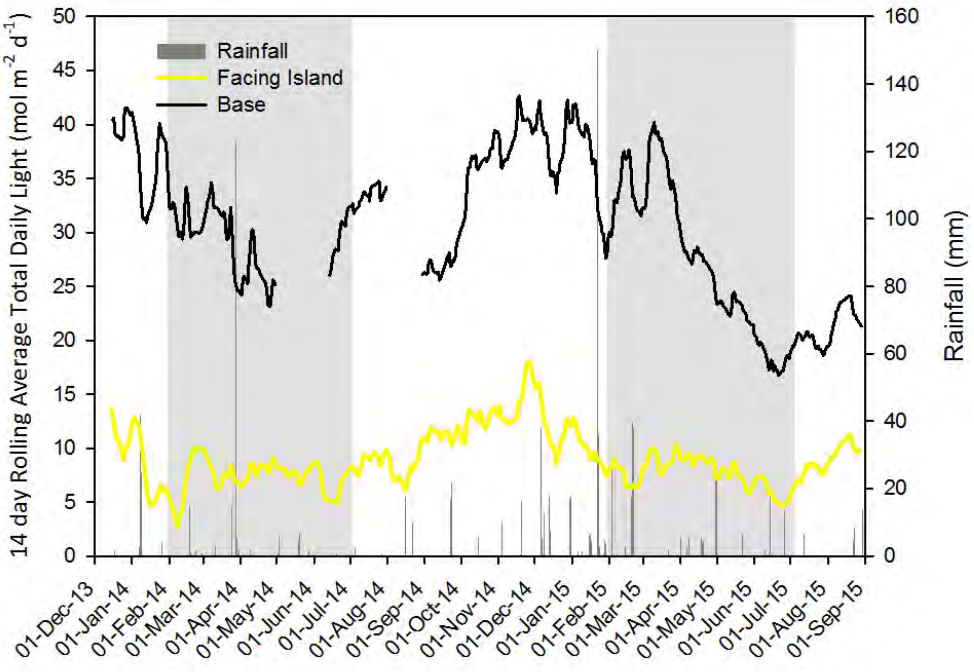

Figure 5. Average total daily benthic light (presented as a 14 day rolling average) at the Facing Island seagrass monitoring site and daily surface light at from Base loggers. Shading represents the seagrass senescent season in Gladstone (February through June).

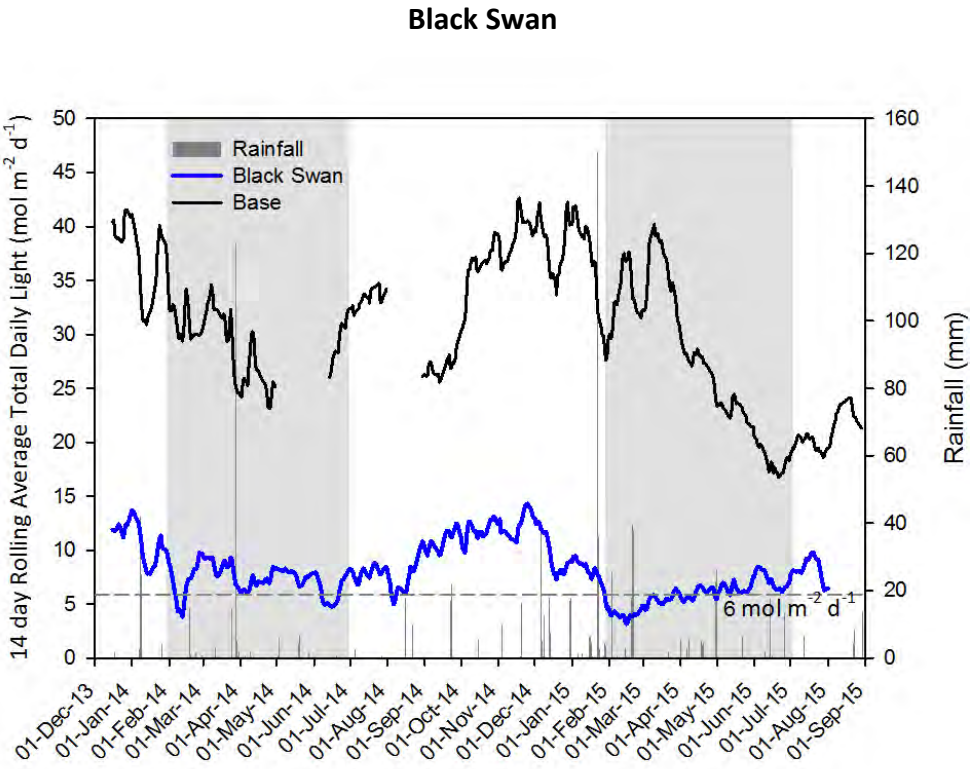

Figure 6. Average total daily benthic light (presented as a 14 day rolling average) at the Black Swan seagrass monitoring site and daily surface light at from Base loggers. Shading represents the seagrass senescent season in Gladstone (February through June).

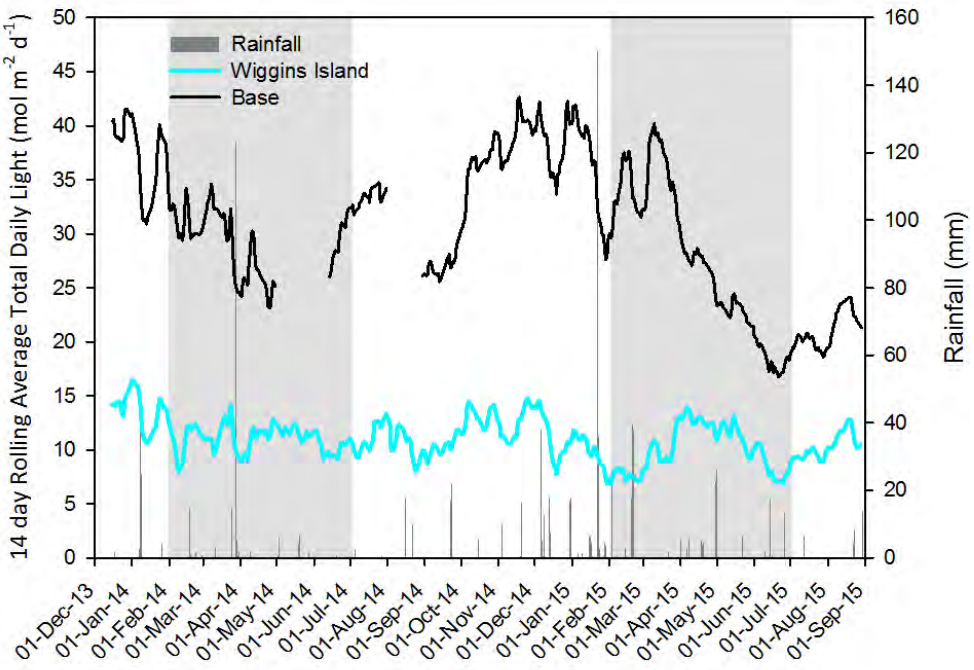

Figure 7. Average total daily benthic light (presented as a 14 day rolling average) at the Wiggins Island seagrass monitoring site and daily surface light at from Base loggers. Shading represents the seagrass senescent season in Gladstone (February through June).

Fisherman's Landing

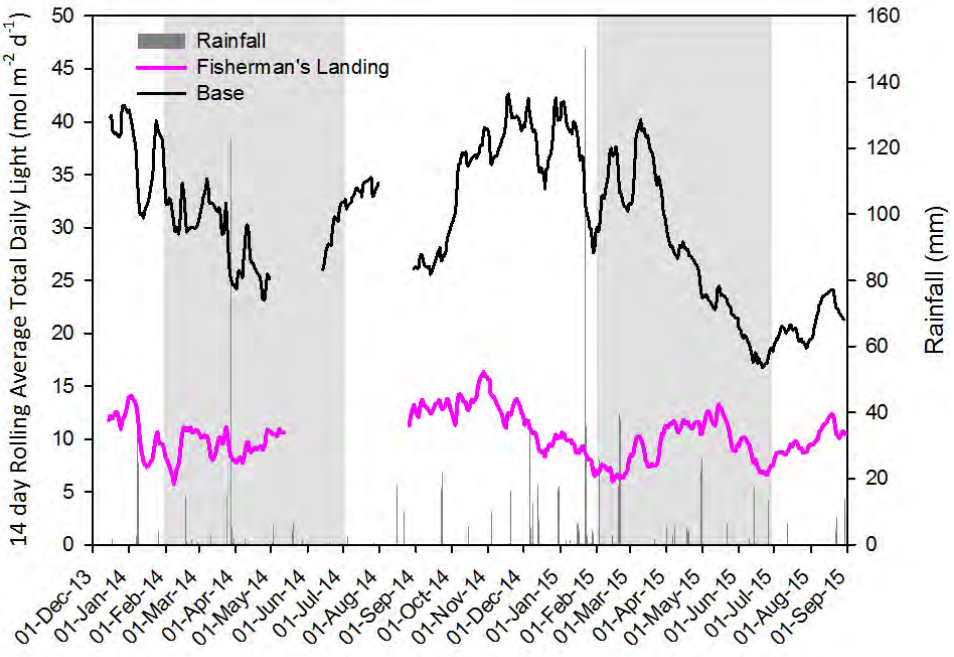

Figure 8. Average total daily benthic light (presented as a 14 day rolling average) at the Fisherman's Landing seagrass monitoring site and daily surface light at from Base loggers. There was damage to equipment shortly after the May deployment and there is no data available for this period. Shading represents the seagrass senescent season in Gladstone (February through June).

Rodds Bay

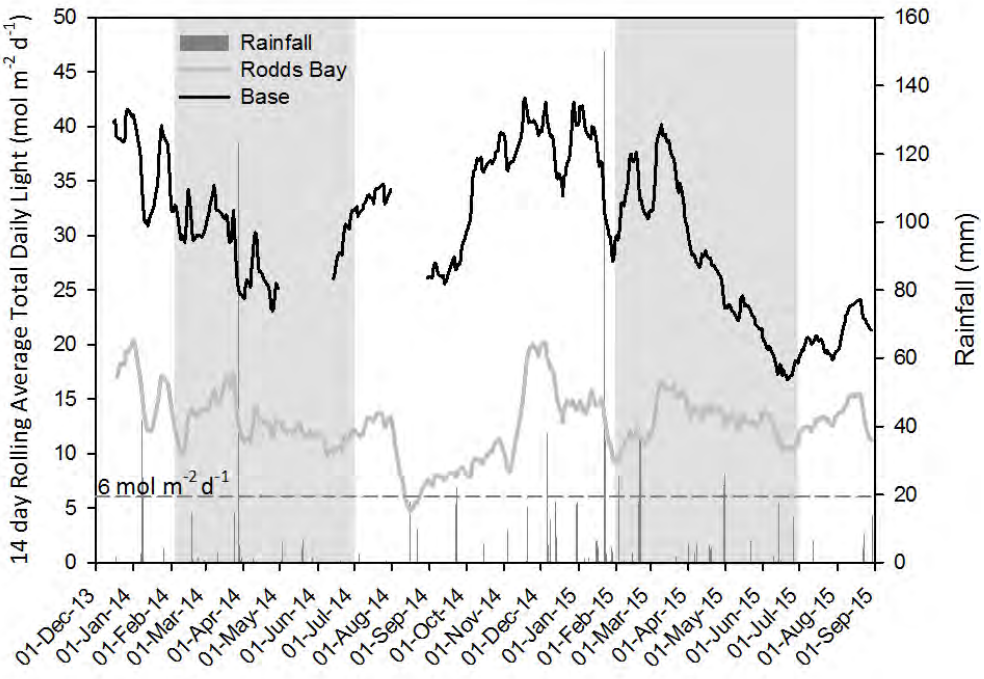

Figure 9. Average total daily benthic light (presented as a 14 day rolling average) at the Rodds Bay seagrass monitoring site and daily surface light from Base loggers. Shading represents the seagrass senescent season in Gladstone (February through June).

Quoin Island

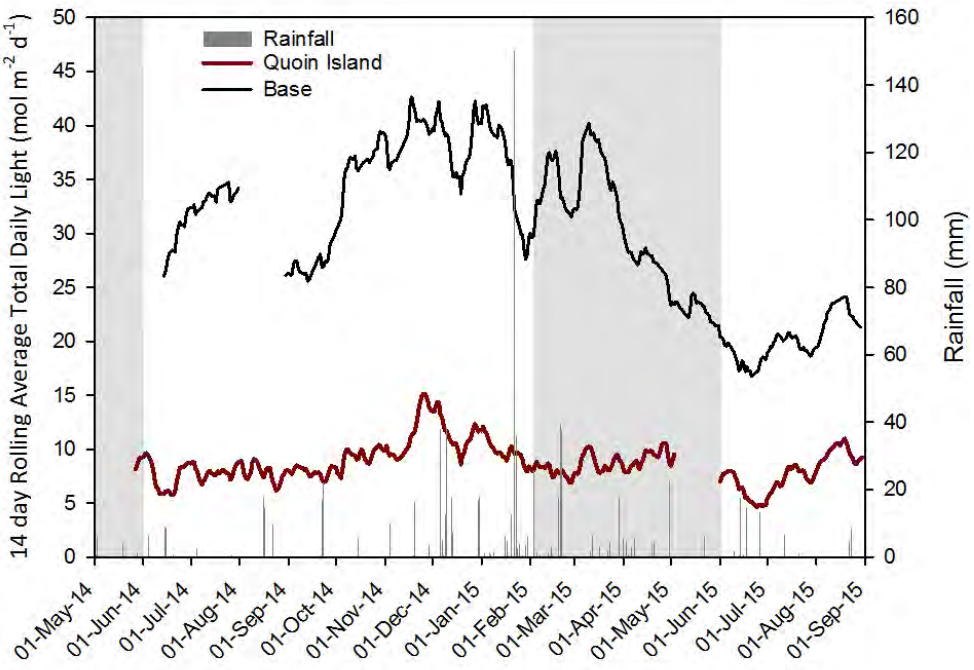

Figure 10. Average total daily benthic light (presented as a 14 day rolling average) at the Quoin Island seagrass monitoring site and daily surface light at from Base loggers. Shading represents the seagrass senescent season in Gladstone (February through June).

South Trees

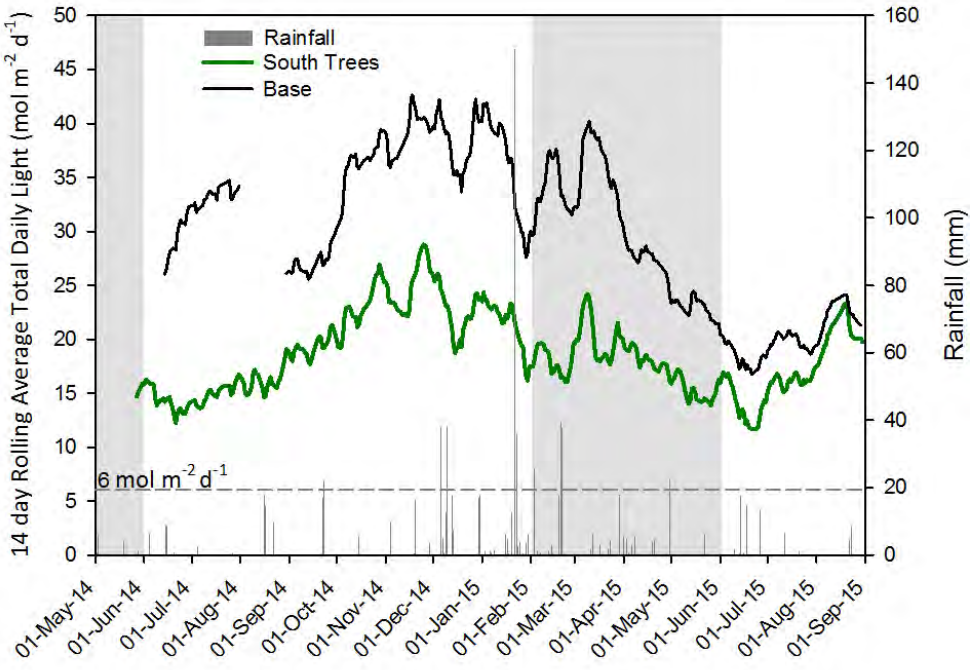

Figure 11. Average total daily benthic light (presented as a 14 day rolling average) at the South Trees seagrass monitoring site and daily surface light at from Base loggers. Shading represents the seagrass senescent season in Gladstone (February through June).

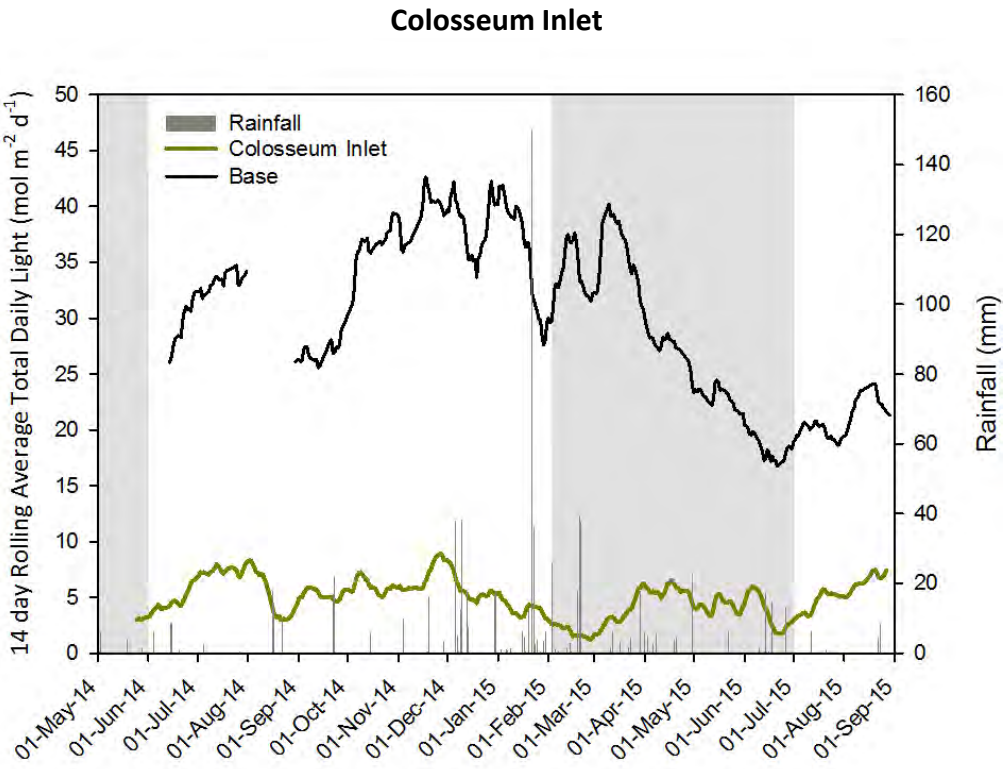

Figure 12. Average total daily benthic light (presented as a 14 day rolling average) at the Colosseum Inlet seagrass monitoring site and daily surface light at from Base loggers. Shading represents the seagrass senescent season in Gladstone (February through June).

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Appendix I Marine algal species known or predicted to occur within the Port Curtis region

Appendix K

Marine algal species known or predicted to occur within the Port Curtis region

Table K1 Wildlife online database results

Algal group	Species name
blue-green algae	*Scytonema hofman-bangii*
blue-green algae	*Blennothrix lyngbyacea*
blue-green algae	*Microcoleus vaginatus*
blue-green algae	*Entophysalis conferta*
blue-green algae	*Entophysalis deusta*
blue-green algae	*Hormothamnium enteromorphoides*
blue-green algae	*Calothrix pilosa*
blue-green algae	*Hormothamnium* sp.
blue-green algae	*Oscillatoria* sp.
blue-green algae	*Schizothrix* sp.
blue-green algae	*Spirulina subsalsa*
brown algae	*Chnoospora implexa*
brown algae	*Colpomenia sinuosa*
brown algae	*Neoralfsia expansa*
brown algae	*Dictyota divaricata*
brown algae	*Lobophora variegata*
brown algae	*Mesospora schmidtii*
brown algae	*Feldmannia globifera*
brown algae	*Hincksia mitchelliae*
brown algae	*Nemacystus decipiens*
brown algae	*Sargassum aquifolium*
brown algae	*Sargassum polycystum*
brown algae	*Dictyota bartayresiana*
brown algae	*Feldmannia irregularis*
brown algae	*Sphacelaria tribuloides*
brown algae	*Hydroclathrus clathratus*
brown algae	*Stypopodium flabelliforme*
brown algae	*Sphacelaria novae-hollandiae*
brown algae	*Asterocladon rhodochortonoides*
brown algae	*Turbinaria ornata*
brown algae	*Petalonia fascia*
brown algae	*Padina australis*
brown algae	*Padina boryana*
brown algae	*Dictyopteris* sp.
brown algae	*Sphacelaria* sp.

Algal group	Species name
brown algae	*Ectocarpus* sp.
brown algae	*Distromium* sp.
brown algae	*Sargassum* sp.
brown algae	*Lobophora* sp.
brown algae	*Padina* sp.
brown algae	*Ralfsia* sp.
green algae	*Codium* sp.
green algae	*Boodlea* sp.
green algae	*Caulerpa* sp.
green algae	*Derbesia* sp.
green algae	*Halimeda* sp.
green algae	*Neomeris* sp.
green algae	*Bornetella* sp.
green algae	*Cladophora* sp.
green algae	*Phaeophila* sp.
green algae	*Avrainvillea* sp.
green algae	*Enteromorpha* sp.
green algae	*Microdictyon* sp.
green algae	*Rhizoclonium* sp.
green algae	*Ulva ralfsii*
green algae	*Halimeda tuna*
green algae	*Cladophoropsis*
green algae	*Bryopsis indica*
green algae	*Blidingia minima*
green algae	*Bryopsis pennata*
green algae	*Caulerpa peltata*
green algae	*Codium geppiorum*
green algae	*Halimeda opuntia*
green algae	*Cladophora dalmatica*
green algae	*Cladophora pellucida*
green algae	*Halimeda cylindracea*
green algae	*Halimeda micronesica*
green algae	*Ostreobium quekettii*
green algae	*Palmophyllum crassum*
green algae	*Parvocaulis clavatus*
green algae	*Parvocaulis parvulus*
green algae	*Penicillus nodulosus*
green algae	*Avrainvillea lacerata*
green algae	*Caulerpa cupressoides*
green algae	*Caulerpa lentillifera*

Algal group	Species name
green algae	*Cladophora herpestica*
green algae	*Endophyton laurenciae*
green algae	*Microdictyon obscurum*
green algae	*Pedobesia ryukyuensis*
green algae	*Phaeophila dendroides*
green algae	*Rhizoclonium riparium*
green algae	*Enteromorpha clathrata*
green algae	*Entocladia marchantiae*
green algae	*Rhipidosiphon javensis*
green algae	*Caulerpa sertularioides*
green algae	*Chlorodesmis fastigiata*
green algae	*Pringsheimiella striata*
green algae	*Microdictyon umbilicatum*
green algae	*Dictyosphaeria versluysii*
green algae	*Pringsheimiella gratulans*
green algae	*Pseudendoclonium submarinum*
green algae	*Ulva flexuosa subsp. Flexuosa*
green algae	*Cladophoropsis vaucheriiformis*
green algae	*Pringsheimiella sanctae-luciae*
green algae	*Caulerpa racemosa var. racemosa*
green algae	*Caulerpa racemosa var. occidentalis*
green algae	*Boodlea composita*
green algae	*Boodlea montagnei*
green algae	*Caulerpa racemosa*
green algae	*Caulerpa webbiana*
green algae	*Codium spongiosum*
green algae	*Neomeris annulata*
green algae	*Udotea orientalis*
green algae	*Ulvaria oxysperma*
green algae	*Ulvella calcicola*
green algae	*Anadyomene plicata*
green algae	*Caulerpa brachypus*
green algae	*Caulerpa serrulata*
green algae	*Chlorodesmis major*
green algae	*Derbesia attenuata*
green algae	*Entocladia robusta*
green algae	*Entocladia viridis*
green algae	*Halimeda discoidea*
green algae	*Halimeda macroloba*
green algae	*Penicillus sibogae*

Algal group	Species name
green algae	*Ulvella setchellii*
green algae	*Valonia macrophysa*
green algae	*Boodleopsis pusilla*
green algae	*Bornetella capitata*
green algae	*Caulerpa fastigiata*
green algae	*Caulerpella ambigua*
green algae	*Entocladia flustrae*
green algae	*Entocladia velutina*
green algae	*Halimeda macrophysa*
green algae	*Neomeris vanbosseae*
green algae	*Ochlochaete hystrix*
green algae	*Parvocaulis exiguus*
green algae	*Blastophysa rhizopus*
red algae	*Hypoglossum heterocystideum*
red algae	*Rhodophyceae Bangia*
red algae	*Hypnea* sp.
red algae	*Crouania* sp.
red algae	*Gelidium* sp.
red algae	*Laurencia* sp.
red algae	*Martensia* sp.
red algae	*Gelidiella* sp.
red algae	*Lomentaria* sp.
red algae	*Audouinella* sp.
red algae	*Dasyclonium* sp.
red algae	*Hydrolithon* sp.
red algae	*Peyssonnelia* sp.
red algae	*Polysiphonia* sp.
red algae	*Herposiphonia* sp.
red algae	*Ceramium codii*
red algae	*Hypnea pannosa*
red algae	*Predaea weldii*
red algae	*Spermothamnion* sp.
red algae	*Taenioma nanum*
red algae	*Amphiroa crassa*
red algae	*Ceramium maryae*
red algae	*Champia parvula*
red algae	*Chondria repens*
red algae	*Dasya iyengarii*
red algae	*Digenea simplex*
red algae	*Hypnea spinella*

Algal group	Species name
red algae	*Jania adhaerens*
red algae	*Wrangelia argus*
red algae	*Ceramium camouii*
red algae	*Galaxaura rugosa*
red algae	*Amphiroa foliacea*
red algae	*Anotrichium tenue*
red algae	*Chondria minutula*
red algae	*Laurencia humilis*
red algae	*Laurencia pannosa*
red algae	*Laurencia pygmaea*
red algae	*Laurencia venusta*
red algae	*Peyssonnelia evae*
red algae	*Plocamium hamatum*
red algae	*Stylonema alsidii*
red algae	*Ceramium flaccidum*
red algae	*Dasya baillouviana*
red algae	*Gelidiella acerosa*
red algae	*Gracilaria arcuata*
red algae	*Halichrysis micans*
red algae	*Laurencia flexilis*
red algae	*Liagora maderensis*
red algae	*Myriogramme carnea*
red algae	*Polysiphonia herpa*
red algae	*Audouinella saviana*
red algae	*Bangia atropurpurea*
red algae	*Pterocladiella caerulescens*
red algae	*Champia vieillardii*
red algae	*Chondria dangeardii*
red algae	*Laurencia intricata*
red algae	*Laurencia majuscula*
red algae	*Laurencia ptychodes*
red algae	*Peyssonnelia calcea*
red algae	*Ceramium clarionense*
red algae	*Crouania minutissima*
red algae	*Euptilota articulata*
red algae	*Haloplegma duperreyi*
red algae	*Laurencia filiformis*
red algae	*Plysiphonia blandii*
red algae	*Spyridia filamentosa*
red algae	*Titanophycus validus*

Algal group	Species name
red algae	*Acrosorium ciliolatum*
red algae	*Ceramium mazatlanense*
red algae	*Chroodactylon ornatum*
red algae	*Crouania capricornica*
red algae	*Erythrotrichia carnea*
red algae	*Eucheuma denticulatum*
red algae	*Gelidiopsis intricata*
red algae	*Herposiphonia secunda*
red algae	*Peyssonnelia inamoena*
red algae	*Portieria hornemannii*
red algae	*Trichogloea requienii*
red algae	*Acanthophora spicifera*
red algae	*Botryocladia leptopoda*
red algae	*Centroceras apiculatum*
red algae	*Centroceras clavulatum*
red algae	*Cryptonemia caliculata*
red algae	*Dichotomaria marginata*
red algae	*Herposiphonia pacifica*
red algae	*Laurencia brongniartii*
red algae	*Laurencia mariannensis*
red algae	*Lomentaria corallicola*
red algae	*Melanamansia glomerata*
red algae	*Platysiphonia delicata*
red algae	*Polysiphonia infestans*
red algae	*Polysiphonia upolensis*
red algae	*Antithamnion percurrens*
red algae	*Asparagopsis taxiformis*
red algae	*Corallophila huysmansii*
red algae	*Coriophyllum setchellii*
red algae	*Mesophyllum mesomorphum*
red algae	*Parviphycus tenuissimus*
red algae	*Peyssonnelia conchicola*
red algae	*Polysiphonia scopulorum*
red algae	*Schizoseris bombayensis*
red algae	*Tricleocarpa cylindrica*
red algae	*Vanvoorstia spectabilis*
red algae	*Antithamnion lherminieri*
red algae	*Audouinella microscopica*
red algae	*Ceratodictyon spongiosum*
red algae	*Griffithsia heteromorpha*

Algal group	Species name
red algae	*Heterosiphonia crispella*
red algae	*Laurencia parvipapillata*
red algae	*Griffithsia subcylindrica*
red algae	*Leptofauchea anastomosans*
red algae	*Polysiphonia sphaerocarpa*
red algae	*Tolypiocladia glomerulata*
red algae	*Chondrophycus carolinensis*
red algae	*Leveillea jungermannioides*
red algae	*Pterocladia caloglossoides*

Source: EHP (2015)

Appendix J
Reef and Nekton Baseline Data Collection

Port of Gladstone Gatcombe and Golding Cutting Channel Duplication Project

Environmental Impact Statement

Reef and Nekton Baseline Data Collection

August 2015

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Acronyms

ANOSIM	Analysis of Similarity
ANOVA	Analysis of Variance
BACI	Before After Control Impact
B-C	Bray-Curtis
BEST	Biota and/or Environmental Matching Model
BRUVS	Baited Remote Underwater Video Stations
CPCe	Coral Point Count estimates software program
CV	Coefficient of Variation
DEHP	Department of Environment and Heritage Protection
DoE	Commonwealth Department of the Environment
DistLM	Distance based Linear Models
DMPA	Dredged Material Placement Areas
EIS	Environmental Impact Statement
GBRMP	Great Barrier Reef Marine Park
GPC	Gladstone Ports Corporation
GPS	Global Positioning System
INTERPOL	Surface Interpolation module in IDRISI®-Selva™
IUCN	International Union for Conservation of Nature
LDPE	Low Density Polyethylene
LNG	Liquefied Natural Gas
LSD	Least Significant Difference
MANOVA	Multivariate Analysis of Variance
NATA	National Association of Testing Authorities
*n*MDS	*non-metric* Multi-Dimensional Scaling
PERMANOVA	Permutational Multivariate Analysis of Variance
PCO	Principle Coordinates Ordination
PSD	Particle Size Distribution
QA/QC	Quality Assurance/Quality Control
RGB	Red, Green and Blue
RWS	Rope, Wand or Shadow
SCUBA	Self-Contained Underwater Breathing Apparatus
± se	Standard error
ToR	Terms of Reference

EXECUTIVE SUMMARY

As part of the baseline data collection strategy to support the process of the Environmental Impact Statement for the Port of Gladstone Gatcombe and Golding Cutting Channel Duplication Project (referred to as 'the Project'), Vision Environment undertook surveys and habitat mapping of coastal fringing reef habitats within the potential Project area of impact. Concurrent surveys in 2014 and 2015 included the examination of reef associated and estuarine/coastal nekton (fish) communities. Surveys were conducted in both the dry and wet seasons to identify potential seasonal differences in habitats and/or communities. The purpose of the surveys was to identify and describe the existing environment, comparing the results to previous studies and where possible, identifying the drivers of change.

Coral Reef Communities: Multiple transects (including those permanently established) were surveyed and photographed by divers at major reef locations (Oyster Rocks/Gatcombe Heads, Sable Chief Rocks, Jenny Lind Banks and Seal Rocks), to determine community demographics of reef assemblages. Substrate cover was calculated based on major and minor substrate categories, in addition to coral health categories. Broadscale high resolution substrate community maps were produced for each reef location based on the community composition, and species lists were produced.

A survey area of 424.3ha was classified within 'the Project potential impact area' (i.e. the Projects potential direct and indirect impact areas as determined by the results of preliminary hydrodynamic modelling). The largest area of reef was encountered at Sable Chief Rocks (~208ha) and the smallest area of reef (< 2ha) at Oyster Rocks. Ten major substrate classes were identified among all locations. The highest average cover was dominated by sediments (i.e. sand, pavement and rock: 55 ± 4.6%), followed by macroalgae (29 ± 3.5%), and hard corals (13 ± 6.4%), with the lowest major classes (< 2%) consisting of soft corals, sponges and ascidians, and other biota. Although the focused study areas were not identical, the results were consistent and within similar ranges, found in previous surveys of the Port Curtis area.

Of the substrate types among locations, the highest noticeable difference was hard coral with over five times more hard coral observed at Sable Chief Rocks compared to all other locations surveyed, indicating this to be the most significant coral reef location in the Projects potential impact area. With the exception of sponges, which made up < 1% of the total cover for all sites, there were no seasonal differences in any of the major category substrates. The results were similar to those of previous studies and also suggest there has been little change in percent cover of substrates from 2012 to 2015.

A number of coral bleaching events have occurred in neighbouring northern inshore reefs of the Great Barrier Reef in the last five years. However, overall, very little coral substrate in this study demonstrated significant signs of stress, with less than 0.5% of corals showing pale colouration and less than 0.1% presenting bleached white colouration. Although some sedimentation was recorded, there were no signs of diseased corals at any of the locations. Growth rates were similar among surveyed locations. Damage to the coral at Seal Rocks, most likely from the passing of tropical Cyclone Marcia in February 2015, was noted.

Reef Associated Fish: Multiple transects were surveyed and photographed by divers at the major reef locations and fish and nekton species (actively swimming aquatic organisms), recorded over two seasons. Obligate reef fish biodiversity was measured as: density, species richness, diversity and species evenness. Baited Remote Underwater Video

Stations were deployed to supplement the transect data. Additionally, coral community data was used to determine if differences in reef associated nekton were related to different types of substrate habitat. To date, no studies on relationships between fish communities and substrate cover have been conducted within Port Curtis.

A total of 6,037 fish from 59 species were encountered throughout the study locations during the 2014 dry season (37%) and the 2015 wet season (63%). The most common fish species was the Yellowtail demoiselle (*Neopomacentrus azysron*) (41%) followed by the Spotted-tail wrasse (*Coris caudimacula*) (11%), and the Wards damsel (*Pomacentrus wardi*) (10%). The highest numbers of fish and species were encountered at Sable Chief Rocks and Jenny Lind Banks. There were no significant differences observed in fish biodiversity between the wet and dry seasons.

Hard coral substrate was found to be the best correlation to explain fish community assemblages among locations**.** However, macroalgae was also correlated with fish assemblages between seasons. The results suggest that different fish communities may be governed by the type of reef structure present. Hence, a change in substrate type (e.g. from coral to macroalgae) is likely to lead to a change in fish communities. This result has also been noted in other studies of reefs throughout the world (Baker et al. 2008, Harding et al. 2006, Hay 1984, Johansen and Jones 2013, Nero and Sealey 2005, Pratchett et al. 2011, Wenger and McCormick 2013, Wenger et al. 2014).

Estuarine and Coastal Fish: A number of coastal and estuarine areas extending from Gatcombe Heads to The Narrows, were targeted for fishing using two netting techniques (cast netting and gill netting) over two seasons. All fish collected were identified to species level, enumerated and measured before being released.

A total of 2,936 fish representing 34 species were caught in cast nets during the 2014 dry season (58.7%) and the 2015 wet season 2015 (41.3%) surveys. The most common species occurring in most locations irrespective of season, was the Southern herring, (*Herklotsichthys castelnaui*) (39%), followed by the Estuary glassfish (*Ambassis marianus*) (21%) and the Spottyface anchovy (*Stolephorus waitei*) (14%). Banana prawns (*Fenneropenaeus merguiensis*) were also common among catches (7%). The highest numbers of fish were encountered at surveyed locations at Lilleys Inlet, South Trees Inlet, Boat Creek South and The Narrows/Targinnie Creek. Despite differences in fish communities among some locations, overall, there were no significant differences observed in fish biodiversity between the dry season and the wet season.

A total of 44 fish were collected from all locations during the 2014 dry season (52%) and 2015 wet season (48%) utilising gill nets. However, a high number of fish were also observed (but not collected) and recorded during the two monitoring campaigns. The most common fish caught, irrespective of season was the Giant queenfish (*Scomberoides commersonnianus*) (27%), followed by the Blue threadfin salmon (*Eleutheronema tetradactylum*) (11%), Beach salmon (*Leptobrama muelleri*) (11%), and the Mud crab (*Scylla serrata*) (11%). A large number of fish caught with gill nets during both seasons, were of recreational or commercial value (67%).

Two organisms, collected by gill net and returned to the water during the two surveys campaigns, are listed under the International Union for Conservation of Nature and Natural Resources Red List of Threatened Species: the Green sea turtle (*Chelonia mydas*) (endangered), and the Blacktip reef shark (*Carcharhinus melanopterus*) (near threatened)

(IUCN 2015). The Green sea turtle is also listed as vulnerable under provisions of the *Environment Protection and Biodiversity Conservation Act 1999*. Several other fish species, previously recorded for this region, are listed as vulnerable or critically endangered on the International Union for Conservation of Nature and Natural Resources Red List, and under provisions of the *Environment Protection and Biodiversity Conservation Act 1999*.

1 INTRODUCTION

Marine ecological monitoring was undertaken by Vision Environment for Gladstone Ports Corporation Ltd (GPC), to provide baseline data to support the process of the Environmental Impact Statement (EIS) for the Port of Gladstone Gatcombe and Golding Cutting Channel Duplication Project (referred to as 'the 'Project').

The deepening of the existing passing channels is needed to improve the Port of Gladstone's operational efficiency and to mitigate an existing and increasing potential shipping incident risk, as the Port's throughput increases. The duplication of the Gatcombe and Golding Cutting shipping channels will facilitate improved flexibility for vessel passing within the Port, which will address the above mentioned need.

The duplication of the existing channels involves dredging to remove approximately 12.6Mm^3 of seabed material and its potential placement at one, or a combination of four, proposed Dredged Material Placement Areas (DMPAs). The marine ecology assessment was proposed to address the Queensland Government's EIS Terms of Reference (ToR) and Commonwealth Department of the Environment (DoE) Guidelines for an EIS requirement to describe the existing marine environment at potential Project direct and indirect impact areas (the potential Project impact area). The potential Project impact area has been defined as the area to be dredged, DMPAs, new navigational aid locations, as well as adjoining areas within potential zones of impact, determined by the results of preliminary hydrodynamic modelling (Aurecon 2014).

This report presents data collected from June to August 2014 (dry season/winter sampling) and January to March 2015 (wet season/summer sampling), for reef, fish and nekton habitats in the potential Project impact area. Major reef locations were examined for reef and fish communities. Fish communities were also examined at a number of additional estuarine locations. A summary of reef and fish communities has been described, quantified and mapped in both wet and dry seasons, to identify any seasonal differences and to confirm reef habitat zonation in more detail. Combined results, interpretation and comparisons to previous assessments are reported in this document.

2 METHODOLOGY

2.1 Coral Reef Communities

2.1.1 Sites

Numerous coastal fringing reefs dominate the open coastal waters east of Port Curtis. The wider geographical area was scanned by TropWater using sled tows undertaken during seagrass habitat mapping in November 2013 (McKenna et al. 2014), to identify the location of minor and major coastal fringing reefs. Using additional historical information from Ayling et al. (2012) and Oceania Maritime Consultants (2011), five reef locations were selected for targeted detailed analyses. Monitoring locations were established taking into account maritime constraints (e.g. shipping channels) and include reef communities at Oyster Rocks/Gatcombe Heads East Point, Sable Chief Rocks, Jenny Lind Banks and Seal Rocks (refer Figure 2.1). Permanent monitoring transects were established at four of the locations for subsequent repeat surveys, which would contribute to a Before, After, Control, Impact (BACI) design. A permanent transect was not included at Oyster Rocks due to high current and low visibility restraints.

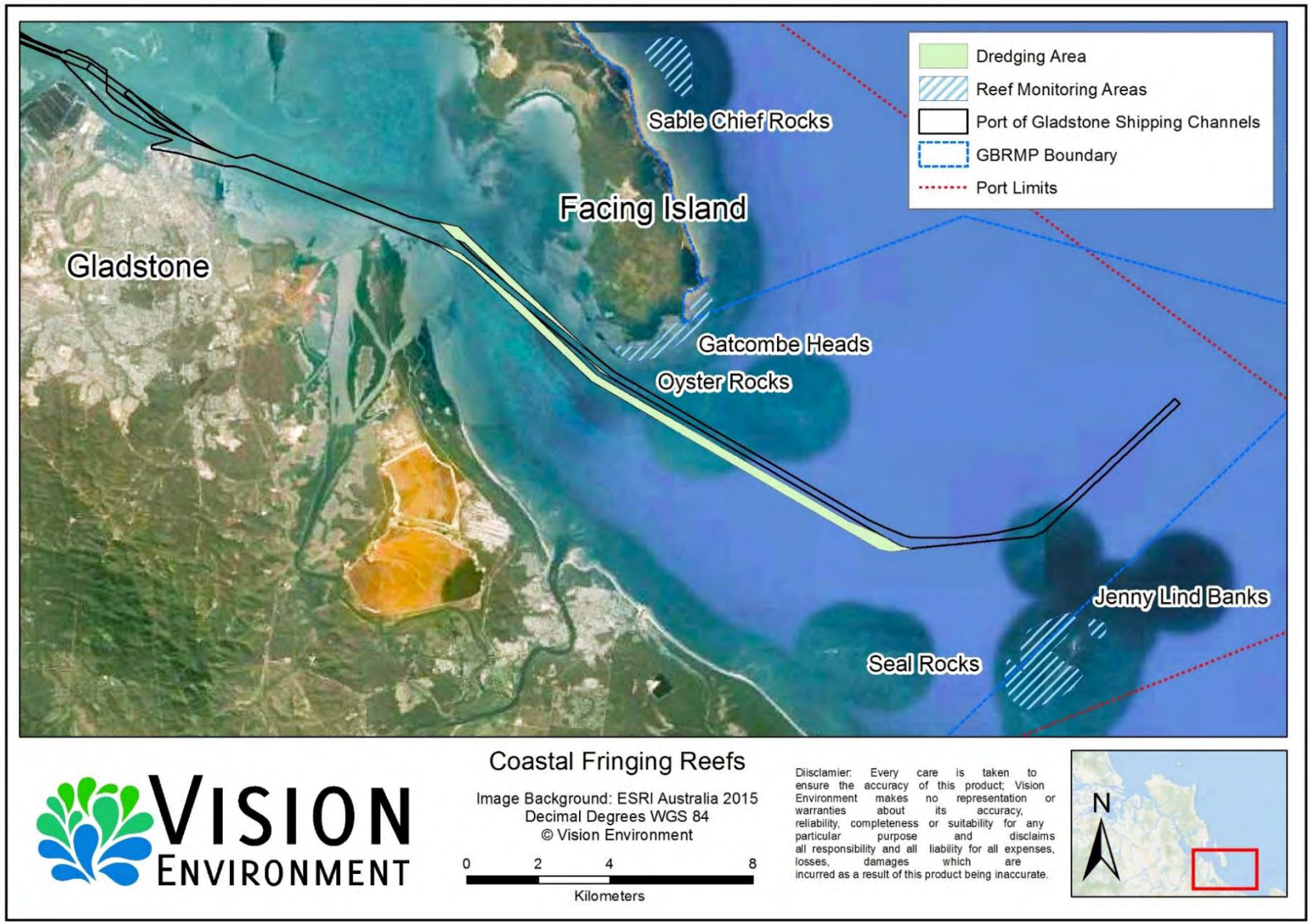


Figure 2.1 Site locations of coastal fringing reefs (in blue) in reference to the Project area

2.1.2 Sampling approach and method

Multiple 50m long belt transects were randomly established along each location, within similar depth contours and continuous reef top (average depth of 4.6m ± 0.6m). The transect start and end was marked with a Global Positioning System (GPS) (Garmin GPS60; WGS84; accuracy 3 to 15m), as well as established track lines, which were attached to a float dragged by tethered divers. Along each transect, photographic quadrats were captured at 5m intervals using a ®Canon G12 (6.1-30.5mm) camera in an underwater housing (WP-DC34) set on the lowest wide angle setting.

Each image was captured 1 m above the benthos using a reference bar to standardise each photograph. A 50mm scale was incorporated into each frame to calculate surface area of each quadrat. Image captures were taken parallel to the benthos to minimise parallax error. The program Coral Point Count estimates (CPCe) (Kohler and Gill 2006), was used to analyse each frame. Depth was recorded for each image capture using a dive computer. Images were taken between June and August 2014. A number of transects (between 6 to 15 per location), were also revisited during the 2015 wet season (January to March 2015), in order to determine if substrate communities had changed seasonally.

2.1.3 Data analyses

Random point count methodology was used to determine community demographics of reef assemblages, pavements and other non-lining substrates. A total of 20 random points were overlayed over each image capture and substrate cover identified beneath each point by an experienced reef taxonomist. A previous survey was used to determine the optimum number of points required to be overlayed over each image, with 20 points showing the highest accuracy per time invested for that particular sized quadrat area (Alquezar and Boyd 2007). When a point intersected a part of an image where rope, wand or shadow (RWS) was present, the substrate underneath was identified and recorded only if there was a high degree of certainty that the substrate underneath the RWS was the same as the neighbouring pixels (e.g. the continuation of a branch from a branching coral). When there was a high degree of uncertainty, points were identified as RWS and these data points were not included in the analysis. Each point was entered and stored as a data code for each frame and calculated as percent substrate cover for each frame in addition to total transect.

A second taxonomist analysed 50% of all transects for quality assurance/quality control (QA/QC) purposes. If the transect accuracy was lower than 95% between the two taxonomists, transects were re-analysed and results discussed. Substrate cover was calculated based on major substrate categories and minor/sub-categories over the whole transect. The major categories for the frame analysis consisted of:

- Sediment (e.g. sands, pavement and rock)
- Macroalgae
- Hard corals
- Soft corals
- Sponges and ascidians
- Other living biota, including mobile organisms, sea pens and tube building worms

Sub-categories consisted of:

- Substrate (e.g. coral, algae)
- Species
- Sub-families

Coral health for each quadrat and transect was also categorised:

- Pale bleached colouration
- Bleached white
- Diseased coral
- Sediment on coral
- Sediment on other biota

Major coral/substrate categories were then further classified using hierarchical clustering analysis. Location data was averaged out to reduce noise. The following classes were derived:

- Sediment (e.g. rock, sand, pavement)
- Hard corals
- Soft corals
- Mixed substrates
- Bare rock/unknown

Further sub-classifications included:

- Dominant substrates (where the dominant substrate consisted of >90% coral cover per frame, for example dominant sand, dominant hard coral or dominant soft coral)
- Dominant substrate with other sparse substrate (where the dominant substrate consisted of 60-89% substrate cover with up to 40% of other substrates)
- Moderate live substrate (consisting of 40-59% with mixed other substrates)
- Mixed substrate (where equal (41-59%) amounts of substrates, including sediments, were encountered)

Major and minor coral/substrate communities, from repeat seasonal transects, were compared between the 2014 dry season and the 2015 wet season.

2.1.4 Statistical analyses

Euclidian distances were calculated using percent substrate cover from major categories (PRIMER Ver. 6.1; Clarke 1993). Euclidian distances were used as the data was standardised from environmental data based on percent substrate cover. Data were square root transformed to increase importance of rarer species. Classifications were clustered based on their similarity and maximised to produce homogeneity among similar classifications. Cluster analysis results were then cross referenced with actual percent substrate cover categories. Four major classes were then derived for mapping purposes.

These included:

- Dominant sediment (i.e. sparse mix and mixed substrates)
- Dominant sediment (i.e. sparse algae)
- Moderate hard coral
- Mixed other

Permutational Multivariate Analysis of Variance (PERMANOVA) hypothesis testing was used to determine significant dissimilarities in substrate community structure (i.e. multiple species and how they relate to each other based on percent cover) among locations and season (PRIMER Ver. 6.1; Clarke 1993). PERMANOVA was used over other techniques because it is a permutational technique that does not require parametric assumptions, which other

analyses would require, for example Multivariate Analysis of Variance (MANOVA) and Analysis of Variance (ANOVA).

Given the highly similar communities among locations and seasons, Principle Coordinates Ordination (PCO), which is based on the PERMANOVA model, was used over *non-metric* Multi-Dimensional Scaling (*n*MDS), due to the *n*MDS model utilising rank methods as opposed to the actual dissimilarity values themselves. This technique graphically separated significantly dissimilar communities based on location and season. Analysis of Similarity (ANOSIM) was also used to determine significantly dissimilar substrate communities among locations and season.

2.1.5 Mapping

Maps were composed using ESRI® ArcGIS™ Version 10.1, and images sourced from Google Earth. Study area outlines reflecting the reef extents were created from visual interpretation of the satellite imagery and confirmed by ground truthing. Reef boundaries were created such that the sampled reef areas were entirely within the boundary, meaning boundary lines were drawn across sand dominant areas immediately adjacent to the known reef area. In order to compensate for potential mapping inaccuracies, a small arbitrary sand dominant buffer area was allowed for in the mapping process. Data derived from the cluster analyses was then used to construct colour coded polygons around reef structure, emphasising dominant community composition (based on major categories). High resolution maps were then produced for each location.

Broadscale substrate community mapping was undertaken using the spatial analysis program IDRISI®-Selva™ 2012 (Clark Labs, Clark University). Transect outlines and 5m point - interval GPS coordinates were entered into the program. Coral/substrate communities were given a numerical value based on the community composition (refer Table 2.1), corresponding to each GPS data point, and entered into a database and linked to a point vector file containing the GPS coordinates.

Table 2.1 Classification descriptions used for substrate surface interpolation

Class	Classification description
1	Sand/pavement dominant
2	Sand/pavement dominant sparse hard coral
3	Sand/pavement dominant sparse macroalgae
4	Sand/pavement dominant sparse soft coral
5	Sand/pavement dominant mixed other
6	Dominant macroalgae
7	Dominant macroalgae sparse pavement/sediment
8	Dominant macroalgae sparse soft coral
9	Dominant macroalgae mixed other
10	Mixed substrates
11	Sponge dominant sparse hard coral
12	Dominant hard coral
13	Dominant hard coral sparse pavement/sediment
14	Dominant hard coral mixed other
15	Dominant soft coral
16	Dominant soft coral sparse pavement/sediment

Classification numerical values were assigned based on natural ecological succession, for example, sediment and substrate classes were assigned lower numbers, followed by macroalgae, with hard coral and soft coral having the highest class numbers (refer Table 2.1). The module INTERPOL (IDRISI®-Selva™ 2012) was then used to generate surface interpolation of the substrate communities, by interpolating nearest neighbour pixel values based on community composition information from the point data. Although surface interpolation is not as accurate as supervised or un-supervised classification techniques, it was accurate enough to determine broad scale substrate classifications without having to use the latest up to date aerial or satellite imagery. A distance weight exponent of five was used for the surface interpolation model, in order to add more weight to the closest sample points in deriving new attributes. Highest substrate accuracies were attained where pixel values were closest to transect data. High resolution maps were then produced using the program ESRI® ArcGis™ 10.1.

2.1.6 Permanent transects

A number of permanent transects (up to 20m per transect) were established during the 2014 dry season and revisited in 2015, to determine if significant changes in corals were observed over the two sampling events. Transects were established at Sable Chief Rocks, Seal Rocks, Gatcombe Heads, and Jenny Lind Banks. Given the limited amount of corals at Gatcombe Heads, there were not enough coral colonies available for a 20m transect, hence only three coral colonies were established. Transects and colonies were tagged using a subsurface buoy and duplicate coral branches were selected from a colony at every 5m in each transect and/or colony, and marked with cable ties (refer Photograph 2.1). Two species of corals were selected for the permanent transects; *Acropora formosa* and *A. hyacinthus* from the Acroporidae family. The two species were selected because they were abundant within all the locations and have previously demonstrated signs of high growth rates in branching corals (Baird and Hughes 2000, Baird and Marshall 2002, Koop et al. 2001, Rogers 1979). Photographs were taken of every tagged colony using an integrated quadrat and measurement scale bar in June 2014 (refer Photograph 2.1). The same colonies were re-visited after 12 months in June 2015 to determine linear extension growth and colour.

Coral linear extension growth was measured using the program Adobe® Photoshop CS6™ Version 6.1. Coral fragments were measured using the ruler tool and standardised to the integrated scale bar on each photograph. Fragments were measured from the edge of the cable tie to the tip of the fragment. The same process was repeated after 12 months, with coral fragments photographed on the same plane and data compared. A minimum of three colonies from each transect were used for comparative purposes. Colour was also measured using Adobe® Photoshop CS6™ Version 6.1. Each fragment was carefully selected using the Quick Selection Tool and colour information attained from frequency histograms. Red, Green and Blue (RGB) channels were selected as well as Luminosity and compared against the white colour on the scale bar of each photograph.

Luminance or luminosity is a measure of perceived brightness in an image and is described as the amount of light absorbed, emitted, or reflected from a particular area. Luminosity was compared in fragments after 12 months and standardised as a percentage to the scale bar of each photograph. Luminosity was used as the main unit of colour as this reflects the amount of potential bleaching based on the increase or decrease of pigment in coral fragments. Coral fragments higher in pigment concentrations would have lower luminance than corals that are bleached. Results were then tabulated.

Photograph 2.1 Marked colonies using a subsurface buoy (left) and photograph quadrat with integrated quadrat and scale bar (right).
Figure note: Red circles denote cable tie markers

2.2 Reef associated nekton - transects

2.2.1 Sites

The same coastal fringing reefs utilised for substrate community mapping were used to determine reef associated nekton species. Data from Oyster Rocks and Gatcombe Heads were pooled together due to small areas and low numbers.

2.2.2 Sampling approach and methodology

At each location, a minimum of four 50m long by 5m wide transects (250m^2) were established in order to determine reef obligatory fish communities. Each transect was swum within 2-8m depth (mean 4.6m ± 0.6m) following a similar depth contour along continuous reef, in order to reduce depth and structure artefacts. Two observers swam along the transect line towing a surface GPS (Garmin GPS60) unit on track mode. A GPS position was recorded at the start, middle and end of each transect. GPS positions were also recorded when a change in course was required. Additionally, a compass course was recorded at the start of each transect. The two observers swam the course of the transect one behind the other along a tethered 5m long line on self-contained underwater breathing apparatus (SCUBA). The first diver counted and recorded all fish within 5m in front and 2.5m either side of the transect. The second diver took photographs of reef from the previous point intercept and recorded depth. This was to ensure obligatory reef fish would not be spooked by two divers. All non-cryptic target fish were recorded at every 5m x 5m before the two divers moved on to the next 5m intercept. Any fish that were within the 0+ year (young of the year) class were not included as these post settlement new recruits are quite cryptic, potentially resulting in under estimating their densities. If an unknown species was encountered, a detailed description was recorded on waterproof paper and a number of close-up images taken with the camera. The descriptions and photographs were later used to identify the species using fish identification keys and books.

2.2.3 Data and statistical analysis

All fish encountered within each 5m x 5m point intercept of each transects were tallied and recorded as per 250m^2 belt transects. Obligate reef fish biodiversity was measured as:

- Density/250m^2 (i.e. total number of organisms per unit area)
- Species richness (i.e. total number of taxa)
- Diversity (i.e. the proportion of total represented by a species, calculated using the Shannon-Wiener diversity index (Shannon and Weaver 1948)
- Species evenness (i.e. how evenly abundance is spread among the various taxa that make up an assemblage)

Diversity values ranged from 0 (low community complexity) to 4 (high community complexity). Species evenness values were between 0 (a few species make up the majority of the abundance) and 1 (even number of species making up the total abundance) (Cai et al. 2006, Hill 1973, McClatchie et al. 1997, Nero and Sealey 2005, Zar 1996).

Any significant differences ($P < 0.05$, 95% confidence intervals) in fish biodiversity among the four locations and seasons were determined using one and two-way ANOVA. Data were tested for homogeneity of variance and normality. Significance levels were increased ($P < 0.01$, 99% confidence intervals) where data did not meet that criterion (O'Neill 2000, Underwood 1997). In order to determine where significant differences among the four locations were, if any, Fisher's Least Significant Difference (LSD) *Post hoc* tests were used.

Fish community assemblages were also examined at each location by multivariate analyses. ANOSIM was used to statistically determine dissimilarities in community structure among locations and season (PRIMER Ver. 6.1; Clarke 1993). Fish community structure was examined using Bray-Curtis similarity measures (Clarke 1993). Bray-Curtis was chosen as the preferred similarity matrix because it performs well in preserving 'ecological distance' in a variety of simulations on different types of data sets (Clarke 1993). Data were dispersion weight corrected to maintain equal weight among common and rare species.

Coral community data was used to determine if differences in reef associated nekton were related to different types of substrate habitat. The model BEST (Biota and/or Environmental Matching) was used to correlate matrix assemblages of major coral classes with fish assemblages using Primer (PRIMER Ver. 6.1; Clarke 1993). DistLM (Distance based Linear Models) was also used to determine the amount of variation explained between the two models and what factors contributed to the correlation.

2.3 Reef associated nekton – baited remote underwater video stations

2.3.1 Sites

Baited Remote Underwater Video Stations (BRUVS) were deployed at four locations within the Project potential impact area. Locations included Seal Rocks, Jenny Lind Banks, Gatcombe Heads and Sable Chief Rocks. BRUVS were deployed at depths ranging from 3 m to 5 m, when conditions and visibility were favourable. BRUVS were placed where possible, next to reef structure visible from the boat and GPS locations were recorded for each deployment. Soak time of each BRUVS deployment ranged from 1 to 1.5 hours, as per previous research outcomes demonstrating that minimum deployment times of between 1 and 2 hours was required to sample between 66-83% of the fish species present (Harasti et al. 2015, Harvey et al. 2012, Unsworth et al. 2014). All observations were then counted as a

1 hour deployment period for all locations, from 10 minutes after the BRUVS were deployed to avoid initial deployment confounding issues.

