SEAGRASS DISTRIBUTION WITHIN MORETON BAY MARINE PARK: SEAGRASS RESPONSE TO THE 2022 FLOOD

Results of two years of post-flood surveys and comparison to pre-flood distribution

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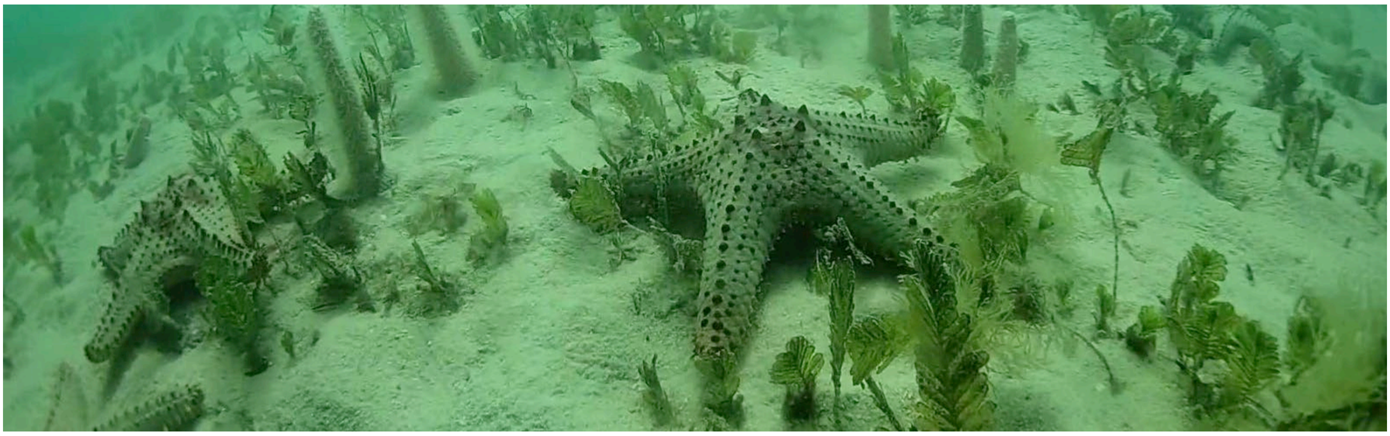

Seagrass Distribution Within Moreton Bay Marine Park: Seagrass Response to the 2022 Flood

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Acknowledgement of Country
Science Under Sail Australia acknowledges the First Nations peoples with Country in and adjoining the Moreton Bay Marine Park and pays respects to elders – past, present and emerging.

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Acknowledgments:
We wish to thank the many Science Under Sail citizen scientists that helped survey the distribution of seagrass throughout Moreton Bay Marine Park, enabling over 8000 sites to be surveyed post-flood (2022/23 & 2023/24) from the Gold Coast Waterways in the south to the northern extent of Pumicestone Passage and Mulgumpin (Moreton Island). We would also like to thank the representatives of the Quandamooka people and the Queensland Parks and Wildlife Service staff who surveyed approximately 900 sites (post-flood) in Eastern, Central and Southern Bay. This project was jointly funded by the Commonwealth and Queensland Government under the Disaster Recovery Funding Arrangement. Healthy Land and Water contributed funding enabling the sampling of additional sites to improve the spatial resolution. Thank you also to Abolghasem Azhdari and Connor Sheidler for their assistance with GIS mapping. Thanks to BMT and JCU/Griffith University for supplying additional pre-flood data. We would like to thank Dr. Abbi Scott for her advice regarding seed sampling protocols.

Executive Summary

Moreton Bay Marine Park (MBMP) extends from the Gold Coast Seaway in the south to Caloundra in the north, incorporating the mouths of many South East Queensland (SEQ) estuaries and extending east of the four sand Islands (Bribie, Moreton, North Stradbroke, and South Stradbroke Islands) to the limit of Queensland State waters. Seagrass meadows are an important ecosystem within the Marine Park, providing a food resource for endangered species (dugong and sea turtles), as well as critical habitat for fish, crabs, and other marine life.

In February/March 2022 a flood event transported large quantities of turbid water to Moreton Bay. As seagrass distribution is strongly influenced by benthic light availability, it was anticipated that the increase in suspended sediment delivered by the flood would impact the distribution and condition of seagrass meadows within MBMP. The Department of Environment, Science and Innovation contracted Science Under Sail Australia to determine the impact of the 2022 flood event on seagrass in MBMP, and the extent of seagrass recovery in the two years following the flood event. Funding was made available from the joint Australian-Queensland government funded Disaster Recovery Funding Arrangements Environmental Recovery Program.

This project surveyed over 5,000 locations within the first year post-flood (2022/23), and over 3,000 locations in the second year post-flood (2023/24). At each location seagrass species and percent cover were recorded, along with sediment type, water depth, and other habitat attributes. This survey data, along with comparable pre-flood data, was used to determine the current spatial distribution of seagrass in MBMP and investigate the impact of the 2022 flood.

Seagrass distribution in MBMP declined from a pre-flood distribution of 327 km^2 to 304km^2 in the first year post-flood (2022/23). In the second year post-flood (2023/24), seagrass distribution further declined to a distribution of 292 km^2. In the 2022/23 survey, seagrass was absent from many areas within which it had previously been observed, with the majority of loss occurring in deep subtidal (> 2m deep) habitats. The intertidal, and shallow subtidal meadows remained relatively stable, with some regional variation in the changes observed post 2022 flood. The average and maximum observed depth of all seagrass species declined post-flood, indicating that seagrass loss is most likely a consequence of reduced benthic light in the aftermath of the February/ March 2022 flood event.

During the second year post-flood (2023/24), seagrass recovered in many locations, despite an overall reduction in areal extent, with the degree and nature of recovery varying between regions. Limited seagrass recovery in the deep subtidal habitats of the Bay suggests elevated turbidity continues to limit seagrass distribution when compared to the pre-flood conditions. Where seagrass growth was not limited by light, the increased nutrient availability post flood may have had a fertilisation effect, with intertidal and shallow subtidal meadows increasing in density in some regions. The post-2022 flood resilience of Moreton Bay's intertidal and shallow subtidal seagrass meadows could provide a source of seed to recolonise deeper areas if water clarity improves in the future.

In the first year post-2022 flood, *Halophila spinulosa* meadows declined significantly, particularly in deep subtidal areas, where *H. spinulosa* dominated meadows were replaced with *H. ovalis*, sparse seagrass or bare sediment. The relatively rare pre-flood, *Syringodium isoetifolium and Cymodocea serrulata* also declined in the first year post-flood. *C. serrulata* and *H. spinulosa* partially recovered in the second year post-flood, however *S. isoetifolium* remained rare. The decline in these species may signify a potential shift towards more transitory coloniser meadows within Moreton Bay's seagrass communities, particularly in the deeper habitats.

Observations of biomass and seed abundance in Moreton Bay support the fact that seagrass meadows in Moreton Bay are resilient and showing signs of recovery following the 2022 flood. High seed abundance in several meadows confirms that seagrass of Moreton Bay is likely to be recovering both sexually and asexually. While the presence of multiple seagrass species at many sites suggests seagrass meadows in Moreton Bay are becoming more diverse in their species composition or undergoing a transition of dominant seagrass species in many locations.

Post-flood surveys undertaken in the first and second year following the 2022 flood reveal declines in seagrass distribution, though they also highlight Moreton Bay seagrass communities' ability to recover following a disturbance. Future management actions in combination with rain and wind patterns will determine if water clarity within the Bay, and hence seagrass distribution, will recover to pre-flood levels.

Recommendations

This report has demonstrated that the 2022 flood not only had an immediate negative impact on Moreton Bay seagrass meadows, but that the mud delivered to the MBMP, by the flood water, has continued to reduce benthic light availability and hence the extent of deep subtidal seagrass meadows has continued to decline during the second year post-flood. The implications of this for management of seagrass meadows within the Moreton Bay Marine Park are two-fold:

1) On-going monitoring of seagrass communities within Moreton Bay annually or biannually in conjunction with more regular water clarity analysis (using either sechii depth or light loggers) is important to inform managers on the changing health of this key habitat within the Marine Park as well as monitoring the most likely environmental variable to limit its distribution.
2) Based on results from the above monitoring program and the predictive modelling it supports, Marine Parks managers will be able to identify any areas within MBMP where specific management actions could enhance seagrass condition. This includes modifications to Marine Park Zones to reduce impacts to active regeneration of seagrass meadows. In areas of MBMP where meadows have been historically lost, but where environmental conditions appear capable of supporting seagrass meadows, active regeneration may be appropriate.

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INTRODUCTION

Study area

Moreton Bay is a large shallow bay on the east coast of Australia, with a catchment that encompasses the majority of the South East Queensland (SEQ) region (Figure 1). Located in the subtropical latitudes, Moreton Bay Marine Park (MBMP) supports a diverse range of interconnected ecosystems, including subtidal and intertidal seagrass meadows, mangroves, soft-sediment benthic invertebrate habitats, as well as reefs supporting hard and soft coral communities. The shallow sheltered nature of Moreton Bay provides optimal conditions for seagrass growth, with seagrass meadows a critical habitat in the larger mosaic of habitats comprising MBMP and the Moreton Bay Ramsar site. The Moreton Bay Ramsar site is a wetland of international significance that covers large parts of MBMP. Seagrass provides an important food source at the base of Moreton Bay's food web (Ebrahim et al. 2014, Connolly et al. 2015, Davis et al. 2015), in addition to a variety of other ecosystem services including: stabilisation of coastal sediments (Potouroglou et al. 2017), carbon storage (Duarte et al. 2005) and nutrient recycling (McGlathery et al. 2007).

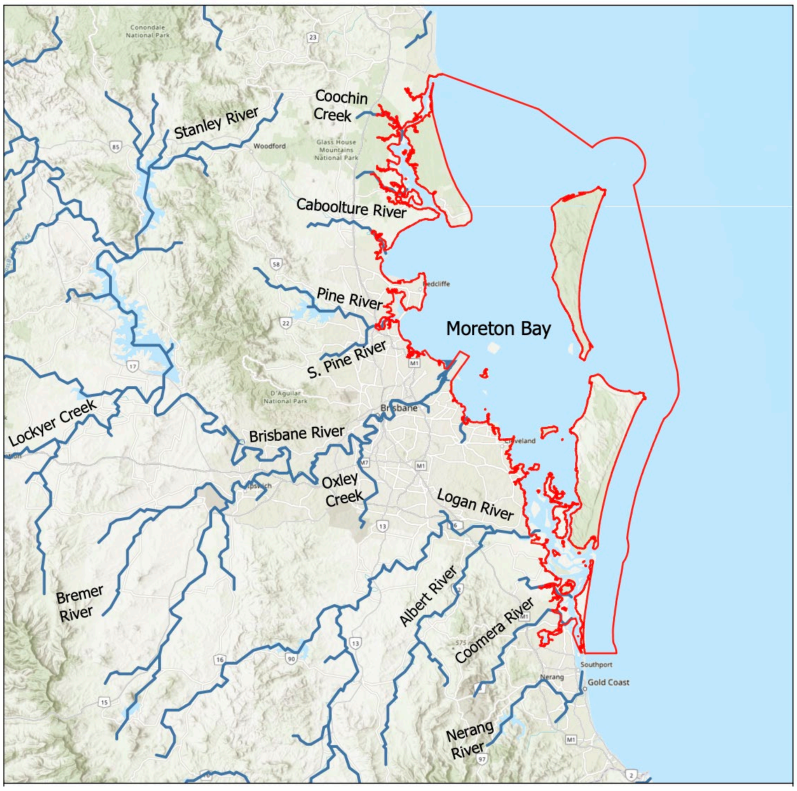


Figure 1. River catchments flowing into the Moreton Bay Marine Park (MBMP boundary outlined in red).

Seagrasses of the Moreton Bay Marine Park

There are seven species of seagrass within MBMP, with interspecies variety in survival/recovery strategies and tolerance to chronic and acute reductions in benthic light (Figure 2) (Longstaff and Dennison 1999, Longstaff 2003, Collier et al. 2016). A review by Kilminster et al. (2015) of seagrass life history strategies and their relevance for management, defines seagrass species on a range from colonising to persistent. Coloniser species such as *Halophila spp*, have low physiological resistance to disturbance but have a rapid ability to recover. Coloniser species have higher shoot turnover, can produce dormant seeds (enabling seed bank formation), and mature to sexual reproduction age quickly (weeks to months); all traits which enable them to quickly recover from a disturbance. In contrast, persistent species have higher physiological resistance to disturbance, yet are much slower to recover. Opportunistic seagrass species sit between these two extremes. Opportunistic species tend to persist in the face of disturbance, while also recovering from seed or new recruits when necessary. MBMP contains both coloniser (C) and opportunistic (O) seagrass species (Figure 2). All species can form enduring seagrass meadows, though coloniser and opportunistic species may also form transitory meadows (Kilminster et al. 2015).

Species	*Halophila decipiens*	*Halophila ovalis*	*Halophila spinulosa*	*Halodule uninervis*	*Zostera muelleri*	*Syringodium isoetifolium*	*Cymodocea serrulata*
Strategy	C	C	C	C/ O	C/ O	O	O
	Coloniser						Opportunistic

Figure 2. Seagrass species found in Moreton Bay Marine Park, with life history strategy as defined by (Kilminster et al. 2015). Seagrass icons from (Collier C, ian.umces.edu/media-library).

Impacts to seagrasses of the Moreton Bay Marine Park

As a marine flowering plant, seagrass requires light to grow and photosynthesise (Dennison 1987, Duarte 1991), and, as such, water clarity is one of the most significant determinants of where seagrass can grow and to what depths. Seagrass is distributed across the Bay in intertidal and subtidal habitats where there is sufficient light for growth (Longstaff 2003) combined with minimal wave or tidal current energy. Due to fluvial inputs to the Bay, a broad water quality gradient exists, from high turbidity and relatively high nutrient concentrations on the western side to low turbidity and relatively low nutrients on the eastern side (Saeck et al. 2019). Although some areas are too turbid for seagrass to grow, seagrass is present across a broad gradient of turbidity in Moreton Bay, with depth range (Abal & Dennison 1996, Longstaff 2003) and diversity of species (Maxwell et al. 2019) increasing eastward.

Water quality in Moreton Bay has suffered considerably since European settlement in the 1820's, with land use changes resulting in increased sediment and nutrient loading rates to the Bay (Saeck et al. 2019). Degraded water quality has led to the loss of seagrass meadows in some parts of the Bay, particularly in southern Deception Bay, Bramble Bay, and the Southern Bay channels (Roelfsema et al. 2009, Maxwell et al. 2019). In response to management actions there has been some improvement in nutrient levels, especially in Western Bay (Saeck et al. 2019), potentially correlated with seagrass recovery proximal to wastewater treatment plant upgrades (i.e. Bramble Bay). In spite of this, the extent of mud/fine sediment distribution in the Bay continues to increase (Saeck et al. 2019). In addition to chronic water quality issues, floods and heavy rainfall events can significantly impact seagrass meadow health. During flood events, extremely large amounts of sediment are discharged to the Bay (Lockington et al. 2017), reducing light penetration to seagrass meadows. The response of seagrass to a disturbance, such as a flood, is affected by the stage of the seasonal cycle at which the disturbance occurs, the spatial extent of the disturbance and the condition of the seagrass meadow pre-disturbance (Kilminster et al. 2015).

During the La Niña phase of the El Niño Southern Oscillation (ENSO), the east coast of Australia experiences increased rainfall, with more frequent extreme high-intensity rain events that have the potential to impact seagrass distribution. In addition to acutely decreasing water clarity in the short term, the large quantities of fine sediment discharged to the Bay during flood events can also contribute to long-term declines in water clarity due to continual resuspension of the newly deposited fine sediments and nutrient enrichment (Longstaff 2003, Saeck et al. 2019, Grinham et al. 2024). Following the 1996 flood of the Caboolture River catchment, approximately 20km^2 of seagrass was lost in southern Deception Bay (Tibbetts et al. 1998). Small patches of *Zostera muelleri* returned to the intertidal zone in 2009, and the coverage of seagrass in this area has continued to expand westward and into deeper water (Maxwell et al. 2019). Following the 2011 Brisbane River flood, localised changes to seagrass distribution and species assemblages were recorded in northern Deception Bay, where subtidal *S. isoetifolium* declined by almost 100% (Hanington et al. 2015). By 2012 there had been some sparse recovery of coloniser species (*H. ovalis*, *Z. muelleri* and *Halodule uninervis*), in areas that had been mono-specific *S. isoetifolium* beds pre-flood, and bare sediment immediately post flood (Hanington et al. 2015). In other areas of the Bay, although seagrass loss was reported immediately after the flood, there was evidence of recovery 12 months later (Babcock et al. 2019). However, at this time there was limited mapping of seagrass distribution pre- and post-flood, thus Bay-wide changes to seagrass distribution were unable to be determined.

Seagrasses within Moreton Bay employ different modes of resilience to resist or recover from environmental or stochastic pressures (e.g. floods). Seagrass resilience, the ability of seagrass ecosystems to withstand and recover from disturbances, is closely linked to factors like biomass and seed counts. There are three critical attributes that affect the resilience of seagrasses: (1) seagrass life history, (2) meadow form, and (3) physical habitat. The combination of these attributes should inform the monitoring and policy required for effective management (Kilminster et al. 2015).

Attribute 1: Seagrass life history

Life-history traits of seagrasses, such as shoot (or ramet) turnover, genet persistence and sexual reproduction characteristics, enable a functional classification at the species level, at which it varies substantially. Broadly, we categorise species as having either persistent or colonising traits based on their ability to resist or recover. Colonising species have low physiological resistance (small biomass) and rapid ability to recover (high seed production)(e.g. *Halophila* *spp*.), whilst persistent species are slow to recover (less energy invested in seed production) but have high physiological resistance (high biomass) (e.g. *Cymodocea serrulata*). Species with a mixture of these traits are categorised as opportunistic (Kilminster et al. 2015) (Figure 3).

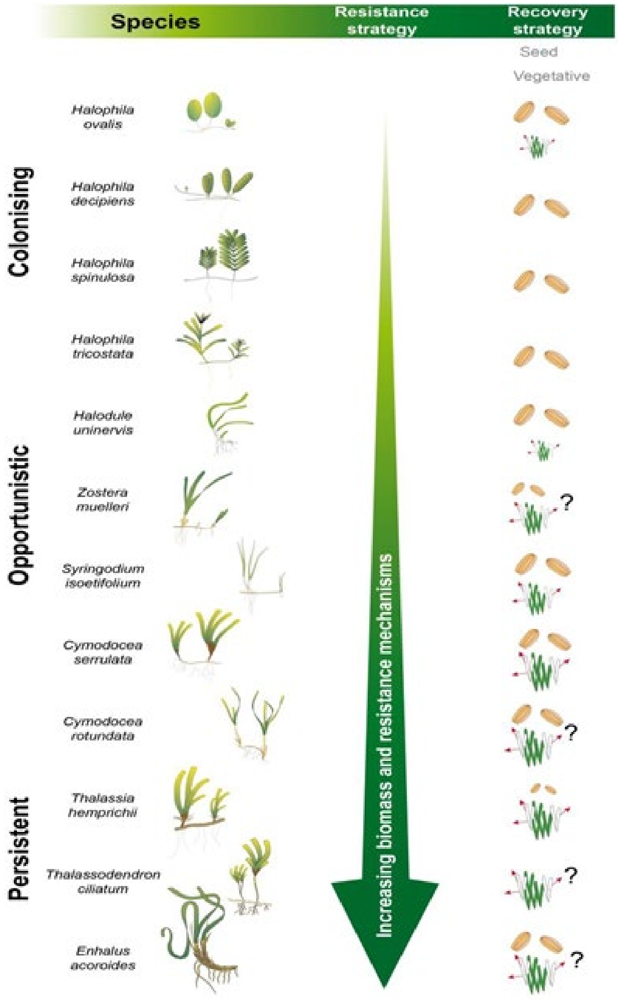


Figure 3. Dominant traits of seagrass species that occur in Queensland and their life history traits (colonising, opportunistic and persistent). Moreton Bay only has colonising or opportunistic species and does not have any of the tropical persistent species (Udy *et. al*. 2019).

Attribute 2: Meadow form

Seagrasses grow in natural units we refer to as 'meadows', which can be defined by the mix of species that occur within them, and variance in density. The functional definition of a meadow is the area in which seagrass can grow continuously, which shares the same environmental *drivers*, and responds to environmental *pressures* in an integrated way (Figure 4). For example, an area exposed to the same hydrological forces with similar water quality and benthic light availability. Thus, although spatially adjacent, a shallow and deep seagrass meadow can have very different characteristics due to different benthic light availability.

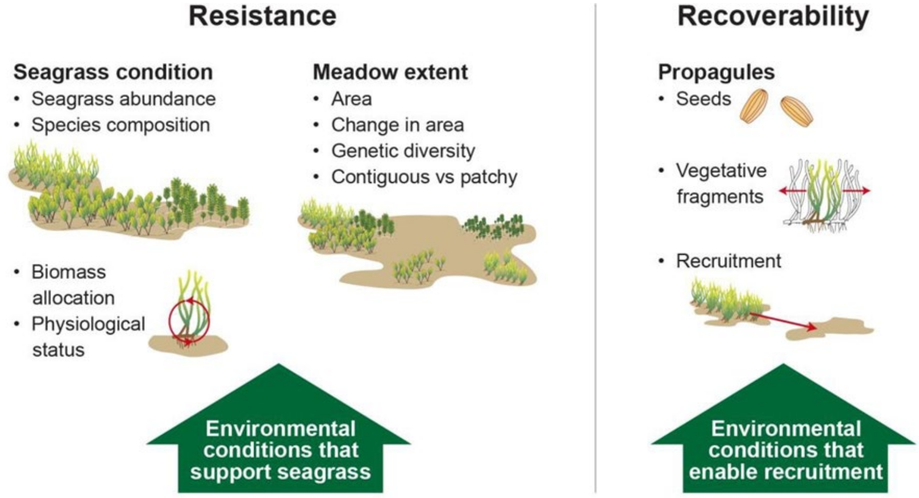


Figure 4. Resistance and recoverability attributes of a seagrass meadow. (Udy *et. al*. 2019).

Attribute 3: Habitat type
The habitat in which seagrasses grow is the final attribute that needs to be considered to understand factors that aid seagrass resilience. Each habitat is impacted by different pressures that influence the physical environment (for example, quantity and variability of light, nutrients, substrate type, freshwater input and hydrodynamic conditions). Moreton Bay has 3 of the 12 seagrass habitat types as defined for the Great Barrier Reef (Udy *et. al*., 2019) (Figure 5). However, the transition from Shallow Subtidal to Deep Subtidal occurs at a shallower depth in Moreton Bay than on the GBR due to higher turbidity.

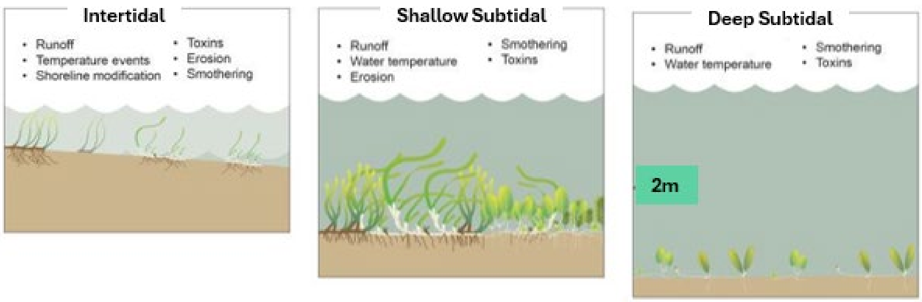


Figure 5. Three seagrass habitat types in Moreton Bay. Small text indicates dominant pressures in each habitat (modified from Udy *et. al*. 2019).

Measuring Resilience
When considering seagrass resilience, there are 2 important factors (Figure 6):

1. A seagrass's ability to resist an environmental pressure, (i.e. not decline or lose percentage cover or biomass)
2. A seagrass's ability to recover/return after a decline or total loss.

Seagrass biomass refers to the amount of seagrass present at a site or in a meadow and is normally represented as the grams Dry Weight (gDW). Higher biomass can indicate a healthier seagrass community with greater resistance to disturbances such as storms, pollution or reduced benthic light. An important factor in predicting seagrasses' resistance to a flood or any environmental pressures likely to reduce light availability, is the above: below-ground biomass ratio. High below-ground biomass can help seagrasses survive a short-term loss of light (e.g. flood), as the rhizomes release stored carbohydrates to maintain metabolic processes until there is sufficient light to support photosynthesis. If an area experiences chronic light limitation, you would expect a larger above: below-ground ratio as the seagrass invests more biomass and energy in leaves to capture the limited light. As mentioned in the above section (Attribute 1: Seagrass Life History), different species of seagrass have very different healthy biomasses depending on their life history. Comparing seagrass total biomass and the above: below-ground ratio within a species can be informative regarding a seagrass meadow's resilience.

Seed counts represent the past reproductive history of seagrass populations. Seeds play a crucial role in the natural regeneration of seagrass meadows, allowing them to recover from disturbances by colonizing new areas or replenishing damaged ones. Higher seed counts suggest a greater potential for seagrass recovery and expansion. However, it is not appropriate to compare seed counts between species as some species have very small seeds that are difficult to identify, and the life history of different species will vary the relative importance of recovery from a seed banks.

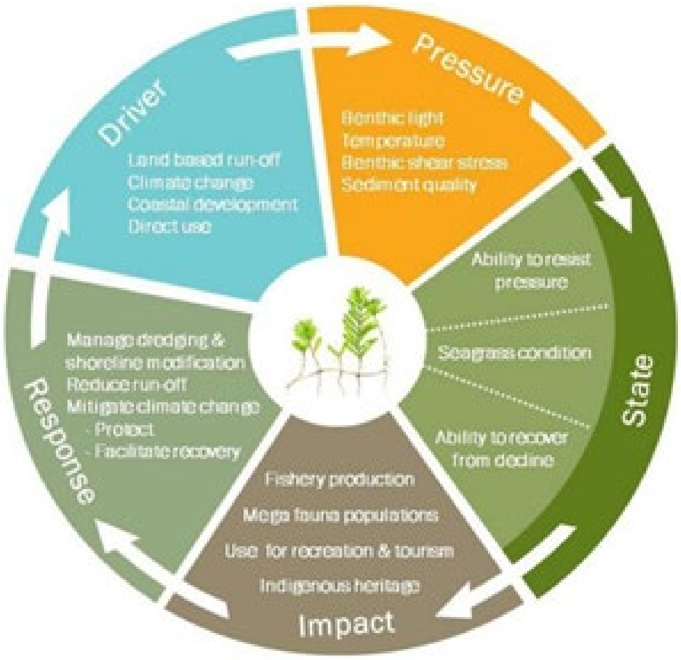


Figure 6. Important factors to consider in relation to seagrass resilience. (Udy *et. al.* 2019)

Flood events in 2022

In spring and summer 2021-2022, a La Niña phase of ENSO combined with the rainfall-promoting phases of the Indian Ocean Dipole (negative IOD) and the Southern Annual Mode (positive SAM), resulted in extremely high rainfall in SEQ. Higher than normal summer rainfalls culminated in a SEQ-wide major flooding event in late February/ early March. Although the 2022 flood of the Brisbane River was less severe than in 2011, in 2022, rainfall spread across the region, with all catchments in SEQ experiencing major flooding; many reaching record discharge volumes during the peak (Table 1). The widespread severity of the 2022 event discharged large volumes of sediment, with fine sediment deposited across 98% of the Bay (Grinham et. al., 2024). The flood plume extended from the

Brisbane River across to Eastern Bay (Grinham et al. 2024) (Figure 7), with a second plume originating from the Logan and Coomera Rivers filling Southern Bay and Broadwater waterways and extending up to southern Eastern Bay. The heavy rainfall also increased turbidity in areas adjacent to smaller catchments, with Coochin Creek (Pumicestone Passage) experiencing a record high discharge. Some catchments, particularly the Lockyer Valley, experienced a second major flood event in May, further extending the duration of low water clarity in the Bay.

Table 1. Comparison of daily mean discharge at February/ March 2022 flood peak to the median daily mean discharge, and previous floods at ten stations distributed within the Moreton Bay Marine Park catchment. Data from http://www.bom.gov.au/waterdata/

Station	Outlet to MBMP	Median (cumecs)	February 2022 peak (cumecs)	Percent above median discharge	2022 February Event percentile occurence bracket	Comparison to Earlier Floods	Records Since
Coochin Creek @Mawsons Road	Pumicestone Passage	0.26	164.7	64,504%	> 99.99 Percentile	**Record discharge**	2006
Caboolture River @Upper Caboolture	Deception Bay	0.09	253.8	269,873%	> 99.99 Percentile	**2nd highest flood peak after 1974**	1965
South Pine River @Drapers Crossing	Bramble Bay	0.09	662.7	720,257%	> 99.99 Percentile	**2nd highest flood peak after 1974**	1965
Lockyer Creek @Rifle Range Road	Bramble Bay via Brisbane River	0.04	777.0	1,992,200%	> 99.90 Percentile	**Lower than 1996, 2011, 2013 Peak**	1988
Bremer River @Walloon	Bramble Bay via Brisbane River	0.03	1019.0	3,396,500%	> 99.99 Percentile	**Record discharge**	1961
Brisbane River @Jindalee	Bramble Bay	18.49	5760.9	31,055%	> 99.90 Percentile	**2nd highest flood peak after 2011**	1996
Oxley Creek @New Beith	Bramble Bay via Brisbane River	0.00	169.5	169,529,900%	> 99.99 Percentile	**Record discharge**	1976
Logan River @Yarrahappini	Southern Moreton Bay	1.50	3419.4	228,467%	> 99.99 Percentile	**2nd highest flood peak after 1974**	1969
Albert River @Bromfleet	Southern Moreton Bay	0.83	1265.6	151,833%	> 99.99 Percentile	**2nd highest flood peak after 1974**	1927
Nerang River @Glenhurst	Gold Coast	0.15	216.5	148,177%	> 99.90 Percentile	**Lower than 1996, 1998 & 2017 Peak**	1986

Seagrass mapping in the Moreton Bay Marine Park

Attempts to map the seagrass of Moreton Bay have been conducted periodically since 1975, however a lack of consistent methodology has made it difficult to properly evaluate changes in temporal distribution and percent cover (Roelfsema et al. 2013). Science Under Sail Australia (SUSA) has collaborated with Healthy Land and Water since 2015 to conduct seagrass surveys in Moreton Bay, using citizen scientists supervised by experienced researchers. From 2015-2019 a different portion of the Bay was surveyed each year. During the 2020/21 spring and summer approximately 2000 locations were surveyed across Moreton Bay, in water depths and regions where seagrass could theoretically occur. James Cook University survey data from 2015 (York et al. 2016) and BMT survey data from 2018-21 (BMT 2018, BMT 2019, BMT 2020, BMT 2021) have been used to supplement SUSA pre-flood coverage in Southern Bay, Broadwater and Pumicestone Passage. The combination of 2015 to 2021 data has been used in this report to provide a pre-flood extent and abundance estimate of seagrass in the Bay.

