Adelaide Outer Harbor Channel Widening Project: Post-dredging Seagrass Survey 2020

Reference: R.B22346.019.01.Seagrass_post-dredging_survey_final.docx
Date: July 2020
Confidential

Document Control Sheet

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	Title:	Adelaide Outer Harbor Channel Widening Project: Post-dredging Seagrass Survey 2020
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Synopsis:	**A report describing the cover and structure of seagrass meadows before and after a capital dredging campaign.**	

REVISION/CHECKING HISTORY

Revision Number	Date	Checked by		Issued by	
0	12th June 2020	CMJ	Conor Jones	LCM	L McKinnon
1	3rd July 2020				

DISTRIBUTION

Destination	Revision										
	0	1	2	3	4	5	6	7	8	9	10
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Executive Summary

Background

Flinders Ports conducted a capital dredging campaign in 2019 at the Port of Adelaide (Outer Harbor Channel Widening Project - OHCW). A seagrass monitoring plan was prepared in accordance with Dredge Licence Conditions a2.4.1(b)(ii) (U-995) and 3.3 (U-988) and a Vegetation Clearance Permit (NVC 2018/3020/010). These conditions/ permits required:

- An assessment of living seagrass extent and percent cover within the Survey Area using a Before-After-Control-Impact (BACI) design
- Quantification of changes to seagrass cover and health known or possibly attributable to the dredging program. The Clearance Permit allowed for the clearance of up to 162 ha of seagrass, based on numerical modelling of the predicted dredge plume. The dredge plume was subsequently remodelled, and the change in seagrass coverage revised to a total reduction of approximately 20ha.

A condition of the Dredge License was that one baseline (pre-dredging) and two post-dredging (2020 and 2022) seagrass assessments were to be undertaken to the satisfaction of the EPA. This report documents the findings of the first two surveys (2019 and 2020).

Approach

A Before-After-Control-Impact (BACI) experimental design was adopted as detailed in the OHCW Seagrass Monitoring Program approved prior to project commencement by EPA. Numerical modelling was used to predict dredge plume extent, and three impact zones were identified:

- Zone of High Impact (ZOHI) = water quality impacts resulting in predicted mortality of ecological receptors (i.e. seagrass meadows)
- Zone of Low to Moderate Impact (ZOLMI) = water quality impacts resulting in sub-lethal impacts to ecological receptors and/or short-term effects to ecological receptors
- Zone of Influence (ZOI)= extent of detectable plume[1], but no predicted ecological impact.

These predicted zones of impact were used to structure the BACI design, as follows.

- 'Impact' sites were located in the ZOI and were analysed using quantitative methods. A total of 10 impact sites were sampled
- 'Control' sites, which are seabed areas in surrounding areas outside the above three impact zones, were analysed using quantitative methods. A total of 25 control sites were sampled.

The purpose of the assessment was to identify whether there had been any impact from dredging outside of predicted areas (i.e. seagrass loss beyond the ZOHI and ZOLMI). 'Recovery' sites were also sampled within ZOHI (three sites) and ZOLMI (eight sites) to confirm predictions about this area being subject to loss of seagrass. These sites were analysed using semi-quantitative methods, and were not subject to detailed statistical analysis in the BACI framework.

[1] 'Detectable' plume in terms of detectable above background conditions by instrumentation deployed in the water column

Each impact and control site consisted of a 400 m by 400 m site box. Five 50 m replicate transects were randomly sampled at each site, and seagrass cover was quantified using point-count methods. Seagrass cover at the recovery sites was estimated using seagrass percentage cover standards.

Cover data from all sites were used to ground-truth satellite mapping which examined changes in seagrass extent and cover. Percent cover data were analysed using generalised linear mixed modelling appropriate to the error structure of the data. The BACI framework was used to test for interactions between control and impact treatments between survey events (i.e. pre-dredge (2019) and post-dredge (2020) surveys).

Findings

Assemblage Structure

Posidonia and *Amphibolis* were the dominant seagrass species in both survey events, found at various depths across the study area. *Halophila* was typically sparse in cover and recorded in up to 16 m water depth. *Heterozostera* was most abundant in shallow (intertidal) areas to the north of the navigational channel.

There were differences in seagrass assemblages between ‘control’ and ‘impact’ sites, reflecting differences in environmental conditions in the Survey Area. Seagrass meadows in the ‘impact’ zone were dominated by *Halophila* species, which are ephemeral, opportunistic species that are intolerant of low light conditions but can recolonise rapidly following disturbance. By contrast, seagrass meadows in the surrounding ‘control’ area were numerically dominated by *Posidonia* and *Amphibolis,* both of which are more tolerant of periodic low light conditions than *Halophila*, but have lower reproduction and recovery rates. The figure below shows the overall change in percent cover between 2019 and 2020 at each survey site.

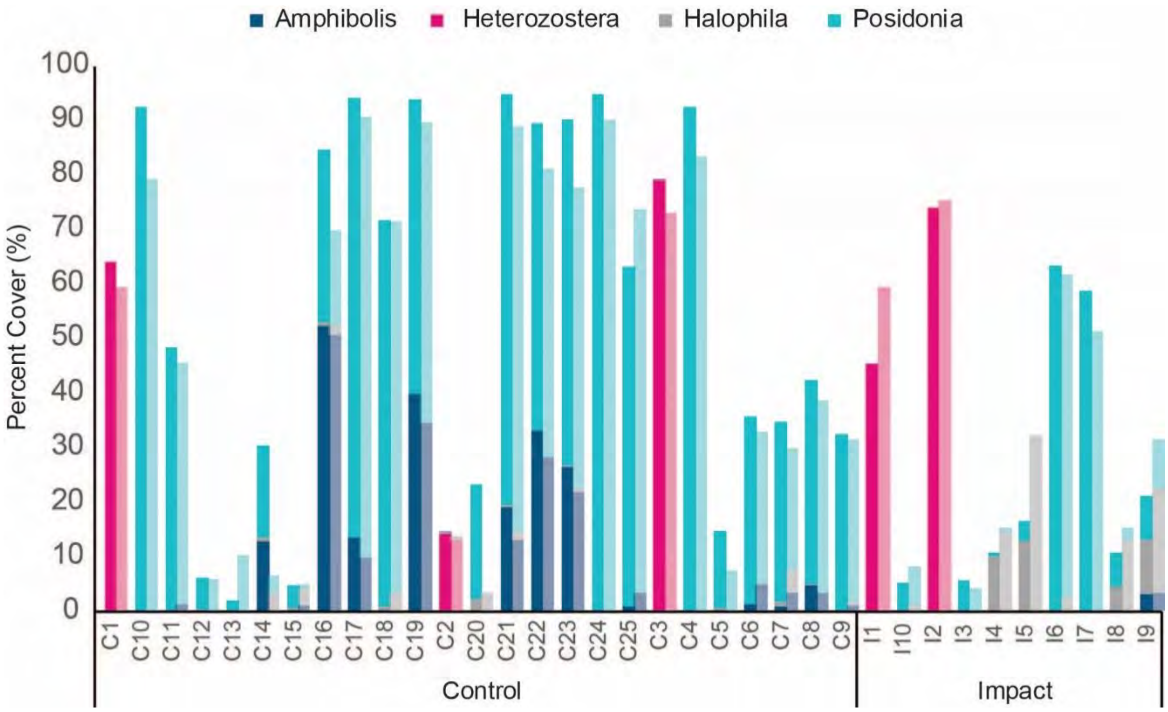

Figure 1 Comparison of seagrass species composition (site-averaged percent cover) at the control and impact sites. 2019 data are shown in full tones, 2020 data are shown in half-tones

Seagrass Percentage Cover

Within the recovery sites, seagrass coverage showed no significant change, although a slight reduction was predicted by modelling. Coverage within the areas of ZOIMI and ZOHI was already very low however.

Within the BACI framework, a potential impact would be indicated if the direction of change in seagrass cover before and after dredging differed between treatments (i.e. significant Time x Control/Impact interaction). For all seagrass taxa and total seagrass cover, no significant difference (p > 0.1) in the Time x Control/Impact interaction was detected. This indicates that there was no detectable difference in seagrass cover before and after dredging activity in 2019.

Complex temporal and spatial patterns in seagrass cover were observed among species and sites. *Halophila* (increase in time) and total seagrass (decease) cover were the only indicators that displayed a significant overall change over time (p < 0.05). And while no statistically significant interaction was detected, analysis of temporal trends within treatments suggested there was a significant (p <0.05) reduction in seagrass cover within the 'control' sites between 2019 (average 56%) and 2020 (average 50%), but no statistically significant change (p > 0.05) in total cover over time within the 'impact' sites. There was variation in seagrass cover within several sites which resulted in inconsistent trends in space and time at the site scale.

The relatively benign effect of the dredging activities on surrounding seagrass communities is most likely related to dredging methodology, the timing of the dredge campaign in winter months outside of the seagrass growth period and dredge management in accordance with turbidity limits. Whilst limits were exceeded on a number of occasions, these were generally of a short duration and localised to within the predicted dredge plume extent.

Seagrass Mapping

Remote sensing was undertaken to map seagrass cover in 2019 and 2020. This mapping provides a general indication of spatial patterns in seagrass communities patterns in the Survey Area. However, there are limitations with this mapping which preclude precise estimates of changes in cover, most notably inherent constraints with seagrass detection in deeper waters, inconsistency in turbidity signals between captures, and low thresholds of detection for substrates with low seagrass cover. Therefore, seagrass mapping should be interpreted with caution and not used to quantify seagrass extent.

Patterns in moderate to dense seagrass in shallow areas were generally consistent between 2019 and 2020. Within the ZOI, sparse seagrass extent was generally consistent over time, the main exception being potential loss in an area immediately north of the dredged channel within an area of already sparse seagrass coverage. Seagrass extent in the western section of the study area (beyond the ZOI) was greater in 2020 than 2019 in the deeper area southwest of the shipping channel, which was inconsistent with field data, and likely an anomaly relating to the quality of the 2019 satellite imagery.

Conclusion

In the DA submission and the Native Vegetation Clearance application, an assumption was made that all seagrass would be lost immediately post-dredging within the zone of high and low to moderate impact, as a conservative measure. The Native Vegetation Clearance permit was awarded on this basis assuming 162 ha of seagrass loss.

Seagrass percentage cover and seagrass mapping comparing the pre-dredging seagrass survey and post-dredging seagrass survey indicate little to no loss of seagrass due to the 2019 Outer Harbor Channel Widening dredging project.

2022 Survey

The Seagrass Monitoring Program consists of three field surveys throughout the Project lifecycle. This current survey is the first post-dredging survey. The 2019 survey was the baseline (pre-dredging survey), conducted in April 2019 and a subsequent survey will be an additional post-dredging survey, scheduled for April 2022, approximately two years post dredging activities ceasing. The surveys have been scheduled at the same time each year to account for seasonal fluctuation in seagrass extent and growth.

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1 Introduction

1.1 Background

1.1.1 Project

The Port of Adelaide is the primary port in South Australia, located at Outer Harbor (approximately 14km west of the Adelaide CBD) in South Australia (Figure 1-1). The port is operated by Flinders Ports Pty Ltd (Flinders Ports) and handles a diverse array of inbound and outbound cargoes, contributing significantly to the State's economic activity.

Flinders Ports conducted the Outer Harbour Channel Widening Project (OHCW) to accommodate Post Panamax class vessels (maximum width of 49m) without operational restrictions. Capital dredging was undertaken to widen: (i) the existing channel by 40 m, to a total width of 170 m; and (ii) the swing basin from 505 m to 560 m. The dredging footprint of the channel and swing basin are presented as a red line in Figure 1-1. Dredged material was transported to a designated dredge material placement area (DMPA), located approximately 30 km offshore in the Gulf of St Vincent (yellow box in Figure 1-2).

The dredge campaign occurred from 7th June and 18th September 2019, inclusive. Dredging was undertaken using a combination of a Trailing Suction Hopper Dredge (TSHD) and Backhoe Dredge (BHD). A sweeper vessel was also used for bed-levelling throughout the project and at its completion. The BHD Magnor, dredged stiffer material, approximately 30% of the overall volume. Generally, BHD create smaller sediment plumes than other forms of dredging. The TSHD Gateway dredged softer material, approximately 70% of the overall volume. A total in-situ nett volume of 1,487,208m^3 of material was removed from the channel (and swing basin). Overall, the total gross volume removed was 2,223,384 m^3, which included over-dredging to ensure that declared depths were reached.

1.1.2 Monitoring Program Requirements

A Development Application (DA) Report was submitted in July 2017 in accordance with the *Development Act 1993.* Flinders Ports received project approval on 28th May 2018 DA 010/V048/17, and a dredge licence was issued by the Environmental Protection Authority (EPA) in March 2019 (Licence No. 50556). In addition, a Vegetation Clearance Permit was issued by the Native Vegetation Council under the Native Vegetation Regulations for the clearance of up to 162ha of seagrass.

The Seagrass Monitoring Program (SMP) was prepared in accordance with the following:

- Dredge License Condition 3.3 (U – 988) – which requires an assessment of live seagrass extent and percent cover within the Survey Area using a Before-After-Control-Impact (BACI) experimental design.
- Vegetation Clearance Permit which requires quantification of the amount of seagrass clearance (i.e. reduced seagrass density or health) attributable to the dredging program[2], in order to confirm the SEB payment amount. This Clearance Permit allows for the clearance of up to 162 ha of seagrass.

[2] i.e. direct loss and indirect losses due to sedimentation/turbidity resulting from dredging

The SMP was designed to meet both condition/permit requirements.

As a condition of the Dredge License, one baseline (pre-dredging) and two post-dredging (2020 and 2022) seagrass assessments must be undertaken to the satisfaction of the EPA and NVC. This report documents the findings of the first post-dredging (2020) seagrass survey. The survey methodology (BMT 2019) was developed in conjunction and approved by the EPA, and forms part of the Project Environmental Monitoring Program (EMP).