2.3.2 Sampling approach and methodology

Two calibrated stereo underwater camera units (SeaGIS Pty. Ltd., Victoria, Australia) containing a pair of Canon Legria HF G25 high definition cameras mounted into a custom made frame were used to capture imagery. A bait bag containing Western Australian pilchards (*Sardinops sagax*), was attached to the extension bar and used to attract the fish towards the cameras. Video imagery was recorded by the two cameras and converted to digital format using Xilisoft Video Converter Ultimate (Xilisoft) for use in photogrammetric calibration and measurement software. Calibration of the stereo-video camera system was done at the end of the sampling campaign using a calibration cube (0.5m × 0.5m × 0.3m).

2.3.3 Data analysis

Once the camera profiles were obtained using the CAL software, which is software used to calibrate the stereoscopic vision of the BRUVS, imagery was then analysed using EventMeasure software. Numbers and types of species were recorded during each frame. Only fish that were in close proximity to the BRUVS, with the ability to identify to species level, were included in the analyses. Data were used to calculate total/cumulative and relative numbers of fish per frame/soak time. Qualitative/semi-quantitative data were recorded and tabulated.

2.4 Estuarine and coastal nekton

2.4.1 Sites

Estuaries, creeks and rivers act as highways for fish and other nekton that commute from one system to another, especially along coastal beaches and fringing rocky reefs during tidal movements, concentrating fish numbers in search for food and shelter (Jones and West 2005, Meynecke et al. 2008a, Meynecke et al. 2008b, Neira and Potter 1992, Rolls 2011, Sousa and Dangremond 2011). Given that the proposed Project dredging activities are to occur in relatively open water, adjacent to a number of beaches and fringing rocky reefs, it can be difficult to discern fish communities along these open areas. In order to determine an accurate estimate of fish communities along these habitats, sampling efforts were concentrated around the mouths of creeks, rivers, inlets and islets (refer Figure 2.2). Using two netting techniques (i.e. cast netting and gill netting), the following areas were targeted:

- South west (SW) Facing Island (including Gatcombe Heads)
- Sand Islet
- Wild Cattle Creek
- Boyne River
- Lilleys Inlet
- South Trees Inlet
- Fisherman's Landing
- The Narrows/Targinnie Creek
- Boat Creek South

2.4.2 Sampling approach and methodology

Cast nets (3m diameter x 2m drop x 6mm mesh size) and gill nets (40m x 1.5m x 75mm and 100mm panel mesh sizes), were used as capture devices to determine fish assemblages from different habitats within each location (refer Photograph 2.2).

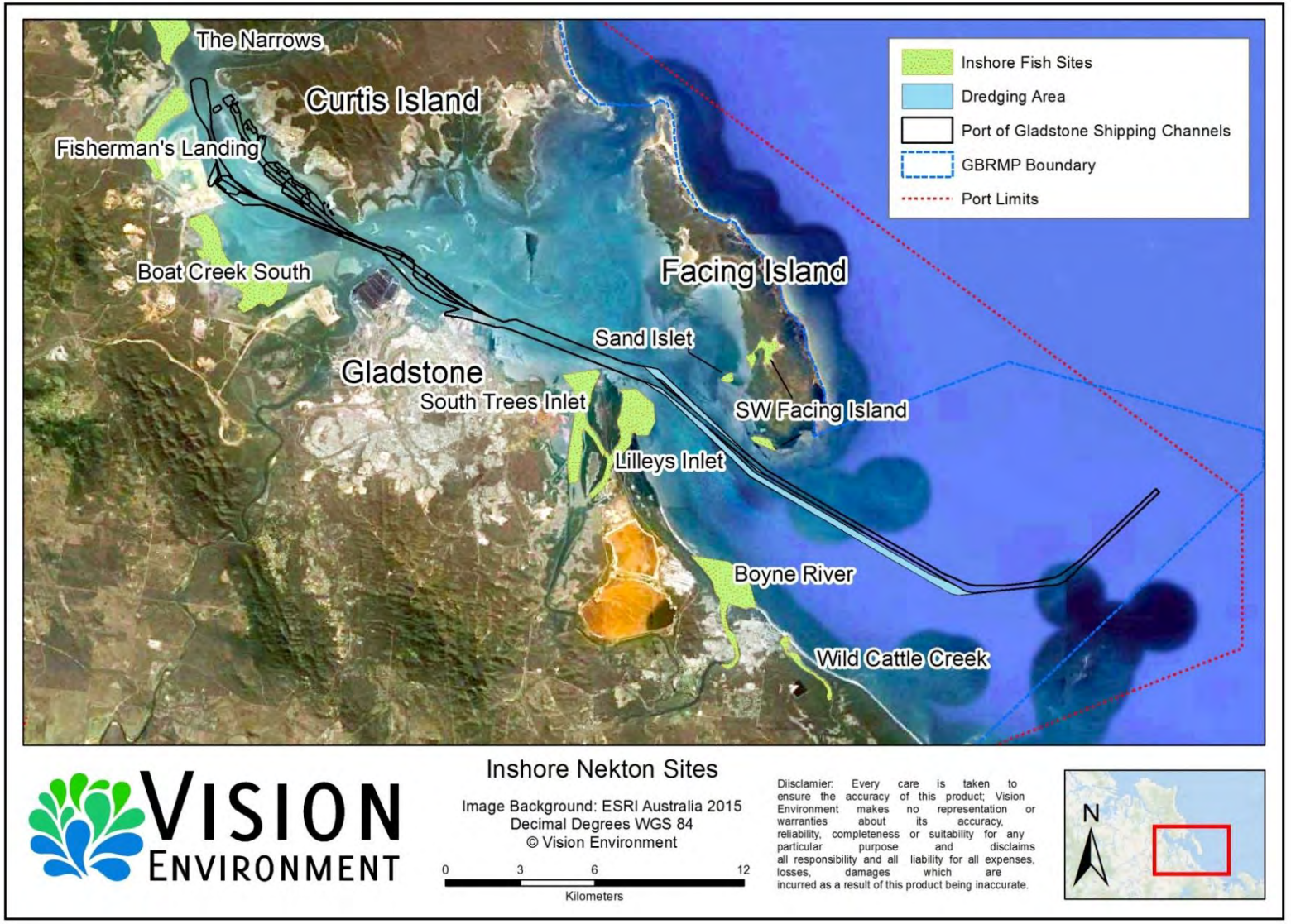


Figure 2.2 Site localities of inshore nekton netting sites

Photograph 2.2 Example of a gill net along a deep channel within South Trees Inlet

A minimum of ten replicate (*n* = 10) casts by an experienced operator (Coefficient of Variation (CV) of cast spread < 3%), were undertaken to sample juvenile and small fish fauna, covering a diversity of small scale habitats that are usually difficult to survey using other netting techniques (e.g. small soft sediment shallow drainage channels, mangrove line feeder channels and other vegetation lined banks). Replicates were randomly spread among sampling locations (up to 1-3km). Gill nets (minimum of one replicate; *n* = 1, soak time of 1-2 hours) were used at each site to sample larger mobile species that may have been underrepresented using the cast net. All fish collected were identified to species level, enumerated and measured before being released. If identity was uncertain, a voucher specimen was collected using ethically humane techniques and sent to the laboratory for further analysis using identification keys and books.

2.4.3 Data and statistical analysis

Fish biodiversity by cast net sampling, was measured as density per $3m^2$, species richness, diversity and species evenness. One and two-way ANOVA was used to determine significant differences ($P < 0.05$, 95% confidence intervals) in fish biodiversity among locations and seasons by cast net sampling. Data were tested for homogeneity of variance and normality. Significance levels were increased ($P < 0.01$, 99% confidence intervals) where data did not meet that criterion (O'Neill 2000, Underwood 1997). In order to determine where significant differences among locations, Fisher's LSD *Post hoc* tests were used.

Fish community assemblages were also examined at each location and season by multivariate analyses and plotted using n-MDS. ANOSIM was used to statistically determine dissimilarities in community structure among locations and season (PRIMER Ver. 6.1; Clarke

1993). Fish community structure was examined using Bray-Curtis similarity measures (Clarke 1993). Data were dispersion weight corrected to maintain equal weight among common and rare species. Given the amount of time required to soak gill nets, number of locations and large tidal runs, gill net data was used for qualitative and semi-quantitative purposes only. Data is shown in tables in Section 3.2 and Section 3.3 for each location.

3 RESULTS AND DISCUSSION

3.1 Coral reef communities

3.1.1 Substrates

Coral reefs are an important part of the marine ecosystem and to human populations. Coastal reefs are often located near estuaries or mangroves, making them important nursery habitats to many juvenile fish species (Birkeland 1997, Blaber et al. 2005, Harding et al. 2006, Patterson and Swearer 2007, Rönnbäck 1999, Schaffelke et al. 2005). However, coastal fringing coral reefs are more susceptible to anthropogenic influences than offshore reefs, specifically because of their proximity to coastal development. Hence, an understanding of reef habitat structure and condition is important in the management of future potential natural and anthropogenic disturbances.

A total area of 424.3ha was classified as potential reef area (based on visual map inspections) within the Project potential impact area. The largest area of reef was encountered at Sable Chief Rocks (~208ha or 49% of the total area observed) and the smallest area of reef (< 2ha or <1%) at Oyster Rocks (refer Table 3.1).

Table 3.1 Total reef area outlines for each location included in the mapping

Location	Area (ha)
Oyster Rocks	1.8
Gatcombe Heads	18.1
Jenny Lind Banks	66.7
Seal Rocks	130.1
Sable Chief Rocks	207.6
Total area	**424.3**

Ten major substrate classes were identified among all locations with four classes pooled (turfing algae, seagrass, other biota and unknowns), due to low cover. The highest average cover (with plus or minus standard error (±se)), was sediments (i.e. sand, pavement and rock: 55 ± 4.6%), followed by macroalgae (29 ± 3.5%), and hard corals (13 ± 6.4%), with the lowest major classes (< 2%) consisting of soft corals, sponges and ascidians, and other biota (refer Table 3.2).

Pavement cover (i.e. rock, rubble and/or sediment) percentages identified in the Project monitoring program were similar to results from a number of studies conducted to the south of the Project potential impact area, encompassing reefs from Elliott Heads to Rodds Bay (Alquezar and Boyd 2007, Alquezar et al. 2007, Alquezar et al. 2011). The studies found similar pavement cover (40-60%), however, dominant live substrate cover included soft corals (30-40%) and macroalgae, which was in contrast found in very small amounts at most study reefs for the Project monitoring program. The Project monitoring program identified macroalgae as the dominant live form cover (~29%), followed by hard coral cover (13%).

This suggests that coastal fringing reefs within Port Curtis are quite different to coastal reefs further south along the coast.

Table 3.2 Mean percent dominant substrate cover among locations

Major substrate classes	Jenny Lind Banks	Oyster Rocks	Gatcombe Heads	Sable Chief Rocks	Seal Rocks	All sites
Sediment (%)	56.4 ± 6.0	53.4 ± 11.0	61.2 ± 4.0	37.8 ± 4.6	67.6 ± 5.5	55.3 ± 4.6
Macroalgae (%)	32.6 ± 5.8	40.0 ± 12.2	29.9 ± 4.5	19.3 ± 2.8	20.8 ± 3.7	28.5 ± 3.5
Hard coral (%)	5.9 ± 1.2	4.1 ± 2.7	4.5 ± 1.2	41.0 ± 6.4	9.7 ± 2.7	13.1 ± 6.4
Soft coral (%)	4.4 ± 1.5	1.0 ± 0.4	0.7 ± 0.2	0.9 ± 0.3	1.5 ± 0.5	1.7 ± 0.6
Sponge and ascidians (%)	0.3 ± 0.1	1.0 ± 0.7	2.2 ± 0.7	0.6 ± 0.3	0.2 ± 0.1	0.9 ± 0.3
Other biota (%)	0.9 ± 0.5	0.5 ± 0.5	1.9 ± 1.4	1.6 ± 0.5	0.8 ± 0.4	1.2 ± 0.2

Results of the broader (> 90,000ha) benthic habitat characterisation of Port Curtis and Rodds Bay undertaken by TropWater, found that the dominant habitat feature was open substrate with a low to medium cover of benthic habitat, benthic fauna and algae (McKenna et al. 2014). A review by DHI (2013) found that reefs along Facing Island had up to 30% hard coral cover. Within the same reef areas, macroalgae cover was between 19-45%. Although the focused study areas were not identical, the results were consistent and within similar ranges found in the Project monitoring program.

Significant ($P < 0.05$) differences in substrate types were observed among study locations (refer Figure 3.1), which included sediments, soft corals, hard corals and sponges. The highest noticeable difference was hard coral, with over five times more hard coral observed at Sable Chief Rocks compared to all other locations (refer Photograph 3.1). There were no significant ($P > 0.05$) differences in substrate cover among locations for macroalgae or other biota.

The results were similar to those found in previous studies of fringing reefs in Port Curtis by Ayling et al. (2012) and Oceania (2011), where Sable Chief Rocks was significantly different to other surrounding study sites along Gatcombe Heads and Facing Island. The authors found that the highest hard coral cover was at Sable Chief Rocks representing 31% cover, with the branching hard coral Acroporidae being the dominant family (~20%). Macroalgae ranged from 15-45% and soft coral was at 2-3% cover (Ayling et al. 2012). These results were similar to the Project monitoring program, where hard coral at Sable Chief Rocks was found at ~40% cover, with *Acropora formosa* being the dominant hard coral species (refer Table 3.2 and 3.3). Macroalgae was at ~20% and soft corals at < 1% (refer Table 3.2) in this monitoring program.

Major substrate cover was significantly dissimilar (PERMANOVA; P (Perm) < 0.001; *df* 4; MS 26.3) among locations (refer Figure 3.2). Sable Chief Rocks had the greatest difference in substrate community composition compared to other locations. The major substrates driving the variation included hard corals, sediments and macroalgae. The findings were similar to those of Ayling et al. (2012) and BMT WBM (2013).

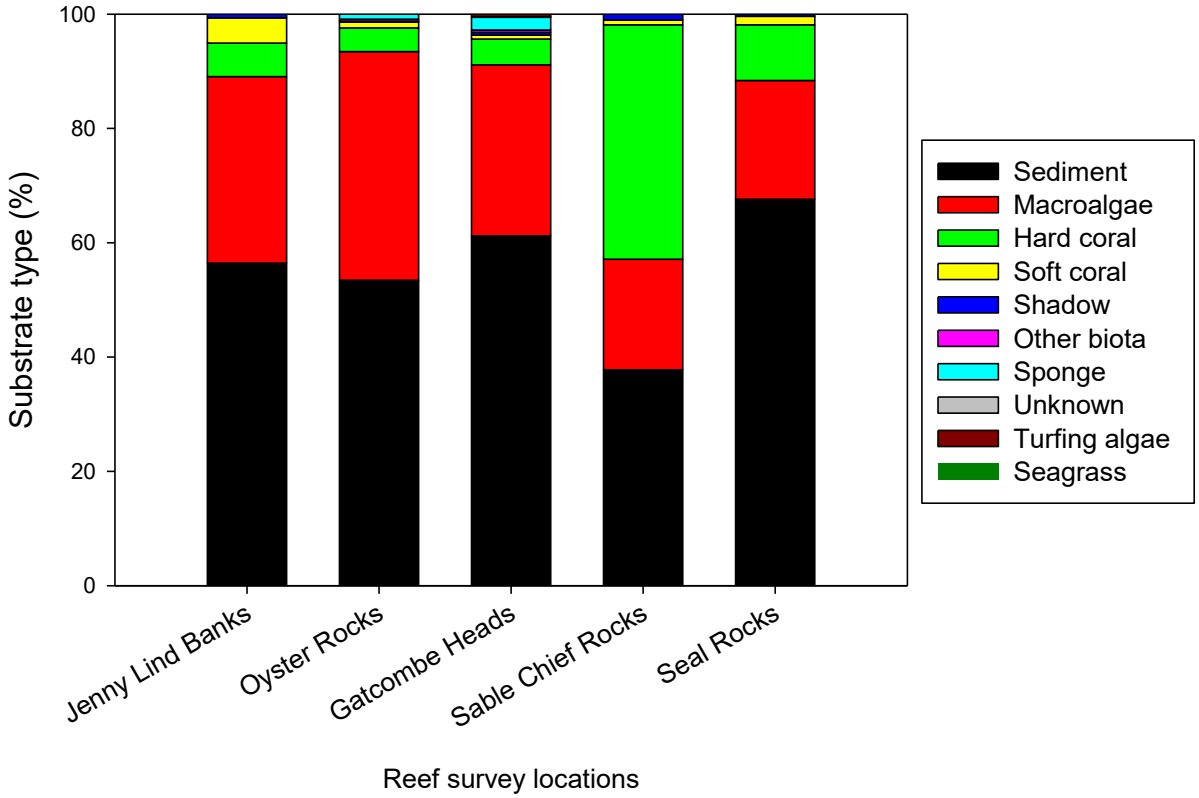


Figure 3.1 **Mean (±se) percent major substrate cover among locations**

Photograph 3.1 **Photograph of extensive hard coral cover at Sable Chief Rocks**

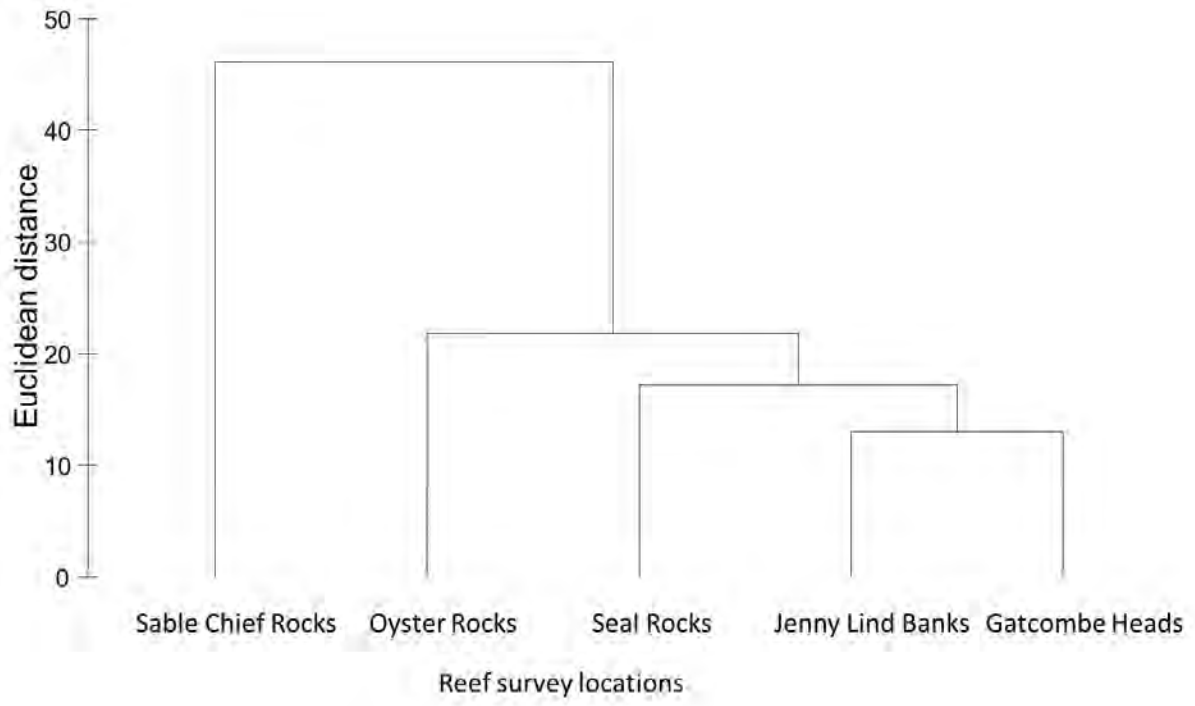


Figure 3.2 Hierarchical cluster analysis using grouped averages of major substrate categories among locations

Substrate sub-categories were classified into 48 classes among all locations (refer Table 3.3). Sand and pavement substrates were the highest non-live classes from all locations (35% and 19%, respectively). Highest living substrate consisted of the macrolagae, *Sargassum* sp. (21%, refer Photograph 3.2), followed by the hard branching coral (*Acropora formosa)* (7%, refer Photograph 3.3), the brown algae genera (*Lobophora* sp.) (2.4%), and the red algae genera (*Asparagopsis* sp.) (1.9%, Photograph 3.4). Similar findings were reported by Ayling et al., (2012) indicating there has been little change in percent cover of substrates from 2012 to 2015.

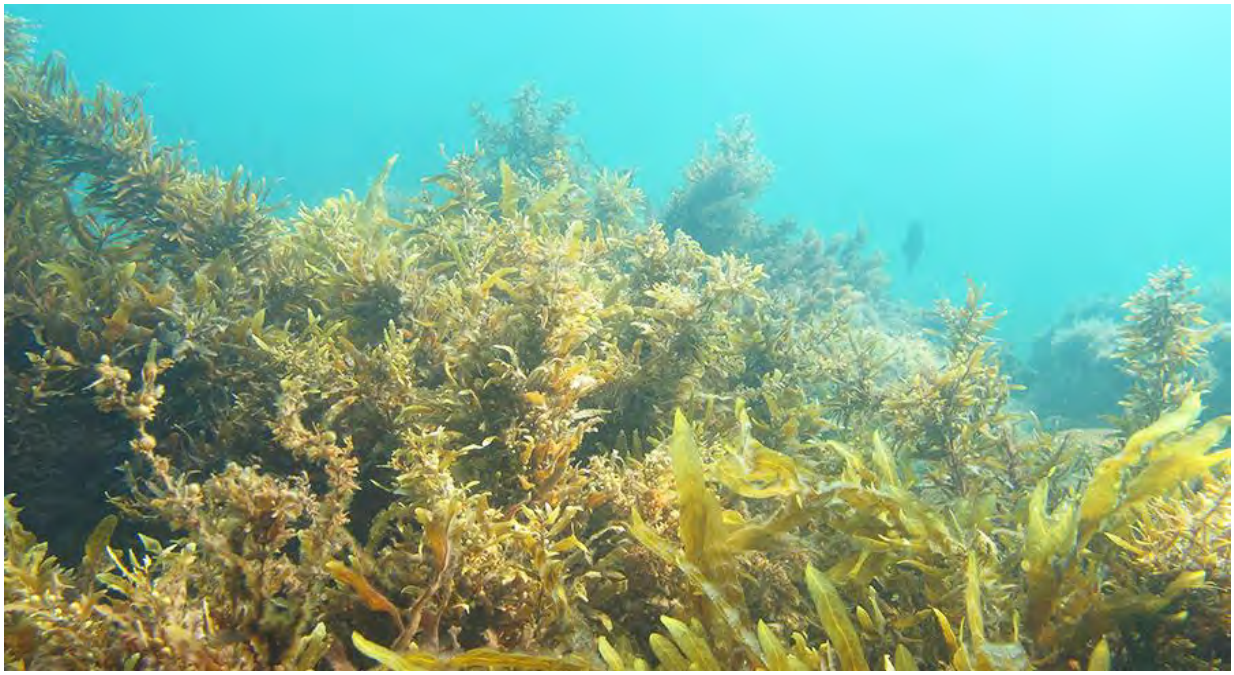
Table 3.3 Mean (±se) percentage of minor substrate class cover among locations

Minor substrate classes (%)	Jenny Lind Banks	Oyster Rocks	Gatcombe Heads	Sable Chief Rocks	Seal Rocks	All sites
Sand (%)	34 ± 8	35 ± 8	43 ± 5	14 ± 3	51 ± 6	35 ± 6
Sargasum sp. (%)	20 ± 4	38 ± 12	23 ± 5	9 ± 2	14 ± 3	21 ± 5
Pavement (%)	22 ± 3	18 ± 5	18 ± 2	22 ± 3	17 ± 2	19 ± 1
Acropora formosa (%)	<0.1	0.0 ± 0.0	0.1 ± 0.1	32 ± 7	0.2 ± 0.2	6.5 ± 6.4
Lobophora sp. (%)	0.9 ± 0.4	0.2 ± 0.2	2.6 ± 0.7	4.1 ± 0.7	4.1 ± 0.9	2.4 ± 0.8
Asparagopsis (%)	6.9 ± 2.1	0.7 ± 0.4	0.3 ± 0.3	1.0 ± 0.4	0.8 ± 0.5	1.9 ± 1.3
Dendrophyllidae turbinaria (%)	1.4 ± 0.7	0.1 ± 0.1	0.7 ± 0.3	0.7 ± 0.2	5.9 ± 2.3	1.8 ± 1.1
Acropora hyacinthus (%)	0.3 ± 0.3	<0.1	1.3 ± 0.8	3.5 ± 1.0	1.3 ± 0.5	1.3 ± 0.6
Padina sp. (%)	2.9 ± 0.9	0.6 ± 0.5	0.2 ± 0.1	0.7 ± 0.3	1.2 ± 0.3	1.1 ± 0.5
Sponge/Ascidian (%)	0.3 ± 0.1	1.0 ± 0.7	2.2 ± 0.7	0.6 ± 0.3	0.2 ± 0.1	0.9 ± 0.4
Poritidae goniopora (%)	0.7 ± 0.3	1.5 ± 1.5	0.9 ± 0.4	0.3 ± 0.2	0.1 ± 0.1	0.7 ± 0.2
Caulerpa racemose (%)	0.3 ± 0.2	0.1 ± 0.1	<0.1	2.7 ± 0.9	<0.1	0.6 ± 0.5
Sinularia sp. (%)	1.7 ± 0.5	0.2 ± 0.2	0.1 ± 0.1	0.3 ± 0.2	0.6 ± 0.2	0.6 ± 0.3
Porities sp. (%)	1.3 ± 0.8	0.6 ± 0.6	0.2 ± 0.2	0.7 ± 0.4	<0.1	0.6 ± 0.2

Minor substrate classes (%)	Jenny Lind Banks	Oyster Rocks	Gatcombe Heads	Sable Chief Rocks	Seal Rocks	All sites
Lobophytum sp. (%)	1.6 ± 0.9	<0.1	0.4 ± 0.2	0.2 ± 0.1	0.6 ± 0.3	0.6 ± 0.3
Coralline algae (%)	0.9 ± 0.3	<0.1	0.9 ± 0.2	0.5 ± 0.1	0.2 ± 0.1	0.5 ± 0.2
Pocillopora damicornis (%)	0.1 ± 0.1	1.1 ± 1.1	<0.1	0.9 ± 0.4	0.2 ± 0.2	0.5 ± 0.2
Faviidae favities (%)	0.6 ± 0.3	0.4 ± 0.2	0.2 ± 0.1	0.2 ± 0.1	0.7 ± 0.5	0.4 ± 0.1
Faviidae goniastrea (%)	0.6 ± 0.5	0.1 ± 0.1	0.5 ± 0.4	0.2 ± 0.1	0.5 ± 0.3	0.4 ± 0.1
Acropora sp. (%)	0.1 ± 0.0	0.1 ± 0.1	<0.1	1.4 ± 0.4	0.3 ± 0.2	0.4 ± 0.3
Halymenia sp. (%)	0.0 ± 0.0	0.2 ± 0.2	1.3 ± 0.6	0.3 ± 0.1	<0.1	0.4 ± 0.2
Sarcophyton sp. (%)	0.7 ± 0.4	0.5 ± 0.3	0.1 ± 0.1	0.1 ± 0.1	0.3 ± 0.1	0.3 ± 0.1
Tape, wand, shadow* (%)	0.2 ± 0.2	0.1 ± 0.1	0.2 ± 0.1	0.7 ± 0.2	0.3 ± 0.1	0.3 ± 0.1
Dead coral with algae (%)	0.1 ± 0.1	<0.1	<0.1	1.3 ± 0.3	0.1 ± 0.1	0.3 ± 0.3
Faviidae favia (%)	0.2 ± 0.1	0.2 ± 0.1	0.6 ± 0.4	0.2 ± 0.1	0.3 ± 0.2	0.3 ± 0.1
Poor quality photograph* (%)	0.3 ± 0.2	<0.1	0.2 ± 0.1	0.5 ± 0.2	0.4 ± 0.2	0.3 ± 0.1
Rubble (%)	0.8 ± 0.4	0.1 ± 0.1	0.1 ± 0.1	0.3 ± 0.1	0.2 ± 0.1	0.3 ± 0.1
Galaxaura sp. (%)	0.3 ± 0.2	<0.1	0.6 ± 0.4	0.4 ± 0.1	<0.1	0.3 ± 0.1
Dictyota sp. (%)	0.1 ± 0.1	0.3 ± 0.2	0.4 ± 0.3	0.2 ± 0.1	0.1 ± 0.1	0.2 ± 0.1
Turfing algae (%)	<0.1	0.0 ± 0.0	0.9 ± 0.9	0.1 ± 0.1	<0.1	0.2 ± 0.2
Xenia sp. (%)	0.1 ± 0.1	0.3 ± 0.2	<0.1	0.2 ± 0.1	<0.1	0.1 ± 0.1
Montipora capricornis (%)	0.1 ± 0.1	0.0 ± 0.0	<0.1	0.4 ± 0.2	0.1 ± 0.1	0.1 ± 0.1
Hydroid (%)	0.3 ± 0.1	<0.1	0.2 ± 0.1	<0.1	<0.1	0.1 ± 0.1
Mobile organism (%)	<0.1	0.2 ± 0.2	0.2 ± 0.1	<0.1	<0.1	0.1 ± 0.1
Unknown biota (%)	0.1 ± 0.1	<0.1	0.1 ± 0.1	0.2 ± 0.1	0.1 ± 0.1	0.1 ± 0.0
Other green (%)	0.1 ± 0.1	<0.1	0.2 ± 0.1	0.0 ± 0.0	0.1 ± 0.1	0.1 ± 0.0
Dictyopteris sp. (%)	0.1 ± 0.1	<0.1	<0.1	0.2 ± 0.1	<0.1	0.1 ± 0.0
Poritidae – Alveopora sp. (%)	0.3 ± 0.2	<0.1	<0.1	<0.1	<0.1	0.1 ± 0.1
Anacropora forbesi (%)	<0.1	<0.1	<0.1	0.1 ± 0.1	0.2 ± 0.1	0.1 ± 0.0
Nephtheidae sp. (%)	0.3 ± 0.1	<0.1	<0.1	0.0 ± 0.0	<0.1	0.1 ± 0.1
Acropora digitifera (%)	0.1 ± 0.1	<0.1	<0.1	0.1 ± 0.1	<0.1	<0.1
Halimeda sp. (%)	<0.1	<0.1	0.1 ± 0.1	0.1 ± 0.0	<0.1	<0.1
Bryozoa (%)	<0.1	0.1 ± 0.1	<0.1	<0.1	<0.1	<0.1
Caluerpa taxifolia (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Halophila sp. (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Gorgonian (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Pocillopora verrucosa (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Codium sp. (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1

Table notes: *Denotes photograph interference. Substrate listed in order of highest to lowest cover

The highest cover of the hard coral species, *Acropora formosa,* was observed at Sable Chief Rocks (> 30%), while Oyster Rocks and Gatcombe Heads exhibited the highest algae cover (*Sargassum* sp.) (38% and 23%, respectively). Lower proportions of the hard coral (*Dendrophyllidae turbinaria*) (1.8%) and the wide flat plate coral (*Acropora hyacinthus*) (1.3%), were observed at most sites, with highest cover observed at Sable Chief (> 3%). Most other classes combined made up less than 1% cover over the whole area.



Photograph 3.2 **Example of the brown algae (*Sargassum* sp.), abundant at Gatcombe Heads (photograph), Oyster Rocks and Seal Rocks**

Photograph 3.3 **Branching coral (*Acropora formosa)* abundant at Sable Chief Rocks**

Similar to the major categories, small but significant ($P < 0.001$) dissimilarities in minor substrate classes were observed (PERMANOVA; P (Perm) < 0.001; *df* 4; MS 111.2) among locations. Sable Chief Rocks displayed the least similar substrate composition to all other locations (refer Figure 3.3), with hard corals driving the main difference between locations at Sable Chief Rocks (PCO1; x-axis), including the branching corals (i.e. *Acropora hyacinthus*, *Pocillopora damicornis*, *Acropora digitifera*, *Acropora formosa*, *Pocillopora verrucosa* and *Anacropora forbesi*). Turfing algae and sediment type also contributed to some differences among locations (PCO2; y-axis).

Photograph 3.4 **Red algae (*Asparagopsis* sp.) abundant at Jenny Lind Banks**

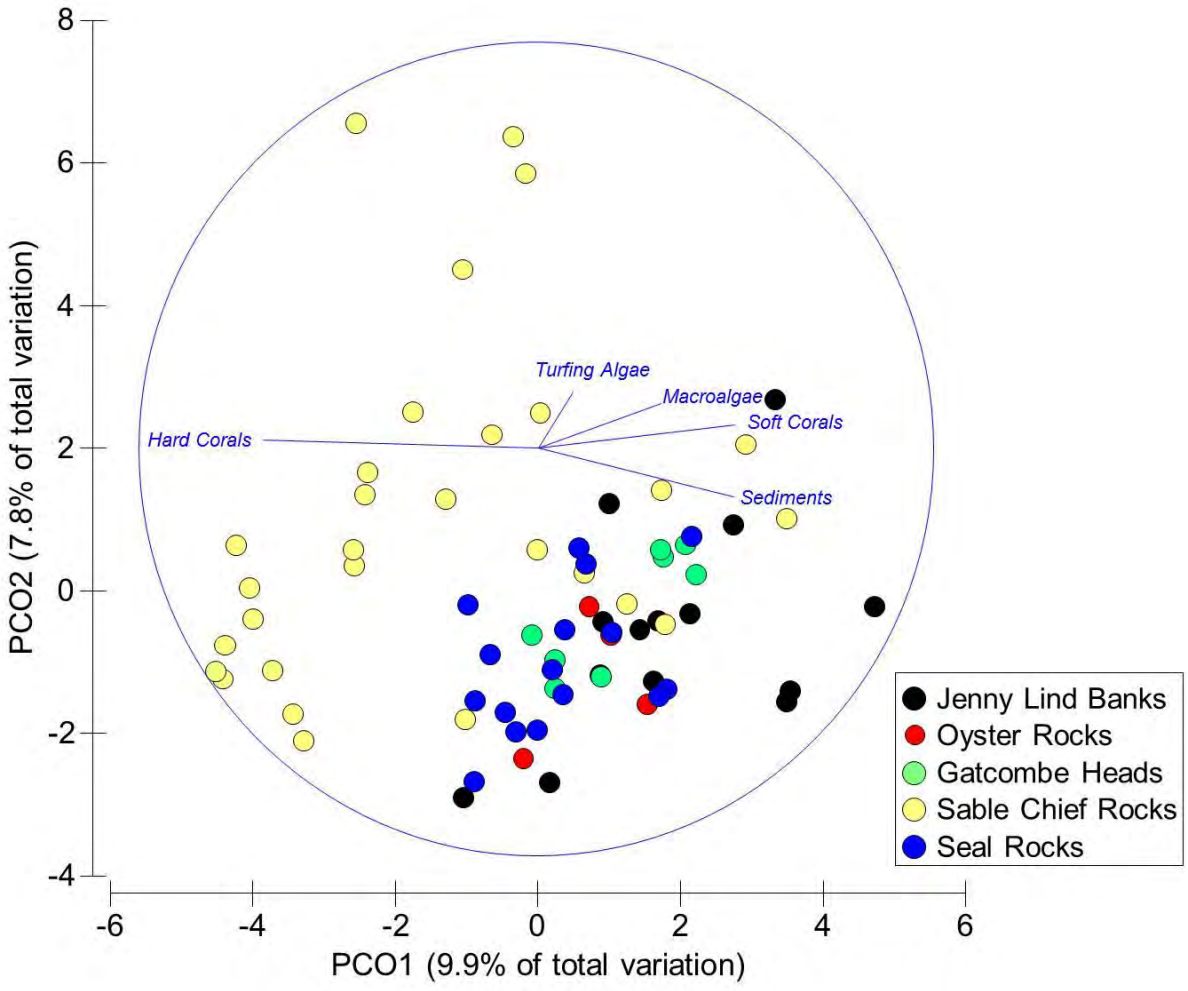


Figure 3.3 **Principle Coordinates Ordination of minor substrate categories among locations**

Figure note: Major categories were overlaid to determine type of structure driving the variation among locations. Data were normalised and square root transformed using Euclidian distances.

Coral bleaching is facilitated by an increase in ocean temperatures and an increase of 1°C can have a significant impact on the frequency of coral bleaching events (Baker et al. 2008, Berkelmans et al. 2004, Berkelmans and Oliver 1999, Glynn 1993). A number of freshwater bleaching events have also occurred at neighbouring northern inshore reefs of the Great Barrier Reef in the last five years (Thompson et al. 2011, Thompson and Dolman 2010, Thompson et al. 2010). Very little coral surveyed as part of the Project monitoring program demonstrated signs of stress, with less than 0.5% of corals showing pale colouration and less than 0.1% presenting bleached white colouration (refer Figure 3.4). The highest amounts of sediments on corals were observed at Seal Rocks, with approximately 0.5% of transects affected, followed by Sable Chief Rocks. The highest amounts of sediment on other biota was observed at Gatcombe Heads and Oyster Rocks (17% and 12%, respectively). There were no signs of diseased corals at any of the locations.

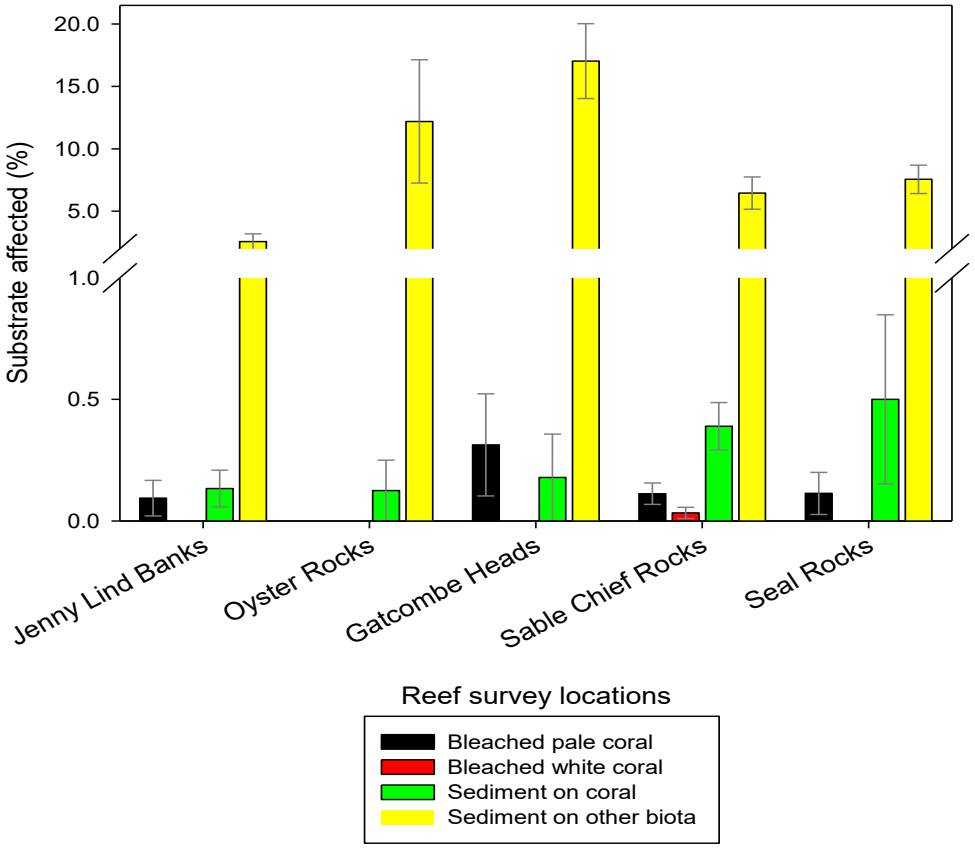


Figure 3.4 Percent substrate affected per site, by stress (bleaching), disease or sediments

3.1.2 Mapping

Reef extent areas were mapped within the Project potential impact area (refer Figure 3.5). Jenny Lind Banks and Oyster Rocks consisted mainly of mixed substrates with approximately 53-56% sediments, 32-40% algae, and 5-10% of hard and soft corals (refer Table 3.2, Figures 3.6 and 3.7). Gatcombe Heads was classified as dominant sediments with sparse algae and comprised approximately 60% sediments, 30% algae and lower amounts (< 10%) of other classes (refer Table 3.2, Figure 3.7). Seal Rocks was classified as dominant sediment with sparse other (refer Figure 3.9), consisting mainly of approximately 68% sediments, 21% algae, 10% hard coral, and < 2% other classes. Sable Chief Rocks exhibited a slightly different classification consisting of moderate hard coral, mixed other

classes, with 41% hard coral, 38% sediments, 19% algae and < 2% other classes (refer Table 3.2, Figure 3.9).

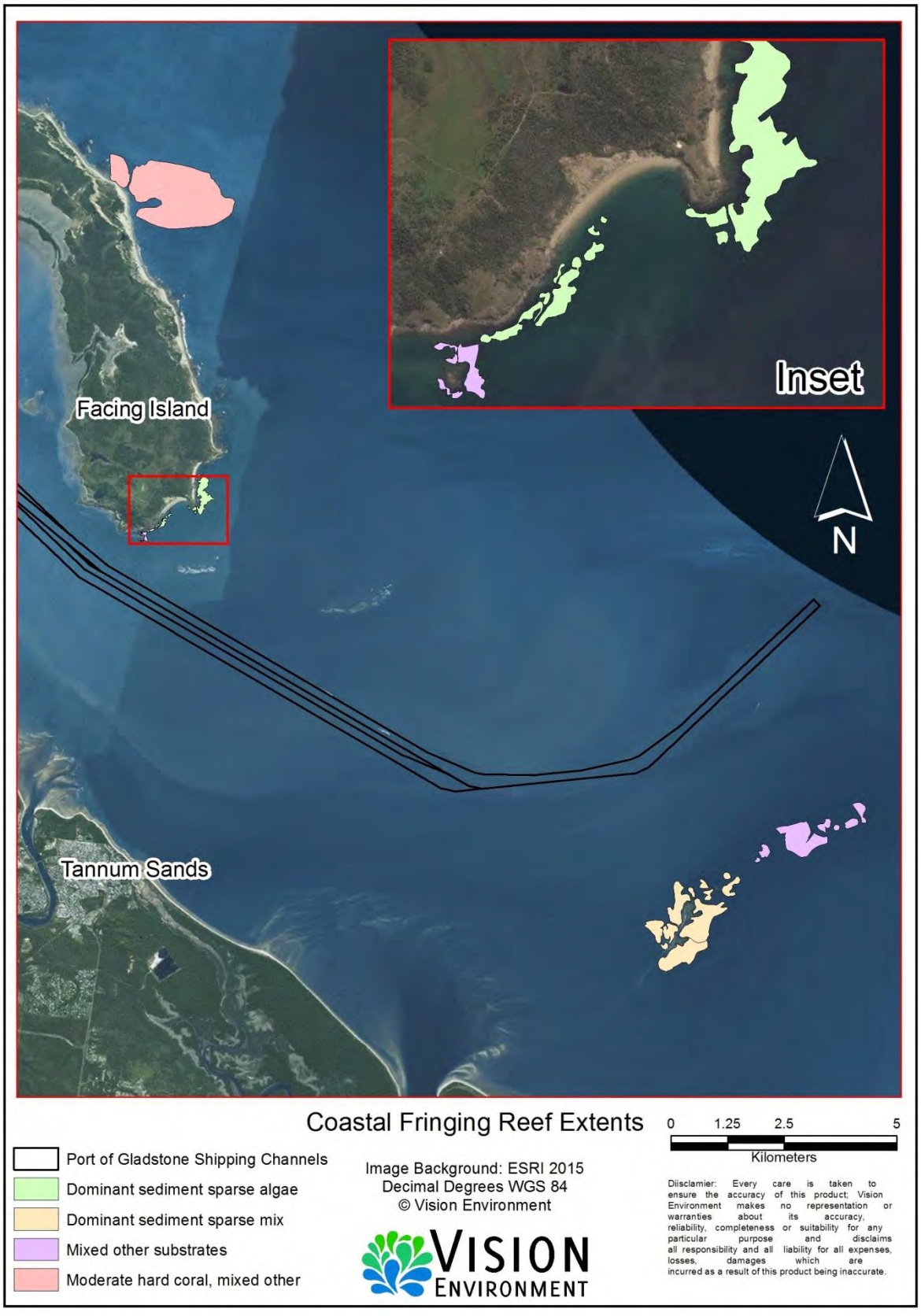


Figure 3.5 Overview map of reef extent areas

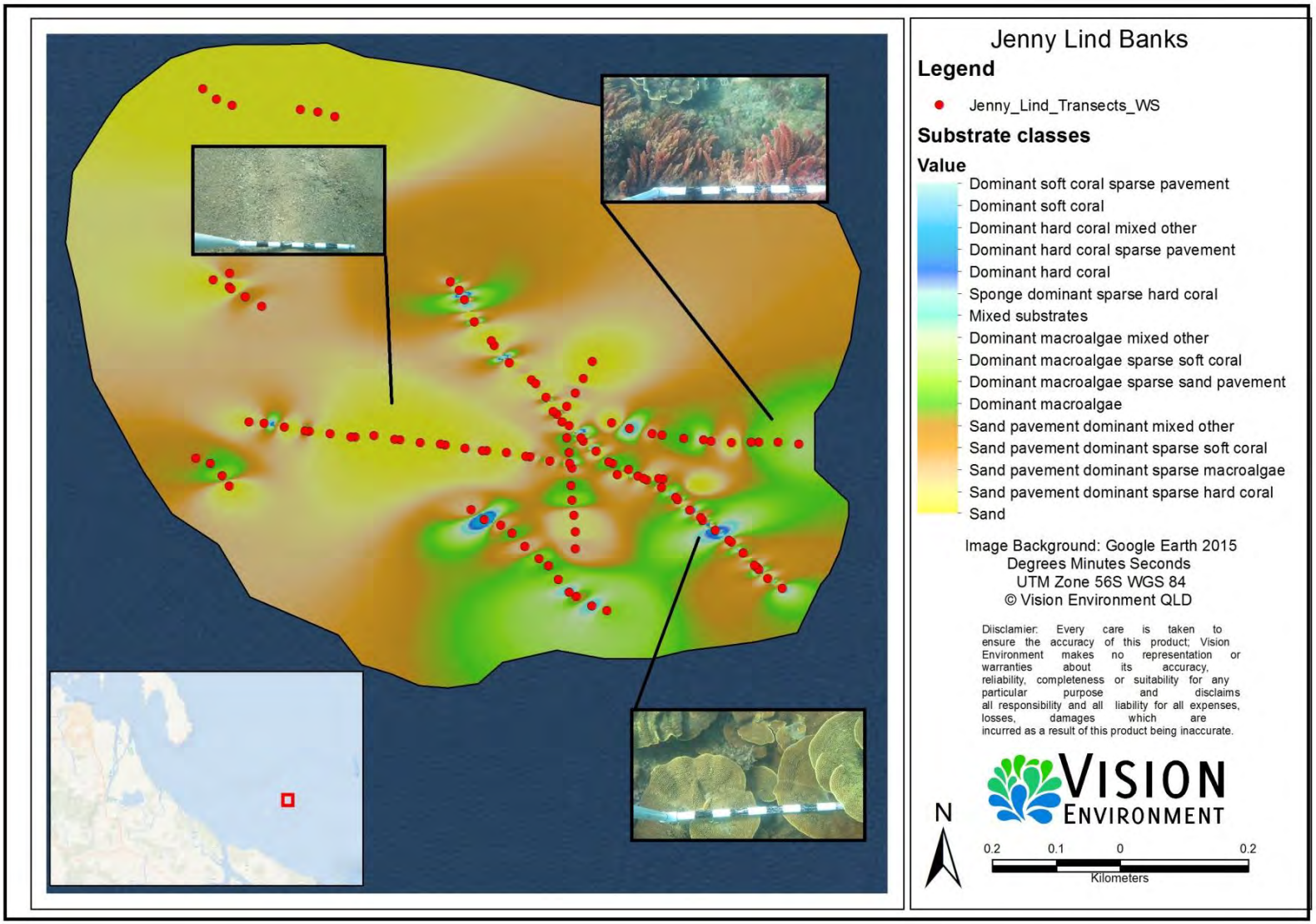


Figure 3.6 Reef area locality and overlaid surface substrate interpolation for Jenny Lind Banks

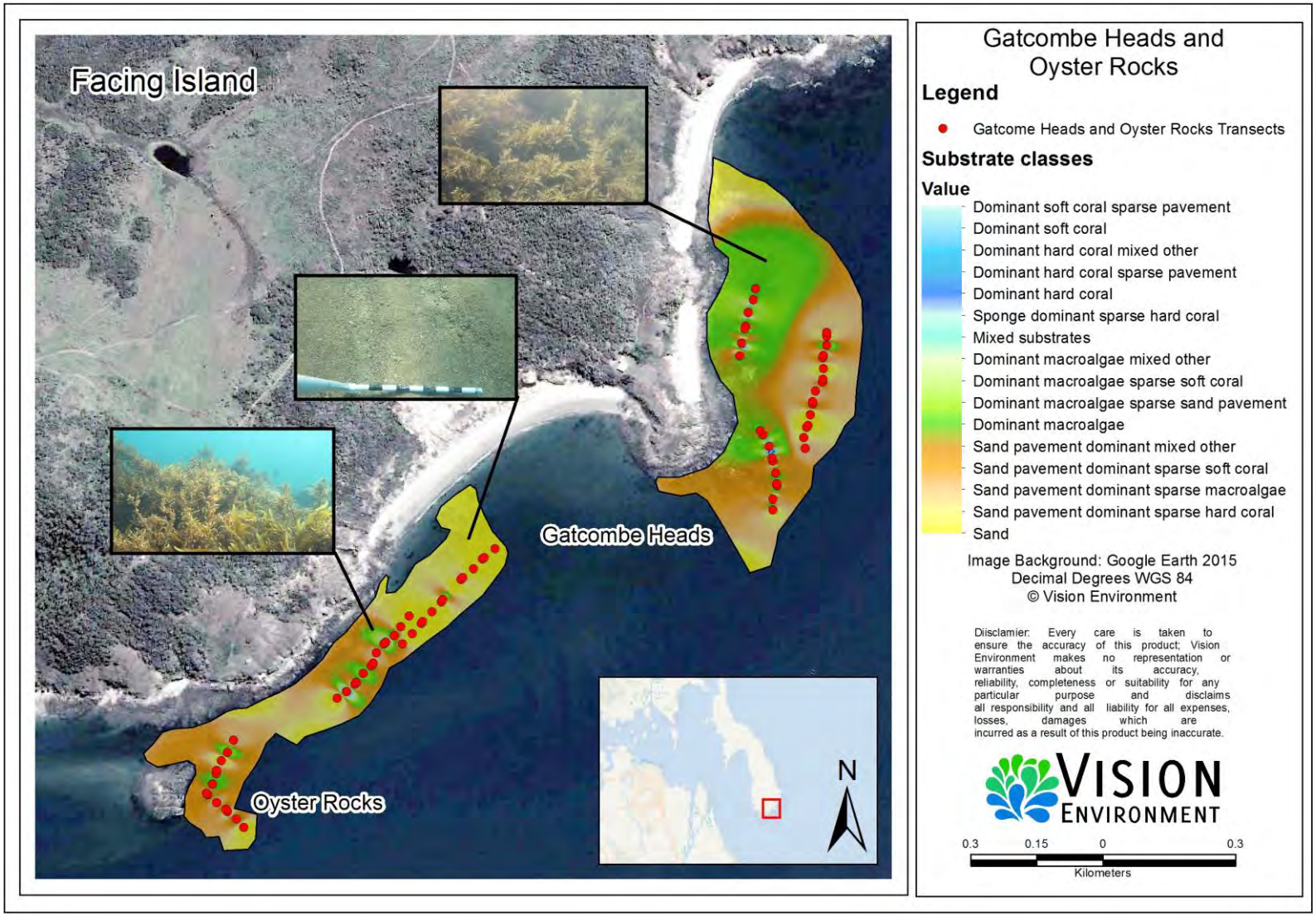


Figure 3.7 Reef area locality and overlaid surface substrate interpolation for Gatcombe Heads and Oyster Rocks

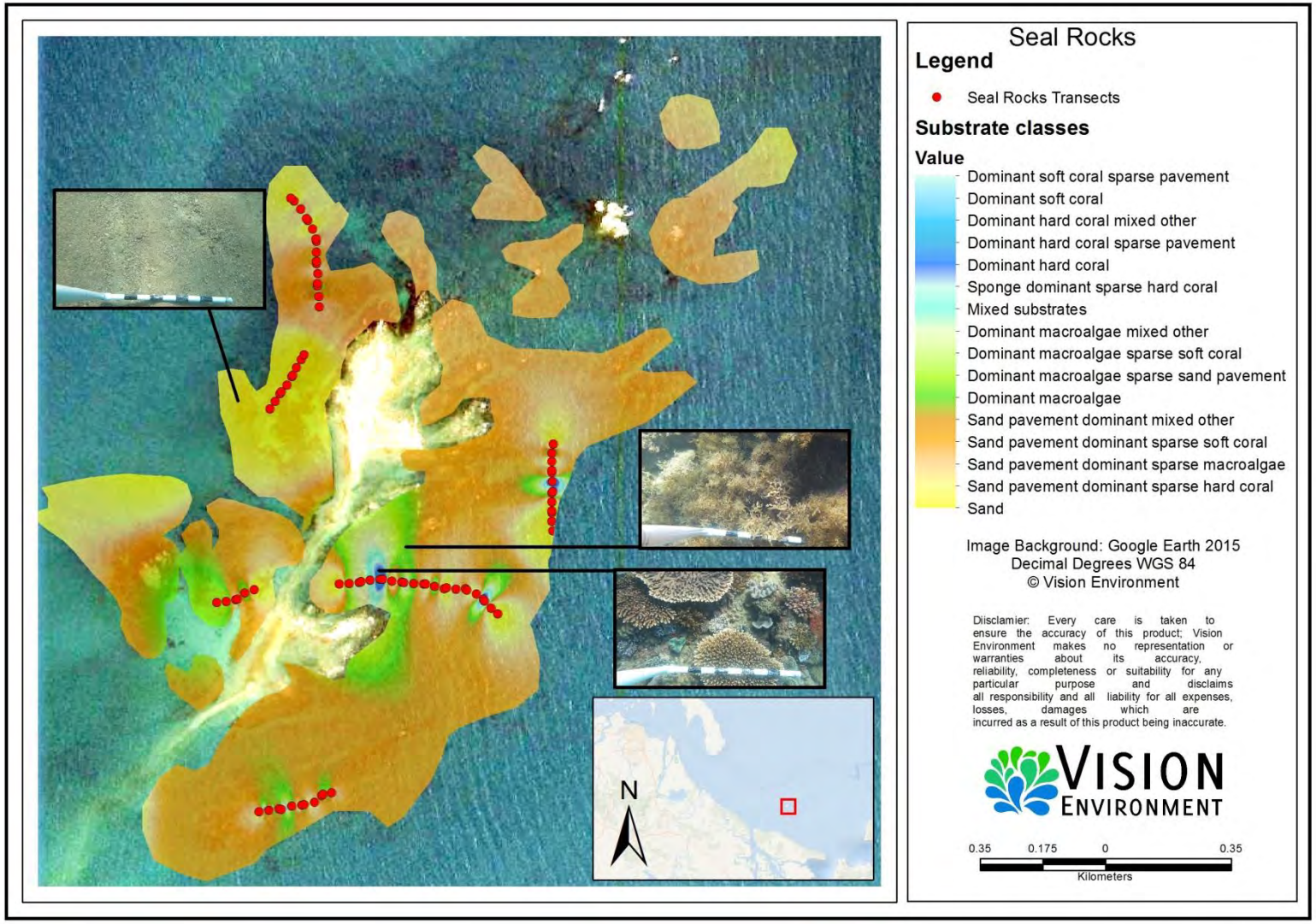


Figure 3.8 Reef area locality and overlaid surface substrate interpolation for Seal Rocks

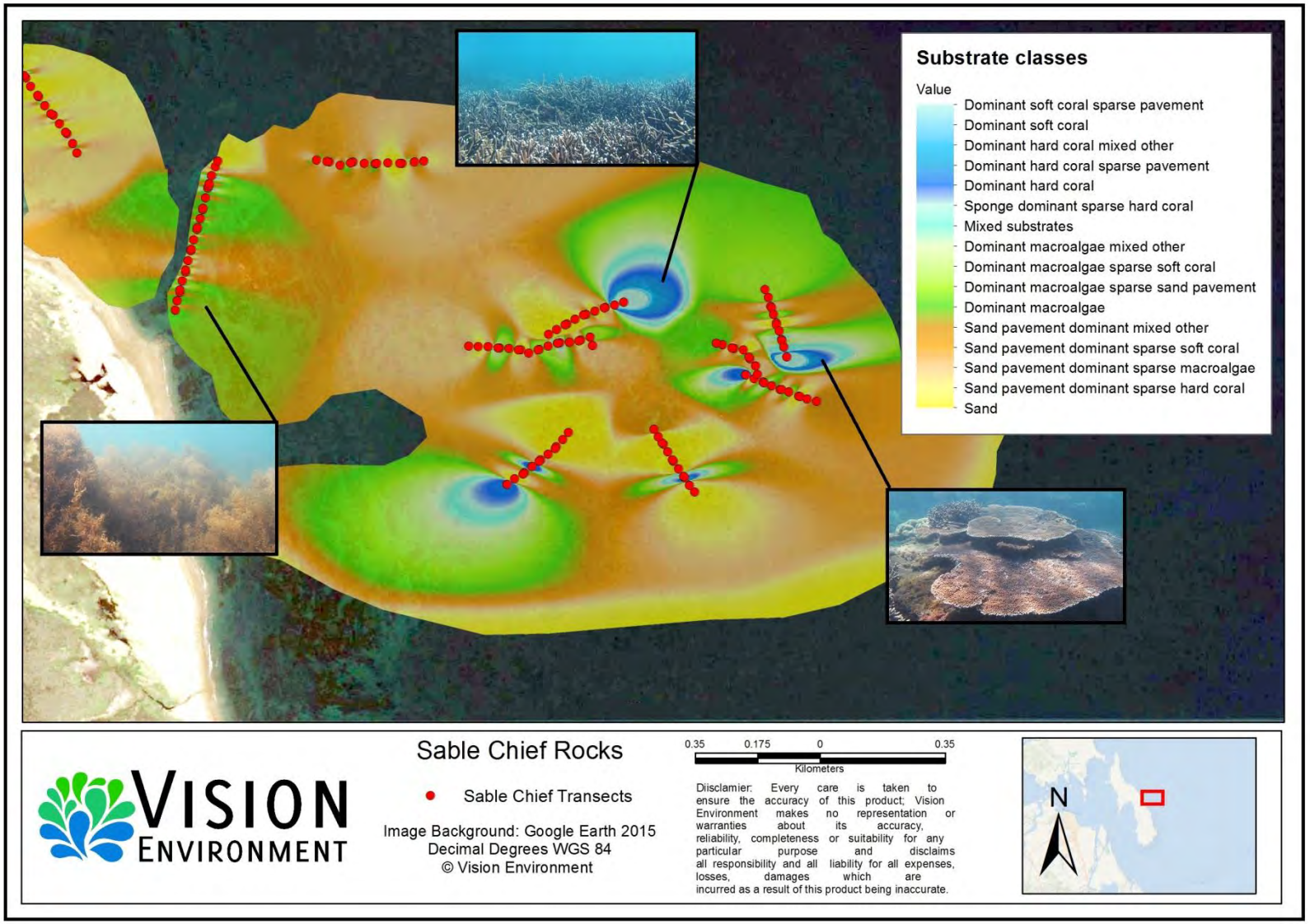


Figure 3.9 Reef area locality and overlaid surface substrate interpolation for Sable Chief Rocks

3.1.3 Seasonal comparisons

Results from transects surveyed in 2014 were compared to results from those undertaken during the 2015 wet season. Although there were significant (P < 0.05) differences among locations (refer Section 3.1.1), there were no significant (P > 0.05) differences in any of the major category substrates. This included hard coral, soft coral, macroalgae and sediment cover within locations, between the 2014 dry season and the 2015 wet season (refer Table 3.4). The exception was sponges, which had higher cover in the 2014 dry season compared to the 2015 wet season. However, sponges made up less than 1% of the total cover for all sites (refer Table 3.4).