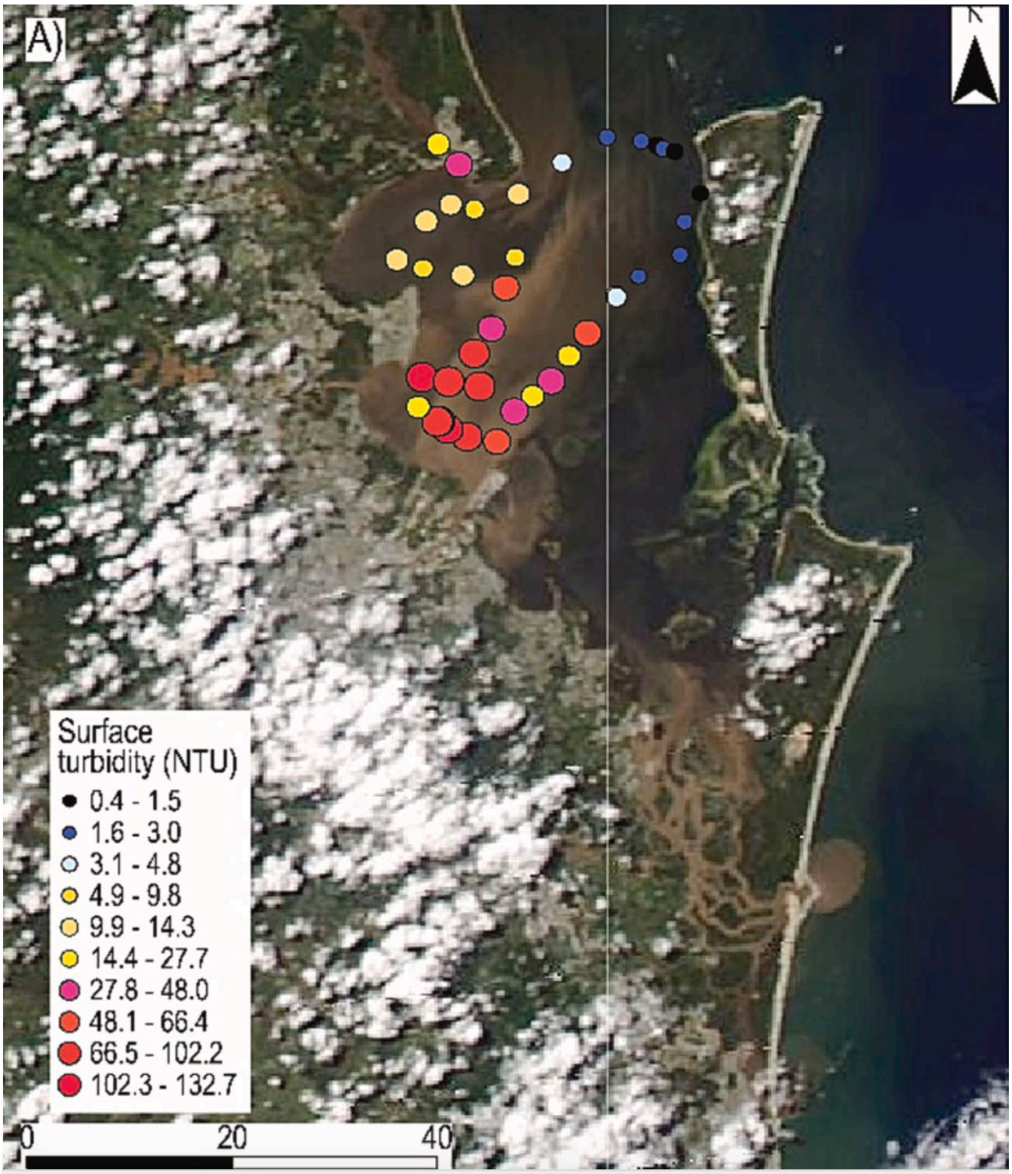


Figure 7. Surface turbidity measurements taken on 2nd March 2022 by Grinham et al. 2024, following the flooding event, overlayed on MODIS satellite image taken the same day. Figure from Grinham et. al. 2024.

In the 2022/23 spring and summer seagrass growth season (8 – 11 months after the 2022 flood), SUSA conducted an extensive survey of benthic habitats in Moreton Bay, revisiting areas that had been surveyed pre-flood, as well as surveying areas of MBMP that had not previously been surveyed or had been surveyed at a low spatial resolution (predominantly deeper or off-shore locations). To determine the extent of seagrass recovery, a follow-up survey was completed in 2023/24 spring-summer, targeting areas where seagrass was likely to occur. The information gathered from these surveys has been compared with pre-flood data to determine how the distribution and composition of seagrass meadows in MBMP has been affected by the February/March 2022 flood. The survey does not capture the loss of seagrass directly following the flood, but instead, indicates the extent of seagrass survival/recovery by the 2022/23 austral summer, and the current distribution of seagrass in the 2023/24 austral summer, 2 years after the flood.

METHODS

Site Description

Seagrass surveys in 2023/24 occurred within the inshore areas of MBMP. The inshore waters of MBMP have an average depth of 6.8m (Dennison & Abal 1999) and stretch from Pumicestone Passage (Caloundra) in the north, to the Gold Coast seaway in the south (approx. 125km). It includes many different habitat types, from wide open bay areas with extensive sandbars and deeper water, to narrow and often shallow waterways of the northern Broadwater and Southern Bay Islands (Figure 1).

The Moreton Bay Catchment covers 21,220km^2, with four main river catchments discharging into the Bay: the Brisbane River, Logan River, Pine River, and Caboolture River (Figure 1). Approximately one-quarter of the catchment remains in original vegetation, with the rest converted to a mix of urban, pastoral, agricultural and forested land uses (Bunn et al. 2007). The proportion of still-vegetated riparian zones is even lower, with unvegetated riparian edges contributing proportionally more suspended sediment to stream discharge (Olley et al. 2015). The changes to land use have also resulted in elevated run-off velocities, greatly increasing erosion and thus sediment transported to the Bay during floods (Bunn et al. 2007). These factors contribute toward recent flood events transporting significantly higher sediment loads to MBMP than historic events in the 1800s and early 1900s (Saeck et al. 2019).

Due to fluvial inputs to the Bay, the western side is dominated by fine, muddy sediment of terrigenous origin, with the proportion of sand increasing towards the eastern side of the Bay. In addition to the wide northern mouth of the Bay (North Passage), openings also occur between Moreton and North Stradbroke Islands (South Passage), North and South Stradbroke Islands (Jumpinpin), and South Stradbroke Island and the Mainland (Gold Coast seaway). These allow clear, oceanic water to enter during flood tides, and remove turbid, catchment-derived run-off during ebb tides. Water clarity and the proportion of sand in the sediment increases closer to these oceanic openings. The extent of muddy sediment has been encroaching further east in Moreton Bay, with double the area now covered in muddy sediments in comparison to 1970 (Lockington et al. 2017). Mud is now the dominant sediment type in the Bay (Lockington et al. 2017; Grinham et al. 2024).

To reflect the gradients in water quality across the Bay, the Healthy Land and Water Report Card splits MBMP into five Bay regions: Eastern Bay, Central Bay, Western Bay, Southern Bay and Broadwater, and the Pumicestone Passage estuarine region (HLW 2023) (Figure 8). Temporal analysis in this report utilises the same regional divisions.

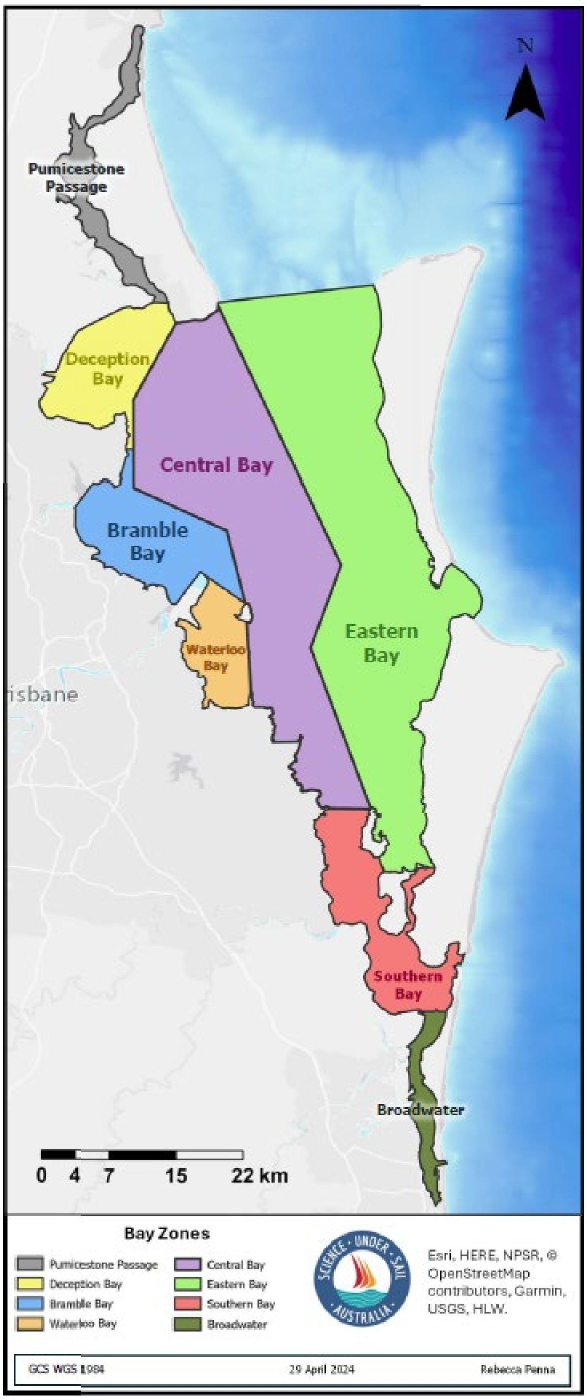


Figure 8. Regions used in Healthy Land and Water Report Card, and for temporal analysis in this report.

Site Selection

This study combines survey data collected by Science Under Sail Australia (SUSA) from 2015 till 2024 (Figure 9). It also utilises survey data collected by James Cook University in Southern Bay and Broadwater in 2015 (York et al. 2016), and by BMT in northern Pumicestone Passage (BMT 2018 - 2021) to supplement SUSA pre-flood sampling coverage. Polygons of seagrass distribution in 2002 (Pumicestone Passage) and 2004, supplied by the Environmental Health Monitoring Program coordinated by Healthy Land and Water, have been compared to the current spatial distribution for Pumicestone Passage.

Pre-flood SUSA surveying (2015 – 2021) utilised a random sampling approach, within the areas of the Bay where seagrass was likely to occur based on depth and water clarity. Site selection was guided by a random site generating program in 'R' that utilised depth (Beaman 30m grid) to create a spread of sites over representative habitats/depth zones. Sampling typically occurred beyond the maximum depth where seagrass was observed in any region, to ensure seagrass extinction depth was captured.

Between August 2022 and January 2023, all regions of the Bay surveyed pre-flood by SUSA were revisited (from the southern tip of Bribie Island south to the Logan River). In addition, other areas of MBMP were added to elucidate a more comprehensive and contemporary understanding of where seagrass habitats occur across the Marine Park (Figure 5). A total of 5367 sites were surveyed from August 2022 till February 2023(first year post-flood) as part of a jointly funded post-flood seagrass and habitat assessment, with the majority of field work undertaken in December 2022 and January 2023. In the second year post-flood (2023/24 austral summer) a follow-up survey sampled 3,414 sites between December 2023 and March 2024. The 2023/24 survey focused on resampling sites proximal to locations that had been sampled previously. Post-flood surveys conducted by SUSA were jointly funded by Queensland Parks and Wildlife Service and Healthy Land and Water.

Sites were visited as close as logistically possible, depending on wind and currents (within 50m of the GPS location). Once in the vicinity of a site, additional sites were occasionally added in areas where water depths were less than 5m to improve the spatial resolution of sampling along depth gradients where changes in seagrass cover were expected to occur. These additional sites were selected by starting from the predetermined site, traveling along a line perpendicular to the shore and sampling every 0.5m to 1m change in depth. These additional sites increased depth threshold precision for each seagrass species as well as maximum seagrass depth in each region (used to map seagrass distribution).

Mapping of seagrass distribution within MBMP has utilised all survey data collected by SUSA between 2015 and 2024, subject to various rules and depending on the time frame of map being produced (see Mapping below). However, to reduce sampling bias in temporal statistical analysis (due to variation in sampling effort between years), the survey data was sub-sampled to create a 'temporal study area'. This 'temporal study area' was defined by drawing a 500m buffer around all survey data within MBMP that was sampled between 2015 and 2021 (pre-flood). All post-flood survey sites from 2022/23 and 2023/24 that were located within this sample area were included in the temporal analysis, but those that were outside were excluded (Figure 9). The total number of sites that contributed to the temporal analysis for each region are displayed in Table 2.

Table 2. Number of observations from each region used in the temporal analysis for each sampling period. Observations that included reliable depth measurements are underlined.

		Eastern Bay	Central Bay	Deception Bay	Bramble Bay	Waterloo Bay	Southern Bay	Broad-water	Pumicestone Passage
Pre-2022 Flood	**2015-16**	839	323	54	4	92	<u>653</u>	<u>536</u>	-
	2016-17	2124	166	219	-	44	-	-	26
	2017-18	654	7	-	-	14	-	-	-
	2018-19	165	3	417	209	132	33	-	6
	2019-20	2611	3	117	-	132	33	-	397
	2020-21	<u>1693</u>	<u>31</u>	<u>242</u>	-	<u>192</u>	<u>80</u>	-	<u>29</u>
Post-2022 Flood	**2022-23**	<u>1531</u>	<u>276</u>	<u>438</u>	<u>75</u>	<u>552</u>	<u>490</u>	<u>352</u>	<u>87</u>
	2023-24	<u>1429</u>	<u>149</u>	<u>224</u>	<u>81</u>	<u>280</u>	<u>305</u>	<u>304</u>	<u>49</u>

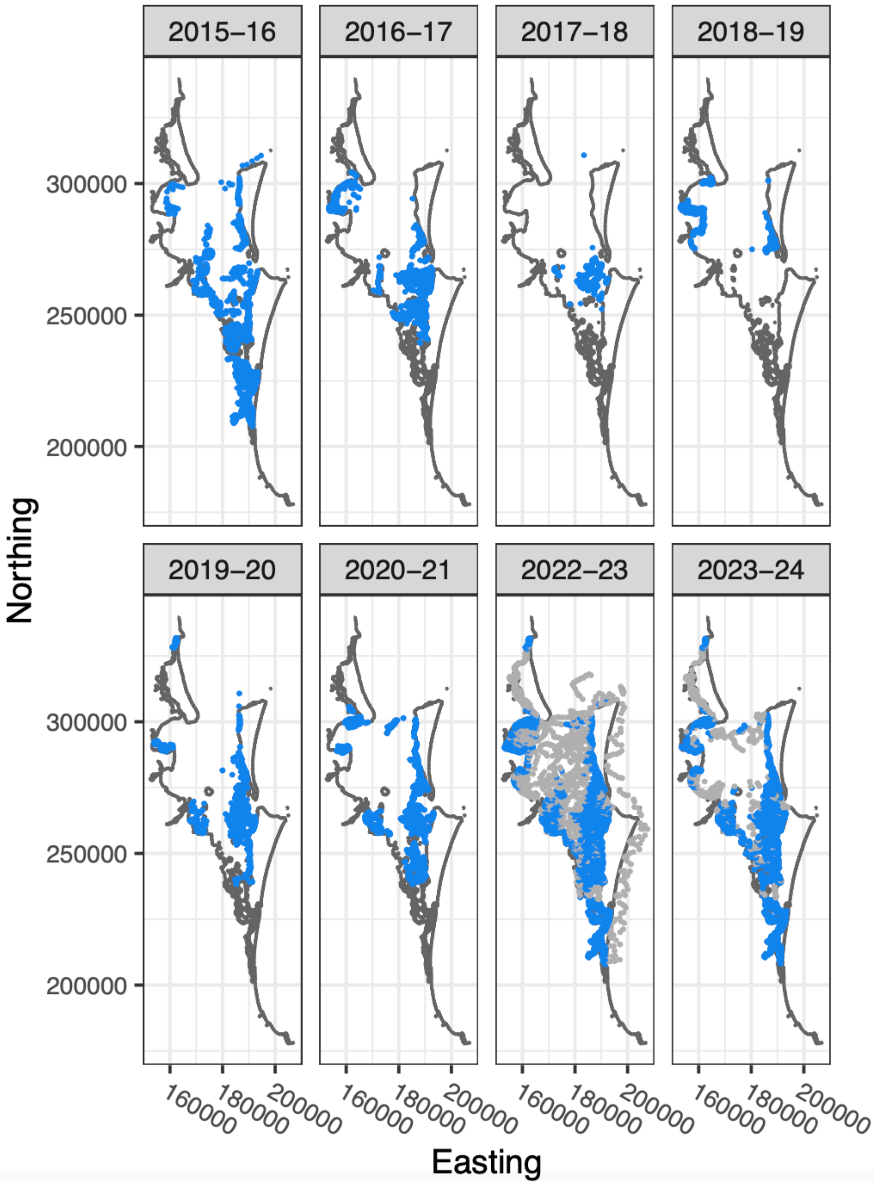


Figure 9. Sites within MBMP sampled each season since 2015 (SUSA, JCU and BMT data combined). Blue sites are within the defined temporal sample area and used for all temporal analysis. Grey sites are not used for temporal analysis but were used when mapping seagrass distribution within MBMP.

Seagrass Monitoring and Habitat Classification

Due to high turbidity levels throughout many of the inshore areas of MBMP, a quadrat approach to estimate seagrass abundance was not chosen as the preferred method. To ensure observations of benthic habitats could be made at the maximum number of locations, we used an ultra-wide-angle lens on an obliquely angled drop camera or a human diver with a camera to undertake the surveys. This enabled the benthic habitats to be surveyed and positive identification of seagrass to occur (if present) in very low water clarity (5-10cm). Hence for this survey, percent cover is based on visual estimations derived from an oblique view, with photos retained and consulted for QA/QC and calibration.

At each site, depth, time, and position (GPS) were recorded in addition to the percent cover of seagrass species, sediment type and other relevant habitat information (Field data sheet, Appendix 3). The attributes measured are a subset of those used in the Queensland Intertidal and Subtidal Ecosystem Classification Scheme (DEHP 2017). Depths were later corrected to account for tidal cycle variations, so that depths used in the analysis are relative to chart datum (Lowest Astronomical Tide). *H. ovalis* and *Halophila decipiens* are particularly difficult to distinguish between in the field (Vy et al. 2014), and thus, for this study, both species were recorded as *H. ovalis*. *Z. muelleri* and *H. uninervis* were also combined into one composite species for mapping and analysis due to the similarity of their above ground appearance when using a drop camera alone. The merging of visually similar seagrass species is also used in rapid visual assessment of seagrass on the Great Barrier Reef, due to the need for detailed inspection and destructive sampling to separate these species (Rasheed, Pers. Com.). All data from 2022/23 and 2023/24 has been QA/QC'd and entered into a data management cloud storage platform with associated photos. As SUSA is a citizen science program, much of the surveying was carried out by volunteers. Volunteers are trained in seagrass identification, under supervision from staff with extensive seagrass knowledge and fieldwork experience. All data has been post-fieldwork verified and checked using video footage and photos collected at the time of sampling.

Figure 10. Methods for surveying seagrass distribution: a & b) drop camera with live stream to boat, c) snorkeller GoPro recording the benthic habitat.

Two different sampling techniques were employed depending on the depth of a site (Figure 10). Please note: these depth categories are based on the survey methodology employed, and do not relate to analysis depth categories.

Shallow/ Intertidal: At shallow and intertidal sites, a snorkeler with a GoPro dove to the bottom and swam approximately 3m along the sea floor. A hand grab sample was taken back to the surface, to confirm seagrass species. The snorkeler would visually estimate the percent cover of species and make a tactile observation of the substrate. Three screen grabs were later taken from each GoPro video and used to validate the snorkeler's observations (Figure 11). Where visibility was < 5cm, a tactile analysis of sediment and the presence of seagrass was utilised. Generally, where visibility was this poor, only mud was present, however if seagrass was felt, a sample was taken and used to determine species, and a sweeping feel of the area, with a bare hand, was used to estimate percent cover. In some instances, intertidal sites were surveyed when exposed at low tide, however given the extremely soft and muddy substrate in many locations, this was not always a feasible option.

Deep: In habitats deeper than free diving range (approx. > 3m), a boat-based drop camera was used to quantify the benthic habitat. The drop camera feeds a live video to an onboard computer and display screen. Five random stills were captured from the video at each site and these, as well as the 30 second to 1 min videos, were used to determine benthic composition (e.g. species of seagrass and percent cover, and sediment type; Figure 12). A value of 1% was assigned if, during the video, a single seagrass plant or other organism was observed, to indicate a presence rather than being an accurate estimate of cover. Covers of 2% - 100% required that the sum seagrass cover observed in all/most frames represented an average of the % cover observed during the entire video.

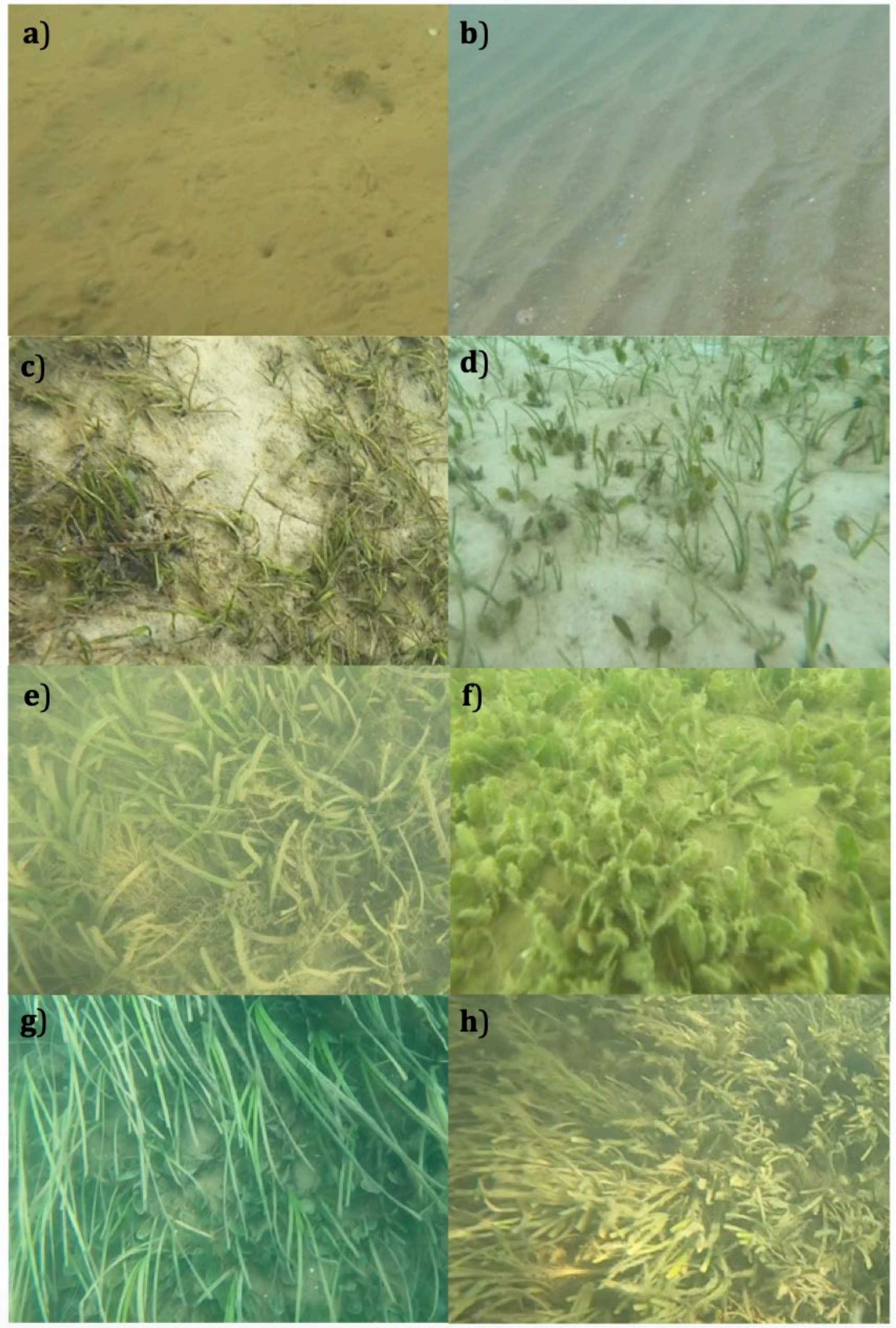


Figure 11. Example of benthic habitat screenshots from GoPro video: a) mud with bioturbation b) sand c) 40% Z. Muelleri and 2% H. Ovalis d) 20% H. ovalis and 10% Z. Muelleri e) 70% C. serrulata f) 75% H. ovalis g) 45% Z. Muelleri and 40% H. ovalis h) 95% Z. Muelleri

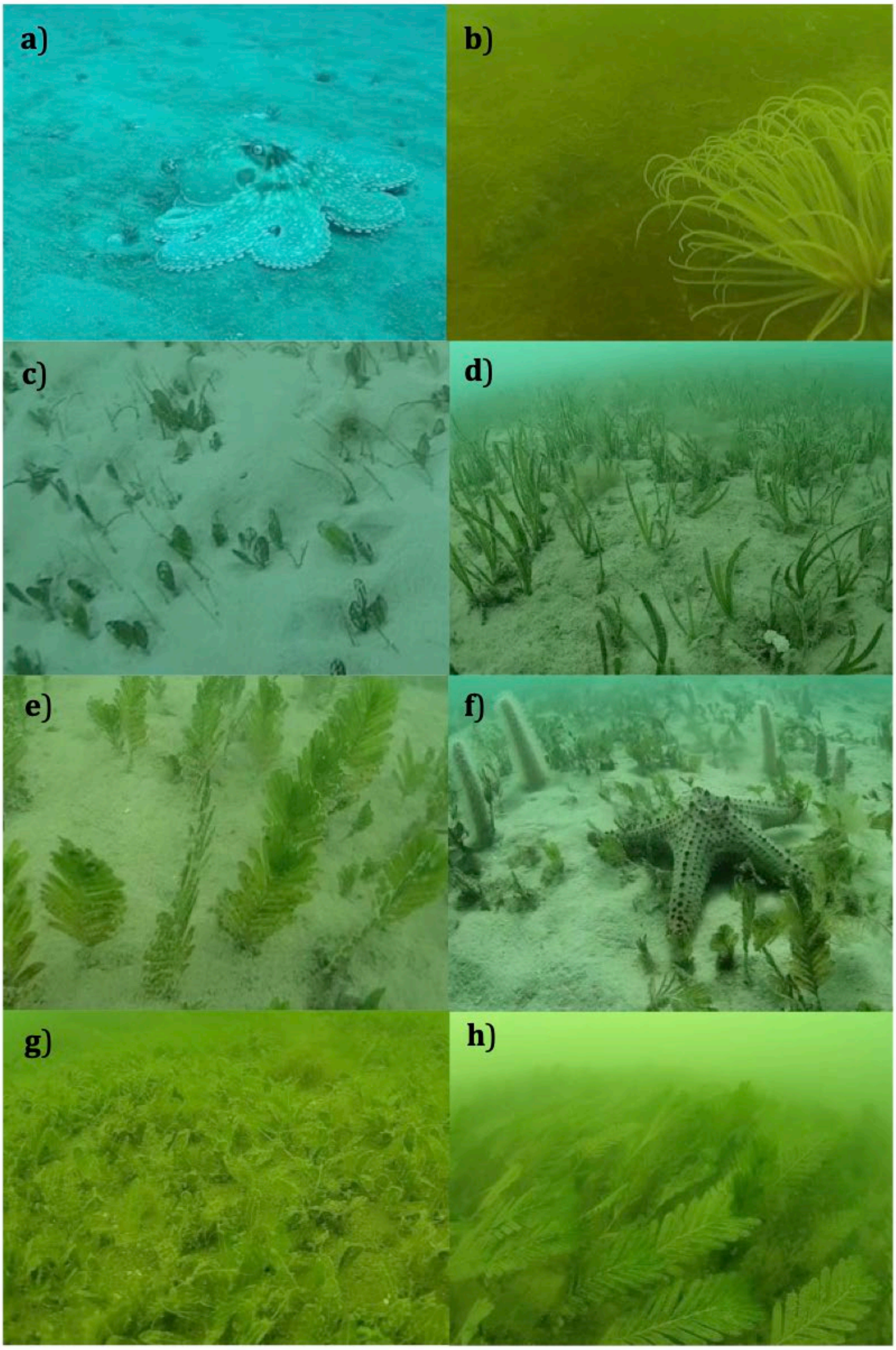


Figure 12. Example of benthic habitat screen shots obtained from the drop camera: a) sand with octopus b) mud with anemone c) 20% *H. ovalis* & 5% *H. uninervis* d) 25% *Z. muelleri* e) 30% *H. spinulosa* f) 20% *H. spinulosa* with sea star and octocoral g) 80% *H. ovalis* h) 80% *H. spinulosa.*

Determination of Meadow Types – cluster analysis

The seagrass community compositions within MBMP are highly variable, ranging from a single species to a five-species presence at a given survey location. Seagrass density ranges from a few leaves observed during a survey (1% cover) to 100% cover. To simplify this complexity and enable comparisons of seagrass meadow distribution and condition over time, observed seagrass coverage at a site has been assigned to one of eight different seagrass meadow types. As a site will often have multiple seagrass species present, grouping sites into meadow types enabled categorisation of the complexity that occurs within seagrass meadows of MBMP, enabling the impact of the 2022 flood to be better defined and interpreted.

To distinguish the predominant seagrass meadow types in MBMP, we used the `pam`, *"partitioning around medoids"* function of the R core package `cluster` (Van der Laan et al. 2003) to look for distinguishable meadow types within all pre-flood data where seagrass was present. The analysis identified six distinct meadow types using seagrass species present and percent cover of each species. The program generalises the notion of a median to multivariate cases, defining the medoid of a meadow type as the observation within a group for which the sum of distances to all other members of that meadow type is minimum. Meadow types were selected to provide the smallest possible *Hellinger distance* within meadow types. Silhouette widths were used to calculate the integrity of the meadow types and ensure they represented real structure in the data (Batool & Hennig 2021). As *S. isoetifolium* was such a rare species in MBMP it did not form a cluster of its own, and thus, to enable analysis of distribution of *S. isoetifolium* dominated meadows, all sites where *S. isoetifolium* was the dominant species (60% or more of the relative seagrass cover) were forced into an *SI* cluster. All sites with no seagrass present were assigned to an Absent cluster, resulting in eight distinct medoids of meadow types identified within the Bay (Table 3).

Once data had been grouped, meadow type names were assigned based on the seagrass species which dominated that cluster. When there were multiple meadow types with the same dominant species, they were separated based on a prefix determined by the average proportional cover of total seagrass within each meadow type:

0% ≥ Very Sparse < 5%,
5% ≥ Sparse <20%,
20% ≥ Moderate <60%,
60% ≥ Dense ≤ 100%

Using the medoids identified with the pre-flood data, all data was then assigned to the meadow type to which it was most similar, as determined by minimising the *Hellinger distance*. In some instances, sites with very sparse seagrass (< 5% cover) were assigned to the *Absent* cluster, as this was their most similar medoid. Although similar statistically, bare substrate and very sparse seagrass are quite different biologically. Therefore, any sites that had sparse seagrass present but had clustered to Absent were split post-processing into a *Very Sparse* meadow type, resulting in a total of nine meadow types used in analysis and mapping

(Table 4, 5). In this report, *SI, CS, HS, HO* and *ZMHU* initials are used when referring to meadow types, while the scientific name of seagrass species is used when referencing the exact species.