LEGEND

- New Location of existing Beacon
- Existing Beacon
- Dredge Material Placement Area
- Proposed Channel Widening Footprint
- Existing Channel Footprint

*Aerial Image © Google Earth

Title:

Project Location

Figure: 1-1

Rev: A

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

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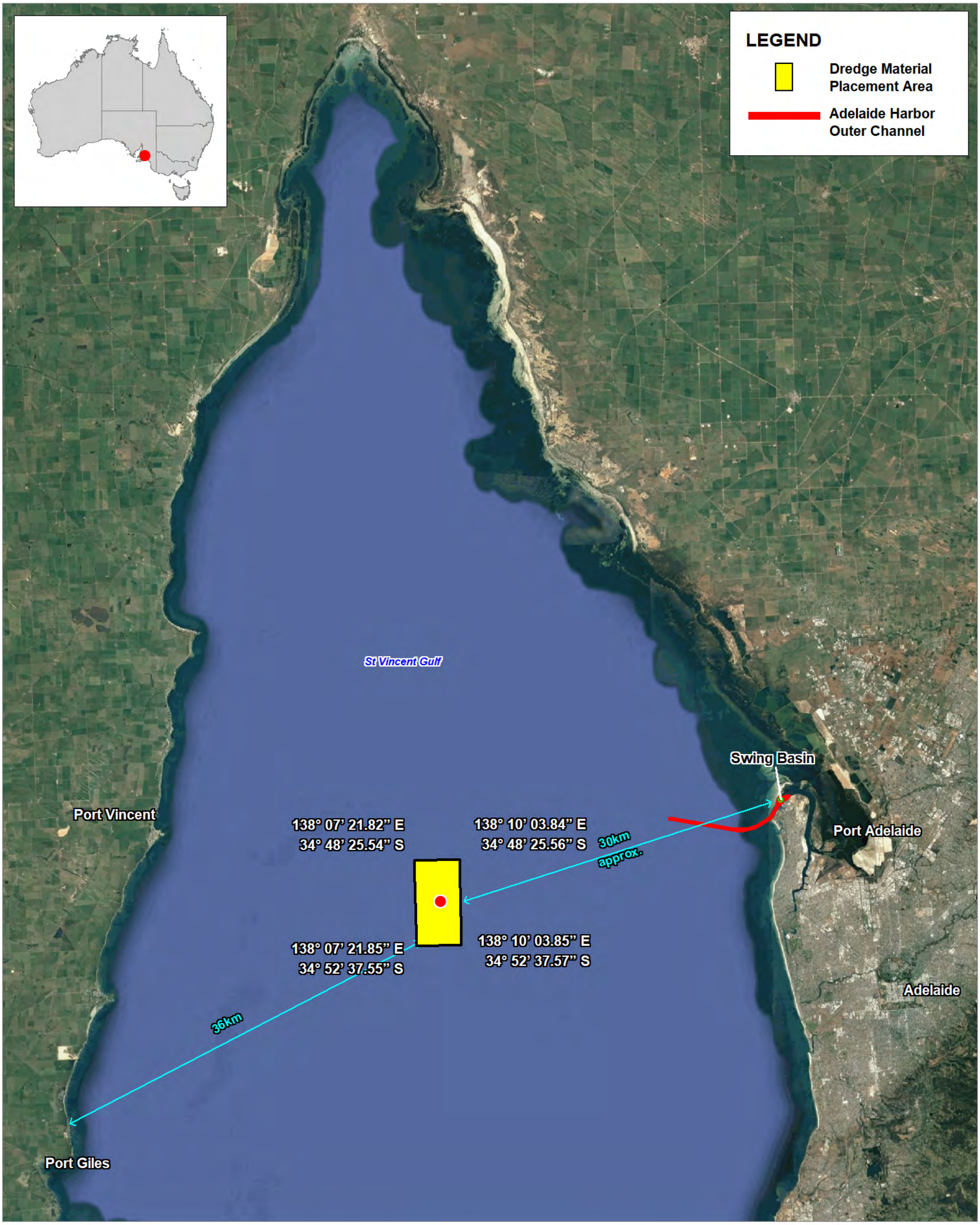

Title:
Dredge Material Placement Area

Figure: 1-2

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1.2 Objectives

The objectives of this first post-dredging survey and their linkages to permit conditions are set out in Table 1-1.

Table 1-1 SMP Objectives

Objective	Condition	Post-Dredging Survey Episode Relevance
(1) Quantify seagrass percentage cover and extent at test and control sites, before and after dredging	Dredge License Condition	This Survey and two years after dredging
(2) Describe the location and full extent of the area of impact associated with dredging operation	Vegetation Clearance Permit	This Survey
(3) Describe changes in condition of seagrass (density or condition) within the area of impact	Vegetation Clearance Permit	This Survey
(4) Assess the extent to which seagrass has recovered two years after the completion of dredging	Vegetation Clearance Permit	Two years after dredging

2 Modelled Impacts to Seagrass

To support the Development Assessment application (approved in 2017), a habitat risk assessment methodology was applied to the project, using the outputs from the predictive dredge plume numerical model. Impact predictions were presented as 'zones of impact', which is now recognised as 'best practice' in dredging environmental assessments and is commonly used in environmental assessments of dredging projects in Australia, building on the methodologies set out in the dredging environmental assessment guidelines produced by the Western Australia Environmental Protection Agency (WA EPA) (2016).

The zones adopted for the habitat risk assessment, include the following:

- Zone of High Impact (ZOHI) = water quality impacts resulting in predicted mortality of ecological receptors with recovery time greater than 24 months
- Zone of Low to Moderate Impact (ZOLMI) = water quality impacts resulting in predicted sub-lethal impacts to ecological receptors and/or mortality with recovery between 6 months (lower end of range) to 24 months (upper end of range)
- Zone of Influence (ZOI)= extent of detectable plume[3], but no predicted ecological impacts.

It is important to note that the recovery times outlined for the various zones should be considered as indicative only, noting that such timeframes are dependent on a range of factors that are extremely complex and difficult to accurately predict. The zones and their 'recovery time frames' represent a means for comparing the likelihood that significant, detectable impact to sensitive receptors could occur, and assume that recovery timeframes are dependent on the magnitude of impact.

A concept design of the zones of impact (sourced from WA EPA 2016) is shown in Figure 2-1.

In the DA report, an assumption was made that all seagrass would be lost immediately post-dredging within the zone of high and low to moderate impact, as a conservative measure.

The tolerance of the seagrass species within the Survey Area varies considerably as does the likelihood of potential impact from dredge plumes. Light is one of the key determinants of species loss and recovery. Colonising /ephemeral species (such as *Halophila* and *Heterozostera* spp.) are characterised by short turnover times (<months) and low physiological resistance to disturbances. However, ephemeral species can recover rapidly, in part due to high investment in sexual reproduction and the resultant ability to build up a seed bank (Erftemeijer & Robin Lewis 2006, Kilminster *et al.* 2015). Conversely, persistent/perennial species (such as *Amphibolis* /*Posidonia* spp.) have long turn-over (months–years) of growth units (i.e. rhizome, shoot and root), clonal vegetative growth and high physiological resistance to disturbance, but are slow to recover from disturbances (Kilminster *et al*. 2015).

[3] 'Detectable' plume in terms of detectable above background conditions by instrumentation deployed in the water column

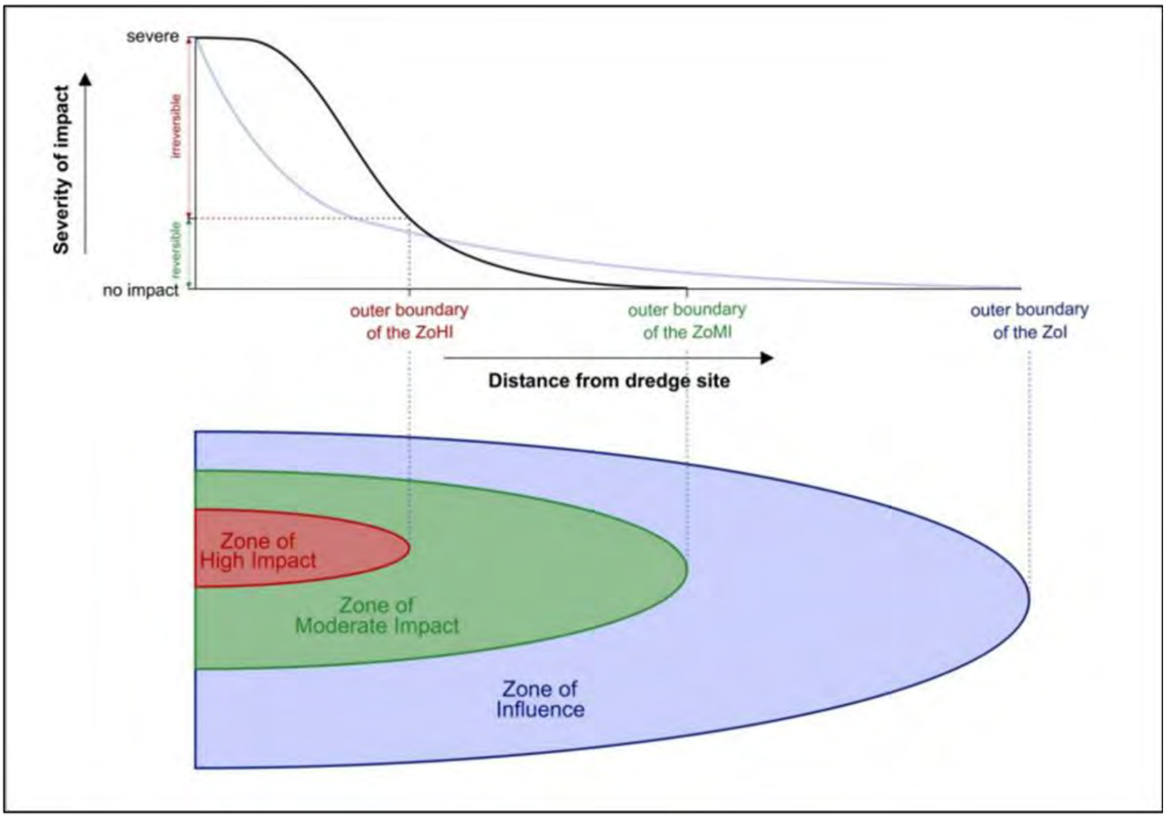

Figure 2-1 Concept design of impact zones (WA EPA 2016)

Gils *et al.* (2017) undertook a review of possible minimum light requirements for seagrass species present in South Australia on behalf of SA Water. It is suggested that *Heterozostera* spp. have an optimal light availability threshold of >20% Light (% of surface irradiance), over a 30-day rolling average period. This is based on a literature review however and has not been field tested. Furthermore, the main area of predicted loss is in an intertidal area, which has complex light requirements in comparison to subtidal areas.

Table 2-1 shows the predicted seagrass losses as a result of dredging activity; these losses were approved by the Native Vegetation Council. Subsequent to submission of the original dredging modelling, the dredging methodology was amended to avoid the need for 'double-handling' of material, which further reduced the predicted zones of impact.

The total area of impacted seagrass (zone of high impact and zone of low to medium impact) was revised to approximately 20ha; this was reduced in comparison to the disturbance area of 158ha originally estimated in the 2017 development application and approved for clearance under the Native *Vegetation Act 1991* and *Planning Act 2016*.

Table 2-1 Estimate of seagrass impacts from direct and indirect dredge plumes approved in the Vegetation Clearance Permit

Classification category	Coverage	Direct Impact	Total Area (ha) within the High to Medium Impact Area (Winter)
Moderate to dense seagrass including Amphibolis and/or Posideonia.	Moderate to dense (35-100%)	-	*0.02*
Sparse seagrass including Halophila australis and/or very sparse Posidonia.	Sparse (1-35%)	*4*	*0.2*
Seagrass dominated by *Heterozostera*	Moderate to dense (35-100%)	-	158
Total		**4**	**158**

3 Conditions During the Dredge Campaign

3.1 Turbidity

The End of Works Report (Boskalis, 2019) documents that:

- The 15-day rolling median exceeded the chronic turbidity HOLD criteria several times at both monitoring sites (Figure 3-1)
- The 6-day rolling median did not exceed the acute turbidity HOLD criteria, as shown in Figure 3-2.

Strong and prolonged weather events during dredging activity contributed to the exceedances listed above, which likely re-mobilised fine sediments produced by dredging activity. This increased localised ambient turbidity through the Survey Area, even after the cessation of dredging.

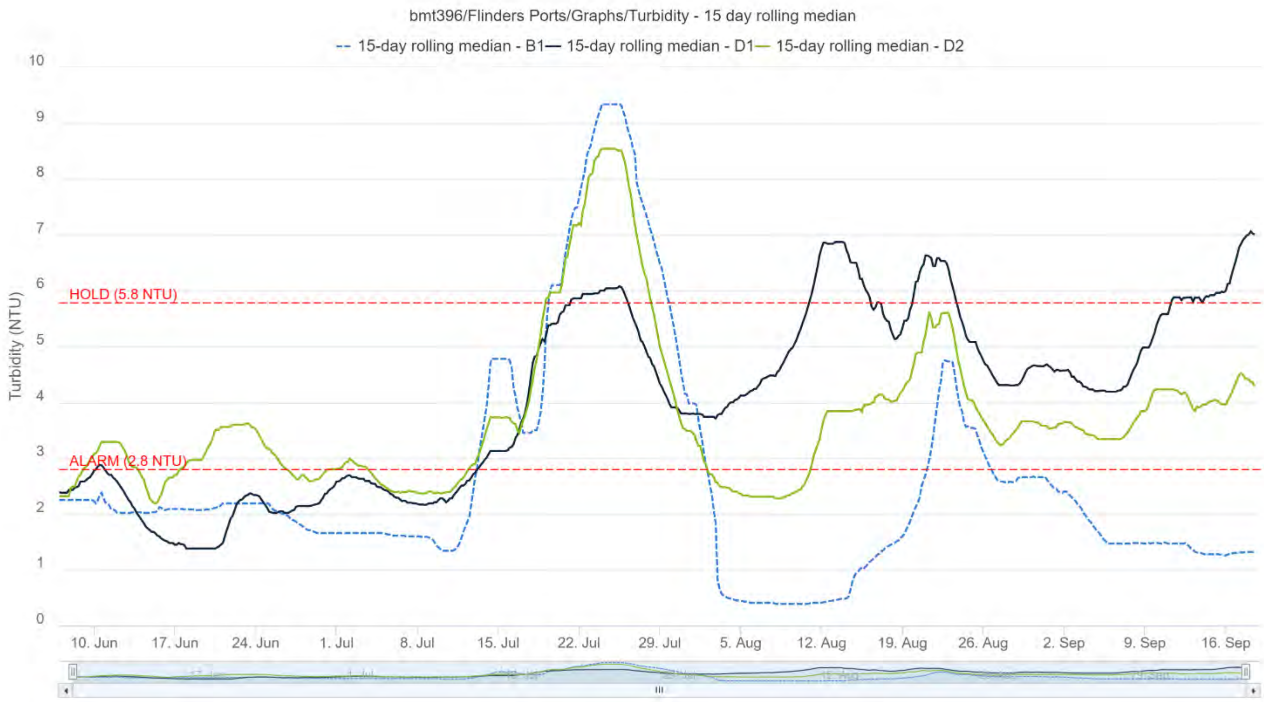

Figure 3-1 15-day rolling median turbidity (NTU)

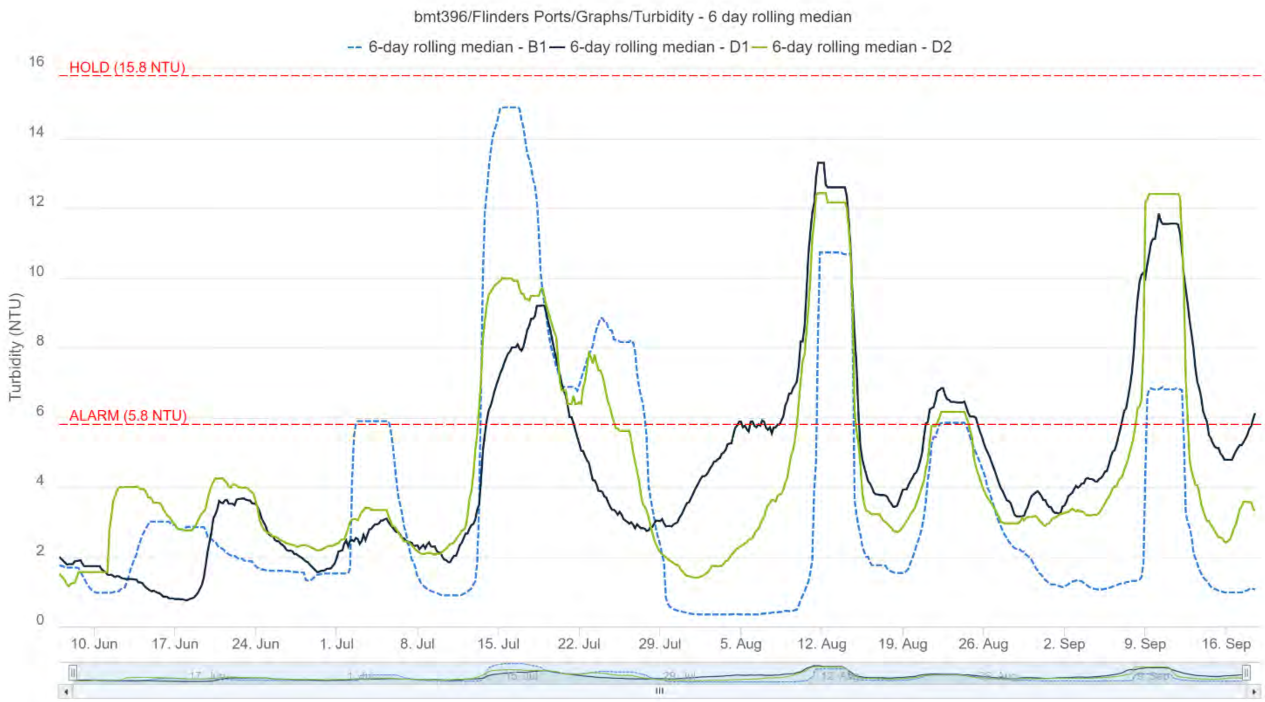

Figure 3-2 Six day rolling median turbidity (NTU)

3.2 PAR

As one of the primary drivers of seagrass condition and resilience to disturbance, understanding the light available, and any loss of light for a prolonged period of time is important. Photosynthetically Available Radiation (PAR) is a way of measuring light available to seagrass. PAR is naturally lower in winter months when daylight hours are reduced. For seagrass loss to occur, light must be limited for a significant period, although the exact duration after which seagrass loss occurs is not well studied in South Australia. South Australian seagrasses would be naturally adapted to low light levels during the winter months.