Slight, but significant dissimilarities in species composition (minor categories) were observed among locations but not between seasons, with highest dissimilarities observed at Jenny Lind Banks (PERMANOVA; P (Perm-Location) < 0.001; *df* 3; MS 129.1). The main classes that influenced the variations among locations were macroalgae and hard coral, particularly for Gatcombe Heads and Sable Chief, respectively (ANOSIM; R-statistic 0.513, P < 0.1%), and to a lower extent, Gatcombe Heads and Jenny Lind Banks (ANOSIM; R-statistic 0.444, P < 0.1%) (refer Table 3.5 and Figure 3.10).

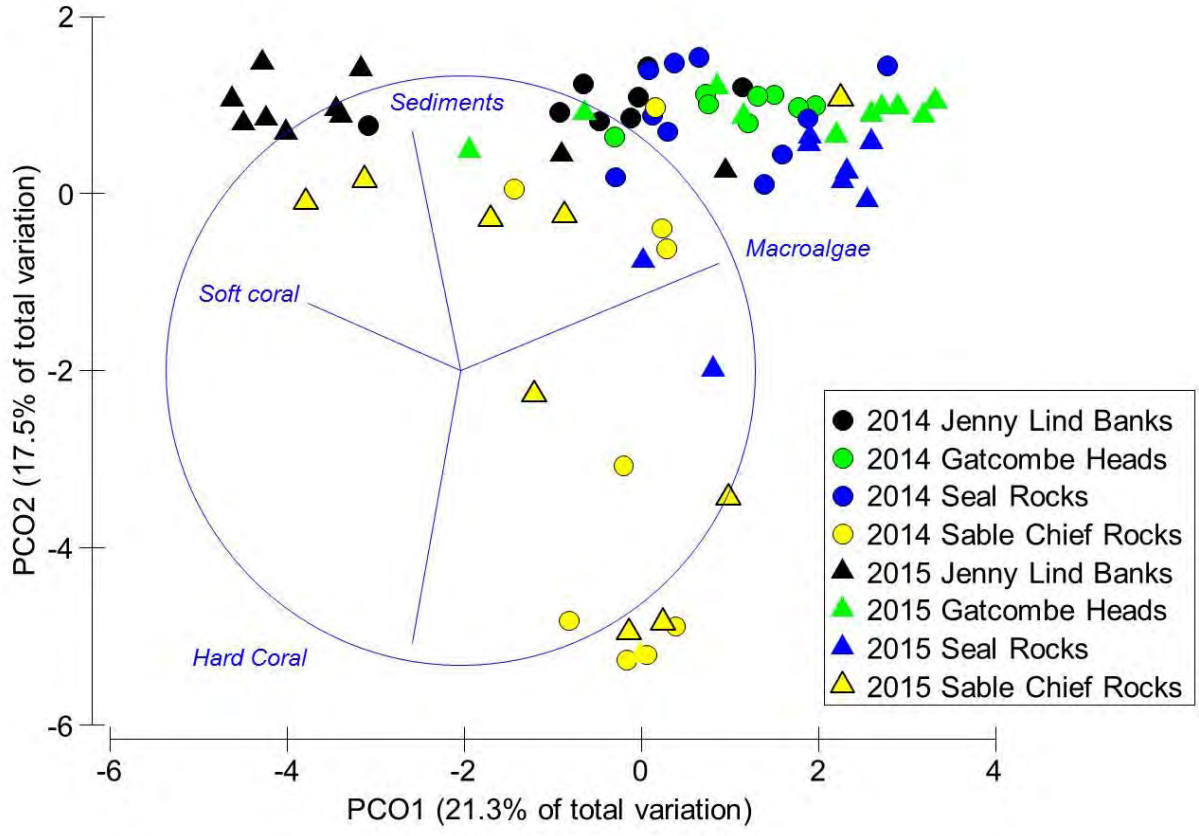


Figure 3.10 Principle Coordinates Ordination of minor substrate categories among seasons for select repeat transects

Figure notes: Major categories were overlaid to determine type of structure driving the variation among locations. Data were normalised and standardised using Euclidian distances; circles denote Dry season 2014; and triangles denote 2015 wet season.

Table 3.4 Mean (±se) percent dominant substrate cover among select transects between the 2014 dry season and 2015 wet season

Site	Jenny Lind Banks		Gatcombe Heads		Seal Rocks		Sable Chief Rocks		All locations	
Season	**Dry 2014**	**Wet 2015**	**Dry 2014**	**Wet 2015**	**Dry 2014**	**Wet 2015**	**Dry 2014**	**Wet 2015**	**Dry 2014**	**Wet 2015**
Sediment (%)	41.1 ± 2.0	72.3 ± 2.7	61.2 ± 3.7	41.1 ± 4.8	71.2 ± 7.5	48.1 ± 4.1	34.2 ± 9.8	52.5 ± 8.1	49.8 ± 4.1	50.7 ± 6.7
Macroalgae (%)	46.7 ± 3.8	10.7 ± 3.0	29.9 ± 4.3	50.4 ± 6.6	16.8 ± 3.8	45.3 ± 7.1	8.8 ± 2.9	6.3 ± 2.5	25.9 ± 3.1	26.9 ± 8.3
Hard coral (%)	6.5 ± 1.8	10.5 ± 2.0	4.5 ± 1.1	6.1 ± 2.7	9.9 ± 3.8	6.6 ± 3.9	54.1 ± 11.1	38.1 ± 9.6	20.8 ± 4.8	19.2 ± 8.5
Soft coral (%)	5.0 ± 2.3	6.4 ± 1.5	0.7 ± 0.2	1.5 ± 0.6	1.7 ± 0.8	0.0 ± 0.0	1.4 ± 0.7	2.5 ± 1.8	2.0 ± 0.6	2.7 ± 1.5
Sponge and ascidians (%)	0.2 ± 0.1	0.0 ± 0.0	2.2 ± 0.7	0.6 ± 0.4	0.2 ± 0.1	0.0 ± 0.0	1.0 ± 0.8	0.6 ± 0.3	0.9 ± 0.3	0.3 ± 0.2
Other biota (%)	0.4 ± 0.5	0.0 ± 0.0	1.5 ± 0.5	0.2 ± 0.2	0.2 ± 0.5	0.0 ± 0.0	0.4 ± 0.3	0.0 ± 0.0	0.6 ± 0.5	0.2 ± 0.3

Table note: Substrate listed in order of highest to lowest cover

Table 3.5 Mean (±se) minor substrate class percent cover among select transects between the 2014 dry season and 2015 wet season

Site	Jenny Lind Banks		Gatcombe Heads		Seal Rocks		Sable Chief Rocks		All locations	
Season	**Dry 2014**	**Wet 2015**	**Dry 2014**	**Wet 2015**	**Dry 2014**	**Dry 2014**	**Wet 2015**	**Dry 2014**	**Wet 2015**	**Dry 2014**
Sand (%)	16.6 ± 3.9	14.8 ± 4.3	42.7 ± 4.5	27.4 ± 3.3	43.8 ± 7.6	34.3 ± 6.8	17.7 ± 6.5	17.7 ± 7.0	30.6 ± 7.2	22.8 ± 5.8
Pavement (%)	23.1 ± 2.7	51.5 ± 5.7	18.3 ± 2.2	13.3 ± 4.9	20.0 ± 3.4	5.5 ± 1.8	12.1 ± 3.9	19.0 ± 6.1	18.3 ± 3.3	23.9 ± 7.8
Sargassium sp. (%)	30.4 ± 3.2	2.1 ± 0.8	23.4 ± 4.5	47.9 ± 6.2	14.1 ± 4.3	14.6 ± 5.8	2.7 ± 1.6	1.3 ± 0.9	17.1 ± 4.9	16.2 ± 7.6
Acropora formosa (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	44.1 ± 12.9	38.3 ± 13.3	11.4 ± 9.0	9.8 ± 8.6
Lobophora sp. (%)	1.1 ± 0.6	5.1 ± 3.1	2.6 ± 0.6	2.3 ± 1.0	5.9 ± 1.2	30.4 ± 6.0	4.1 ± 1.4	3.2 ± 1.9	3.6 ± 1.2	9.1 ± 4.9
Dead coral with algae (%)	0.1 ± 0.1	1.5 ± 0.5	<0.1	0.3 ± 0.3	<0.1	8.3 ± 5.2	1.6 ± 0.5	7.0 ± 2.8	0.4 ± 0.3	4.0 ± 2.9
Acropora hyacinthus (%)	0.5 ± 0.4	1.7 ± 0.9	1.3 ± 0.8	0.5 ± 0.5	1.0 ± 0.6	<0.1	7.5 ± 2.3	4.7 ± 1.5	2.6 ± 1.6	1.8 ± 1.1
Asparagopsis sp. (%)	9.7 ± 2.3	2.3 ± 1.1	0.3 ± 0.2	0.3 ± 0.3	0.7 ± 0.5	<0.1	0.2 ± 0.2	<0.1	2.5 ± 1.7	0.7 ± 0.7
Montipora verrucosa (%)	<0.1	4.6 ± 1.8	<0.1	0.1 ± 0.1	<0.1	6.0 ± 4.0	<0.1	0.8 ± 0.5	<0.1	2.8 ± 2.1
Turbinaria sp. (%)	1.0 ± 0.6	<0.1	0.7 ± 0.3	<0.1	8.8 ± 3.5	<0.1	0.6 ± 0.3	<0.1	3.1 ± 2.2	<0.1
Lobophytum sp. (%)	2.2 ± 1.4	4.0 ± 1.3	0.4 ± 0.2	0.2 ± 0.2	0.6 ± 0.5	<0.1	0.1 ± 0.1	1.4 ± 1.1	0.8 ± 0.7	1.5 ± 1.0

Site	Jenny Lind Banks		Gatcombe Heads		Seal Rocks		Sable Chief Rocks		All locations	
Season	**Dry 2014**	**Wet 2015**	**Dry 2014**	**Wet 2015**	**Dry 2014**	**Dry 2014**	**Wet 2015**	**Dry 2014**	**Wet 2015**	**Dry 2014**
Mussidae - *Favia* sp. (%)	0.2 ± 0.2	2.7 ± 1.3	0.6 ± 0.3	0.3 ± 0.3	0.2 ± 0.2	<0.1	0.7 ± 0.4	1.3 ± 0.7	0.4 ± 0.3	1.2 ± 0.8
Sarcophyton sp. (%)	1.0 ± 0.7	2.4 ± 0.8	0.1 ± 0.1	0.6 ± 0.5	0.3 ± 0.2	<0.1	0.2 ± 0.2	0.6 ± 0.5	0.4 ± 0.4	1.0 ± 0.6
Poritidae goniopora (%)	0.7 ± 0.4	1.7 ± 0.7	0.9 ± 0.4	0.9 ± 0.7	0.1 ± 0.1	<0.1	0.2 ± 0.1	0.7 ± 0.5	0.4 ± 0.3	0.9 ± 0.6
Sponge/ascidian (%)	0.2 ± 0.1	0.1 ± 0.1	2.2 ± 0.7	0.5 ± 0.3	0.3 ± 0.1	<0.1	1.1 ± 0.8	0.5 ± 0.2	0.9 ± 0.6	0.3 ± 0.2
Padina sp. (%)	3.8 ± 1.2	<0.1	0.2 ± 0.1	<0.1	1.0 ± 0.4	<0.1	<0.1	<0.1	1.2 ± 0.8	<0.1
Faviidae - *Favites* sp. (%)	0.4 ± 0.3	1.4 ± 1.0	0.2 ± 0.1	<0.1	1.1 ± 0.8	<0.1	0.4 ± 0.3	0.6 ± 0.6	0.5 ± 0.5	0.6 ± 0.6
Poritidae - *Porities* sp. (%)	2.1 ± 1.2	0.9 ± 0.4	0.2 ± 0.2	<0.1	<0.1	<0.1	0.4 ± 0.3	0.7 ± 0.5	0.7 ± 0.6	0.4 ± 0.4
Coralline algae (%)	0.9 ± 0.4	0.6 ± 0.5	0.9 ± 0.2	0.5 ± 0.3	0.2 ± 0.1	<0.1	0.5 ± 0.1	0.5 ± 0.6	0.6 ± 0.2	0.4 ± 0.4
Faviidae *goniastrea* spp. (%)	1.1 ± 0.8	<0.1	0.5 ± 0.4	0.4 ± 0.3	0.3 ± 0.2	<0.1	0.2 ± 0.1	0.5 ± 0.6	0.5 ± 0.4	0.2 ± 0.3
Sinularia sp. (%)	1.6 ± 0.7	<0.1	<0.1	<0.1	0.5 ± 0.2	<0.1	0.5 ± 0.3	<0.1	0.7 ± 0.4	<0.1
Halymenia sp. (%)	<0.1	0.5 ± 0.3	<0.1	0.3 ± 0.2	0.1 ± 0.1	0.2 ± 0.2	0.1 ± 0.1	<0.1	0.3 ± 0.3	0.3 ± 0.2
Montipora capricornis (%)	0.2 ± 0.2	0.3 ± 0.4	<0.1	0.7 ± 0.6	0.2 ± 0.2	<0.1	0.7 ± 0.7	<0.1	0.3 ± 0.4	0.3 ± 0.4
Acropora sp. (%)	0.1 ± 0.1	<0.1	<0.1	<0.1	<0.1	<0.1	1.8 ± 1.1	0.3 ± 0.4	0.5 ± 0.6	0.1 ± 0.2
Montipora sp. (%)	<0.1	<0.1	<0.1	1.5 ± 1.6	<0.1	<0.1	<0.1	0.4 ± 0.3	<0.1	0.5 ± 0.8
Rubble (%)	1.3 ± 0.6	<0.1	0.1 ± 0.1	<0.1	0.2 ± 0.1	0.0 ± 0.0	0.2 ± 0.1	<0.1	0.4 ± 0.3	<0.1
Galaxaura sp. (%)	<0.1	<0.1	0.6 ± 0.3	<0.1	<0.1	<0.1	<0.1	<0.1	0.3 ± 0.2	0.1 ± 0.1
Pocillopora verrucosa (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.3 ± 0.3
Poor quality photogprah* (%)	<0.1	<0.1	0.2 ± 0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.3 ± 0.2	<0.1
Pocillopora damicornis (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.3 ± 0.3	<0.1
Alveopora sp. (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.2 ± 0.3
Tape, Wand, Shadow* (%)	<0.1	<0.1	0.2 ± 0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.2 ± 0.1	<0.1
Turfing algae (%)	<0.1	<0.1	0.9 ± 0.8	<0.1	<0.1	<0.1	<0.1	<0.1	0.2 ± 0.4	<0.1
Unknown biota (%)	<0.1	<0.1	0.1 ± 0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.2 ± 0.1	<0.1

Site	Jenny Lind Banks		Gatcombe Heads		Seal Rocks		Sable Chief Rocks		All locations	
Season	**Dry 2014**	**Wet 2015**	**Dry 2014**	**Wet 2015**	**Dry 2014**	**Dry 2014**	**Wet 2015**	**Dry 2014**	**Wet 2015**	**Dry 2014**
Other green (%)	<0.1	<0.1	0.2 ± 0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.1 ± 0.1	<0.1
Nephtheidae (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.1 ± 0.2
Acropora digitifera (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Hydroid (%)	<0.1	<0.1	0.2 ± 0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Xenia (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Dictyota sp. (%)	<0.1	<0.1	0.4 ± 0.3	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Mobile organism (%)	<0.1	<0.1	0.2 ± 0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Anacropora forbesi (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Gorgonian (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	0.1 ± 0.1
Dictyopteris sp. (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Caluerpa racemose (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Zostera sp. (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Agariciidae - Agaricia sp. (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Halimeda sp. (%)	<0.1	<0.1	0.1 ± 0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1
Caluerpa taxifolia (%)	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1	<0.1

Table Note: *Denotes photograph interference. Substrate listed in order of highest to lowest cover

Results of the Project monitoring program were similar to that of Ayling et al. (2012) who found in a survey of fringing reefs adjacent to Facing Island, that although hard and soft coral cover did not change significantly temporally, macroalagae cover did. However, the study by Ayling et al. (2012) was conducted twice over the same seasonal period. The Project monitoring program showed distinct wet season/dry season changes in macroalgae cover, particularly for Jenny Lind Banks (refer Table 3.4 and Figure 3.10). This may be attributed to a number of factors, including changes in temperature, light, nutrient enrichment and grazer activity (Coman et al. 2003, Costanzo et al. 2003, Lapointe et al. 2005, Lapointe et al. 2010), as many macroalgal species have demonstrated marked growth during short time periods when conditions are favourable (Burkholder et al. 2007, Costanzo et al. 2004, Lin and Fong 2008, Pihl et al. 1999).

3.1.4 Permanent transects

Permanent transects were established at four locations (Gatcombe Heads, Sable Chief Rocks, Jenny Lind Banks, and Seal Rocks). Transects were resurveyed after 12 months from three of the four locations (Gatcombe Heads, Sable Chief Rocks and Jenny Lind Banks). Due to the relatively shallow depth of the colonies at Seal Rocks, all transect corals were severely damaged during the passing of Tropical Cyclone Marcia in February 2015. Hence, no temporal data was able to be collected for this location.

There were no significant ($P > 0.05$) differences in coral growth per year among locations, with an average growth rate of 9.7 ± 1.2mm/year for the two species of Acroporidae at all sites (refer Figure 3.11). The growth rates were measurably lower (approximately 11-22%) than growth rates reported for the same corals in other experiments (Harriott 1998, Oliver 1984). However, the growth rate differences reported in other studies were most probably attributed to differences in temperature, geographic location and water quality, as these variables are known to have a large effect on coral growth (Baird and Marshall 2002, Harriott 1998, Koop et al. 2001, Rogers 1979).

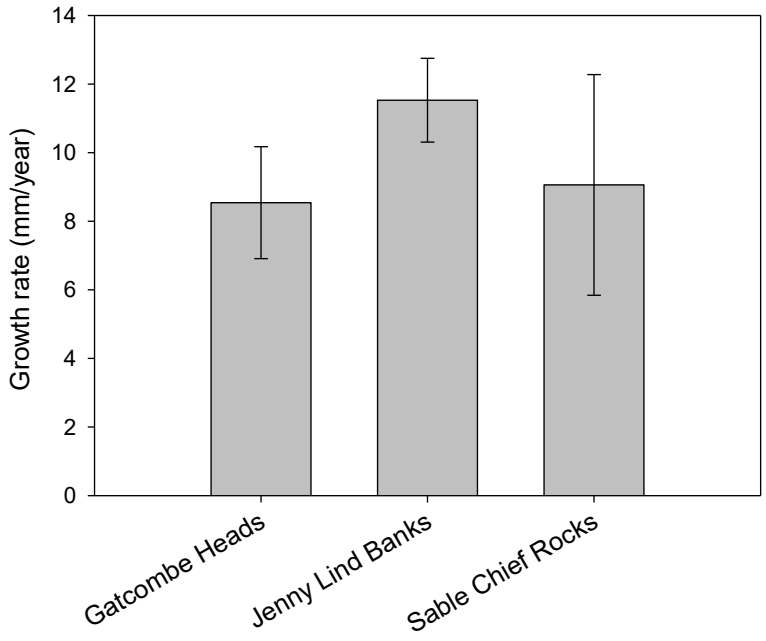


Figure 3.11 Mean ± (se) growth rate, per year, among locations in two *Acropora* sp.

Luminosity and pigmentation were measured at the three locations. Branching corals from the Acroporidae family were slightly lighter in pigmentation compared to that measured the previous year at Gatcombe Heads and Sable Chief Rocks, but there was no significant ($P > 0.05$) difference between the two locations. However, branching corals at Jenny Lind Banks had become significantly darker in pigmentation since the previous year and compared to Gatcombe Heads and Sable Chief Rocks (refer Table 3.6). Changes in pigment concentrations attributed to changes in concentrations of algae and/or zooxanthellae, can be affected by a number of stressors including but not limited to, water clarity, light availability, temperature, ocean acidification and season (Chen et al. 2005, Crawley et al. 2010, Fabricius 2006).

In general, ±8-10% difference in luminosity was observed in pigment variation between June 2014 and June 2015 at all sites, however, no bleaching was observed in any of the colonies.

Table 3.6 Mean ± (se) percentage difference in coral pigment change, for *Acropora* sp., over 12 months

Site	RGB (%)	Red (%)	Green (%)	Blue (%)	Luminosity (%)
Gatcombe Heads	-11.2 ± 4.8	-16.3 ± 6.0	-11.0 ± 5.1	-5.7 ± 3.7	-12.2 ± 5.2
Jenny Lind Banks	11.0 ± 3.3	16.6 ± 4.2	14.5 ± 8.9	22.5 ± 8.1	8.6 ± 2.6
Sable Chief	-4.8 ± 2.4	-5.2 ± 3.1	-8.2 ± 4.3	-0.2 ± 1.4	-6.6 ± 3.4
Seal Rocks	-	-	-	-	-

Table notes: Negative numbers denote corals have gone lighter compared to the previous year and positive numbers denote corals have gone darker compared to previous years. RGB denotes Red/Green/Blue colour channels

3.2 Reef associated nekton

3.2.1 Transects

Reef associated fish are important components of shallow water habitats, in that they form complex ecosystem functions. For example, some reef associated fish species are obligate corallivores, meaning that they consume only live coral tissue. Other fish species are grazers, which rely on different types of algal growth thus keeping macroalgal habitats within check. Some species, such as parrot fishes, graze on the coral structure, contributing to links within the carbon cycle and reef re-formation (Fabricius et al. 2005, Glynn 1993, Hay 1984, Hughes 1994, Johansen and Jones 2013, Kavanagh 2000, Letourneur et al. 1998, Pratchett et al. 2011). It is suggested that up to 75% of fish found within a coral reef rely on live corals for food, shelter or settlement (Pratchett et al. 2011). Hence, there is a strong positive relationship between coral and other live substrate cover, and fish reef associated assemblages.

To date, no studies on relationships between fish communities and substrate cover have been conducted within Port Curtis and therefore this Project monitoring program is a first for Port Curtis.

A total of 6,037 fish from 59 species were encountered throughout the four study locations during the 2014 dry season (37%) and the 2015 wet season (63%). The most common fish species was the Yellowtail demoiselle (*Neopomacentrus azysron*) (41%) (refer Photograph 3.5), followed by the Spotted-tail wrasse (*Coris caudimacula*) (11%), and the Wards damsel (*Pomacentrus wardi*) (10%). The most common fish families throughout the

monitoring program included Pomacentridae (Damsel fish) (73%), followed by Labridae (Wrasse) (21%), and Apogonidae (Cardinal fishes) (2%). Table 3.7 lists the reef community species encountered during the surveys.

Photograph 3.5 **Example of the abundant Yellowtail demoiselle (*Neopomacentrus azysron*) at Sable Chief Rocks**

Table 3.7 **List of families, species and common names of reef associated fish species encountered throughout the study**

Family	Species	Common Name
Acanthuridae	*Acanthurus bahianus*	Ocean surgeonfish
Acanthuridae	*Acanthurus nigrofuscus*	Brown surgeonfish
Acanthuridae	*Acanthurus pyroferus*	Mimic surgeonfish
Apogonidae	*Apogon bandanensis*	Bigeye cardinalfish
Apogonidae	*Cheilodipterus quinquelineatus*	Five-lined cardinalfish
Chaetodontidae	*Amphichaetodon melbae*	Narrow-barred butterflyfish
Chaetodontidae	*Chaetodon aureofasciatus*	Goldern butterflyfish
Chaetodontidae	*Chaetodon plebeius*	Blueblotch butterflyfish
Chaetodontidae	*Chaetodon trifascialis*	Chevron butterflyfish
Chaetodontidae	*Chaetodon ulietensis*	Doublesaddle butterflyfish
Cheilodactylidae	*Cheilodactylus ephippium*	Painted morwong
Diodontidae	*Diodon nichthemerus*	Slender-spined porcupinefish
Elapidae	*Aipysurus laevis*	Olive sea snake
Epinephelidae	*Plectropomus maculatus*	Bar-cheeked trout
Gobiidae	*Cyptocentrus* sp.	Ventral-barred shrimp goby
Haemulidae	*Diagramma pictum*	Painted sweetlip
Haemulidae	*Plectorhinchus flavomaculatus*	Goldspotted sweetlip
Hemiscylliidae	*Hemiscyllium ocellatum*	Epaulette shark
Labridae	*Bodianus mesothorax*	Split-level hogfish
Labridae	*Cheilinus fasciatus*	Red-breasted wrasse

Family	Species	Common Name
Labridae	*Choerodon schoenleinii*	Blackspot tuskfish
Labridae	*Coris caudimacula*	Spotted-tail wrasse
Labridae	*Ctenolabrus rupestris*	Goldsinny wrasse
Labridae	*Pseudolabrus guentheri*	Gunther's wrasse
Labridae	*Suezichthys arquatus*	Painted rainbow wrasse
Labridae	*Synodus binotatus*	Two-spot lizardfish
Labridae	*Thalassoma lunare*	Moon wrasse
Lutjanidae	*Lutjanus carponotatus*	Spanish flag snapper
Lutjanidae	*Lutjanus russellii*	Moses' snapper
Nemipteridae	*Scolopsis bilineata*	Two-lined monocle bream
Pomacanthidae	*Chaetodontoplus duboulayi*	Scribbled angelfish
Pomacentridae	*Abudefduf bengalensis*	Bengal sergeant
Pomacentridae	*Abudefduf sexfasciatus*	Scissortail sergeant
Pomacentridae	*Chromis nitida*	Yellowback puller
Pomacentridae	*Chrysiptera cyanea*	Blue demoiselle
Pomacentridae	*Chrysiptera rollandi*	Rolland's demoiselle
Pomacentridae	*Neopomacentrus azysron*	Yellowtail demoiselle
Pomacentridae	*Pomacentrus amboinensis*	Ambon damsel
Pomacentridae	*Pomacentrus bankanensis*	Speckled damselfish
Pomacentridae	*Pomacentrus coelestis*	Neon damselfish
Pomacentridae	*Pomacentrus milleri*	Millers damselfish
Pomacentridae	*Pomacentrus moluccensis*	Lemon damsel
Pomacentridae	*Pomacentrus wardi*	Wards damsel
Pomacentridae	*Stegastes apicalis*	Yellowtip gregory
Pomacentridae	*Stegastes nigricans*	Dusky gregory
Pomacentridae	*Stegastes* sp.	Gregory damsel
Pseudochromidae	*Pseudochromis fuscus*	Yellow dottyback
Pseudochromidae	*Pseudochromis porphyreus*	Purple dottyback
Pseudochromidae	*Pseudochromis porphyreus*	Magenta dottyback
Pseudochromidae	*Pseudochromis* sp.	Blue dottyback
Pseudochromidae	*Pseudochromis* sp.	Pink dottyback
Serranidae	*Diploprion bifasciatum*	Barred soapfish
Serranidae	*Epinephelus merra*	Dwarf spotted rockcod (wire netting cod)
Siganidae	*Siganus corallinus*	Blue spotted rabbitfish
Sparidae	*Acanthopagrus australis*	Yellowfin bream
Sphyraenidae	*Sphyraena obtusata*	Yellowtail barracuda
Tetraodontidae	*Arothron firmamentum*	Starry toado

Significant ($P < 0.05$) differences in fish density, species richness, diversity and species evenness was observed among sampling locations, with highest numbers of fish and species encountered at Sable Chief Rocks and Jenny Lind Banks (refer Figure 3.12). Lowest numbers and species were encountered at Seal Rocks and Gatcombe Heads. No significant ($P > 0.05$) differences were observed in fish biodiversity among seasons.

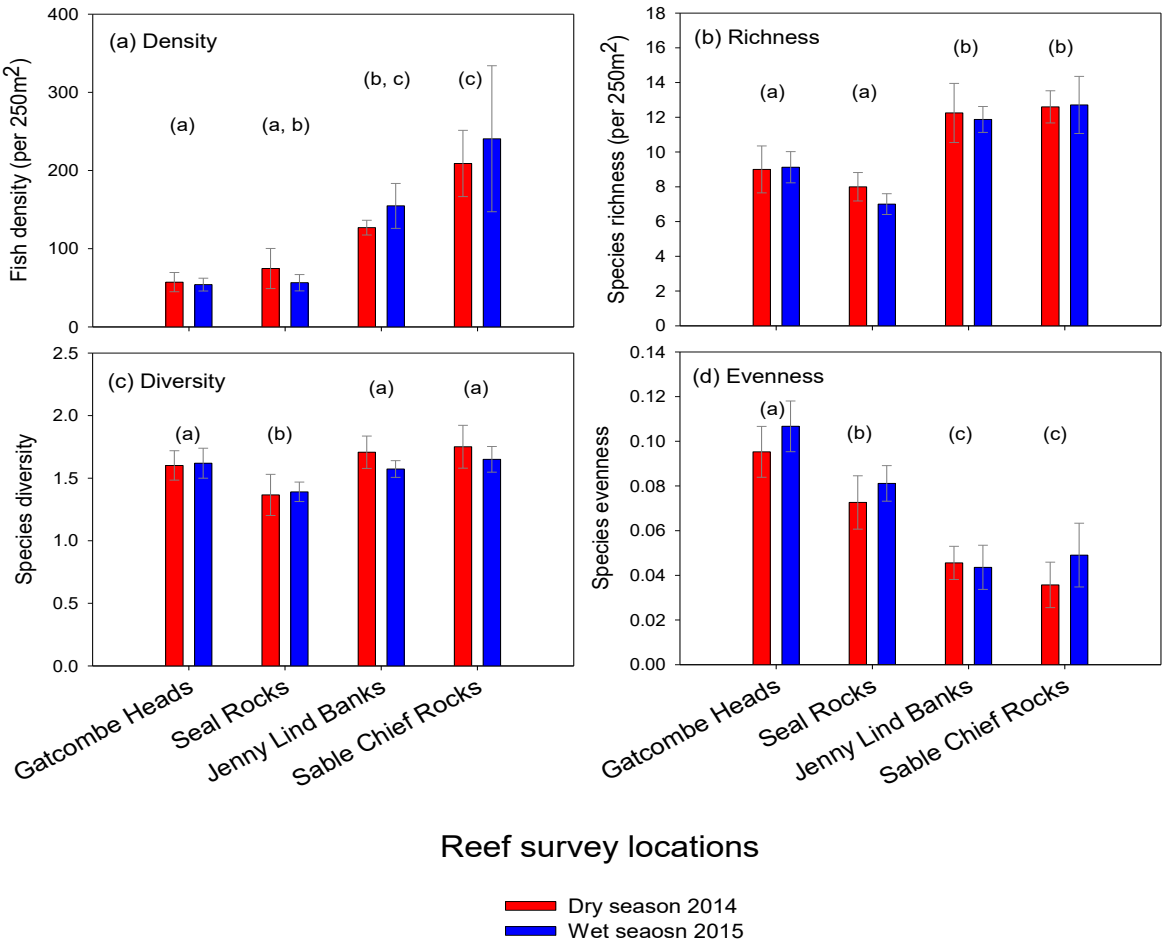


Figure 3.12 Mean (±se) (a) fish density, (b) species richness, (c) diversity, and (d) species evenness at four reef locations

Figure note: Data for Gatcombe Heads and Oyster Rocks have been pooled due to low numbers, and geographical proximity. Same letters denote no significant (P > 0.05) differences among locations, different letter denote significant (P < 0.05) differences among locations.

Reef obligate and associated fish communities tend to be territorial or have very limited homing ranges, hence they do not move large distances throughout their life histories. This is likely why no differences in biodiversity were observed among seasons.

Significantly dissimilar reef associated fish assemblages were observed among locations (R-statistic 0.197; P < 0.2%), and season (R-statistic 0.92; P < 0.1%), with highest differences in fish community structure encountered between Gatcombe Heads and Sable Chief Rocks (ANOSIM; R statistic 0.41), and Seal Rocks and Sable Chief Rocks (ANOSIM; R statistic 0.3). However, fish communities were similar between Jenny Lind Banks and Sable Chief Rocks, as well as Gatcombe Heads and Seal Rocks.

Minor correlations between substrate community matrices and fish assemblages were encountered, with hard coral substrate being the best correlation to explain fish community assemblages among locations (refer Figure 3.13). However, macroalgae was highly correlated with fish assemblages between seasons. The results suggest that different fish communities may be governed by the type of reef structure present. Hence, a change in substrate type is likely to lead to a change in fish communities. Hard corals represented 16% of the total variation in the total data and macroalgae contributed to 45% (refer Figure 3.13).

This result has also been noted at other reefs throughout the world. A change in reef habitat type can affect fish biodiversity and community structure (Baker et al. 2008, Harding et al. 2006, Hay 1984, Johansen and Jones 2013, Nero and Sealey 2005, Pratchett et al. 2011, Wenger and McCormick 2013, Wenger et al. 2014).

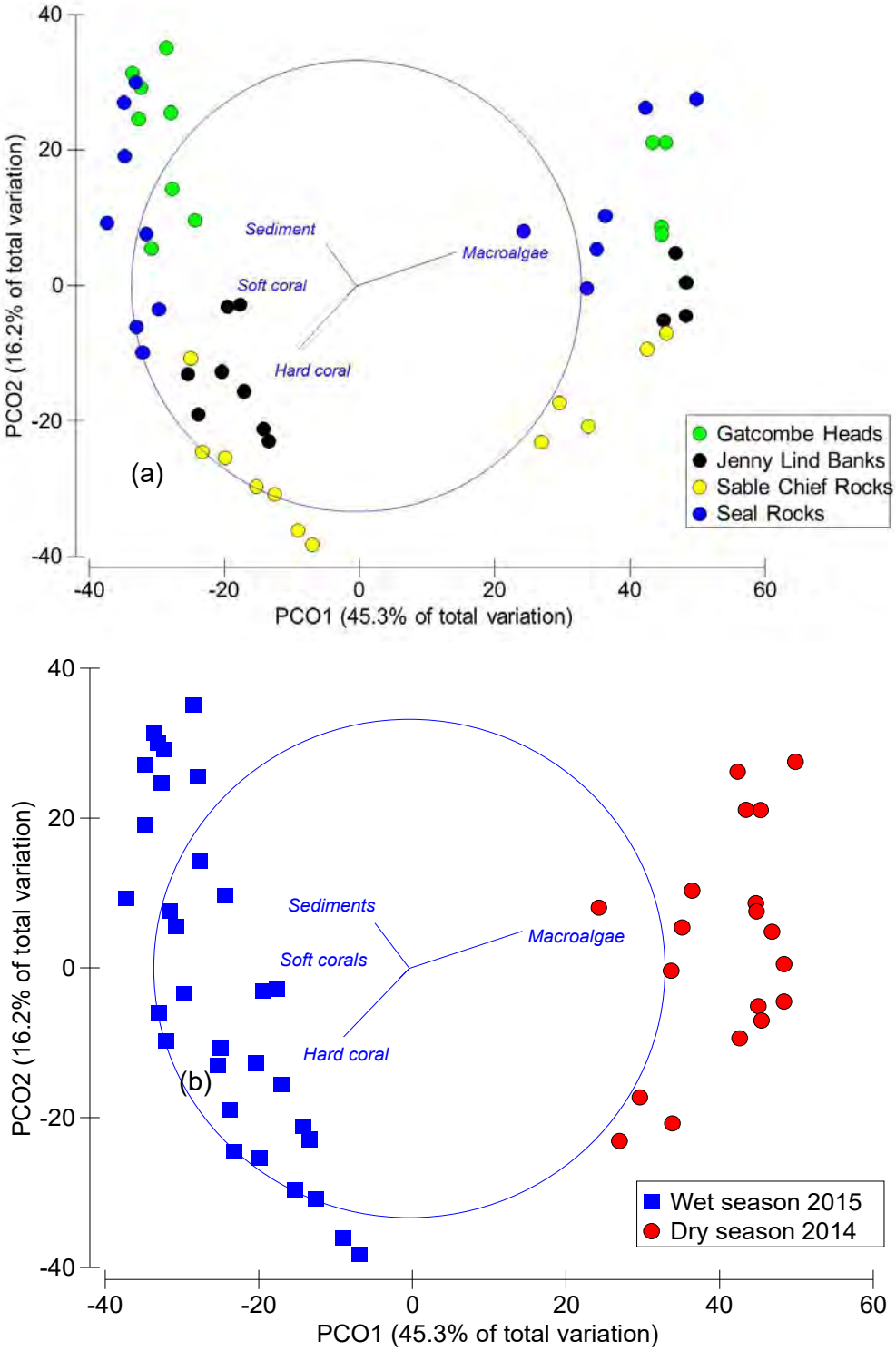


Figure 3.13 Principle Coordinates Ordination of reef associated nekton among (a) locations, and (b) season

Figure note: Major substrate categories were overlaid to determine type of structure driving the variation among fish assemblages. Data were standardised and dispersion weighted

3.2.2 Baited Remote Underwater Video Stations

During the 2014 dry season, BRUVS were deployed at four locations, including Seal Rocks, Jenny Lind Banks, Gatcombe Heads and Sable Chief Rocks. BRUVS have their advantages in capturing observations and measurements in larger mobile animals that normally avoid SCUBA divers, including sharks and rays, in addition to the ability to make observations within deep water habitats that are limited to SCUBA surveys. However, visibility plays a major factor in recording these organisms (Assis et al. 2013, Harasti et al. 2015, Harvey et al. 2012, Lowry et al. 2012, Mallet and Pelletier 2014, Unsworth et al. 2014).

Limited data was collected from these stations due to regular poor water clarity, thus providing very few opportunities to utilise this type of technique. Unlike the diver transects, the BRUVS being a stationary image capture require very high visibility to obtain useable data. No useable data was able to be collected in the 2015 wet season due to consistent poor water visibility during multiple deployments and no data was collected at Seal Rocks in both seasons due to visibility constraints.

Highest number of occurrences of fish in frame in a one hour block was at Gatcombe Heads, followed by Jenny Lind Banks and Sable Chief Rocks (refer Table 3.8), suggesting that fish resided in the vicinity of the BRUVS longer at Gatcombe Heads compared to other locations. There were also significantly ($P < 0.05$) higher numbers of fish observed per frame at Gatcombe Heads compared to Jenny Lind Banks and Sable Chief Rocks, however there were no significant ($P > 0.05$) differences in species richness among locations. Highest numbers of species were recorded at Jenny Lind Banks. For a list of species observed at all locations, refer to Table 3.9.

Table 3.8 Mean ± (se) number of fish species occurring per frame and over a one hour soak time (N_{max}) using BRUVS

Locations	Gatcombe Heads	Sable Chief Rocks	Jenny Lind Banks
Total species per 1 hour deployment	5	5	8
Mean max No. per Frame (N_{max})	1.9 ± 0.1	1.3 ± 0.1	1.2 ± 0.1
Mean richness per frame	1.1 ± 0.02	1.1 ± 0.05	1 ± 0
Total number of occurrences	122	30	50

Table 3.9 List of species occurring at different locations

Family	Species name	Common name	Present at location		
			Gatcombe Heads	Sable Chief Rocks	Jenny Lind Banks
Balistidea	*Sufflamen* sp.	Black trigger fish	No	Yes	No
Chaetodontidae	*Chaetodon plebeius*	Blue-blotch butterflyfish	No	No	Yes
Chaetodontidae	*Chaetodon trifascialis*	Chevron butterflyfish	No	Yes	Yes
Labridae	*Pseudolabrus guentheri*	Gunther's wrasse	No	No	Yes
Labridae	*Thalassoma lunare*	Moon wrasse	Yes	No	Yes
Lutjanidae	*Lutjanus russellii*	Moses' snapper	Yes	No	No

Family	Species name	Common name	Present at location		
			Gatcombe Heads	Sable Chief Rocks	Jenny Lind Banks
Pomacentridae	*Chrysiptera rollandi*	Rolland's demoiselle	No	No	Yes
Lutjanidae	*Lutjanus carponotatus*	Spanish flag snapper	Yes	Yes	Yes
Labridae	*Coris caudimacula*	Spotted-tail wrasse	No	No	Yes
Pomacentridae	*Chromis nitida*	Yellowback puller	No	Yes	No
Sparidae	*Acanthopagrus australis*	Yellowfin bream	Yes	No	No
Sphyraenidae	*Sphyraena obtusata*	Yellowtail barracuda	Yes	No	Yes
Pomacentridae	*Neopomacentrus azysron*	Yellowtail demoiselle	No	Yes	No

Due to the limited capture moments, the only new species that was recorded, which was not observed using SCUBA based visual survey techniques, was the Black trigger fish (*Sufflamen* sp.). All other fish were also encountered using visual surveys and in greater numbers and with higher statistical rigor.

3.3 Estuarine and coastal nekton communities

3.3.1 Estuarine systems

Estuaries are complex environments that support a wide range of flora and fauna, composed of a number of subtidal and intertidal habitats, including salt marshes, mangrove forests, soft muddy and sandy sediments, and seagrasses (Faunce and Serafy 2006, Meynecke 2009, Sheaves et al. 2007).

There are two large offshore islands (Curtis Island and Facing Island) that protect the waters of Port Curtis forming a narrow coastal embayment approximately 200 km^2 in area. Freshwater inputs come from two major rivers, the Boyne River and the Calliope River, and the numerous creeks and tributaries that discharge into the Port and surrounding waters. Strong tidal currents and a 5m tidal range also have a major influence on the area's marine intertidal ecosystems. Port Curtis (below the lowest astronomical tide mark) forms part of the Great Barrier Reef World Heritage Area (GBRMPA 1998), this marine environment is home to many tropical reef, estuarine and coastal fishes. Port Curtis also hosts a number of marine mammals including dugongs and turtles that rely on these habitats to survive (Taylor et al. 2007, Witting 2009). Therefore, similar to reef associated nekton, estuarine and coastal nekton are also important contributors to the marine ecosystem.

3.3.2 Cast net sampling

A total of 2,936 fish representing 34 species were caught in cast nets during the 2014 dry season (58.7%) and the 2015 wet season (41.3%) surveys at the inshore fish monitoring sites (refer Figure 2.2). The most common species occurring in most locations, irrespective of season, was the Southern herring, (*Herklotsichthys castelnaui*) (39%), followed by the Estuary glassfish (*Ambassis marianus*) (21%) and the Spottyface anchovy (*Stolephorus waitei*) (14%). Banana prawns (*Fenneropenaeus merguiensis*) were also common among catches (7%). Most other fish encountered each made up less than 1% of the total catch,

except for a select few species, including Green back mullet (*Liza subviridis*), Common ponyfish (*Leiognathus equulus*), and Slender garfish (*Hyporhamphus dussumieri*), which made up ~13% of the combined total catch. The most common fish families included Clupeidae (39%), Ambassidae (21%), and Engraulidae (14%). Table 3.10 shows the list of species encountered during cast netting.

Table 3.10 List of species encountered and proportions from all locations between the dry season 2014 and wet season 2015

Family name	Species name	Common name	Dry (%)	Wet (%)	Total (%)
Clupeidae	*Herklotsichthys castelnaui*	Southern herring	51.8	20.0	38.7
Ambassidae	*Ambassis marianus*	Estuary glassfish	25.7	15.0	21.3
Engraulidae	*Stolephorus waitei*	Spottyface anchovy	10.2	20.2	14.3
Penaeidae	*Fenneropenaeus merguiensis*	Banana prawn	1.3	16.0	7.4
Leiognathidae	*Leiognathus equulus*	Common ponyfish	3.9	4.8	4.3
Hemiramphidae	*Hyporhamphus dussumieri*	Slender garfish	0.1	10.1	4.2
Mugilidae	*Liza subviridis*	Greenback mullet	3.6	4.4	3.9
Gerreidae	*Gerres subfasciatus*	Common silverbiddy	0.6	3.1	1.6
Atherinidae	*Atherinomorus ogilbyi*	Common hardyhead	0.0	2.5	1.0
Sparidae	*Acanthopagrus butcheri*	Black bream	0.5	0.5	0.5
Hemiramphidae	*Arrhamphus sclerolepis*	Subnose garfish	0.1	0.9	0.4
Batrachoididae	*Tetractenos hamiltoni*	Common toadfish	0.6	0.0	0.4
Batrachoididae	*Tetractenos glaber*	Smooth toadfish	0.3	0.3	0.3
Sillaginidae	*Sillago ciliata*	Sand whiting	0.2	0.2	0.2
Siganidae	*Siganus lineatus*	Goldlined rabbitfish	0.0	0.5	0.2
Tetraodontidae	*Chelonodon patoca*	Milk-spotted pufferfish	0.3	0.0	0.2
Terapontidae	*Amniataba percoides*	Barred grunter	0.3	0.0	0.2
Latidae	*Lates calcarifer*	Barramundi	0.1	0.2	0.1
Portunidae	*Portunis pelagicus*	Blue swimmer crab	0.1	0.0	0.1
Mugilidae	*Mugil cephalus*	Sea mullet	0.1	0.1	0.1
Sparidae	*Acanthopagrus australis*	Yellowfin bream	0.0	0.2	0.1
Uranoscopidae	*Kathetostoma laeve*	Common stargazer	0.0	0.2	0.1
Engraulidae	*Engraulis australis*	Australian anchovy	0.0	0.2	0.1
Sphyraenidae	*Sphyraena jello*	Pick-handle harracuda	0.0	0.2	0.1
Terapontidae	*Amniataba percoides*	Barred Grunter	0.1	0.0	<0.1
Tetraodontidae	*Arothron manilensis*	Narrow-lined pufferfish	0.1	0.0	<0.1
Belonidae	*Strongylura strongylura*	Spottail needlefish	0.0	0.1	<0.1
Lutjanidae	*Lutjanus russelli*	Moses perch	0.0	0.1	<0.1
Carangidae	*carangoides humerosus*	Epaulette trevally	0.0	0.1	<0.1
Terapontidae	*Terapon jarbua*	Crescent perch	0.0	0.1	<0.1
Clupeidae	*Nematalosa erebi*	Bony bream	0.0	0.1	<0.1
Leptobramidae	*Leptobrama muelleri*	Beach salmon	0.0	0.1	<0.1
Belonidae	*Strongylura strongylura*	Spot tail needlefish	0.0	0.1	<0.1
Platycephalidae	*Platycephalus indicus*	Bar tail flathead	0.1	0.0	<0.1

Table note: Species listed in order of highest to lowest proportion

Only a limited number of nekton studies have been conducted within and surrounding Port Curtis (Alquezar 2010, Alquezar 2011, Alquezar and Scannell 2010, Currie and Connolly 2004, Graham et al. 2007). A comprehensive, whole of port trawl survey of the fish community in Port Curtis and The Narrows was conducted in the 2003 wet season, with over 80 species of demersal fish identified (Currie and Connolly 2004). The Common ponyfish, (*Leiognathus equulus*) was the most abundant and widely distributed fish encountered in the survey, with over 30% of the total abundance observed in half of all the sampling sites (Currie and Connolly 2004).

In the Project monitoring program, the Southern herring was the most abundant fish species contributing to 39% of the total catch for both seasons (refer Table 3.10), but was more prominent in the dry season than the wet. The Common ponyfish was also common during both seasons, although not in the same proportions that were observed by Currie and Connolly (2004). Other studies have also shown similar results to the Project monitoring program, with the following species being within the top five in their survey locations for shallow water habitats, including the Banana prawn (34-41%); the Spotty-face anchovy (17%); the Greenback mullet (12-16%); the Southern herring (10-35%), and the Common ponyfish (Alquezar 2010, Alquezar 2011, Alquezar and Scannell 2010).

Significant differences ($P < 0.05$) in density, species richness, diversity and species evenness (biodiversity) were encountered among locations (refer Figure 3.14), with highest numbers of fish encountered at the Lilleys Inlet, South Trees Inlet, P2B (Calliope River mouth) and QE3 (Narrows/Targinnie Creek). Lowest numbers were encountered at Sand Islet and SW Facing Island. Species richness was also significantly ($P < 0.05$) lower at Sand Islet, FL8 (Fisherman's Landing) and Wild Cattle compared to all other locations. In general, there were no significant differences ($P > 0.05$) in nekton biodiversity between the dry season and the wet season, however significant interactions among season and location were observed ($P < 0.05$; Location x Season), suggesting that differences in biodiversity were encountered between seasons for some locations but not others (refer Figure 3.14).

Similar to biodiversity, fish community assemblages were also significantly dissimilar (R-statistic 0.204; $P < 0.1\%$) among locations, irrespective of season (refer Figure 3.15), with largest differences observed between the South Trees Inlet and Sand Islet (R-statistic 0.6), in addition to Sand Islet and Lilleys Inlet (R-statistic 0.372; $P < 0.1\%$). Fish communities were similar ($P > 0.1\%$) in composition between FL8 and P2B, Wild Cattle Creek and Boyne River, and South Trees Inlet and Boyne River (refer Figure 3.15). There were no significant dissimilarities (R-statistic 0.032; $P > 0.1\%$) in fish communities among season, irrespective of location (refer Figure 3.15).

Differences in fish communities among locations can be attributed to a number of factors. Numerous studies have shown that different habitat, such as vegetated versus non-vegetated, can influence community structure (Able 2005, Abrantes and Sheaves 2009, Baker and Sheaves 2009, Clynick and Chapman 2002, Faunce and Serafy 2006, Gilkinson et al. 2003, Griffiths 2001, Guidetti 2000, Hindell 2006, Ikejima et al. 2003, Jaureguizar et al. 2003, Johnston and Sheaves 2007, Ley et al. 1999). In addition, physicochemical parameters, including turbidity and natural tidal flow, have been demonstrated to influence abundance of new recruits within Port Curtis and other locations around Queensland (Currie and Connolly 2004, Currie and Small 2005). Thus results are not surprising given the difference in the habitats surveyed. A change in habitat by anthropogenic influence may potentially lead to a change in nekton communities.

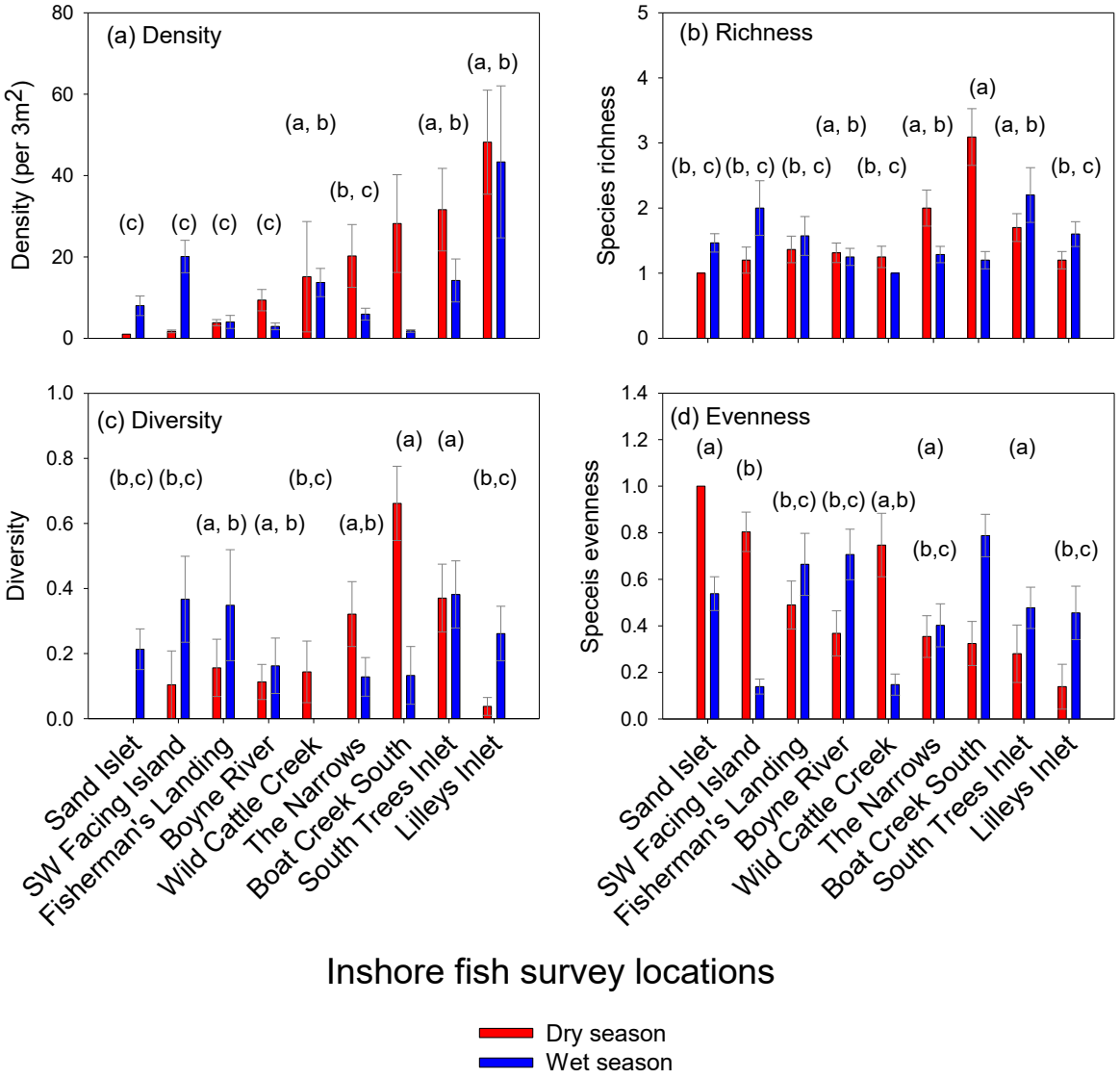


Figure 3.14 Mean (± se) nekton (a) density, (b) richness, (c) diversity and (d) evenness among locations and season for cast net sampling (n ≥ 10)

Figure note: Same letters denote no significant (P > 0.05) differences among locations, different letter denotes significant (P < 0.05) differences

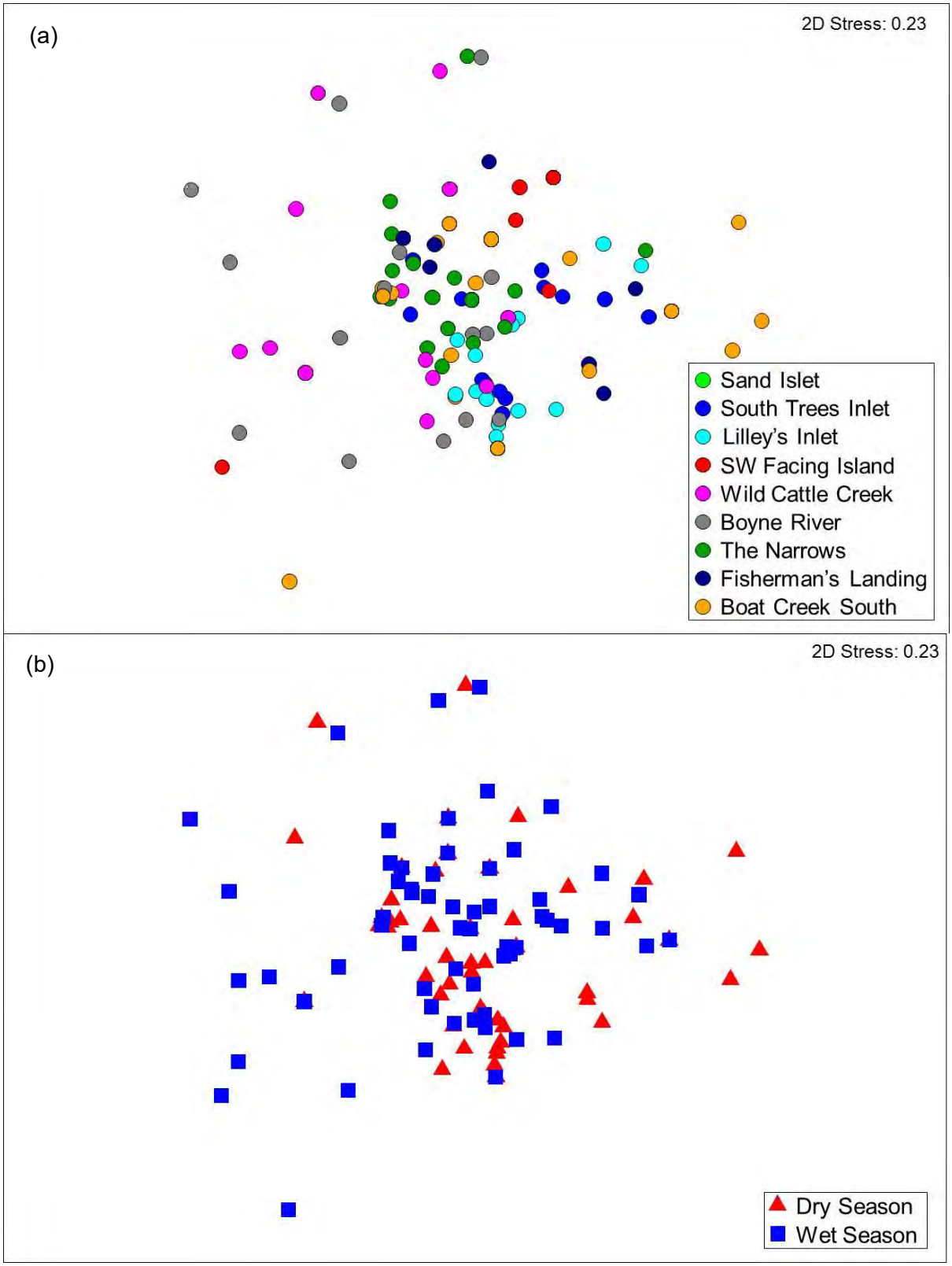


Figure 3.15 **Non-metric Multidimensional Scaling (n-MDS) ordination plot (Bray-Curtis similarity) of fish community assemblages utilising shallow complex habitats among (a) locations, and (b) seasons.**

Figure note: Data were standardised and dispersion weight corrected (Bray-Curtis similarity).