Table 3. Medoids of each meadow type present in Moreton Bay Marine Park, as determined by cluster analysis on all pre-flood data.

Meadow Type	*Halophila spinulosa*	*Halophila ovalis*	*Zostera muelleri/ Halodule uninervis*	*Cymodocea serrulata*	*Syringodium isoetifolium*	*Bare*
Absent	0.0	0.0	0.0	0.0	0.0	100.0
SI	0.0	0.0	0.0	0.0	40.0	60.0
CS	0.0	0.0	0.0	70.0	0.0	30.0
Moderate HS	40.0	0.0	0.0	0.0	0.0	60.0
Sparse HS	9.9	0.1	0.0	0.0	0.0	90.0
HO	0.0	5.0	0.0	0.0	0.0	95.0
Moderate ZMHU with HO	0.0	2.0	25.0	0.0	0.0	73.0
Dense ZMHU	0.0	0.7	69.3	0.0	0.0	30.0

Table 4. Average cover of sites that clustered into each meadow type pre-flood.

Meadow Type	*Halophila spinulosa*	*Halophila ovalis*	*Zostera muelleri/ Halodule uninervis*	*Cymodocea serrulata*	*Syringodium isoetifolium*	*Bare*
Absent	0.0	0.0	0.0	0.0	0.0	100.0
Very Sparse	0.4	0.2	1.6	1.1	0.2	96.5
SI	1.1	3.5	3.0	1.0	40.5	50.9
CS	0.7	0.6	1.1	67.0	0.2	30.4
Moderate HS	43.87	1.9	1.3	0.1	0.2	84.9
Sparse HS	12.5	2.5	0.5	0.2	0.0	84.9
HO	0.6	11.5	1.0	0.1	0.1	86.8
Moderate ZMHU with HO	1.6	5.5	24.0	0.2	0.3	68.5
Dense ZMHU	2.0	3.5	66.5	0.3	0.1	27.7

Table 5. Average cover of sites that clustered into each meadow type post 2022 flood.

Meadow Type	*Halophila spinulosa*	*Halophila ovalis*	*Zostera muelleri/ Halodule uninervis*	*Cymodocea serrulata*	*Syringodium isoetifolium*	*Bare*
Absent	0.0	0.0	0.0	0.0	0.0	100.0
Very Sparse	0.5	0.4	1.1	0.1	0.0	97.9
SI	7.4	10.4	4.2	1.0	31.0	47.0
CS	0.4	1.4	3.1	65.4	0.3	29.5
Moderate HS	45.0	4.6	3.3	0.1	0.2	46.8
Sparse HS	11.5	2.8	0.5	0.1	0.0	85.2
HO	0.8	16.4	1.0	0.1	0.1	81.7
Moderate ZMHU with HO	1.1	9.0	23.2	0.4	0.2	66.2
Dense ZMHU	0.6	4.7	72.3	0.2	0.1	22.1

Mapping seagrass distribution, density, species composition, meadow type and sediment type

Seagrass characteristics (distribution, density (% cover), species composition and meadow type) were mapped using ArcGIS Pro Geographic Information System (GIS). Each survey site included:

- Site information:
  - Site number, survey date and time, latitude and longitude, Depth (m), survey method (drop camera or snorkeler), and vessel.
- Seagrass cover information:
  - Seagrass species present, seagrass percent cover per species.
- Meadow type (as defined by cluster analysis in 'R')
- Additional information: sediment type, and comments

Seagrass distribution, density (% cover), species composition and meadow type have been displayed in separate maps. The steps/rules used to create each map are outlined below:

Seagrass (all species) polygons:

Polygons were created for the three survey periods pre-flood (2015 - 2021), first year post-flood (2022/23) and second year post-flood (2023/24). These polygons represent the most likely extent of seagrass (all species) in MBMP at the relevant period by applying the following steps/rules in order:

- A seagrass 'presence'/'absence' point map was created using data from the survey period represented (2015-2021; 2022/23; 2023/24).
- All sites where seagrass was observed as 'present' were included within the seagrass polygon, if their nearest neighbour also had seagrass 'present'.
- When a nearest neighbour was 'absent', the extent of the polygon was drawn at the halfway point between the 'present' and 'absent' sites. If there was a single 'absent' site surrounded by 'present' sites or 'present' and 'absent' sites were spatially mixed, then the area was assumed to have patchy seagrass and included in the polygon.
- If only 1% seagrass cover was present, and the site was surrounded by sites where seagrass was absent it could be excluded.
- Where no observations were made in the survey period of the polygon, the following information was used to interpolate between observations or extrapolate beyond observations:
  - Drawn to the 95th percentile with depths rounded to the next deepest 0.5m contour unless doing so exceeded the max depth for that zone or if the difference was < 0.1 from the shallower 0.5m contour; in these instances, it was rounded to the closest contour (using the new Moreton Bay DEM provided by the Queensland Wetland group within DES) (Table 6),
  - Seagrass present observations made during a different survey period were included in the period being mapped if the observation was equal to or shallower than the 95th percentile depth for all seagrass in the period being mapped,

- All post-flood seagrass present observations were included in the pre-flood polygon time period, unless a pre-flood absence was in the close vicinity.

Table 6. Depths contours that polygons of different survey periods were drawn to.

Area	Preflood 95th Percentile (-m)	2022 - 2023 95th Percentile (-m)	2023 – 2024 95th Percentile (-m)
Eastern Bay	4.5	4.5	4.0
Central Bay	2.5***	4.5	3.5
Deception Bay	3.0*	3.0*	2.5*
Bramble Bay	N/A	3.0*	2.5*
Waterloo Bay	3.5*	3.0*	3.5*
Southern Bay	0.5*	0.5*	1.0*
Broadwater	1.5**	1.0**	1.0**
Pumicestone Passage	N/A	1.5*	0.5*

**Drawn to DEM where possible (limited DEM accuracy due to turbidity), otherwise drawn to observations with buffer. Pumicestone Passage DEM not used at all due to inaccuracy when compared with observed depths.*
*** Broadwater does not have a DEM, so polygon drawn to observations and satellite imagery.*
****not enough preflood observations from this area for this to be a valid 95th percentile depth for the area. Polygon drawn to observations with buffer.*

Seagrass (all species) density (% cover):
This map shows the average percent cover of seagrass (all species combined) in MBMP in 2023/24. It has been created using only the 2023/24 seagrass survey data and the seagrass distribution (all species) 2023/24 polygon by applying the following steps/rules in order:

- A total seagrass cover point map was created for all survey sites within MBMP (2023/24 survey only)
- The percentage cover for each site were averaged within a 1km x 1km grid using ‘Fishnet’ geoprocessing tool in ArcGIS Pro.
- The colour of each pixel was determined by the average % cover of all observations within a pixel (including 0% for ‘absent’ sites)
- All pixels were combined and clipped to fill the seagrass distribution (all species) 2023/24 polygon (see above for method).

Individual point maps for each seagrass species showing the % cover separated into five density categories are also included in Appendix 1.

Seagrass meadow type:
This map shows the dominant seagrass meadow types in different areas of MBMP in 2023/24. It has been created using the 2023/24 seagrass survey data, after allocating a seagrass meadow type to each site, and the seagrass distribution (all species) 2023/24 polygon by applying the following steps/rules in order:

- A 1km x 1km grid was created over the 2023/24 sites which had seagrass ‘present’ using ‘Fishnet’ geoprocessing tool in ArcGIS Pro
- The dominant meadow type in each $1km^2$ pixel was represented by giving that pixel the colour associated with that particular meadow type.
- All pixels were combined and clipped to fill the seagrass distribution (all species) 2023/24 polygon (see above for method).

Change in Seagrass distribution:
Four maps were created in ArcGIS Pro to show spatial 'expansion', 'loss', and 'no change' variation between the following survey periods: Historic data (2002) and pre-flood (Pumicestone Passage only), pre-flood and 2022/2023, 2022/2023 and 2023/2024, and pre-flood and 2023/2024 (Baywide). These maps were all created following the same process listed below:

- The seagrass presence polygon for one survey period was clipped against the other using the geoprocessing tool, 'Erase', alternating between the polygons as to which was the 'input feature' and which was the 'erase feature'. This process resulted in the 'expansion' and 'loss' polygons for change map each map.
- The 'no change' polygon was created by using the geoprocessing tool, 'Clip' to create a polygon showing areas where seagrass was present in both survey periods.

Determination of changes in seagrass meadow type, density, and depth distribution

Using data from within the seagrass 'temporal study area' we calculated changes in the proportion of meadow type, density (% cover when present) for all seagrass species, and depth distribution of each seagrass species. To quantify the frequency of seagrass meadow occurrence at different depths, we categorised sites into Intertidal (above chart datum), Shallow Subtidal (< 2m deep) and Deep Subtidal (> 2m deep) for analysis. We recognise this depth cut-off is much shallower than that used elsewhere in Queensland to define "deep" habitats, however given the high turbidity of MBMP, the change in community composition from shallow to deep water species occurs at a much shallower depth that is close to the 2m depth contour throughout MBMP.

The proportion of meadow types gives an indication of how frequently that meadow type occurred in each sampling period, region, and depth zone, while the percent cover graphs indicate the changes in the density of each seagrass species for each sampling period, region, and depth zone.

The maximum depth of each seagrass species (when present in a region) was calculated in 2 ways. Firstly, by taking the depth of the 98th percentile of that species within each region, and secondly, by analysing for the 98th percentile depth at which 10% cover of each species was recorded. Very sparse seagrass often extended much deeper than the denser seagrass meadows, and hence calculating the maximum depth of observations with percent cover $\geq$ 10% provided a more realistic representation of the depth to which seagrass was flourishing. Seagrass with percent cover < 10% likely represent a declining (dying) meadow or a meadow in the early stages of recovery. Every effort has been made to check on outlier observations, but to ensure misidentified observations (e.g. detritus recorded as sparse seagrass) did not distort the results, the 98th percentile of depth distribution has been used to estimate maximum depth in this report.

Non-parametric Wilcoxon tests were used to analyse significant differences in percent cover and depth range between sampling periods within a region and depth zone. This analysis method was chosen as the data was not normally distributed. Please note: the Wilcoxon Test is testing for a difference in the distribution of observed percent cover and depths, and while it does identify where changes have occurred, it will not always line up with what would appear to be significant by visually appraising the overlap of s.e. bars on the average bar graphs. We used the cut-off $n > 2$, for calculating average percent cover of each species. If any species did not occur more than 2 times in the given region, and depth zone no average percent cover was calculated for it.

Biomass and seed bank assessment

In January and February 2024, seagrass cores were collected from nine locations within Moreton Bay Marine Park. Locations were selected that had relatively typical seagrass composition for its area, and represented the Shallow and Deep Subtidal habitats, as these were the habitats most affected by the 2022 flood. At each location 3 x 15cm diameter cores were collected by snorkel and returned to the surface with minimal disturbance of the surface sediment (Figure 13). All sediment and seagrass from the cores were then filtered through a 2cm sieve to collect the seagrass biomass and a 500µ sieve to capture any seeds or seed casings.

Biomass

Seagrass biomass from each core was separated into separate species (if multiple species were present) and then into the above and below-ground components of the seagrass (leaves, rhizomes and roots). Biomass samples were dried initially by airing in sunlight for approximately 4 hrs to reduce the moisture content and then overnight in a drying oven at 60^{0}C (Figure 13). Samples were then weighed to an accuracy of 0.1mg using scientific scales at Moreton Bay Research Centre. The resulting dry weights of the three replicate cores were averaged and normalised to represent the average gDW m^{-2} of seagrass leaves, rhizomes and roots at each location (i.e. 1gDW.$core^{-1} \sim 57$gDW.m^{-2}).

Seeds

The diameter of seagrass seeds varies significantly between species (also potentially between different seagrass morphology, within the same species). The diameter of *Cymodocea serrulata* seeds are approximately 5mm, *Halodule uninervis*, approximately 2mm, *Zostera muelleri,* approximately 1mm and *Halophila spp*. Seeds, approximately 500µ (Waycott et. al. 2004). As all sediment was passed through a 500µ sieve, we expected to find all *C. serrulata, H. uninervis,* and *Z. muelleri*, seeds. Moderate to large *Halophila spp*. seeds are also likely to have been caught in the 500µ sieve (if present), but we only found a single *Halophila spp*. seed in our samples (Figure 50).

Everything that did not go through the 500µ sieve was inspected with a dissecting scope to identify seeds or seed casings. We also filtered sediment with the 250µ sieve to look for smaller *Halophila spp*. seeds, but too much sediment was retained by this sieve to perform a thorough microscopic investigation.

The total number of seeds/ seed casings identified from each core, were averaged between the 3 replicate cores from each location and normalised to the number of seeds m^{-2} (i.e. 1 seed. Core^{-1} ~ 57 seeds.m^{-2}).

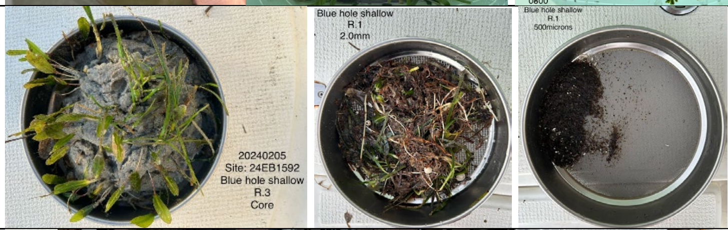
<table>
<tr><td>Three replicate 15 cm diameter cores collected at each of 9 locations across Moreton Bay. Some samples were stored in plastic tubs prior to sieving.</td><td>
</td></tr>
<tr><td>Samples sieved through a 2mm and 500µ sieve to separate seagrass biomass from detritus with potential seeds or seed casings.</td><td>
</td></tr>
<tr><td>Everything caught in the 2mm sieve was sorted into seagrass species and leaves, rhizomes, roots. Detritus left after sorting was checked for any seeds using a dissecting microscope.</td><td></td></tr>
<tr><td>Everything caught in the 500µ sieve was sorted using a dissecting microscope to identify and count seeds. Seagrass samples (after sorting) were dried at 60°C in a drying oven then weighed on a high precision scientific scale.</td><td></td></tr>
</table>

Figure 13. Process of collecting, sieving, processing and weighing biomass and seed abundance samples from 9 seagrass meadows in Moreton Bay

RESULTS

Seagrass Distribution and Meadow Types in Moreton Bay

Seagrass in Moreton Bay is clustered around the edges of the Bay, predominantly occurring in the shallow embayments in Western Bay, the sandbars of Eastern Bay, the intertidal edges of Pumicestone Passage, Southern Bay and Broadwater (Figure 14). Seagrass has never been observed in the middle of Moreton Bay, except for occasional sparse seagrass on the sandbars of North Passage.

Seagrass distribution in MBMP declined from a pre-flood distribution of 327 km^2 to 304 km^2 in the first year post-flood (2022/23). In the second year post-flood (2023/24), seagrass distribution further declined to a distribution of 292 km^2 (Figure 15). The reduced seagrass distribution is mainly attributable to the decline in seagrass depth distribution post-flood (Table 7). This contraction in depth range of seagrasses in Moreton Bay constrains seagrass distribution into shallower water, limiting the spatial extent of seagrass in the subtidal zones of many regions of Moreton Bay (Figure 16).

Seagrass distribution declined along the deeper edges of subtidal meadows between pre-flood (2015-21) and the first year post-flood (2022/23); particularly in the Central, Eastern and Southern Bay regions (Figure 16). Between the first and second year post flood (2023/24) there was some recovery along the deeper edges of meadows in the Waterloo, Southern and Eastern Bay regions. Seagrass distribution declined in Deception Bay and Bramble Bay, and some parts of Eastern Bay, between the first and second year post flood. Overall seagrass extent in the second year post flood (2023/24) is lower than the pre-flood distribution.

Table 7. The 95th percentile depth (-m) of seagrass occurrence in each region of Moreton Bay used to interpolate between sites when mapping distribution.

Moreton Bay region	Pre-flood	First year post flood (2022/23)	Second year post flood (2023/24)
Eastern Bay	4.48 (2020-21)	4.30	3.72
Central Bay	N/A	4.15	3.29
Deception Bay	3.08 (2020-21)	2.70	2.41
Bramble Bay	N/A	3.05	2.36
Waterloo Bay	3.41 (2020-21)	2.95	3.28
Southern Bay	0.31 (2015-16)	0.55	0.73
Broadwater	1.25 (2015-16)	0.76	0.67
Pumicestone Passage	N/A	1.23	0.44

N/A – insufficient pre-flood sampling within this region to calculate an accurate 95th percentile depth.

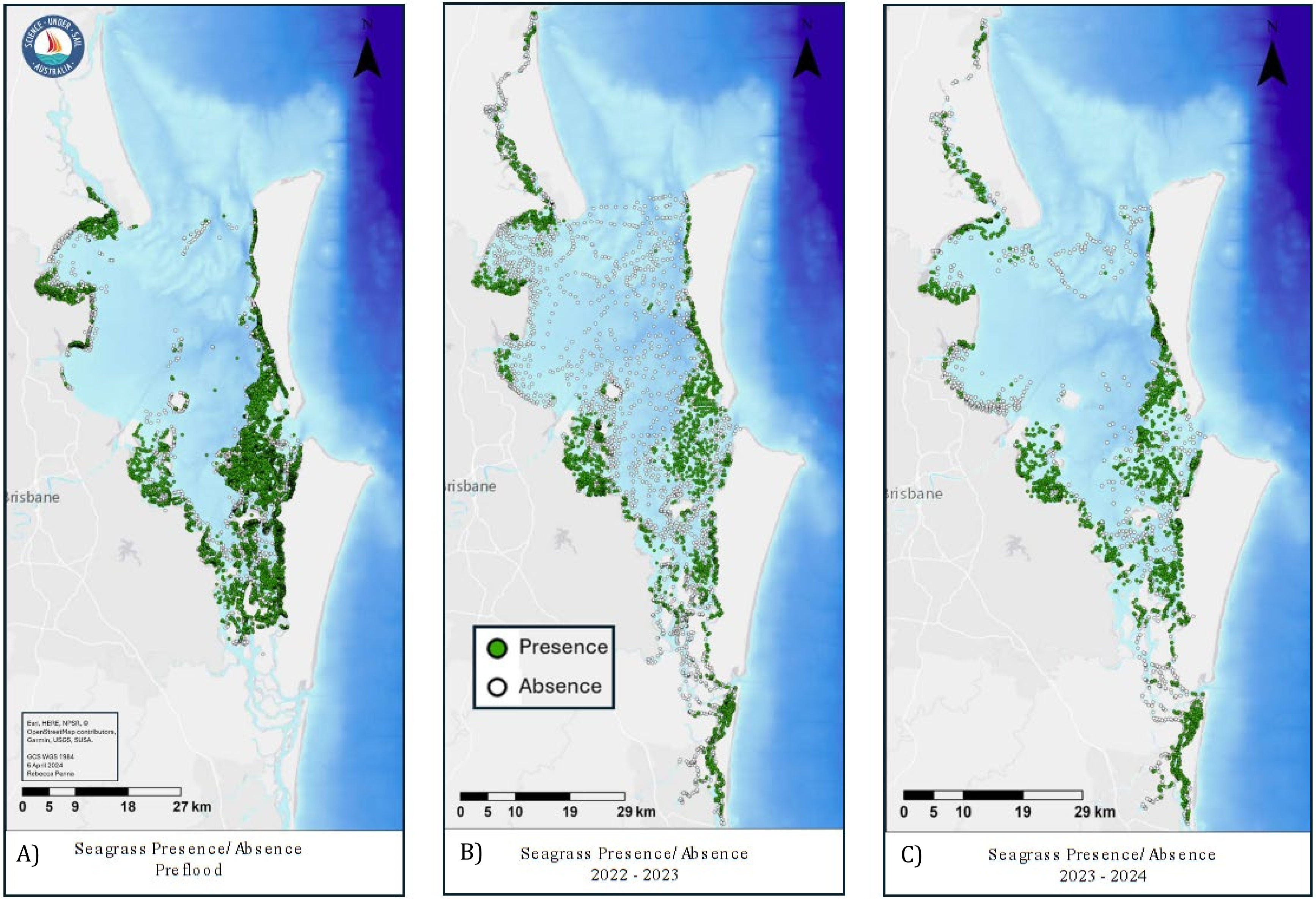


Figure 14. Map of seagrass presence/absence at all surveyed sites during the three survey periods: A) Pre- flood (2015-21) B) First year post-flood (2022/2023) C) Second year post-flood (2023/ 2024). Presence indicated by green, absence indicated by white.

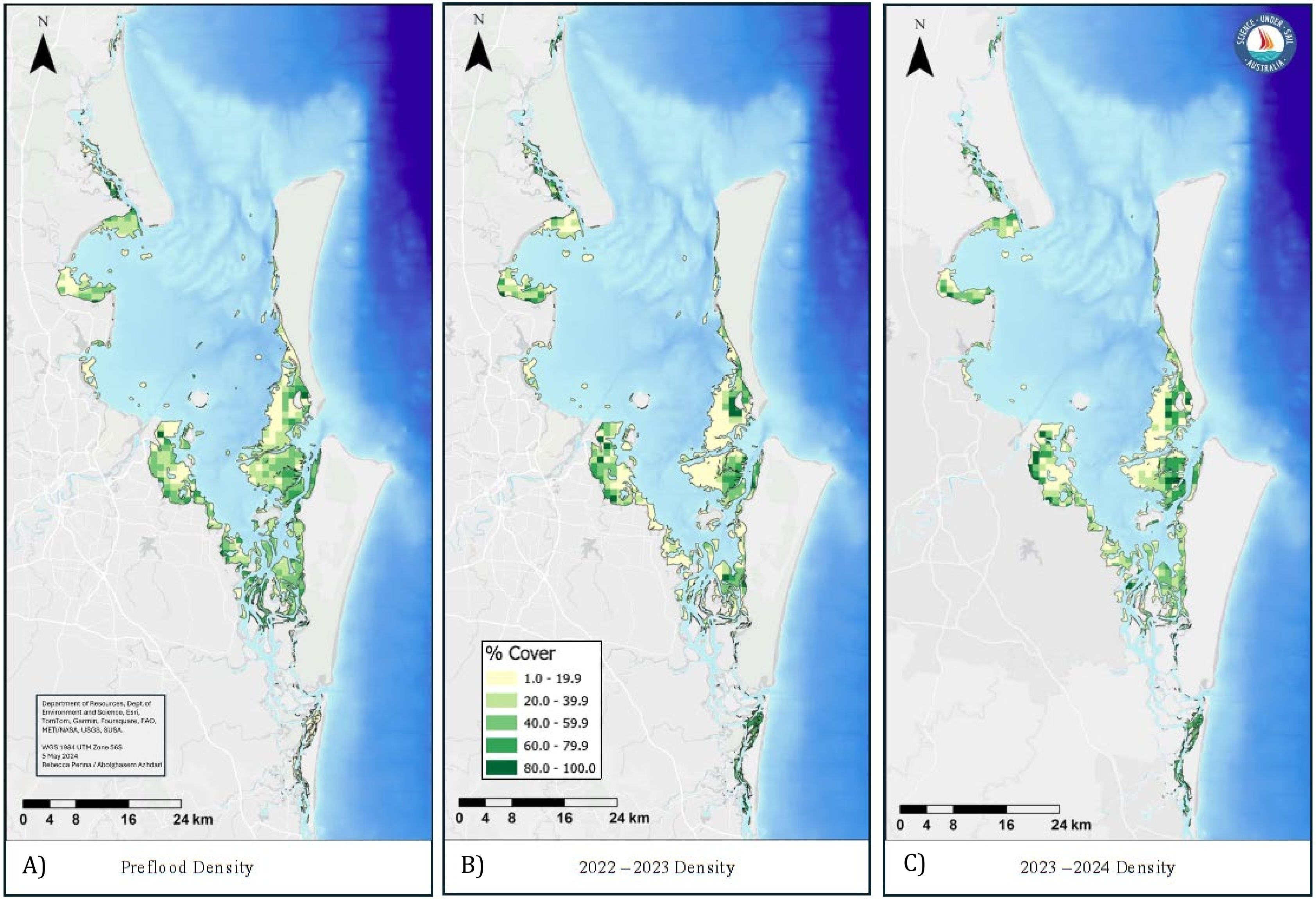


Figure 15. Seagrass distribution in Moreton Bay: A) Pre- flood (2015-21) B) First year post-flood (2022/2023) C) Second year post-flood (2023/2024). Polygons represent likely seagrass distribution based on approx. 21,000 benthic habitat observations between 2015 and 2024. Extent of seagrass polygon is determined by in-situ observations, the maximum depth distribution of seagrass in each zone and a high-resolution DEM (supplied by DES). The density shown represent the average density of seagrass in each 1km².

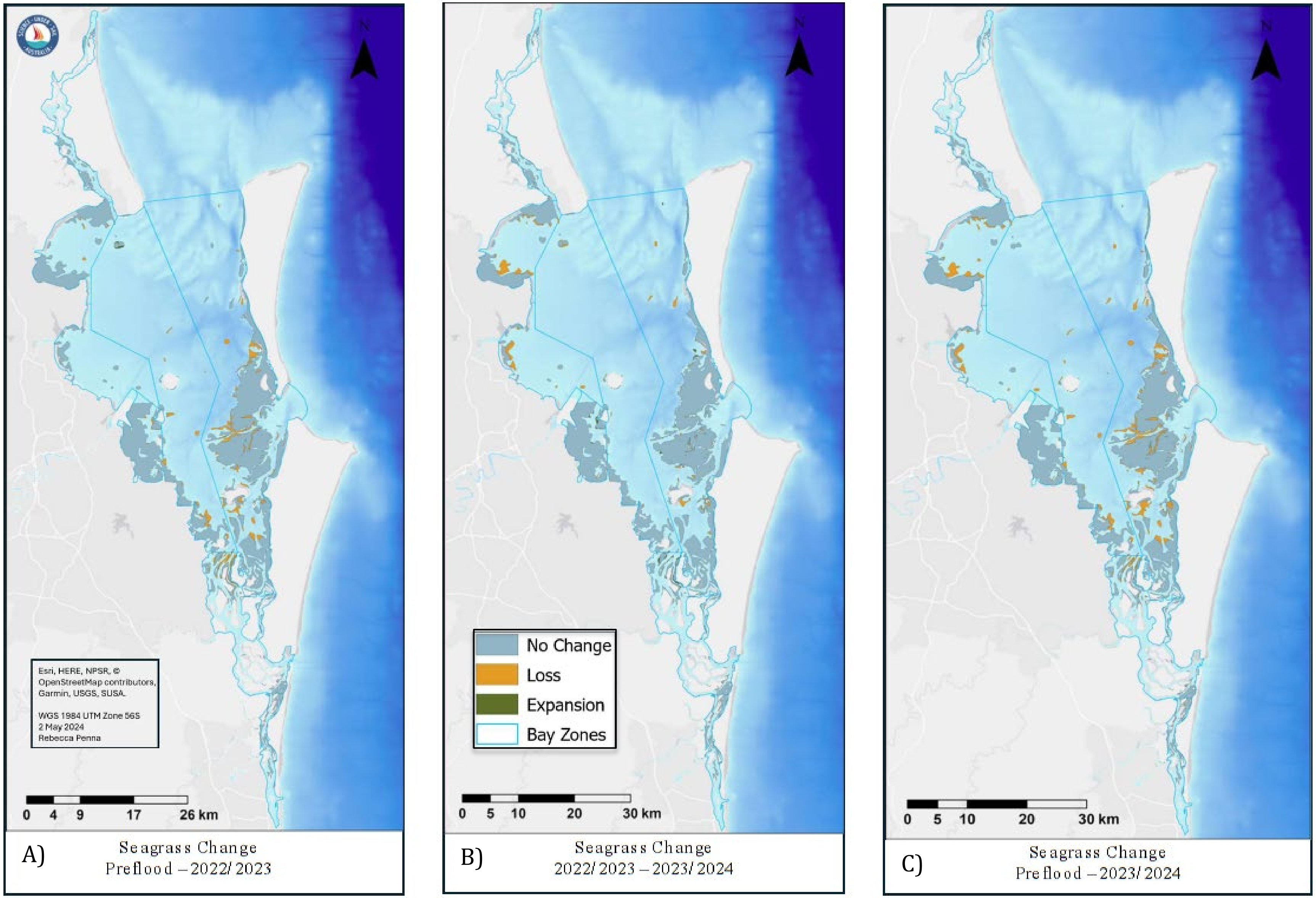


Figure 16. Change in seagrass distribution change in Moreton Bay: A) Pre-flood (2015-2021) to 2022/2023 B) 2022/2023 to 2023/2024 C) Pre-flood (2015-2021) to 2023/2024.

The proportion of surveyed sites where seagrass was present decreased in all depth zones following the 2022 floods, with the largest decline observed in the Shallow Subtidal and Deep Subtidal zones (Figure 17). In the second year post-flood (2023/24), seagrass occurrence had recovered to pre-flood proportions in the Intertidal and Shallow Subtidal zones, however some *Moderate HS* and *Sparse HS* meadows had been replaced by *HO*. The proportion of sites where seagrass was observed in the Deep Subtidal zone recovered slightly since the first year post-flood (2022-23), but is not yet back to pre-flood levels, with *Sparse HS* and *Moderate HS* accounting for most of the loss (Figure 17).

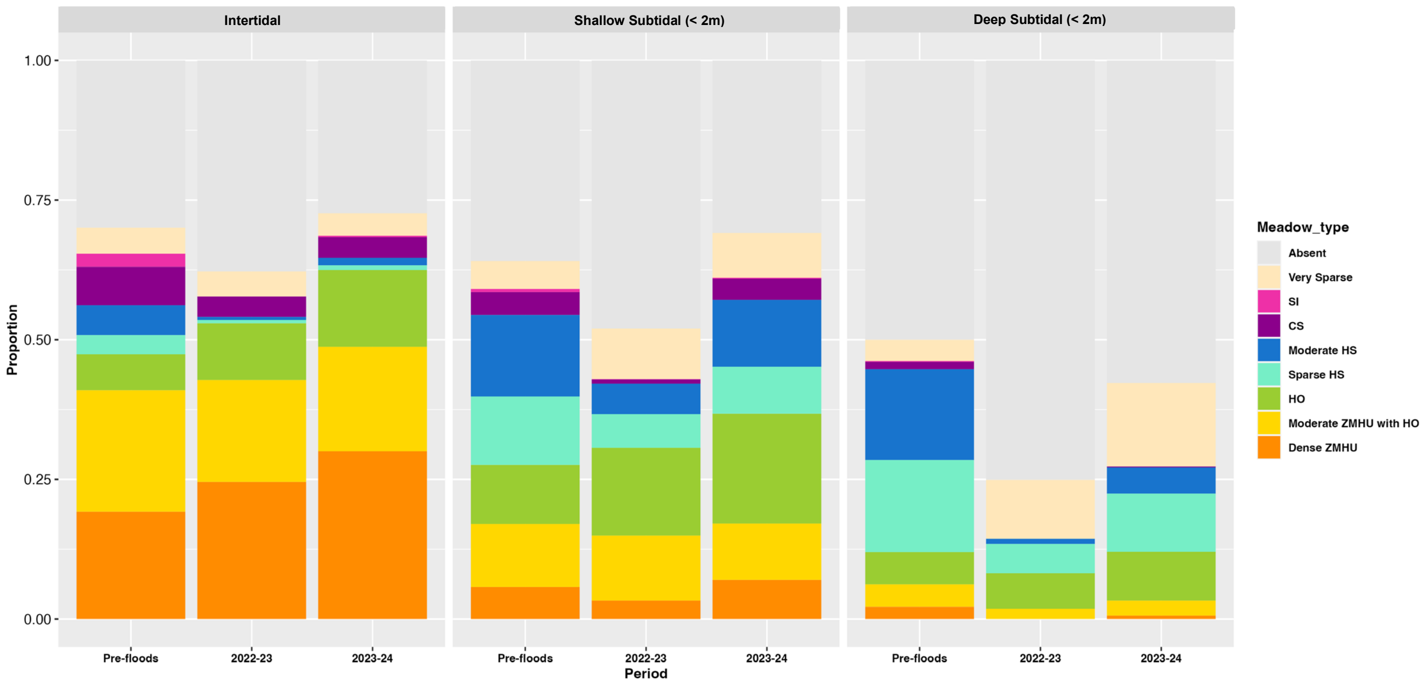


Figure 17. Comparison of the proportion of meadow types observed within the Moreton Bay 'temporal sample area' between the pre-flood, 2022/23 and 2023/24 survey periods (excluding Pumicestone Passage).