Although a PAR target was not set, BMT recorded PAR data at the three monitoring stations. Figure 3-3 shows that lower levels of benthic PAR often coincided with low surface irradiance (i.e. cloudy days).

Figure 3-4 shows the percentage of surface irradiance available at the seafloor. There were several occasions where light levels were significantly reduced at the compliance sites either due to poor weather conditions or turbidity plumes. Whilst available PAR was reduced, light conditions are generally lower in winter months anyway when seagrass is not growing.

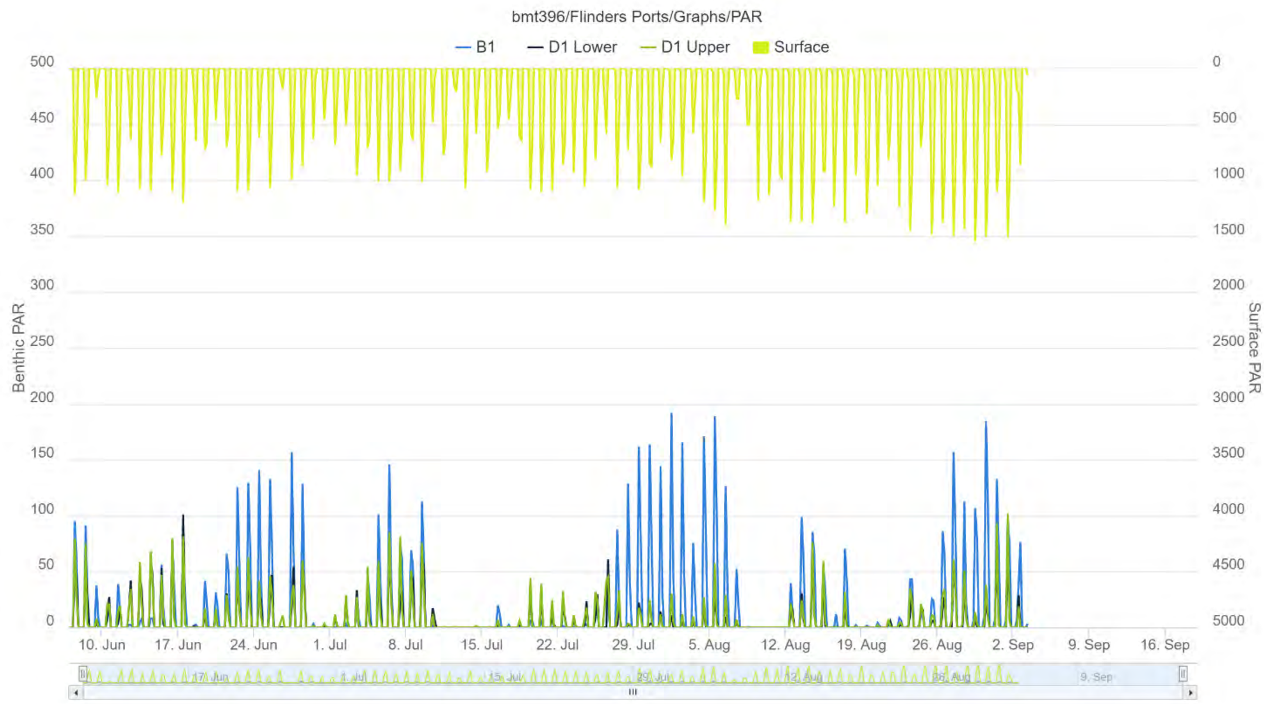

Figure 3-3 Total PAR at surface compared to benthic loggers

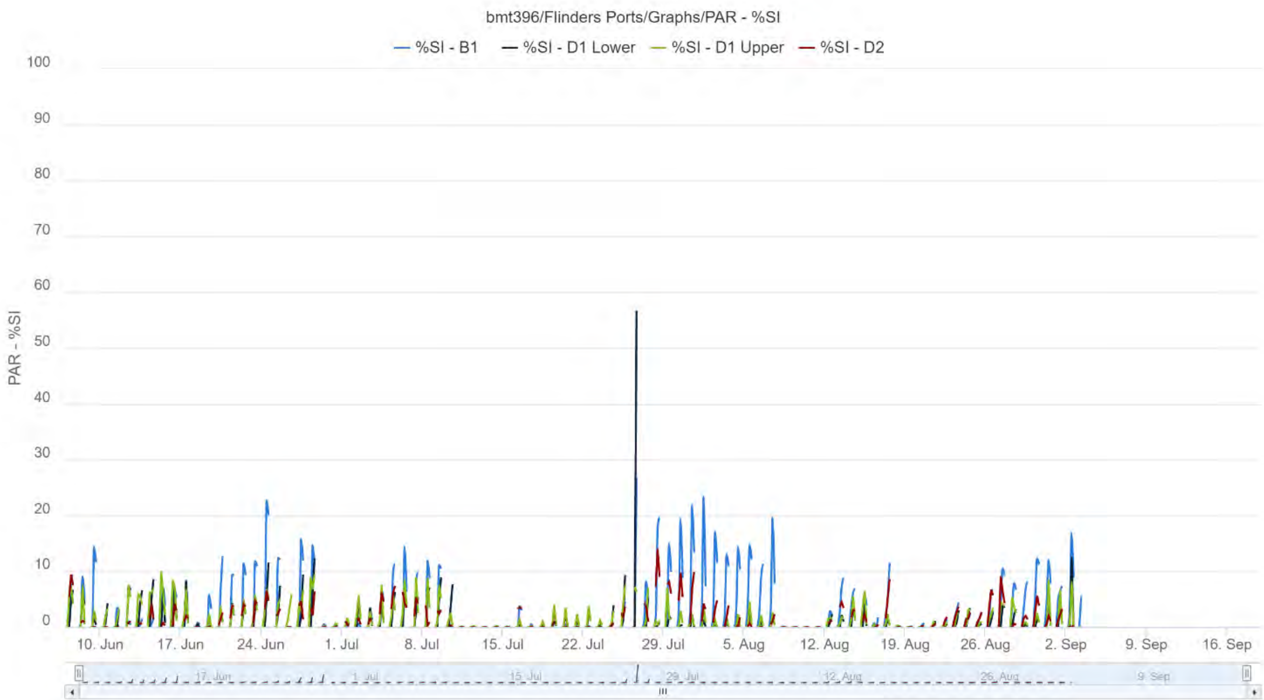

Figure 3-4 % Surface Irradiance Benthic PAR

3.2.1 Dredge Plume Extent

3.2.1.1 Modelled Plumes

Turbidity percentiles predicted for the project are shown in Figure 3-5. The acute exceedance level 95th percentile (worst case) is shown on the left and the chronic 50th percentile exceedance level 50th percentile on the right.

Predicted sediment deposition rate percentiles for the dredging campaign are shown in Figure 3-6. The acute exceedance level (95th percentile) is shown on the left and the chronic exceedance level (50th percentile) on the right. The final predicted distribution of net sediment deposition at the end of the dredging campaign is shown in Figure 3-7.

3.2.1.2 Real-time Plumes

Throughout the dredge campaign, satellite images from the MODIS satellite were collected (roughly every 2 days) showing the raw turbidity in the broader area. Typically, the images show that turbidity was widespread the length of the coastline during high wind events, with a rapid reduction within 1-2 days of the weather event, after which it was confined to the area immediately surrounding the dredge footprint (usually concentrated on either side of the channel just beyond the breakwater, rather than the outer sections), and was generally in accordance with the modelled plume extent.

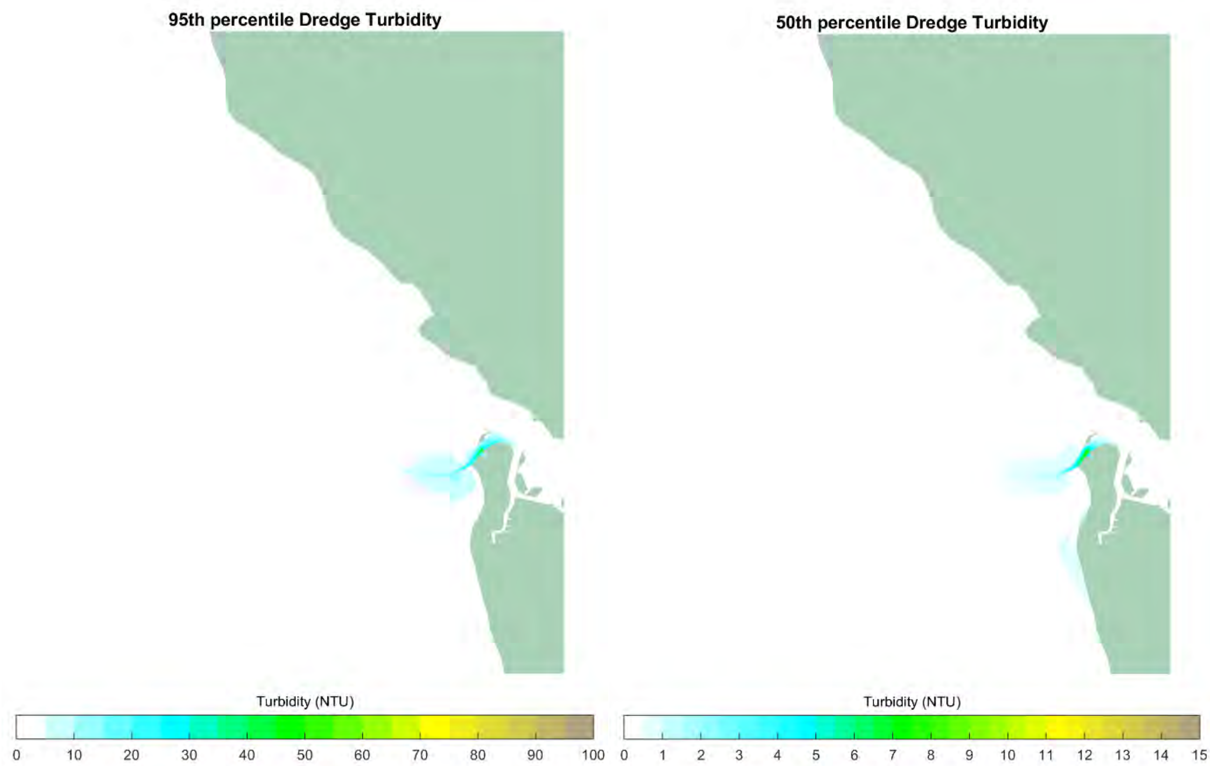

Figure 3-5 Turbidity Percentile Contours. Acute 95th Percentile (Left); Chronic 50th Percentile (right).

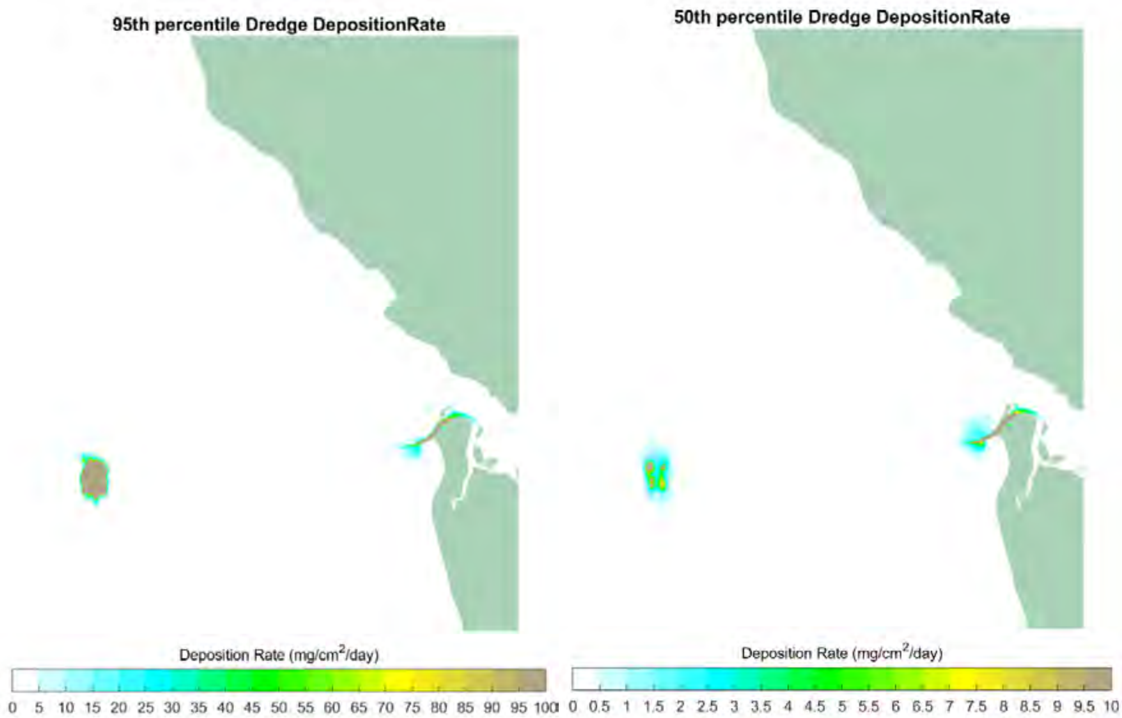

Figure 3-6 Sediment Deposition Rate Percentile Contours. Acute 95th Percentile (left); Chronic 50th Percentile (right).