3.3.3 Gill net sampling

A total of 44 fish were collected from all locations during the dry (52%) and wet (48%) seasons utilising gill nets (refer Table 3.11). However, a high number of fish were observed (but not collected) and recorded during the two monitoring campaigns (refer Table 3.12). The most common fish caught, irrespective of season was the Giant queenfish (*Scomberoides commersonnianus*) (27%), followed by the Blue threadfin salmon (*Eleutheronema tetradactylum*) (11%), Beach salmon (*Leptobrama muelleri*) (11%), and the Mud crab (Scylla serrata) (11%). The most common families included Carangidae (31%), Leptobramidae (11%), Polynemidae (11%) and Portunidae (11%).

Table 3.11 List and numbers of species collected at each site using a gill net

Location	Family name	Species name	Common name	Dry	Wet	Total
Lilleys Inlet	Scatophagidae	*Scatophagus argus*	Spotted scat	3	0	3
	Carangidae	*Scomberoides commersonnianus*	Giant queenfish	1	1	2
	Terapontidae	*Amniataba percoides*	Barred grunter	1	0	1
	Scatophagidae	*Scatophagus argus*	Spotted scat	1	0	1
	Carangidae	*carangoides humerosus*	Epaulette trevally	0	1	1
Fishermans Landing	Leptobramidae	*Leptobrama muelleri*	Beach salmon	1	2	3
	Polynemidae	*Eleutheronema tetradactylum*	Blue threadfin salmon	3	0	3
	Portunidae	*Scylla serrata*	Mud crab	4	0	4
	Ariidae	*Ariopsis graeffei*	Blue catfish	1	0	1
	Rhinobatidae	*Aptychotrema rostrata*	Eastern shovelnose ray	0	1	1
Boat Creek South	Ariidae	*Ariopsis graeffei*	Blue catfish	2	0	2
	Leptobramidae	*Leptobrama muelleri*	Beach salmon	1	1	2
	Pomatomidae	*Pomatomus saltatri*	Tailor	0	1	1
	Cheloniidae	*Chelonia mydas*	Green turtle	0	1	1
South Trees Inlet	Polynemidae	*Eleutheronema tetradactylum*	Blue threadfin	1	0	1
	Carangidae	*Scomberoides commersonnianus*	Giant queenfish	0	1	1
	Carangidae	*Carangoides humerosus*	Epaulette trevally	0	1	1
	Portunidae	*Scylla serrata*	Mud crab	0	1	1
SW Facing Island	Cheloniidae	*Chelonia mydas*	Green turtle	1	0	1
	Terapontidae	*Amniataba percoides*	Barred grunter	1	0	1
	Polynemidae	*Eleutheronema tetradactylum*	Blue threadfin	1	0	1
Boyne River	Cheloniidae	*Chelonia mydas*	Green turtle	1	0	1
	Carangidae	*Scomberoides commersonnianus*	Giant queenfish	0	9	9
Sand Isle	Carcharhinidae	*Carcharhinus melanopterus*	Blacktip reef shark	0	1	1
Wild Cattle	-	-	-	0	0	0
The Narrows / Targinnie Creek	-	-	-	0	0	0

Table 3.12 List of nekton species netted and/or observed during the 2014 dry season and 2015 wet season sampling campaigns

Family name	Species name	Common name
Ambassidae	*Ambassis marianus*	Estuary glassfish
Ariidae	*Ariopsis graeffei*	Blue catfish
Atherinidae	*Atherinomorus ogilbyi*	Common hardyhead
Batrachoididae	*Tetractenos glaber*	Smooth toadfish
Batrachoididae	*Tetractenos hamiltoni*	Common toadfish
Belonidae	*Strongylura strongylura*	Spottail needlefish
Carangidae	*carangoides humerosus*	Epaulette trevally
Carangidae	*Scomberoides commersonnianus*	Giant queenfish
Carcharhinidae	*Carcharhinus melanopterus*	Black tip reef shark
Cheloniidae	*Chelonia mydas*	Green turtle
Cheloniidae	*Eretmochelys imbricata*	Hawksbill turtle
Clupeidae	*Herklotsichthys castelnaui*	Southern herring
Clupeidae	*Nematalosa erebi*	Bony bream
Clupeidae	*Sprattus novaehollandiae*	Australian sprat
Engraulidae	*Engraulis australis*	Australian anchovy
Engraulidae	*Stolephorus waitei*	Spottyface anchovy
Gerreidae	*Gerres subfasciatus*	Common silverbiddy
Hemiramphidae	*Arrhamphus sclerolepis*	Subnose garfish
Hemiramphidae	*Hyporhamphus dussumieri*	Slender garfish
Latidae	*Lates calcarifer*	Barramundi
Leiognathidae	*Leiognathus equulus*	Common ponyfish
Leptobramidae	*Leptobrama muelleri*	Beach salmon
Lutjanidae	*Lutjanus russelli*	Moses perch
Mictyridae	*Mictyris longicarpus*	Blue soldier crab
Mugilidae	*Liza subviridis*	Greenback mullet
Mugilidae	*Liza vaigiensis*	Diamondscale mullet
Mugilidae	*Mugil cephalus*	Sea mullet
Myliobatidae	*Aetobatus narinari*	Spotted eagle ray
Penaeidae	*Fenneropenaeus merguiensis*	Banana prawn
Platycephalidae	*Platycephalus indicus*	Bartail flathead
Polynemidae	*Eleutheronema tetradactylum*	Blue threadfin
Pomatomidae	*Pomatomus saltatri*	Tailor
Portunidae	*Portunis pelagicus*	Blue swimmer crab
Portunidae	*Scylla serrata*	Mud crab
Rhinobatidae	*Aptychotrema rostrata*	Eastern shovelnose ray
Scatophagidae	*Scatophagus argus*	Spotted scat
Siganidae	*Siganus lineatus*	Goldlined rabbitfish
Sillaginidae	*Sillago ciliata*	Sand whiting
Sparidae	*Acanthopagrus australis*	Yellowfin bream
Sparidae	*Acanthopagrus butcheri*	Black bream
Sphyraenidae	*Sphyraena jello*	Pick-handle Barracuda
Terapontidae	*Amniataba percoides*	Barred grunter

Family name	Species name	Common name
Terapontidae	*Terapon jarbua*	Crescent perch
Tetraodontidae	*Arothron manilensis*	Narrow-lined Pufferfish
Tetraodontidae	*Chelonodon patoca*	Milk-spotted Pufferfish
Uranoscopidae	*Kathetostoma laeve*	Common stargazer

Table note: Species include those listed in Tables 3.10 and 3.11

Highest numbers of nekton caught in gill nets, irrespective of season, were encountered at FL8 (12 individuals), followed by the Boyne River (10 individuals) and Lilleys Inlet (eight individuals). No nekton were collected by gill nets at Wild Cattle Creek or QE3 during both dry and wet season campaigns. Species richness was also highest at FL8 and Lilleys Inlet (refer Table 3.11).

There is a significant recreational and commercial fishing industry within Gladstone. However, recreational fishing catch rates have shown a steady decline since 1991 (Platten 2004). The most common inshore commercial/recreational fish caught in Port Curtis in 2004 was Whiting (*Sillago* sp. 81%), with the rest of the catch being comprised of a combination of Bream (*Acanthopagrus* sp.), Flathead, (*Platycephalus* sp.), Garfish (*Arrhamphus* sp.), Longtom (*Tylosurus* sp.), Beach salmon (*Leptobrama mulleri*) and Grunter (*Pomadasys* sp).

Recent studies have also shown a decrease in recreational fish catch rates by around 35%, in conjunction with a steady increase in fishing effort (25%), and boat registrations (>37%) from 2006-2014 (Sawynok et al. 2013, Sawynok et al. 2014). Highest catch rates were comprised of Mud crabs (~22%), followed by Bream (21%), and Barramundi (9%) (Sawynok et al. 2014).

Other known recreational and commercially important nekton within Port Curtis include Barramundi (*Lates calcarifer*), King salmon (*Onorhynchus* sp.), Blue salmon (*Eleutheronema tetradactylum*), Sea mullet (*Mugil cephalus*), Grunter (*Pomadasys kaakan*), Mud crab (*Scylla serrata*), and several shark species (DPI&F 2008). In the Project monitoring program, a large number of nekton caught during both seasons with gill nets, were of recreational or commercial value (67%), including some of the above mentioned species (refer Tables 3.11 and 3.12).

Several fish species found in Port Curtis (Alquezar 2010, Alquezar 2011, Alquezar and Scannell 2010, Currie and Connolly 2004, Graham et al. 2007) are listed on a number of endangered species lists. The Narrowsnout sawfish (*Pristis zijsron*) is listed as vulnerable under provisions of *Environment Protection and Biodiversity Conservation Act* (DoE 2015) and critically endangered on the IUCN Red List (IUCN 2015). However, this species has not been recorded in the local area since the late 1960s. Other species such as the Eastern grey nurse shark (*Carcharias taurus*), which have potential distributions within Port Curtis, are listed as critically endangered under provisions of the EPBC Act and the IUCN Red List (DoE 2015)

The IUCN Red List also lists the Estuary stingray (*Dasyatis fluviorum*) as vulnerable, and the Blacktip reef shark (*Carcharhinus melanopterus*) and the Bull shark (*Carcharhinus leucas*) as near threatened (IUCN 2015). Overall, information on fish assemblages, particularly quantitative data, is limited, including information supplied from recreational fishers.

The only organisms listed as near threatened or endangered, collected by gill net or observed during the two surveys campaigns, were the Blacktip reef shark (*Carcharhinus*

melanopterus), which is listed as near threatened on the IUCN Red List, and the Green sea turtle (*Chelonia mydas*), which is listed as Vulnerable under provisions of the EPBC Act but endangered on the IUCN Red List (DoE 2015).

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Appendix J2
Reef and Nekton Baseline Data Collection

Facing Island Reef Surveys

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Synopsis: An assessment of reef communities adjacent to Facing Island, located near Gladstone on the central Queensland coast.

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Executive Summary

Background

An extensive network of nearshore reefs occurs on the eastern side of Facing Island. This reef system is located more than 5 km from the East Banks Spoil Disposal Site (EBSDS). To meet conditions of the sea dumping permit for maintenance material, GPC monitors the condition of reef assemblages at Facing Island every five years. Monitoring tests the impact hypothesis that maintenance dredging activities have not affected adjacent reef communities. Reefs in the study area were previously surveyed in 2011, 2012, 2013 and 2014, thereby providing a (recent) historical data-set to assess changes over time.

The overall aim of this study was to characterise patterns in the structure and condition of benthic reef communities along Facing Island and 'control' sites at Rundle Island, and trial the use of various technologies for future monitoring studies.

Approach

Three sampling techniques were used in the present study:

- using high-resolution multispectral satellite imagery to provide broad-scale mapping of reefs and to investigate opportunities for more frequent monitoring
- quantitative sampling of epibenthic reef communities – diver-based sampling to characterise the benthic community structure and condition of reefs for comparisons with previous surveys.
- trial 3D photogrammetry mapping – diver-based sampling to construct 3D images of the seafloor, which was undertaken for the purposes of stakeholder engagement and as a potential monitoring tool.

Reef Mapping

Remotely-sensed data based on high-resolution multispectral satellite imagery greatly improved the accuracy of reef extent. In total, 21.71 km^2 of reefs were mapped, compared with 13.90 km^2 that were digitised in the previous mapping using aerial imagery alone. This was due to better resolution of the interface between deep reef and sands using remote sensing data.

The multispectral data also provided a basis for classifying the broad-scale distribution of dominant reef communities. The classification was most successful east of Facing Island, followed by the Seal Rocks Reef area, and was least successful at Rundle Island. This was likely due to differences in optical water types, especially turbidity conditions. This trial has provided improved habitat mapping at Facing Island and Seal Rocks but suggests that object-based classification of water-column corrected high-resolution multispectral data would be required for objective change detection over broad spatial scales.

Reef Community Monitoring

The results of the present study indicate that there were strong spatial gradients in epibenthic community structure and condition throughout the study area. The sites differ in their exposure to anthropogenic and catchment disturbance, and the revised design attempted to partition these processes as much as practical. Revised treatments include the harbour entrance (potentially affected by catchment processes and dredging), eastern Facing Island (potentially affected by dredged material placement), and Rundle Island (serving as a broad-scale environmental control). In summary:

- Hard coral cover was greatest at Rundle Island and sites along the eastern side of Facing Island, and the lowest cover was recorded at sites in the harbour entrance treatment.
- Macroalgae cover showed an inverse relationship to coral cover – sites with low coral cover had the highest algae cover.
- Harbour entrance sites had greatly reduced in coral cover between 2013 and 2014, and do not appear to have recovered substantially between 2014 and 2018.
- While the spatial patterns in hard coral and macroalgal cover are consistent with catchment disturbances, the relative influence of dredged material placement impacts cannot be completely partitioned. However, based on dredge plume monitoring, modelling studies, and the 2018 cover data, it is considered extremely unlikely that sediment plumes created by maintenance dredging are driving these spatial patterns. Both monitoring and modelling indicate that sediment plumes created by material placement are short term features (measurable for < 1 hour) that do not have significantly large enough concentrations to impact reef communities.
- A large degree of uncertainty exists regarding temporal changes in community mainly to do with low levels of replication and a lack of raw data. Future monitoring should consider the use of photo-quadrat data as the unit of replication, rather than transects to improve the potential to detect changes. This can be done with the raw data collected by BMT in 2018, and using Gladstone Healthy Harbour Partnership (GHHP) data, provided GHHP increases point ID frequency above 20.

3D Imaging

Four 3D models were produced over 100 m^2 sites using diver-based photogrammetry techniques. They allow the user to observe the site from a variety of perspectives in space. The technique potentially allows visualisation of small changes in benthic cover over a reasonably large area that may not be encompassed in point-based or line-intercept monitoring approaches. Observing orthomosaics side by side provides an opportunity to assess cover without the difficulties of measurement in a highly rugose 3D environment. Benthic cover classifications could be performed over the orthomosaics to determine percent cover of particular groups using image analysis software.

The disadvantages of this form of monitoring are that the ability to build these models is underpinned by the heterogeneity of the substrate. Substrates with a highly repetitive pattern, or that move through time are difficult to match using pixel matching algorithms. These limitations mean that data collection and model outputs are vulnerable to missing data. The risk of missing data increases with substrate heterogeneity and turbidity, as capture frames become smaller with less overlap, and fewer features for pixel matching algorithms.

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1 Introduction

1.1 Background

1.1.1 Project Need

The East Banks Sea Disposal Site (EBSDS) has a long history of use for the disposal of uncontaminated capital and maintenance dredged material, and is currently used by Gladstone Ports Corporation (GPC) exclusively for the disposal of maintenance dredged material. Fine sediments within EBSDS are re-mobilised and transported by waves and currents, and as such functions as a dispersive site.

The closest coral reef receptor to the EBSDS is the network of nearshore reefs located on the eastern side of Facing Island. This reef system is located more than 5 km from the EBSDS (Figure 1-1).

The Port of Gladstone sea dumping permit for maintenance material prescribes a Long Term Monitoring and Management Plan for Maintenance Dredging Sea Disposal (2015-2018). The reef monitoring aspect of this plan seeks to test the impact hypothesis that Maintenance dredging activities do not result in long-term changes to adjacent reef communities. This is established by reef condition surveys conducted every 5 years in accordance with Technical Advisor Consultative Committee recommendations.

The study area for this project includes reefs located on the eastern side of Facing Island and Rundle Island to the north of Facing Island (Figure 1-1). Some of the reefs in the study area were surveyed in 2011, 2012, 2013 and 2014; providing a (recent) historical data-set to assess changes over time. The 2013 data (Sea Research 2013) was the most recent study to include all monitoring program sites. These reef systems were found to support rich and abundant epibenthic communities comprised of hard and soft corals, macroalgae and a wide range of other invertebrate fauna (Sea Research 2013; BMT WBM 2014).

1.1.2 Aim and Objectives

The overall aim of this study is to characterise patterns in the structure and condition of benthic epibenthic reef communities along Facing Island and ‘control’ sites at Rundle Island. The specific objectives of this study were to:

- compare spatial patterns in the structure and condition of epibenthic reef communities at Facing Island and Rundle Island as a part of 5-yearly compliance monitoring
- assess changes in epibenthic reef communities using sea floor imagery at representative sites over time.
- based on the above, describe spatial and temporal patterns in within the context environmental variability and potential human impacts.
- Build upon the reef mapping work of BMT WBM (2014), by using remote sensing techniques to describe the distribution of nearshore reefs at Facing Island.
- trial remote sensing and 3D mapping technologies as ways of performing more accurate and regular monitoring of benthic reef communities.

Title:

Study Area and Previous Reef Mapping

Figure: 1-1

Rev: A

BMT endeavours to ensure that the information provided in this map is correct at the time of publication. BMT does not warrant, guarantee or make representations regarding the currency and accuracy of information contained in this map.

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Approx. Scale

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1.2 Monitoring Design Considerations

Effective environmental monitoring and interpretation relies on an understanding of natural gradients, confounding processes, as well as the intensity, duration, and geographical extent of the putative anthropogenic impacts to be assessed. In this case, putative anthropogenic impacts include plumes generated by material placement and resuspension from the EBSDS. These issues need to be considered in the design and interpretation of the monitoring program, and are discussed in more detail below.

1.2.1 Reef Locations and Physical Setting

Prior to BMT WBM (2014), the most comprehensive spatial layers describing reefs within Port Curtis and surrounding Facing Island was the Great Barrier Reef Marine Park's Google Earth layer for reefs of the marine park (Lawrey and Stewart 2016). This area was updated by BMT WBM (2014) based on manual digitisation from aerial imagery and ground surveys. Maps from the two data sources are shown in Figure 1-1. *Note: The present survey updated the reef mapping based on interpretation of 10 m resolution multispectral Sentinel-2 imagery (see Section 3.1).*

The existing (2014) mapping illustrates that an extensive reef system occurs along almost the entire length of the eastern side of Facing Island. Smaller offshore reefs systems occur to the south (Seal Rocks) and north (Rundle Island) of Facing Island. These reef systems differ greatly in terms of their physical setting, including reef size, water depth, geological setting, wave exposure, and hydrodynamic setting (exposure to currents). These reefs also have different water quality characteristics and vulnerabilities to flood plumes, as described below.

1.2.2 Water Quality Characteristics

BMT's hydrodynamic model provides a basis for predicting suspended sediment concentrations associated with dredged material placement at the EBSDS (initial plumes and subsequent remobilisation), and concentrations of suspended sediment and salinity during ambient conditions and major flood events. A summary of relevant findings from various technical reports is provided below.

Dredged Material

Sediment movements associated with dredged material placement at the EBSDS was simulated for placement associated with the maintenance dredging, where 340,000 m^3 of material would be placed on the EBSDS (BMT WBM 2017). The 95th percentile exceedance plot shows where concentrations of total suspended solids (TSS) would be exceeded for the worst 4-day period(s) in the simulation (Figure 1-2). This plot shows that plumes emanating from the EBSDS are generally low in TSS concentration, with the worst-affected reef being those surrounding Sable Chief Rocks (6-7 mg/L TSS). Reefs between Sable Chief Rocks and Gatcombe Head would experience 5 mg/L TSS or less, and North Point Reef, near North Entrance, would experience 2-3 mg/L TSS peak levels.

The increase in TSS due to spoil placement was relatively minor compared to ambient wind-generated resuspension and TSS released by flood plumes (BMT WBM 2017).

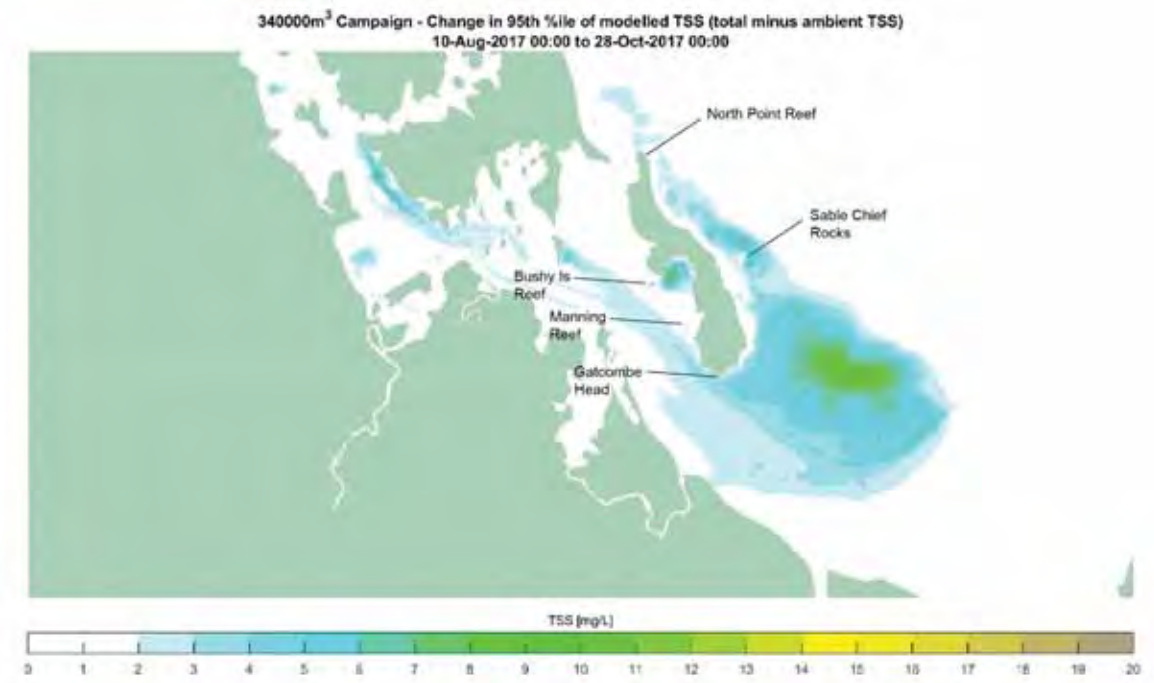


Figure 1-2 95th percentile TSS for a 2.5 month simulation of spoil placement at the EBSDS

The reef remediation prioritisation study (BMT WBM 2014), developed a catchment model for integration with the Port Curtis hydrodynamic model. This was used to hindcast the changes in turbidity and salinity that may have affected corals during the 2013 flood event. Total suspended solids generated from major floods are orders of magnitude higher than disposal plumes associated with dredged maintenance material disposal.

The effects of the simulated 2013 flood were consistent with catastrophic impact to all reefs west of Gatcombe Head and inside North Entrance, attributable to drastic reductions in salinity. The 2013 model showed significant bodies of freshwater developed west of Facing Island and a plume with salinity well below 20 ppt extended into Rodds Bay and over Seal Rocks reefs. Salinities of approximately 15 ppt or less were experienced for three days in the cluster of reefs around North Entrance, while Bushy Island and Manning Reef experienced salinities less than 10 ppt. Measured time-series data from an instrument located between Bushy Island and Manning Reef, showed that salinity fell below 5 ppt at the peak of the 2013 event.

Berkelmans *et al.*, (2012) suggested a salinity dose-time threshold for acroporid corals, based on observed responses to the Fitzroy River 2010-2011 flood plume and its effects on reefs in the Keppel group. The Keppel reefs are the nearest significant inshore coral community to Port Curtis, and acroporid corals (among the more sensitive genera) have a dose-time linear threshold of 22 PSU (roughly equivalent to PPT) for three days grading to 28 PSU over 16 days (Berkelmans *et al.,* 2012).

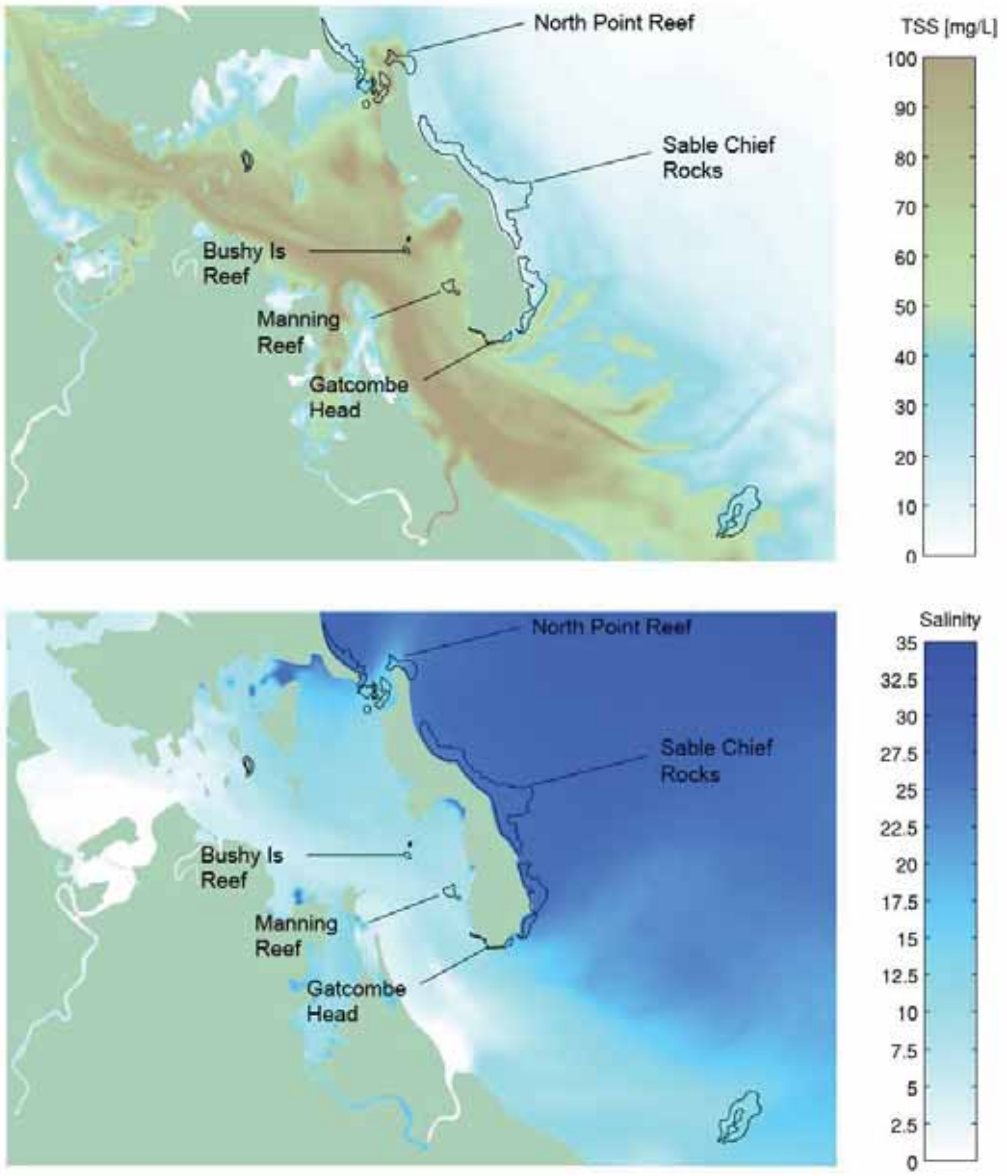


Figure 1-3 Hindcast median TSS (above) and 95th percentile salinity (below) from the 2013 floods (BMT WBM 2014)

BMT visited a selection of sites from the original monitoring program in 2014, but not all 2013 monitoring sites, because the objective was to prioritise remediation sites, not execute the monitoring program. The remediation prioritisation study identified major declines in coral cover between 2013 and 2014, and also between 2009 and 2014, using data collected at sites not included in the monitoring program from BMT WBM (2009). Sites that experienced major reductions in coral cover fell within the Berkelmans *et al.,* (2012) salinity threshold, applied to the 2013 model hindcast. The loss of hard coral was likely the result of freshwater flows from the Boyne and Calliope Rivers, leaving parts of eastern Facing Island relatively unaffected.

Thermal Bleaching

Reefs in the monitoring program are also at risk of thermal bleaching, like the rest of the Great Barrier Reef (GBR). Degree heating weeks (DHW) showing the number of weeks that the summer average sea surface temperature were exceeded for the three world-scale mass-bleaching events are shown in (Hughes *et al.,* 2017).

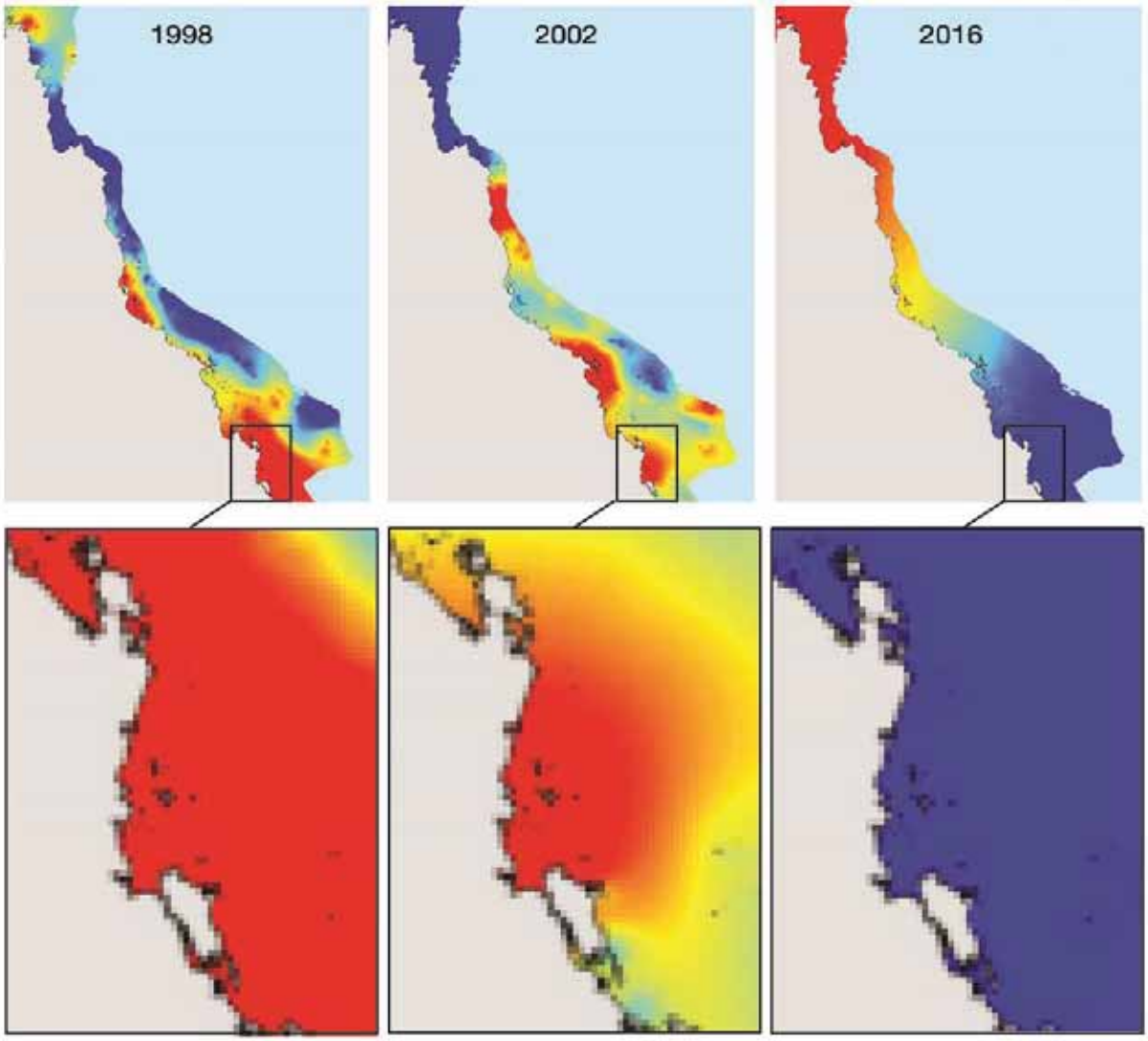


Figure 1-4 Spatial heat stress for bleaching events in 1998 (left) 2002 (middle) and 2016 (right). Dark blue shading indicates 0 Degree heating weeks (DHW), and red is the maximum DHW for each year (7, 10 and 16, respectively). Hughes *et al.,* (2017)

The study area is shown as insets the bottom of each pane in Figure 1-4. The reefs within the monitoring program have experienced variable temperature stress in each of these events, with blue shading indicating zero DHW, and red shading being the maximum DHW for each bleaching event. Other colours are shown as a continuous colour spectrum between the zero and maximum DHW extremes. In the 1998 event, all reefs experienced extreme warming, in the 2002 event there was a north-south gradient in temperature stress, and in 2016, none of the reefs experienced sustained

warming above the average sea surface temperature. This shows that unlike catchment impacts, where there are clearly defined spatial gradients in impact, thermal stress varies inconsistently, and is often consistent throughout the study area.

1.2.3 Original Monitoring Program Design

The past monitoring design (last implemented by Sea Research, 2013) consisted of a control-impact framework, with sites east of Facing Island considered to be 'impact', and sites at Rundle Island considered to be 'control' (Figure 1-5). Putative impact was considered the movement of dredged material placed on the EBSDS (Figure 1-1). At each site, four line-intercept transects were taken over 50 m belts, giving four replicates per site. The design had seven impact sites and three control sites.

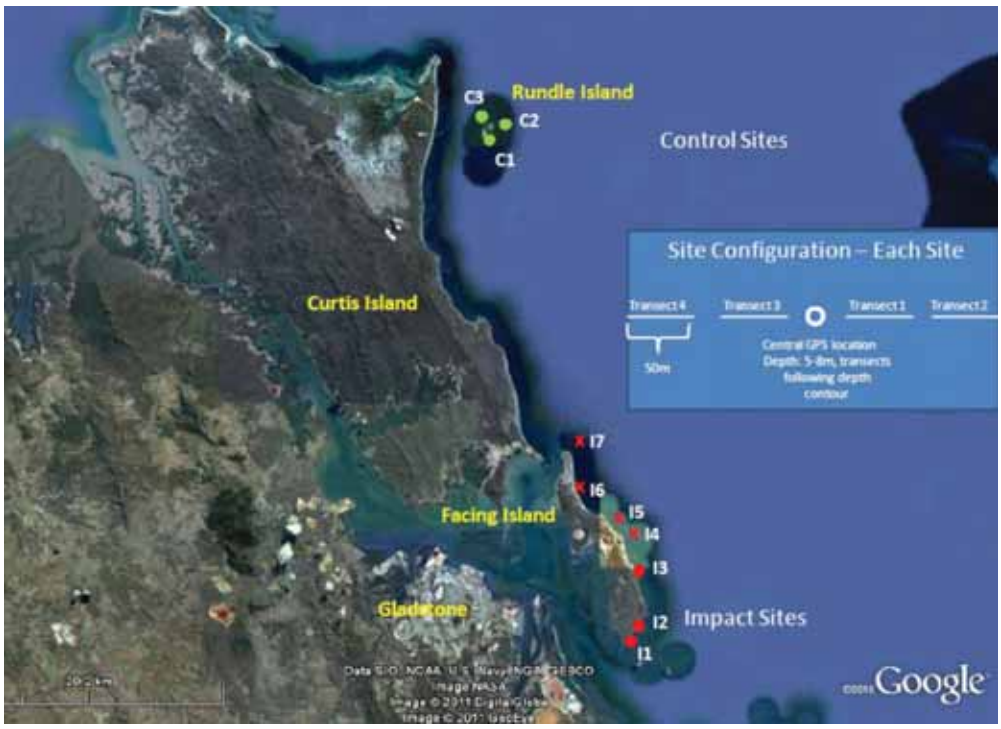


Figure 1-5 Monitoring design last implemented by Sea Research (2013)

Ideally more control sites at alternative locations would be employed to better encompass site-specific variability, but suitable sites (located inshore, with hard coral communities, at similar depths) were not available. Alternative sites investigated on the east coast of Curtis island were dominated by sponges and not appropriate controls for eastern Facing Island (Sea Research 2012) and eventually removed from the monitoring program.

1.2.4 Revised Nomenclature and Design

The original design included three sites in the 'impact' treatment that experience chronic and acute water quality stress from turbid (and occasionally fresh) water from Port Curtis. As described earlier (Sea Research 2013, BMT WBM 2014), these benthic communities also have very different composition, and are dominated by different species subject to prevailing water quality conditions.

For these reasons, a revised design that includes three treatments: harbour entrance; outer Facing Island; and Rundle Island is presented in Table 1-1 and Figure 1-6.

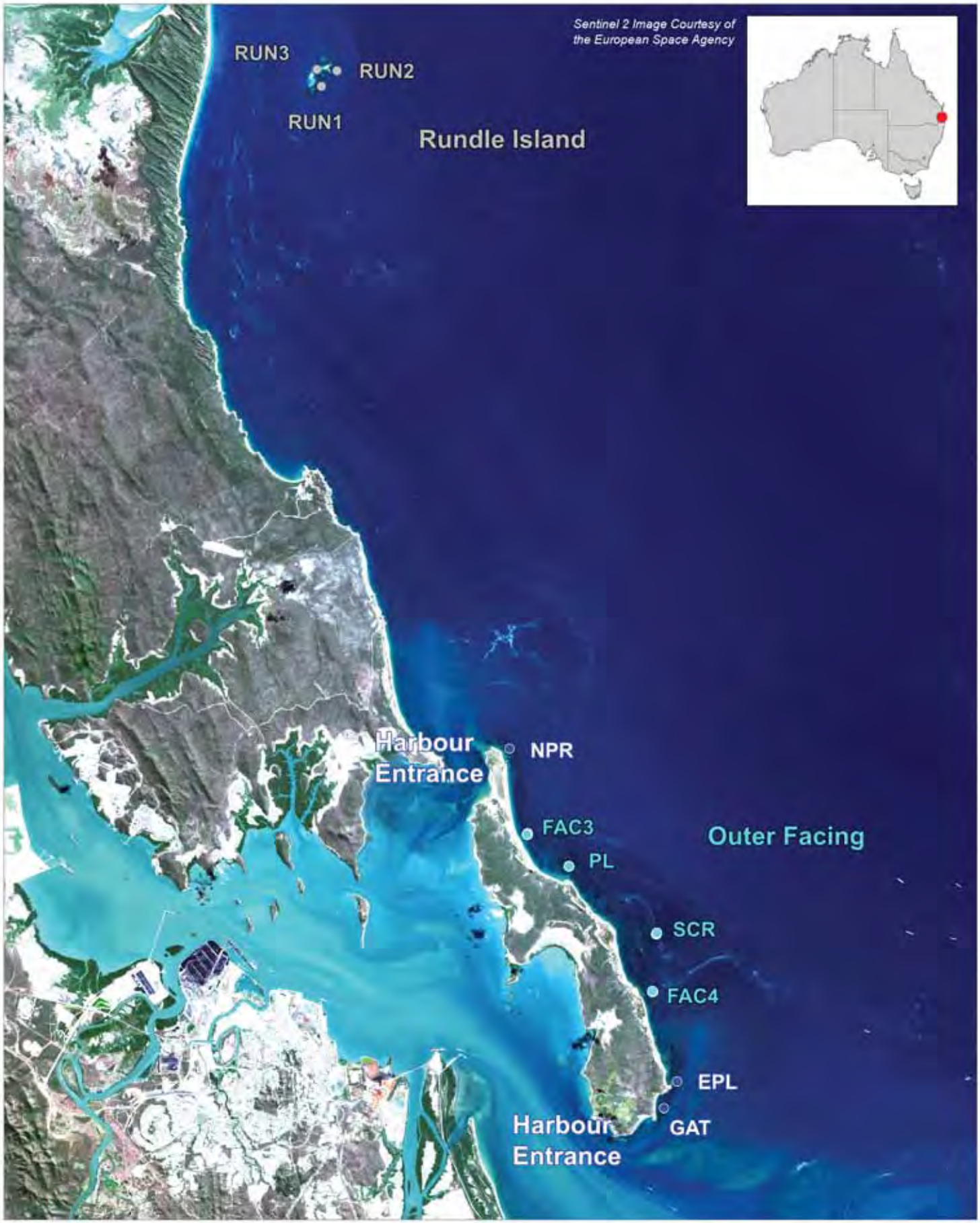
Ideally the study terminology would follow the naming convention applied by the GBRMPA spatial layer. However, individual reefs mapped by GBRMPA form a contiguous fringing reef in front of Facing Island, while other monitoring sites exist over reefs that are not mapped or named. The site labels adopted in Table 1-1 endeavour to follow GBRMPA terminology where possible within the limitations stated above.

The revised design groups sites into the following treatments:

- Background sites (RUN1, RUN2, and RUN3 in Figure 1-6) - Reefs at Rundle Island represent control sites in the context of assessing potential dredging impacts. These sites are also not as strongly influenced by catchment runoff as the harbour entrance treatment.
- Harbour entrance sites (NPR, EPL, and GAT in Figure 1-6) – these sites are more strongly influenced by fluvial discharges than the other two treatments. Two of the sites in this treatment (GAT, EPL) are also closest to the EBSDS.
- Outer Facing Island (FAC3, PL, SCR, and FAC4 in Figure 1-6) – these sites are not predicted to be grossly affected by fluvial discharges and are >10 km from EBSDS, representing putative material placement impacts.

Table 1-1 Revised and previous designs, and terminologies

BMT WBM 2014 site (GBRMPA reef Name)	2014 Data Availability	Sea Research Sites (2013)	2018 site terminology	2018 Treatments
Gatcombe East (Facing Island #6)	Assessed	Impact 1	**GAT** (Gatcombe)	harbour entrance
East Point Ledge (Facing Island #5)	Assessed	Impact 2	**EPL** (East Point Ledge)	harbour entrance
Facing Island #4 (Facing Island #4)	Assessed	Impact 3	**FAC4** Facing Island #4	outer Facing
Sable Chief Rocks Reef (Sable Chief Rocks)	Assessed	Impact 4	**SCR** (Sable Chief Rocks)	outer Facing
Pearl Ledge (Facing Island #3)	Assessed	Impact 5	**PL (**Pearl Ledge)	outer Facing
North Facing Island (not charted or named)	Not assessed	Impact 6	**FAC3** (Facing Island #3)	outer Facing
North Entrance (North Point Reef/ Facing Island #1)	Not assessed	Impact 7	**NPR** (North Point Reef)	harbour entrance
Rundle Island 1 (Rundle Reef)	Not assessed	Control 1	**RUN1** (Rundle Reef)	Rundle Island
Rundle Island 2	Not assessed	Control 2	**RUN2** (Rundle Reef)	Rundle Island
Rundle Island 3	Not assessed	Control 3	**RUN3** (Rundle Reef)	Rundle Island



Title:

Sites and Treatments

Figure: 1-6

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2 Methods

Three sampling techniques were used in the present study:

- quantitative (transect-based) sampling of epibenthic reef communities as a part of 5-yearly compliance monitoring
- broad-scale mapping of reefs using high-resolution multispectral satellite imagery
- trial photogrammetry to produce 3D models.

Transect-based methods were used to perform 5-yearly reef monitoring, while remote sensing and 3D modelling methodologies were used to provide more accurate reef mapping, and as trial methodologies for potential for more regular monitoring.

2.1 Remote Sensing Analysis

Multispectral remotely sensed imagery was analysed by calculating the benthic reflective index (BRI), and the depth invariant index (DII) for Sentinel-2 scenes encompassing the study area. The depth invariant index has been used extensively for benthic classification of coral reef environments, while the benthic reflective index can have better classification efficacy in turbid-water environments (Sagawa *et al.,* 2010). Scenes of the study area from August 11th and September 15th 2017 were downloaded and processed, with the latter scene offering the best visibility, but poorer atmospheric conditions. Of the imagery available in the last two years, the two scenes chosen had the best combination of water and atmospheric clarity.

The DII (Lyzenga 1981) was calculated from atmospherically corrected, de-glinted images. Scenes were atmospherically corrected using the Sentinel application platform (SNAP 6.0) and de-glinted using the procedures outlined in Hedley *et al.,* (2005). Shallow-water signals for use in both procedures were obtained from sandy areas east of Facing Island known to be devoid of marine plants or reef based on ground-truthing in 2014 and 2018. The BRI was calculated using a 10 m resolution bathymetric grid, developed by BMT and GPC for hydrodynamic modelling purposes. For the DII, the coastal band 1 (0.443 µm central wavelength, 60m resolution) was resampled to 10 m and used with the blue band (0.490 µm) and green band (0.560 µm) as band pairs to provide maximum signal penetration in turbid waters.

The DII was visualised as a, false colour RGB image with the B1-B2 band combination as the red channel, the B2-B3 combination as the green channel, and the B1-B3 combination as the blue channel. Results for the August capture were not presented or further processed due to poor visibility, particularly at North Entrance and Gatcombe Head, where reefs were partially or fully obstructed by turbidity.

The results of the DII and BRI indices were classified using Maximum Likelihood methods for pixels (Foody *et al*., 1992) in Arcmap 10.5. Polygon areas over dense *Acropora* stands at Facing Island and Rundle Island were used as training polygons for coral habitat, while macroalgal dominated reef at Gatcombe Head, North Entrance, and Seal Rocks Reef were used to train macroalgal reef polygons. Habitat classification signature files were used to classify the BRI and DII indices. Classifications for the DII were attempted with five band pairs, and three band pairs; both of which

produced classification results inferior to the BRI. Therefore, the classification based on BRI is presented in section 3.1.

2.2 Coral Monitoring Transects

2.2.1 Data Collection

The 10 sites surveyed by Sea Research (2013) were revisited in the present survey (Figure 1-6) between May 31st and June 4th, 2018. The 2013 survey used a line intercept approach over a 50 m transects, with four replicate samples collected at each site. The present survey was designed to be compatible with the original sampling program, but able to be improved in the future by using photo-quadrats (taking random photos along a fixed transect) to increase sample replication and sample precision. Photo-quadrat methods are also similar to those used in the Long-term Inshore Reef Monitoring Program by AIMS and are compatible with the transect data collected by the GHHP.

The start of each transect location was marked with a shot line and located using a waterproof GPS unit tied to the diver's surface float. Start and finish points for each transect were determined from point tracking on the diver tether.

Each location was surveyed along the -3 to -5 m lowest astronomical tide (LAT) contour or as appropriate given site conditions. Transects occasionally became shallower or deeper while following available reef habitat. A 20 m transect tape was lain on the substrate, and a diver swam over the substrate underneath using paired high-definition underwater cameras capturing still images (every second) and video. Distance from the tape varied depending on visibility and macroalgal canopies encountered; however, a distance of 15-50 cm was generally adhered to as per procedures used by AIMS and the Hawaii Coral Reef Assessment and Monitoring Program. Incidence of coral disease and numbers of coral recruits were quantified within 34 cm of the transect tapes from video footage taken over the entire transect.

All diving was undertaken by BMT marine scientists with ADAS commercial dive qualifications aboard the 2c certified vessel *Candela*.

2.2.2 Data Analysis

The video footage was used to assist in identification, and for coral health, recruitment counts, and general observation purposes. Still imagery was quantified to determine benthic cover and community composition. Coral Point Count was used to quantify photo-quadrats, with coral and algae identified to the lowest practical level (typically genus). Twenty-point identifications were made per photo. Photos were selected randomly (10 per transect), giving a total of 200 point IDs per transect, or 40 photo-quadrat samples per site.

Raw data from Sea Research (2013) were unavailable; therefore, qualitative temporal comparisons between 2013 and 2018 were made. Approximate means and standard errors for sites were digitised from the original report and re-apportioned into sites within the 2018 treatment groups by averaging means and errors in the new treatment groups. Transect-level means and standard errors were visually compared between the four sites that were surveyed in both 2014 (BMT WBM 2014) and 2018 (present study).

Differences among sites and treatments in 2018 were explored at the photo-quadrat level, to increase replication and improve power. A chi-squared test was conducted to test the differences in coral cover among different sites and treatment groups within a generalised linear model. Patterns in quantitative community attributes (such as cover) using photo-quadrats as replicates were explored using generalised linear modelling in R. Percent cover data converted to proportions best fit the Poisson distribution, and this was used as the error structure in linear modelling with the Chi-squared test statistic. This approach mitigated the need to comply with standard ANOVA assumptions.

Patterns in assemblage structure at different sites were analysed using the non-metric muti-dimensional scaling plots (nMDS), Similarity Percentages (SIMPER), and Analysis of Similarity (ANOSIM) routines using Primer 7 software package. ANOSIM and nMDS routines were used to identify differences in assemblage structure among sites and among treatments. SIMPER and shade plots were used to identify taxa contributing to differences among sites and treatments and to identify possible drivers of these patterns.

2.3 3D Mapping

Three-dimensional mapping of reefs was trialled at three sites; North Point Reef (NPR), Facing Island #3 (FAC3), and Sable Chief Rocks (SCR). Imagery was also collected from Seal Rocks Reef, but was not able to be built into a 3D model due to a dynamic algal canopy that prevented photo-matching during photogrammetry.

At each site, 500-800 georeferenced photos were taken using a wide-angle still camera interfaced with a surface RTK GPS. Photos and positions were built into 3D models using proprietary techniques involving:

- Batch processing of photos to remove poorly focused images.
- Colour correction to bring up red tones and reduce green and blue levels.
- Removal of lens vignetting to homogenise lighting across the field of view.
- Photogrammetry using tie points at precisely known locations.
- Generation of 3D models and orthomosaics from aligned imagery.

3 Results and Discussion

3.1 Remote Sensing

3.1.1 Facing Island

False-colour imagery of the DII at Facing Island shows shallow-water reef communities resolved as bright pink regions (Figure 3-1). Turbidity and other water-column effects make the reef-sand boundary difficult to distinguish just north of Pearl Ledge, but a clear sand (black) versus reef (pink) boundary is present over much of the image. The previous mapping polygons from BMT WBM (2014) and GBRMPA are shown as blue and yellow polygons, respectively.

The delineation between reef edge and sand (dark pixels) is clear along most of Eastern Facing Island area except the area of reef near Pearl Ledge, where water column artefacts are difficult to distinguish from reef. The DII mapping is mostly consistent with the BMT WBM (2014) reef extent, but there are some areas previously mapped as reef that appear to be sand, and there are other areas of reef beyond the 2014 extent that should be included. These differences in extent are presumably methodological. Increases in reef extent include the following areas:

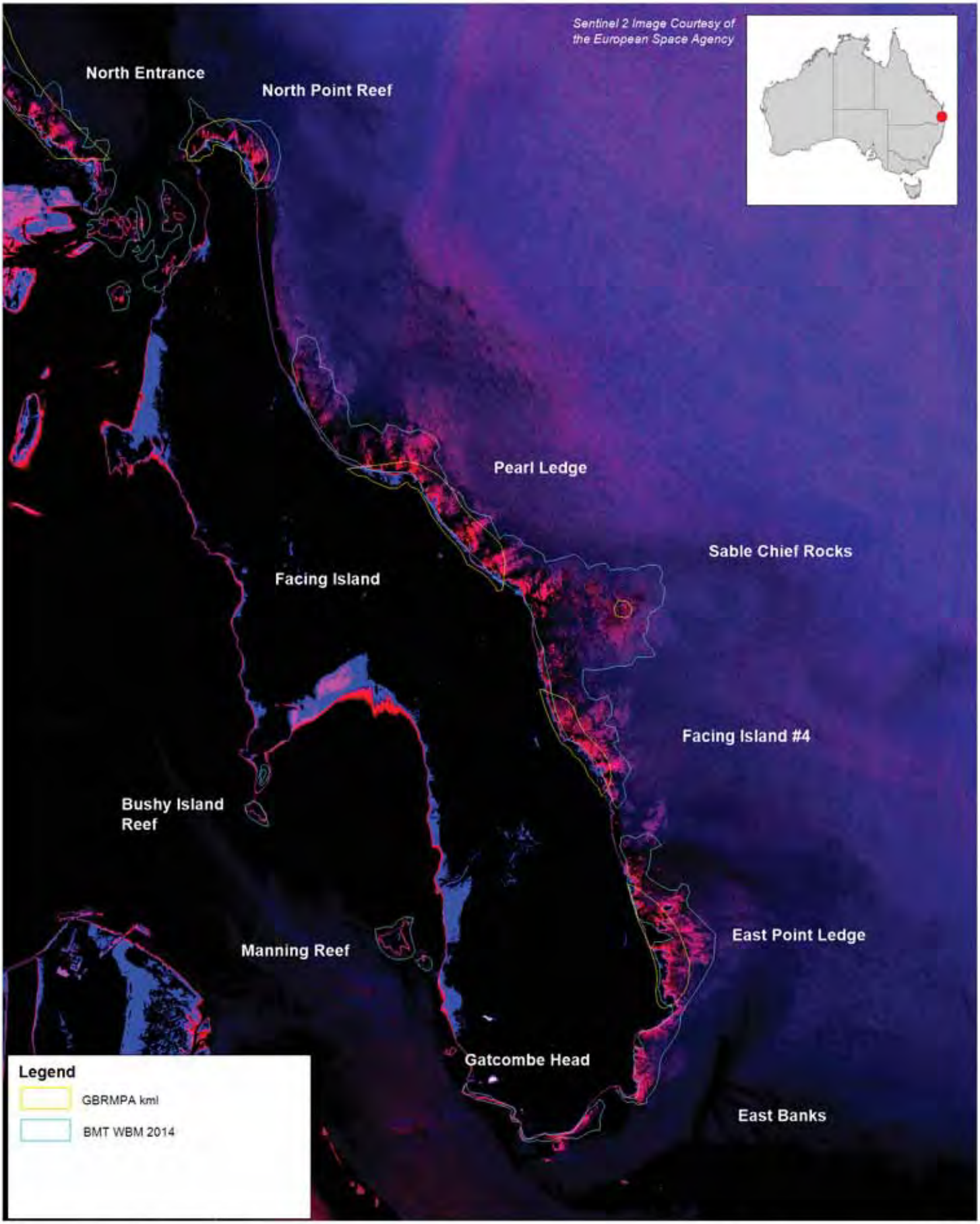
- close to the shore between North Point Reef and FAC3.
- near Pearl Ledge.
- south-east of Sable Chief Rocks.
- between East Point Ledge and Facing Island #4.

Reductions in the BMT WBM (2014) reef extent are between Pearl Ledge and FAC3, near Gatcombe Head, and between Gatcombe Head and East Point Ledge.

The maximum likelihood classification of eastern Facing Island using the BRI has produced a map of coral and macroalgal dominated reefs along its length (Figure 3-2). This map was produced using ground-truthing data where coral was the dominant cover form (polygon areas at Rundle Island and Sable Chief Rocks) and where macroalgae was the dominant cover (reefs at East Point Ledge, and Gatcombe Head). The maximum likelihood method predicts classes based on training samples and does not provide quantitative cover data for coral or macroalgae. The map shows that reefs from East Point Ledge south, and inside North Entrance are macroalgal dominated, while communities between Sable Chief Rocks and North Entrance re dominated by coral.

This map largely agrees with ground-truthing observations conducted in 2014 and 2018; however, several anomalous classifications have been produced, including patches of macroalgal dominated reef inside Port Curtis and patches of coral-dominated habitat on East Banks. The anomalies appear to be the result of turbidity and/or seagrass patches. Reefs such as Manning Reef and Bushy Island are poorly characterised by this classification.

Interestingly, the area of reef between Pearl Ledge (more specifically FAC3) and North Entrance was not ground-truthed in 2014 or 2018, yet both the DII and BRI classifications suggest that reef exists in this area.