Baywide, the *Dense ZMHU* and *Moderate ZMHU with HO* meadow types dominate the Intertidal zone and were the least impacted by the 2022 flood (Figure 17, 18). The Shallow Subtidal zone represents a transition region with *Z. muelleri/ H. uninervis, H. ovalis* and *H. spinulosa* all contributing to the major meadow types. *CS* and *SI* meadows occur predominantly in Eastern Bay (Figure 18). CS declined in all depth zones in the first year post flood, only recovering in the Intertidal and Shallow Subtidal in the second year post-flood. Post flood the *SI* meadow type existed very rarely (Figure 17, 18). Pre-flood *H. spinulosa* was the dominant species in the Deep Subtidal zone, with *Moderate HS* and *Sparse HS* meadows representing 33% of observations in the Deep Subtidal zone. However, the abundance of *H. spinulosa* dramatically declined post-flood, with *HS* meadow types representing only 6% of observations in the first year post flood (2022/23). By the second year post-flood *HS* had partially regrown, accounting for 15% of observations (half of its pre-flood frequency). In the second year post-flood the *HO* meadow type increased beyond its pre-flood proportion to become an equal dominant meadow type with *Sparse HS* in the Deep Subtidal zone in 2023/24.

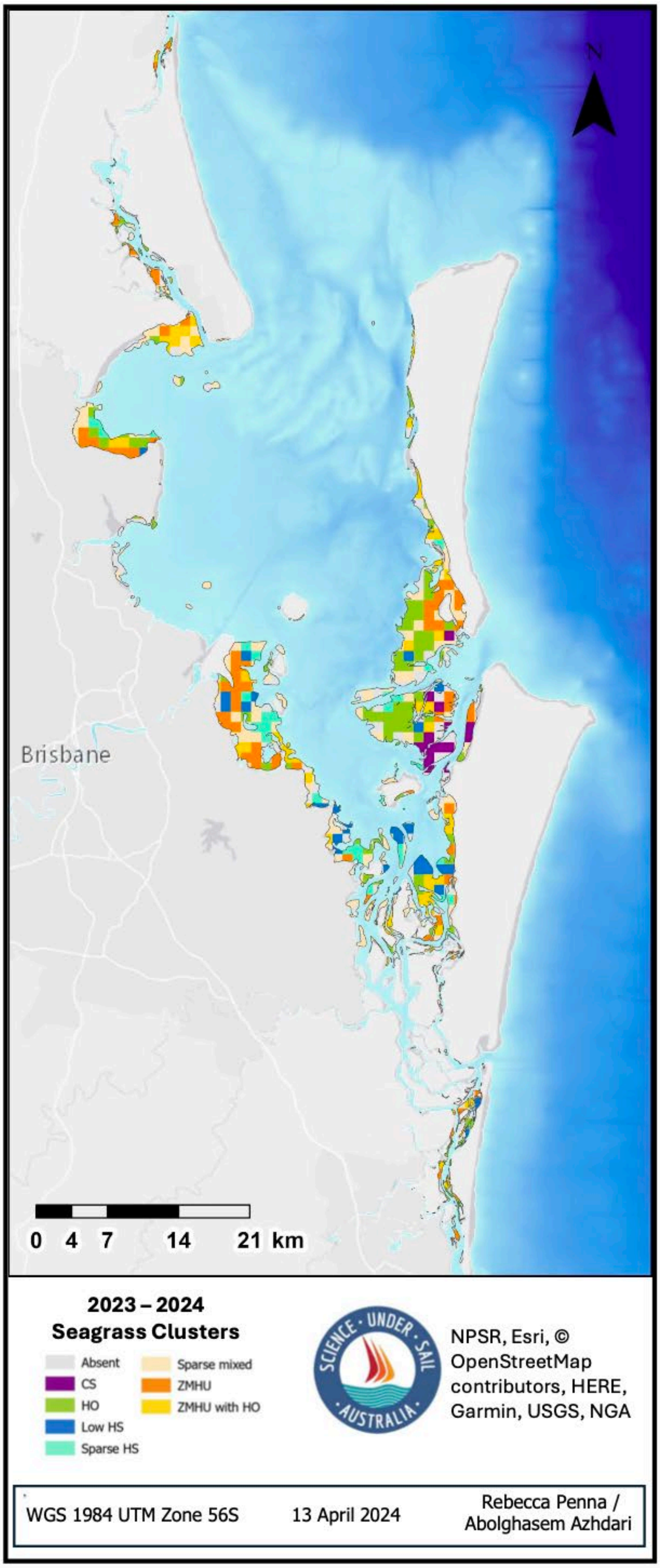


Figure 18: Distribution of meadow types across Moreton Bay during the 2023/24 survey period (second year post 2022 flood). Dominant meadow type displayed for each 1km².

When seagrass was present in the Intertidal zone, it existed at higher density post-flood compared to pre-flood (Figure 19). However, in the Shallow Subtidal and Deep Subtidal, the seagrass density declined in the first year following the 2022 flood. During the second year post-flood seagrass density had recovered to pre-flood levels in the Shallow Subtidal, however the Deep Subtidal seagrass remained sparser than pre-flood densities (Figure 19).

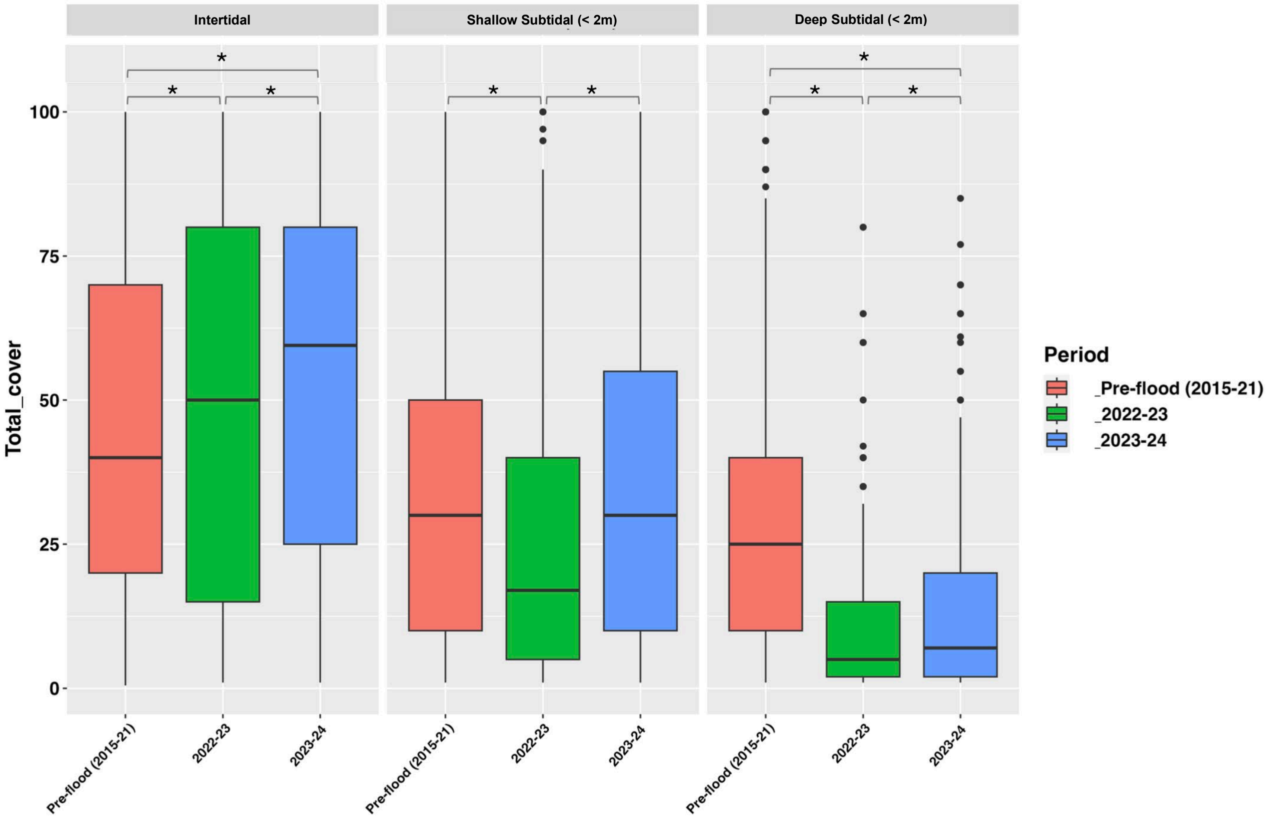


Figure 19. Box and whisker plot comparison of the density of seagrass (all species), when present, observed in the temporal comparison sample area. * Indicate significance ($p < 0.05$) as determined by pairwise Wilcoxon tests between periods.

Eastern Bay

Seagrass in Eastern Bay is concentrated along the western shoreline of Moreton and North Stradbroke Island and on the Eastern Banks (Figure 20A). In the second year post-flood high density seagrass meadows existed along the western side of North Stradbroke Island and in some deeper areas on the Eastern Banks, although much of this region had sparse seagrass cover. Eastern Bay contains the majority of Moreton Bay's seagrass meadows, with seagrass covering approximately 169km^2 prior to the 2022 flood. The seagrass meadows reduced in areal extent by approximately 15km^2 in the first year post-flood (2022/23), with a further 5km^2 reduction in the second year post-flood (2023/24)(Figure 20B). Most of the seagrass loss occurred in the deeper areas.

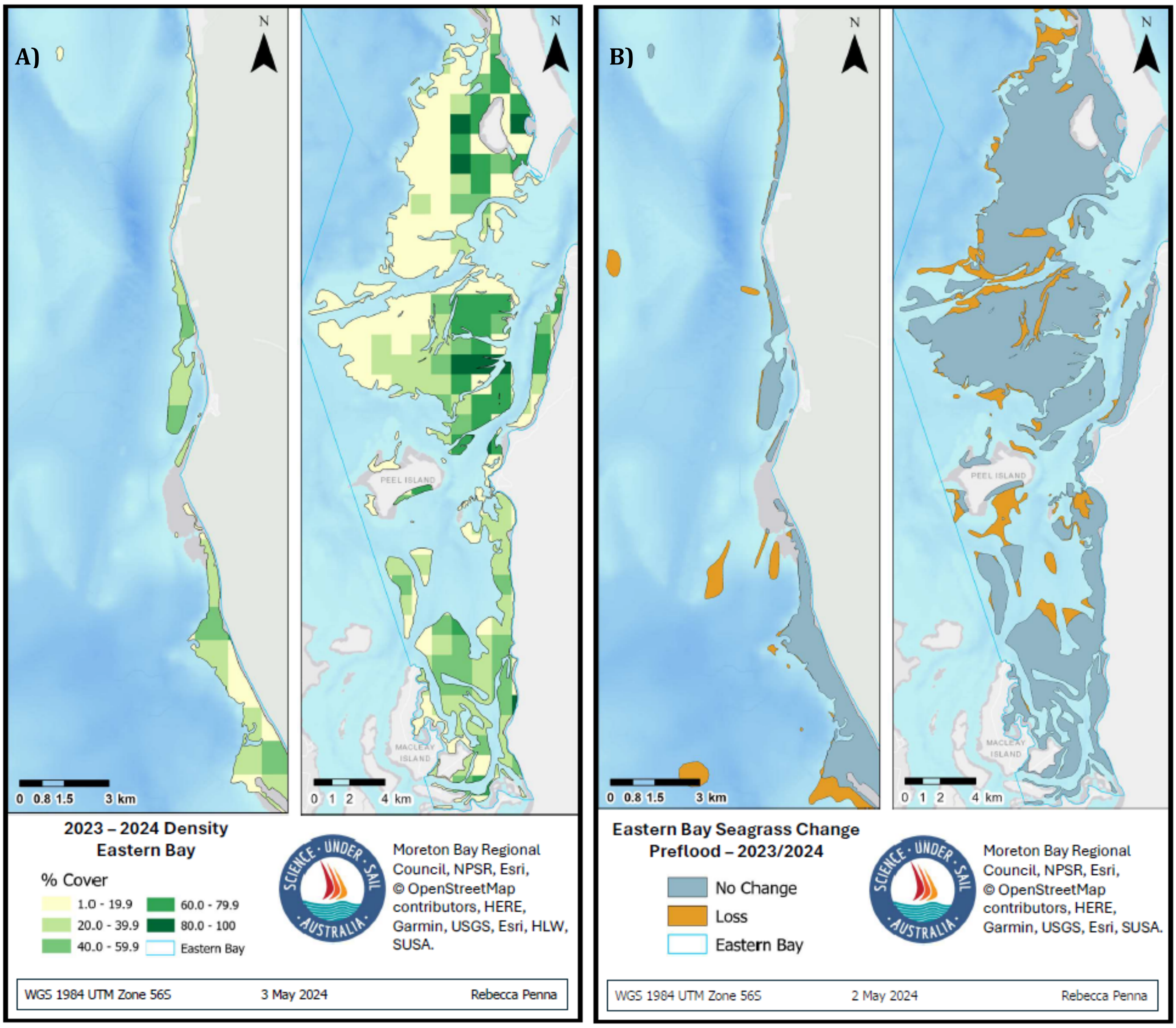


Figure 20. Areas in Eastern Bay where seagrass (all species) was present or likely to occur.

A) Average density of seagrass cover in 2023/24 within a 1km^2 pixel (second year post-flood), darker green indicates higher density (% cover) B) Changes in seagrass distribution between pre-flood (2015-21) and the second year post-flood (2023/24).

The proportion of sites where seagrass was present declined in all depth zones of Eastern Bay during the first year following the flood (2022/23) (Figure 21). The biggest decline was in the Deep Subtidal zone, with *Absent* observations increasing from 46% pre-flood to 77% in the first year post-flood. The two *HS* meadow types accounted for 37% of pre-flood observations (all zones), with *Moderate HS* occurring in 19% of the observations (2015-2021). Both *Moderate* and *Spares HS* declined post-flood. *Moderate HS* almost disappeared from the Deep Subtidal zone (0.4%) in the first year post-flood (2022/23) and only accounted for 4% of observations in 2023/24. *SI* meadows were already rare pre-flood, accounting for approximately 3.5% of survey sites in the Eastern Bay Intertidal zone. In the first year post-flood, *SI* meadows were not observed, and by the end of the second year post-flood *SI* meadows accounted for only 0.5% of observations in the Intertidal zone. *CS* meadows also declined in the Shallow and Deep Subtidal zones following the 2022 flood, however the proportion of *CS* meadows in the Shallow Subtidal zone had recovered by the 2023/24 sampling season. *CS* meadows have not returned to the Deep Subtidal zone since the flood. The proportion of *HO* and *Very Sparse* meadows has increased since pre-flood across all depth zones, in some cases replacing the meadow types that declined.

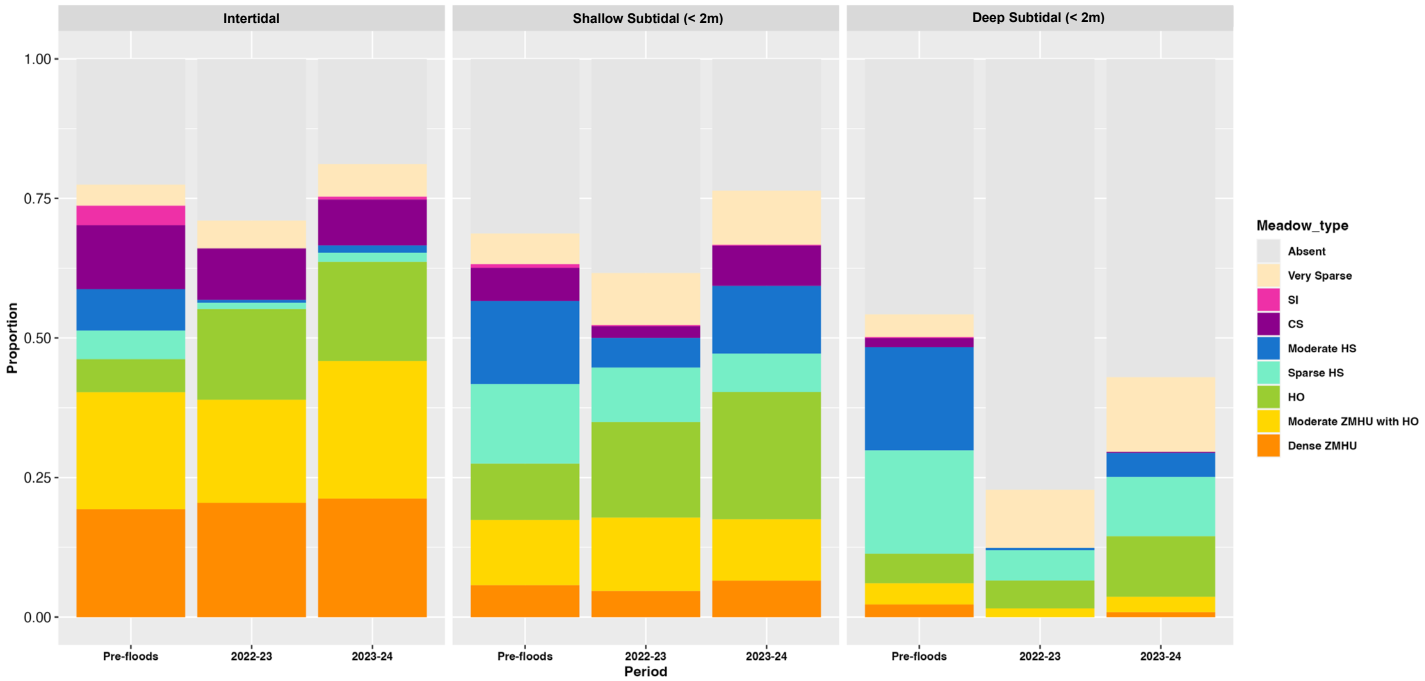


Figure 21. Comparison of the proportion of meadow types observed within the 'temporal sample area' in Eastern between the pre-flood, 2022/23 and 2023/24 survey periods.

When *H. spinulosa* was present, its density (% cover) declined across all depth zones following the 2022 flood. In the Intertidal, *H. spinulosa* has remained sparse post-flood. In the Shallow Subtidal and Deep Subtidal zone, *H. spinulosa* density has recovered significantly since 2022/23 though it remains sparser than pre-flood (Figure 22). The average density of *H. ovalis* declined in the Intertidal and Shallow Subtidal zones in the first year post-flood (2022/23), yet in the second year since the flood (2023/24), had recovered to pre flood levels. In the Deep Subtidal zone, *H. ovalis* density did not change immediately post flood, however in 2023/24 it was sparser than pre-flood observations. Average density of *Z.*

muelleri/ H. uninervis has declined in all depth zones since pre-flood with the greatest decline in the Deep Subtidal zone. The density of *C. serrulata* in the Intertidal zone declined in 2022/23 but had recovered by the 2023/24 survey. The density of *C. serrulata* has not changed significantly in the Shallow Subtidal zone. *C. serrulata* was not observed in the Deep Subtidal zone in 2022/23, with rare observations in the Deep Subtidal zone in 2023/24 significantly sparser than pre-flood (Figure 22).

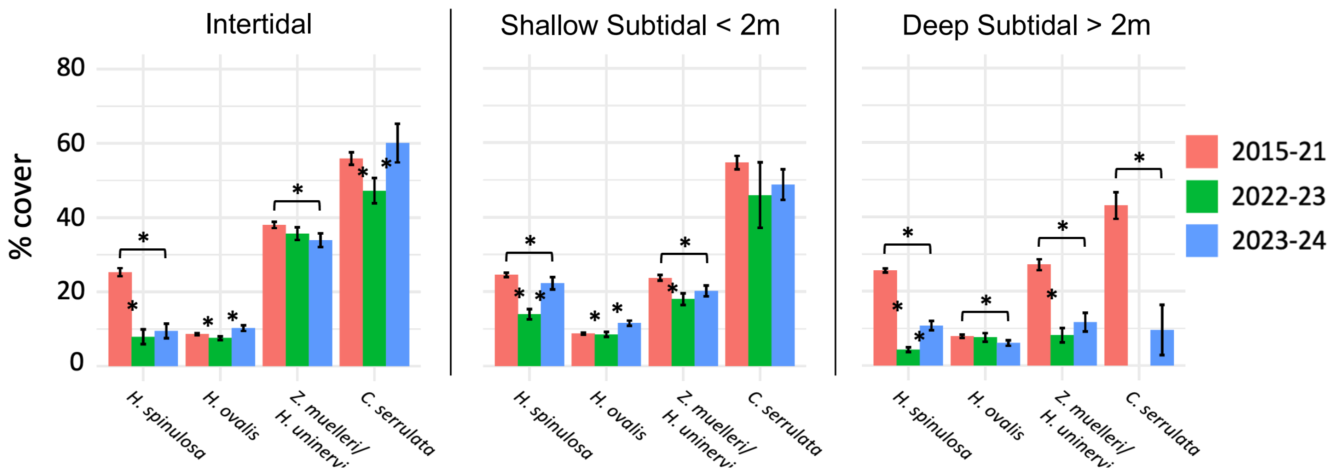


Figure 22. Change in seagrass density (% cover) between pre-flood (2015/2021), 2022/23 and 2023/24 in Eastern Bay. * Indicates significant difference between adjacent years on bar graph. * Over bracket represents significant difference between 2015/21 and 2023/24.

The average depth of *H. spinulosa* in Eastern Bay (1.8m) has not significantly changed since pre-flood surveys (2020/21), however, the maximum depth become shallower in both the first (2022/23) and second year (2023/24) post-flood (Figure 23). The average depth of *H. ovalis, Z. muelleri / H. uninervis* and *C. serrulata* were shallower in the first year post-flood, however the average depth of all three species recovered in the second year post-flood to be similar to pre-flood depths (Figure 23).

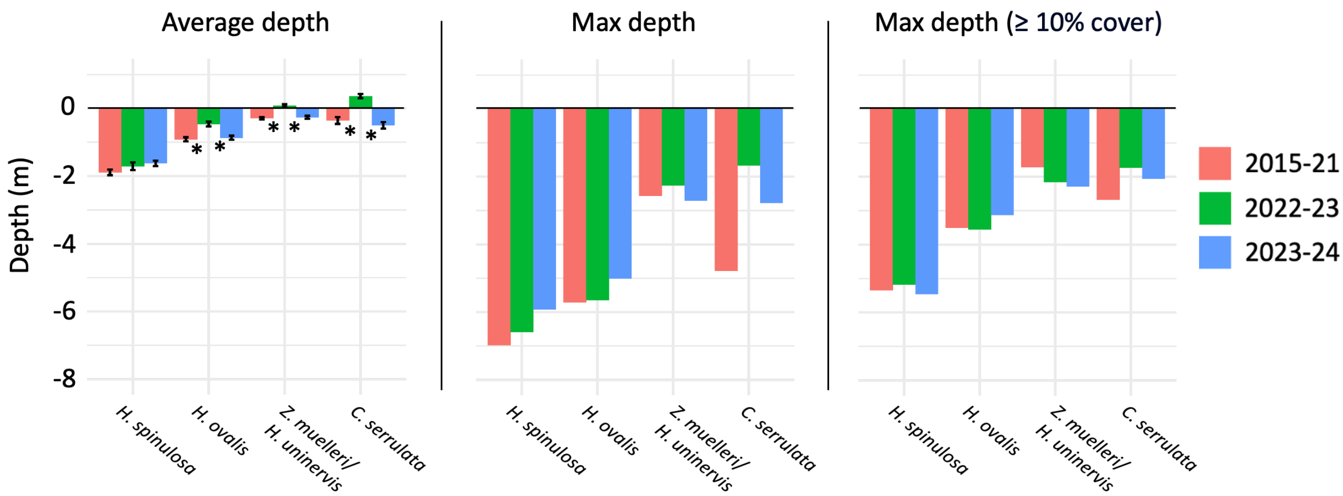
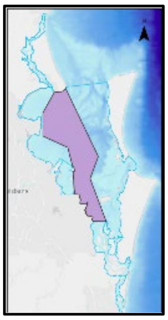


Figure 23. Changes in seagrass average depth, maximum depth and maximum depth when density ≥ 10% cover for each species in Eastern Bay. Comparing 2020/21, 2022/23 and 2023/24 depth distributions. * Indicates significant difference between adjacent years on bar graph. * Over bracket represents significant difference between 2015/16 and 2023/24.



Central Bay

Seagrass in Central Bay occurs along the southern edge of Bribie Island as well as along the coastline from Wellington Point south to Coochiemudlo Island. Small areas of seagrass have also been observed in the deeper water west of the Eastern Banks, on the eastern side of St Helena and Green Island and around Mud Island (Figure 24A). Seagrass density and areal extent is greatest in the southern sections of Central Bay between Wellington Point and Coochiemudlo Island. Pre- flood seagrass covered approximately 29km^2 in Central Bay. In the first year post-flood (2022/23) seagrass distribution had declined by 3.6km^2, with a further 0.4km^2 loss in the second year post flood (2023/24) (Figure 24B).

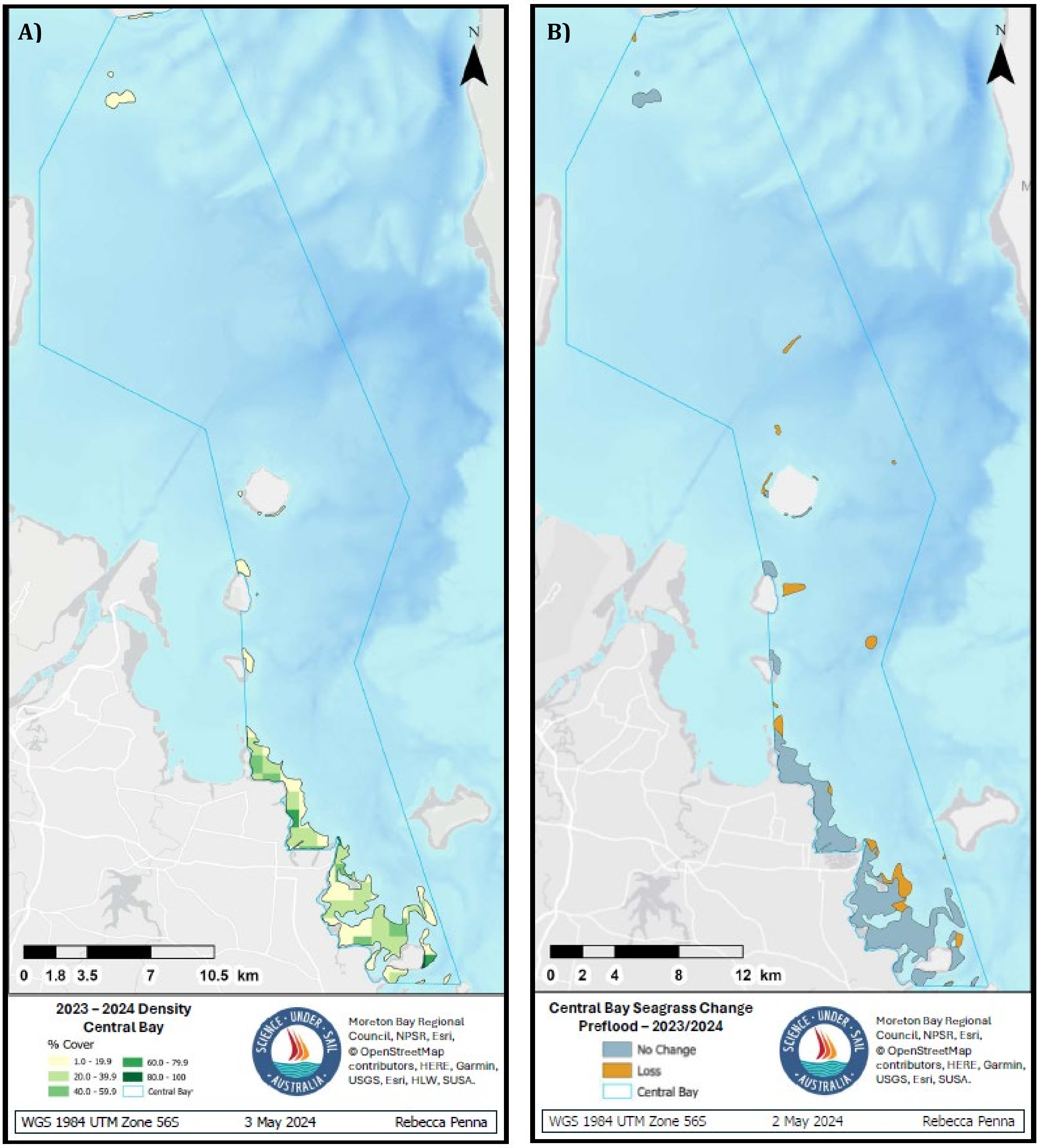
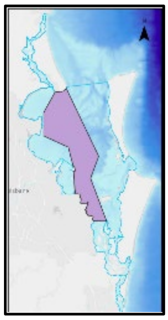


Figure 24. Areas in Central Bay where seagrass (all species) was present or likely to occur. A) Average density of seagrass cover in 2023/24 within a 1km^2 pixel (second year post-flood), darker green indicates higher density (% cover) B) Changes in seagrass distribution between pre-flood (2015-21) and the second year post-flood (2023/24).



In the temporal study area of Central Bay, the proportion of sites where seagrass was observed declined in all depth zones following the flood in 2022/23. The proportion of seagrass has shown strong signs of recovery in the 2023/24 survey however there has been a shift in meadow type post-flood in the Intertidal and Deep Subtidal zones (Figure 25). In the Intertidal, *Dense ZMHU* and *HS* meadow declined in the first year post-flood (2022/23). By 2023/24 *Dense ZMHU* had partially recovered but had also been replaced by *Moderate ZMHU with HO* and *HO.* In the Shallow Subtidal zone *Moderate HS* and *Sparse HS* declined in the first year post-flood (2022/23), however were then observed at a higher proportion of sites in Central Bay in 2023/24 than pre-flood. *Dense ZMHU* has not been observed in the Shallow Subtidal since the flood. In the Deep Subtidal zone, Moderate *HS, CS*, *Dense ZMHU* and *Moderate ZMHU with HO* have disappeared since the flood, while the proportion of sites with *Very Sparse*, *Sparse HS* and *HO* has increased. *HO* has become the dominant meadow type in the Deep Subtidal zone since the flood (Figure 25).