Figure 3-7 Final Net Sediment Deposition Contours

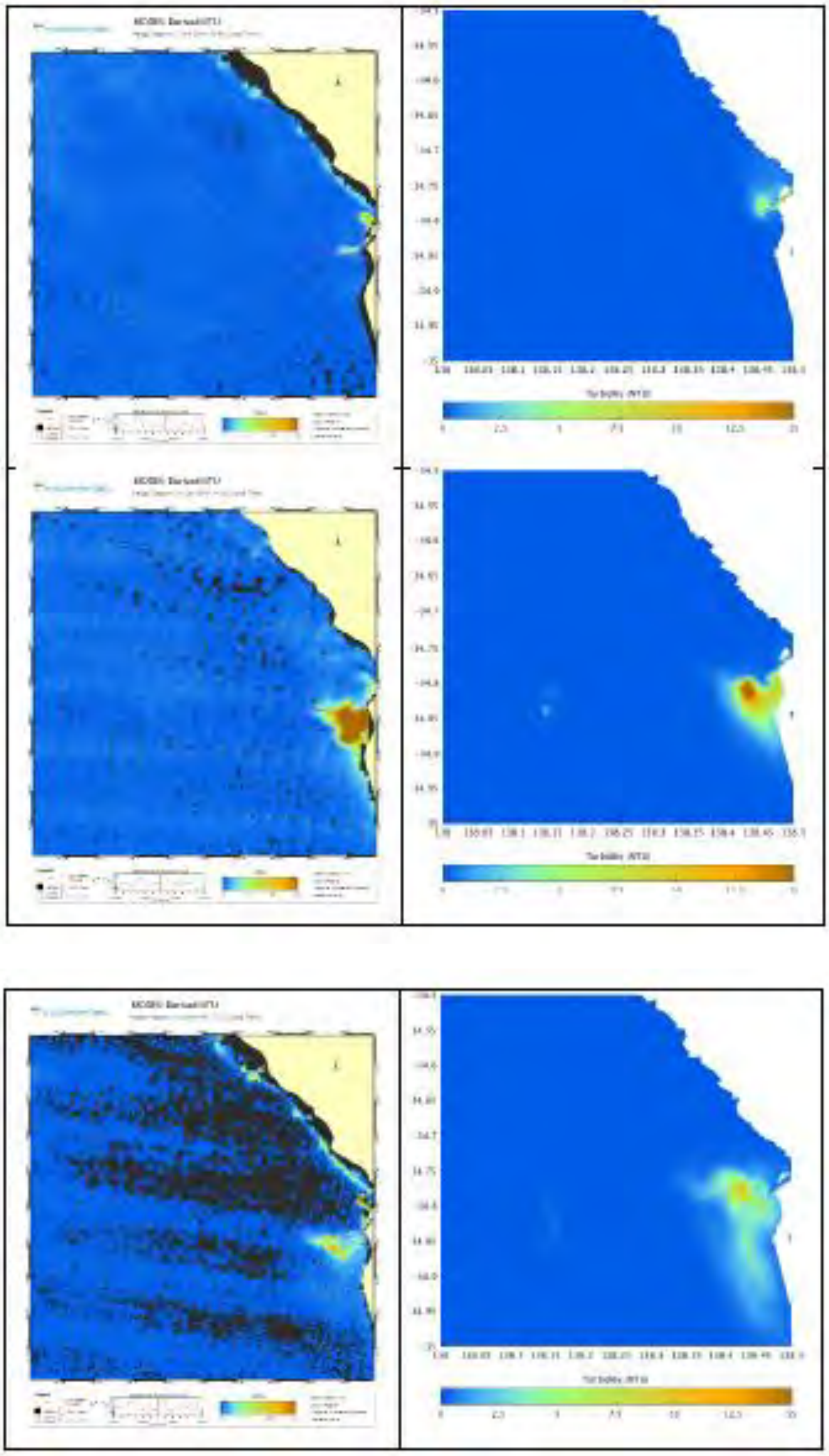

Figure 3-8 Comparison of satellite image derived turbidity (including background) and model turbidity predictions (excluding background)

4 Survey Methodology

The survey methodology outlined below is consistent with that proposed in the approved SMP. The 2019 survey was conducted by the BMT marine science team. The 2020 survey was undertaken by Social and Ecological Assessment Pty Ltd (SEA)[4], in accordance with the prescribed methodology, and using BMT equipment and on-call support.

4.1 Survey Timing

The pre-dredging survey was conducted over two survey campaigns due to weather constraints. The first survey was conducted on the 14-16th of April 2019. The weather was fine, with light to moderate winds of variable direction which provided workable conditions. The second pre-dredging survey was conducted on the 28-29th of April 2019. The weather was fine with calm to light winds which provided ideal survey conditions.

The post-dredging seagrass survey was due to be conducted in April 2020; however, due to COVID-19 pandemic restrictions and unworkable weather (elevated wind and swell), the survey was conducted on the 4-6th of May 2020. The weather was overcast, with light to moderate north to north-west winds which provided workable conditions.

4.2 Survey Sites

4.2.1 Dredging Licence Compliance Sites (Control and Impact Sites)

To comply with the Dredge Licence conditions, and the Before-After, Control-Impact (BACI) approach required, the survey locations of the following (refer to Figure 4-2):

- **Control sites** – sites were outside of the turbidity zone of influence and therefore not modelled to be affected by dredging or dredge-related turbidity.
- **Putative Impact sites** – sites modelled as being within the turbidity zone of influence (no ecological impact). Thus, these sites were surveyed to test the hypothesis of the impact modelling i.e. that there had been no ecological impact.

The survey area was identified by the EPA (Attachment C of the approved dredge licence). Sites were selected based on previous seagrass mapping (April 2017) undertaken by BMT to ensure all representative benthic habitat types were sampled. These sites were further refined in consultation with the EPA to include sites at the location of water quality monitoring buoys, some additional sites on shallow intertidal areas north of the inner channel and to allow for sufficient statistical power to compare sampling events.

Each of the control and impact sites cover an area of 16 ha, composed of a 400 m by 400 m quadrant. There were 25 control sites and 10 impact sites within the survey area and within each of the sites, five 50 m replicate transects were surveyed. The five transect replicates within each site ensured that the sampling was representative within the site and there was sufficient power to detect a difference between sampling events at the site level. The survey design was estimated to provide a detection power (the power for detecting an effect of a given size) of 87% and therefore was designed

[4] Due to government imposed travel restrictions in response to the COVID-19 pandemic

to provide the necessary power at a critical feature cover of 15%. The survey power calculator output is shown in Figure 4-1.

Daily differing weather conditions meant that transect surveys approved in the SMP could not be exactly replicated within each site. On arrival at each transect point and prior to any video footage collection, the direction of drift of the boat due to winds/tides was determined. The direction of the 50 m transect was decided by the Master of the vessel and BMT staff but was collected within the pre-defined site boundary. Results were not impacted due to the level of replication between sites.

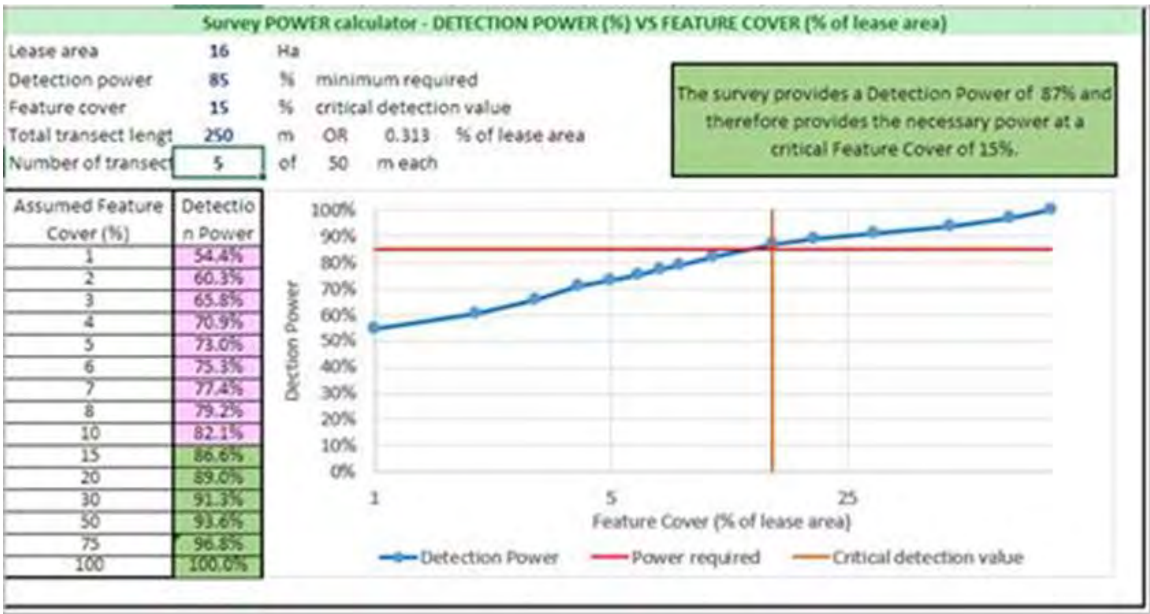

Figure 4-1 Screen shot of Survey Power calculator (provided by S. Gaylard, EPA)

4.2.2 Native Vegetation Significant Environmental Benefit (SEB) Site (Recovery sites)

In addition to sites surveyed to comply with the dredge licence requirements, additional 'recovery' sites were surveyed to understand seagrass impacts within the zones of impact (i.e. low to medium and high impact (as shown in Figure 4-2). The recovery sites are intended to determine seagrass recovery rates and SEB offset requirements post-dredging.

Recovery transect sites, consisted of 25 transects (one at each site) spaced along the channel. The transect sites did not have pre-defined start and finish points and consisted of a single start point. A 50 m transect was surveyed at each of the 25 recovery transect sites. The transect points were conducted at the same start locations, but transect orientation varied among survey campaigns due to the prevailing wind and tidal conditions.

Note that the recovery sites were used in addition to the BACI site (control and impact) to assist in ground-truthing satellite mapping and examine changes in the extent and distribution of seagrass percent cover and composition. Data from the recovery sites were analysed based on estimated cover, rather than using point-count methods. Therefore, these sites have not been compared quantitatively with the control or impact sites for the Dredge Licence Condition Seagrass Assessment (BACI design). Three recovery sites fell within the ZOHI, eight sites fell within the ZOLMI, 18 sites fell within the ZOI.

LEGEND

- Existing Channel Footprint
- Channel Widening Footprint
- Control site
- Impact Site
- Control/Impact Transect Point
- Recovery Transect Point (Native Vegetation SEB survey)
- Turbidity Monitoring Buoy Location
- Turbidity zone of influence (no ecological impact)
- Turbidity zone of Low to Moderate Impact
- Turbidity zone of High Impact
- Survey Area (as per Dredge Licence)

Title:

Seagrass Monitoring Program Transect Locations

Figure: **4-2**

Rev: **A**

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4.3 Survey Method

The field survey involved the use of towed video camera transects in sub-tidal and intertidal areas to identify seagrass species composition and percentage cover.

Benthic trawls using a modified seagrass rake were used to collect seagrass specimens for species identification during the baseline survey. Trawls were not required in the post-dredging survey (2020) as all seagrass identifications (to genus) were possible using the towed video camera.

Towed video transects used a high-definition camera (3840 x 2160 pixels per frame) with a wide-angle lens. The camera was flown at ~1 m above the substratum at a speed of 1–2 km/h facing downwards at a 90-degree angle. All footage was recorded onto the internal camera memory, while composite standard definition footage was relayed to a screen on the vessel for real-time data analysis by a trained marine ecologist. All equipment (laptop, GPSs and the camera) was set to the same date/time to ensure that habitat data collected along each transect could be geo-referenced onto mapping (as appropriate). The length of the 50 m transects began at the point of clear footage at the seafloor.

For recovery sites, cover was estimated using seagrass percentage cover standards (McKenzie 2003, see Figure 4-3). The video footage collected was reviewed post-field to ensure a consistent approach to habitat classification and to match the survey data to the GPS tracklog (by date and time). The percentage cover, seagrass species/genus and other variables of interest (e.g. algae, bivalves and benthic habitat types) were recorded, taking additional GPS marks at locations where the densities and/or species composition changed.

Specifically, for the Dredge Licence Condition Seagrass Assessment (i.e. for the transects conducted at the control and impact sites), the video camera was set to take a still image every five seconds (concurrent with the video footage). These images were used to inform the quantitative seagrass percentage cover and statistical assessment. The point count software (CORALNET) was used for the image analysis, with classifications made from a random assignment of five points. This methodology removes the need to know the depth from the seabed and allowed transects to be completed in a more efficient manner.

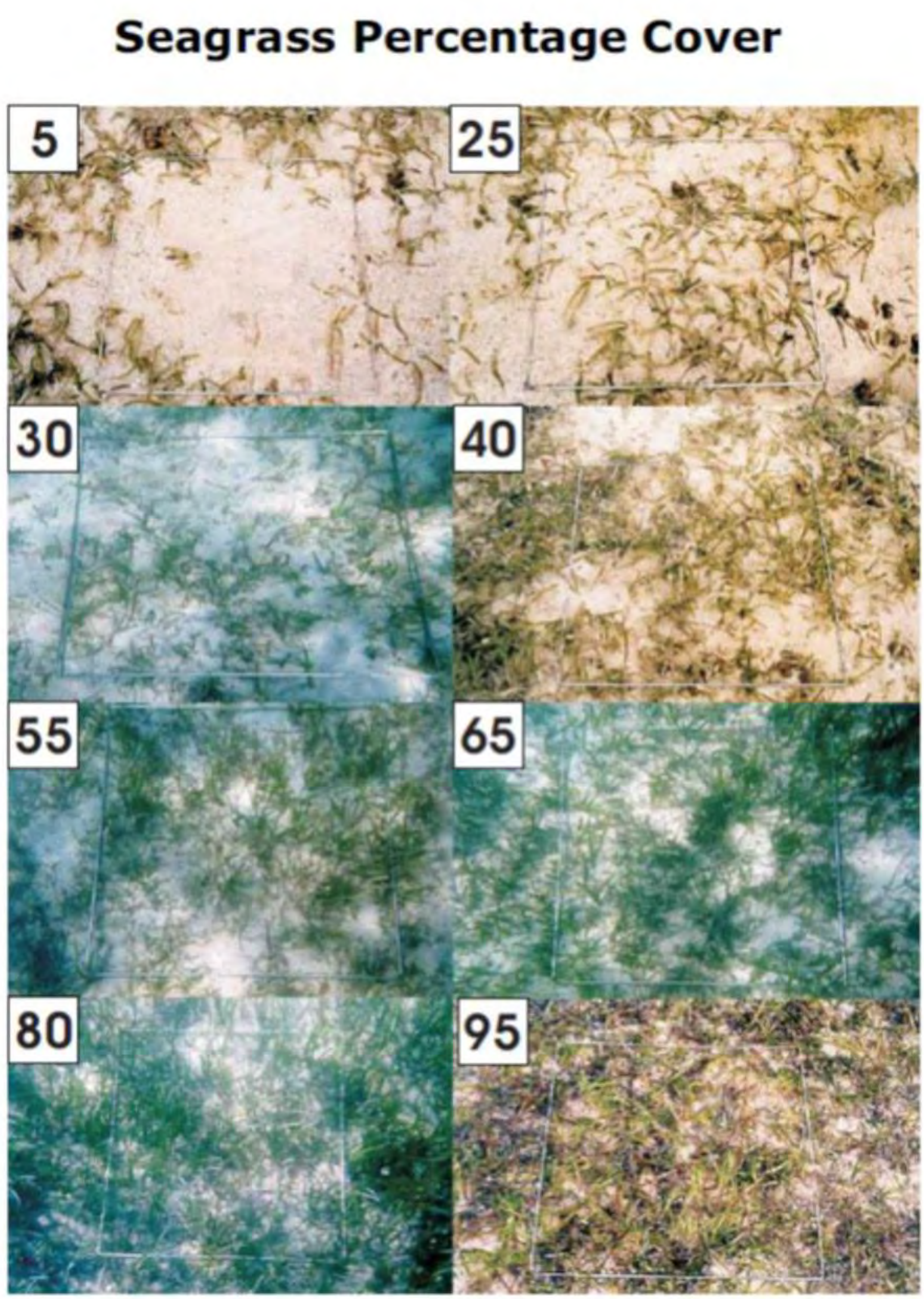


Figure 4-3 Seagrass percentage cover estimates (McKenzie 2003)

4.4 Survey Results Analysis and Reporting

4.4.1 Dredge Licence Condition Seagrass Assessment

In accordance with the SMP, after each survey was undertaken, field survey data was analysed using suitable techniques and documented (including mapping).