Title:

Facing Island Depth Invariant Index

Figure: 3-1

Rev: A

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N

0 1.5 3km

Approx. Scale

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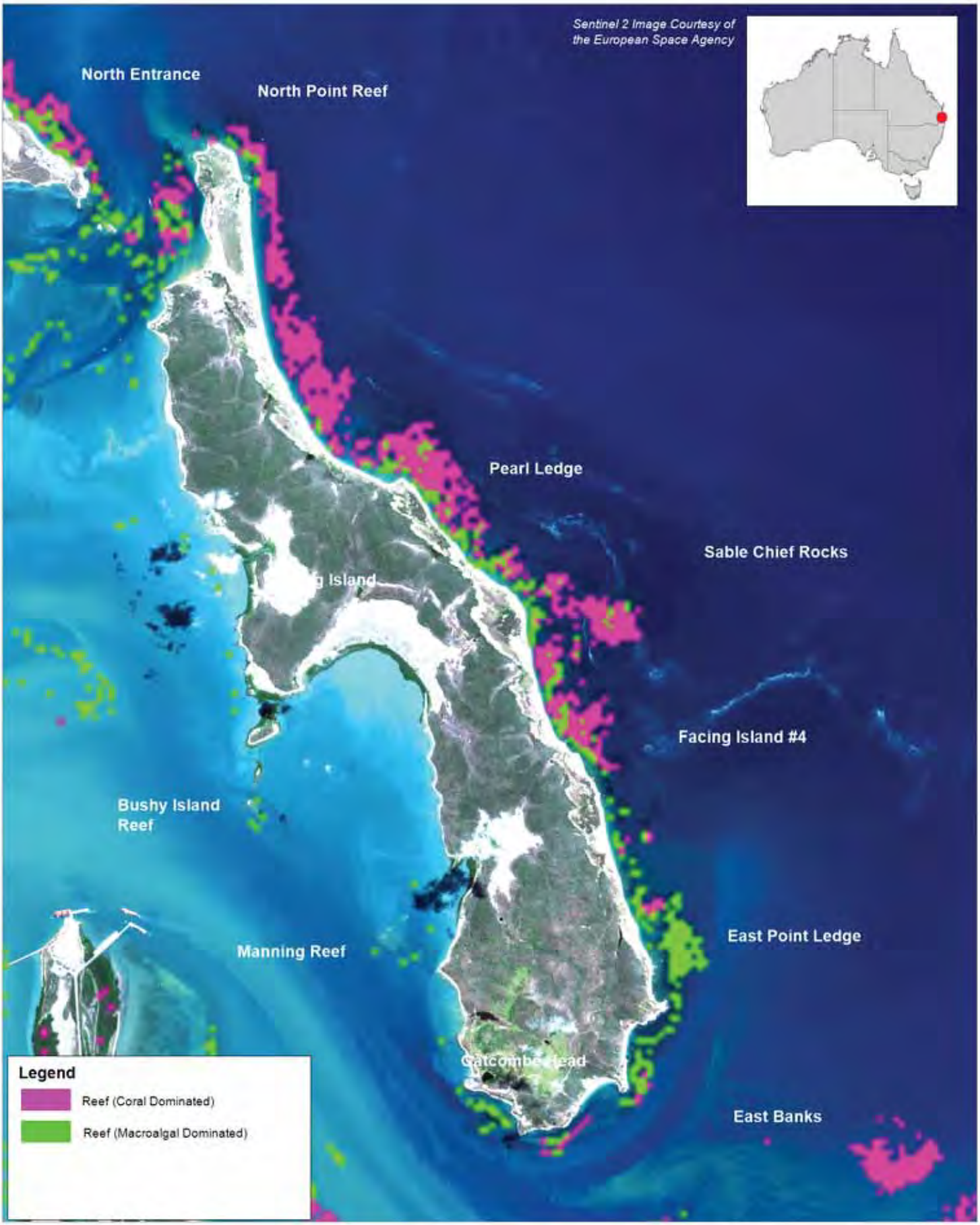

Title:

Facing Island Benthic Reflective Index (Maximum Likelihood Classification)

Figure: 3-2

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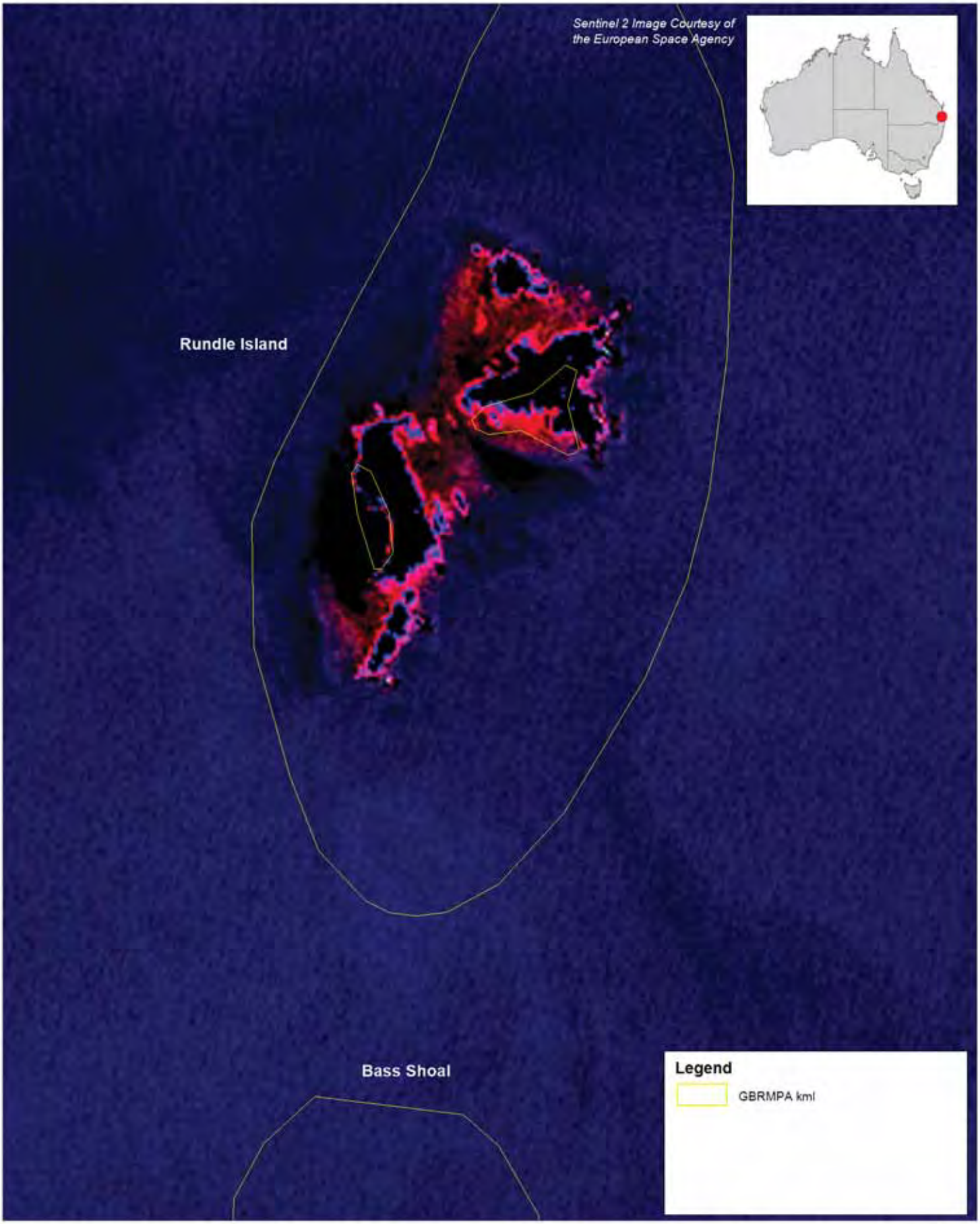
3.1.2 Rundle Island

False-colour imagery of the DII at Rundle Island again shows shallow-water reef communities resolved as bright pink regions (Figure 3-3). This site was not mapped by BMT WBM (2014), but the GBRMPA reef mapping extent is shown as a yellow polygon. Based on ground-truthing, the reef mapping appears consistent with patterns observed on the north-west side of the Island, but has failed to resolve communities deeper than 10 m on the eastern side of the Island. Video transect sites conducted on this side of the Island at RUN 2 do not overlap with the visible area of reef, suggesting that more extensive deep-water communities (>10 m) exist beyond the DII mapping.

The area of Bass Shoals mapped as a large circle to the south of Rundle Island did not show any features consistent with coral at this location.

The maximum likelihood classification of Rundle Island using the BRI has produced a map of coral and macroalgal dominated reefs over some parts of the reef, but mis-classified large sections of reef (Figure 3-4). This map in inconsistent with ground-truthing observations conducted in 2018 and includes only partial classifications of reef habitat and some anomalous classifications of open water.



Title:

Rundle Island Depth Invariant Index

Figure: 3-3

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Title:

Rundle Island Benthic Reflective Index (Maximum Likelihood Classification)

Figure: 3-4

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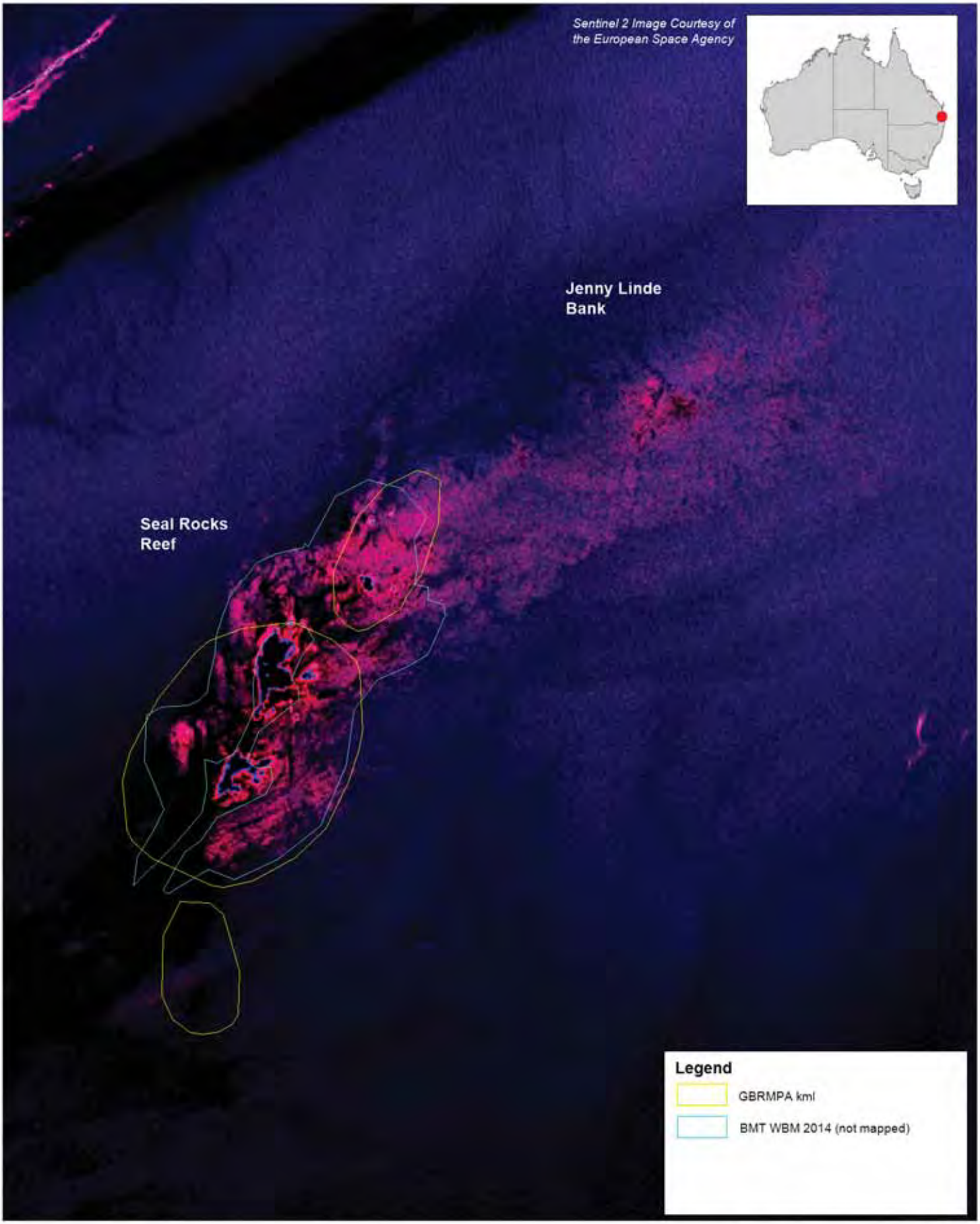
3.1.3 Seal Rocks

False-colour imagery of the DII at Seal Rocks Reef shows shallow-water reef communities resolved as bright pink regions, running between Seal Rocks and Jenny Lind Bank (Figure 3-5). Darker blue areas probably represent areas of sea floor that are not sand (rubble or shell) or possibly water column artefacts. Previous mapping polygons from BMT WBM (2014) and GBRMPA are shown as blue and yellow polygons, respectively. Unlike previous mapping, the DII shows that reef runs continuously between Seal Rocks Reef and Jenny Lind bank, and the two features should probably be classified and managed as one reef.

The delineation between reef edge and sand (dark pixels) is clear along the northern reef edge. The reef edge becomes more diffuse and patchy along the southern edge, indicating either obstruction of the remotely sensed signal by water clarity, or perhaps that the reef is patchy by nature, consisting of smaller bomboras and outcrops. The DII imagery can greatly improve the accuracy of the BMT WBM (2014) reef extent mapping. In particular, the south-western area previously mapped as reef appears to be sand, the area of seafloor between Seal Rocks and Jenny Lind bank should be included as reef extent. These differences in mapped extent are presumed to be methodological mapping differences rather than an increase in reef extent.

The maximum likelihood classification of Seal Rocks Reef using the BRI produced a map of scattered coral- and macroalgal dominated reef along its length (Figure 3-6). The map suggests that reef in the north-east extremity of the feature (near Jenny Lind Bank) is more coral-dominated, while the reef surrounding Seal Rocks Reef is a mixture of substrates. Extensive ground-truthing of Seal Rocks Reef was not part of the present scope of work, but it appears that some anomalous classifications have been produced, including mis-classifications of reef as sandy substrates along the deeper sections of the reef, and parts of the northern reef edge.



Title:

Seal Rocks Depth Invariant Index

Figure: 3-5

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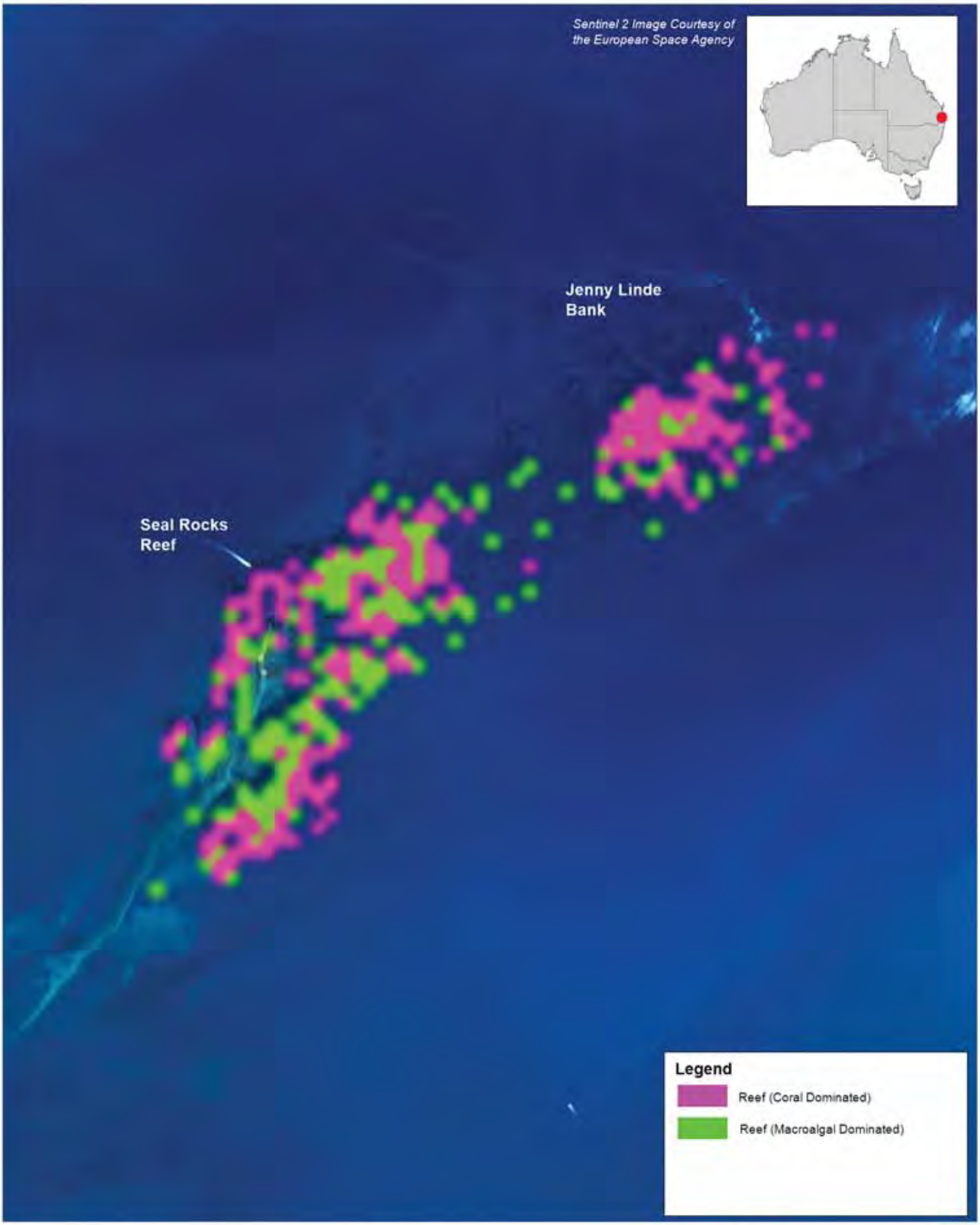

Title:

Seal Rocks Benthic Reflective Index (Maximum Likelihood Classification)

Figure: 3-6

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3.1.4 2018 Revised Reef Layer

The reef layer presented in Figure 3-7 is a revision to the BMT WBM (2014) layer based on interpretation of the DII and BRI indices, and both atmospherically corrected true-colour images. Small reefs along Curtis Island have also been digitised, although not part of the present study. There is no evidence in any of the imagery that the reef situated over East Banks in the GBRMPA layer (Figure 1-1) exists. This area was visited in 2014 and consisted only of sand. The 2018 reef layer represents a total 21.71 Km^2 of reef, compared to 13.90 km^2 mapped in 2014.

Cover types derived using the BRI in this study can be used to map areas of reef dominated by coral or macroalgae. However, this is only considered appropriate at Facing Island; mapping of cover types is considered too anomalous to be of use at Seal Rocks or Rundle Island using Sentinel Imagery, and without dedicated water column removal. For quantitative cover comparisons across the study area, more complicated (and expensive) remote sensing methods involving ultra-high resolution (2 m) imagery and physics-based water column correction should be employed.

Title:

Adjusted Reef Layer 2018

Figure: 3-7

Rev: A

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3.2 Coral Survey Results

3.2.1 2018 Univariate Cover Analysis

Site-level coral cover data from the 2018 survey followed a Poisson distribution and was analysed using a generalised linear model appropriate to this distribution. There was a significant difference between coral cover at different sites (X^2 residual deviance [7, N = 399) = 6980.5; p<0.001]) and between treatment groups (X^2 residual deviance [2, N = 399) = 10632.4; p<0.001]). Greatest cover of live coral was at sites located at Rundle Island and at outer Facing Island sites FAC3 and SCR (Figure 3-8). At a treatment level the highest coverage of coral was at Rundle Island followed by the outer reefs of Facing Island. Sites at the northern and southern extent of Facing Island (harbour entrance sites) which are exposed to tidal currents and turbidity from Gladstone Harbour had the lowest coral cover.

Soft coral was observed at low levels (<5%) across most sites except RUN2 (Figure 3-9). There were no clear patterns in the distribution of soft coral among treatment groups.

Macroalgae was present at all survey sites in moderate abundance with highest cover seen at harbour entrance sites and lowest at Rundle Island sites (Figure 3-10). Generally, total algae cover followed an inverse distribution to coral cover – i.e. typically where macroalgal cover was higher, cover of corals was lower.

The contribution of macroalgae and turfing algae to total algal cover was examined at each site. At sites where coral cover was highest (Rundle Island sites as well as FAC4 and SCR at outer Facing Island) the algal community was dominated by low lying algal turfs (see Figure 3-8 and Figure 3-10). Algal communities at other sites were comprised of both macroalgae and turfing algae. This relationship is discussed further in Section 3.2.2. Dense macroalgae can form large canopies that shade the substrate, potentially outcompeting corals (McCook *et al.,* 2001); however, both macroalgae and turfing algae can inhibit coral recruitment (Arnold *et al.,* 2010). Phase shifts between coral and algal dominated communities have been well documented in the literature and may be caused by a number of ecological processes or environmental conditions (see McManus and Polsenberg 2004) but regardless of the drivers, cover of hard corals and macroalgae are generally inversely related and this was the pattern observed in the present study.

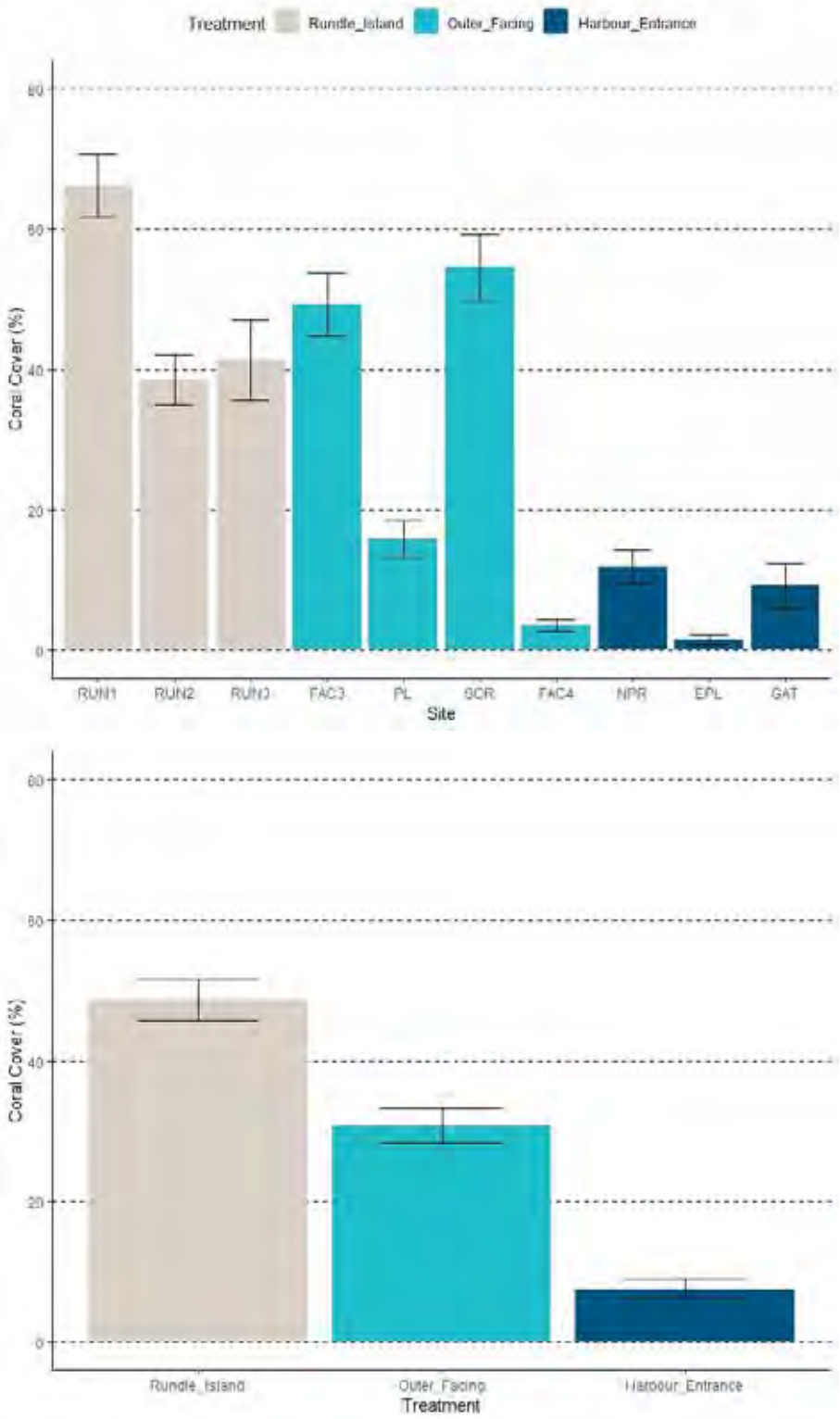

Figure 3-8 Mean coral cover (±SE) by site (upper) and treatment (lower)

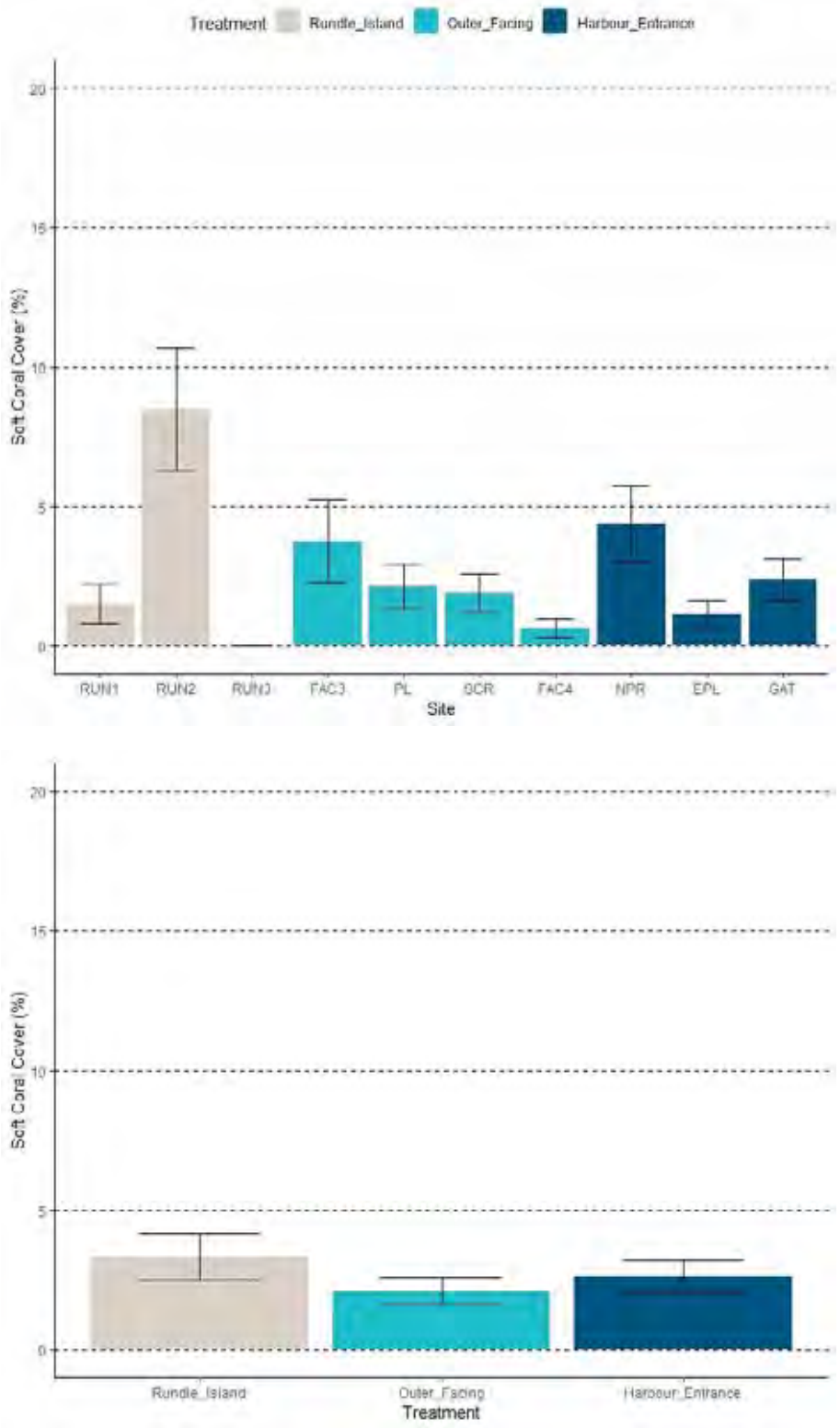

Figure 3-9 Mean soft coral cover (±SE) by site (upper) and treatment (lower)

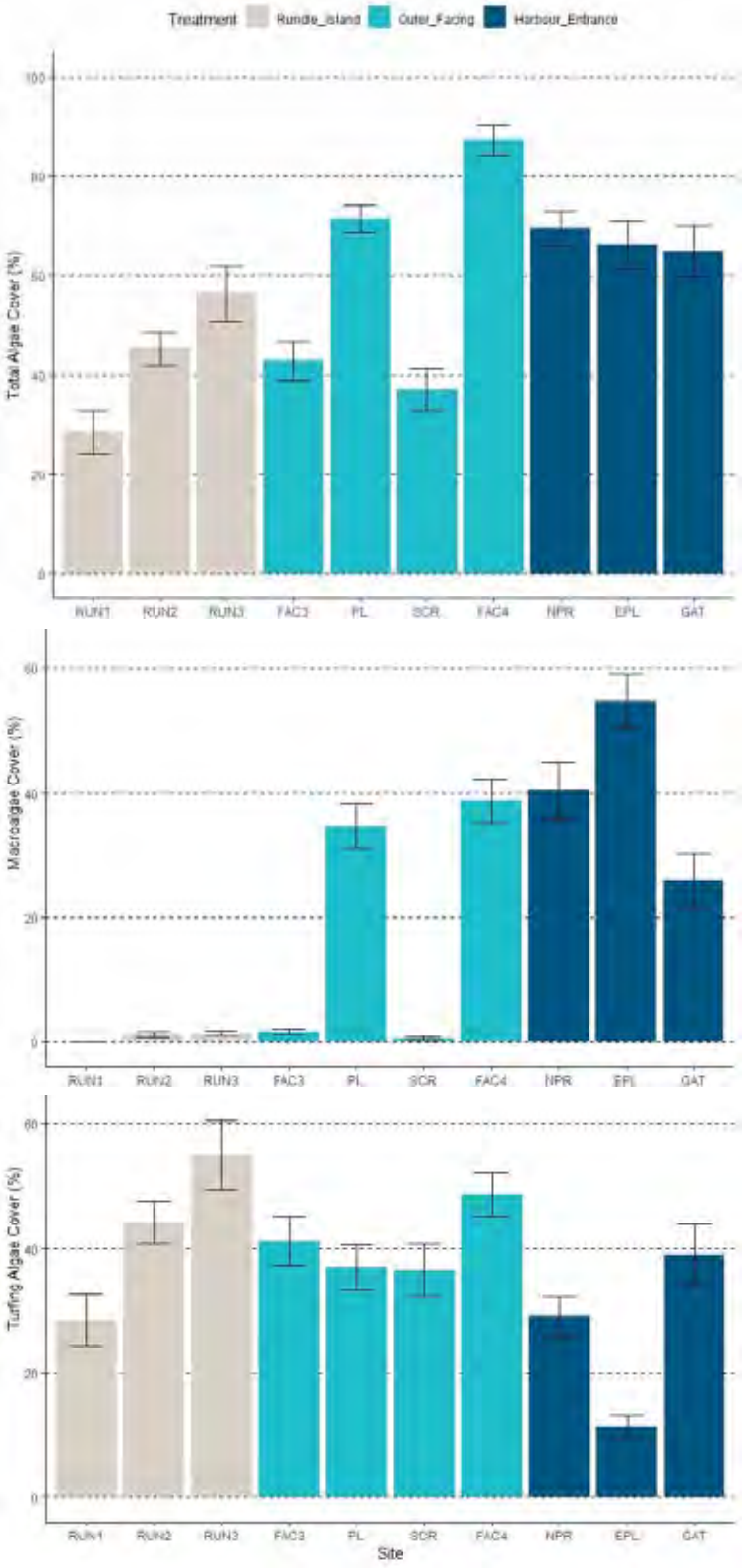

Figure 3-10 mean algae cover (±SE) for total algae (upper) and separated to show cover of macroalgae (middle) and turfing algae (lower)

3.2.2 2018 Multivariate Cover Analysis

Figure 3-11 is an nMDS ordination showing patterns in similarity of benthic communities among sites in two-dimensional space. Samples with similar communities are positioned closer together, while samples with dissimilar communities are positioned further apart. Two communities were distinguished at the 50% Bray Curtis similarity level:

- Rundle Island sites and outer Facing Island sites FAC3 and SCR had similar communities, grouping together at the 50% similarity level on the right side of the ordination. Some sites within this group had a relatively distinct community – most notably sites SCR and RUN1 which were more similar to each other than they were to other sites. Most treatments displayed great within-site variability in communities, often being more similar to other sites than to each other.
- Harbour entrance sites and outer Facing Island sites FAC4 and PL were more similar to one another and were grouped close together on the left-hand site of the plot.

The shade plot in Figure 3-12 provides a graphical representation of the relative abundance of different taxa groups at each site (dark grey = high abundance, white = zero abundance). As discussed above, coral cover was much higher overall at Rundle Island sites, FAC3 and SCR. The shade plot shows that coral cover at these sites was dominated by *Acropora* which was observed in large monospecific stands that were sometimes interspersed with other corals and benthic cover types (Figure 3-13). The corals *Pocillopora* and *Goniastrea* showed similar distributions to *Acropora* while *Porites* and *Montipora* were present more broadly throughout the study sites but were generally more abundant at Rundle Island, FAC3 and SCR. Turfing algae was observed across all sites while bare substrate (Sand/Mud) and the macroalgae *Sargassum* was more common at outer Facing Island sites PL and FAC4 and at harbour entrance sites. The macroalgae *Lobophora* and *Dictoya* showed similar patterns to *Sargassum* in their distribution. These macroalgae species were the most abundant and their distribution is consistent with that for macroalgae discussed above and shown in Figure 3-10. Crustose coralline algae was sparse at all sites but was more common at outer Facing Island and harbour entrance sites and was not widely seen at Rundle Island.

ANOSIM indicated that there were significant differences in communities among sites. Pairwise tests for differences between sites show that most sites were significantly different to each other (see Appendix A for the full test results). Exceptions were sites that did not demonstrate a significant difference (i.e. communities at these sites were very similar) and included sites EPL and GAT (R=0.375; p=0.086); GAT and NPR (R=0.302; p=0.114); NPR and PL (R=0.073; p=0.314) and; RUN1 and SCR (R=0.26; p=0.086).

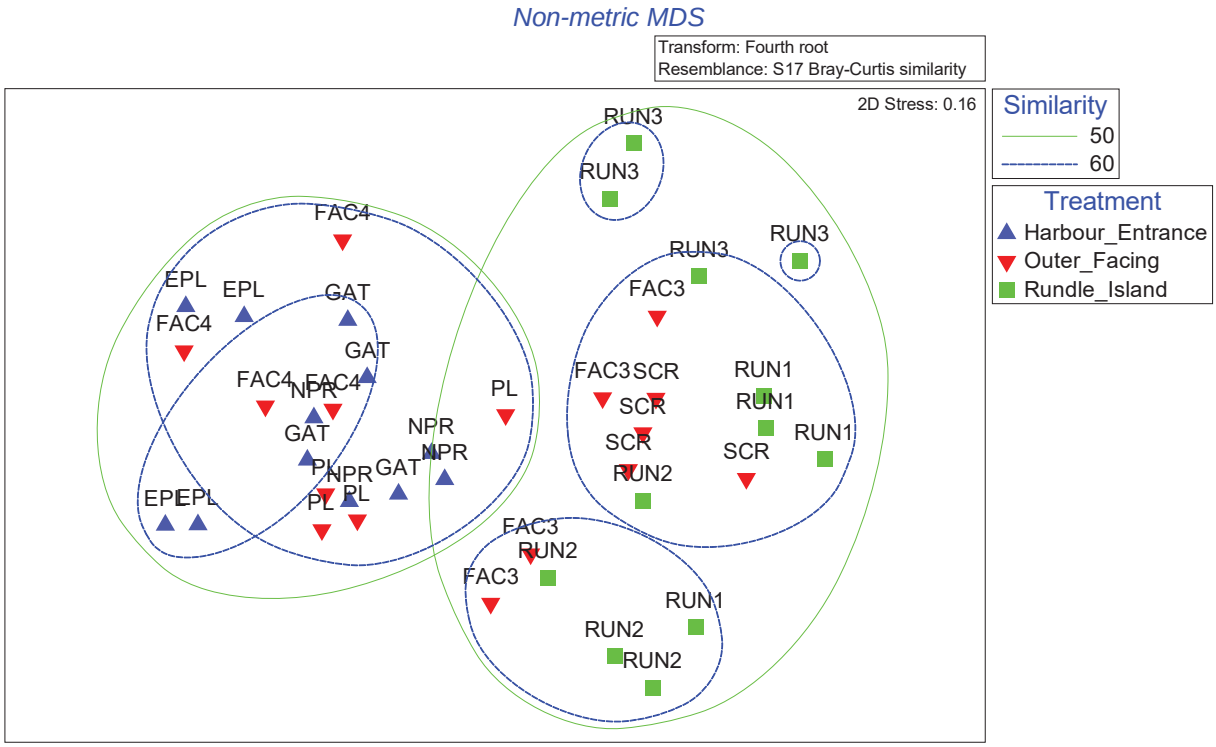

Figure 3-11 nMDS plots showing similarity among benthic communities at different sites, differentiated by treatment groupings

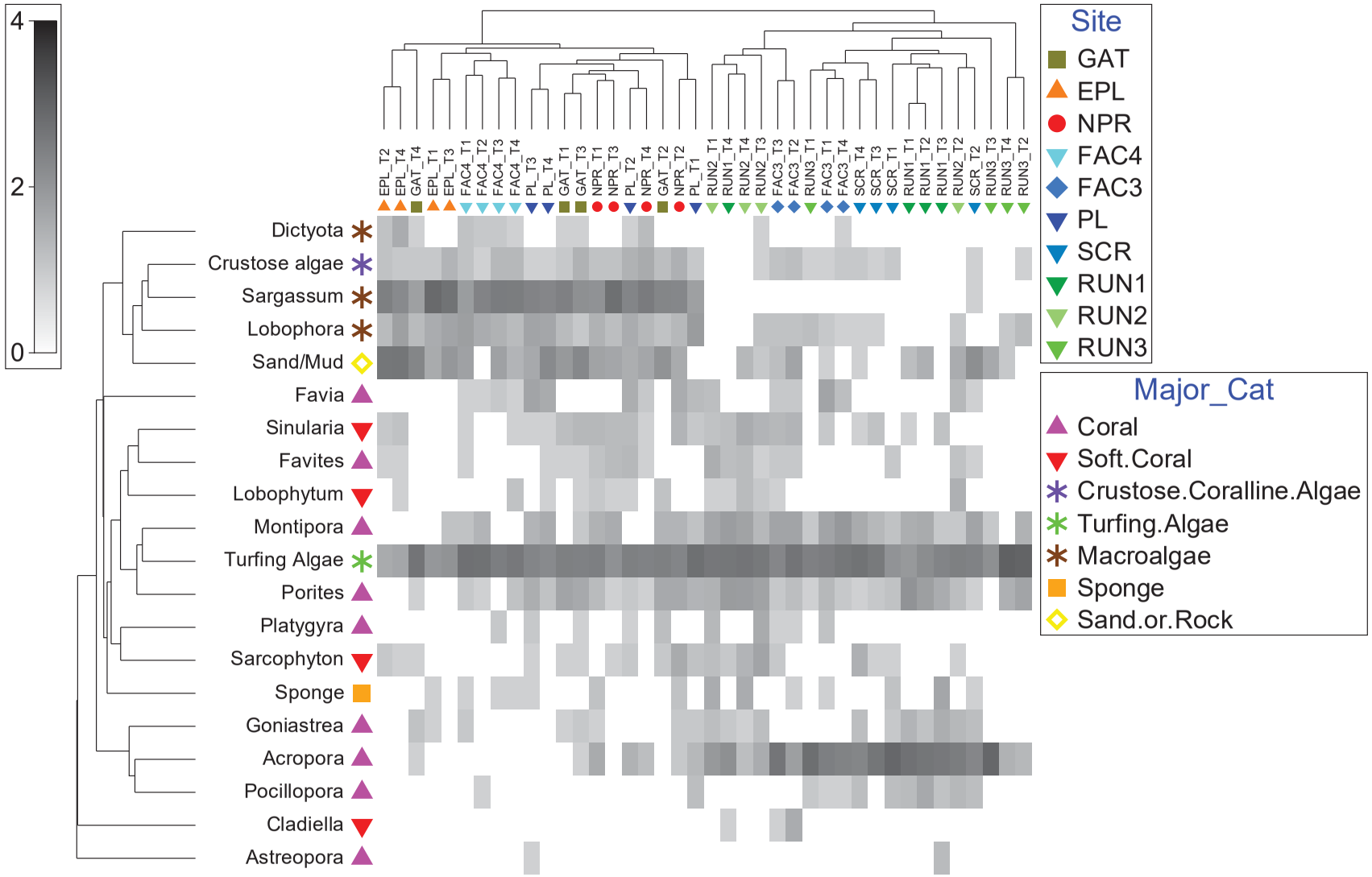

Figure 3-12 Shade plots provide a visual representation of SIMPER results

Figure 3-13 Large stand of *Acropora* at RUN3 and transect tape used to enumerate recruits and incidence of disease

3.2.3 Comparisons Between 2013 and 2018

This section presents qualitative visual comparisons between the data collected in 2013, a subset of sites that were sampled in 2014, and the results of the present survey. Graphs are presented using transect-level data. It should also be noted that whether the algae category includes turfing algae in the 2013 survey is unclear. Sites sampled on all three occasions include:

- Sable Chief Reef (SCR).
- Pearl Ledge (PL).
- Gatcombe (GAT).
- Facing Island Reef # 4 (FAC4).
- East Point Ledge (EPL).

Due to a lack of raw data, no quantitative comparisons have been made between the 2013 and 2018, nor have they been made between 2014 and 2018 due to different levels of transect replication.

Coral cover appeared to have declined between 2013 and 2014 at harbour entrance sites, and has likely remained at similar levels in 2018 to what was observed in 2014 (Figure 3-14). The variability in cover (variability within and between sites) at Rundle Island increased dramatically in the 2018 survey, and this was a reflection of site-level changes that occurred there. Figure 3-14 shows that site RUN1 likely increased in coral cover while site RUN3 likely decreased in coral cover between 2013 and 2018. This divergence in coral cover resulted in increased transect level variability (error) in Figure 3-14. Site RUN3 had recently experienced a large dieback event, and there was active evidence of coral disease (white and brown band disease) at RUN1 and RUN3 (section 3.2.4). Coral cover appeared to remain steady at FAC3 and RUN2 and may have increased at SCR. Recent dieback may have been the result of coral disease and/or thermal bleaching.

Changes in algal cover should be considered cautiously, given the uncertainty regarding the treatment of turfing algae in 2013. However, given the likely reductions in hard coral cover observed outer Facing Island and harbour entrance sites, a commensurate increase in algal cover would be expected, and is consistent with the patterns observed in Figure 3-15. The inclusion of turfing and macroalgae was consistent methodologically between 2014 and 2018, and a slight reduction in algal cover post 2014 would be consistent with some recovery in years after the 2013 floods at harbour entrance sites. Similarly, an increase in macroalgae at Rundle Island would be possible after a significant dieback event. A relatively large increase in algal cover occurred at site RUN3, where numerous incidences of coral disease were encountered.

Looking at site-based data, it is probable that sites PL, FAC4, and FAC3 had increased algal cover between 2013 and 2018, and NPR to a lesser degree. Algal cover was probably similar between 2013 and 2018 at sites RUN1 and RUN2. The site SCR probably had less algae in 2018 than 2013.

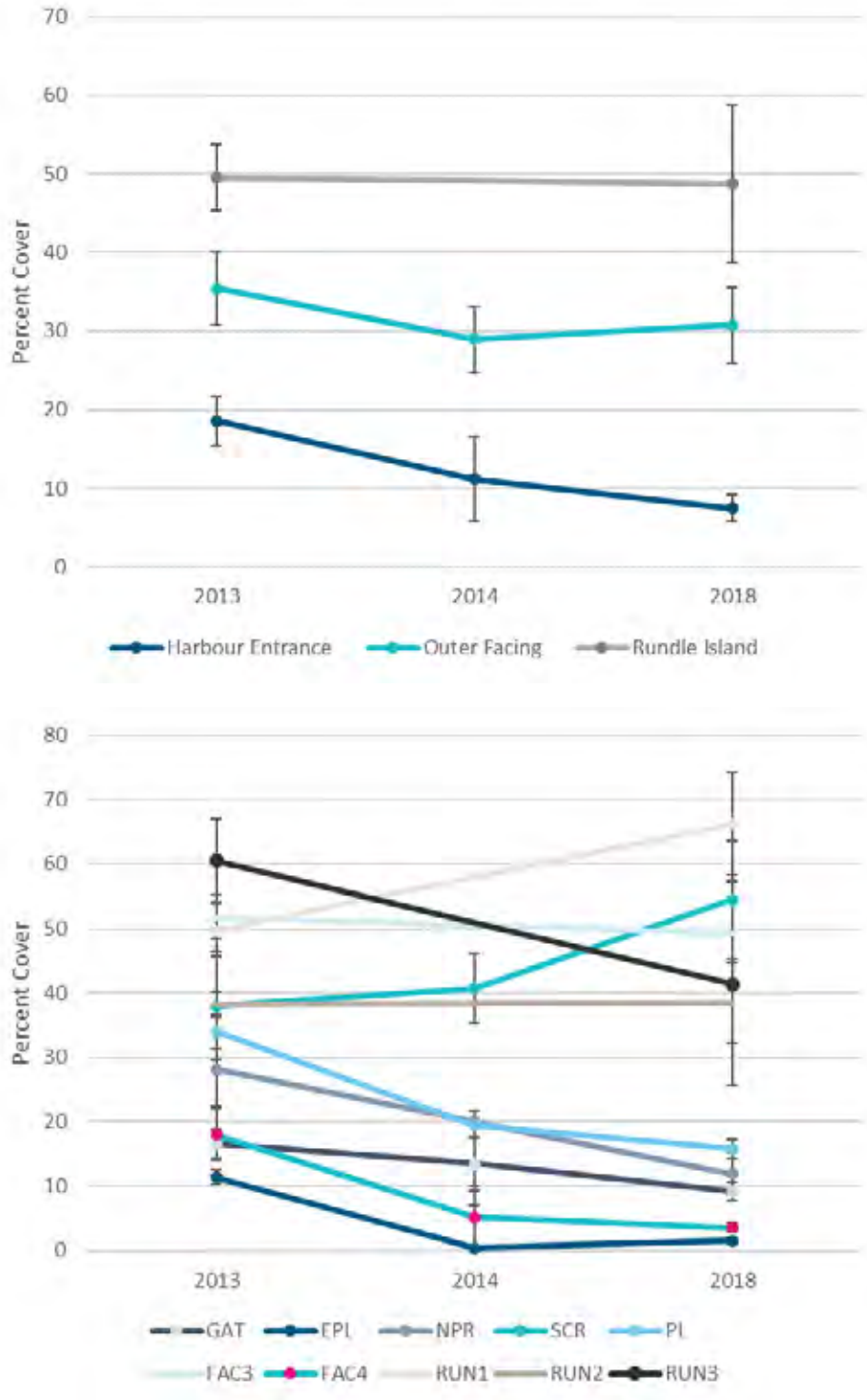

Figure 3-14 Temporal changes in hard coral cover among treatments (above) and sites (below)

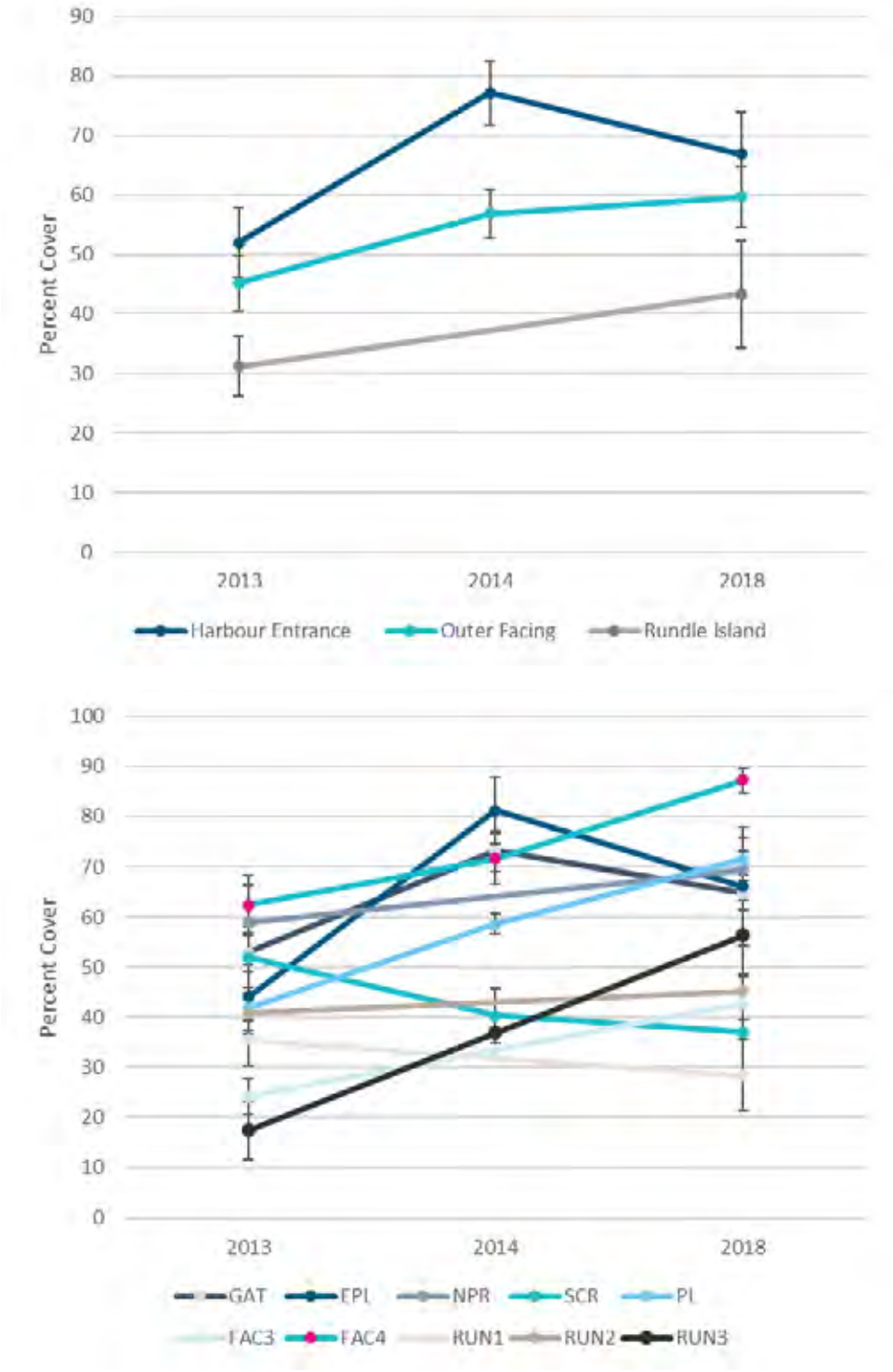

Figure 3-15 Temporal changes in algal cover among treatments (above) and sites (below)

Changes in soft coral cover among sites and treatments were highly variable among sites, treatments and time periods (Figure 3-16). This was attributed to the very low prevalence of this taxon. Differences are unlikely to be significant or meaningful between any of the monitoring periods given the present form of the data.

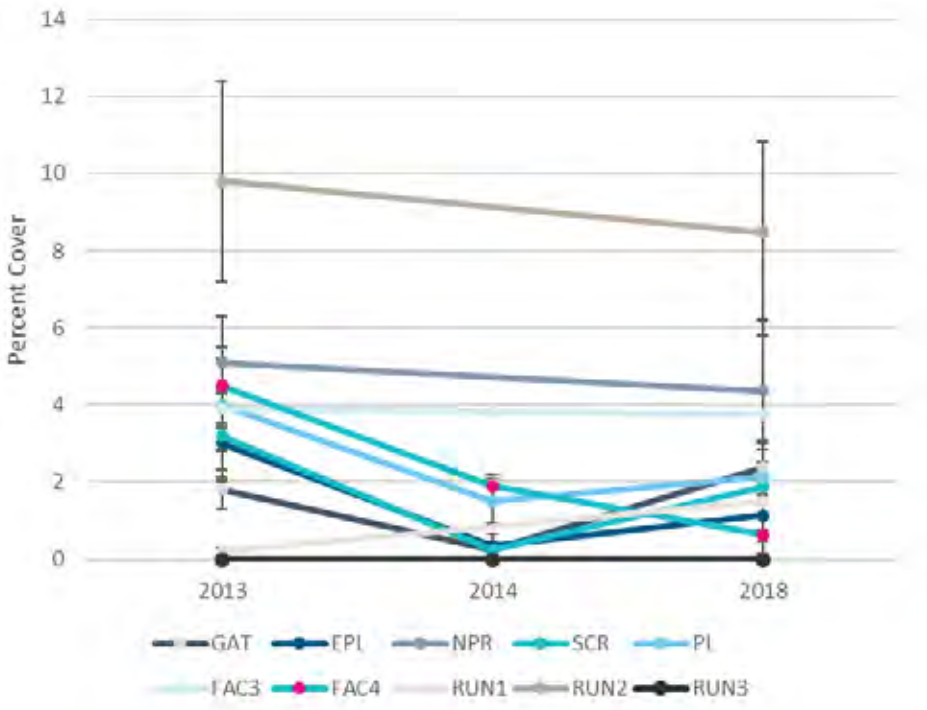

Figure 3-16 Temporal changes in soft coral cover among sites

3.2.4 2018 Coral Disease

Incidence of white and brown band diseases and coral bleaching are shown in Figure 3-17. White and brown band diseases were recorded collectively and show that incidences of coral disease were highest at Rundle Island, particularly at site RUN3, and to a lesser degree at RUN1.

Coral bleaching was observed at more sites than coral disease and again it was observed most frequently at Rundle Island. It was observed at all sites except NPR and GAT (Figure 3-17). Bleaching tended to be on individual colonies or partial colony bleaching, rather than widespread areas of bleaching as is often the case with major thermal bleaching events. In 2017, no severe bleaching was observed on the southern GBR (Swains or Capricorn-Bunker group), with the southernmost severe bleaching restricted to the Mackay Region (GBRMPA unpublished monitoring data).

In many cases, thin algal coatings were on adjacent tissues suggesting that colonies had recently experienced mortality. It is possible that some bleaching counts were actually coral disease, as partial colony bleaching and coral disease can be difficult to distinguish. Sample imagery of this recently deceased coral and white band diseases are shown in Figure 3-18

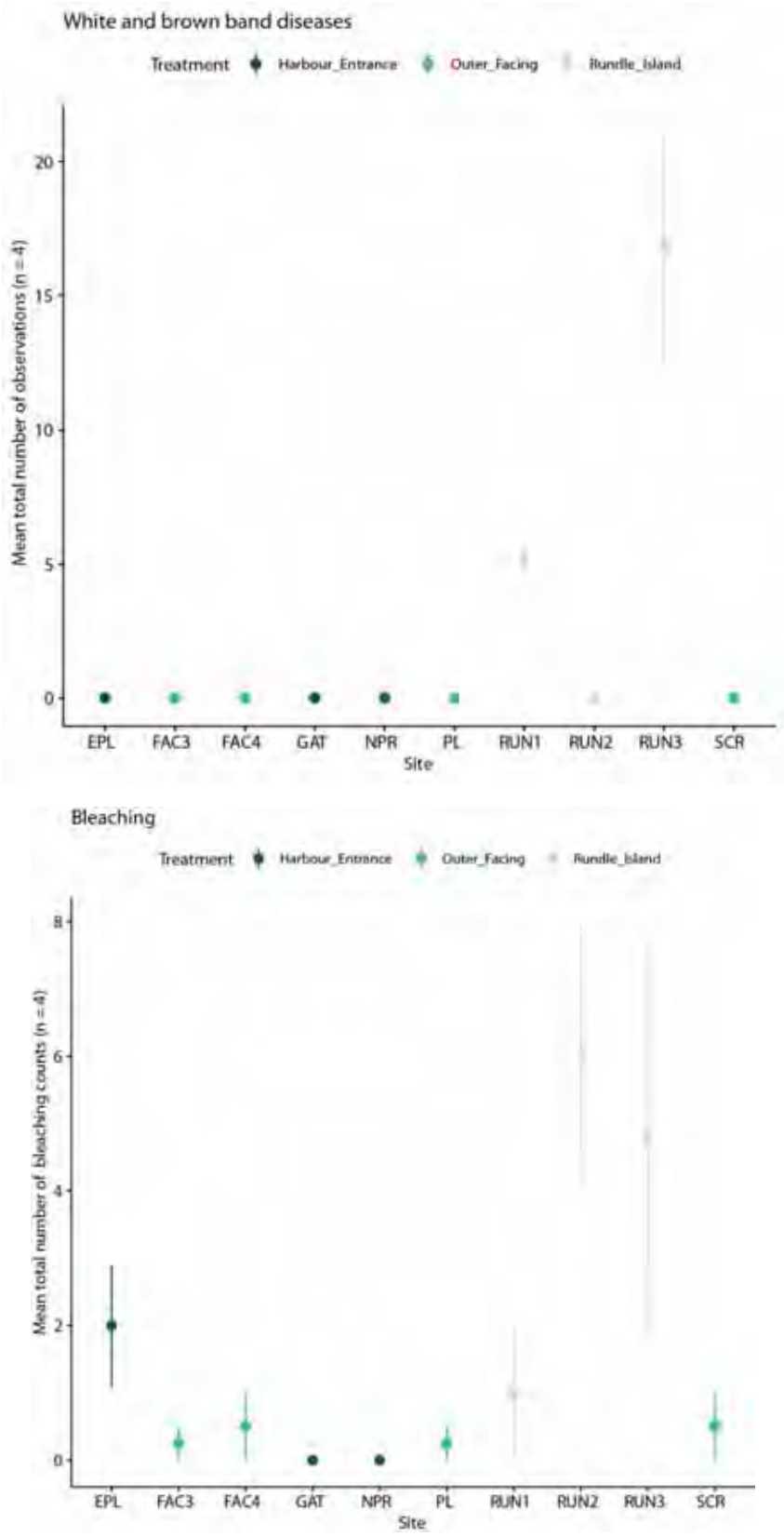

Figure 3-17 Mean (+/- SE) observations of white and brown banding disease (above) and bleaching

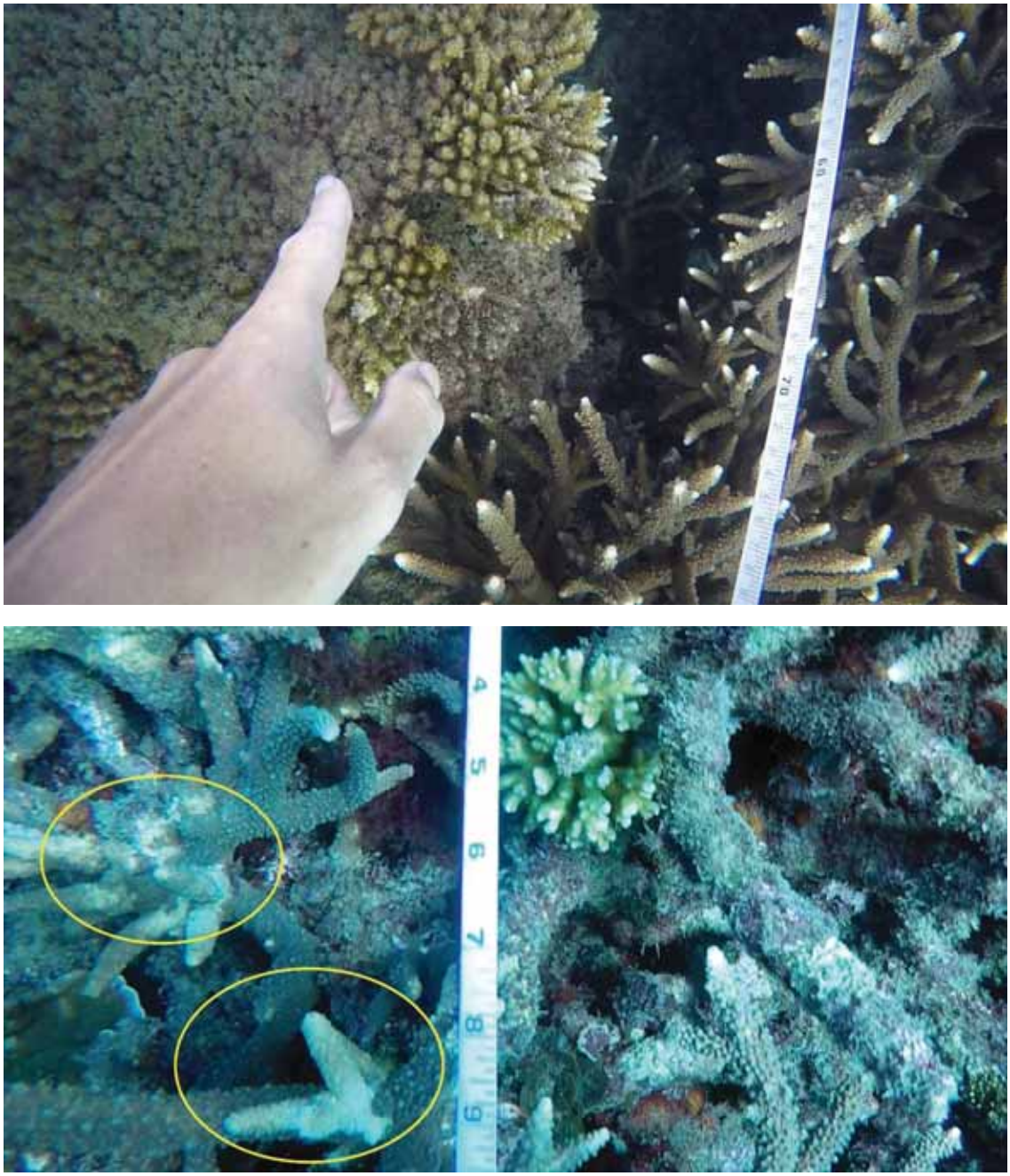

Figure 3-18 Sample imagery of recently dead tissue at Sable Chief Rocks (above) and suspected white band disease at Rundle Island (below)

3.2.5 Coral Recruit Densities 2018

Figure 3-19 shows that the total number of coral recruits was heavily dominated by *Turbinaria*. Recruiting corals were observed at all sites, with the highest levels of recruitment observed at outer Facing Island sites, particularly FAC3 and SCR. Recruit numbers were highly variable at sites where large numbers of recruits were observed, as indicated by large error bars for each site (Figure 3-19).