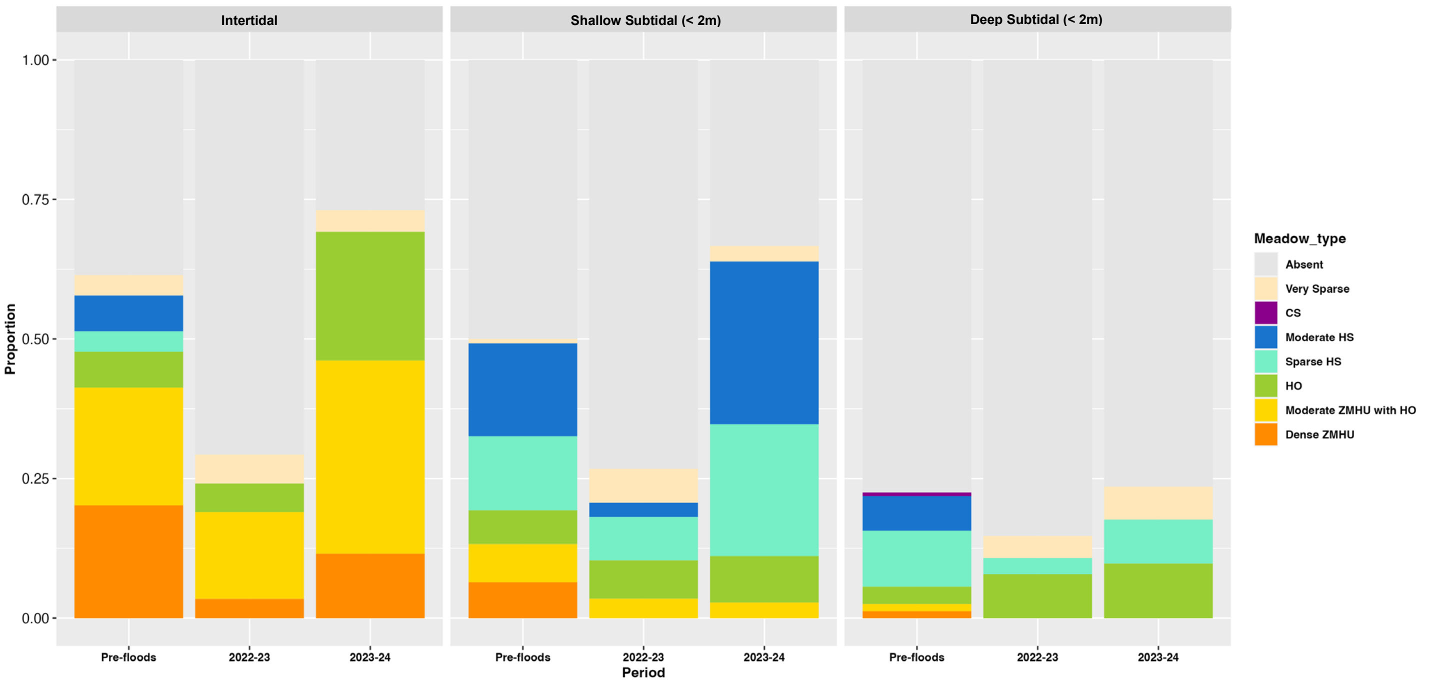
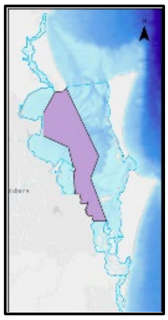


Figure 25. Comparison of the proportion of meadow types observed within the sample area in Central Bay between the pre-flood, 2022/23 and 2023/24 survey periods.

Where seagrass was present, the density of different species exhibited various responses to the 2022 flood in Central Bay (Figure 26). *H. spinulosa* and *Z. muelleri/ H. uninervis* both declined and remain either absent or sparse post-flood in all depth zones whereas *H. ovalis* varied between zones. Where *H. spinulosa* was observed, its density declined in the Shallow and Deep Subtidal in the first year post flood (2022/23). In the second year post flood (2023/24) *H. spinulosa* density had recovered to pre-flood levels in the Shallow Subtidal, however it has remained sparse in the Deep Subtidal. Where *H. ovalis* was observed in the Shallow Subtidal, it was significantly sparser in the second year post-flood (2023/24) than in the pre-flood and 2022/23 surveys. *H. ovalis* density was not significantly different between sampling years in the Intertidal and Deep Subtidal.



Where *Z. muelleri/ H. uninervis* was observed, it was significantly sparser following the 2022 flood in all depth zones. It has remained sparse in the Intertidal, though there has been some recovery in the Shallow Subtidal (Figure 26).

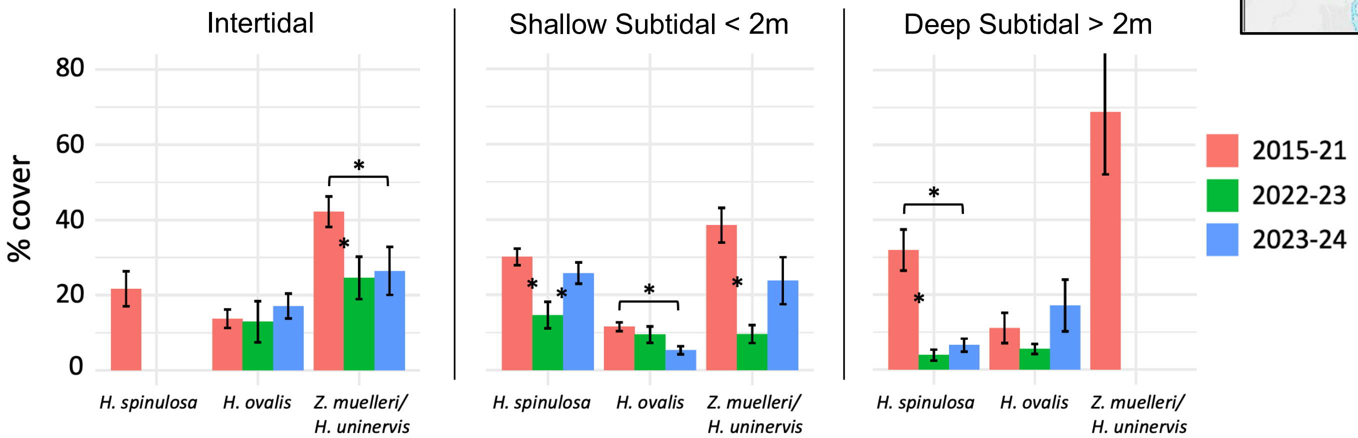
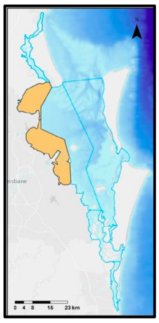
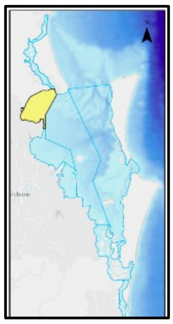


Figure 26. Change in seagrass density (% cover) between pre-flood, 2022/23 and 2023/24 in Central Bay. * Indicates significant difference between adjacent years on bar graph. * Over bracket represents significant difference between 2015/21 and 2023/24.



Western Bay

The seagrass meadows of the Western Bay covered a total areal extent of approximately 97km^2 pre-flood. In general, the Western Bay meadows had a delayed response to the 2022 flood compared to other regions, with seagrass distribution declining by approximately 1 km^2 in the first year post-flood (2022/23) and a further 8km^2 in the second year post-flood (2023/24). Western Bay is comprised of Deception Bay, Bramble Bay, and Waterloo Bay. Due to the difference in proximity of the three Bays to the Brisbane River, the impact of the 2022 flood has varied significantly, and thus, the three Bays have been analysed separately.



Deception Bay

Deception Bay has seagrass meadows in its northern and southern section, but a bare area in the middle, due to turbidity from the Caboolture River (Figure 27). Poor water clarity limits the depth to which seagrass grows, with sparse *H. spinulosa* and *H. ovalis* meadows transitioning into bare substrate as it gets deeper. The total areal extent of seagrass pre-flood was 40km^2, with only a 0.6km^2 decline observed in the first year post-flood (2022/23), but a 5km^2 reduction occurring in the second year post-flood (2023/24), with significant loss of the deeper seagrass meadows.

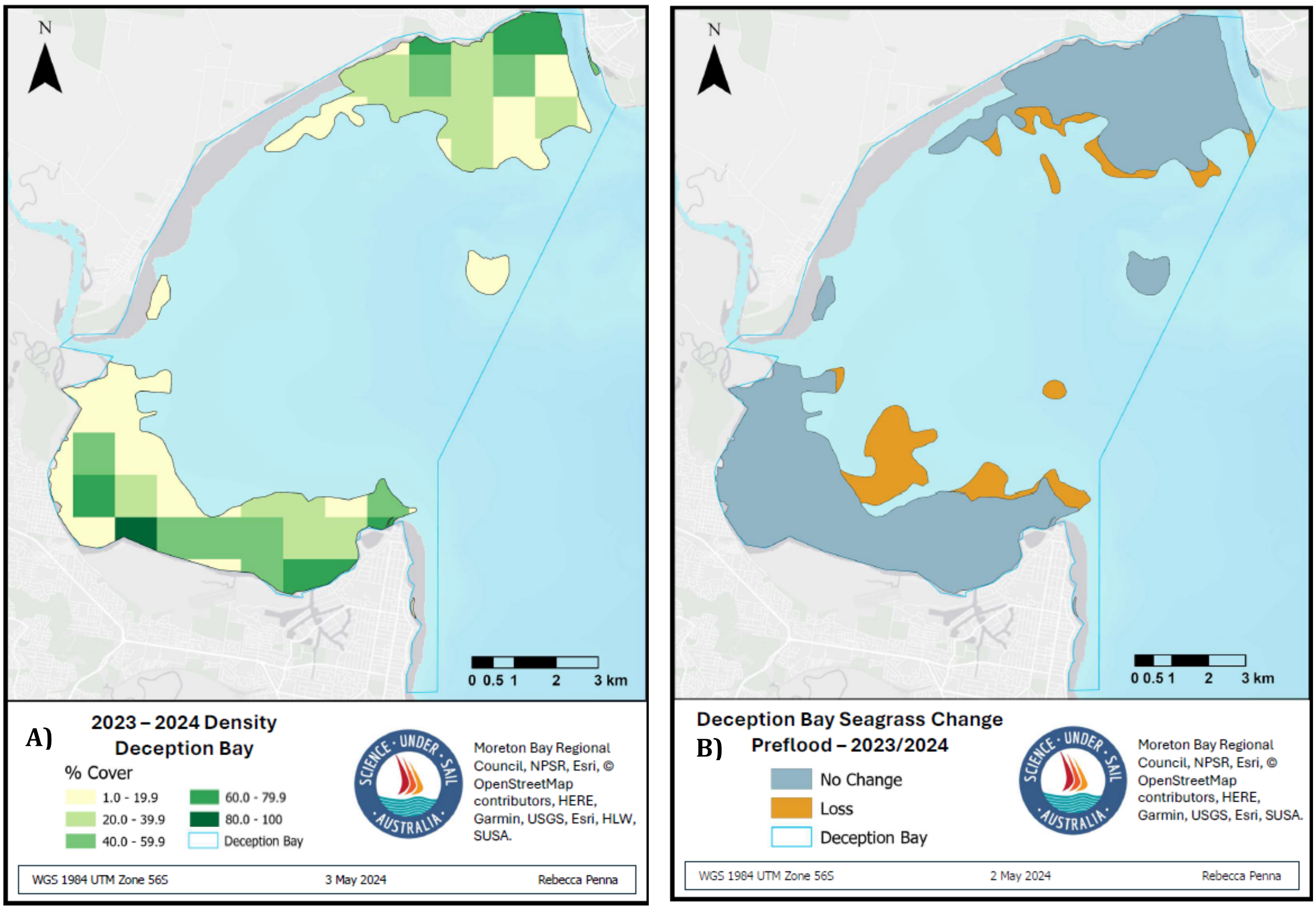
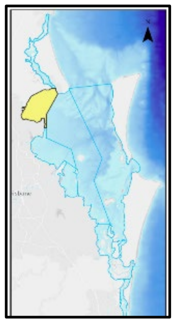


Figure 27. Areas in Deception Bay where seagrass (all species) was present or likely to occur. A) Average density of seagrass cover in 2023/24 within a 1km² pixel (second year post-flood), darker green indicates higher density (% cover) B) Changes in seagrass distribution between pre-flood (2015-21) and the second year post-flood (2023/24).

The overall proportion of sites with seagrass observed was higher in the second year post flood than pre- flood across all depth zones, however this is mainly attributable to an increase in Very Sparse seagrass. In the Intertidal zone, *Dense ZMHU* and *Moderate ZMHU with HO* meadows became more dominant post-2022 flood, with the proportion of sites where seagrass meadows exist increasing in the second year post-flood (2023/24) (Figure 28). *SI* meadows however have not been observed in the Intertidal zone since the flood. In the Shallow Subtidal *Moderate HS, Sparse HS and SI* meadows declined in the first year post-flood, replaced by *Very Sparse* seagrass. By the second year post-flood, *Moderate HS, Sparse HS and SI* meadows had recovered, with the total proportion of sites with seagrass in the Shallow Subtidal zone greater than pre-flood. Dense ZMHU had also recovered to greater than pre-flood proportions by the second year post-flood. In the Deep Subtidal *Moderate HS, Sparse HS, CS and Dense ZMHU* disappeared following the 2022 flood, however the proportion of *HO* and *Moderate ZMHU with HO* meadows remained stable. In the second year post-flood, the proportion of *HO* meadows had declined, with *Very Sparse* seagrass becoming the dominant meadow type in the Deep Subtidal.



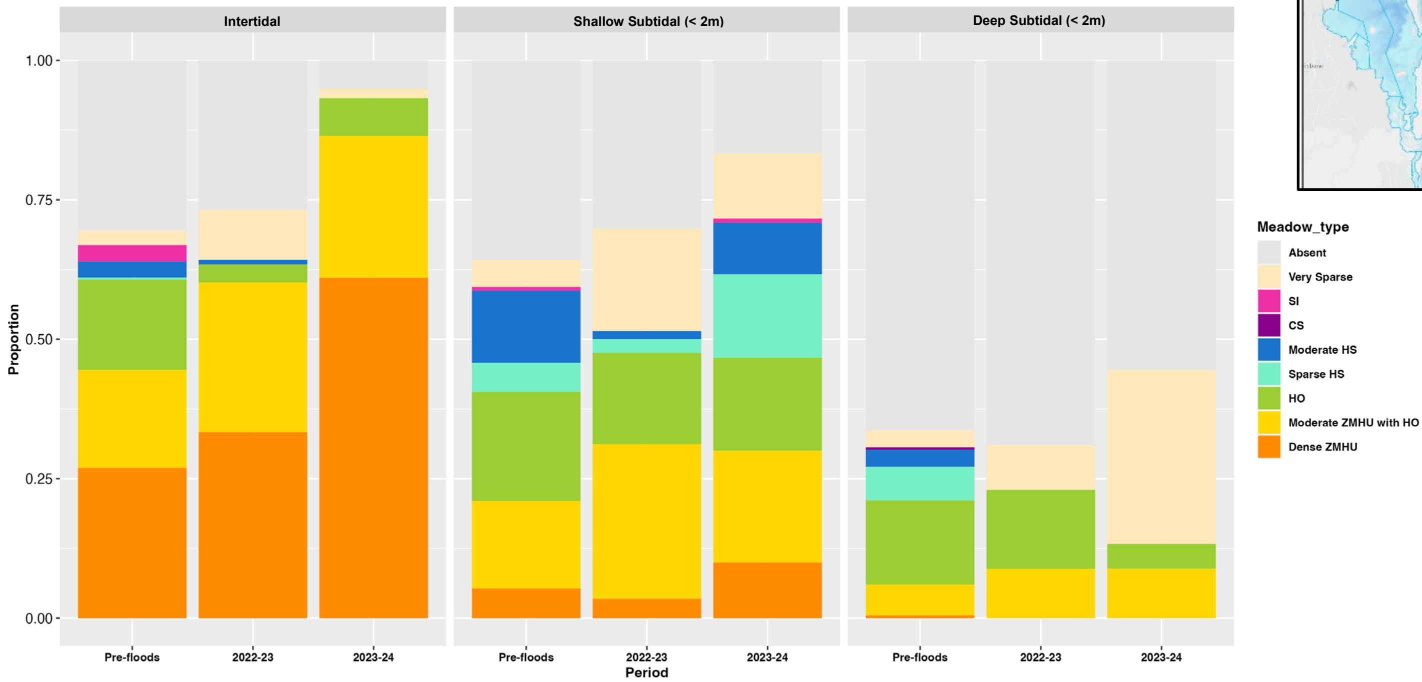


Figure 28. Comparison of the proportion of meadow types observed within the sample area in Deception Bay between the pre-flood, 2022/23 and 2023/24 survey periods.

Where *H. spinulosa* was observed, its density declined in all depth zones post 2022 flood and had not recovered by the second year post flood (2023/24) (Figure 29). The density of *H. ovalis* did not change in any depth zone in the first year following the 2022 flood, however in the Deep Subtidal zone *H. ovalis* was significantly sparser in the second year post-flood. Z*. muelleri/ H. uninervis* density was not significantly affected by the flood in the Intertidal and Shallow Subtidal zones. In the Deep Subtidal zone however Z*. muelleri/ H. uninervis* was significantly sparser in the first year post flood, and the density has continued to decline in the second year post-flood.

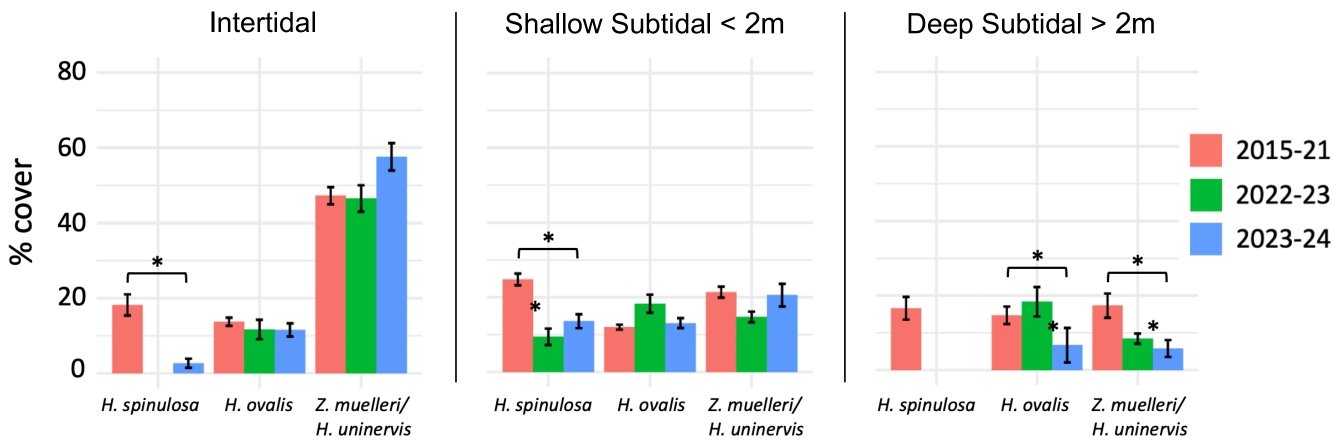
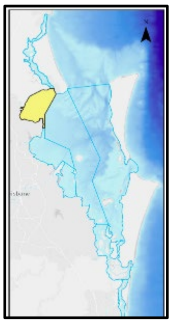


Figure 29. Change in seagrass density (% cover) between pre-flood, 2022/23 and 2023/24 in Deception Bay. * Indicates significant difference between adjacent years on bar graph. * Over bracket represents significant difference between 2015/21 and 2023/24.



The average and maximum depth of *H. spinulosa* and *H. ovalis* did not change in the first year post-flood (2022/23) in Deception Bay (Figure 30). However, in the second year post-flood (2023/24) the average and maximum depth of *H. spinulosa* was significantly shallower than pre-flood, and the average and maximum depth of *H. ovalis* was shallower than in 2022/23. The average depth of *Z. muelleri / H. uninervis* became slightly deeper in the two sampling years post-flood, in comparison to pre-flood depth.

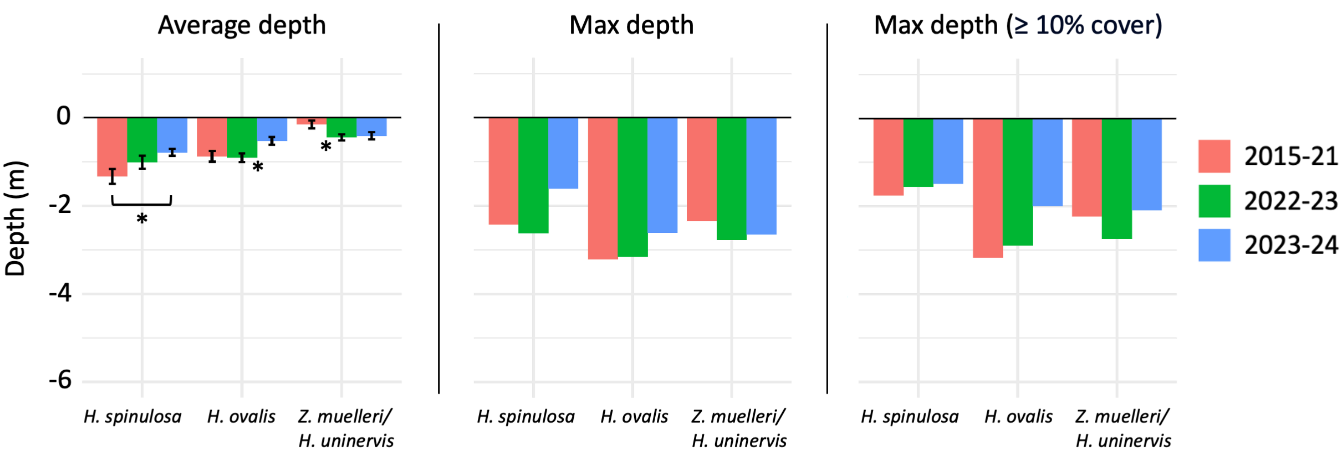
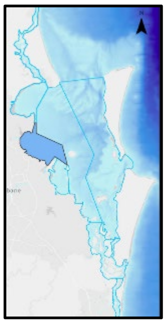


Figure 30. Changes in seagrass average depth, maximum depth and maximum depth when density ≥ 10% cover for each species in Deception Bay. Comparing 2020/21, 2022/23 and 2023/24 depth distributions. * Indicates significant difference between adjacent years on bar graph. * Over bracket represents significant difference between 2015/16 and 2023/24.

Bramble Bay



Due to its proximity to the mouth of the Brisbane River, Bramble Bay receives large amounts of mud from the Brisbane River catchment during floods and periods of heavy rainfall. Bramble Bay had only approximately 9km^2 of seagrass pre-flood, mainly occurring along its northern shoreline, adjacent to Woody Point (Figure 26A). Intermittent seagrass patches have been observed along the Sandgate foreshore, with seagrass density and frequency of occurrence decreasing as you move towards the Brisbane River.

There was no observed change in seagrass areal extent in Bramble Bay in the first year post flood (2022/23), however this is possibly due to limited pre-flood sampling in southern Bramble Bay. A significant decline in seagrass areal extent occurred during the second year post-flood with approximately 3km^2 loss observed in 2023/24 compared to 2022/23. Despite the inability to quantify an accurate reduction in areal seagrass extent in Bramble Bay between pre-flood and 2022/23 (due to pre-flood sampling occurring only in northern Bramble Bay), there does appear to have been a significant flood effect with the proportion of sites in Bramble Bay where seagrass was observed declining in all depth zones post-2022 flood within the temporal study area (Figure 31).

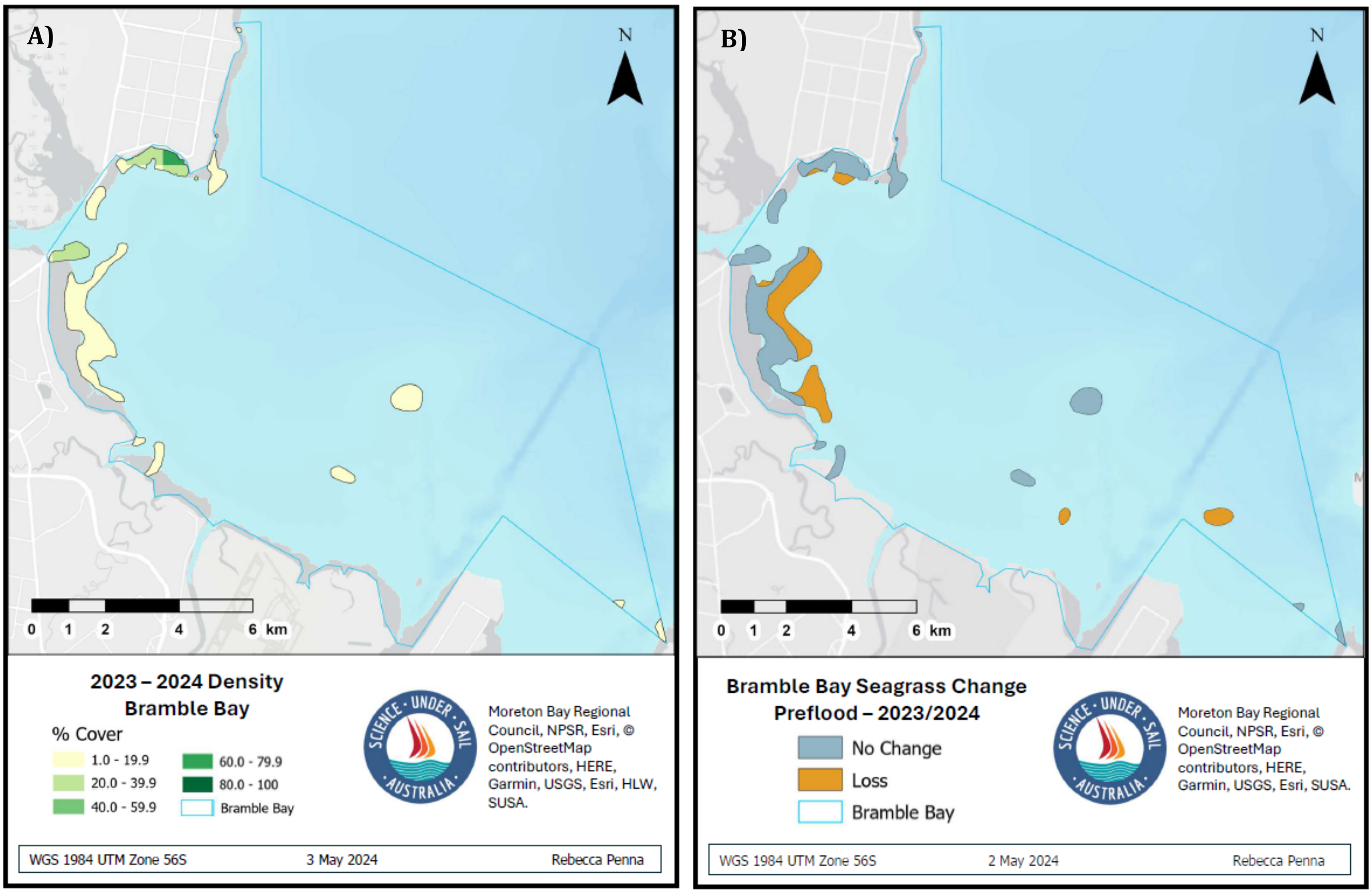
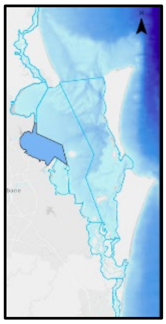


Figure 31. Areas in Bramble Bay where seagrass (all species) was present or likely to occur. A) Average density of seagrass cover in 2023/24 within a 1km² pixel (second year post-flood), darker green indicates higher density (% cover) B) Changes in seagrass distribution between pre-flood (2015-21) and the second year post-flood (2023/24).

In the Intertidal zone only *Very Sparse* seagrass and *Moderate ZMHU* with *HO* were observed in the first year post-flood (2022/23), with *Dense ZMHU* and *HO* meadows recovering by the second year post-flood (2023/24). *Moderate HS* and *Sparse HS* were not observed post-flood in the Intertidal zone. In the Shallow Subtidal zone only *HO* and *Very Sparse* meadows were observed in the first year post-flood. *Dense ZM* and *Moderate ZM with HO* had recovered to pre-flood levels by 2023/24, however *Moderate HS* had not returned and there was only slight recovery of *Sparse HS*. In the Deep Subtidal zone *Moderate HS*, *Moderate ZMHU with HO* and *Dense ZMHU* were not observed post-flood. The proportion of sites with seagrass has remained lower than pre-flood, with *HO* and *Very Sparse* now the dominant meadow type.



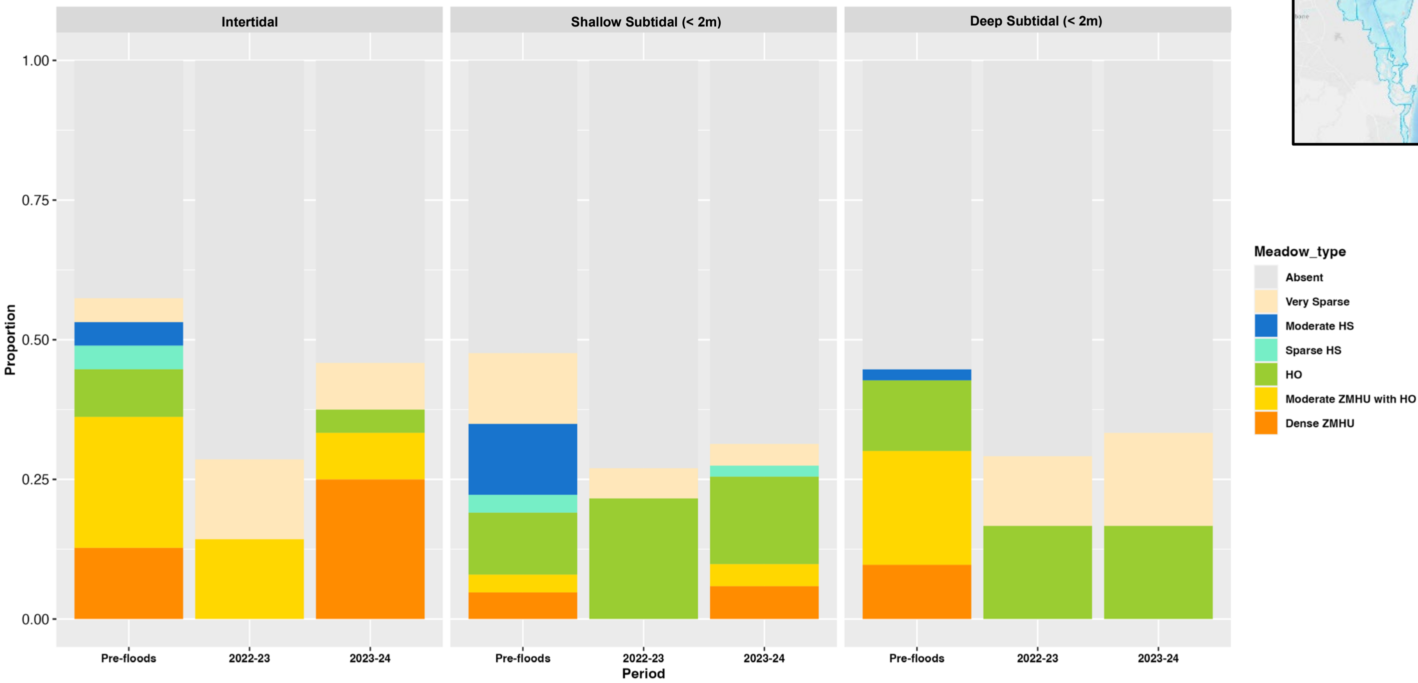


Figure 32. Comparison of the proportion of meadow types observed within the sample area in Bramble Bay between the pre-flood, 2022/23 and 2023/24 survey periods.