4.4.1.1 Point Count Analysis

Automated annotation software (CORALNET) was used to quantify cover of each seagrass taxa on each transect. An average of 14 frames were sampled per transect were sampled in 2019, whereas an average of 28 frames per transect were sampled in 2020. The difference in numbers of frames between surveys was a consequence of the higher image quality in 2020. A total of five sub-sample points were classified in each frame, resulting in an average of 70 and 139 sub-samples points per transect in 2019 and 2020, respectively. Each transect was used as the sample unit (i.e. sub-samples within transects were not considered in statistical analyses).

The cover of major benthic categories and seagrass taxa was quantified for each sub-sample point. Fully automated approaches have provided comparable results to human expert classifications for coral substrates, while algal substrates have required semi-automated annotation (Beijbom *et al*., 2015). The CORALNET program allows the user to set the level of automated assistance based on estimates of the machine accuracy (Cohen's kappa) for major functional groups of benthos. This estimate is progressively updated with more training (manual classification) as the project progresses. Beijbom *et al*. (2015) show that a 5% reduction in accuracy does not have a significant impact on the cover estimates of functional groups, therefore, the level of automation was set throughout the project to ensure at least 95% automation accuracy. This was not achieved; therefore, all automated identifications were confirmed by a marine ecologist.

4.4.1.2 Statistical Methods

Frequency distributions for the fully quantitative BACI data, including total seagrass cover, seagrass wrack, and each of the seagrass taxa were visualised to better understand the structure of the data and appropriate analyses. These are shown in Figure 4-4, and display error structures fitting that of either beta or Poisson distributions. Response variables including all the individual taxa were heavily biased towards zero counts and rarely 'capped' at 100% cover. Due to the extreme zero-bias in these data, including complete absence at some locations, these variables were analysed using quasi-Poisson error distributions, while total seagrass cover was analysed with beta regression.

Percent cover data of total seagrass were analysed using the generalised linear mixed modelling package (glmmTMB) in R. Sampling year and treatment were considered random factors with site treated as a fixed factor. The beta regression used zero-one inflated with counts converted to percent, while quasi-Poisson models used point-count data. Quasi-Poisson models were fitted without specific inclusion of site as a fixed factor within generalised linear models. These methods are resilient to many of the issues affecting this dataset including non-normality (strong zero-bias), heteroscedacity (differing levels of variation associated with treatments), and design imbalance (more sites in control than impact treatments).

The BACI framework was used to interrogate the interaction between treatment (control and impact) and survey event (effectively before and after dredging). A significant interaction in these terms, specifically a reduction in cover at impact sites relative to controls, was interpreted as an impact potentially related to dredging.

For recovery sites, a quasi-Poisson generalised linear model was used to investigate differences in cover among the various zones of impact (ZOI, ZOLMI, and ZOHI) occurring between 2019 and 2020. These data were analysed separately to the fully quantitative BACI data due to differences in replication and cover estimation methodology.

Multiple comparisons among sites and treatments were investigated with the enmeans package, and graphs were produced with ggplot2 in R.

4.4.1.3 Satellite Mapping

Satellite imagery was used to establish the full extent of seagrass coverage outside of the surveyed transects. Sentinel imagery was downloaded for analysis, with captures from April 12, 2019 and April 14, 2020 used in the analysis. These images were selected as the clearest imagery available for April i.e. the least cloud cover or turbidity. Depth Invariant Indices (DII) were created for images remove water-column artefacts for benthic classification. Habitat classifications (including seagrass percent cover and composition) were derived using the DII. All of these operations were conducted in ArcGIS version 10.5. For additional detail regarding these methods see Appendix B.

The DII was also classified for distribution of species and their coverage, this was split into eight classes:

- Bare substrate
- Macroalgae – dominant coverage
- Low density – <35% seagrass coverage
- Moderate density – 35–70% seagrass coverage
- High density – >70% seagrass coverage
- *Heterozostera* low density cover – <35% coverage
- *Heterozostera* moderate density cover – 35-70% coverage
- *Heterozostera* high density cover – >70% coverage.

While mapping based on remote sensing was performed within the entire Survey Area, it should be noted that the western (deeper) half of this area was considered a low-confidence region due to occasional excessive turbidity in combination with a reduced seafloor signal due to depth. The western (low confidence) area should be considered as indicative, while the eastern half can be considered more reliable. Also note that the repeatability low-density seagrass mapping, particularly in deeper areas is also subject to variations in water clarity, and should be interpreted using other lines of evidence, such as quantitative cover data. Seagrass extents in each of the modelled impact zones were calculated.

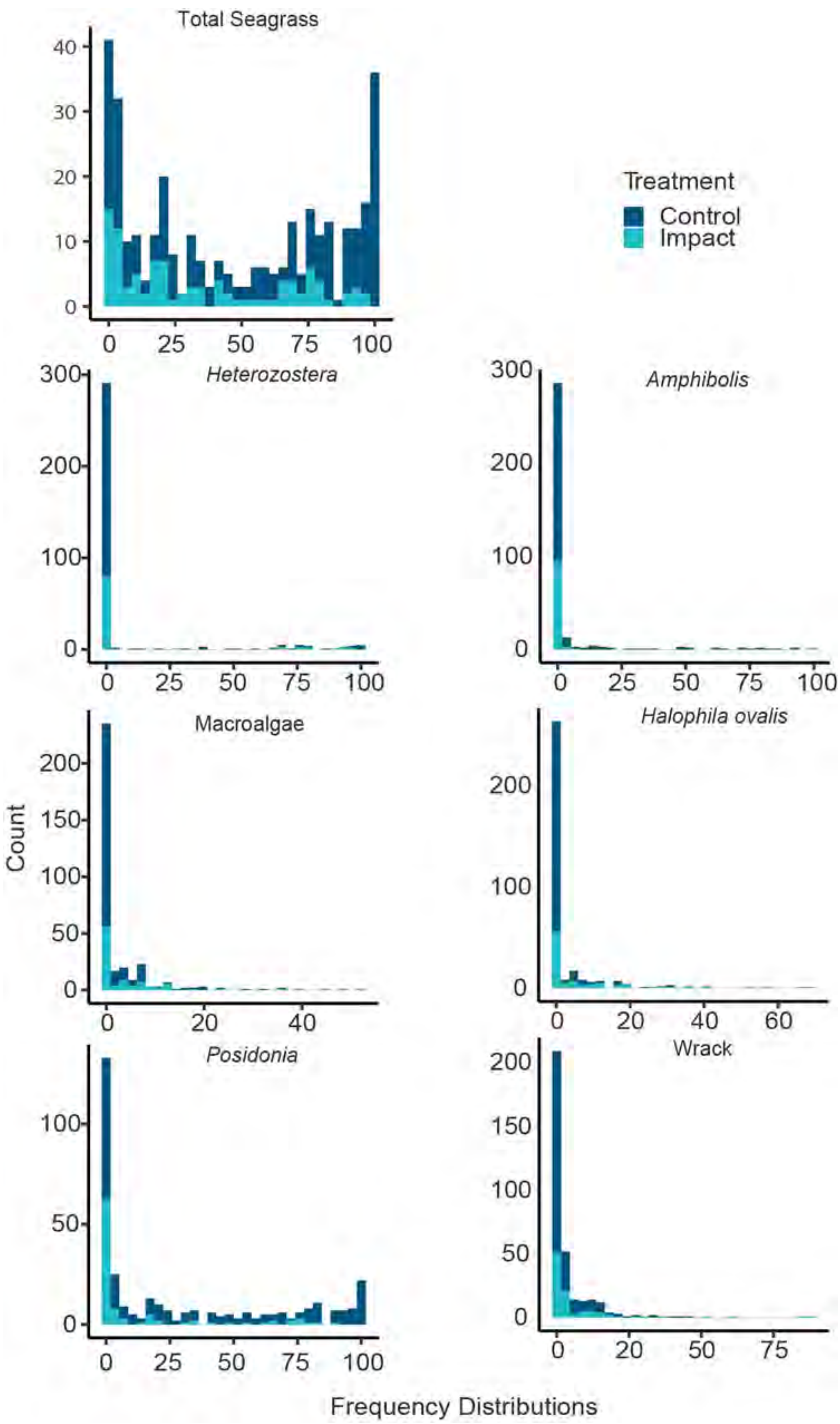


Figure 4-4 Frequency distributions for major univariate response variables including total seagrass, *Heterozostera, Amphibolis*, macroalgae, *Halophila, Posidonia*, and wrack

5 Results

5.1 Benthic Primary Producer Species

5.1.1 Species Composition

Four seagrass genera were recorded: *Posidonia*, *Amphibolis*, *Halophila* and *Heterozostera*. The total percent cover varied widely across transects with individual transects ranging between 0 and 100% seagrass cover, while site averages (of transect data) varied between 2.1 and 94.8%.

The seagrass community composition is shown in Figure 5-1. Most of the control sites were dominated by the three perennial seagrass genera: *Amphibolis*, *Heterozostera* and *Posidonia.* Impact sites typically had a higher proportion of *Halophila* and where cover exceeded 50%, this was made up of either *Posidonia* or *Heterozostera*.

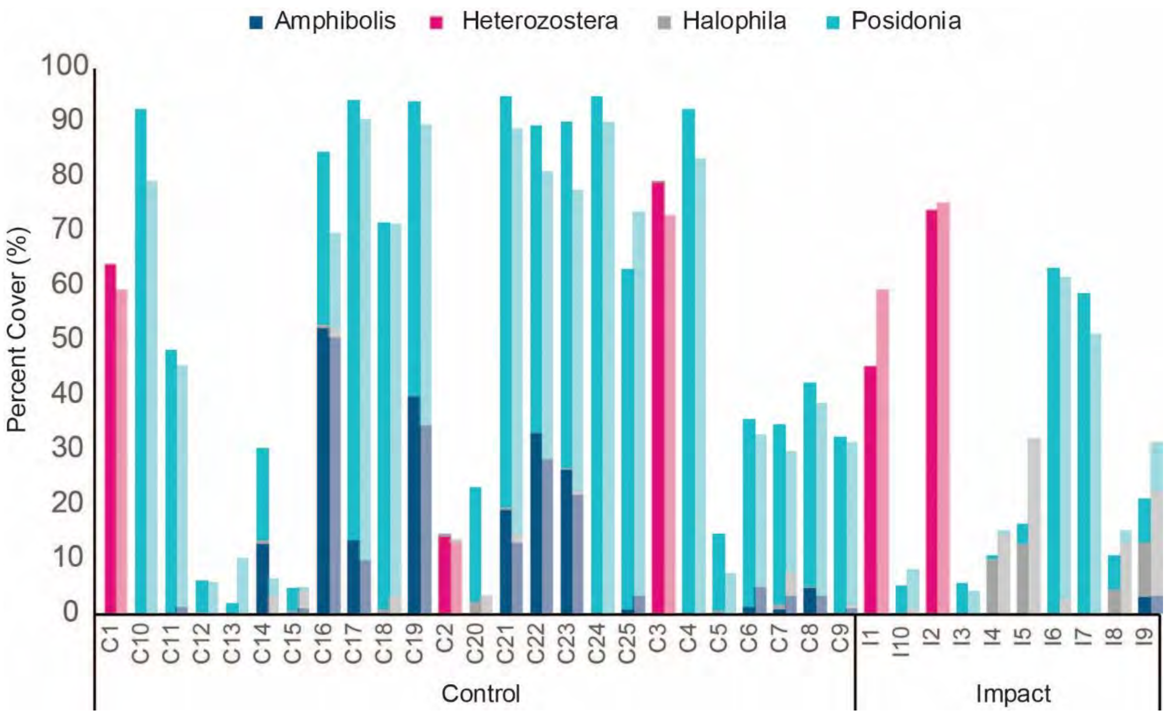


Figure 5-1 Comparison of seagrass species composition (site-averaged percent cover) at the control and impact sites. 2019 data are shown in full tones, 2020 data are shown in half-tones

5.1.2 Statistical Analysis

5.1.2.1 Univariate BACI Analyses

Univariate comparisons in the cover of each of the seagrass species, seagrass wrack, and total seagrass suggest that putative impact areas did not decline significantly in total cover compared to control areas (Figure 5-2). For total seagrass cover there was a significant effect of year ($p = 0.028$), but neither the effect of treatment, or the interaction between treatment and event were significant (Table 5-1). Independent contrasts for each treatment showed that the decline in cover at the control treatment between years was significant (p = 0.008) while the temporal differences for the impact treatment were not significant.

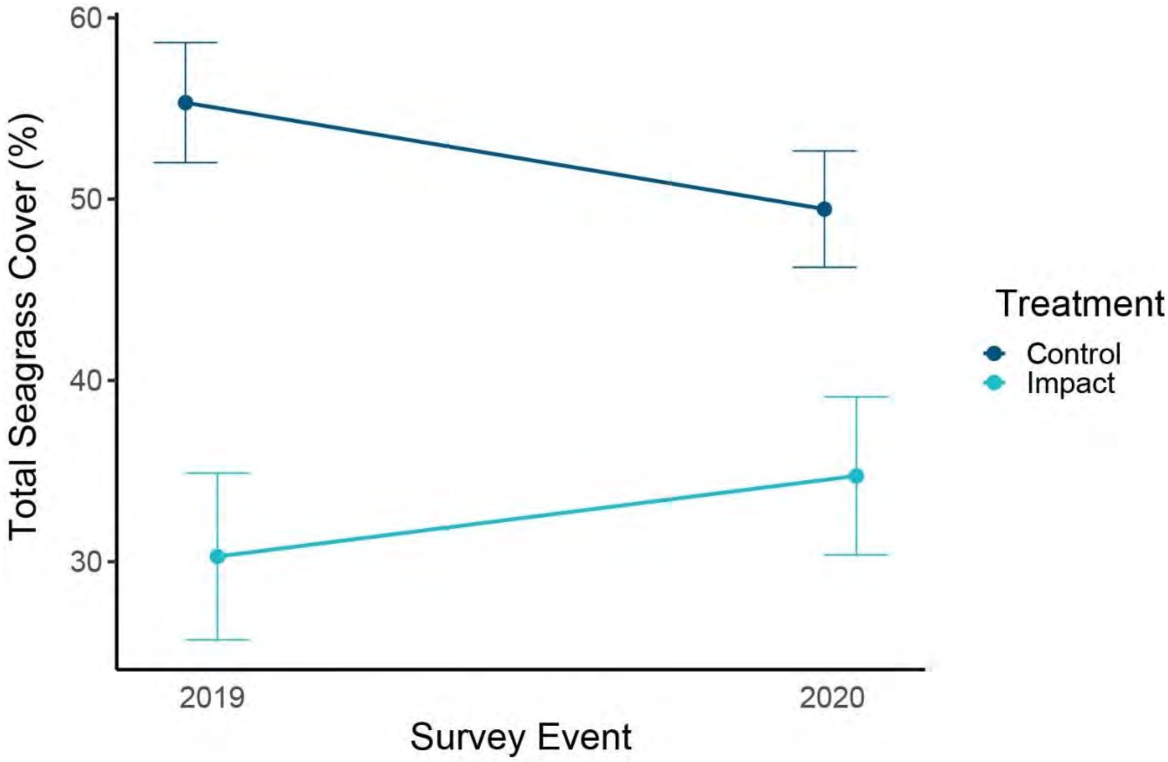


Figure 5-2 Changes in total seagrass cover (mean ± se) between 2019 and 2020 at control and putative impact BACI sites.