Rundle Island and harbour entrance sites had similar levels of recruitment, with the lowest numbers of recruits observed at EPL. It should be noted that recruitment counts may be lower than recorded

at some harbour entrance sites where *Sargassum* canopies were extensive. At these locations, particularly EPL and GAT, macroalgae often became entangled in the camera array and frequently obstructed the view of the camera and the diver, and counts were made with these limitations.

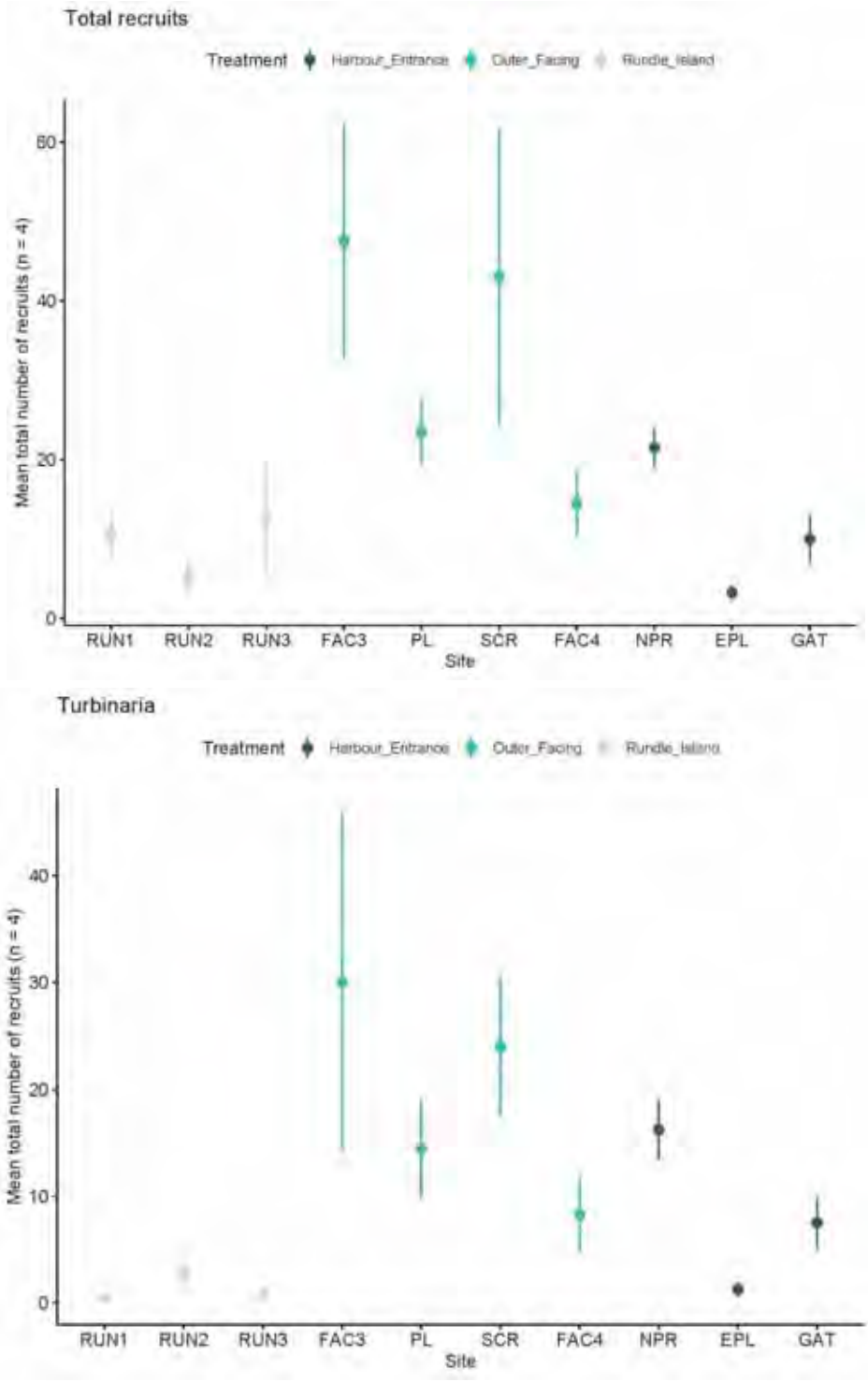

Figure 3-19 Mean (+/- SE) number of total recruits (above) and *Turbinaria* recruits (below)

3.3 3D Models

Four 3D models were produced at a subset of monitoring sites. One model was produced for NPR and SCR, and two models were produced at FAC3. These are shown in Figure 3-20 to Figure 3-23. The bottom image in each figure shows the orthomosaic of the compiled imagery. Colour in all cases appears more red in the shallower parts of the model due to loss of the red spectrum of light at greater depths.

The model at NPR shows reef elevated above the sea floor as a series ridges running north to south and entering the sea floor obliquely (Figure 3-20). A range of faviid and soft corals can be seen atop these ridges. The inter-reefal substrate consists of rubble and sand with bedforms and macroalgae. The macroalgae can be seen sitting mostly on the inter-reefal substrate directly adjacent to the reef ridges.

The first model generated at FAC3 shows a large bombora rising approximately 4 m up from the seafloor (Figure 3-21). Numerous plating acroporid corals and soft corals can be seen on and around the outcrop, including a large bleached coral. Some holes in the image are present along steep surfaces of the model.

The second model generated at FAC3 shows an undulating coral surface including four smaller coral bomboras rising approximately 1 m up from the seafloor (Figure 3-22). Numerous plating acroporid corals and soft corals can be seen throughout the model as it gently slopes away. While the generated surface is intact and free of holes, some of the model is missing from the deepest and shallowest sections of the capture area.

The model generated for SCR shows two large coral outcroppings in the corner of a field of staghorn (A*cropora*) coral (Figure 3-23). The site contains numerous plating acroporid corals amongst dense stands of branching or staghorn coral (*Acropora*). Part of the capture area could not be built into the 3D model due to extremely homogeneous view fields, where 100% *Acropora* was present in many images. Sufficient overlap between distinguishing features could not be achieved due to a combination of substrate homogeneity and water clarity.

A sample shot of the level of detail in the orthomosaic from the top of the outcrop (Figure 3-21) is shown in Figure 3-24.

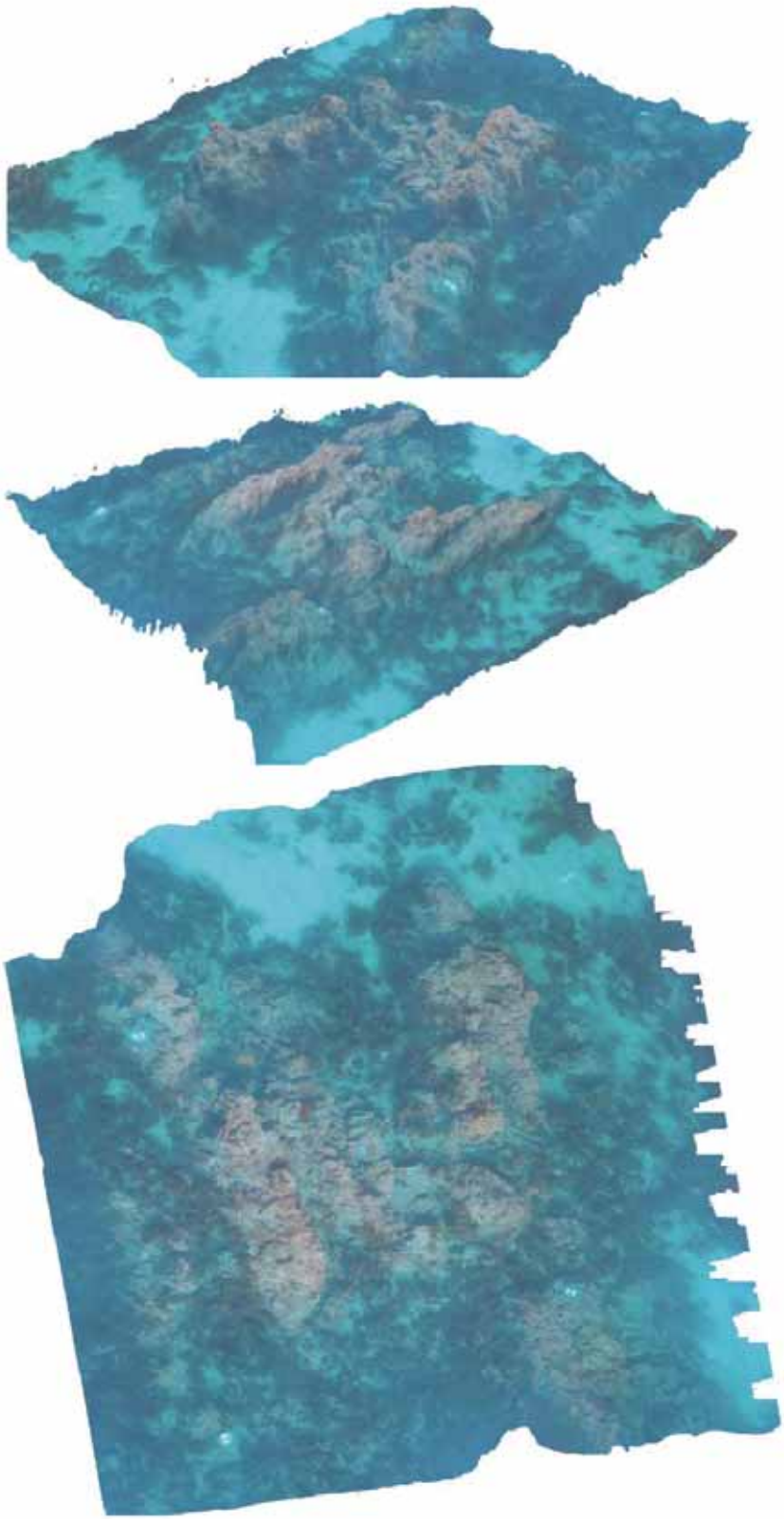

Figure 3-20 Example images of the 3D model from NPR

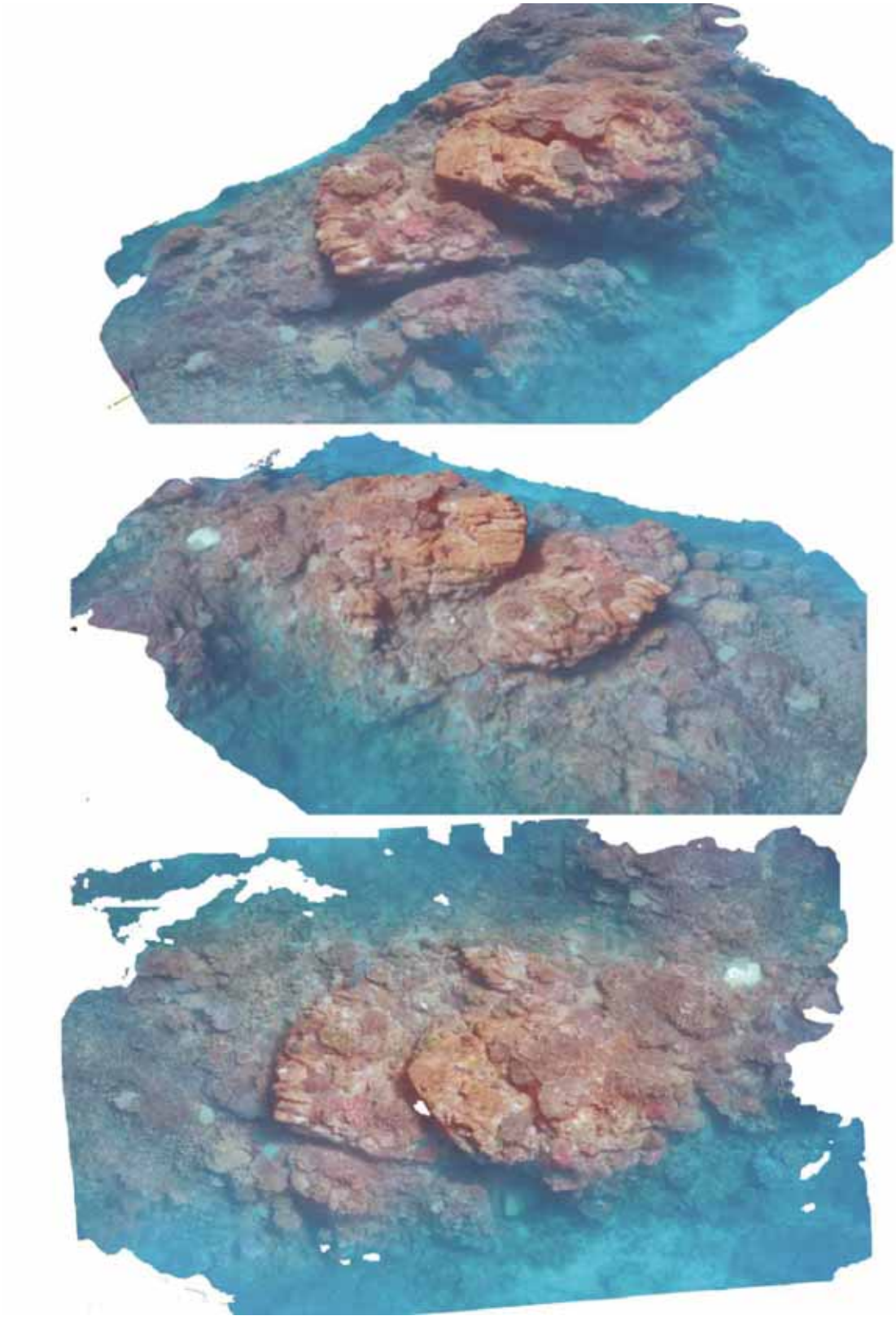

Figure 3-21 Example images of a 3D model from FAC3

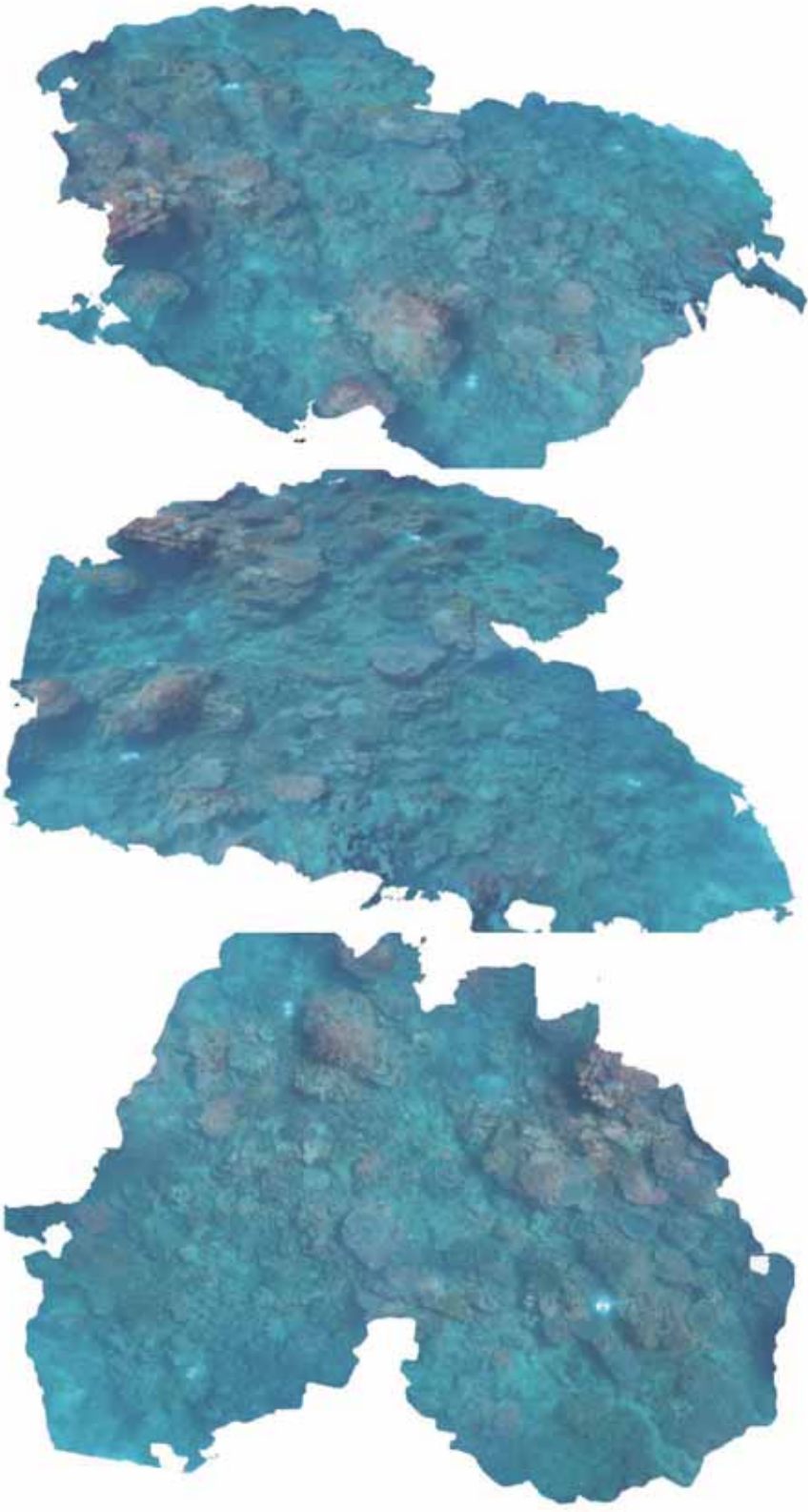

Figure 3-22 Example images of the second model from FAC3

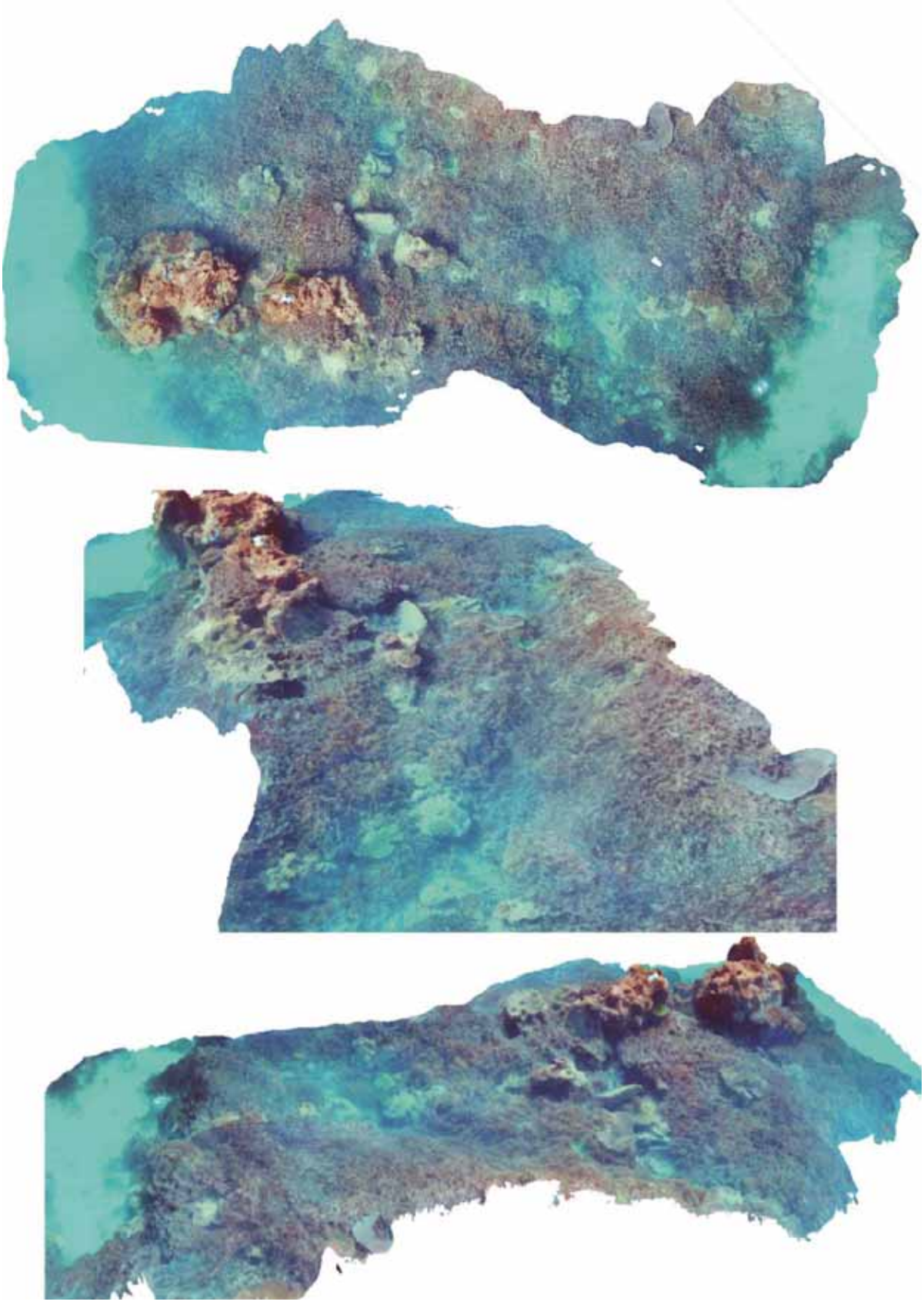

Figure 3-23 Example images of a 3D model from SCR

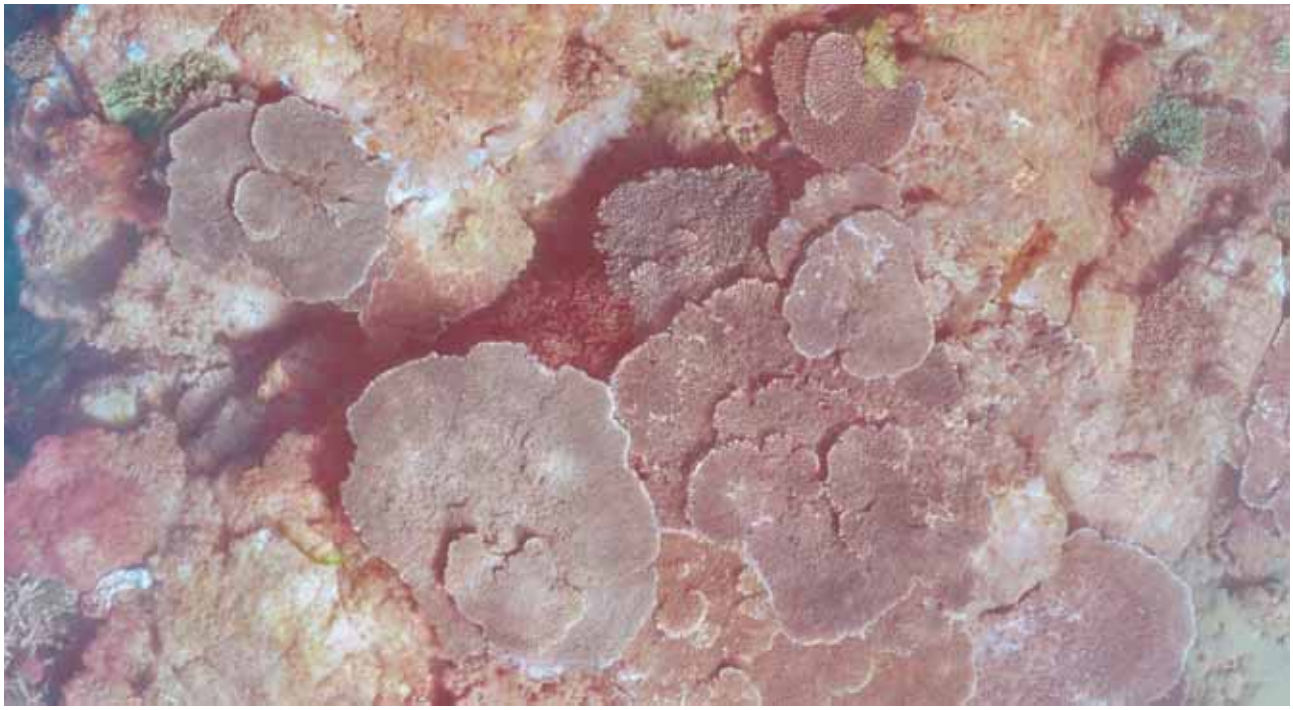

Figure 3-24 Example detail from the orthomosaic generated for top of the bombora shown in in the first FAC3 model (Figure 3-21)

4 Conclusions and Recommendations

4.1 Remote Sensing

The BRI and DII indices when applied to Sentinel-2 scenes, have improved the accuracy of the digitised reef extent performed by BMT WBM (2014). The BRI produced a much better classification of benthic habitats than the DII, which was highly anomalous and not presented. Sagawa *et al.,* (2010) show that their technique is a superior index to the DII in defining seagrass meadows in turbid water conditions (Jerlov water types II and III).

In the present study some anomalous classifications of the BRI required the user to accept or reject classifications, which would hinder the use of this imagery for objective broad-scale monitoring. Training areas were based on known conditions (sand, macroalgal dominated reefs, and coral dominated reefs) in surrounding Facing Island. As such, the classification was most successful east of Facing Island, followed by the Seal Rocks Reef area, and it was least successful at Rundle Island. This is likely due to differences in optical water type, where attenuation of the band signals was based on the water surrounding Facing Island, rather than the typically much clearer water of Rundle Island. Some of the worst classification errors at Rundle Island occurred in the deeper waters east of the island, which were not mapped as coral. Ideally, both indices would be repeated at Rundle Island using ground truthing and attenuation relationships from water in the vicinity. This was not part of the present scope.

While Sentinel-2 represents highly cost-effective imagery at reasonable resolution and high re-visit times, the biggest hurdles to using it for monitoring are the turbidity of the ambient environment, and its resolution. Finer-scale resolution data (1-2 m) would offer more reliable classifications with object-based classification techniques (Roelfsema *et al.,* 2018). However, higher resolution imagery would still be obstructed by ambient turbidity. This can be overcome by using a physics-based water column correction (Ohlendorf *et al.,* 2011), which can be performed at greater cost. This would allow quantitative spatial comparisons in cover types to be made through time. While these methods are costlier to perform than the methods used in this study, the need for more regular monitoring of broad-scale cover data may warrant the additional cost.

While such techniques can show the change in cover types through time over broad areas, such monitoring cannot provide the same level of detail regarding coral health or community change seen in site-based monitoring (divers or remote cameras).

4.2 Coral Monitoring

Coral community data and cover metrics suggest that some of the sites in the monitoring program are fundamentally different based on their living coral cover, health, and composition. The sites differ in their exposure to anthropogenic and catchment disturbance, and the revised design attempts to partition these processes as much as practical. However, some sites will experience inseparable natural and anthropogenic disturbance. Harbour entrance sites GAT and EPL are exposed to plumes generated from maintenance dredging placement on the EBSDS and chronic and acute catchment impacts. Catchment impacts likely exist as a gradient from both harbour entrances to the centre of Facing Island. The site FAC4 could probably sit within either treatment group (outer Facing Island

or harbour entrance). The sites SCR, PL, and FAC3 have sufficient community similarity to sites RUN1-3 for meaningful future comparisons.

The SCR and PL sites are very important in the interpretation of potential impacts as they (SCR in particular) experience the most influence from dredged material placement plumes emanating from the EBSDS and are highly protected from catchment impacts.

Cover metrics in 2018 showed that SCR had the second highest living coral cover. While a lack of raw transect data prevented quantitative comparisons between 2013 and 2018, some trends appeared relatively robust and are worthy of mention. Hard coral cover (the primary metric of health) likely increased at SCR, while it may have decreased at PL. Cover appeared relatively unchanged at FAC3. These changes are not consistent with widespread impacts generated by dredge plumes impacting coral communities. If this were the case, reductions in cover would be expected at SCR and FAC3. Communities at Rundle Island, particularly RUN3 were experiencing extensive coral disease and appeared to have recently experienced substantial dieback.

Harbour entrance sites experienced a substantial reduction in living coral cover between 2013 and 2014 (BMT WBM 2014), and do not appear to have recovered between 2014 and 2018. Some coral disease was also present in 2018 at SCR on plate corals, and variable changes in coral cover through time at outer Facing Island sites may be the result of variable recruitment and coral mortality since the last monitoring period. Outer Facing Island sites typically had the highest levels of recruitment.

A large degree of uncertainty exists regarding temporal changes in community mainly to do with low levels of replication and a lack of raw data. Future monitoring should consider the use of photo-quadrat data as the unit of replication, rather than transects to greatly improve the potential to detect changes. Sampling random quadrats from a pool of continuously collected photographs will allow replicate units to be assessed without creating issues of non-independence, provided photos are collected at sufficient intervals to prevent overlap.

4.3 3D Modelling

3D models provide a stakeholder engagement tool to provide a better perspective on the type of communities that exist in each location, particularly for non-specialists, or an audience that is unfamiliar being in the water or reef environments. They are also useful to managers and scientists as visual aids to other numerical data.

As a monitoring tool, they provide the potential to visualise small changes in benthic cover over a reasonably large area that may not be encompassed in point-based or line-intercept monitoring approaches. Observing orthomosaics side by side gives the viewer an opportunity to assess cover without the difficulties of measurement in a highly rugose 3D environment. Benthic cover classifications could be performed over the orthomosaics to determine percent cover of particular groups using image analysis software.

The disadvantages of this form of monitoring are that the ability to build these models is underpinned by the heterogeneity of the substrate. Substrates with a highly repetitive pattern, such as beds of staghorn coral, sands, or seagrasses are difficult to match using pixel matching algorithms. Similarly, moving substrates, such as schools of fish or beds of macroalgae change their orientation through time and create major challenges for photogrammetry. These limitations mean that data collection

and model outputs are vulnerable to missing data. The risk of missing data increases with substrate heterogeneity and turbidity, as capture frames become smaller with less overlap, and fewer features for pixel matching algorithms. Other issues include maintaining consistent colour through time with variable turbidity and lighting regimes.

Despite these issues, the models provide visually captivating imagery with the potential to produce high-resolution orthoimagery for subsequent classification and enumeration, in a selection of suitable potential habitats.

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Appendix A ANOSIM Results

Tests for differences between unordered Site groups

Global Test

Sample statistic (R): 0.725

Significance level of sample statistic: 0.1%

Number of permutations: 999 (Random sample from a large number)

Number of permuted statistics greater than or equal to R: 0

Pairwise Tests

Groups	R Statistic	Significance Level (p)	Possible Permutations	Actual Permutations	Number >= Observed
EPL, FAC4	0.417	0.029	35	35	1
EPL, GAT	0.375	0.086	35	35	3
EPL, FAC3	1	0.029	35	35	1
EPL, NPR	0.406	0.029	35	35	1
EPL, PL	0.521	0.029	35	35	1
EPL, RUN1	1	0.029	35	35	1
EPL, RUN2	1	0.029	35	35	1
EPL, RUN3	0.99	0.029	35	35	1
EPL, SCR	1	0.029	35	35	1
FAC4, GAT	0.792	0.029	35	35	1
FAC4, FAC3	0.969	0.029	35	35	1
FAC4, NPR	0.573	0.029	35	35	1
FAC4, PL	0.552	0.029	35	35	1
FAC4, RUN1	1	0.029	35	35	1
FAC4, RUN2	1	0.029	35	35	1
FAC4, RUN3	0.948	0.029	35	35	1
FAC4, SCR	0.99	0.029	35	35	1
GAT, FAC3	0.958	0.029	35	35	1
GAT, NPR	0.302	0.114	35	35	4
GAT, PL	0.542	0.029	35	35	1
GAT, RUN1	1	0.029	35	35	1
GAT, RUN2	0.885	0.029	35	35	1
GAT, RUN3	0.906	0.029	35	35	1

ANOSIM Results

Groups	R Statistic	Significance Level (p)	Possible Permutations	Actual Permutations	Number >= Observed
GAT, SCR	0.948	0.029	35	35	1
FAC3, NPR	0.792	0.029	35	35	1
FAC3, PL	0.667	0.029	35	35	1
FAC3, RUN1	0.729	0.029	35	35	1
FAC3, RUN2	0.646	0.029	35	35	1
FAC3, RUN3	0.615	0.029	35	35	1
FAC3, SCR	0.5	0.029	35	35	1
NPR, PL	0.073	0.314	35	35	11
NPR, RUN1	0.99	0.029	35	35	1
NPR, RUN2	0.896	0.029	35	35	1
NPR, RUN3	0.917	0.029	35	35	1
NPR, SCR	0.917	0.029	35	35	1
PL, RUN1	1	0.029	35	35	1
PL, RUN2	0.885	0.029	35	35	1
PL, RUN3	0.948	0.029	35	35	1
PL, SCR	0.865	0.029	35	35	1
RUN1, RUN2	0.531	0.057	35	35	2
RUN1, RUN3	0.521	0.029	35	35	1
RUN1, SCR	0.26	0.086	35	35	3
RUN2, RUN3	0.875	0.029	35	35	1
RUN2, SCR	0.5	0.057	35	35	2
RUN3, SCR	0.5	0.029	35	35	1

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Appendix K
Priority Back on Track fauna species

Appendix K Priority Back on Track fauna species

The 'Back on Track species prioritisation framework' (Back on Track [BoT]) identifies priority threatened species for conservation and recovery management in Queensland.

Gladstone is situated within the Fitzroy NRM. The biodiversity document relevant for the Project is '*Back on track actions for biodiversity: Taking action to conserve species in the Fitzroy Natural Resource Management Region*' (DERM 2010).

There are 81 BoT priority species for the Fitzroy NRM, including 49 invertebrate and fauna species (DERM 2010) (refer Table 2), including seven invertebrates, four fish species, one amphibian, 18 reptiles, seven birds and 12 mammals.

Six BoT priority fauna were confirmed to occur within the Project database search area following the database review and field assessments, including three marine turtles (refer Section 14), one migratory shorebird (refer Section 12) and two marine mammals (refer Section 15). A further 15 BoT priority fauna species were considered to have a moderate likelihood of occurrence within the Project impact areas, including two fish species (refer Section 9), one migratory shorebird (refer Section 12), one invertebrate species, one marine turtle (refer Section 14), one marine mammal species (refer Section 15), five terrestrial reptiles and four mammal species (refer Section 13 for intertidal and terrestrial fauna species). Twenty-eight of the 49 BoT priority species were considered to have a negligible or low likelihood of occurrence within the Project impact areas, based on the category definitions outlined in Table 1.

Table 1 Likelihood of occurrence category definitions

Likelihood of occurrence category	Definition
Confirmed	Multiple records for the species in the Project EIS database search area from a reliable data source (e.g. previous studies or specimen-backed database records) or confirmed during Project EIS field investigations, and suitable habitat exists within the Project impact areas
Moderate	Records for the species in the Project EIS database search area from a reliable data source but not specifically recorded within the Project impact areas. Suitable habitat for this species exists within the Project impact areas
Low	Records for the species from the Project EIS database search area from a reliable data source but not specifically recorded within the Project impact areas. Suitable habitat for this species does not exist within the Project impact areas
Negligible	Identified from database searches only and no suitable habitat exists within the Project impact areas

Table note:
Refer Section 2.2 for further details.

Table 2 Back on Track priority invertebrate and fauna species for the Fitzroy NRM region

Scientific name	Back on Track status (Fitzroy NRM)	Habitat association	Likelihood of occurrence within the Project direct and/or potential indirect areas
Fish and other nekton			
Molluscs			
Black knight semi-slug (*Helicarionidae BL 22*)	Critical priority	The species is an air-breathing terrestrial semi-slug	**Low** – The species may be present within terrestrial environments adjacent to the WBE reclamation area. However there is limited information regarding the species habitat associations
Blackdown Tableland snail (*Camaendiae BL 11*)	High	The species is associated with area surrounding the Blackdown Tableland National Park	**Negligible** – The Project impact area is outside of the species known area of distribution
Boggomoss snail (*Adclarkia dawsonensis*)	Critical priority	The species is known from the Dawson Valley which lies within the Brigalow Belt. Habitat is characterised by *Acacia harpophylla* (Brigalow) forest and woodland on clay soils. The species is restricted to the moister riparian and boggomoss habitats on alluvial flats between Taroom and Theodore	**Negligible** – The Project impact areas are outside of the area of known species distribution and does not contain areas of suitable habitat
Dawson River dark snail (*Sphaerospira mossmani*)	High	A native, air-breathing terrestrial snail associated with the Dawson River region of Queensland	**Negligible** – The Project impact areas are outside of the species known area of distribution. There is limited information regarding the species habitat associations
Lotus Creek banded snail (*Camaenidae BL 4*)	High	An air-breathing terrestrial snail associated with the Lotus Creek area. Lotus Creek is situated between Mackay and Rockhampton in central Queensland	**Negligible** – The Project impact areas are outside of the species known area of distribution
Marine fish			
Estuary stingray (*Dasyatis fluviorum*)	High	The species typically inhabits shallow, mangrove-lined tidal rivers, estuaries and bays	**Moderate** – Suitable habitat is present within the WBE reclamation area
Great white shark (*Carcharodon carcharias*)	High	Great white sharks can be found from close inshore around rocky reefs, surf beaches and shallow coastal bays to outer continental shelf and slope areas	**Moderate** – Suitable habitat is present within the Project impact areas
Green sawfish (*Pristis zijsron*)	High	The species inhabits muddy bottom habitats and enters estuaries. The species has been recorded in inshore marine waters, estuaries, river mouths, embankments and along sandy and muddy beaches	**Low**– The Project impact areas are situated outside of the current known distribution for the Green sawfish This species has not been recorded in the vicinity of the Project impact areas since the late 1960s

Scientific name	Back on Track status (Fitzroy NRM)	Habitat association	Likelihood of occurrence within the Project direct and/or potential indirect areas
Freshwater fish			
Ornate rainbowfish (*Rhadinocentrus ornatus*)	High	The species is associated with freshwater rivers and streams. The species distribution is restricted to isolated locations in the Greater Brisbane/Redlands region and has a patchy distribution through coastal creeks from Byfield, Queensland to Nambucca in New South Wales	**Negligible** – There are no areas of suitable habitat present within the Project impact areas. Rivers and creeks within the Project impact areas are tidally influenced
Marine turtles			
Flatback turtle (*Natator depressus*)	Critical priority	The Flatback turtle resides in the Australian continental shelf and requires sandy beaches to nest	**Confirmed** – Suitable foraging and nesting habitat is present within the Project impact areas The Flatback turtle is known to occur within Port Curtis and is the dominant nesting species of marine turtles on beaches within the Gladstone region
Green turtle (*Chelonia mydas*)	Critical priority	The Green turtle inhabits ocean currents and utilises shallow benthic foraging habitats such as tropical tidal and sub-tidal coral and rocky reef habitat or inshore seagrass	**Confirmed** – Suitable foraging and nesting habitat is present within the Project impact areas The Green turtle has been recorded regularly within Port Curtis and Green turtles nest on beaches within Port Curtis (refer Section 14)
Hawksbill turtle (*Eretmochelys imbricata*)	Critical priority	Hawksbill turtles typically occur in tidal and sub-tidal coral and rocky reef habitats throughout tropical waters	**Moderate** – Suitable foraging habitat is present within the Project impact areas Significant feeding populations of Hawksbill turtles have not been recorded within the Port Curtis region. No Hawksbill turtle nesting has been recorded within the Port Curtis region.
Leatherback turtle (*Dermochelys coriacea*)	Critical priority	Leatherback turtles are known from marine waters all around Australia and requires sandy beaches to nest	**Low** – Suitable foraging habitat is present within the Project impact areas Scattered nesting is known to occur from Bundaberg to Round Hill Head. Only one Leatherback turtle has been recorded nesting near Port Curtis
Loggerhead turtle (*Caretta caretta*)	Critical priority	The Loggerhead turtle is known from marine waters across Australia and requires sandy beaches to nest	**Confirmed** - Suitable foraging habitat is present within the Project impact areas The species has been recorded within Port Curtis

Scientific name	Back on Track status (Fitzroy NRM)	Habitat association	Likelihood of occurrence within the Project direct and/or potential indirect areas
Megafauna			
Australian snubfin dolphin (*Orcaella heinsohni*)	Critical priority	Boat-based surveys along the east coast of Queensland indicate that Australian snubfin dolphins are primarily found in shallow waters less than 20m deep, close to the coast, close to river and creek mouths and in the proximity of seagrass meadows	**Moderate** – Suitable habitat is present within the Project impact areas The Australian snubfin dolphin has not been identified within the Port Curtis region during extensive survey efforts conducted as part of the Ecosystem Research and Monitoring Program coordinated by GPC (refer Section 15)
Dugong (*Dugong dugong*)	Critical priority	Dugongs are associated with coastal waters and estuarine creeks and streams. Feeding aggregations tend to occur in wide, shallow protected bays; wide, shallow mangrove channels; and in the lee of large inshore islands. These areas are coincident with sizeable seagrass meadows	**Confirmed** – Suitable habitat is present within the Project impact areas The Dugong is known to occur within Port Curtis (refer Section 15)
Australian humpback dolphin (*Sousa sahulensis*)	Critical priority	Australian humpback dolphins inhabit shallow coastal, estuarine, and occasionally riverine habitats, in tropical and subtropical regions. The species usually occurs close to the coast, generally in depths of less than 20m, but they have been seen 55km offshore in shallow water	**Confirmed** – Suitable habitat is present within Project impact areas The Australian humpback dolphin is known to occur within Port Curtis
Migratory shorebirds			
Beach stone-curlew (*Esacus magnirostris*)	High	The Beach stone-curlew is predominately found on open, undisturbed beaches, islands, reefs and estuarine intertidal sand and mudflats. The species demonstrates preference for beaches with estuaries or mangroves nearby	**Confirmed** – The species has been identified within areas directly adjacent to the WBE reclamation area
Little tern (*Sterna albifrons*)	High	In Australia, Little terns inhabit sheltered coastal environments, including lagoons, estuaries, river mouths and deltas, lakes, bays, harbours and inlets, especially those with exposed sandbanks or sand-spits, and also on exposed ocean beaches	**Moderate** – Suitable habitat is present within Project impact areas
Intertidal and terrestrial fauna			
Mammals			
Bridled nailtail wallaby (*Onychogalea fraenata*)	Critical priority	The Bridled nailtail wallaby occurs in woodlands, particularly in Brigalow scrub. This species is usually associated with the edges of Brigalow woodlands and regrowth as these areas provide maximum shelter at the height of the individual. The species is confined to Tauton National Park, near the town of Dingo	**Negligible** – The Project impact areas do not have suitable habitat for the species and is situated outside of the species known distribution

Scientific name	Back on Track status (Fitzroy NRM)	Habitat association	Likelihood of occurrence within the Project direct and/or potential indirect areas
Coastal sheathtail bat (*Taphozous australis*)	High	The Coastal sheathtail bat forages within distances of 1km from the ocean, in habitat, including sand dune scrub, mangroves, Melaleuca swamps, coastal heathlands, open Eucalypt forest and grasslands	**Moderate** – Suitable foraging habitat for the species adjacent to the WBE reclamation area. Suitable foraging resources are not present within the WBE reclamation area direct impact area. The boulders which define the bund wall of the existing Western Basin reclamation area may provide potential roosting habitat for the species. The species has been previously recorded in areas approximately 1.3km west of the WBE reclamation area.
South-eastern long-eared bat (*Nyctophilus corbeni*)	High	The Eastern long-eared bat occurs in a range of inland woodland vegetation types, including Box, Ironbark and Cypress pine woodlands. This species prefers habitat with a distinct canopy and a dense clustered shrub layer	**Low** – Suitable habitat is not considered to be present with respect to the coastal nature of the Project impact areas
Ghost bat (*Macroderma gigas*)	Critical priority	The Ghost bat occurs in a range of habitats from rainforest, monsoon and vine scrub, to open woodlands in arid areas. Roosting sites include undisturbed caves or mineshafts which have several openings	**Low** – Suitable habitat is not present within the Project impact areas
Golden-tipped bat (*Kericoula papuensis*)	High	The Golden-tipped bat is found in rainforest and adjacent sclerophyll forest. It roosts in abandoned hanging yellow-throated scrubwren and brown gerygone nests located in rainforest gullies on small (first and second order) streams	**Low** – Suitable habitat is not present within the Project impact areas
Grey-headed flying-fox (*Pteropus poliocephalus*)	Critical priority	The Grey-headed flying-fox is associated with a wide range of habitats, including rainforest, mangroves, paperbark forests, wet and dry sclerophyll forests and cultivated areas. This species is capable of nightly flights of up to 50km from a roost site in order to forage (Eby 1991)	**Moderate** – The Eucalyptus woodlands present within and adjacent to the Project impact areas provide suitable foraging habitat for the species. There are no breeding camps for the species present within the Project impact areas. The nearest Grey-headed flying-fox camp to the Project impact areas is situated approximately 25 km south of the WBE reclamation area, recorded to support between 1-499 Grey-headed flying-fox individuals during a survey conducted in November 2015) (DoE 2018). No other active Grey-headed flying-fox camps are recorded by DoE (2018) within a 50km radius of the Project impact areas

Scientific name	Back on Track status (Fitzroy NRM)	Habitat association	Likelihood of occurrence within the Project direct and/or potential indirect areas
Spotted-tailed quoll (*Dasyurus maculatus maculatus*)	High	The Spotted-tail quoll is associated with mature wet forest habitats and utilises escarpments and gullies for foraging and breeding. Species required suitable den structures such as hollow logs, tree hollows, rocky outcrops or caves	**Low** – Suitable habitat is not present within the Project impact areas
Water mouse (*Xeromys myoides*)	Critical priority	The species inhabits mangroves and associated saltmarsh, sedgelands, clay pans, heathlands and freshwater wetland communities	**Moderate** – Suitable habitat is present directly adjacent to the WBE reclamation area
Yellow-bellied glider (*Petaurus australis australis*)	High	The Yellow-bellied glider occurs in tall Eucalypt open forests. Habitat is often restricted to the ecotone between rainforest and drier woodland ecosystems	**Moderate** – Suitable habitat is present adjacent to the WBE reclamation area, and adjacent to the Quarry to WBE reclamation area
Spiders			
Australian featherleg spider (*Selenotypus plumipes*)	Critical priority	The species inhabits arid-zone grassland regions	**Negligible** – Suitable habitat is not present for the species within the Project impact areas
Queensland whistling tarantula (*Selenocosmia crassipes*)	Critical priority	The species lives in rainforests and gullies or other sheltered cooler spots in open forests along the Queensland Central coast	**Moderate** – Suitable habitat is present adjacent to the WBE reclamation area
Reptiles			
Allan's lerista (*Lerista allanae*)	High	A robust burrowing skink, the species is confined to the undulating, black soil downs of the ventral Brigalow Belt region	**Negligible** – The Project impact areas are outside of the species known area of distribution
Capricorn worm-skink (*Anomalopus brevicollis*)	High	The species is found in well-mulched friable soil under logs, rock and leaf litter in open dry sclerophyll forest to rainforest habitats (e.g. rainforest, vine scrub, semi-evergreen vine thicket, monsoon forest). They are found on sand and cracking clay based soils and rock outcrops	**Moderate** – Suitable habitat is present, adjacent to the WBE reclamation area and adjacent to the Quarry to WBE reclamation area
Collared delma (*Delma torquata*)	High	The Collared delma is predominately associated with Eucalyptus and Brigalow dominated woodlands within the Brigalow Belt. At the eastern extent of the species range, the species is associated with exposed rocky outcrops	**Low** – Suitable habitat is not present for the species within the Project impact area
Common death adder (*Acanthophis antarcticus*)	High	The Common death adder is found in a range of habitats in association with deep leaf litter, including rainforests, wet sclerophyll forests, woodlands, grasslands, chenopod dominated shrublands and coastal heathlands	**Moderate** – Potential species habitat is present within areas of open dry forest adjacent to the WBE reclamation area and Quarry adjacent the WBE reclamation area haul routes.