Where *H. ovalis* had recovered in the Intertidal zone by the second year post-2022 flood (2023/24) it was significantly sparser than pre-flood (Figure 33). In the Shallow Subtidal zone *H. ovalis* density was not impacted in the first year post-flood, however it was significantly sparser in the second year post-flood. There was insufficient observations of *Z. muelleri/ H. uninervis* and *H. spinulosa* post-flood to enable a comparison of density.

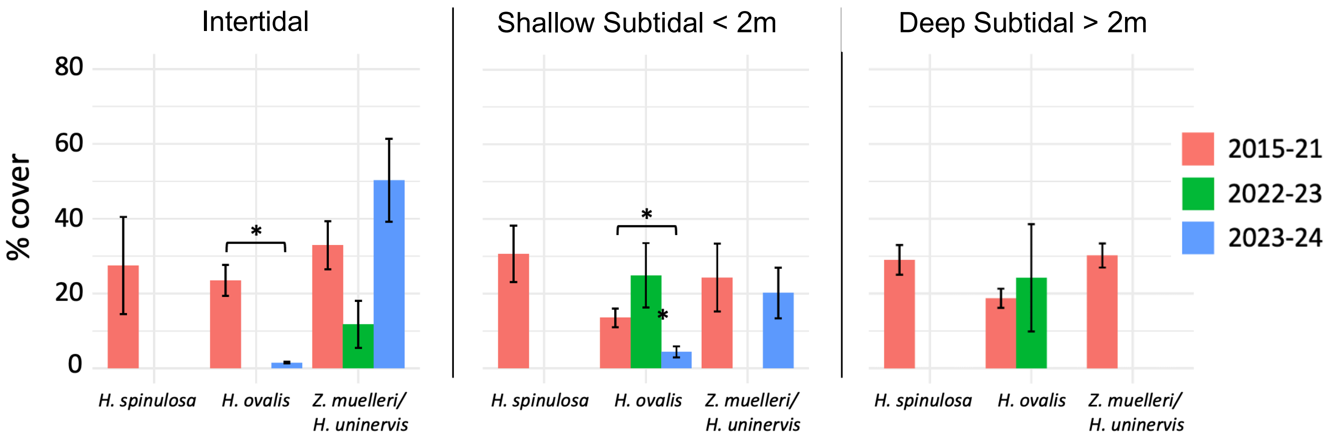
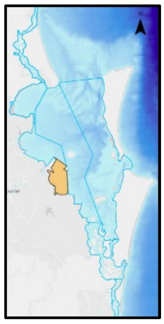


Figure 33. Change in seagrass density (% cover) between pre-flood, 2022/23 and 2023/24 in Bramble Bay. * Indicates significant difference between adjacent years on bar graph. * Over bracket represents significant difference between 2015/21 and 2023/24. * Indicates significant difference between adjacent years on bar graph.



Waterloo Bay

Seagrass covers the majority of Waterloo Bay, with an areal extent of 49km^2 (Figure 34). The intertidal zone around its edge had moderate to dense seagrass meadows dominated by *Zostera muelleri*, while the deeper areas transitioned to *H. spinulosa* and *H. ovalis* meadows. Waterloo Bay exhibited the smallest impact of the 2022 flood in areal extent of seagrass, with only a 0.6km^2 decline in the first year post-flood (2022/23) and no extra loss in the second year (2023/24) (Figure 34B).

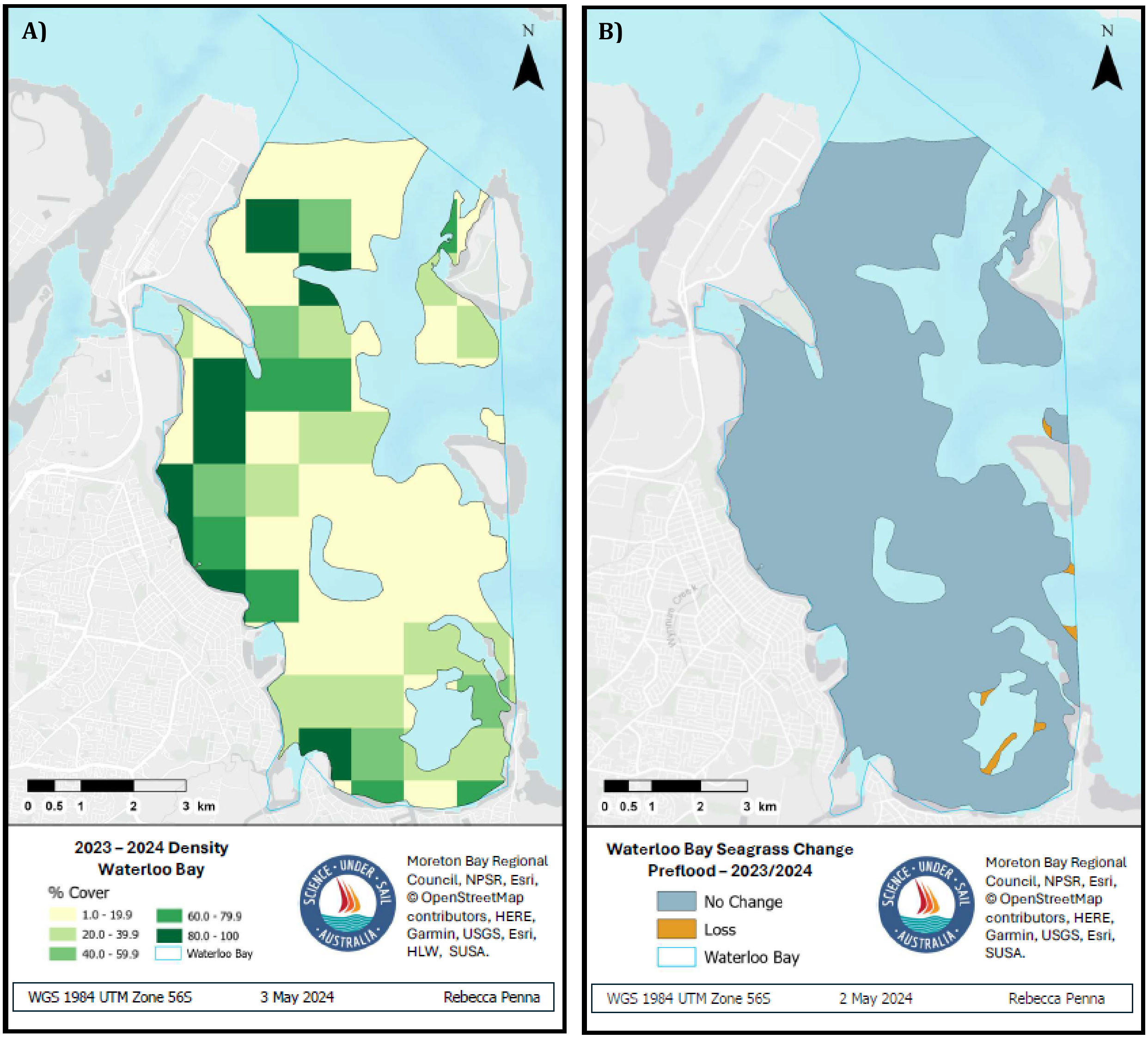
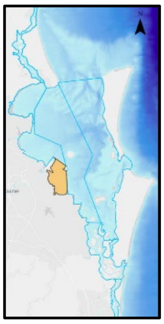


Figure 34. Areas in Waterloo Bay where seagrass (all species) was present or likely to occur. A) Average density of seagrass cover in 2023/24 within a 1km^2 pixel (second year post-flood), darker green indicates higher density (% cover) B) Changes in seagrass distribution between pre-flood (2015-21) and the second year post-flood (2023/24).

In Waterloo Bay the overall proportion of sites with seagrass within the temporal study area is now higher than it was pre-flood. This is mainly attributable to an increase in *HO* and *ZM* meadows in the shallow subtidal and Very Sparse seagrass in the deep subtidal (Figure 35). In the Intertidal zone there was minimal impact



from the 2022 flood on the overall proportion of sites with seagrass (Figure 35). However, in the first year post-flood (2022/2023) the proportion of *Dense ZMHU*, *Sparse HS* and *Moderate HS* declined, partially replaced by *HO* and *Moderate ZMHU with HO*. By the second year post-flood, *Dense ZMHU* had recovered in the Intertidal zone and occurred at a higher proportion of sites than pre-flood. In the Shallow Subtidal zone, the total proportion of sites with seagrass has increased every year since the flood, though the composition has altered. In the first year post-flood *HO* meadows had replaced some *Dense ZMHU* and *Moderate HS* meadows. However, by the second year post flood, *Dense ZMHU* and *Moderate HS* had recovered to pre-flood levels, with the overall proportion of sites with seagrass now higher than pre-flood. The Deep Subtidal zone was the most impacted depth zone in Waterloo Bay, with *Moderate HS* significantly declining in the first year post-flood, replaced by *Very Sparse* and a slight increase in *HO*. By the second year post-flood, *Moderate HS* had recovered, and a high proportion of *Very Sparse* seagrass remained.

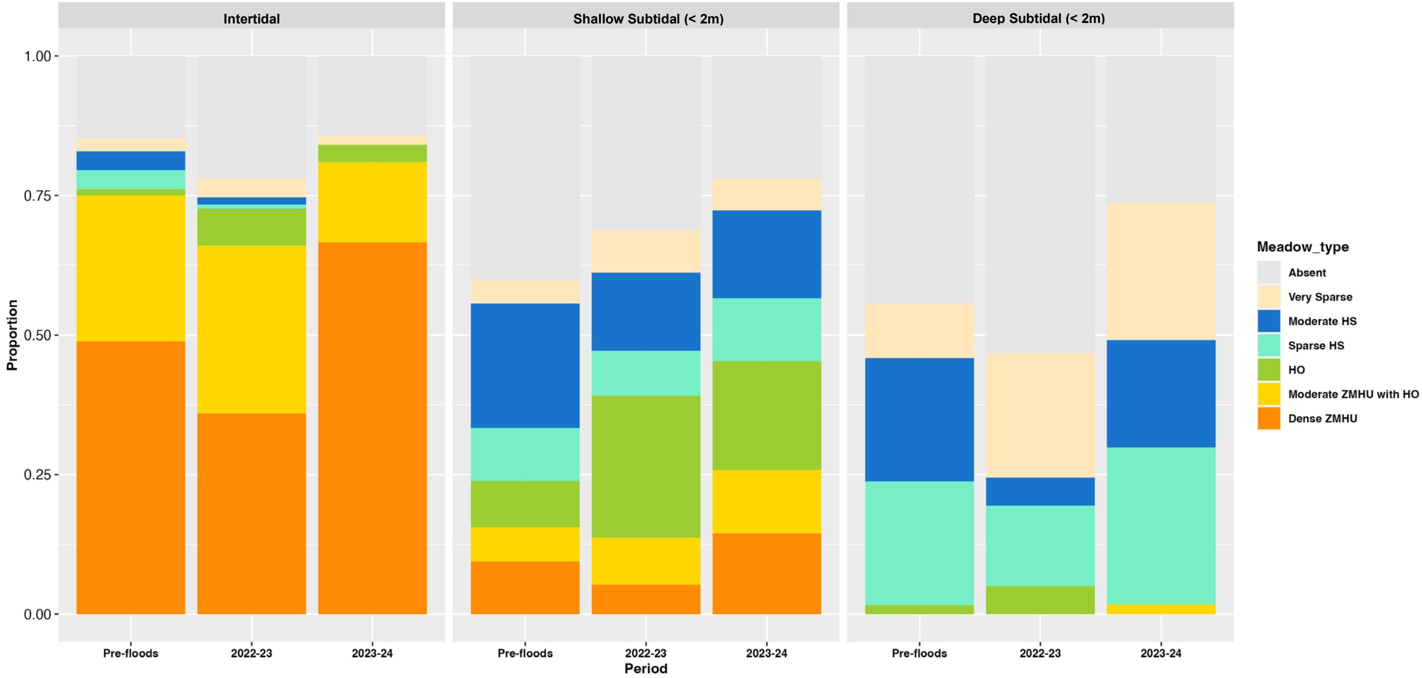
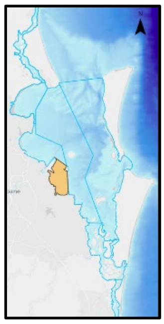


Figure 35. Comparison of the proportion of meadow types observed within the sample area in Waterloo Bay between the pre-flood, 2022/23 and 2023/24 survey periods.

Where *H. spinulosa* was observed in the Shallow and Deep Subtidal zones it has been significantly sparser in both survey years post-2022 flood (Figure 36). *H. spinulosa* density did not significantly change in the Intertidal. *H. ovalis* density has responded differently across the depth zones in Waterloo Bay. In the Shallow Subtidal zone, *H. ovalis* was most dense in the first year post-flood, and by 2023/24 had returned to pre-flood densities. In the Deep Subtidal zone *H. ovalis* has been sparser since the 2022 flood. *Z. muelleri/ H. uninervis* density increased in the second year post-flood in the Intertidal zone, compared to pre-flood. In the Shallow Subtidal *Z. muelleri/ H. uninervis* density declined in the first year post-flood but had recovered by the second year.



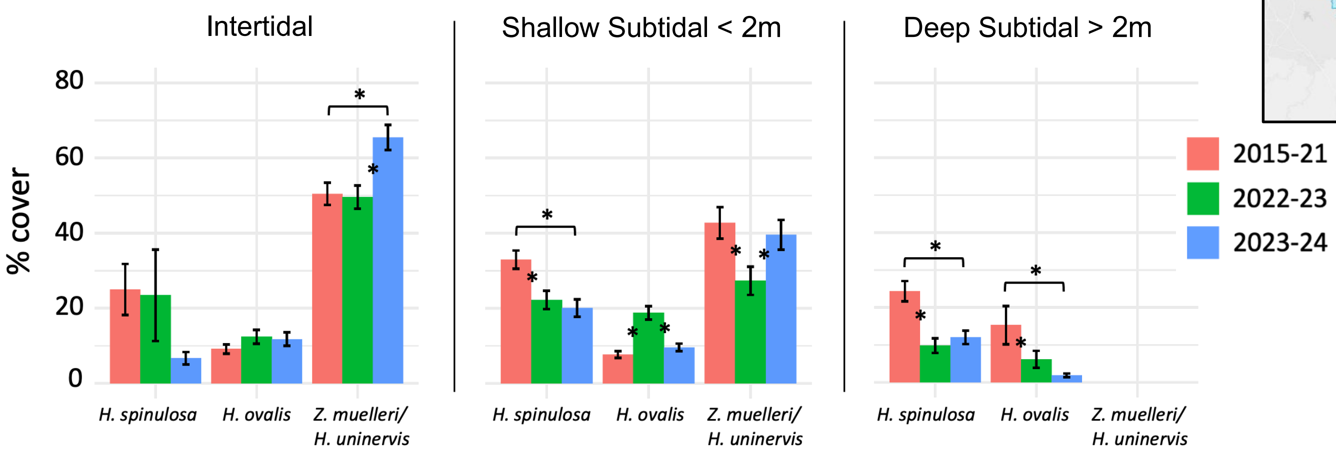


Figure 36. Change in seagrass density (% cover) between pre-flood, 2022/23 and 2023/24 in Waterloo Bay. * Indicates significant difference between adjacent years on bar graph. * Over bracket represents significant difference between 2015/21 and 2023/24.

The depth range of *H. spinulosa* in Waterloo Bay did not change significantly between years (Figure 37). The maximum observed depth of *H. ovalis*, and *H. ovalis* with ≥ 10% cover was deepest in the first year post-flood, however by the second year post-flood, the average depth of *H. ovalis* was shallower than both pre-flood and 2022/23. *Z. muelleri/ H. uninervis* grew shallower on average post-2022 flood in Waterloo Bay, though the maximum depth of sparse observations remained similar.

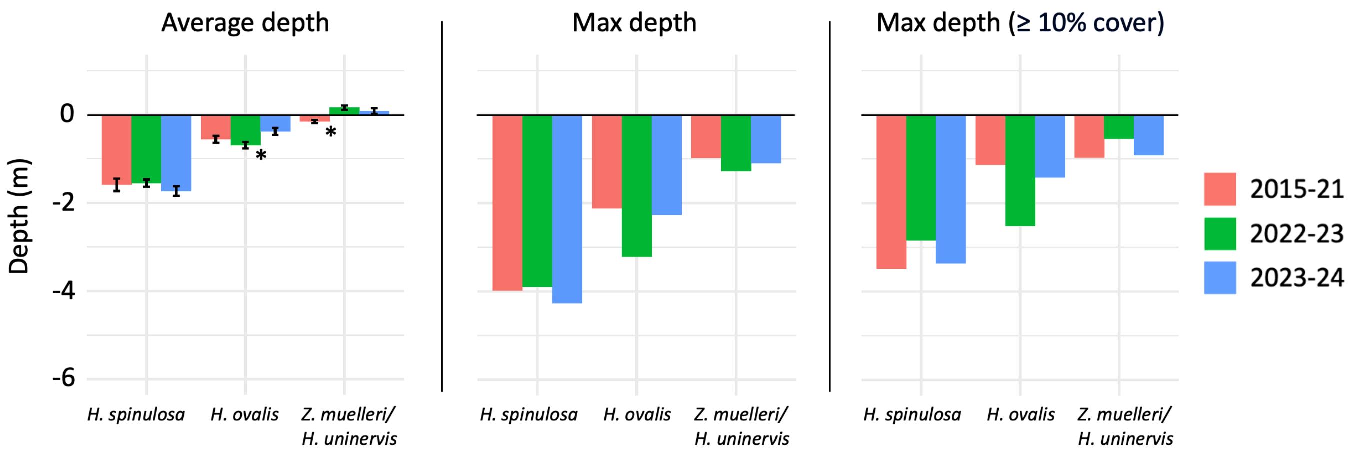
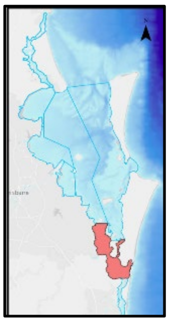


Figure 37. Changes in seagrass average depth, maximum depth and maximum depth when density ≥ 10% cover for each species in Waterloo Bay. Comparing 2020/21, 2022/23 and 2023/24 depth distributions. * Indicates significant difference between adjacent years on bar graph. * Over bracket represents significant difference between 2015/16 and 2023/24.



Southern Bay

Seagrass in Southern Bay is heavily impacted by run-off from the Logan River, with a seagrass 'Dead Zone' surrounding the River mouth. This 'Dead Zone' expands and contracts in response to past weather and run-off events, always expanding after floods or heavy rainfall in the catchment. Seagrass meadows in this region were estimated to cover 15km^2 pre 2022 flood (most of the data was collected in 2015). Post-flood there was an approximately 3km^2 reduction in 2022/23 and a 1km^2 recovery in 2023/24. In 2023/24 seagrass began to be present at the northern end of Long Island in the Intertidal zone (Figure 38). As you moved north and further away from the Logan River seagrass generally increases its depth distribution and density. *Z. muelleri/ H. uninervis* was the dominant species in the Intertidal zone, and *H. spinulosa* in the Shallow Subtidal zone.

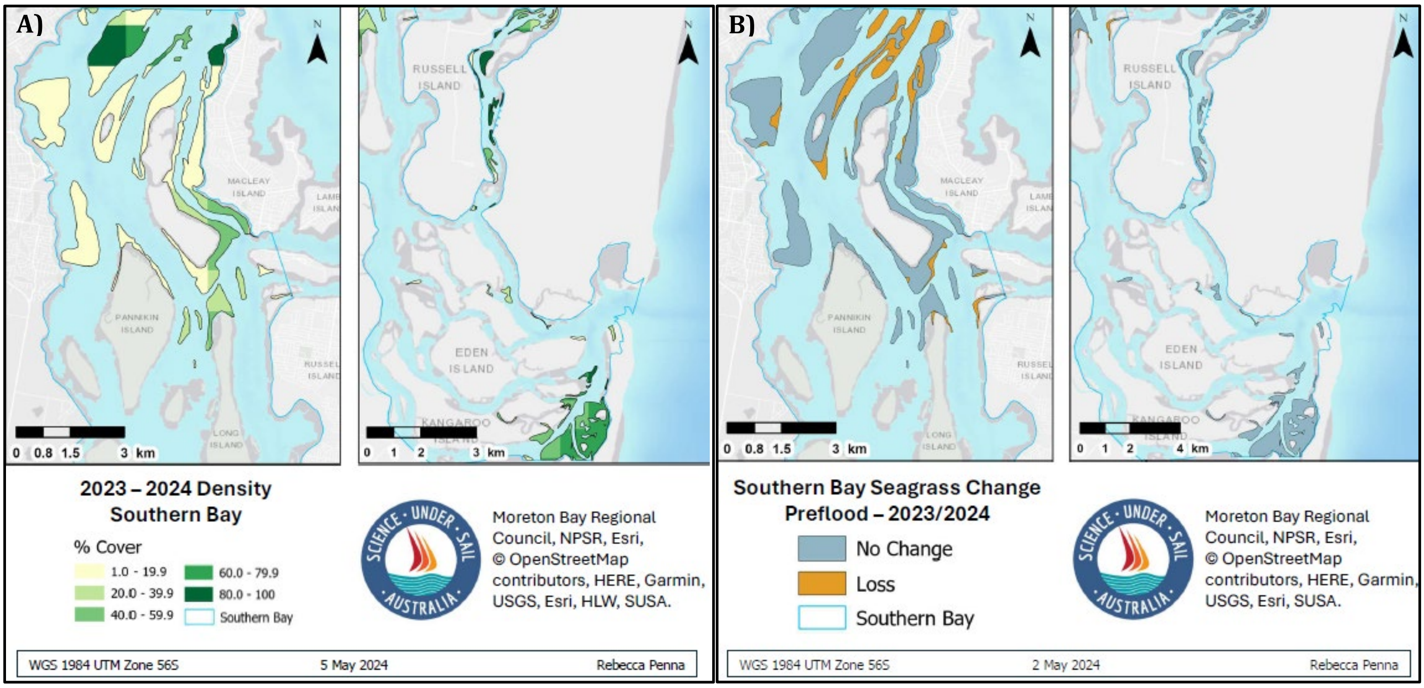
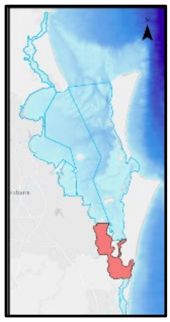


Figure 38. Areas in Southern Bay where seagrass (all species) was present or likely to occur. A) Average density of seagrass cover in 2023/24 within a 1km^2 pixel (second year post-flood), darker green indicates higher density (% cover) B) Changes in seagrass distribution between pre-flood (2015-21) and the second year post-flood (2023/24).

The majority of seagrass in the Southern Bay occurs within the Intertidal zone where the proportion of meadow types has remained relatively stable within the temporal study area since pre-flood (Figure 39). *Dense ZMHU* meadows have partially replaced *Moderate ZMHU with HO* meadows in the Intertidal zone post-flood. In the Shallow and Deep Subtidal zones the total proportion of sites with seagrass present declined post-flood, with little sign of recovery by the second year post-flood (2023/24). Moderate HS and Sparse HS accounted for most of the loss in the Shallow and Deep Subtidal zones. No seagrass was observed in the Deep Subtidal zone post-flood.



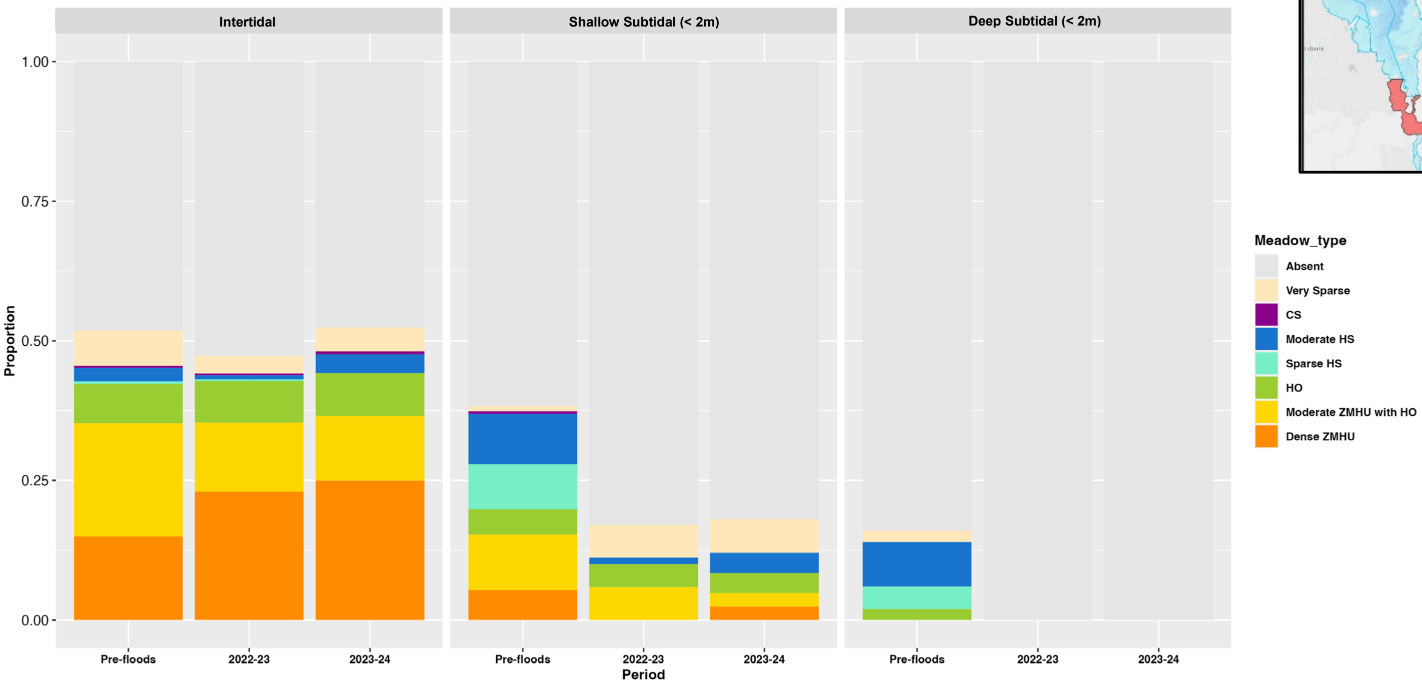


Figure 39. Comparison of the proportion of meadow types observed within the sample area in Southern Bay between the pre-flood, 2022/23 and 2023/24 survey periods.

Where *H. spinulosa* was present in the Intertidal and Shallow Subtidal zones there has been no change in its density (Figure 40). In the Intertidal zone, when *H. ovalis* and *Z. muelleri/ H. uninervis* occur they have been significantly denser since the 2022 flood. However, in the Shallow Subtidal zone *Z. muelleri/ H. uninervis* have been significantly sparser since the 2022 flood. There has been no significant change in the density of *H. ovalis* in the Shallow Subtidal.

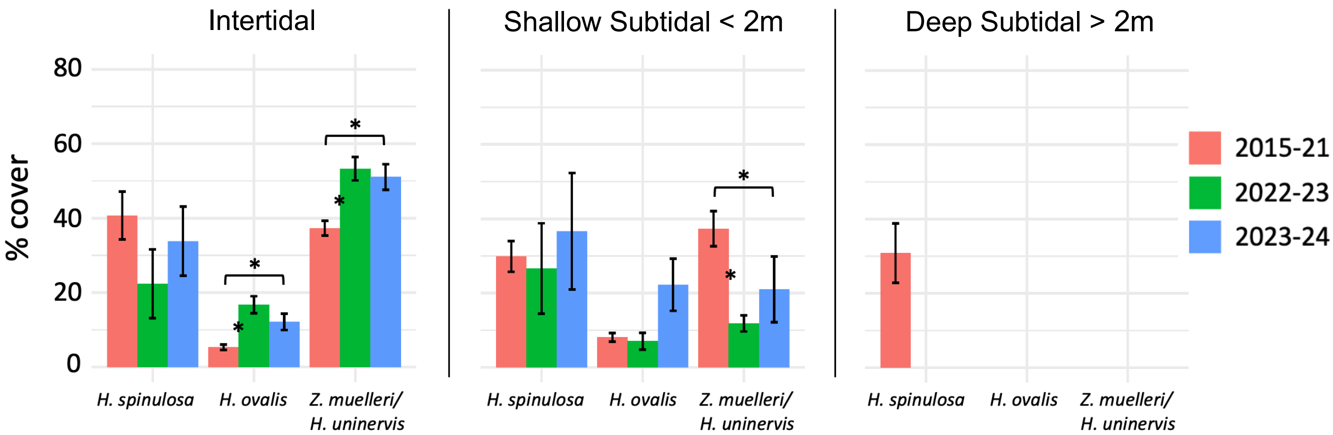
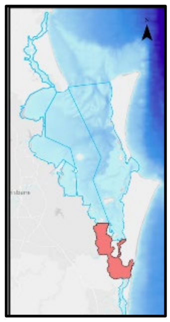


Figure 40. Change in seagrass density (% cover) between pre-flood, 2022/23 and 2023/24 in Southern Bay. * Indicates significant difference between adjacent years on bar graph. * Over bracket represents significant difference between 2015/21 and 2023/24.



Seagrass in Southern Bay had a much shallower depth distribution than other regions prior to the 2022 flood, with majority of Southern Bay seagrass occurring in the Intertidal zone (Figure 41). The average depth of *H. spinulosa* has not changed since 2015/16. The average and maximum observed depth of *H. ovalis* ≥ 10% cover in Southern Bay was deeper in the first year post-2022 flood than in 2015/16, however the maximum observed depth had been deeper in 2015/16. The average depth of *Z. muelleri/ H. uninervis* did not change in the first year post-flood but was shallower in the second year post-flood than in 2015/16 and 2022/23.

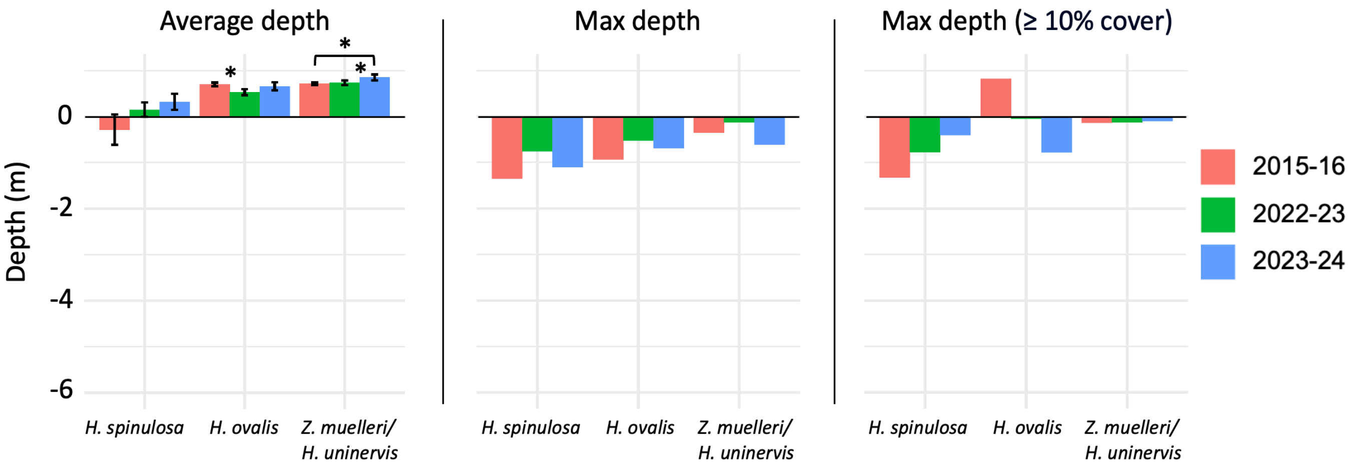
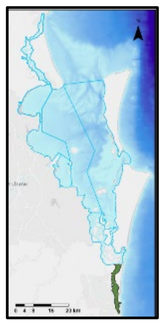


Figure 41. Changes in seagrass average depth, maximum depth and maximum depth when density ≥ 10% cover for each species in Southern Bay. Comparing 2015/16, 2022/23 and 2023/24 depth distributions. * Indicates significant difference between adjacent years on bar graph. * Over bracket represents significant difference between 2015/16 and 2023/24.