Site-averaged changes in total seagrass cover provide a more detailed perspective of how seagrass cover varied the two survey events at control and impact sites (Figure 5-3). Most control sites varied little with slight reductions in cover between 2019 and 2020. Exceptions to this were control sites C14, C16, C20, which had reasonably large reductions in the cover total cover, related to a loss of *Posidonia* (Figure 5-1). Conversely, most impact sites also varied little in total cover between 2019 and 2020, apart from impact site I5 which had a relatively large increase in total cover (Figure 5-3) driven by an increase in the cover of *Halophila* (Figure 5-1). The reduction in cover at site I7 was also associated with very large variation (Figure 5-3) which was due to an inconsistent response within the site; two transects did not change and reductions in cover occurred at three other transects (see Figure 5-7).

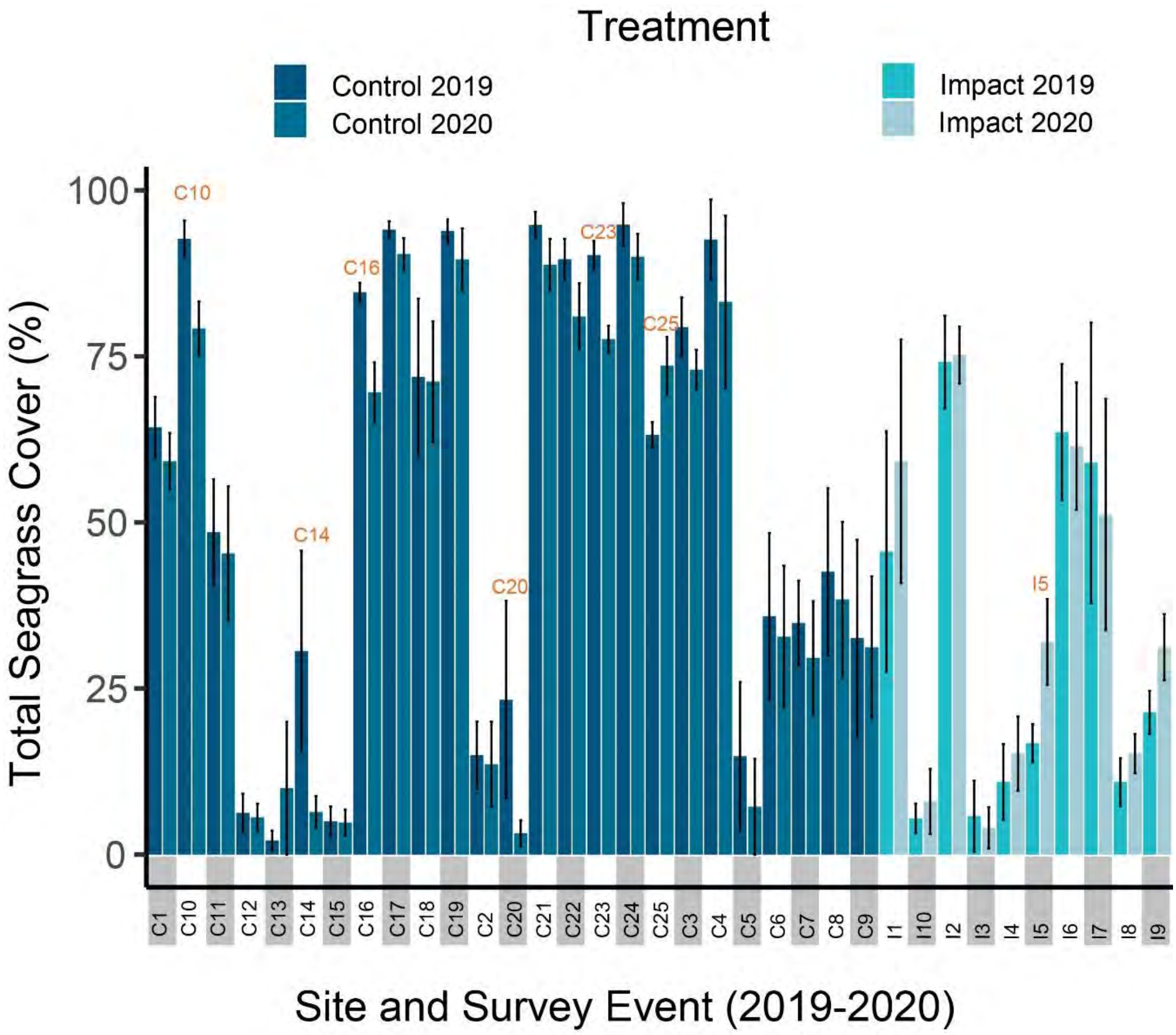


Figure 5-3 Site-level changes in total seagrass cover (mean ± se) between 2019 and 2020 at control and putative impact BACI sites

Temporal changes in individual seagrass taxa and macroalgae between treatments and surveys did not suggest that reductions in seagrass cover had occurred at putative impact sites relative to control areas (Figure 5-4). The effect of Treatment was significant for all individual seagrass taxa, suggesting that overall the cover of each seagrass taxon differed between control and impact sites, but there were no significant BACI interaction terms for any of these variables (Table 5-1). *Halophila* was the only taxon that differed significantly between survey events, with an increase in cover in 2020 (Figure 4-4, Table 5-1). Overall these results do not show any form of reduction in seagrass cover overall or for any individual seagrass taxa in impact sites.

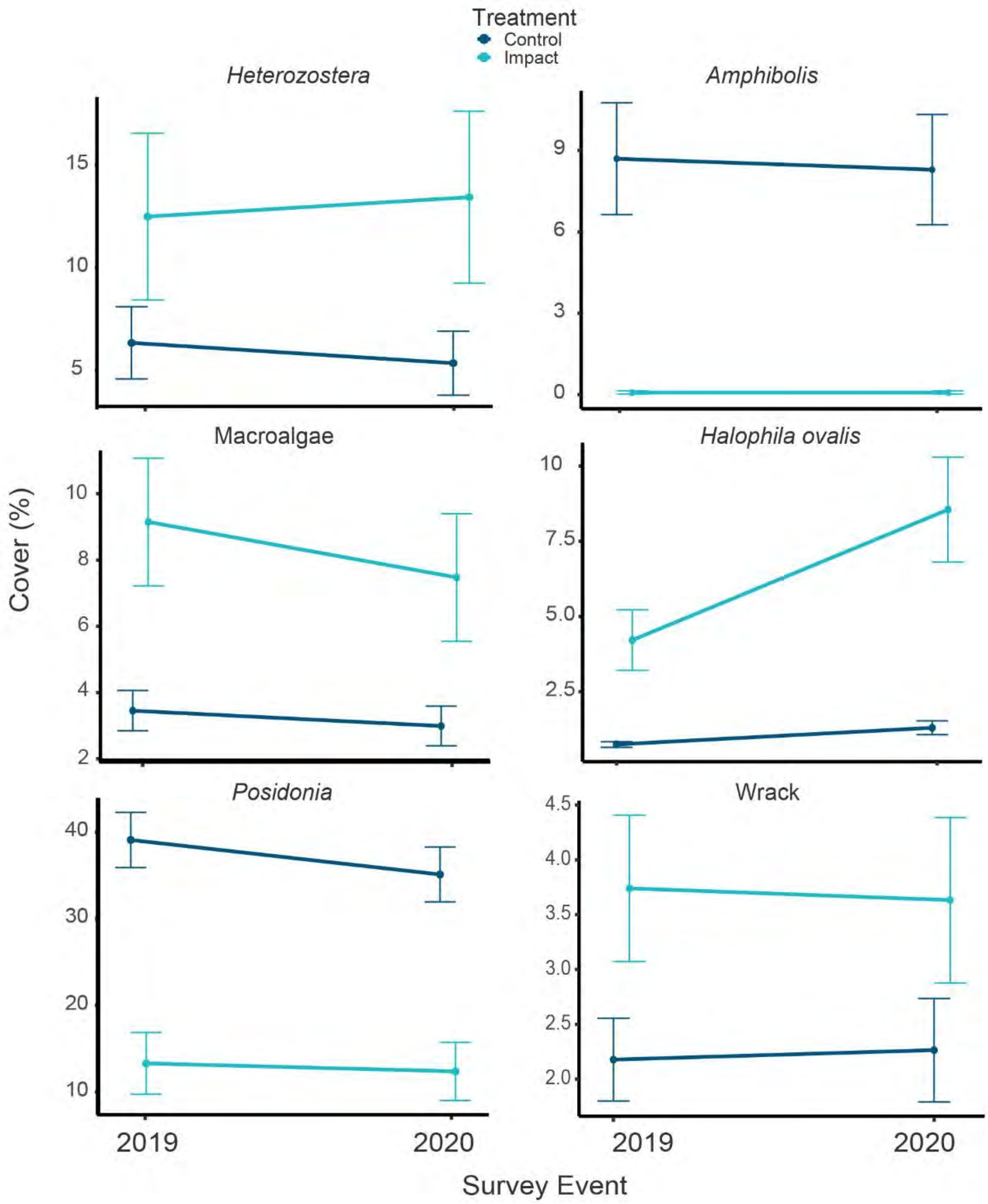


Figure 5-4 Changes in the cover of individual benthic primary producer taxa (mean ± se) between 2019 and 2020 at control and putative impact BACI sites. Significant interactions suggesting impact, were observed for Amphibolis and Halophila.

Table 5-1 Summary Analysis of Deviance Table (Type II Wald) for generalized linear models. Significance codes: 0 = *, 0.001 =**, 0.01 = *, 0.05 = .**

Response	Parameter	Chi Sq	df	P>Chi
Amphibolis				
	Year	0.3361	1	0.5621
	Treatment	27.7034	1	**1.414e-07** ***
	Year*Treatment	0.0187	1	0.8912
Macroalgae				
	Year	0.7357	1	0.3911
	Treatment	21.3723	1	**3.782e-06** ***
	Year*Treatment	0.0211	1	0.8846
Heterozostera				
	Year	0.0360	1	0.849500
	Treatment	6.8444	1	**0.008892** **
	Year*Treatment	0.1656	1	0.684052
Halophila				
	Year	13.497	1	**0.000239** ***
	Treatment	102.066	1	**< 2.2e-16** ***
	Year*Treatment	0.102	1	0.749466
Wrack				
	Year	0.0036	1	0.95203
	Treatment	5.8055	1	**0.01598** *
	Year*Treatment	0.0275	1	0.86820
Posidonia				
	Year	0.744	1	0.3884
	Treatment	43.265	1	**4.78e-11** **
	Year*Treatment	0.008	1	0.9294
Total Seagrass				
	Year	4.7973	1	**0.0285** *
	Treatment	2.1089	1	0.1464
	Year*Treatment	2.3036	1	0.1291

5.1.2.2 Recovery Site Analysis

Average seagrass cover was less than 1% before and after dredging at recovery sites (Figure 5-5). There was a trend of reduced total seagrass cover over time, but this was not statistically significant (*p* = 0.058) (Figure 5-5, Table 5-2). Within each of the zones individually, there were no significant differences in cover between 2019 and 2020; the largest difference observed at the ZOI had a p-value of 0.11.

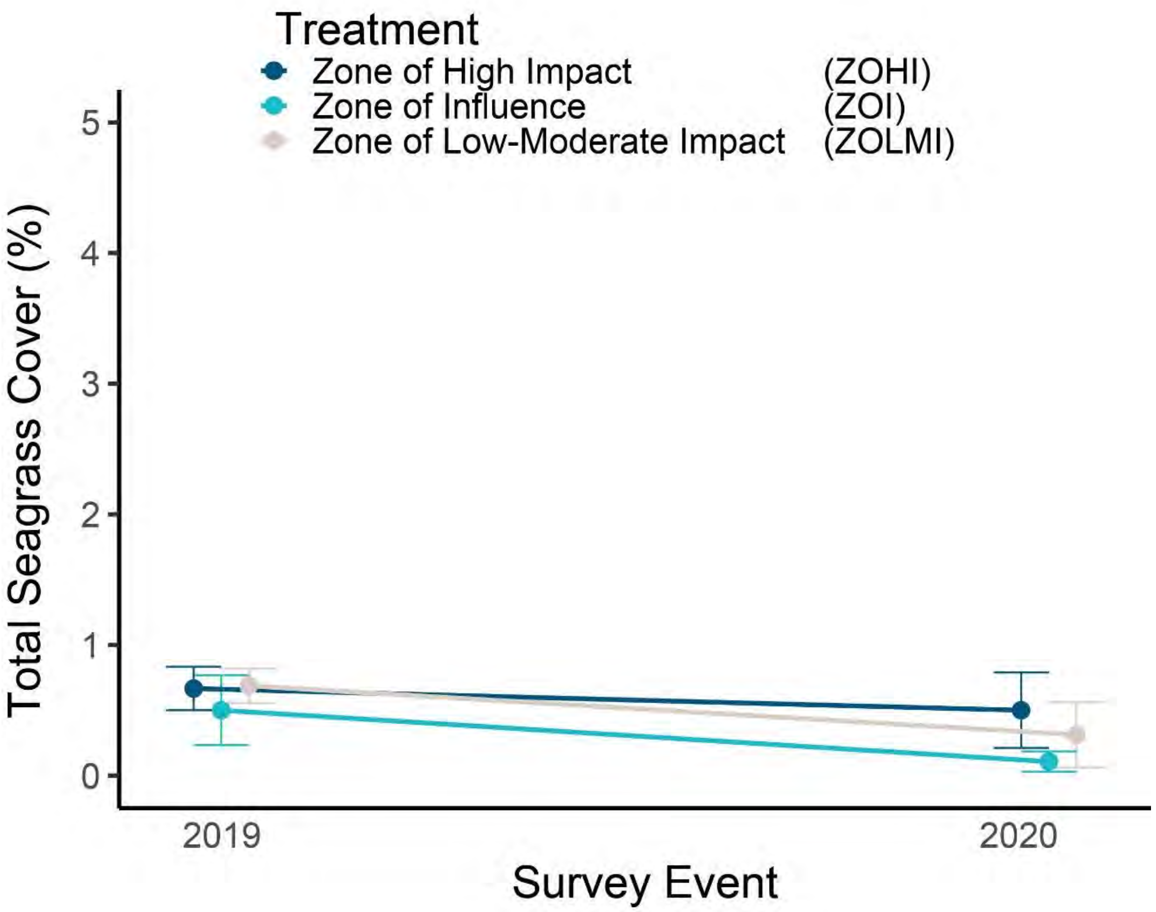


Figure 5-5 Changes in total seagrass cover (mean ± se) between 2019 and 2020 at the different type of recovery site (ZOHI, ZOI, ZOLMI)

Table 5-2 Summary Analysis of Deviance Table (Type II Wald) for generalized linear models. Significance codes: <0.0001 = **, <0.001 =**, 0.01 = *, <0.05 = .**

Response	Parameter	Chi Sq	df	P>Chi
Total Seagrass				
	Year	3.58	1	**0.058** .
	Treatment	1.27	1	0.52
	Year*Treatment	0.75	1	0.69

5.2 Seagrass Habitat Mapping

5.2.1 Spatial and Temporal Patterns

Seagrass meadows were dominated by perennial *Posidonia* or *Amphibolis* species, with ephemeral, *Halophila* genera highly abundant in meadows that ranged in depth between 6-15 m. *Heterozostera* was predominately observed in shallow intertidal areas (typically <1 m) but was also observed in depths up to 4.5 m in lower densities.