Scientific name	Back on Track status (Fitzroy NRM)	Habitat association	Likelihood of occurrence within the Project direct and/or potential indirect areas
Fitzroy River turtle (*Rheodytes leukops*)	High	The species is restricted to the Fitzroy River and its tributaries, and occurs within permanent freshwater riverine reaches and permanent waterholes	**Negligible** – The Project impact areas are outside of the species known area of distribution. Water sources within the Project impact areas are tidally influenced
Golden-tailed gecko (*Strophurus taenicauda*)	High	The species inhabits open woodlands and open forests, with the majority of species records collected from the Brigalow Belt region	**Negligible** – The Project impact areas are outside of the species known area of distribution. The species has not been identified in database sources
Ornamental snake (*Denisonia maculata*)	High	The species is predominately associated with environments within the Brigalow Belt bioregion. The Ornamental snake is commonly associated with Gilgai formations which are found where deep-cracking alluvial soils with high clay contents occur. The species is usually found in areas mapped as Landzone 4 by the Queensland RE mapping	**Low** – Suitable habitat is not present for the species within the Project impact areas
Phyllurus campionae (no common name (NCN))	Critical priority	A leaf-tailed gecko for which limited information regarding the species habitat association is limited. Leaf-tailed geckos are associated with rainforest environments	**Negligible** – The Project impact area does not contain suitable habitat for leaf-tailed geckos
Ringed thin-tailed gecko (*Phyllurus caudiannulatus*)	High	The species is found in rainforest habitats, restricted to the Bulburin State Forest on the Dawes Range and Many Peaks Range	**Negligible** – The Project impact areas are outside of the species known area of distribution
Rusty monitor (*Varanus semiremex*)	High	The Rusty monitor is associated with mangrove environments	**Moderate** –Areas adjacent to the WBE reclamation area provide suitable habitat for the species
Stephens' banded snake (*Hoplocephalus stephensii*)	High	The species lives in rainforests, moist forests, heaths and vine thickets	**Moderate** – Suitable habitat is present, adjacent to the WBE reclamation area
White-throated snapping turtle (*Elseya albagula*)	High	The species is found within the Burnett, Fitzroy, Raglan and Mary River waterways. The species is found in permanent flowing water areas where there is suitable shelter resources	**Negligible** – The Project impact areas are outside of the species known area of distribution. Water sources within the Project impact area are tidally influenced
Yakka skink (*Egernia rugosa*)	High	The species is found amongst dense ground vegetation, fallen timber or rock outcrops in open dry sclerophyll forest (ironbark) or woodland, Brigalow forest, open shrub land, and lancewood forest on coarse gritty soils in the vicinity of low ranges, foothills and undulating terrain with good drainage	**Moderate** – Potential species habitat is present within areas of open dry forest adjacent to the WBE reclamation area and Quarry adjacent the WBE reclamation area haul routes.
Frogs			
Kroombit tinkerfrog (*Taudactylus pleione*)	High	The species is known from 12 small patches of rainforest at Kroombit Tops National Park, southwest of Gladstone	**Negligible** – The Project impact areas are outside of the species known area of distribution

Scientific name	Back on Track status (Fitzroy NRM)	Habitat association	Likelihood of occurrence within the Project direct and/or potential indirect areas
Birds			
Black-breasted button-quail (*Turnix melanogaster*)	Critical priority	The Black-breasted button-quail habitat is associated with coastal scrubs and fragments of vine forest and thickets that are periodically water-stressed. The species may also be found in low thickets or woodlands, acacia thickets and lantana patches (in association with these vegetation types). They have also been regularly observed within hoop pine plantations	**Low** – Suitable habitat is not present for the species within the Project impact area
Diamond firetail (*Stagonopleura guttata*)	High	The Diamond firetail is found in grassy woodlands, heath, farmland and grasslands with scattered trees. The species range from the Carnarvon Ranges in Queensland, and the Eyre Peninsula and Kangaroo Island in South Australia	**Negligible** – The Project impact areas are outside of the species known area of distribution
Painted honeyeater (*Grantiella picta*)	High	The species inhabits Boree, Brigalow and Box-Gum Woodlands and Box-Ironbark Forests. The species is one of Australia's most specialised honey eaters and is dependent on mistletoe species, primarily *Amyema cambagei* and *Amyema quandang* as a food source	**Low** – Based on the known distribution it is unlikely that this species will occur within the Project impact areas
Red goshawk *(Erythrotriorchis radiatus*)	High	The species generally occur where there is a mosaic of vegetation types close to permanent water, including coastal and sub-coastal tall open forest, lightly treed tropical savannahs containing wooded or forested watercourses, woodlands, the edges of rainforest and gallery forests along watercourses and wetlands	**Low** – Suitable habitat is not present for the species within the Project impact area
Yellow chat (Dawson) (*Epthianura crocea macgregori*)	High	The Yellow chat (Dawson) inhabits marine plain wetlands that are subject to seasonal inundation and varying degrees of both fresh and saltwater influence. The species usually inhabits marine plains that have a network of shallow drainage channels which support a mosaic of vegetation that consist of grasslands, dense beds of rush or sedge, patches of samphire, and areas of bare or sparsely vegetated mudflats	**Low** – Suitable habitat is not present for the species within the Project impact area

Appendix L
Fish and fisheries values lists

Appendix L Fish and fisheries values lists

Table 1 Species of fish and other nekton recorded during Project EIS surveys at estuarine and coastal habitats during the 2014 dry season and the 2015 wet season

Family name	Species name	Common name	Percent of observed species (%)
Clupeidae	*Herklotsichthys castelnaui*	Southern herring	38.7
Ambassidae	*Ambassis marianus*	Estuary glassfish	21.3
Engraulidae	*Stolephorus waitei*	Spottyface anchovy	14.3
Penaeidae	*Fenneropenaeus merguiensis*	Banana prawn	7.4
Leiognathidae	*Leiognathus equulus*	Common ponyfish	4.3
Hemiramphidae	*Hyporhamphus dussumieri*	Slender garfish	4.2
Mugilidae	*Liza subviridis*	Greenback mullet	3.9
Gerreidae	*Gerres subfasciatus*	Common silverbiddy	1.6
Atherinidae	*Atherinomorus ogilbyi*	Common hardyhead	1.0
Sparidae	*Acanthopagrus butcheri*	Black bream	0.5
Hemiramphidae	*Arrhamphus sclerolepis*	Subnose garfish	0.4
Batrachoididae	*Tetractenos hamiltoni*	Common toadfish	0.4
Batrachoididae	*Tetractenos glaber*	Smooth toadfish	0.3
Sillaginidae	*Sillago ciliata*	Sand whiting	0.2
Siganidae	*Siganus lineatus*	Goldlined rabbitfish	0.2
Tetraodontidae	*Chelonodon patoca*	Milk-spotted pufferfish	0.2
Terapontidae	*Amniataba percoides*	Barred grunter	0.2
Latidae	*Lates calcarifer*	Barramundi	0.1
Portunidae	*Portunis pelagicus*	Blue swimmer crab	0.1
Mugilidae	*Mugil cephalus*	Sea mullet	0.1
Sparidae	*Acanthopagrus australis*	Yellowfin bream	0.1
Uranoscopidae	*Kathetostoma laeve*	Common stargazer	0.1
Engraulidae	*Engraulis australis*	Australian anchovy	0.1
Sphyraenidae	*Sphyraena jello*	Pick-handle harracuda	0.1
Terapontidae	*Amniataba percoides*	Barred grunter	< 0.1
Tetraodontidae	*Arothron manilensis*	Narrow-lined pufferfish	< 0.1
Belonidae	*Strongylura strongylura*	Spottail needlefish	< 0.1
Lutjanidae	*Lutjanus russelli*	Moses perch	< 0.1
Carangidae	*carangoides humerosus*	Epaulette trevally	< 0.1
Terapontidae	*Terapon jarbua*	Crescent perch	< 0.1
Clupeidae	*Nematalosa erebi*	Bony bream	< 0.1
Leptobramidae	*Leptobrama muelleri*	Beach salmon	< 0.1
Belonidae	*Strongylura strongylura*	Spot tail needlefish	< 0.1
Platycephalidae	*Platycephalus indicus*	Bar tail flathead	< 0.1

Table note: Species listed in order of highest to lowest proportion

Source: Adapted from Vision Environment (2015a)

Table 2 Species of fish and other nekton recorded during the Project EIS reef habitat surveys

Family	Species	Common name
Acanthuridae	*Acanthurus bahianus*	Ocean surgeonfish
Acanthuridae	*Acanthurus nigrofuscus*	Brown surgeonfish
Acanthuridae	*Acanthurus pyroferus*	Mimic surgeonfish
Apogonidae	*Apogon bandanensis*	Bigeye cardinalfish
Apogonidae	*Cheilodipterus quinquelineatus*	Five-lined cardinalfish
Chaetodontidae	*Amphichaetodon melbae*	Narrow-barred butterflyfish
Chaetodontidae	*Chaetodon aureofasciatus*	Goldern butterflyfish
Chaetodontidae	*Chaetodon plebeius*	Blueblotch butterflyfish
Chaetodontidae	*Chaetodon trifascialis*	Chevron butterflyfish
Chaetodontidae	*Chaetodon ulietensis*	Doublesaddle butterflyfish
Cheilodactylidae	*Cheilodactylus ephippium*	Painted morwong
Diodontidae	*Diodon nichthemerus*	Slender-spined porcupinefish
Epinephelidae	*Plectropomus maculatus*	Bar-cheeked trout
Gobiidae	*Cyptocentrus* sp.	Ventral-barred shrimp goby
Haemulidae	*Diagramma pictum*	Painted sweetlip
Haemulidae	*Plectorhinchus flavomaculatus*	Goldspotted sweetlip
Hemiscylliidae	*Hemiscyllium ocellatum*	Epaulette shark
Labridae	*Bodianus mesothorax*	Split-level hogfish
Labridae	*Cheilinus fasciatus*	Red-breasted wrasse
Labridae	*Choerodon schoenleinii*	Blackspot tuskfish
Labridae	*Coris caudimacula*	Spotted-tail wrasse
Labridae	*Ctenolabrus rupestris*	Goldsinny wrasse
Labridae	*Pseudolabrus guentheri*	Gunther's wrasse
Labridae	*Suezichthys arquatus*	Painted rainbow wrasse
Labridae	*Synodus binotatus*	Two-spot lizardfish
Labridae	*Thalassoma lunare*	Moon wrasse
Lutjanidae	*Lutjanus carponotatus*	Spanish flag snapper
Lutjanidae	*Lutjanus russellii*	Moses' snapper
Nemipteridae	*Scolopsis bilineata*	Two-lined monocle bream
Pomacanthidae	*Chaetodontoplus duboulayi*	Scribbled angelfish
Pomacentridae	*Abudefduf bengalensis*	Bengal sergeant
Pomacentridae	*Abudefduf sexfasciatus*	Scissortail sergeant
Pomacentridae	*Chromis nitida*	Yellowback puller
Pomacentridae	*Chrysiptera cyanea*	Blue demoiselle
Pomacentridae	*Chrysiptera rollandi*	Rolland's demoiselle
Pomacentridae	*Neopomacentrus azysron*	Yellowtail demoiselle
Pomacentridae	*Pomacentrus amboinensis*	Ambon damsel
Pomacentridae	*Pomacentrus bankanensis*	Speckled damselfish
Pomacentridae	*Pomacentrus coelestis*	Neon damselfish
Pomacentridae	*Pomacentrus milleri*	Millers damselfish

Family	Species	Common name
Pomacentridae	*Pomacentrus moluccensis*	Lemon damsel
Pomacentridae	*Pomacentrus wardi*	Wards damsel
Pomacentridae	*Stegastes apicalis*	Yellowtip gregory
Pomacentridae	*Stegastes nigricans*	Dusky gregory
Pomacentridae	*Stegastes* sp.	Gregory damsel
Pseudochromidae	*Pseudochromis fuscus*	Yellow dottyback
Pseudochromidae	*Pseudochromis porphyreus*	Purple dottyback
Pseudochromidae	*Pseudochromis porphyreus*	Magenta dottyback
Pseudochromidae	*Pseudochromis* sp.	Blue dottyback
Pseudochromidae	*Pseudochromis* sp.	Pink dottyback
Serranidae	*Diploprion bifasciatum*	Barred soapfish
Serranidae	*Epinephelus merra*	Dwarf spotted rockcod (wire netting cod)
Siganidae	*Siganus corallinus*	Blue spotted rabbitfish
Sparidae	*Acanthopagrus australis*	Yellowfin bream
Sphyraenidae	*Sphyraena obtusata*	Yellowtail barracuda
Tetraodontidae	*Arothron firmamentum*	Starry toado

Source: Vision Environment (2015a)

Table 3 Species of fish, crustacean and molluscs known from the Calliope River

Family	Species	Common name
Fish		
Ambassidae	*Ambassis marianus*	Estuary perchlet
Ambassidae	*Ambassis vachelli*	Perchlet
Amblyopidae	*Brachyamblyopus* sp.	Worm goby
Anguillidae	*Anguilla reinhardtii*	Long-finned eel
Apogonidae	*Siphamia roseigaster*	Pinkbreast siphonfish
Apogonidae	*Ostorhinchus fasciatus*	Striped cardinalfish
Apogonidae	*Apogon* sp. 1	Cardinalfish
Ariidae	*Arius graeffei*	Blue catfish
Ariidae	*Arius thalassina*	Salmon catfish
Atherinidae	*Atherinomorus vaigiensis*	Common hardyhead
Belonidae	*Strongylura strongylura*	Black spot longtom
Belonidae	*Tylosurus crocodilus*	Crocodile longtom
Carangidae	*Alectis indica*	Diamond trevally
Carangidae	*Caranx melampygus*	Spotted trevally
Carangidae	*Gnathanodon speciosus*	Golden trevally
Carangidae	*Caranx ignobilis*	Giant trevally
Carangidae	*Parastromateus niger*	Black pomfret
Carangidae	*Scomberoides commersonianus*	Giant leatherskin queenfish
Carangidae	*Megalaspis cordyla*	Finny scad
Carangidae	*Selaroides leptolepis*	Smooth-tailed trevally
Carangidae	*Pentolabus radiatus*	Fringe-finned trevally

Family	Species	Common name
Carangidae	*Carangoides caeruleopinnatus*	Onion-ring trevally
Carangidae	*Ulua mentalis*	Cale trevally
Carcharhinidae	*Carcharhinus obscurus*	Black whaler shark
Carcharhinidae	*Carcharhinus leucas*	Bull shark
Carcharhinidae	*Rhizoprionodon acutus*	White-eyed shark
Carcharhinidae	*Carcharhinus fitzroyensis*	Creek whaler shark
Centropomidae	*Lates calcarifer*	Barramundi
Chaetodontidae	*Scatophagus argus*	Spotted scat
Chaetodontidae	*Selenotoca multifasciata*	Butter-fish
Chaetodontidae	*Chaetodon tricinctus*	Three-band coralfish
Chaetodontidae	*Parachaetodon ocellatus*	Ocellate coralfish
Chanidae	*Chanos chanos*	Milkfish
Chirocentridae	*Chirocentrus dorab*	Wolf herring
Clupeidae	*Herklotsichthys castelnaui*	Southern herring
Clupeidae	*Herklotsichthys koningsbergeri*	Spotted herring
Clupeidae	*Nematalosa come*	Hairback herring
Clupeidae	*Nematalosa erebi*	Bony bream
Clupeidae	*Hyperolophus translucidus*	Glassy sprat
Dasyatidae	*Himantura uarnak*	Long-tailed ray
Dasyatidae	*Dasyatis fluviorum*	Brown stingray
Dasyatidae	*Dasyatis kuhlii*	Bluespot stingray
Diodontidae	*Diodon nichthemerus*	Porcupine fish
Drepanidae	*Drepane punctata*	Spotted sickle fish
Echeneididae	*Remora remora*	Remora suckerfish
Elopidae	*Elops hawaiensis*	Giant herring
Engraulidae	*Stephorus* sp.	Anchovy
Engraulidae	*Thryssa hamiltoni*	Hamilton's anchovy
Engraulidae	*Thryssa setirostris*	Longjaw glassnose
Engraulidae	*Thryssa aestuaria*	Estuary anchovy
Ephippidae	*Zabidius nevemaculeatus*	Short-finned batfish
Gerreidae	*Gerres subfasciatus*	Silver biddy
Gerreidae	*Gerres filamentosus*	Threadfin silver-belly
Gerreidae	*Gerres oyena*	Common silver-biddy
Gerreidae	*Gerres subfasciatus*	Common silverbiddy
Gobiidae	*Arenigobius frenatus*	Goby
Gobiidae	*Glossogobius biocellatus*	Sleepy goby
Gobiidae	*Leme* sp.	Burrowing goby
Gobiidae	*Periopthalmus koelreuteri*	Mud-skipper
Gobiidae	*Goby species*	Juvenile too small to identify species
Gymnuridae	*Gymnura australis*	Rat-tailed ray
Haemulidae	*Pomadasys argenteus*	Small spotted javelin fish

Family	Species	Common name
Haemulidae	*Pomadasys kaakan*	Spotted javelin fish
Haemulidae	*Plectorhynchus gibbosus*	Brown morwong
Haemulidae	*Diagramma pictum*	Painted sweetlip
Hemiramphidae	*Arrhamphus sclerolpis*	Snub-nose garfish
Hemiramphidae	*Hemirhamphus robustus*	Three by two garfish
Hemiramphidae	*Hyporhamphus regularis*	River garfish
Hemiramphidae	*Hyporhamphus quoyi*	Short-nosed garfish
Hemiramphidae	*Hyporhamphus australis*	Sea garfish
Hemiramphidae	*Zenarchapterus biffonis*	Buffon's river garfish
Lapridae	*Choerodon cephalotes*	Purple tuskfish
Leiognathidae	*Leiognathus equulus*	Common ponyfish
Leiognathidae	*Leiognathus* sp.	Ponyfish
Leiognathidae	*Leiognathus* sp.	Ponyfish
Lutjanidae	*Lutjanus argentimaculatus*	Mangrove jack
Lutjanidae	*Lutjanus johni*	Golden snapper
Lutjanidae	*Lutjanus russelli*	Moses snapper
Megalopidae	*Megalops cyprinoides*	Tarpon/oxeye herring
Monacanthidae	*Paramonacanthus* sp.	Filefish
Monacanthidae	*Meuschenia* sp. 1	Leatherjacket
Monacanthidae	*Meuschenia* sp. 2	Leatherjacket
Monacanthidae	*Meuschenia* sp. 3	Leatherjacket
Monacanthidae	*Meuschenia* sp. 4	Leatherjacket
Monodactylidae	*Monodactylus argenteus*	Diamond fish
Monodactylidae	*Leptobramidae*	Beach salmon
Monodactylidae	*Leptobrama muelleri*	Beach salmon
Mugilidae	*Liza subviridis*	Greenback mullet
Mugilidae	*Liza argentea*	Flat-tail mullet
Mugilidae	*Liza vaigiensis*	Diamond scale mullet
Mugilidae	*Mugil cephalus*	Sea mullet
Mugilidae	*Paramugil georgii*	Fantail mullet
Mugilidae	*Myxus elongatus*	Sand mullet
Mugilidae	*Valamugil seheli*	Blue-tailed mullet
Mullidae	*Upeneus tragula*	Bar-tailed goatfish
Muraenesocidae	*Muraenesox cinereus*	Pike eel
Myliobatidae	*Aetobatus narinari*	White spotted eagle ray
Nemipteridae	*Nemipterus theodorei*	Yellow-lip butterfly bream
Nemipteridae	*Pentapodus paradiseus*	Blue-faced whiptail
Paralichthyidae	*Pseudorhombus arsius*	Large-toothed flounder
Paralichthyidae	*Pseudorhombus argent*	Flounder
Platycephalidae	*Platycephalus fuscus*	Dusky flathead
Platycephalidae	*Platycephalus indicus*	Bar-tailed flathead

Family	Species	Common name
Platycephalidae	*Suggrundus* sp.	Flathead
Platycephalidae	*Platycephalus arenius*	Sand flathead
Polynemidae	*Eleutheronerna tetradactylum*	Blue threadfin
Polynemidae	*Polydactylus sheridani*	King salmon
Polynemidae	*Polydactylus macrochir*	King threadfin
Rachycentridae	*Rachycentron canadum*	Cobia
Rhinobatidae	*Aptychotrema rostrata*	Eastern shovelnose ray
Scatophagidae	*Selenotoca multifasciata*	Striped butterfish
Sciaenidae	*Argyrosomus japonicus*	Jewfish/Mulloway
Sciaenidae	*Protonibea diacanthus*	Black jewfish
Scombridae	*Scomberomorus semifasciatus*	Broad-barred mackerel
Scombridae	*Scomberomorus queenslandicus*	School mackerel
Scombridae	*Auxis thazard*	Frigate mackerel
Scorpaenidae	*Notesthes robusta*	Bullrout
Scorpaenidae	*Scorpaenidae* sp.	Scorpion fish
Scorpaenidae	*Paracentropogon vespa*	Spot fin waspfish
Scorpaenidae	*Scorpaenopsis* sp.	Scorpionfish
Serranidae	*Ephinephelus coioides*	Estuary cod
Serranidae	*Ephinephelus malabaricus*	Malabar cod
Serranidae	*Ephinephelus quoyanus*	Longfin rockcod
Serranidae	*Ephinephelus lanceolatus*	Queensland groper
Serranidae	*Cephalopholis cyanostigma*	Blue spotted rock cod
Siganidae	*Siganus guttatus*	Golden-lined spinefoot
Siganidae	*Siganus rivulatus*	Happy moments/Marbled spinefoot
Sillaginidae	*Sillago analis*	Golden-lined whiting
Sillaginidae	*Sillago maculata*	Winter whiting
Sillaginidae	*Sillago ciliata*	Sand whiting
Soleidae	*Achlyopa nigra*	Black sole
Soleidae	*Phylichthys sclerolepsis*	Sharp-headed sole
Sparidae	*Acanthopagrus australis*	Yellowfin bream
Sparidae	*Acanthopagrus berda*	Black/pikey bream
Sparidae	*Acanthopagrus* sp. 2	Bream
Sparidae	*Lethrinus laticaudis*	Grass emporer
Sphyraenidae	*Sphyraena barracuda*	Great barracuda
Sphyraenidae	*Sphyraena jello*	Pickhandle barracuda
Synodontidae	*Saurida undosquamis*	Large-scaled grinner
Syngnathidae	*Sygnathidae* sp. 1	Pipefish
Syngnathidae	*Hippocampus* sp. 1	Seahorse
Synanceiidae	*Synanceia horrida*	Estuarine stonefish
Terapontidae	*Pelates quadrilineatus*	Trumpeter
Terapontidae	*Pelates sexlineatus*	Eastern striped grunter

Family	Species	Common name
Terapontidae	*Amniataba percoides*	Barred grunter
Terapontidae	*Terapon jarbua*	Crescent perch
Tetraodontidae	*Marilyna pleurostictus*	Banded toadfish
Tetraodontidae	*Chelonodon patoca*	Milk-spotted puffer
Tetraodontidae	*Torquigener* sp.	Toadfish
Tetraodontidae	*Tetractenos hamiltoni*	Common toadfish
Tetraodontidae	*Arothron manilensis*	Narrow-lined toadfish
Triacanthodidae	*Tripodichthys angustifrons*	Yellow-fin tripod fish
Triacanthodidae	*Triacanthus biaculeatus*	Short-nosed tripod fish
Crustaceans		
Alpheidae	*Apheus* sp.	Clicker prawn
Callianassidae	*Trypaea australiensis*	Yabby
Palaemonidae	*Macrobrachium* sp.	Shrimp
Penaeidae	*Melicertus plebejus*	Eastern king prawn
Penaeidae	*Fenneropenaeus merguiensis*	Banana prawn
Penaeidae	*Metapenaeus ensis*	Greasyback prawn
Penaeidae	*M. bennettae*	Bay prawn
Penaeidae	*Trachypenaeus fulvus*	Hardback prawn
Portunidae	*Portunus pelagicus*	Blue swimmer crab
Portunidae	*Scylla serrata*	Mud crab
Sergestidae	*Acetes* sp.	Shrimp
Squillidae	*Squilla* sp.	Mantis shrimp
Molluscs		
Batillariidae	*Telescopium telescopium*	Mud whelks
Loliginidae	*Loliolus* sp.	Bay squid
Sepiolidae	*Euprymna* sp.	Mickey mouse squid
Sepiolidae	*Idiosepius* sp.	Seagrass squid

Source: Adapted from NPRSR (2014)

Table 4 Species of fish, crustacean and molluscs considered to be of economic importance from the Calliope River

Family	Species	Common name
Fish		
Anguillidae	*Anguilla reinhardtii*	Long-finned eel
Ariidae	*Ariidae* 2 spp.	Catfish
Belonidae	*Strongylura strongylura*	Black spot longtom
Carangidae	*Monodactylus argenteus*	Diamond fish
Carangidae	*Scomberoides commersonianus*	Giant leatherskin queenfish
Carangidae	*Carangidae* 4 spp.	Trevally
Centropomidae	*Lates calcarifer*	Barramundi
Chaetodontidae	*Scatophagus argus*	Spotted scat
Chirocentridae	*Chirocentrus dorab*	Wolf herring

Family	Species	Common name
Drepanidae	*Drepane punctata*	Spotted sickle fish
Haemulidae	*Haemulidae* 4 spp.	Javelin fish
Hemiramphidae	*Zenarchapterus biffonis*	Buffon's garfish
Hemiramphidae	*Hemiramphidae* 6 spp.	Garfish
Lapridae	*Choerodon cephalotes*	Purple tuskfish
Lutjanidae	*Lutjanidae* 3 spp.	Tropical snapper species
Megalopidae	*Megalops cyprinoides*	Oxeye herring
Monodactylidae	*Leptobrama muelleri*	Beach salmon
Mugilidae	*Mugilidae* 7 spp.	Mullet
Myliobatidae	*Aetobatus narinari*	White spotted eagle ray
Platycephalidae	*Platycephalidae* 4 spp.	Flathead
Polynemidae	*Polynemidae* 3 spp.	Threadfin
Rachycentridae	*Rachycentron canadum*	Cobia
Rhinobatidae	*Aptychotrema rostrata*	Eastern shovelnose ray
Scatophagidae	*Selenotoca multifasciata*	Butterfish
Sciaenidae	*Protonibea diacanthus*	Black jewfish
Sciaenidae	*Argyrosomus japonicus*	Jewfish/Mulloway
Scombridae	*Scombridae* 3 spp.	Mackerel
Serranidae	*Serranidae* 5 spp.	Groper and Cod species
Sillaginidae	*Sillaginidae* 3 spp.	Whiting
Sparidae	*Sparidae* 4 spp.	Bream species
Crustaceans		
Portunidae	*Portunus pelagicus*	Blue swimmer crab
Portunidae	*Scylla serrata*	Mud crab
Panaeidae	*Panaeidae* 5 spp.	Prawns

Source: Adapted from NPRSR (2014)

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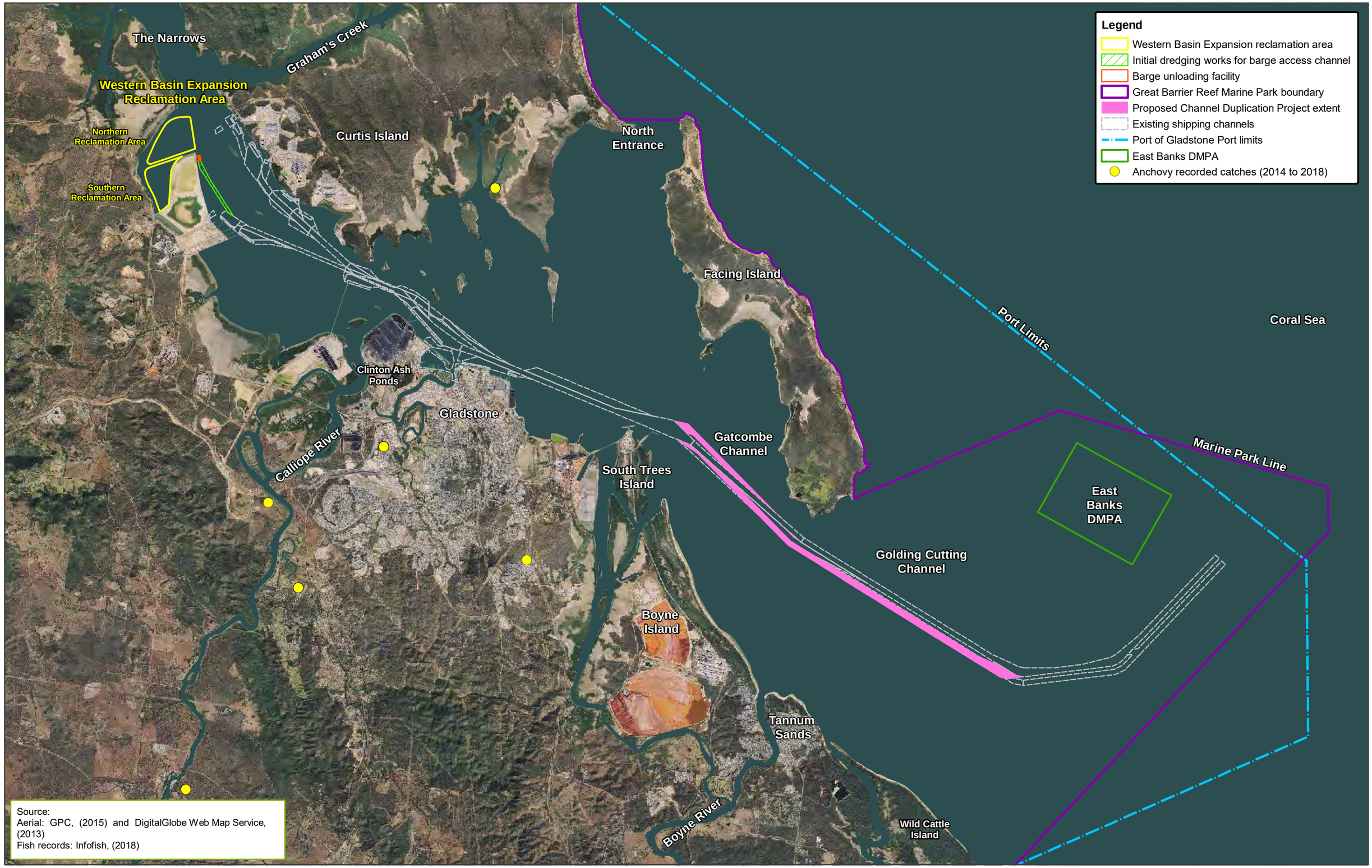

Source:
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Fish records: Infofish, (2018)

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Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L1: Anchovy recorded catches (2014 to 2018)

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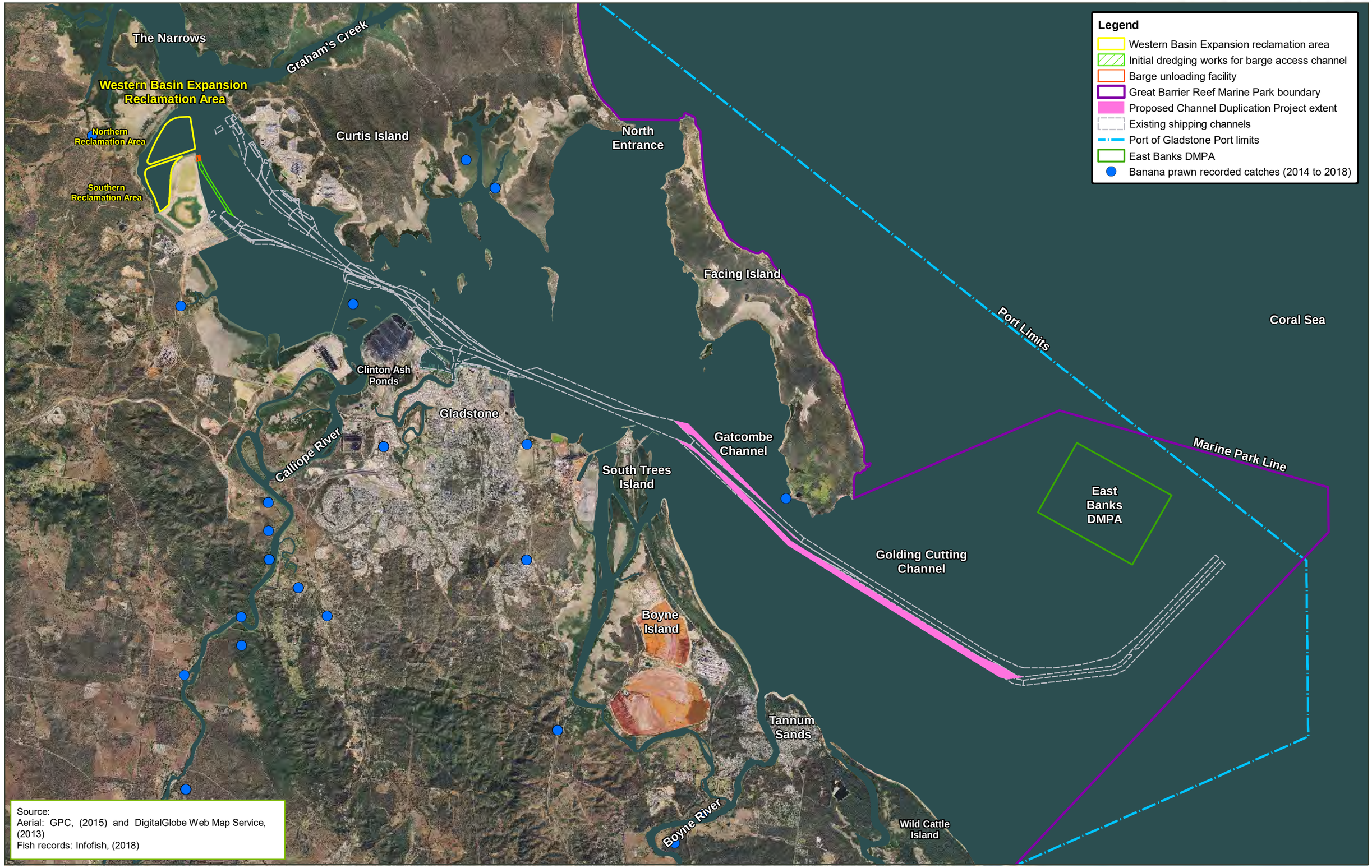

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Metres
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Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L2: Banana prawn recorded catches (2014 to 2018)

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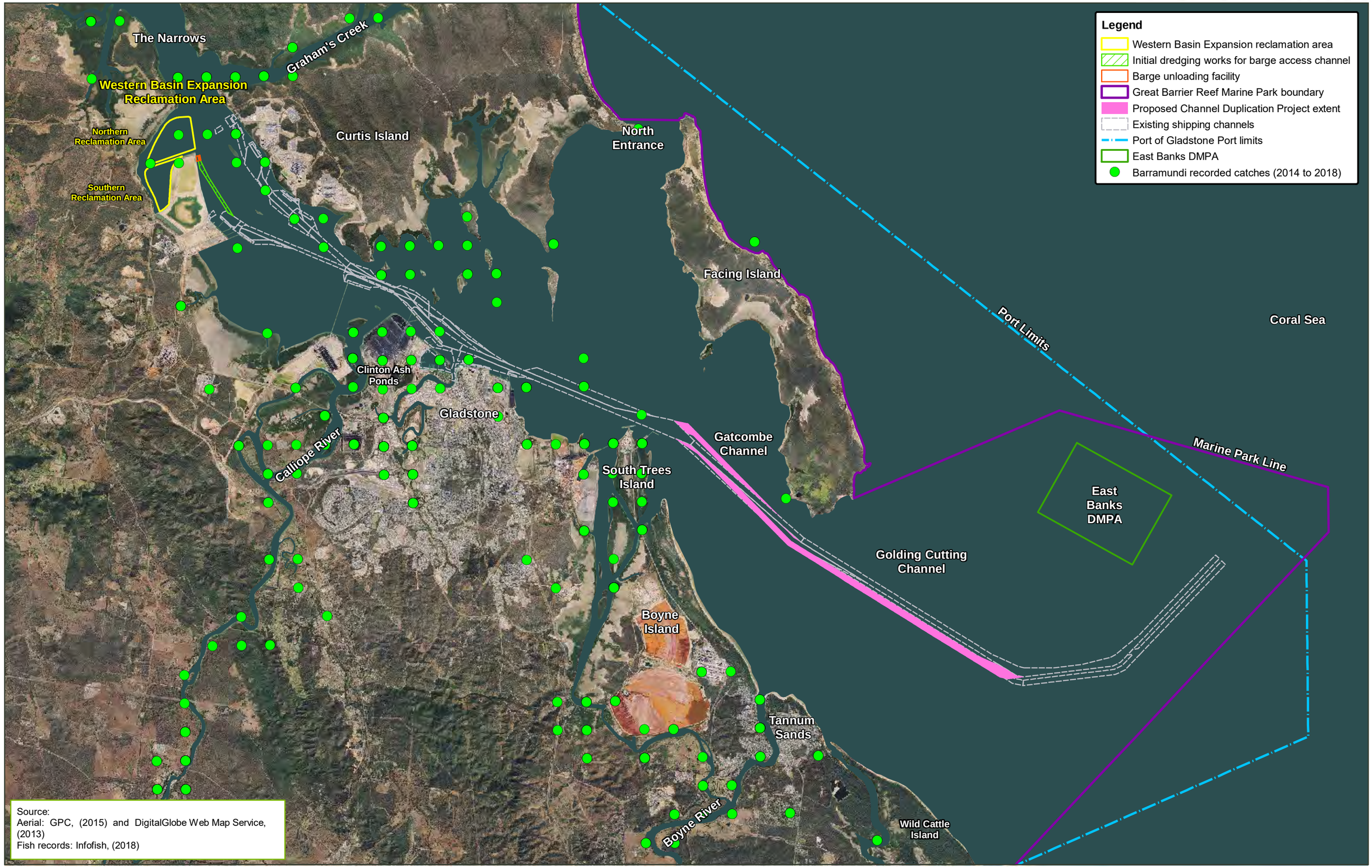

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Fish records: Infofish, (2018)

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0 1,500 3,000 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L3: Barramundi recorded catches (2014 to 2018)

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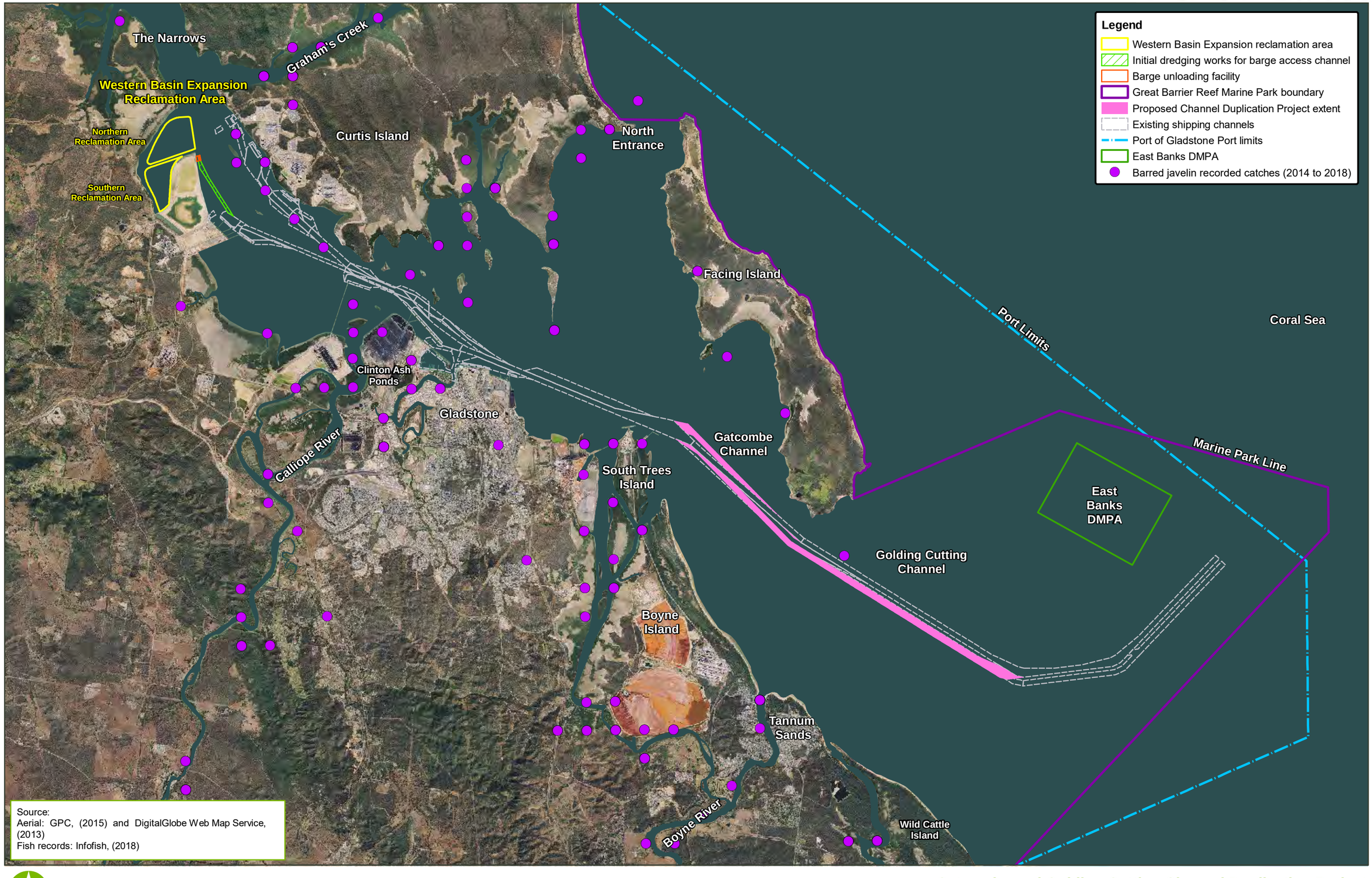

Source:
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Fish records: Infofish, (2018)

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0 1,500 3,000 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

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Figure L4: Barred javelin recorded catches (2014 to 2018)

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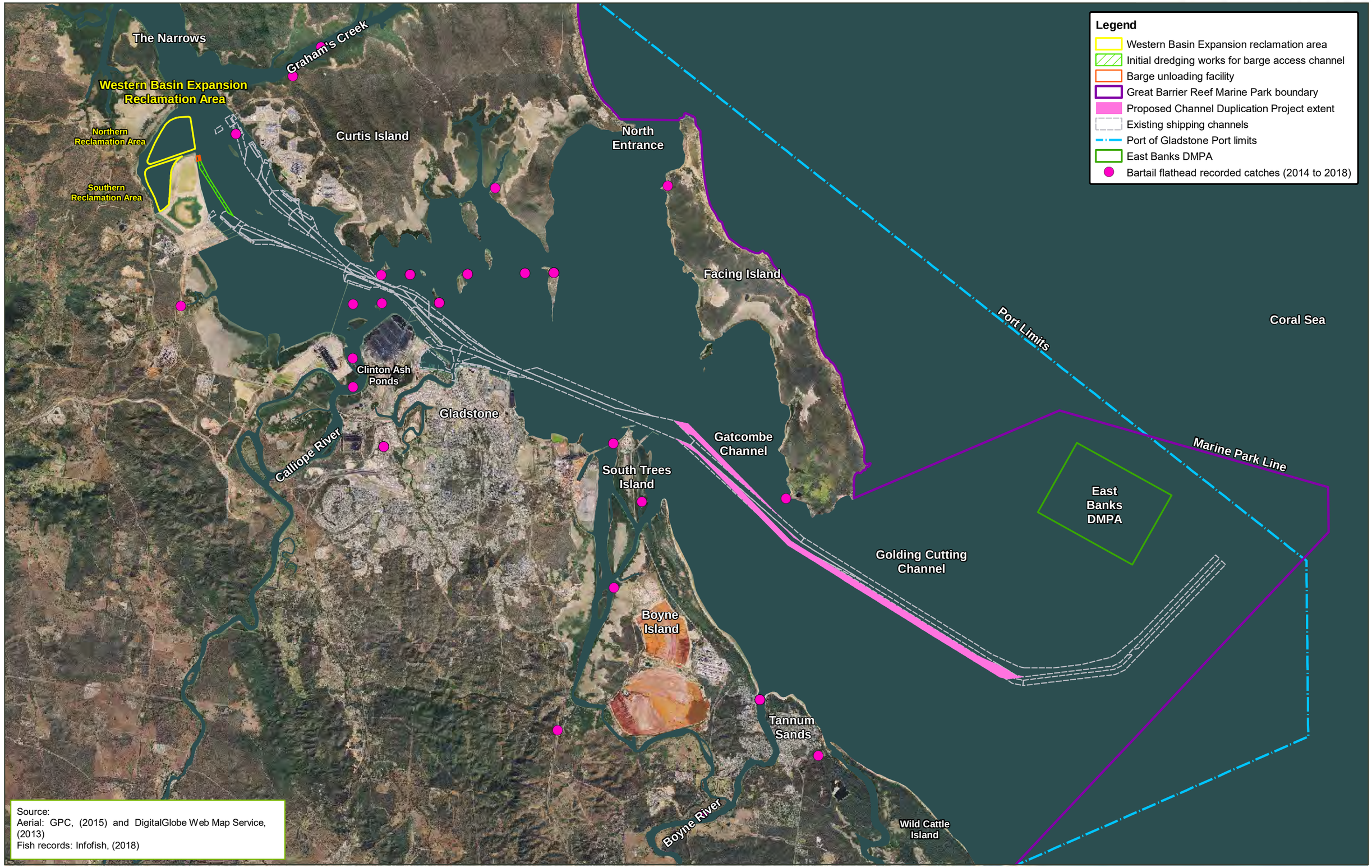

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Fish records: Infofish, (2018)

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Metres
0 1,500 3,000

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

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Figure L5: Bartail flathead recorded catches (2014 to 2018)

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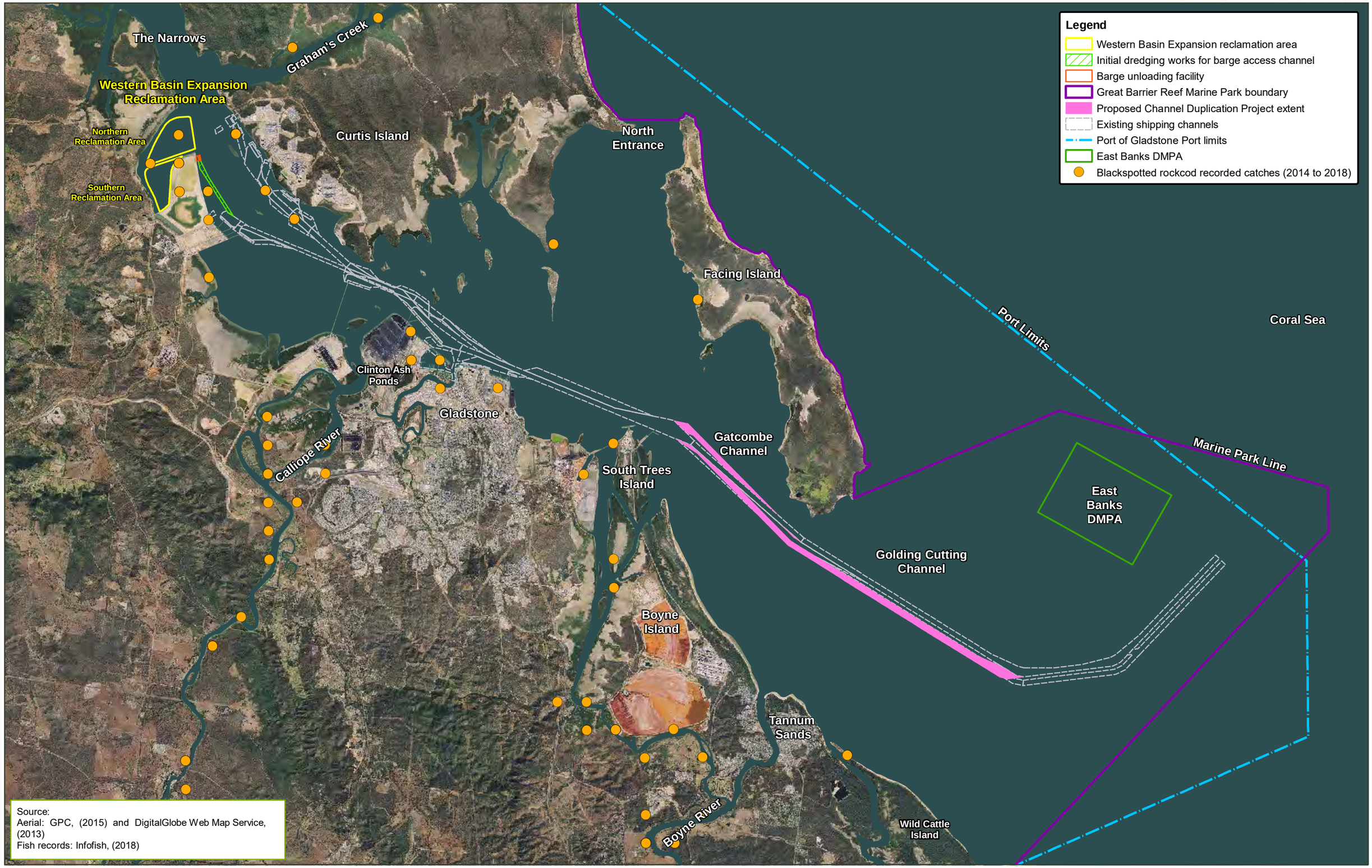

Source:
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Fish records: Infofish, (2018)

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0 1,500 3,000 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

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Figure L6: Blackspotted rockcod recorded catches (2014 to 2018)

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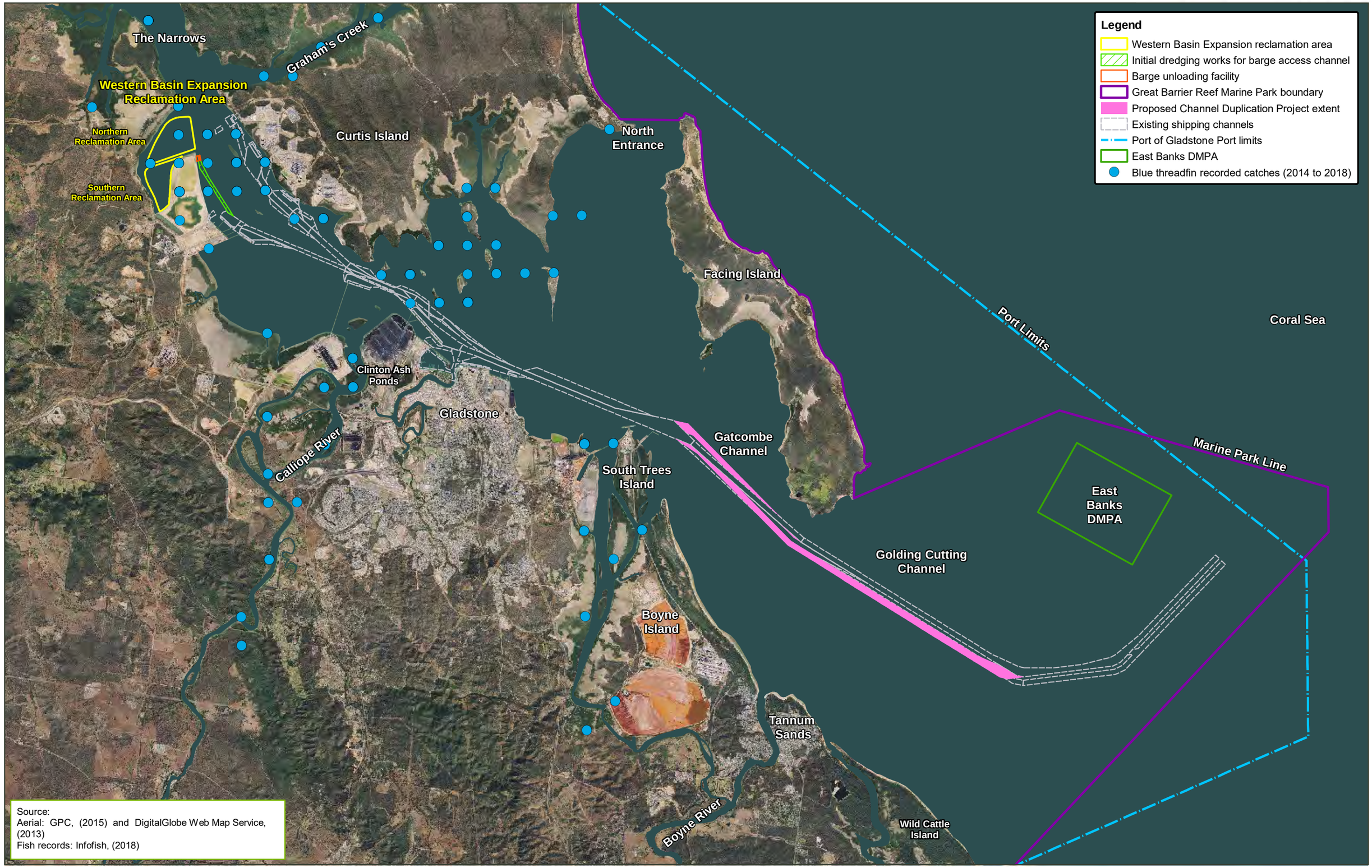

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Fish records: Infofish, (2018)

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0 1,500 3,000 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

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Figure L7: Blue threadfin recorded catches (2014 to 2018)

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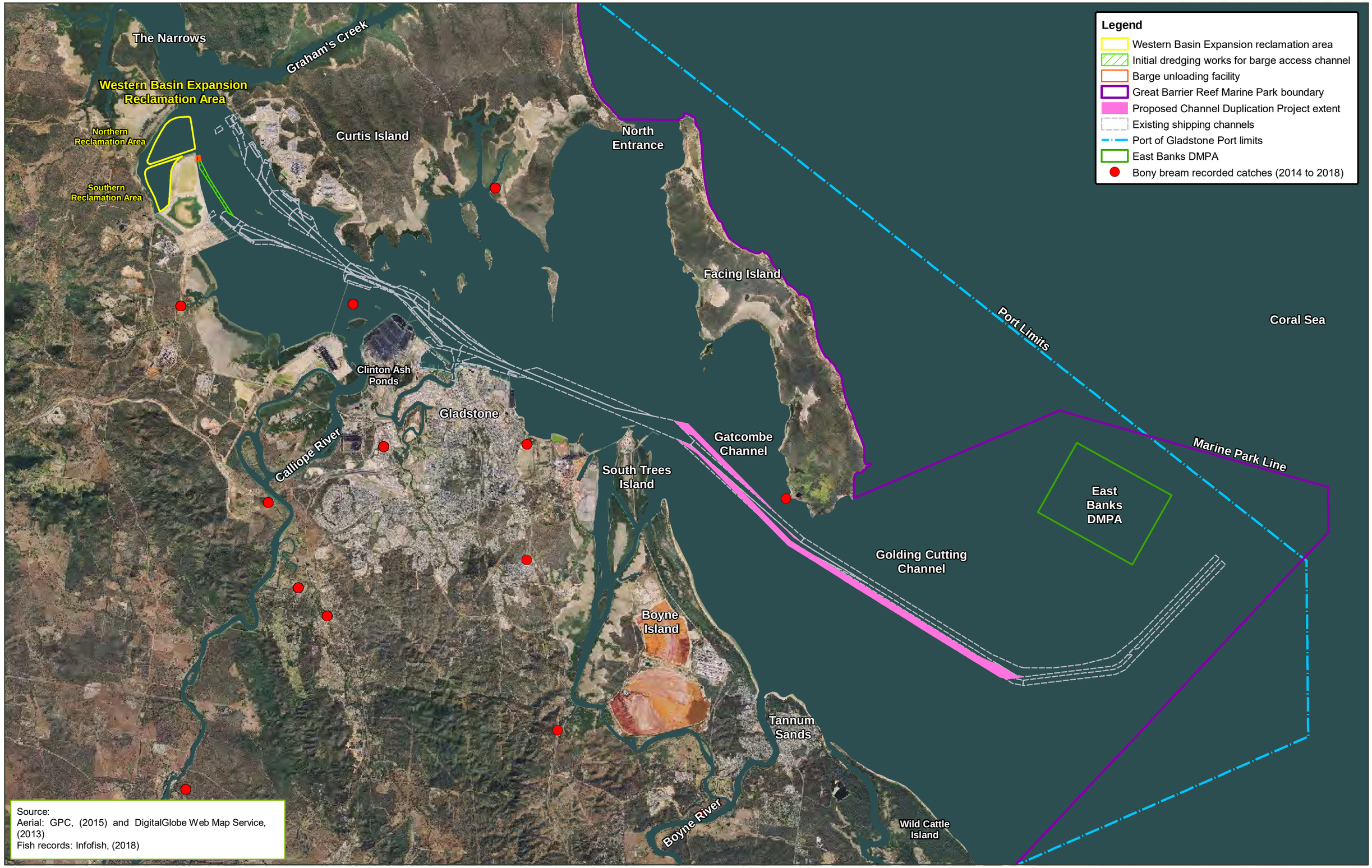

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Fish records: Infofish, (2018)

Map by: RB
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0 1,500 3,000 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L8: Bony bream recorded catches (2014 to 2018)

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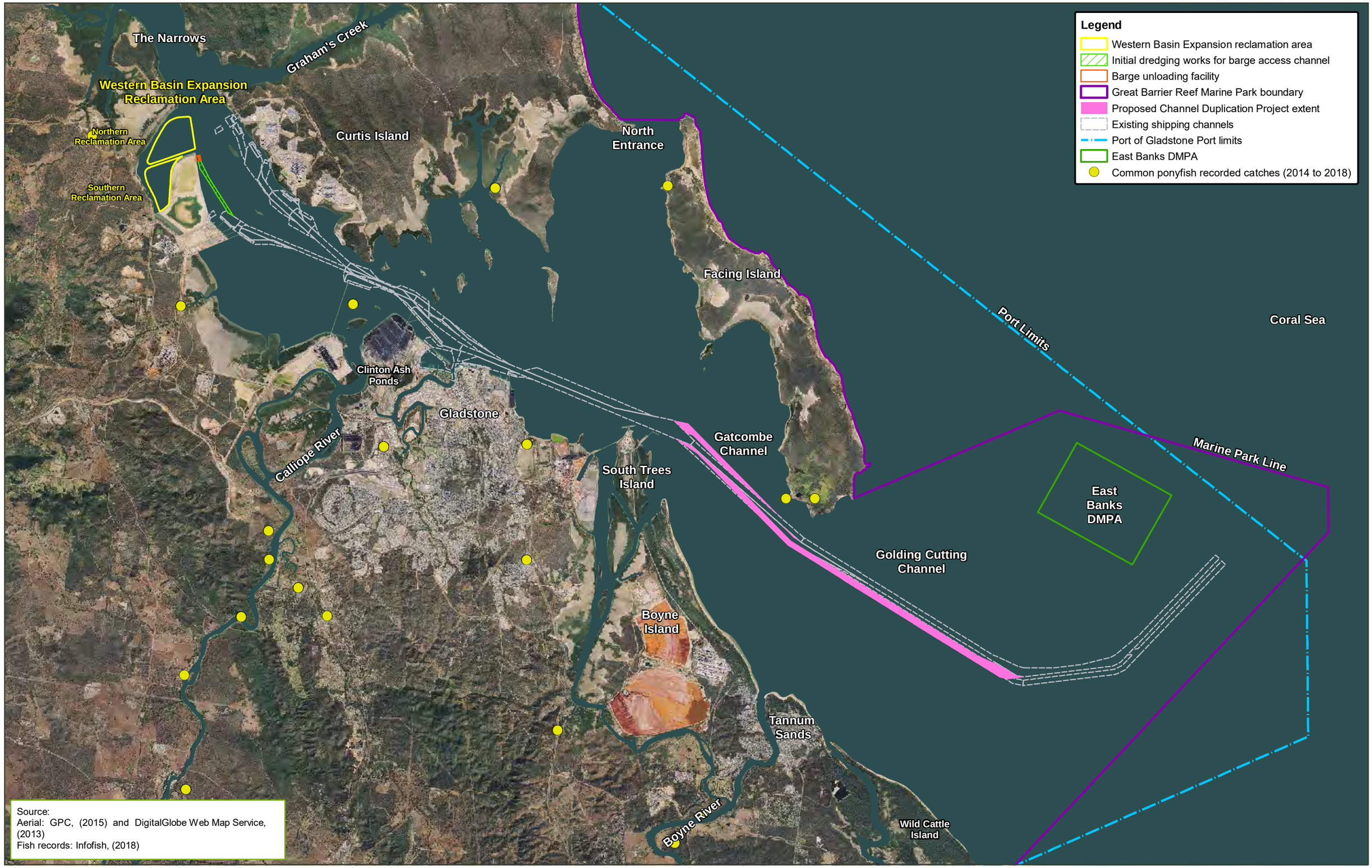

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0 1,500 3,000 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L9: Common ponyfish recorded catches (2014 to 2018)

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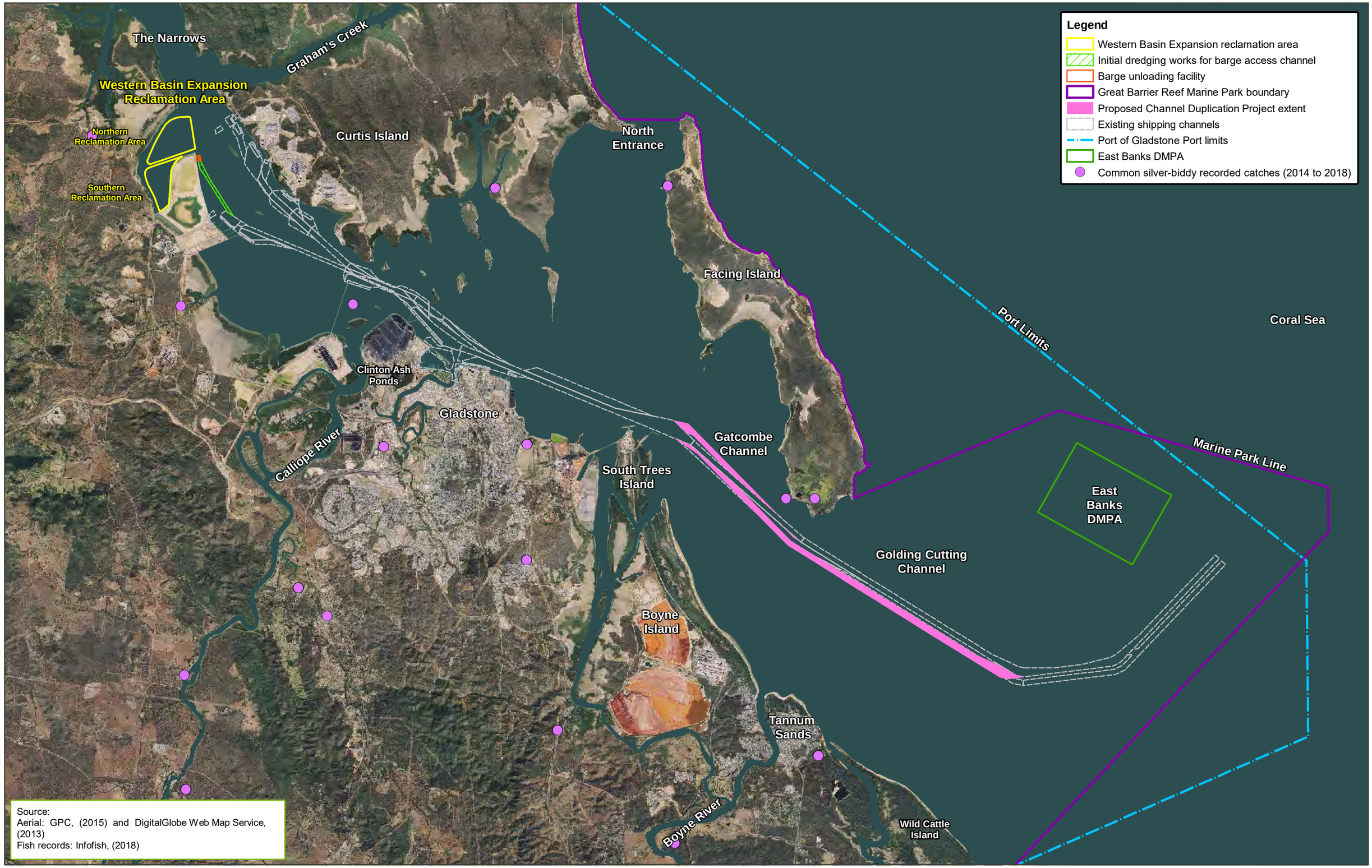

Source:
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0 1,500 3,000 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L10: Common silver-biddy recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

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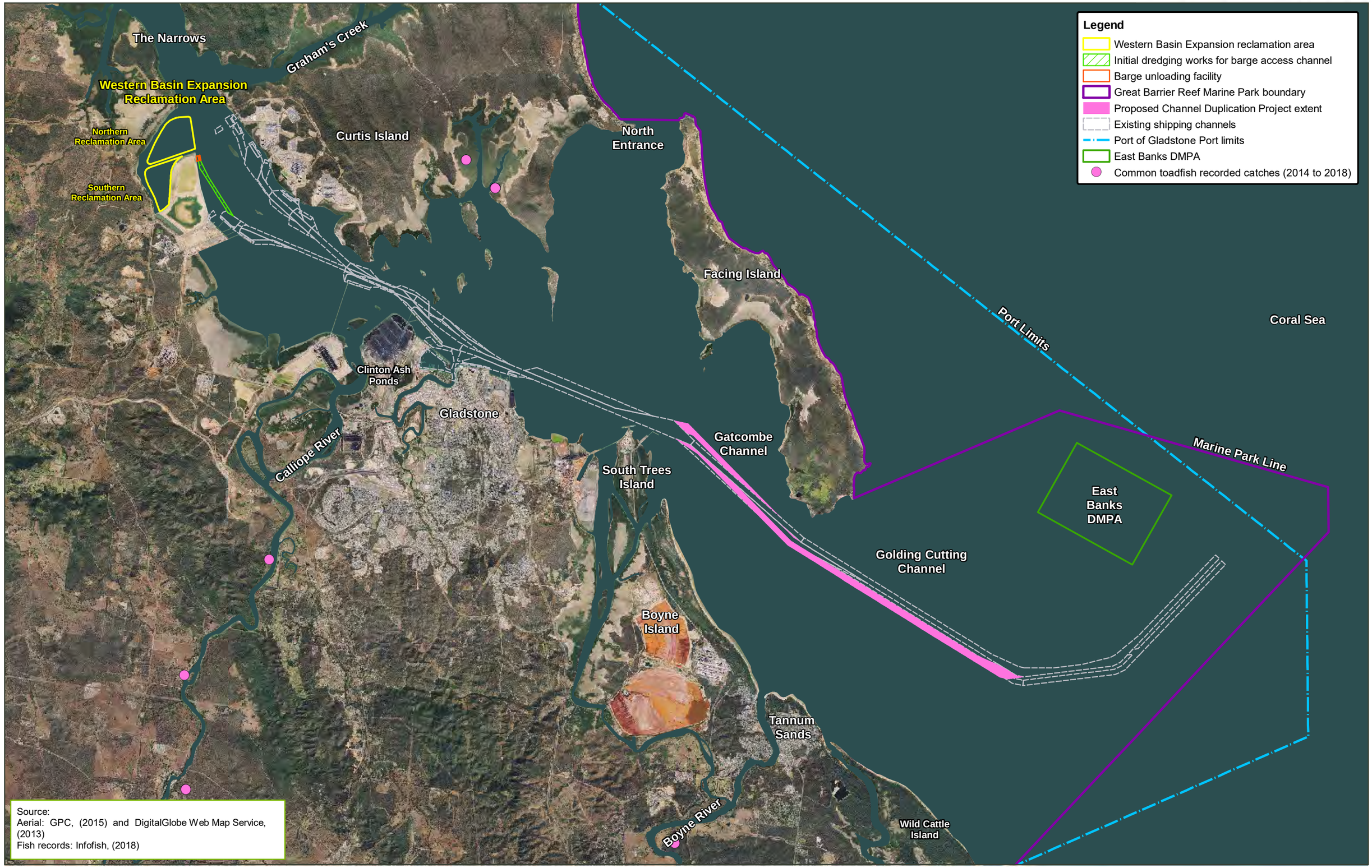