Broadwater



The Broadwater has extensive seagrass meadows near the Gold Coast Seaway and along the Western side of South Stradbroke Island, with a total areal extent of 6.2km^2 (Figure 42). As you move away from the clean ocean water towards the Coomera River the seagrass depth distribution declines, but seagrass remains present in the Intertidal zone. No seagrass loss was observed in the post-flood sampling, when compared with 2015 observations (Griffith University/ James Cook University). However, as majority of seagrass meadows in the Broadwater grow in the Intertidal and Shallow Subtidal zone and there are often steep sided channels with strong tidal currents at the edge of seagrass meadows it is not surprising that there is considerable consistency between meadows that had seagrass in 2015 and the two years of post-flood sampling covered in this report (2022/23 & 2023/24). *Z. muelleri/ H. uninervis* was the dominant species in the Intertidal zone, and *H. ovalis* in the Shallow Subtidal zone.

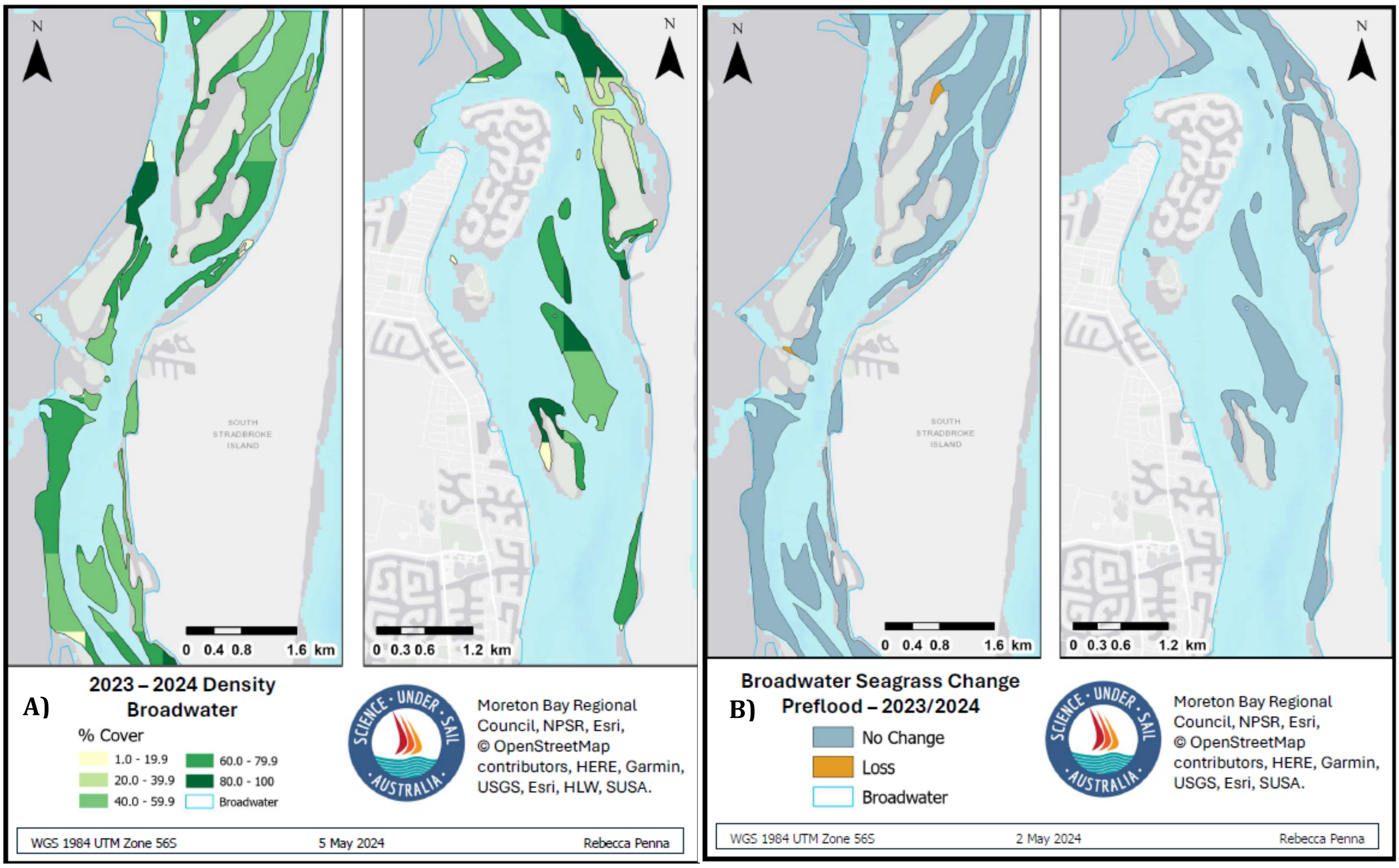
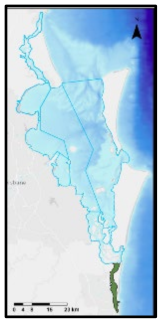


Figure 42. Areas in Broadwater where seagrass (all species) was present or likely to occur.

A) Average density of seagrass cover in 2023/24 within a 1km² pixel (second year post-flood), darker green indicates higher density (% cover) B) Changes in seagrass distribution between pre-flood (2015-21) and the second year post-flood (2023/24).

The proportion of sites with seagrass in the Broadwater Intertidal has been increasing since the 2022 flood (Figure 43). The proportion of *Dense ZMHU* and *HO* in the Intertidal increased in the first year post flood, while the proportion of *Very Sparse* and *Moderate ZMHU with HO* decreased. In the second year post-flood there were further increases in the proportion of *HO, Sparse HS* and *CS* meadows. In the Shallow Subtidal zone *Sparse HS, Very Sparse* and *Moderate ZMHU with HO* meadows declined in the first year post-flood. In the second year post-flood *HO* and *Dense ZMHU* meadows increased to occur more frequently than pre-flood, and there was also recovery of *Moderate HS* and *Sparse HS* meadows. *CS* was only observed pre-flood. In the Deep Subtidal zone seagrass had already been rare pre-flood. In the first year post-flood rare *HO* meadows still existed, and there was an increase in *Very sparse* seagrass. By the second year post-flood *Very Sparse* was the only meadow type in Deep Subtidal zone.



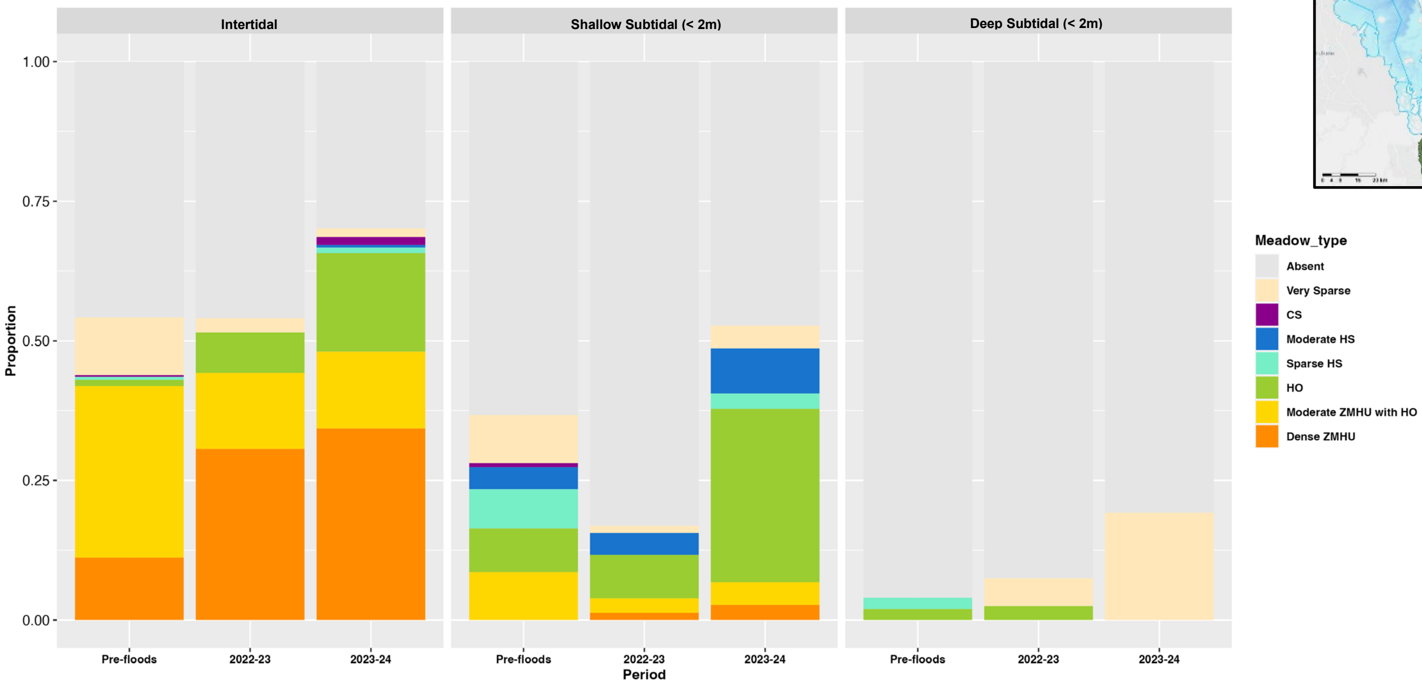


Figure 43. Comparison of the proportion of meadow types observed within the sample area in Broadwater between the pre-flood, 2022/23 and 2023/24 survey periods.

Where seagrass is present in Broadwater its density (% cover) has generally increased since the 2022 flood (Figure 44). *H. spinulosa* density in the Intertidal zone did not significantly increase between pre-flood and the first year post-flood, but in the second year post-flood it was significantly denser than pre-flood. *H. ovalis* density increased in the Intertidal and Shallow Subtidal zones in the first year post-flood and continued to get denser in the second year post-flood. *Z. muelleri/ H. uninervis* in the Intertidal and Shallow Subtidal zones increased in density in the first year post-flood and has remained denser than pre-flood observations in the second year post-flood. There were not enough seagrass observations in the Deep Subtidal to analyse changes in density.

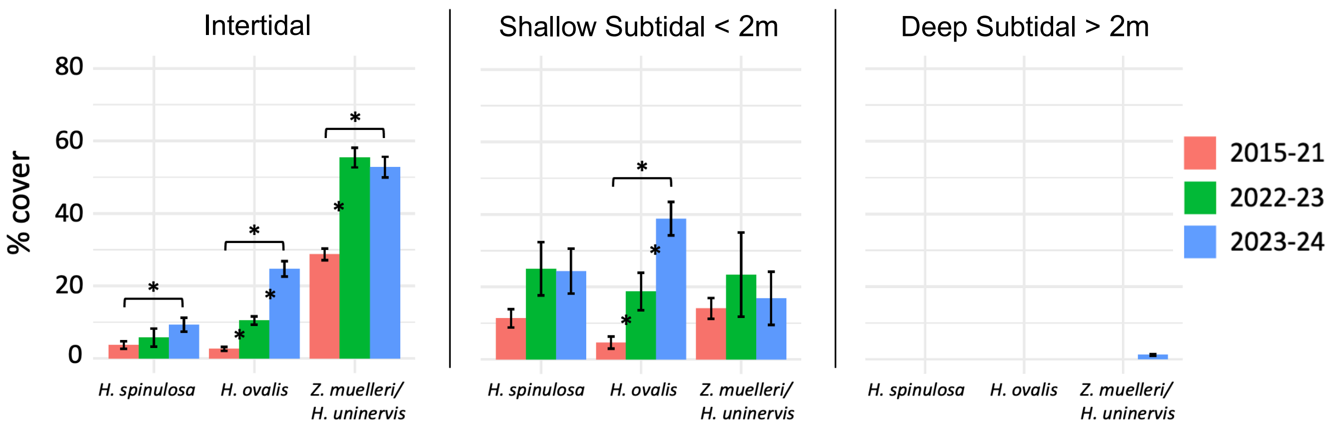
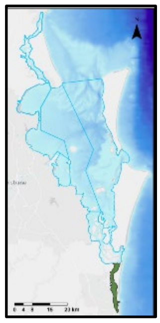


Figure 44. Change in seagrass density (% cover) between pre-flood, 2022/23 and 2023/24 in Broadwater. * Indicates significant difference between adjacent years on bar graph. * Over bracket represents significant difference between 2015/21 and 2023/24.



Similarly to Southern Bay, seagrass in Broadwater had a much shallower depth distribution than other regions prior to the 2022 flood, with majority of seagrass occurring in the Intertidal zone (Figure 45). *H. spinulosa* depth distribution remained similar to 2015/16 in the first year post-2022 flood (2022/23) however became significantly shallower in the second year post-flood (2023/24). The maximum observed depth of *H. ovalis* and maximum depth of *H. ovalis* ≥ 10% cover, has gotten shallower since 2015/16 though there has been no significant change in the average depth. *Z. muelleri/ H. uninervis* depth range has shallowed since 2015/16 in Broadwater, although even pre-flood it was only rarely observed in the subtidal.

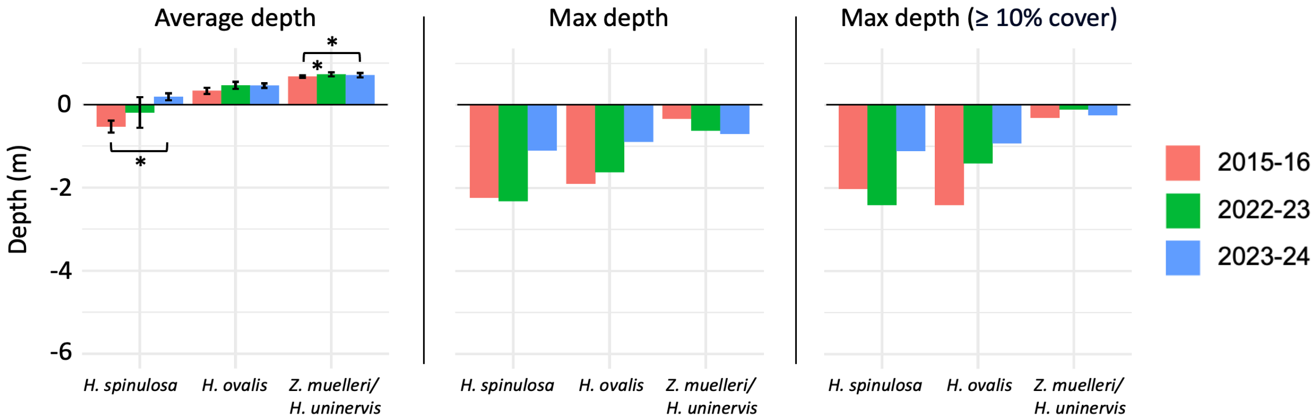


Figure 45. Changes in seagrass average depth, maximum depth and maximum depth when density ≥ 10% cover for each species in Broadwater. Comparing 2015/16, 2022/23 and 2023/24 depth distributions. * Indicates significant difference between adjacent years on bar graph. * Over bracket represents significant difference between 2015/16 and 2023/24.

Pumicestone Passage

Seagrass in Pumicestone Passage is absent or very sparse in the middle of the Passage in the vicinity of including Tripcony bight, with seagrass density and frequency of occurrence increasing at both ends of the Passage (Figure 46A). There is limited pre-flood data for Pumicestone Passage, especially in its middle section, and as such the apparent absence of loss in the pre-flood – 2023/24 areal extent comparison (Figure 46B) is likely a result of limited pre-flood sampling. Due to this we have also conducted a change analysis between historic 2002 data and the recent data set (Figure 46C). This analysis shows that considerable loss of seagrass has occurred in Pumicestone Passage during the past 20 years, with total seagrass in the Passage declining from approximately 13km^2 in 2002 to 9km^2 in 2023/24. Most of this loss (3.7km^2) occurred in the Tripcony Bight area of the Passage. As there have been three major floods (2011, 2013, 2022), and numerous smaller run-off events during this period it is impossible to know when this loss occurred. The patches of expansion present in the historic 2002 to current distribution comparison (Figure 46C) are likely only a result of DEM inaccuracies in defining the intertidal zone, as opposed to a real expansion of meadows.

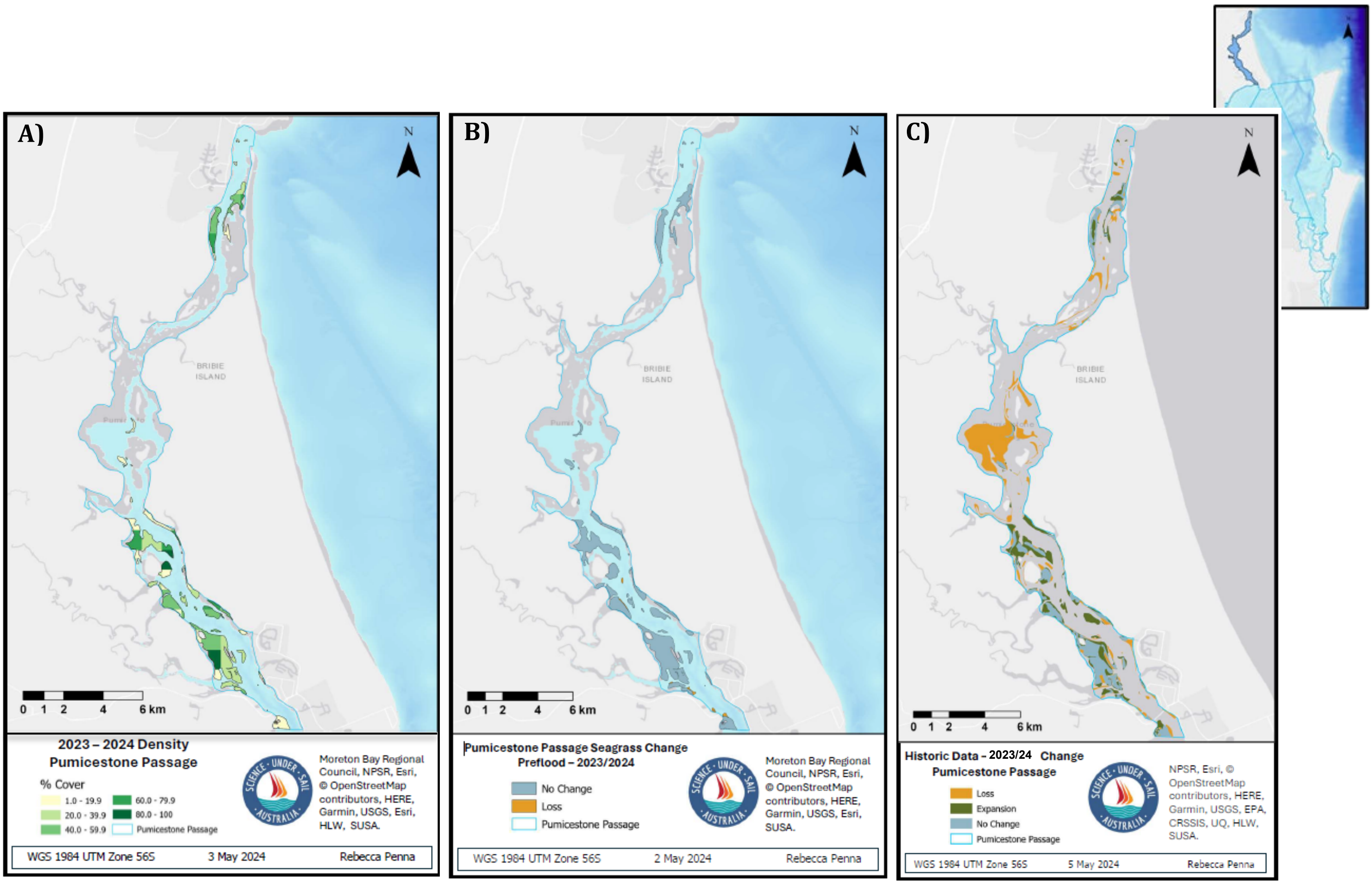


Figure 46. Areas in Pumicestone Passage where seagrass (all species) was present or likely to occur. A) Average density of seagrass cover in 2023/24 within a 1km^2 pixel (second year post-flood), darker green indicates higher density (% cover) B) Changes in seagrass distribution between pre-flood (2015-21) and the second year post-flood (2023/24) C) Changes in seagrass distribution between historic 2002 data and 2023/24.

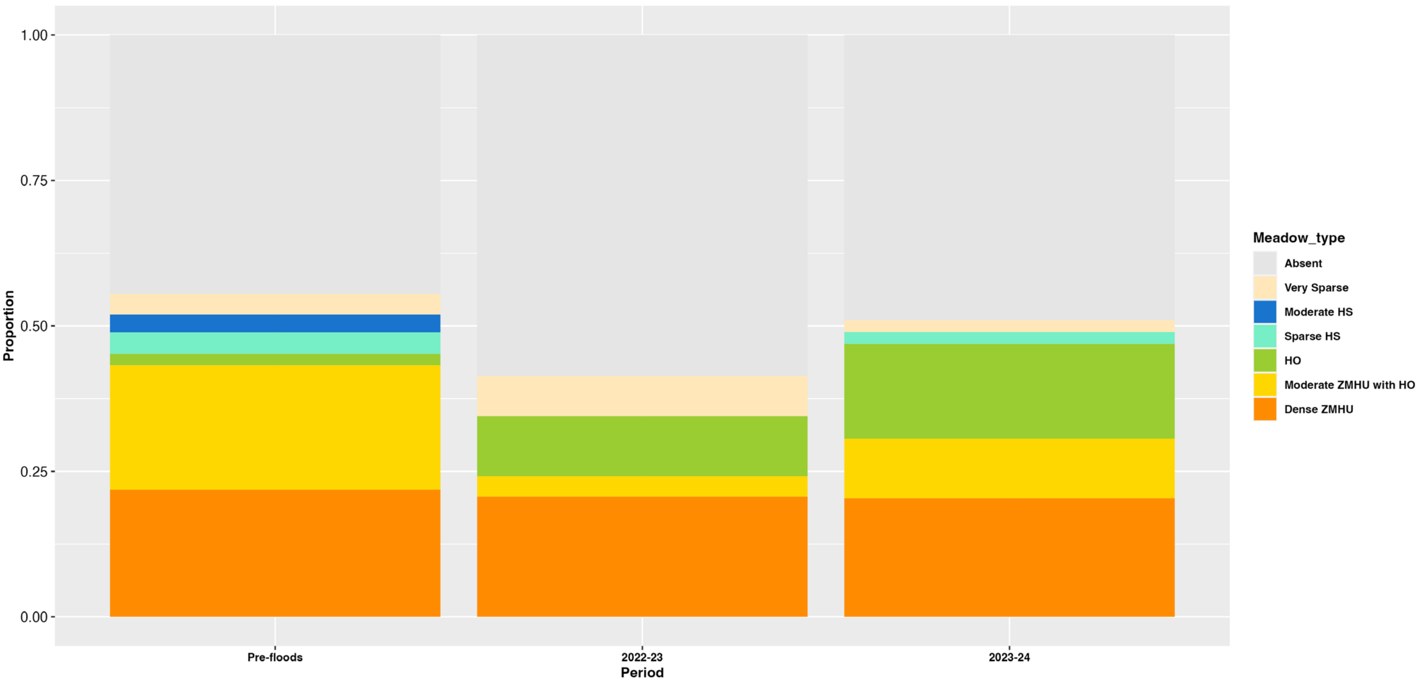


Figure 47. Comparison of the proportion of meadow types observed within the sample area in Pumicestone Passage between the pre-flood, 2022/23 and 2023/24 survey periods for all depth zones combined.

The proportion of *Dense ZMHU* meadows remained stable from pre-flood till the second year post-flood in the areas of the Passage where reliable pre-flood data existed for temporal statistical analysis (Figure 47). However, the proportion of *Moderate ZMHU with HO*, *Moderate HS* and *Sparse HS* declined after the 2022 flood, and there was a slight increase in the proportion of *HO* meadows. In the second year post-flood (2023/24) *HO* meadows became more common than pre-flood, and there was a slight recovery of *Sparse HS*. *Moderate HS* meadows however were not observed post-2022 flood.

Where *H. ovalis* was present, its density increased in the first year following the 2022 flood and maintained this higher density in the second year post-flood compared to pre-flood (Figure 48). When, *H. spinulosa* or *Z. muelleri/ H. uninervis* were present, there was no significant change in their density.

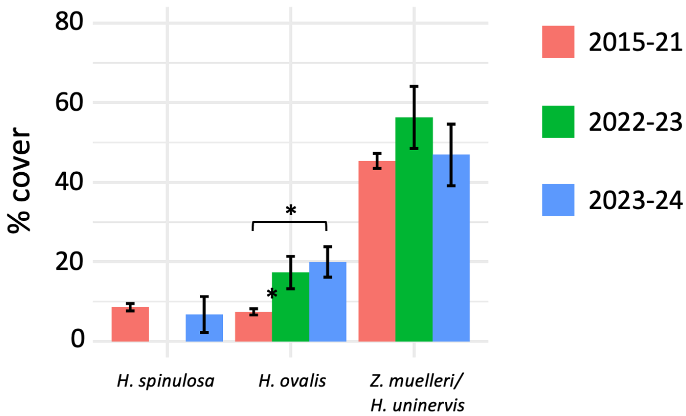


Figure 48. Change in seagrass density (% cover) between pre-flood, 2022/23 and 2023/24 in Pumicestone Passage. * Indicates significant difference between adjacent years on bar graph. * Over bar represents significant difference between 2015/21 and 2023/24.

Biomass and seed bank assessment

Seagrass biomass across Moreton Bay ranged from a maximum of 606gDWm^{-2} for a *C. serrulata* meadow on Wanga Wallen Banks (Wanga Wallen 2) to a minimum of 24gDWm^{-2} for a mixed *H. spinulosa* and *H. ovalis* meadow (Eastern Banks 1) (Figure 49). The largest variations occurred between meadows with different dominant species *C. serrulata* > *Z. muelleri* > *H. spinulosa*. However, there was also significant variation in total biomass and the above: below-ground ratio (A:B) of seagrass within the same species (Table 8). For example, *H. spinulosa* biomass ranged from a high of 96gDWm^{-2} with a 2.1A:B (Eastern Banks 2) to a low of 17gDWm^{-2} and a 0.7A:B (Eastern Banks 1). While the lowest A:B ratio of 0.15 was observed for an intertidal *Z. muelleri* meadow in the Southern Bay, with very small and thin leaves, but a substantial below-ground biomass of rhizomes and roots.

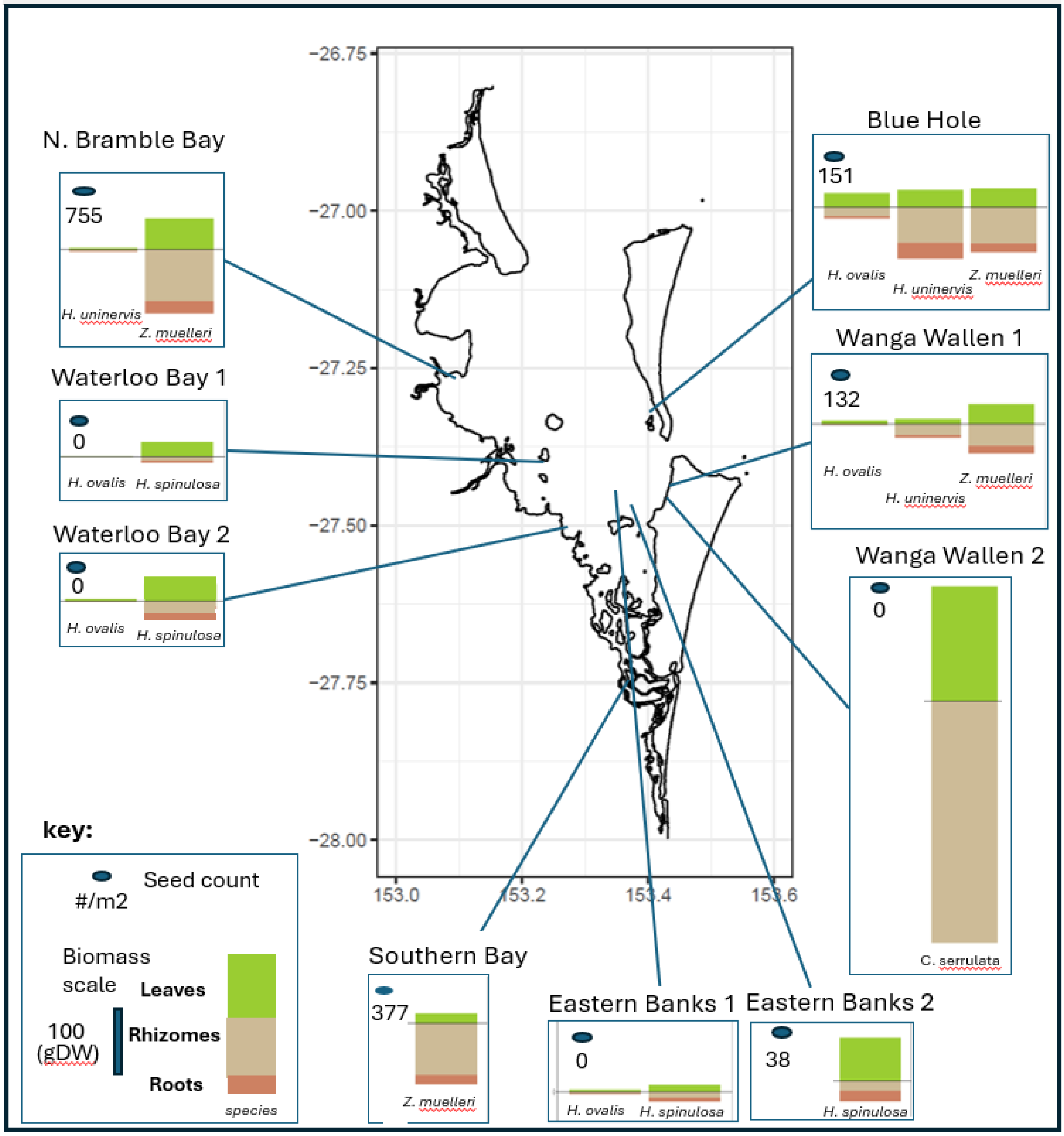


Figure 49. Biomass and seed abundance from 9 seagrass meadows in Moreton Bay.

Seed counts were also highly variable across Moreton Bay. Ranging from no seeds being found in many cores to a maximum of 7 seeds of *Z. muelleri* found in a core from North Bramble Bay (Figure 50). Despite sieving multiple cores of *C. serrulata* (20 cores), and the relatively large size of its seeds, no *C. serrulata* seeds were identified from Wanga Wallen 2. It is important to note that every seed found in a core, changes by 19 the average seeds m^{-2}, so a difference of only a few seeds being found can lead to a large change in the seeds m^{-2} in Table 8 (below). However, the good correlation between replicate cores from the same location confirmed that the average seed densities reported in Table 8 are likely to be representative of seagrass meadows sampled l in January and February 2024.