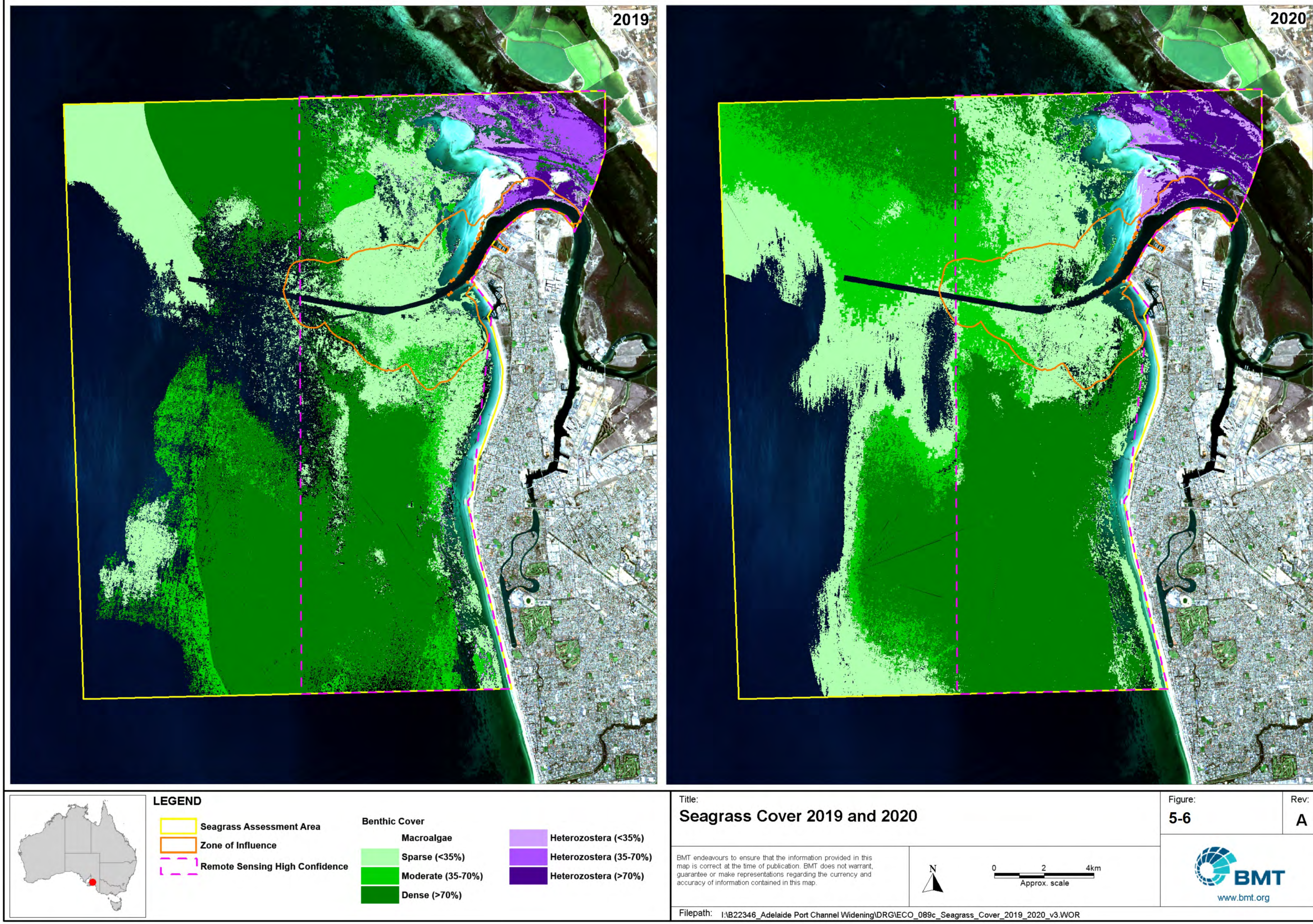
The seagrass habitat maps from the April 2019 and May 2020 are shown in Figure 5-6. Within the entire Survey Area, the total seagrass cover within the Survey Area (combining all species and percentage coverages) increased from 62.4% in 2019 to 74.1% in 2020 (Figure 5-6, Figure 5-7). Most of the mapped 'gains' occurred in deep waters in the western portion of the Survey Area, which had low mapping confidence (Section 5.2.2). Based on field surveys in this area, there is a high likelihood that seagrass meadow extent in this area was under-represented in the 2019 mapping.

The mapped seagrass extents within modelled impact zones are shown in Table 5-3. Remote sensing typically did not detect seagrass in and immediately adjacent to the shipping channel (within ZOHI) in both surveys. This is supported by field observations at recovery sites, which indicated that these areas were mostly bare, with some macroalgae was observed towards its western end and in isolated patches throughout, and seagrass cover <1% (see Section 5.1.2.2).

Within the zone of low to moderate impact, there was a gain in *Heterozostera* cover between 2019-2020, but no change in other seagrass categories. Within the zone of influence, while there was a decrease of approximately 50 ha of sparse seagrass cover in 2020, 'gains' in *Heterozostera* cover and moderate mixed assemblages, and no change in dense seagrass meadows.

Table 5-3 Mapped Seagrass Extent (ha) by Modelled Impact Zones - 2019 and 2020

Classification Category	Classification Sub-category*	ZOHI/ZLMI		ZOI	
		2019	2020	2019	2020
Moderate to dense mixed species seagrass (35-100%)	Moderate seagrass	≤0.5	≤0.5	196	329
	Dense seagrass	≤0.5	≤0.5	101	111
Sparse mixed species seagrass (1-35%)	Sparse seagrass	7	8	1329	1278
Seagrass dominated by *Heterozostera* (1-100%)	Sparse *Heterozostera*	19	31	118	141



Title:

Seagrass Cover 2019 and 2020

Figure: 5-6

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While many of these changes are likely the result of changes in density classifications, or artefacts of depth and turbidity in the western extent from the 2019 scene, it is clear that there has not been a consistent reduction in seagrass cover in the zone of influence, consistent with field observations described in Section 5.1.2.1.

Figure 5-7 shows 2020 seagrass cover and seagrass cover changes at each site. As shown in earlier in Figure 5-3, field observations indicated that most control sites varied little over time. Exceptions to this were control sites C10, C14, C16, C20, which had reductions in total cover. Site C20 also had a large overall reduction, but there was great within -site heterogeneity (Figure 5-7).

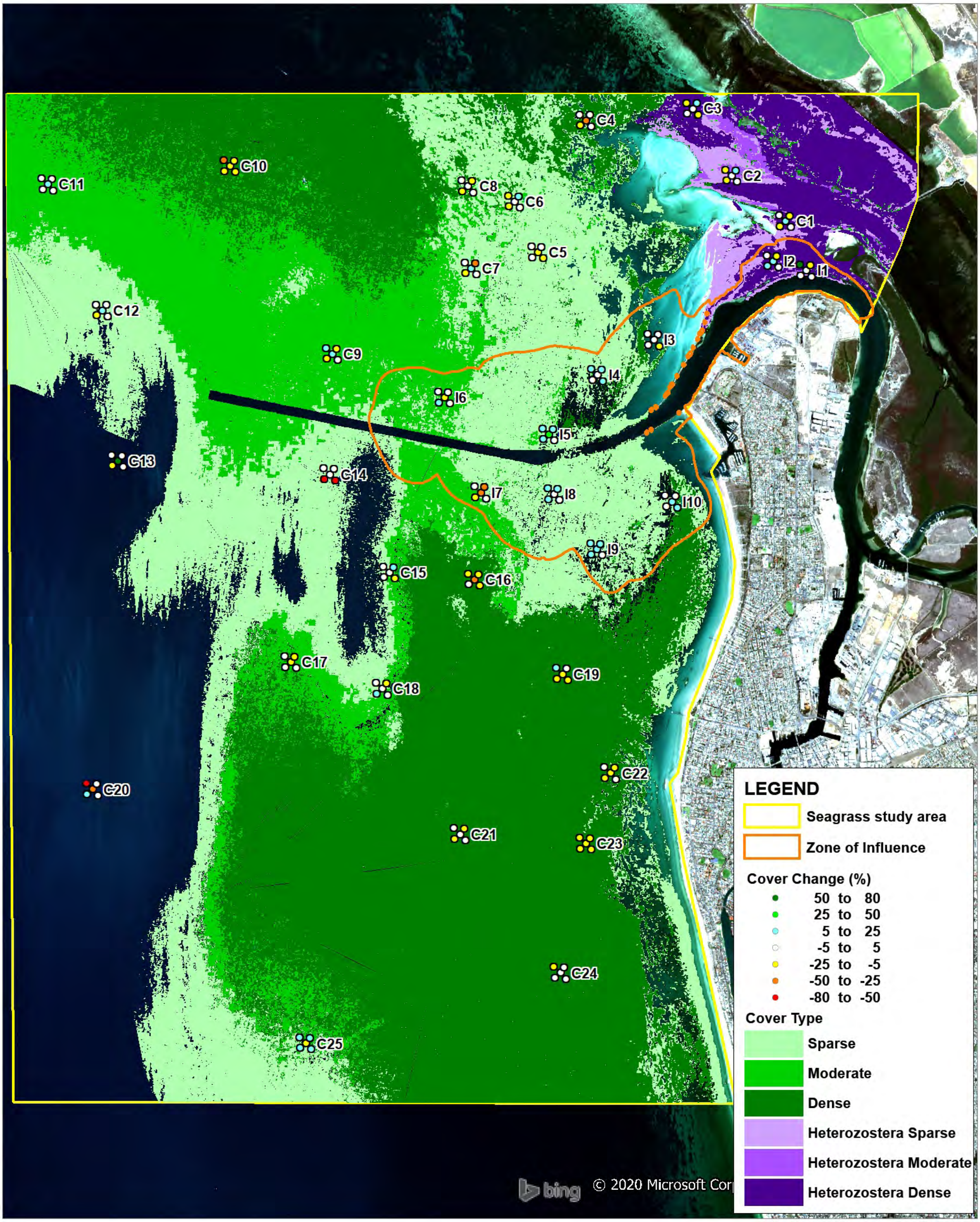
Most impact transect also varied little in total cover between 2019 and 2020, or slightly increased, apart from impact site I5 which had a relatively consistent increase in total cover (Figure 5-7) driven by an increase in the cover of *Halophila* (Figure 5-1). The only consistent reduction in cover for the impact sites was at site I7 where two transects remained unchanged and cover reduced at three other transects (see Figure 5-7).

The spread and inconsistency of response in control and impact sites do not indicate a dredging impact or geographically consistent change within the control area.

5.2.2 Mapping Limitations

The quality of the remotely sensed estimate of seagrass cover was affected by water depth, differences in turbidity signal between captures, and thresholds of detection for substrates with low seagrass cover. Generally, the remote sensing limit for satellite imagery is approximately 20 m and the ease of detection and classification lessons with depth (Mishra *et al*. 2006).

Initial, more complex classifications attempting to separate many of the seagrass genera were performed and assessed. Point overlays were used to determine the quality of the classifications, but these classifications were not able to differentiate between the seagrass genera in the deeper habitats. This resulted in the deeper communities being classified on the basis of total percentage cover, rather than species composition. Many of the field transects were conducted approximately 20 m depth, at the edge of the classification and differentiation threshold. Most of the seagrass meadows were also composed of mixed communities and therefore the spectral signature was very similar. Ephemeral seagrass species were difficult to map due to their low density (0-10%) and as they were often interspersed between other species. *Heterozostera* could be differentiated from the other seagrass genera as it was predominately found in the shallow and intertidal regions of the Survey Area and therefore classifications had stronger spectral signatures for differentiation. The remote sensing capabilities in the zone of impact were also affected by the sparse cover of seagrass. Mixed communities, heterogeneity within transects, and low cover affected the precision of the classification.



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Seagrass Cover Change

Figure: 5-7

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6 Summary

The pre- and post-dredging seagrass surveys found the dominant seagrass genera were *Posidonia* and *Amphibolis,* which occurred throughout the Survey Area. The other notable seagrass genera were *Halophila* (typically sparse in cover at depths of up to 16 m) and *Heterozostera* (predominately found in shallow intertidal areas). The zone of impact was typically comprised more of *Halophila* than the surrounding 'control' area, which was dominated by *Posidonia* and *Amphibolis*.

Recovery transects in the shipping channel showed that cover was typically very low in 2019, and there was some evidence that this was reduced further in 2020.

Overall, the BACI design testing for dredge plume effects outside of the channel did not indicate a significant interaction, where cover was reduced in impact sites relative to control sites. Instead, it was more likely that cover in control areas was lower in 2020, and cover in putative impact areas remained unchanged, based on BACI data. The relatively benign effect of the dredging activities on surrounding seagrass communities is likely related to dredging methodology and dredge management in accordance with turbidity limits. While dredge plumes were observed in MODIS satellite data, these were relatively insignificant compared to turbidity generated by several extended periods of south-westerly gales (BMT 2019 a, b, c, d).

There was a non-significant increase in the overall average seagrass cover for the impact treatment. While cover typically increased in many transects in the ZOI due to small increases in *Halophila,* the inconsistency of the response within and among sites meant that the overall effect of 'impact' was not significant.

Consistently reduced cover at several control sites indicates an actual reduction in cover. However, sites where a strongly inconsistent response was observed, particularly where slow growing species greatly increased in cover, are likely to be the result of variable transect placement over patchy habitat. The location of sites with a consistent reduction in total cover did not suggest there had been a defined dredging impact or geographically consistent change in the control area.

Patterns in the remote-sensing mapping of moderate to dense seagrass in 2018 and 2019 were generally consistent, with an increase in seagrass cover observed in the zone of influence and wider Survey Area. These increases typically occurred in the deeper area southwest of the shipping channel and are likely an anomaly relating to the quality of the 2019 capture.

Difficulties in remote sensing deeper waters, anomalous turbidity signals between captures, and thresholds of detection for substrates with low seagrass cover, require remotely sensed changes in seagrass extent to be interpreted cautiously. These patterns should be used in addition to more robust but spatially limited methods such as the BACI and ground-truthing data. Remote sensing assessments in the shallow, higher-confidence zone of influence do not suggest seagrass extent was reduced between 2019 and 2020.

The Seagrass Monitoring Program consists of three field surveys throughout the Project lifecycle. This current survey is the first post-dredging survey. The 2019 survey was the baseline (pre-dredging survey), conducted in April 2019 and a subsequent survey will be an addition post-dredging survey, scheduled for April 2022, approximately two years post dredging activities ceasing. The surveys have been scheduled at the same time each year to account for seasonal fluctuation in seagrass extent and growth.