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Fish records: Infofish, (2018)

Map by: RB
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0 1,500 3,000 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L11: Common toadfish recorded catches (2014 to 2018)

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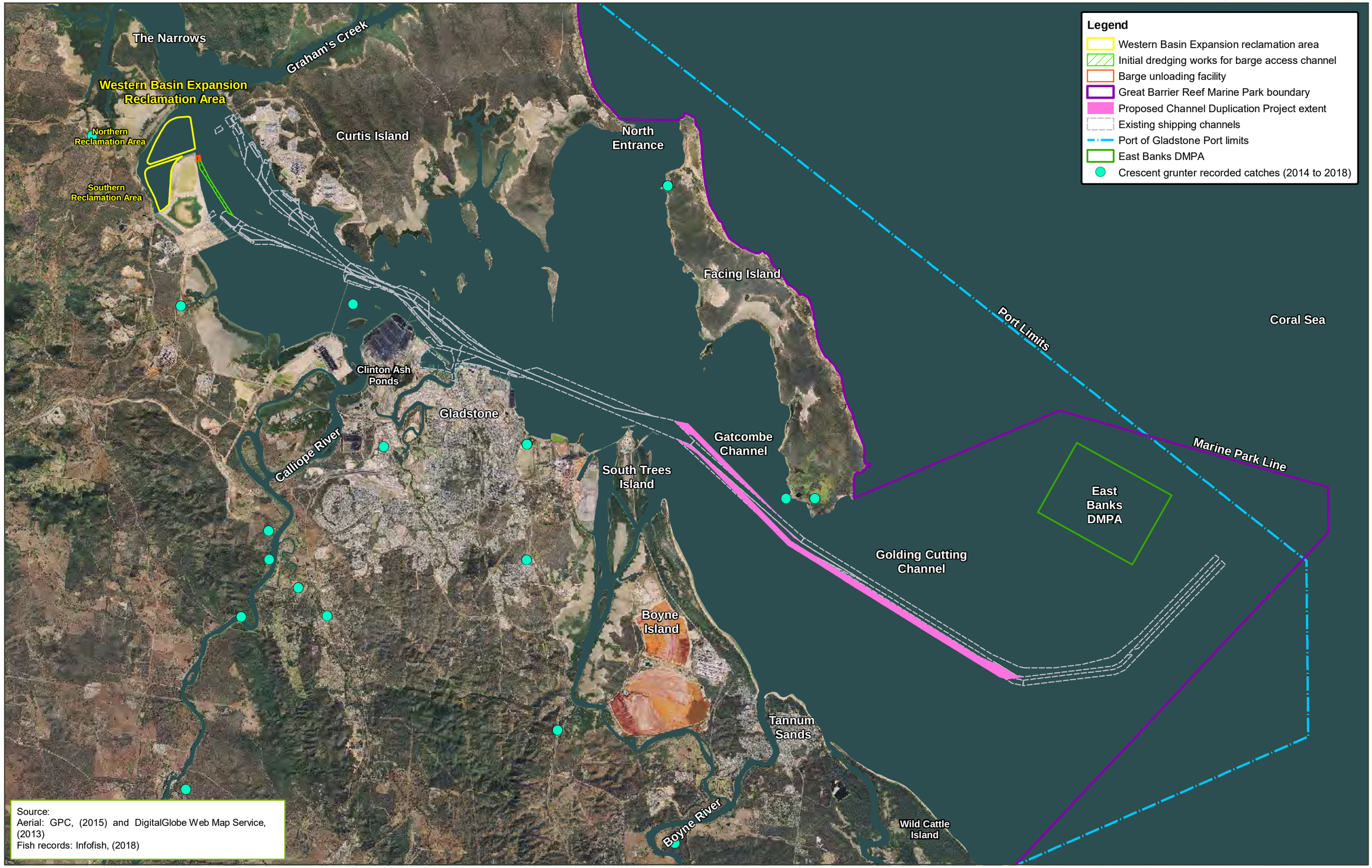

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Map by: RB
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0 1,500 3,000 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L12: Crescent grunter recorded catches (2014 to 2018)

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The Narrows
Graham's Creek
Western Basin Expansion Reclamation Area
Northern Reclamation Area
Southern Reclamation Area
Curtis Island
North Entrance
Facing Island
Port Limits
Coral Sea
Clinton Ash Ponds
Gladstone
Calliope River
Gatcombe Channel
South Trees Island
Marine Park Line
East Banks DMPA
Golding Cutting Channel
Boyne Island
Tannum Sands
Boyne River
Wild Cattle Island

Legend
- Western Basin Expansion reclamation area
- Initial dredging works for barge access channel
- Barge unloading facility
- Great Barrier Reef Marine Park boundary
- Proposed Channel Duplication Project extent
- Existing shipping channels
- Port of Gladstone Port limits
- East Banks DMPA
- Diamond scale mullet recorded catches (2014 to 2018)

Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
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0 1,500 3,000 Metres

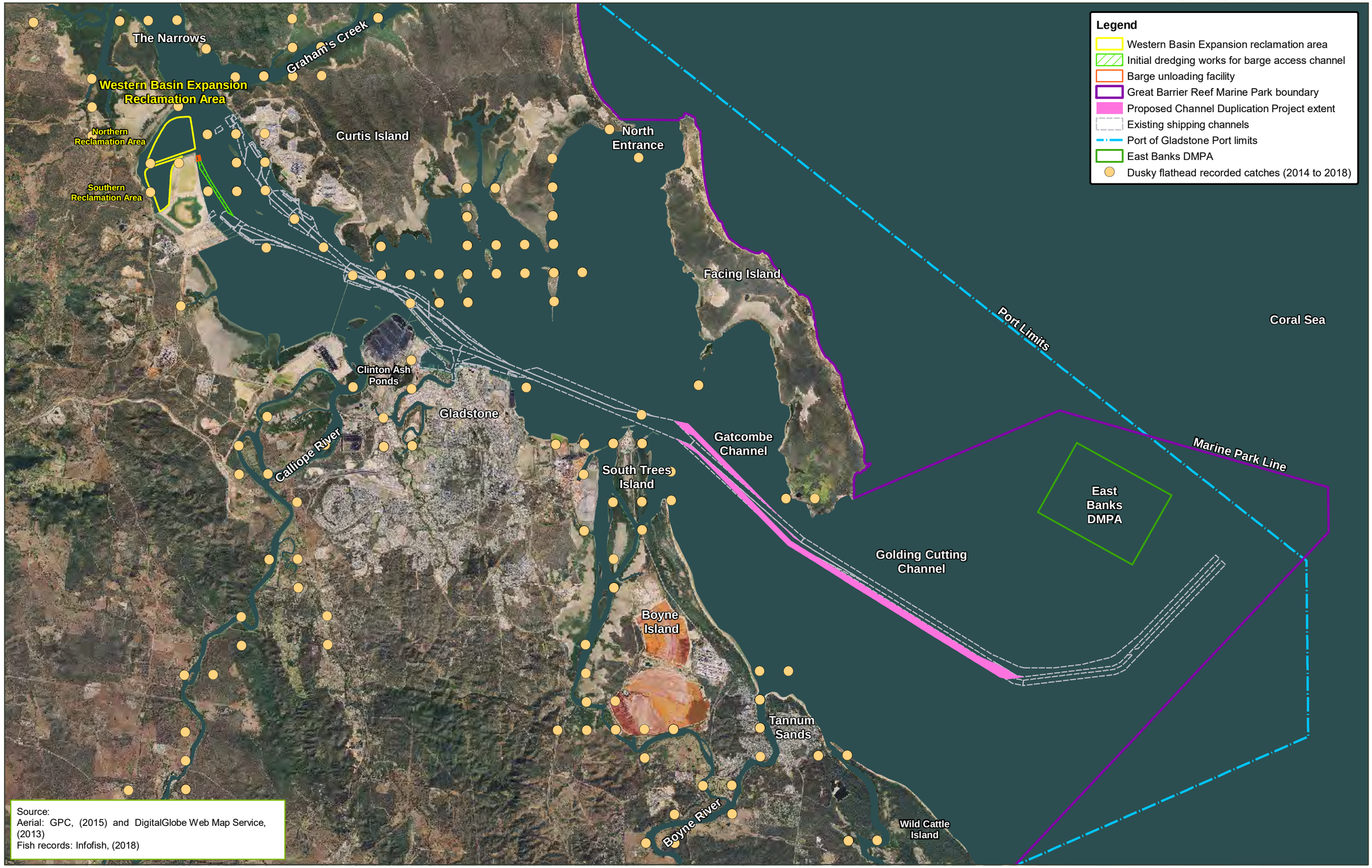
Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L13: Diamond scale mullet recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon



Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_327.mxd 24/01/2019 14:07

0 1,500 3,000 Metres

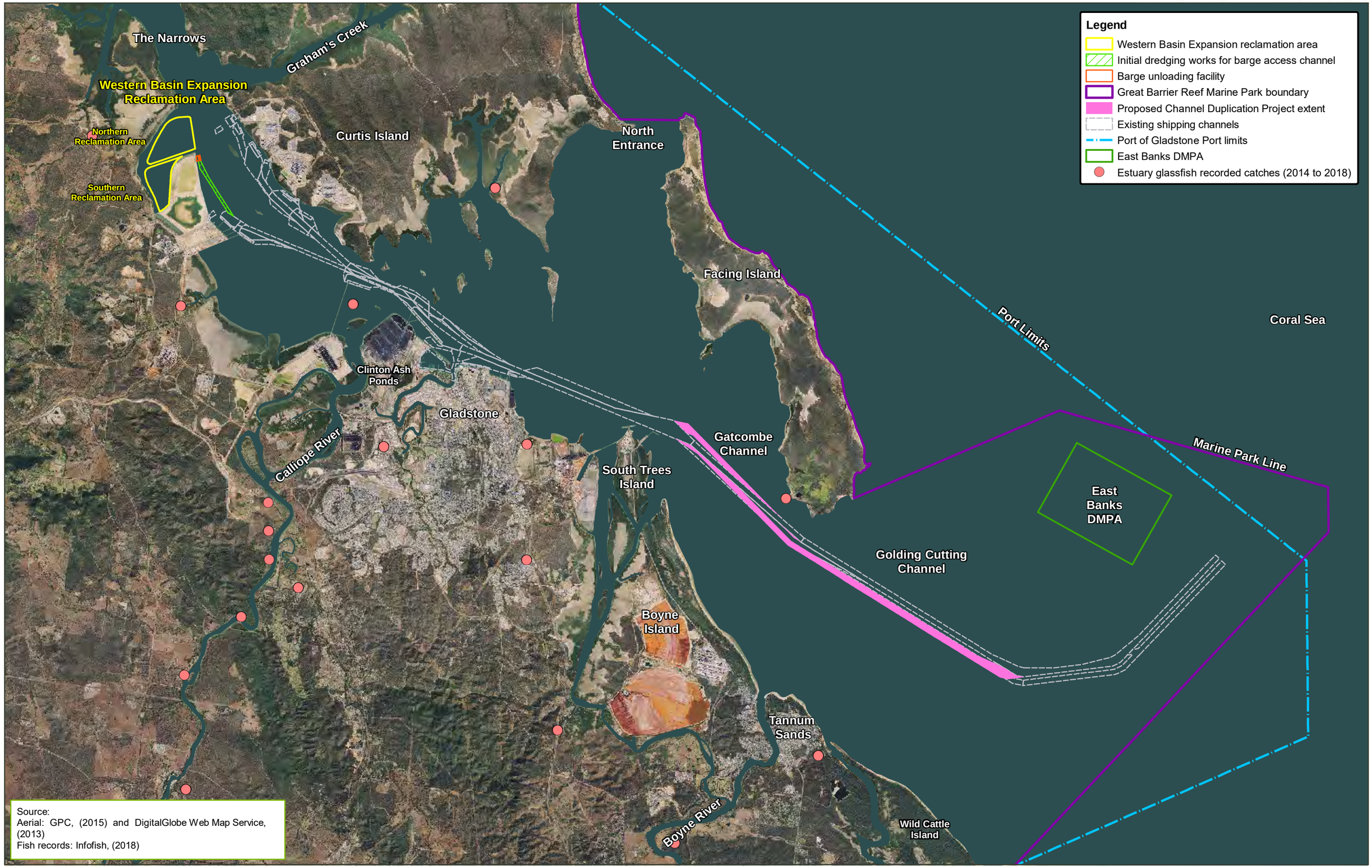
Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L14: Dusky flathead recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon



Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_328.mxd 24/01/2019 14:08

0 1,500 3,000 Metres

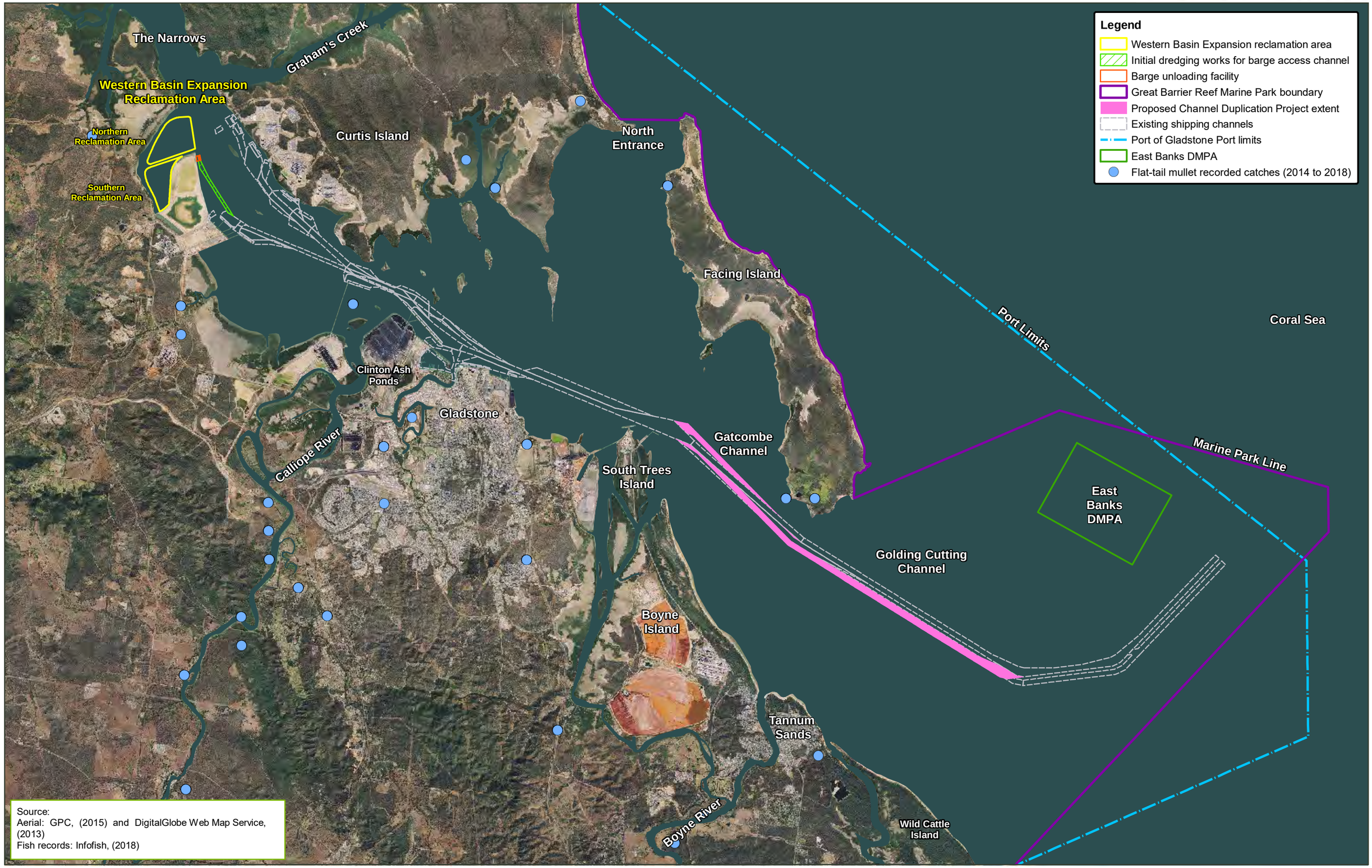
Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L15: Estuary glassfish recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon



Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_329.mxd 24/01/2019 14:35

0 1,500 3,000 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L16: Flat-tail mullet recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon

The Narrows
Graham's Creek
Western Basin Expansion Reclamation Area
Northern Reclamation Area
Southern Reclamation Area
Curtis Island
North Entrance
Facing Island
Port Limits
Coral Sea
Clinton Ash Ponds
Gladstone
Calliope River
Gatcombe Channel
South Trees Island
Marine Park Line
East Banks DMPA
Golding Cutting Channel
Boyne Island
Tannum Sands
Boyne River
Wild Cattle Island

Legend
- Western Basin Expansion reclamation area
- Initial dredging works for barge access channel
- Barge unloading facility
- Great Barrier Reef Marine Park boundary
- Proposed Channel Duplication Project extent
- Existing shipping channels
- Port of Gladstone Port limits
- East Banks DMPA
- Golden snapper recorded catches (2014 to 2018)

Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_330.mxd 24/01/2019 14:38
Map by: RB

0 1,500 3,000 Metres

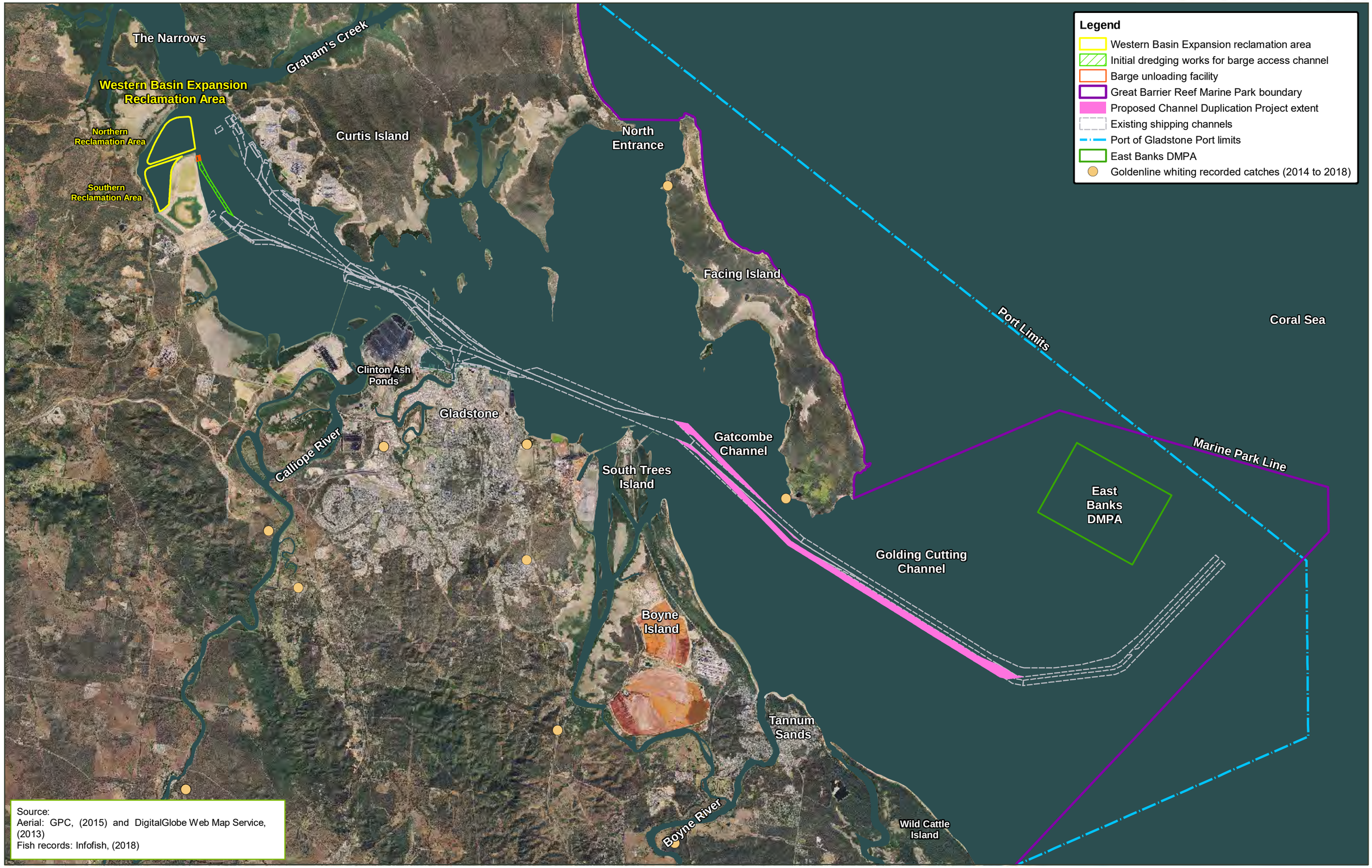
Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L17: Golden snapper recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon



Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_331.mxd 24/01/2019 14:39

0 1,550 3,100 Metres

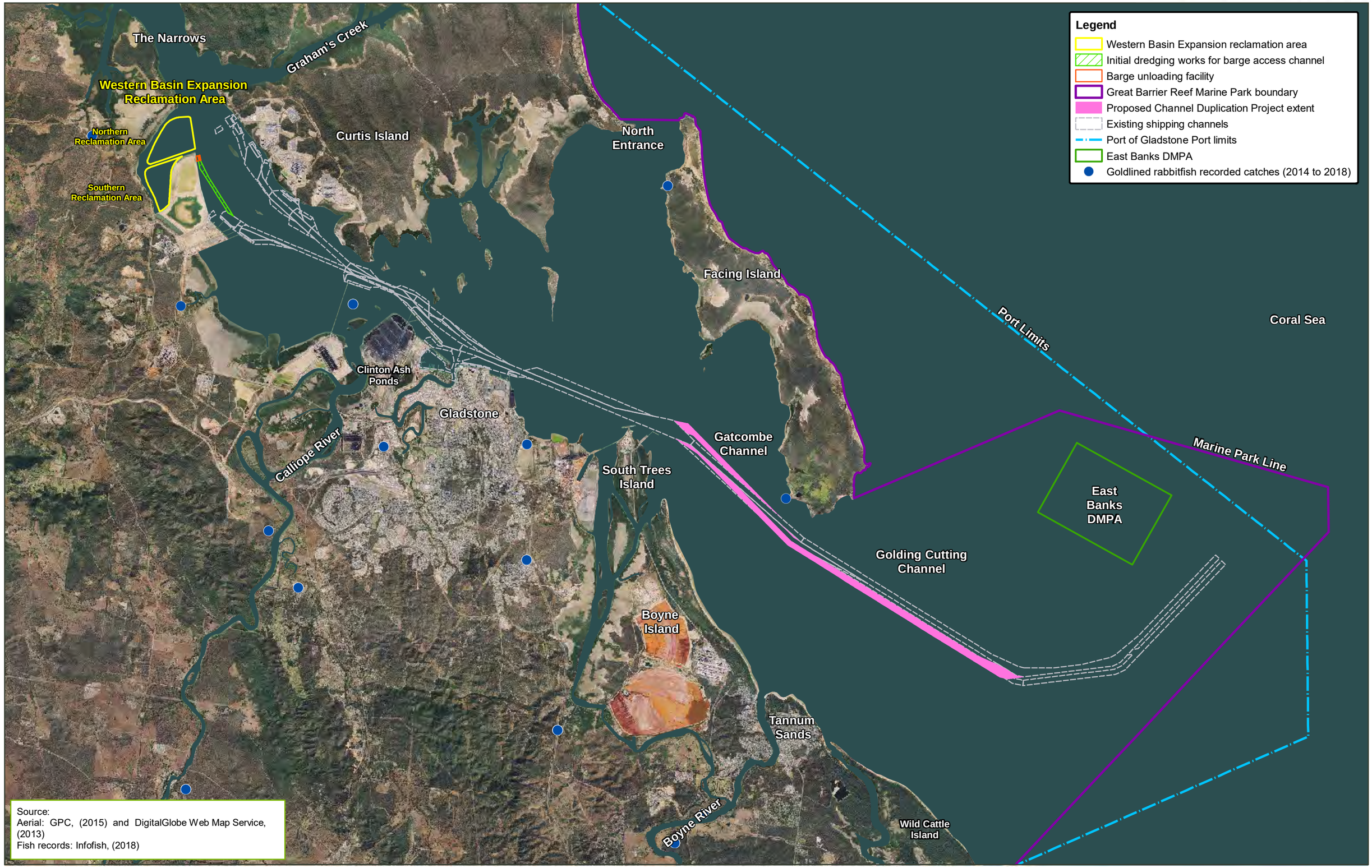
Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L18: Goldenline whiting recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon



Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_332.mxd 24/01/2019 14:40

0 1,550 3,100 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L19: Goldlined rabbitfish recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

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The Narrows
Graham's Creek
Western Basin Expansion Reclamation Area
Northern Reclamation Area
Southern Reclamation Area
Curtis Island
North Entrance
Facing Island
Port Limits
Coral Sea
Clinton Ash Ponds
Gladstone
Calliope River
Gatcombe Channel
South Trees Island
Marine Park Line
East Banks DMPA
Golding Cutting Channel
Boyne Island
Tannum Sands
Boyne River
Wild Cattle Island

Legend
- Western Basin Expansion reclamation area
- Initial dredging works for barge access channel
- Barge unloading facility
- Great Barrier Reef Marine Park boundary
- Proposed Channel Duplication Project extent
- Existing shipping channels
- Port of Gladstone Port limits
- East Banks DMPA
- Goldspotted rockcod recorded catches (2014 to 2018)

Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_333.mxd 24/01/2019 14:40

0 1,550 3,100 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L20: Goldspotted rockcod recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon

The Narrows
Graham's Creek
Western Basin Expansion Reclamation Area
Northern Reclamation Area
Southern Reclamation Area
Curtis Island
North Entrance
Facing Island
Port Limits
Coral Sea
Clinton Ash Ponds
Gladstone
Calliope River
Gatcombe Channel
South Trees Island
Marine Park Line
East Banks DMPA
Golding Cutting Channel
Boyne Island
Tannum Sands
Boyne River
Wild Cattle Island

Legend

- Western Basin Expansion reclamation area
- Initial dredging works for barge access channel
- Barge unloading facility
- Great Barrier Reef Marine Park boundary
- Proposed Channel Duplication Project extent
- Existing shipping channels
- Port of Gladstone Port limits
- East Banks DMPA
- Mangrove jack recorded catches (2014 to 2018)

Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_334.mxd 24/01/2019 14:41

0 1,550 3,100 Metres

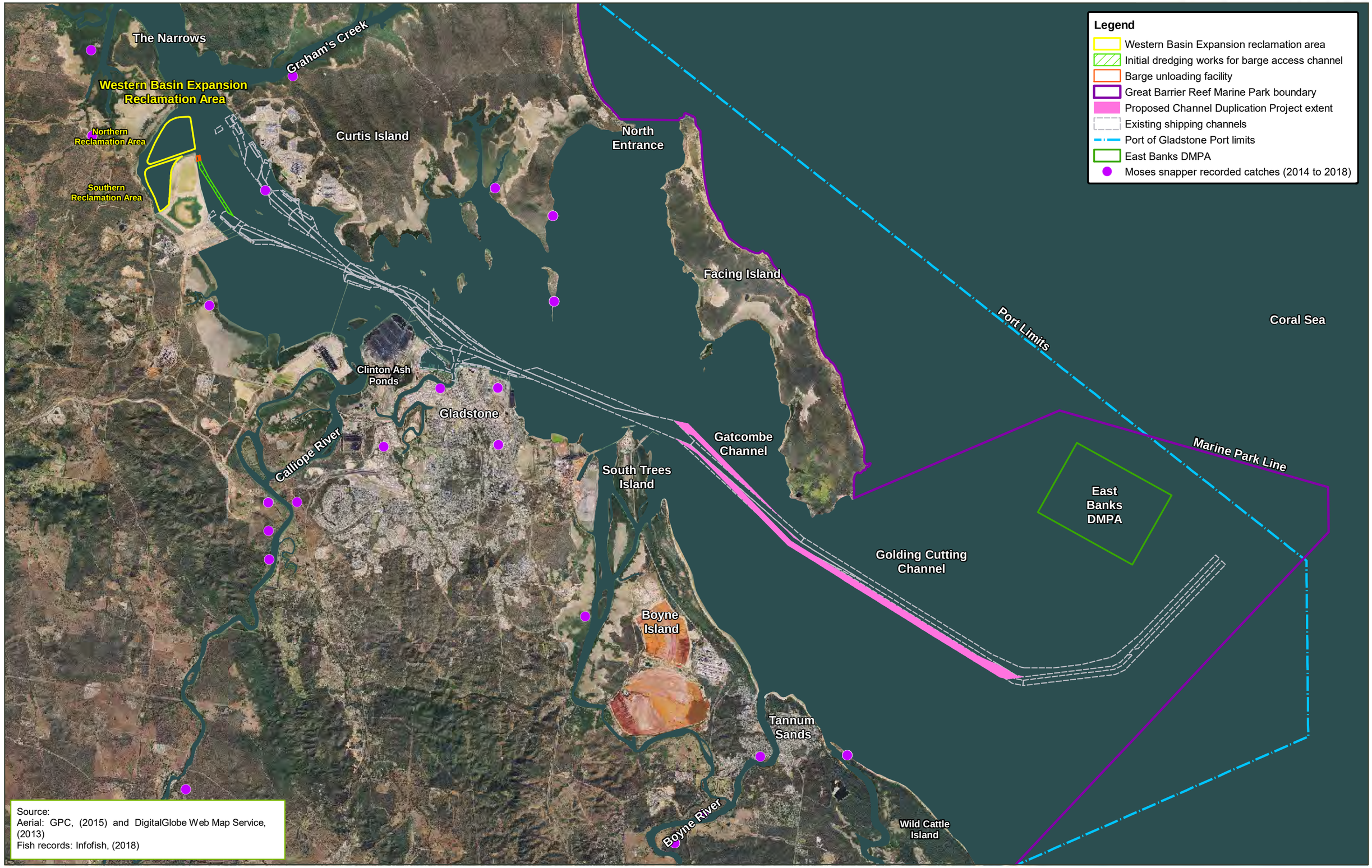
Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L21: Mangrove jack recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon



Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_335.mxd 24/01/2019 14:42

0 1,550 3,100 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L22: Moses snapper recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon

The Narrows
Graham's Creek
Western Basin Expansion Reclamation Area
Northern Reclamation Area
Southern Reclamation Area
Curtis Island
North Entrance
Facing Island
Port Limits
Coral Sea
Clinton Ash Ponds
Gladstone
Calliope River
Gatcombe Channel
South Trees Island
Marine Park Line
East Banks DMPA
Golding Cutting Channel
Boyne Island
Tannum Sands
Boyne River
Wild Cattle Island

Legend
- Western Basin Expansion reclamation area
- Initial dredging works for barge access channel
- Barge unloading facility
- Great Barrier Reef Marine Park boundary
- Proposed Channel Duplication Project extent
- Existing shipping channels
- Port of Gladstone Port limits
- East Banks DMPA
- Pikey bream recorded catches (2014 to 2018)

Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_336.mxd 24/01/2019 14:43

0 1,550 3,100 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L23: Pikey bream recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon

The Narrows
Graham's Creek
Western Basin Expansion Reclamation Area
Northern Reclamation Area
Southern Reclamation Area
Curtis Island
North Entrance
Facing Island
Port Limits
Coral Sea
Clinton Ash Ponds
Gladstone
Calliope River
Gatcombe Channel
South Trees Island
Marine Park Line
East Banks DMPA
Golding Cutting Channel
Boyne Island
Tannum Sands
Boyne River
Wild Cattle Island

Legend
- Western Basin Expansion reclamation area
- Initial dredging works for barge access channel
- Barge unloading facility
- Great Barrier Reef Marine Park boundary
- Proposed Channel Duplication Project extent
- Existing shipping channels
- Port of Gladstone Port limits
- East Banks DMPA
- Sand whiting recorded catches (2014 to 2018)

Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_337.mxd 24/01/2019 14:44

0 1,550 3,100 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L24: Sand whiting recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

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The Narrows
Graham's Creek
Western Basin Expansion Reclamation Area
Northern Reclamation Area
Southern Reclamation Area
Curtis Island
North Entrance
Facing Island
Port Limits
Coral Sea
Clinton Ash Ponds
Gladstone
Calliope River
Gatcombe Channel
South Trees Island
Marine Park Line
East Banks DMPA
Golding Cutting Channel
Boyne Island
Tannum Sands
Boyne River
Wild Cattle Island

Legend

- Western Basin Expansion reclamation area
- Initial dredging works for barge access channel
- Barge unloading facility
- Great Barrier Reef Marine Park boundary
- Proposed Channel Duplication Project extent
- Existing shipping channels
- Port of Gladstone Port limits
- East Banks DMPA
- Sea mullet recorded catches (2014 to 2018)

Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_338.mxd 24/01/2019 14:45

0 1,550 3,100 Metres

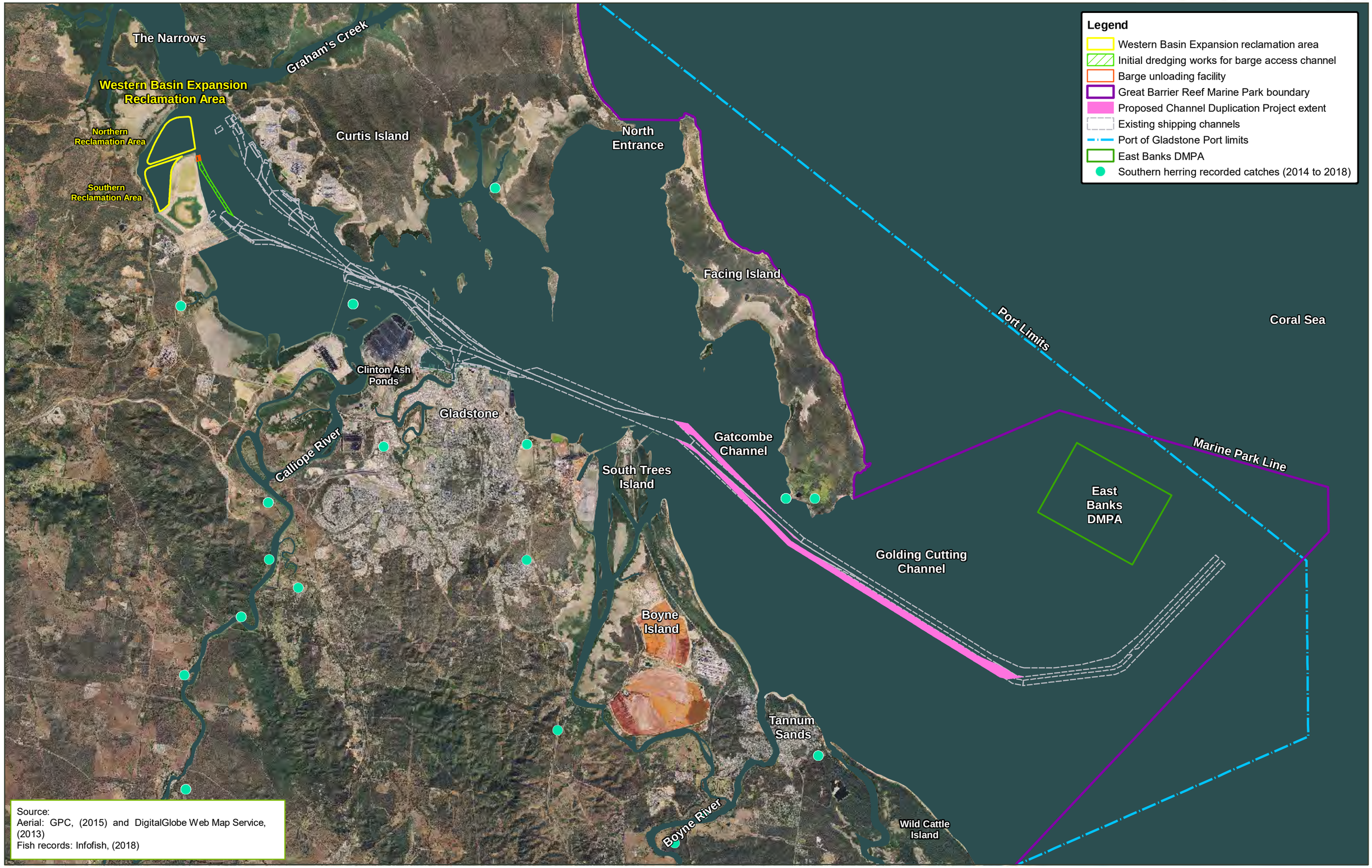
Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L25: Sea mullet recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon



Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
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0 1,550 3,100 Metres

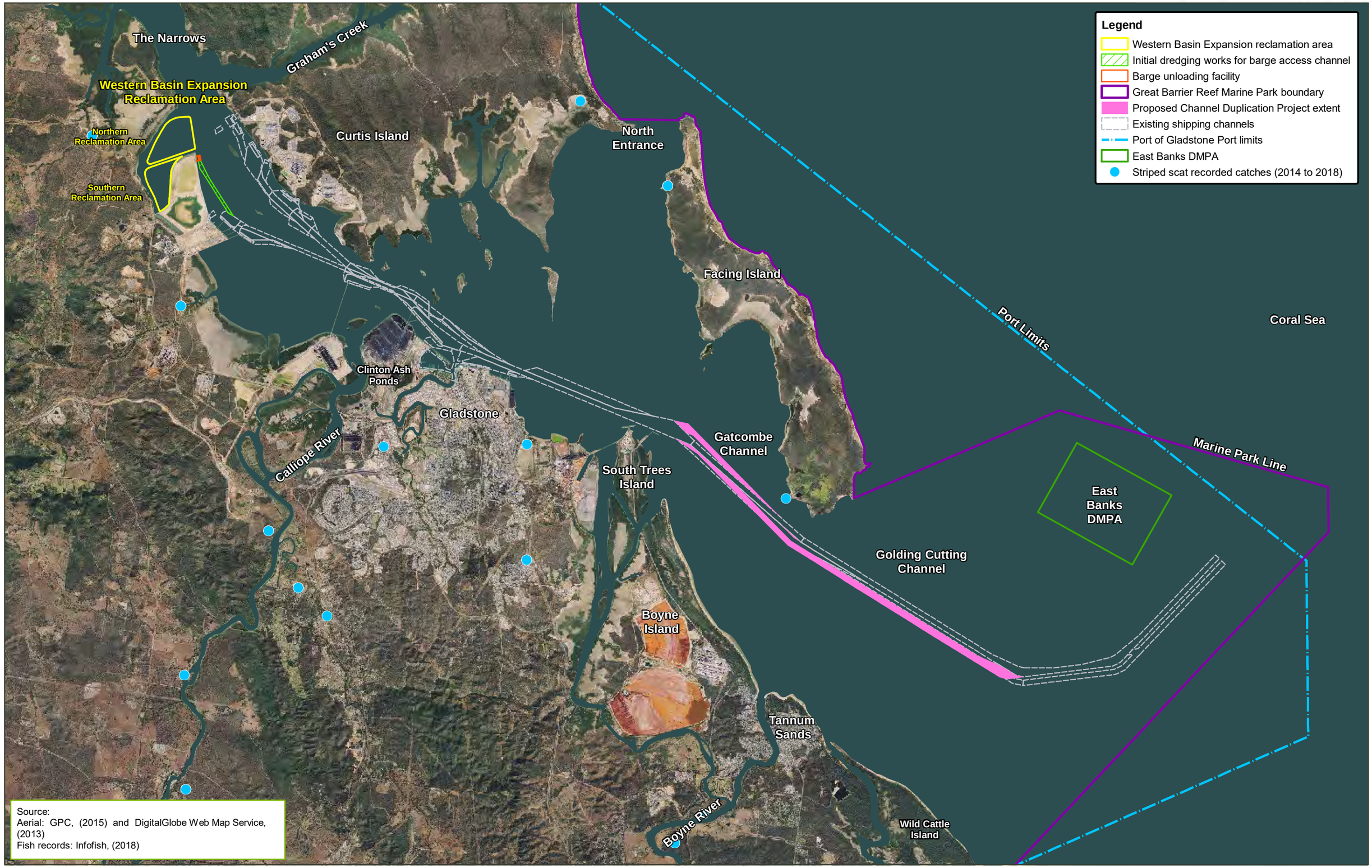
Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L26: Southern herring recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon



Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
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0 1,550 3,100 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L27: Striped scat recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

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The Narrows
Graham's Creek
Western Basin Expansion Reclamation Area
Northern Reclamation Area
Southern Reclamation Area
Curtis Island
North Entrance
Facing Island
Port Limits
Coral Sea
Clinton Ash Ponds
Gladstone
Calliope River
Gatcombe Channel
South Trees Island
Marine Park Line
East Banks DMPA
Golding Cutting Channel
Boyne Island
Tannum Sands
Boyne River
Wild Cattle Island

Legend
- Western Basin Expansion reclamation area
- Initial dredging works for barge access channel
- Barge unloading facility
- Great Barrier Reef Marine Park boundary
- Proposed Channel Duplication Project extent
- Existing shipping channels
- Port of Gladstone Port limits
- East Banks DMPA
- Tarwhine recorded catches (2014 to 2018)

Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_341.mxd 24/01/2019 14:47

0 1,550 3,100 Metres

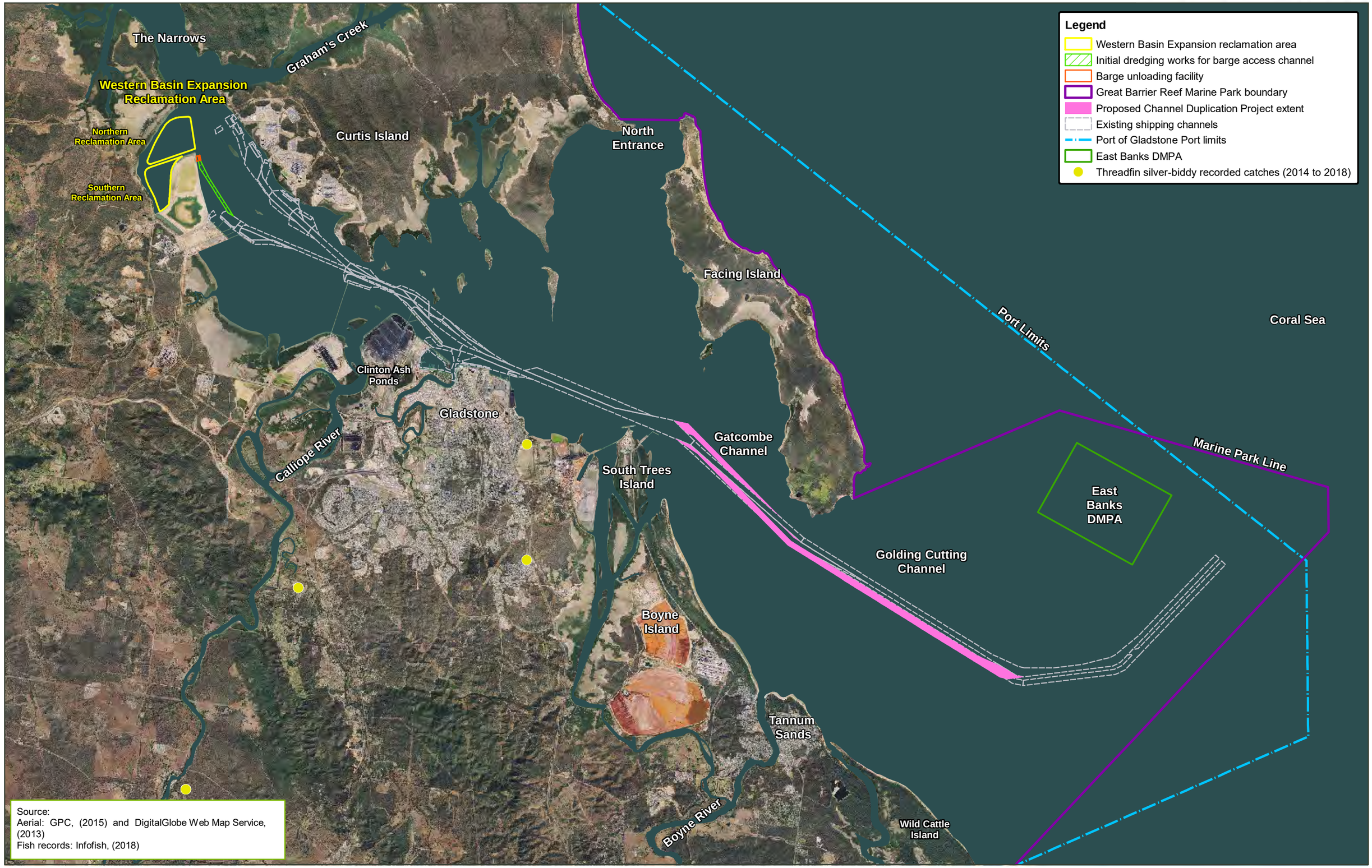
Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L28: Tarwhine recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon



Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_342.mxd 24/01/2019 14:48

0 1,550 3,100 Metres

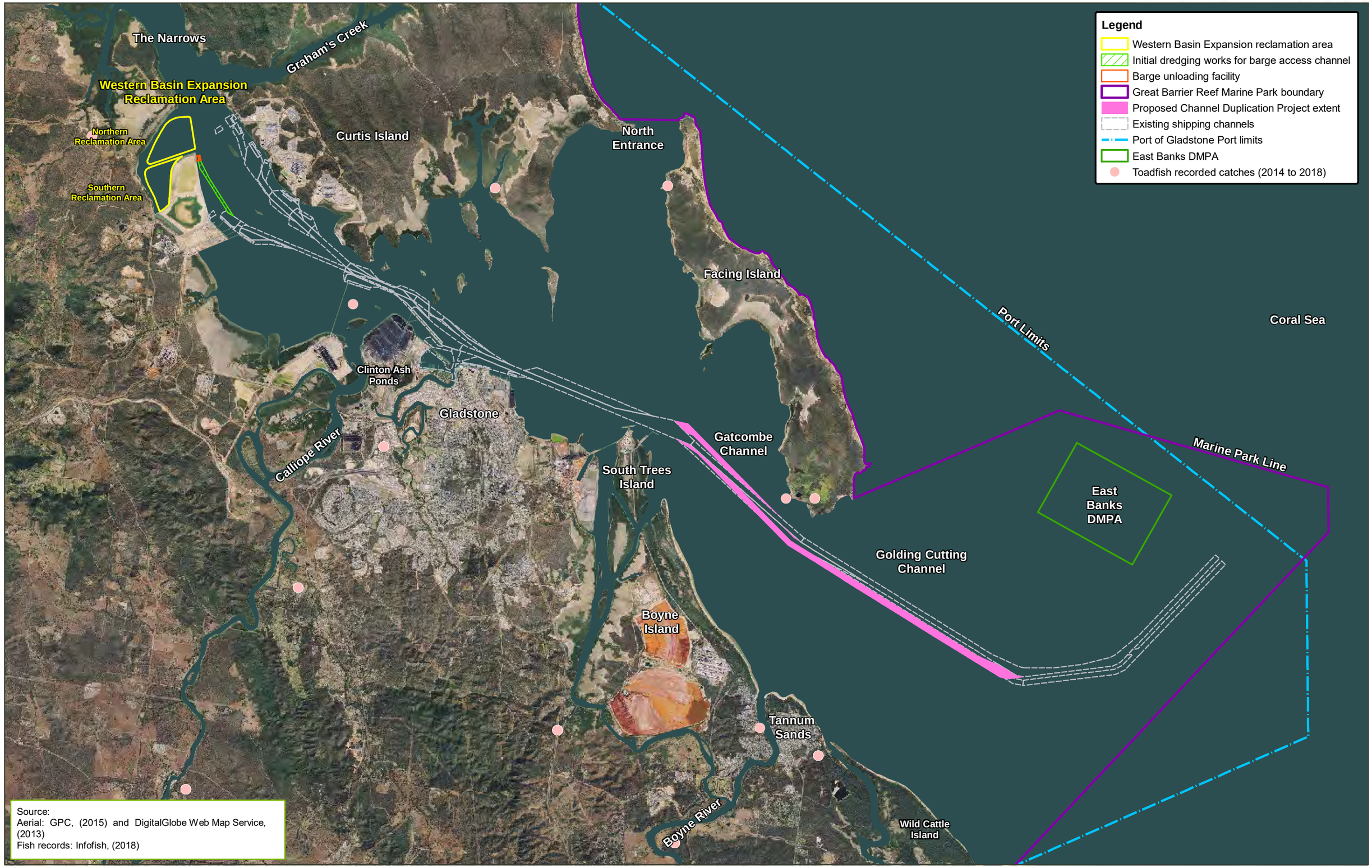
Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L29: Threadfin silver-biddy recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon



Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_343.mxd 24/01/2019 14:49

Metres
0 1,550 3,100

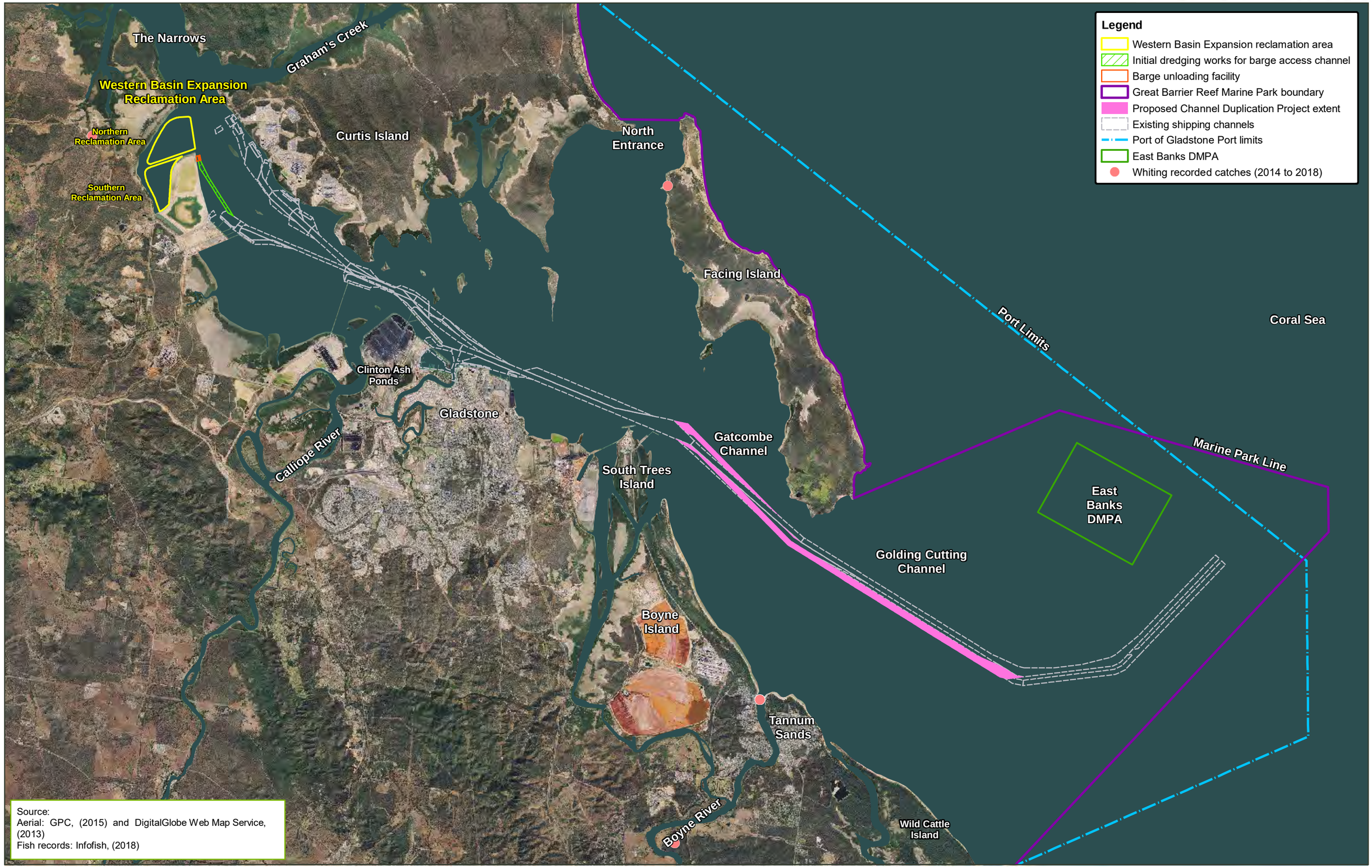
Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L30: Toadfish recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon



Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
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0 1,550 3,100 Metres

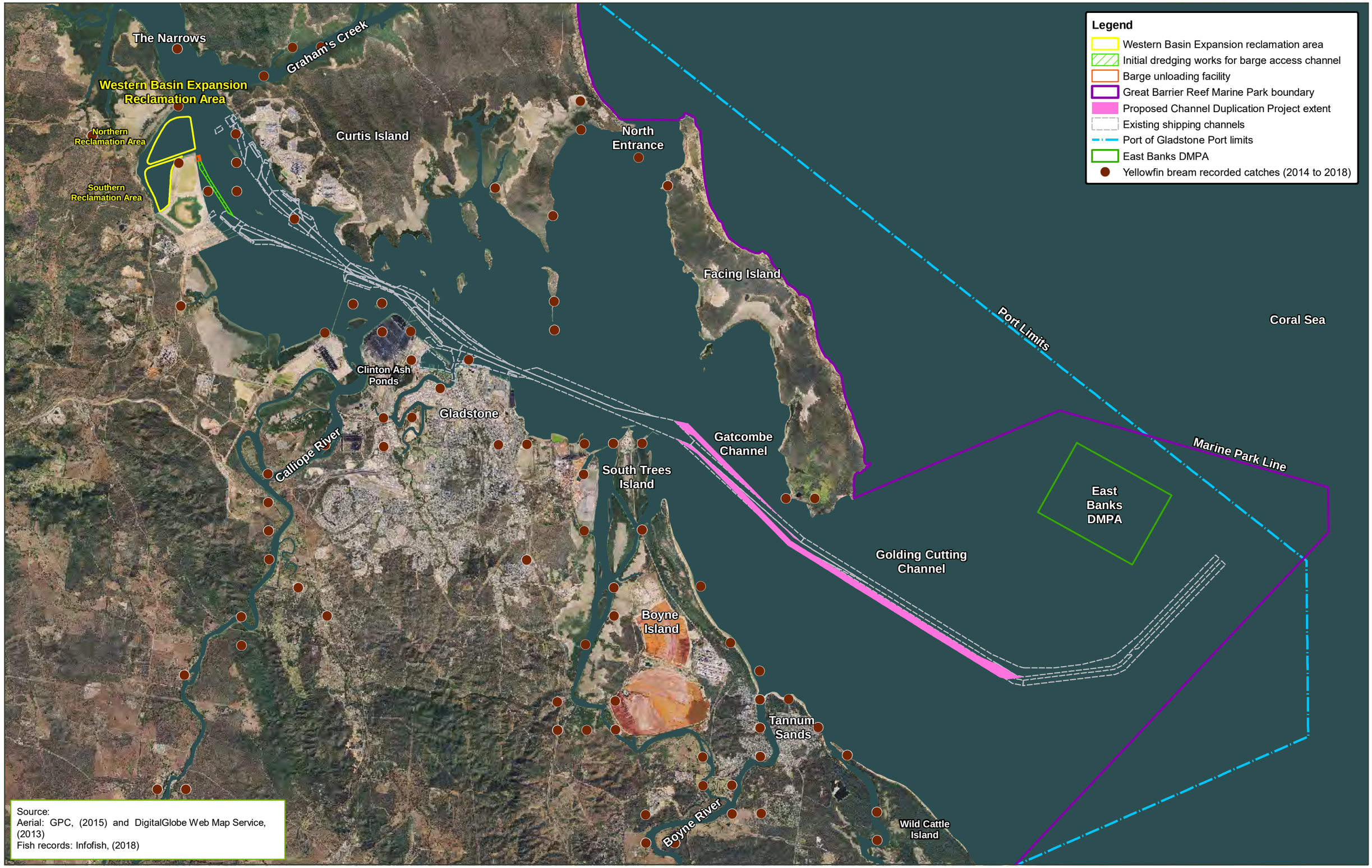
Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L31: Whiting recorded catches (2014 to 2018)

Gladstone Ports Corporation
Growth, Prosperity, Community.

aurecon



Source:
Aerial: GPC, (2015) and DigitalGlobe Web Map Service, (2013)
Fish records: Infofish, (2018)

Map by: RB
P:\GIS\Projects\237374_GPC_Channel_Duplication_EIS\237374_EIS_345.mxd 24/01/2019 14:51

0 1,550 3,100 Metres

Date: 24/01/2019 Version: 0 Job No: 237374
Coordinate system: GDA_1994_MGA_Zone_56

Gatcombe and Golding Cutting Channel Duplication Project

Figure L32: Yellowfin bream recorded catches (2014 to 2018)