7 References

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http://www.epa.wa.gov.au/sites/default/files/Policies_and_Guidance/Technical%20Guidance%20-%20EIA%20of%20Dredging%20Proposals-131216.pdf

Appendix A Generalised Linear Models

```
> mod_Amphibolis <- glm(Amphibolis ~ Year * Treatment, data=sgcover,
family=quasipoisson())
> summary(mod_Amphibolis)

Call:
glm(formula = Amphibolis ~ Year * Treatment, family = quasipoisson(),
    data = sgcover)

Deviance Residuals:
    Min       1Q   Median       3Q      Max
-4.0478  -4.0478  -3.7262  -0.7534  16.2709

Coefficients:
                      Estimate Std. Error t value Pr(>|t|)
(Intercept)           336.3901   565.6043   0.595    0.552
Year                   -0.1656     0.2801  -0.591    0.555
TreatmentImpact      -620.6103  4524.0660  -0.137    0.891
Year:TreatmentImpact    0.3057     2.2401   0.136    0.892

(Dispersion parameter for quasipoisson family taken to be 35.93559)

    Null deviance: 7887.9  on 339  degrees of freedom
Residual deviance: 6877.2  on 336  degrees of freedom
AIC: NA

Number of Fisher Scoring iterations: 7

> Anova(mod_Amphibolis)
Analysis of Deviance Table (Type II tests)

Response: Amphibolis
               LR Chisq Df Pr(>Chisq)
Year             0.3361  1     0.5621
Treatment       27.7034  1  1.414e-07 ***
Year:Treatment   0.0187  1     0.8912
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
>
> mod_Macroalgae <- glm(Macroalgae ~ Year * Treatment, data=sgcover,
family=quasipoisson)
> summary(mod_Macroalgae)

Call:
glm(formula = Macroalgae ~ Year * Treatment, family = quasipoisson,
    data = sgcover)

Deviance Residuals:
   Min      1Q  Median      3Q     Max
-4.287  -2.635  -2.453   0.544  11.995

Coefficients:
                      Estimate Std. Error t value Pr(>|t|)
(Intercept)          290.76987  578.20941   0.503    0.615
Year                  -0.14340    0.28632  -0.501    0.617
TreatmentImpact      119.27063  815.44639   0.146    0.884
Year:TreatmentImpact  -0.05859    0.40379  -0.145    0.885

(Dispersion parameter for quasipoisson family taken to be 16.11039)

    Null deviance: 4132.1  on 339  degrees of freedom
```

```
Residual deviance: 3777.2  on 336  degrees of freedom
AIC: NA

Number of Fisher Scoring iterations: 6

> Anova(mod_Macroalgae)
Analysis of Deviance Table (Type II tests)

Response: Macroalgae
               LR Chisq Df Pr(>Chisq)    
Year             0.7357  1     0.3911    
Treatment       21.3723  1  3.782e-06 ***
Year:Treatment   0.0211  1     0.8846    
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
> 
> mod_Ho <- glm(Halophila ~ Year * Treatment, data=sgcover, family=quasipoisson)
> summary(mod_Ho)

Call:
glm(formula = Halophila ~ Year * Treatment, family = quasipoisson, 
    data = sgcover)

Deviance Residuals: 
    Min       1Q   Median       3Q      Max  
-4.0708  -1.3606  -0.8524  -0.8524  10.1779  

Coefficients:
                       Estimate Std. Error t value Pr(>|t|)  
(Intercept)          -1889.5182  1011.4549  -1.868   0.0626 .
Year                     0.9354     0.5008   1.868   0.0627 .
TreatmentImpact        361.2775  1132.4860   0.319   0.7499  
Year:TreatmentImpact    -0.1778     0.5607  -0.317   0.7514  
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

(Dispersion parameter for quasipoisson family taken to be 8.010928)

    Null deviance: 2820.4  on 339  degrees of freedom
Residual deviance: 1885.5  on 336  degrees of freedom
AIC: NA

Number of Fisher Scoring iterations: 6

> Anova(mod_Ho)
Analysis of Deviance Table (Type II tests)

Response: Halophila
               LR Chisq Df Pr(>Chisq)    
Year             13.497  1   0.000239 ***
Treatment       102.066  1  < 2.2e-16 ***
Year:Treatment    0.102  1   0.749466    
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
> 
> mod_Wrack <- glm(Wrack ~ Year * Treatment, data=sgcover, family=quasipoisson)
> summary(mod_Wrack)

Call:
glm(formula = Wrack ~ Year * Treatment, family = quasipoisson, 
    data = sgcover)

Deviance Residuals: 
    Min       1Q   Median       3Q      Max  
-2.7350  -2.1281  -2.0870   0.8399  12.4198  
```

```
Coefficients:
                       Estimate Std. Error t value Pr(>|t|)
(Intercept)           -78.00332  521.61088  -0.150    0.881
Year                    0.03902    0.25829   0.151    0.880
TreatmentImpact       138.23537  829.92273   0.167    0.868
Year:TreatmentImpact   -0.06820    0.41095  -0.166    0.868

(Dispersion parameter for quasipoisson family taken to be 9.036426)

    Null deviance: 2158.8  on 339  degrees of freedom
Residual deviance: 2106.0  on 336  degrees of freedom
AIC: NA

Number of Fisher Scoring iterations: 6

> Anova(mod_Wrack)
Analysis of Deviance Table (Type II tests)

Response: Wrack
               LR Chisq Df Pr(>Chisq)
Year             0.0036  1    0.95203
Treatment        5.8055  1    0.01598 *
Year:Treatment   0.0275  1    0.86820
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
>
> mod_HetZ <- glm(Heterozostera ~ Year * Treatment, data=sgcover,
family=quasipoisson)
> summary(mod_HetZ)

Call:
glm(formula = Heterozostera ~ Year * Treatment, family = quasipoisson,
    data = sgcover)

Deviance Residuals:
   Min      1Q  Median      3Q     Max
-5.237  -3.580  -3.580  -3.283  17.260

Coefficients:
                       Estimate Std. Error t value Pr(>|t|)
(Intercept)            351.5212   840.4509   0.418    0.676
Year                    -0.1732     0.4162  -0.416    0.678
TreatmentImpact       -497.3883  1225.7072  -0.406    0.685
Year:TreatmentImpact     0.2467     0.6069   0.406    0.685

(Dispersion parameter for quasipoisson family taken to be 61.80473)

    Null deviance: 11548  on 339  degrees of freedom
Residual deviance: 11113  on 336  degrees of freedom
AIC: NA

Number of Fisher Scoring iterations: 7

> Anova(mod_HetZ)
Analysis of Deviance Table (Type II tests)

Response: Heterozostera
               LR Chisq Df Pr(>Chisq)
Year             0.0360  1   0.849500
Treatment        6.8444  1   0.008892 **
Year:Treatment   0.1656  1   0.684052
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
>
```

```
> mod_Posid <- glm(Posidonia ~ Year * Treatment, data=sgcover, family=quasipoisson)
> summary(mod_Posid)

Call:
glm(formula = Posidonia ~ Year * Treatment, family = quasipoisson,
    data = sgcover)

Deviance Residuals:
   Min      1Q  Median      3Q     Max
-8.984  -6.417  -2.865   4.236  14.560

Coefficients:
                      Estimate Std. Error t value Pr(>|t|)
(Intercept)          223.45983  261.64529   0.854    0.394
Year                  -0.10885    0.12956  -0.840    0.401
TreatmentImpact      -68.81314  764.01718  -0.090    0.928
Year:TreatmentImpact   0.03354    0.37832   0.089    0.929

(Dispersion parameter for quasipoisson family taken to be 39.05479)

    Null deviance: 15126  on 339  degrees of freedom
Residual deviance: 13401  on 336  degrees of freedom
AIC: NA

Number of Fisher Scoring iterations: 6

> Anova(mod_Posid)
Analysis of Deviance Table (Type II tests)

Response: Posidonia
               LR Chisq Df Pr(>Chisq)
Year              0.744  1     0.3884
Treatment        43.265  1   4.78e-11 ***
Year:Treatment    0.008  1     0.9294
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
>
> mod_TotalGrass <- glmmTMB(Total_Seagrass_pc ~ Year * Treatment + (1|Site),
data=sgcover_select, family=beta_family)
> summary(mod_TotalGrass)
 Family: beta  ( logit )
Formula:          Total_Seagrass_pc ~ Year * Treatment + (1 | Site)
Data: sgcover_select

     AIC      BIC   logLik deviance df.resid
  -487.6   -464.6    249.8   -499.6      334

Random effects:

Conditional model:
 Groups Name        Variance Std.Dev.
 Site   (Intercept) 2.324    1.524
Number of obs: 340, groups:  Site, 35

Overdispersion parameter for beta family (): 4.25

Conditional model:
                      Estimate Std. Error z value Pr(>|z|)
(Intercept)           651.5477   244.6551   2.663  0.00774 **
Year                   -0.3226     0.1212  -2.663  0.00775 **
TreatmentImpact      -710.8349   467.7859  -1.520  0.12862
Year:TreatmentImpact    0.3516     0.2316   1.518  0.12907
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
> Anova(mod_TotalGrass)
```

```
Analysis of Deviance Table (Type II Wald chisquare tests)

Response: Total_Seagrass_pc
                Chisq Df Pr(>Chisq)
Year           4.7973  1     0.0285 *
Treatment      2.1089  1     0.1464
Year:Treatment 2.3036  1     0.1291
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1
```

Appendix B Remote Sensing

B.1 Methodology

B.1.1 Pre-processing

The best Sentinel-2 imagery capture between March and April from each survey year was downloaded for analysis. The selected imagery provided the least water column turbidity and full Survey Area coverage, these were captured on April 12th, 2019 and on April 14th, 2020 for the respective study years. These satellite images were atmospherically corrected using the Sen2cor atmospheric correction module within the Sentinel Application Platform. Images were resampled to 10 m^2 pixel dimensions, de-glinted using the procedures in Hedley *et.al* (2004) and land-masked using band 8 (832 nm central wavelength) reflectance. Sample RBG imagery for the two captures is shown in Figure B-1.

The attenuation of the remotely-sensed benthic reflectance signal through water was improved for classification using depth invariant approaches detailed in Lyzenga (1981). Depth invariant indices (DIIs) were calculated for three band combinations, namely coastal band 1, blue band 2 and green band 3, with central wavelengths of 442, 492, and 559 nm, respectively. Both satellite captures used the same DII band combinations to produce geotiffs for subsequent classification.

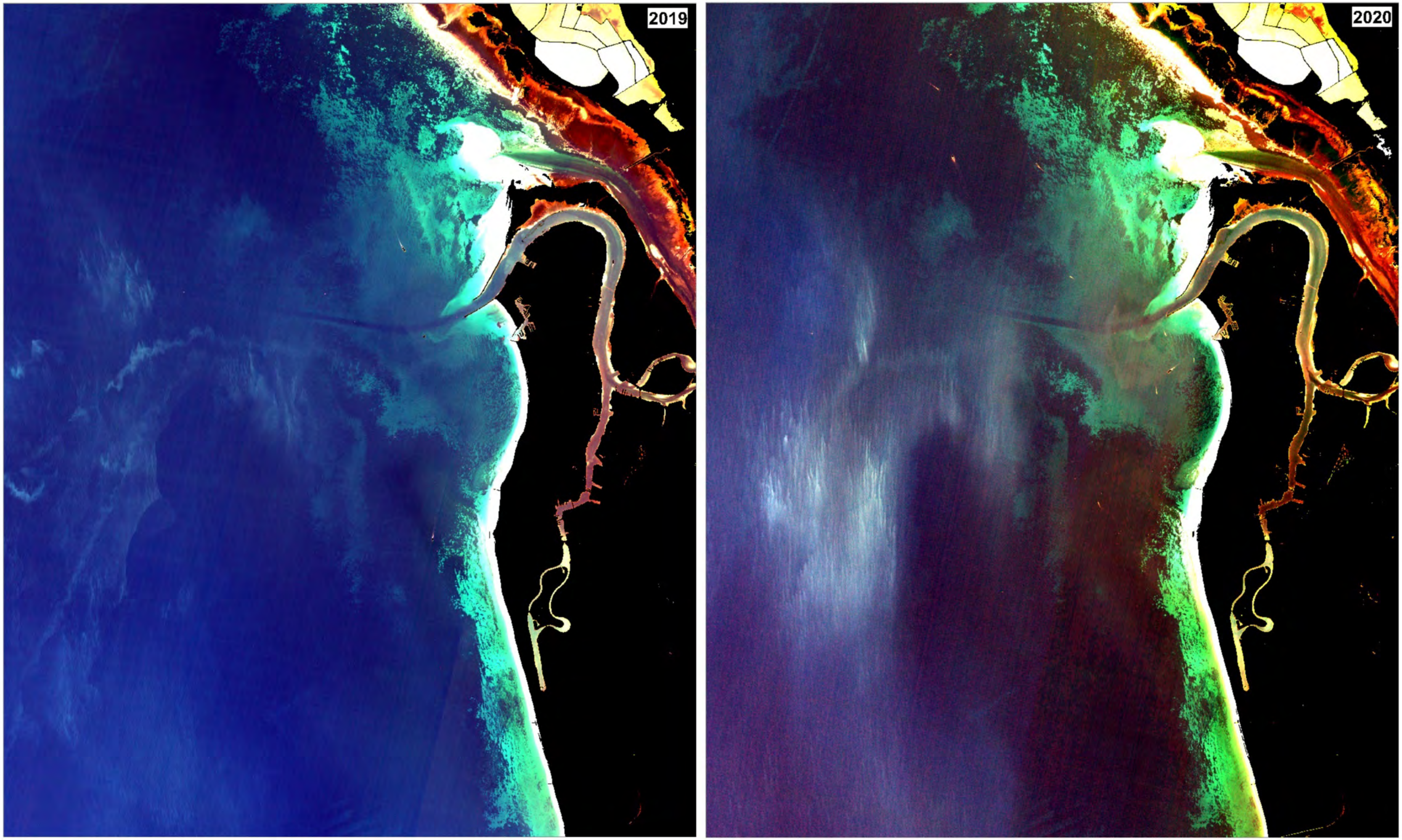
B.1.2 Classification

The corrected DII images were imported into ArcGIS version 10.5 for analysis. The images were classified and trained using ground-truthing data from recovery sites and BACI transect locations. Classifications were performed using Maximum-Likelihood methods applied similarly to each image. The 2020 image included a significant band 1 anomaly between two satellite passes, resulting in different signal strengths in different parts of the image. The 2020 image was classified twice using training datasets applicable to each side of the band-1 anomaly to produce consistent classes on both sides.

Classifications included the following substrates:

- Bare substrate
- Macroalgae – dominant coverage
- Low density – <35% seagrass coverage
- Moderate density – 35–70% seagrass coverage
- High density – >70% seagrass coverage
- *Heterozostera* dominated low density cover – <35% coverage
- *Heterozostera* dominated moderate density cover – 35-70% coverage
- *Heterozostera* dominated high density cover – >70% coverage.

Amphibolis, *Halophila* and *Posidonia* habitats could not be differentiated due to similarities in their spectral signatures, significant mixing within communities at the scale of pixel resolution (10 m), and the relatively low density of *Halophila*. This classified output was filtered and cleaned using filter and cleaning settings common to all classificaitons. An overlay of the BACI and recovery transect cover classes was used to verify the accuracy of habitat classifications.



Title:

De-glinted, Atmospherically Corrected and Land-masked RGB Images from 2019 and 2020

Figure: **B-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.

N

0 2 4km
Approx. scale

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Filepath: I:\B22346_Adelaide Port Channel Widening\DRG\ECO_091_Satellite_Imagery_2019_2020.WOR

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