Table 8. Biomass and seed abundance from 9 seagrass meadows in Moreton Bay.

- When more than 1 species is present in a core, each species is shown in a separate row.

Site	Species	Leaves (gDW m^{-2})	Rhizomes (gDW m^{-2})	Roots (gDW m^{-2})	A: B ratio	Seeds (#m^{-2})	Seed Casings (#m^{-2})
N^th Bramble Bay	*Z. muelleri*	46	77	19	0.48	660	94
N^th Bramble Bay	*H.uninervis*	2	3	1	0.62	-	-
Waterloo Bay 1 (St Helena Island)	*H. spinulosa*	22	6	3	2.5	0	0
Waterloo Bay 1 (St Helena Island)	*H.ovalis*	1	1	0.02	1.1	-	-
Waterloo Bay 2 (Lota Foreshore)	*H. spinulosa*	31	11	4	2.0	0	0
Waterloo Bay 2 (Lota Foreshore)	*H.ovalis*	3	1	0.25	2.1	-	-
Southern Bay (Mackley Island)	*Z. muelleri*	14	77	14	0.16	170	208
Eastern Bay 1 (Maroom Banks)	*H. spinulosa*	7	6	4	0.89	0	0
Eastern Bay 1 (Maroom Banks)	*H.ovalis*	3	3	1	0.72	-	
Eastern Bay 2 (Maroom Hole)	*H. spinulosa*	65	15	16	2.1	19 (probably *H.spinulosa*)	19 (*Z. muelleri*)
Wanga Wallen 2 (N^th Myora)	*C. serrulata*	171	359	76	0.39	0	0
Wanga Wallen 1 (S^th Amity)	*Z. muelleri*	30	32	12	0.69	0	132
Wanga Wallen 1 (S^th Amity)	*H.uninervis*	8	17	4	0.39	-	-
Wanga Wallen 1 (S^th Amity)	*H.ovalis*	6	1	1	3.2	-	-
Blue Hole (Crab Island)	*Z. muelleri*	28	54	13	0.42	57	94
Blue Hole (Crab Island)	*H.uninervis*	25	53	24	0.33	-	-
Blue Hole (Crab Island)	*H.ovalis*	21	13	4	1.24	-	-

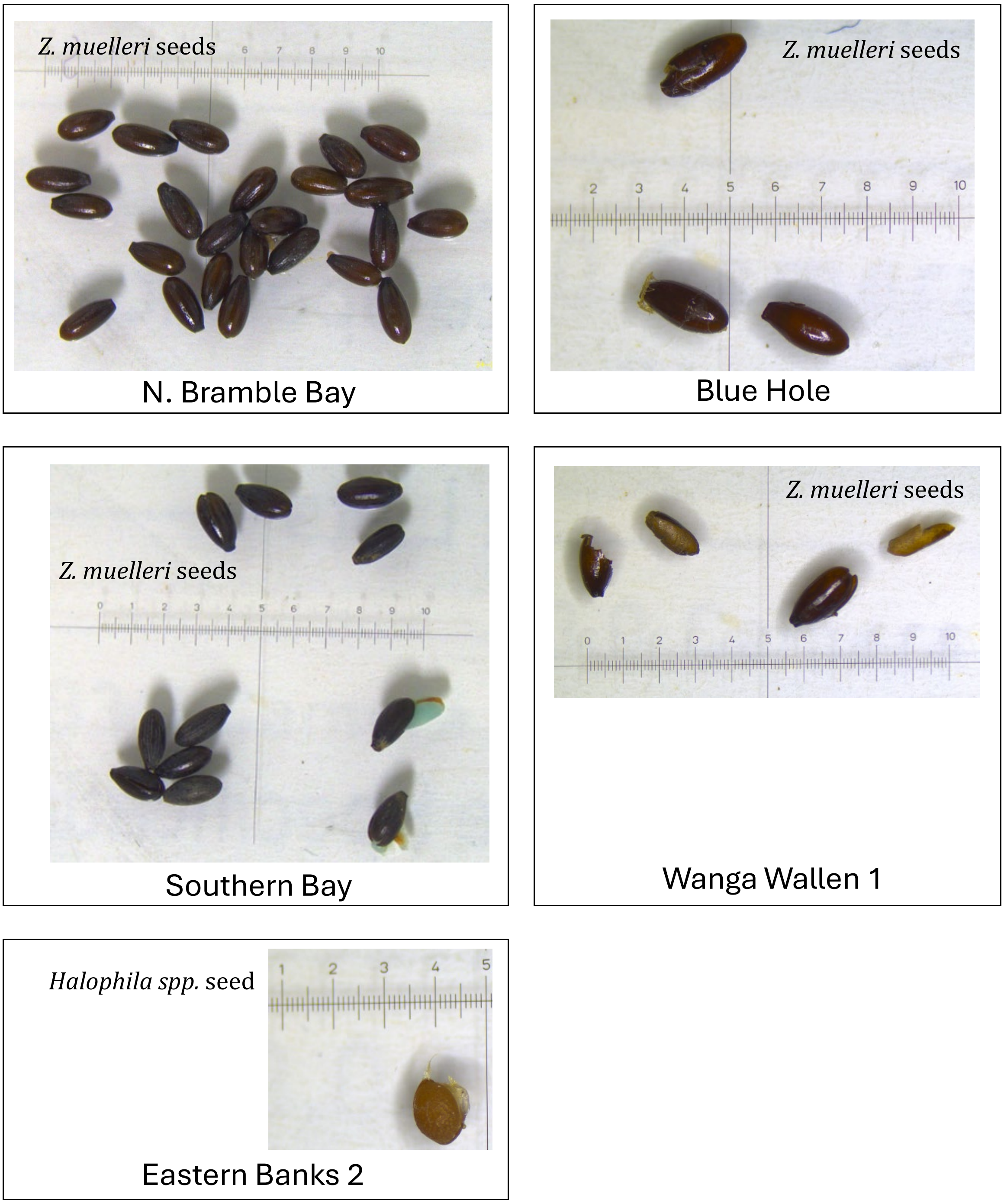


Figure 50. Example photos of seeds/casings identified from each location where seeds were present. Seed casings are above the ruler and whole seeds are below the ruler.

DISCUSSION

The 2022 flood in the Moreton Bay catchment reduced the distribution of seagrass in MBMP. The decline in depth range of all seagrass species in the first year post-flood indicates that benthic light availability was a limiting factor for seagrass growth in the aftermath of the 2022 floods. The greatest declines in seagrass distribution and abundance were observed in the Deep Subtidal zone (>2m below chart datum), suggesting that the large quantities of sediment delivered by river systems during the 2021/22 austral summer reduced the available benthic light in these habitats for an extended period to below the minimum level required for seagrass to survive (Abal & Dennison 1996, Longstaff 2003). The depth range of seagrass has recovered in many regions during the second year post-flood, however there is evidence of chronic water quality issues hampering full meadow recovery in the Deep Subtidal zone. In the second year post-flood, seagrass in some areas of Moreton Bay had a higher density of seagrass cover in the Intertidal and Shallow Subtidal zones, than observed pre-flood, indicating where light was not a limiting factor, the 2022 flood may have had a positive fertilisation effect.

The Intertidal and Shallow Subtidal meadows in most of the Bay had recovered by the second year post-flood, however the density and frequency of occurrence of seagrass in the Deep Subtidal zone remained supressed compared to pre-flood conditions. This lack of full recovery in the Deep Subtidal zone indicates a long-term reduction of benthic light, likely due to the increase in mud delivered by the flood. Despite a switch to the statistically drier El Niño cycle in 2023, the SAM remained positive, with Moreton Bay Catchment receiving above median rainfall in the 2023/24 austral summer (BOM 2024). Continued rain combined with re-suspension of fine sediments deposited by the 2022 flood is likely to have maintained higher turbidity levels within Moreton Bay than during pre-flood conditions.

The largest difference between pre- and first year post-flood seagrass communities was observed in Eastern Bay, despite this region having comparatively better water clarity than Western Bay. It is likely this difference was caused by meadows in Western Bay already having significantly shallower depth distributions than Eastern Bay, due to chronically poor water clarity (Roelfsema et al. 2009, Leigh et al. 2013). It is also possible that the seagrasses resilience to changes in water clarity are greater in Western Bay. Seagrass communities that live in chronically low light conditions have altered physiological and morphological characteristics that better enable them to cope with acute reductions in benthic light caused by severe events such as floods (Abal et al. 1994, Maxwell et al. 2014). By the second year post-flood, seagrass in Eastern Bay had recovered to a frequency of occurrence and density similar to pre-flood in the Intertidal and Shallow Subtidal zones. However, the density and proportion of seagrass in Deep Subtidal meadows is still lower than pre-flood indicating a continued depression in water clarity.

In contrast to Eastern Bay, the seagrass depth range in Deception Bay was not affected in the first year post-flood, likely due to the already chronic water quality discussed above (Roelfsema et al. 2009, Leigh et al. 2013). However, in the second year post-flood the depth range of *H. spinulosa* and *H. ovalis* became significantly shallower. The decline in depth range observed in the second year post-flood may be indicative of a more long-term decline in light availability, due to continual re-suspension of fine sediment deposited during the flood that has maintained benthic light availability below minimum light requirements for an extended period (Longstaff 2003). Parts of Deception Bay that were muddy sand pre-flood are now sandy mud, with a higher proportion of fine particles available for re-suspension (Grinham et. al 2024). Deception Bay is exposed to the dominant south-easterly wind, making it more vulnerable to re-suspension than other regions of the Bay. During sampling, we observed a highly turbid benthic layer in the water column that initiated approximately 50cm above the seafloor despite relatively good surface water clarity. If this turbid layer regularly occurs above the seafloor, it will significantly reduce the light available for photosynthesis to support seagrass survival and growth.

The loss/ decline of *H. spinulosa* within all regions in the first year post-flood, suggests that in many locations *H. spinulosa* was already growing close to its minimum benthic light levels. Though there has been some recovery of *H. spinulosa* in the second year post-flood, *H. ovalis* or very sparse seagrass has replaced *H.spinulosa* in the Deep Subtidal zone of most regions. As a coloniser species, *H. ovalis* is often the first to recover and thus, the increase in proportion of *HO* meadows since the 2022 flood is likely due to *H. ovalis* colonizing bare substrate left by the loss of other species (Rasheed 2004, Kilminster et al. 2015). Although fast growing, *H. ovalis* and very sparse meadows will not be particularly resilient in the event of future reductions in benthic light (Longstaff et al. 1999, Collier et al. 2016), indicating a potential shift to a more transitory coloniser seagrass community in the Deep Subtidal zone of Moreton Bay.

Pre-flood *C. serrulata* and *S. isoetifolium* were already rare in Moreton Bay, occurring predominantly in parts of the Bay with the cleanest water: Eastern Banks (including Wanga Wallen on the western side of North Stradbroke) and to a lesser extent, North Deception Bay and Southern Bay adjacent to Jumpinpin. *C. serrulata* and *S. isoetifolium* are tropical seagrass species, adapted to clear water reef environments (Carter et al. 2021). Following the 2011 floods the loss of a large meadow of *S. isoetifolium* was recorded in North Deception Bay (Hanington et al. 2015), and pre-2022 flood, SUSA had only observed limited recovery of *S. isoetifolium* in the area. In the first year post-flood, the occurrence of *C. serrulata* and *S. isoetifolium* significantly declined, with distribution restricted to the Eastern Bay only, with the exception of a few observations in Deception Bay and Southern Bay (around Jumpinpin). The decline in distribution of *C. serrulata* in Eastern Bay coincided with a decline in depth distribution in the first year post-flood, indicating light availability had likely been a limiting factor. In the second year post-flood, there had been good recovery of *C. serrulata* in the Intertidal and Shallow Subtidal zones in Eastern Bay but limited recovery in the Deep Subtidal zone. *S. isoetifolium* remained rare in Eastern Bay and other regions in both sampling years post-flood. Although opportunistic species such as *C. serrulata* and *S. isoetifolium*, can withstand longer periods of acute low light than coloniser species (Collier et al. 2016), the flood events in Moreton Bay appear to have decreased benthic light levels beyond their survival capacity in parts of their pre-flood distribution. These species are much slower to recover than the opportunistic *Halophila* spp. The decline in these opportunistic species in Moreton Bay could have long-term implications for the biodiversity and stability of Moreton Bay's seagrass meadows.

Although some declines were observed in the intertidal assemblages, this zone was less affected by the flood, with the majority of seagrass loss in subtidal habitats. Intertidal meadows are the least affected by declines in water clarity as they are the shallowest meadows and receive direct light for at least the low tide portion of the day. *Z. muelleri* is also more resilient to lower light conditions than *Halophila* spp., capable of surviving up to 2 months of near-total light deprivation (Abal et al. 1994, Longstaff et al. 1999). The minimal seagrass loss in the Intertidal zone is consistent with findings in North Deception Bay, Waterloo Bay and Southern Bay following the 2011 floods, where no mortality of *Z. muelleri* was recorded (Hanington et al. 2015, Maxwell et al. 2014). There were, however, declines in the frequency and density of *Z. muelleri/ H. uninervis* and *H. ovalis* in the Intertidal zone of Bramble Bay and Central Bay in the first year post-flood. These declines may have been caused by acute low salinities, diluted toxicants present in the floodwater, or smothering by deposited fine sediment. In the second year post-flood, Intertidal meadows had recovered in both Bramble and Central Bay. In general, the response of seagrasses in Moreton Bay to the 2022 flood are in direct contrast to the results of the post-flood seagrass survey in the Great Sandy Marine Park, where the intertidal meadows declined, and the majority of remaining seagrass was observed in the deep subtidal (York et al. 2022). This indicates the primary environmental stressors likely differed between these systems, suggesting future management approaches to improve seagrass recovery and resilience may need to differ in these two marine parks.

In the Western Bay (Deception and Waterloo), Southern Bay and Broadwater, the density and proportion of seagrass increased in the Intertidal and Shallow Subtidal zones post-flood. These regions experienced chronic poor water clarity prior to the flood and thus, were likely to be more resilient to acute reductions in benthic light (Abal et al. 1994, Maxwell et al. 2014). The 2022 flood significantly increased available nutrients (O'Mara et al. 2019, Grinham et. al. 2024), and thus, as long as there was sufficient light for growth, the flood is likely to have produced a positive fertilisation effect (Udy and Dennison 1997).

It is important to note that although only limited flood effects were observed in the spring/summer sampling in the first year post flood (2022/23) in the Western Bay, this survey is a representation of the distribution of seagrass that had survived and also re-grown in the 9-11 months since the flood. Preliminary surveying in the Western Bay in August 2022 (6 months post flood) observed significantly lower densities of *H. spinulosa* and *H. ovalis* than the spring/summer sampling (Udy et al. 2023). Hence the current study may not accurately represent the short term declines in seagrass distribution that occurred immediately following the flood, that would have had implications for the ecosystem and species of concern (e.g. turtles and dugongs) that rely on seagrass as their primary food source (Preen 1995).

Bramble Bay suffered more seagrass loss following the 2022 flood than the rest of the Western Bay region. Bramble Bay is immediately north of the mouth of the Brisbane River and is heavily affected by fluvial inputs of fine sediment (Figure 3). Prior to the 2022 flood, the seagrass in Bramble Bay had primarily been observed in the northern half, furthest from the mouth of the Brisbane River. Since the 2022 flood, aside from a small amount of moderately dense *Z. muelleri/ H. uninervis* in the Intertidal zone (mainly at the northern end), Bramble Bay seagrass is predominantly comprised of sparse *H. ovalis*. These appear to be transient meadows susceptible to future disturbances.

Pumicestone Passage seagrass is dominated by dense Intertidal *Z. muelleri/ H. uninervis* meadows on the side of the passage, with the proportion of *H. ovalis* increasing as you move into the Shallow Subtidal zone. The deeper channels rarely have seagrass, though prior to the 2022 flood, some *H. spinulosa* had been observed. Only a minor flood impact was observed in the comparable temporal sites analysed for Pumicestone Passage near the two entrances. Comparison of historic 2002 data to current distribution indicates there has been substantial loss of seagrass in Tripcony Bight at some point in the last 20 years. BMT surveying of northern Pumicestone Passage shows a continual decline in seagrass extent in northern Pumicestone Passage from 2019 till 2023 (BMT 2023). The decrease in seagrass extent in northern Pumicestone Passage is most likely a result of the Bribie Island break-through, which has dramatically altered sand movement, as well as wave and tidal energy in the area (BMT 2023). The increase in proportion of coloniser *H. ovalis* observed in the second year post-flood may either be an indication of recovery or a result of sampling bias, given the limited spatial coverage of pre-flood sampling.

Seagrass biomass observed in Moreton Bay was highly variable between locations (within the same species) and between seagrass species. Above-ground biomasses ranged from $10gDWm^{-2}$ for a mixed *H. spinulosa*, *H. ovalis* meadow (equivalent to the maximum biomasses observed in Hervey Bay; York et. al. 2022) to $171gDWm^{-2}$ for *C. serrulata* in Eastern Bay.

The high biomass observed for *C. serrulata* (both above and below-ground) is consistent with it being the most persistent seagrass species in Moreton Bay. Where persistent species invest more energy into vegetative growth, accumulate higher biomass, but reduce their energy investment in sexual reproduction (flowers and seed formation). The total *C. serrulata* biomass observed in Moreton Bay in the current study ($530gDWm^{-2}$) was very similar to the total biomass observed in 1994 ($508gDWm^{-2}$) in Wanga Wallen Banks (slightly north of the 2024 site) (Udy and Dennison, 1997). However, the current study observed a higher proportion of biomass as leaves, with the Above:Below biomass ratio being higher in 2024 than in 1994 (0.39 in 2024 compared to 0.26 in 1994). Given the limited number of samples, it is hard to confirm if this reflects natural variation or a fertilisation effect in the 2024 *C. serrulata* meadow. Previous studies in Moreton Bay demonstrated that an increase in nutrient availability at Wanga Wallen Banks would be likely to cause an increase in leaf growth rates and may explain the observed increase in the A:B ratio (Udy and Dennison 1997).

H. spinulosa meadows had both the second highest (65gDWm^{-2}) and lowest (7 gDWm^{-2}) above ground biomass. They generally had very low below ground biomass and high A:B ratios (0.9 to 2.5), consistent with their life history as a colonising species. Colonising species expand rapidly when conditions are favourable and invest in sexual reproduction (flowers and seed production). *H. spinulosa* meadows were the most severely affected by the 2022 flood and are likely to be undertaking their rapid recovery phase. *H. spinulosa* has expanded its distribution and increased its percentage cover since the August 2022 survey, either through regrowth of the limited remaining living rhizomes or by seed germination.

Mixed meadows, where *Z. muelleri* was the dominant species, had slightly less variation in their above-ground and total biomass with above-ground biomass ranging from 14 to 46gDWm^{-2} and total biomass ranging from 74 – 142gDWm^{-2}. Their A:B ratios ranged from 0.16 to 0.69, demonstrating a high morphological plasticity in this species, with the small leaf intertidal meadows having the lowest A:B (0.16) and the larger leaf Shallow Subtidal meadows having the highest A:B (0.69).

When the *Z. muelleri* meadows surveyed in the current study are compared with two baywide surveys conducted in Moreton Bay in January 1999 and February 2013, the total biomass observed in the current study was less, suggesting that across Moreton Bay the density of *Z. muelleri* may have declined. The current study had a total *Z. muelleri* biomass range from 74 – 142gDWm^{-2}, yet in 2013 (Samper-Villarreal et. al. 2017), *Z. muelleri* biomass was observed to range between 98 and 328 gDWm^{-2} and between 140 and 650gDWm^{-2} in 1999 (Udy et. al., unpublished, Appendix 4). Although the total *Z. muelleri* biomass may have reduced over the last 25 years, the relative proportion of above to below-ground biomass (A:B) appears to have remained in a similar range to the 2013 (0.11 - 0.44) and 1999 (0.18 – 0.75) ratios.

The sampling locations were different between studies, so it is unreasonable to directly compare these results, but as all studies surveyed multiple locations with sites representative of Western and Eastern Bay conditions (some within the same bay or bank), the reduction in the higher biomass values for *Z. muelleri* meadows suggests the density of *Z. muelleri* meadows in Moreton Bay may be reducing.

Relatively high seed/ seed casing abundance was recorded in the current study for *Z. muelleri* meadows, but no *C. serrulata* or *H.uninervis* seeds or seed castings were observed, despite these species being present in the cores. We also observed a single *Halophila spp*. seed and suspect more may have been missed, due to their very small size. The highest seed abundance (755seeds.m^{-2}) was observed in North Bramble Bay, which is a seagrass meadow that we believe has been expanding since 2018, when we first observed recovery of seagrass in Bramble Bay. Despite the 2022 flood, likely slowing the recovery of this meadow, the high seed count and biomass of this meadow is a good sign for the meadow's resilience and recovery.

The other *Z. muelleri* meadows surveyed had approximately ½ to ¼ of the seed abundance as North Bramble Bay with Southern Bay having 377seeds.m^{-2} and the two Eastern Bay locations having 132 and 155seeds.m^{-2}. This reduction in seed abundance as you move towards more stable meadows and away from nutrient sources is consistent with the understanding of abiotic and biotic factors that influence seed production (Zhaxi et. al. 2022, Lekammudiyanse et. al. 2024).

The authors are not aware of a previous study in Moreton Bay that measured the abundance of seagrass seeds, but was advised by a representative at JCU, who regularly counts seeds, that the number of seeds and seed castings observed in the Moreton Bay samples were much higher than what would be expected in Hervey Bay samples (Scott pers. comm). The seed counts in Moreton Bay were also an order of magnitude larger than seed densities recorded from mixed species seagrass meadows near Cairns (*Z. muelleri* 3 – 24seeds.m^{-2} *H. uninervis* 8 – 13seeds.m^{-2} *C. serrulata* 0 – 1seeds.m^{-2})(Jarvis et. al. 2021). This could be for two reasons: (1) Moreton Bay has denser seagrass meadows with a higher *Z. muelleri* biomass; (2) Seagrass meadows that are in an active expansion/recovery phase are thought to invest more energy into sexual reproduction (flower and seed production).

The results from the current study indicate that although the 2022 flood had widespread impacts on the distribution and composition of seagrass meadows within MBMP, there has been some recovery in the second year post-flood. The observations of biomass and seed abundance in Moreton Bay support the fact that seagrass meadows are recovering from the 2022 flood and in a reasonably healthy condition. The lack of full recovery of seagrass in the Deep Subtidal zone raises significant implications for the ongoing survival of Moreton Bay's deeper seagrass communities. The decline in the occurrence of *S. isoetifolium, C. serrulata* and *H. spinulosa* may also indicate a change in the resilience of Moreton Bay's seagrass communities. Despite the declines in depth range, the resilience of Moreton Bay's Intertidal and Shallow Subtidal meadows could provide a future source of seed to recolonise deeper areas if water clarity improves. There is evidence that seagrass meadows are potentially reducing their total biomass (lower maximum biomass observed in Moreton Bay compared to 1999 survey). There is also anecdotal evidence that historically monospecific seagrass meadows are becoming more diverse. This is evident in the majority of biomass cores collected for the current study having multiple species (two or three) present, compared with the 1999 survey where cores of *Z. muelleri*, *H.uninervis* and *C. serrulata* only had a single species present. It also supports the analysis of the bay wide observations that suggest seagrass meadows in Moreton Bay are becoming more diverse in their species composition.

Grinham et al. (2024) reported that between 2015 and 2019 there was a reduction in the mud content of sediment in the Bay, indicating that the Bay has the potential to recover from the large quantities of fine sediment delivered by floods if given adequate recovery time. Future management actions in combination with rain and wind patterns will determine if water clarity within the Bay is able to recover.

Recommendations

This report has demonstrated that the 2022 flood had an immediate negative impact on Moreton Bay seagrass meadows during the first year post-flood, likely due to the flood plume and turbid water following the flood reducing benthic light availability. It has also demonstrated that the flood has had an ongoing negative impact on the Deep Subtidal seagrass meadows (below 2m AHD), likely due to the mud delivered to Moreton Bay by the flood water continuing to be resuspended by wind and tidal action, resulting in a chronic decrease in benthic light availability across the Bay.

The effective management of seagrass meadows within MBMP requires an understanding of the current condition of seagrass meadows, any trajectory of change (contracting or expanding) and an understanding of benthic light availability, as this is the dominant environmental impact determining seagrass distribution and condition within MBMP. To efficiently obtain this information it is recommended that Marine Park managers work with other organisations within SE Qld (including other sections of DESI) to maintain an on-going monitoring program within MBMP that will monitor seagrass communities in all areas of Moreton Bay annually or biannually.

The analysis of changes in seagrass distribution in this report has focused on two main areas; spatial change in distribution, and temporal change in composition and density. The sampling undertaken to complete this report has been extensive and it is therefore not necessarily feasible to continually monitor seagrass distribution in MBMP at this resolution. Therefore, we have suggested two possible future monitoring methods to capture the temporal and spatial change.

Analysis of temporal change in density and composition could be achieved by picking a representative meadow in each tidal zone of each region, within the existing temporal study area. Additional meadows that include rare or declining seagrass species such as *C. serrulata* and *S. isoetifolium* should also be included from each region where they occur. By making a min of 30 random observations in each meadow/year it would provide an ability to identify changes in species composition and or density over time. This would require approximately 1000 observations/ year and could be adjusted as per available funding to either improve or reduce spatial resolution. If there is limited funding, prioritisation of the sub-tidal meadows should occur. These meadows are difficult to observe using remote sensing and experienced the largest declines from the 2022 flood. While poor water clarity is the dominant environmental variable impacting on seagrass in MBMP, the best indication of seagrass health to inform marine park managers of its condition will be surveys that include depth distributions for different species.

Identification of spatial change (as opposed to average change in a representative meadow) will require a higher resolution of sites. This report has highlighted that the biggest loss of seagrass has been in the Deep Subtidal zone, as a result of declines in seagrass depth range. As such, future monitoring should focus on determining the depth range of seagrass in each Bay / region. This can be achieved with a series of transects, with sites every 0.5m change in depth. Future changes in depth distributions would then be able to be compared to current distribution as determined by depth contours or in-situ measurements. It's important to note that while the recently improved MBMP DEM (commissioned by the Qld Wetlands Program, DESI) improved bay wide estimates of depth where water clarity allowed satellites to see the bottom, the accuracy of depth models was limited in areas of the bay with poor water clarity. This meant that regions of MBMP where seagrass is likely to decline (due to limited light availability) are the areas where depth estimates are the least accurate. To resolve this new bathymetry data is required, especially in turbid areas of MBMP. This could be collected as part of a future seagrass monitoring program or separately. It could be done at a high resolution only in areas where seagrass distribution transects are monitored (see above depth transect recommendation). But an improved bay wide bathymetry, including areas with poor water clarity, would be useful to support many aspects of Marine Park Management. One specific application would be the ability to model likely seagrass responses to future floods more accurately as well as the ability to produce more accurate seagrass distribution maps with a minimal/optimised sampling strategy.

In addition to monitoring the distribution of seagrass species and density (% cover) it is important for Marine Park managers to have up to date information on the dominant environmental drivers of seagrass change. Based on the results of this report and previous analysis (Udy et. Al. 2021, Udy et al. 2023), it is believed water clarity and its impact on benthic light availability is the primary environmental factor limiting seagrass distribution in Moreton Bay, it also directly impacts on seagrass density, with density usually decreasing as benthic light decreases. To improve Marine Park managers understanding of how seagrass meadows within MBMP respond to ongoing floods and catchment run-off, as well as if active regeneration of seagrass may be beneficial, it is important for them to have access to regular assessment of water clarity and benthic light availability within MBMP (using either sechii depth or light loggers). Access to regular assessments of this key environmental driver will enable them to predict seagrass meadows likely to be impacted by future floods and identify areas where seagrass loss or slow recovery of seagrass may be due to factors other than water clarity (e.g. toxins or loss of seed bank). Where seagrass meadows are thought to be at risk of loss or have experienced an acute decline/loss it would be useful to undertake more intensive field sampling – including seed bank assessment, reproductive effort, and temporal assessments of biomass and/or seagrass growth rates. These seagrass response variables provide an indication of the resilience of the seagrass meadow and likely recovery outcomes, enabling managers to decide if more intensive intervention/restoration efforts are justified. Other environmental variables that would assist marine park managers in predicting future impacts on seagrass meadows and the likelihood of seagrass to recover (following loss) in various regions of the Marine Park include improved bathymetry information for MBMP, hydrodynamic information (including the behaviour of flood/catchment runoff scenarios, wind-driven turbidity plumes, and seed dispersal).

Integrating of causal environmental variables with seagrass responses into a seagrass presence/absence model for MBMP will provide managers with an important initial assessment of likely impacts of future extreme weather events as well as indicate what management actions will best support seagrass resilience and recovery of impacted meadows.

Seagrass distribution in Moreton Bay has been consistent over the last 20 years with benthic light being the primary driver of distribution and nutrient availability and benthic light influencing the density of seagrass cover and probably also species present. In areas where water clarity is not limiting seagrass growth or recovery, it is possible that different management strategies will be needed to achieve optimal environmental outcomes, including the possibility of active regeneration if there is an absence of seed bank or viable seagrass to enable meadow regeneration.

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Appendix 1: Areal extent of seagrass polygons created for Moreton Bay Marine Park and seagrass change calculations used in report.

Table A1 1: Seagrass distribution areal extent as determined by polygon analysis.

Polygon	Area
Pre-flood (2015-2021)	327.02 km^2
2022/ 2023	304.17 km^2
2023/2024	291.83 km^2
2004 (Baywide) + 2002 (Pumicestone Passage)	198.17km^2
2004	185.48 km^2
2002 (Pumicestone Passage)	12.69 km^2

Table A1 2: Change in seagrass areal extent between survey periods as determined by polygon analysis.

Change Map	No change (km^2)	Loss (km^2)	Expansion (km^2)
Pre-flood (2015-21) – 2022/23	303.68	23.33	0.49
2022/23 – 2023/24	289.23	14.93	2.59
Pre-flood (2015-21) – 2023/24	291.79	35.23	0

Table A1 3: Change in seagrass areal extent between Pre-flood (2015-21) surveying and first year post-flood (2022/23) for each region as determined by polygon analysis.

Zone	No change (km^2)	Loss (km^2)	Expansion (km^2)
Pumicestone Passage	9.21	0.01	0
Deception Bay	39.05	0.58	0
Bramble Bay	9.61	0	0
Waterloo Bay	48.12	0.65	0
Central Bay	25.28	3.64	0.44
Eastern Bay	154.20	15.25	0
Southern Bay	12.48	3.19	0
Broadwater	6.21	0	0

Table A1 4: Change in seagrass areal extent between Pre-flood (2015-21) surveying and second year post-flood (2023/24) for each region as determined by polygon analysis.

Zone	No change (km^2)	Loss (km^2)	Expansion (km^2)
Pumicestone Passage	9.07	0.14	0
Deception Bay	33.86	5.77	0
Bramble Bay	5.30	2.97	0
Waterloo Bay	48.51	0.26	0
Central Bay	24.84	4.08	0
Eastern Bay	149.31	20.11	0
Southern Bay	13.84	1.84	0
Broadwater	6.18	0.03	0

Appendix 2

Presence and density of individual seagrass species within Moreton Bay Marine Park, based on the 2023/24 survey.

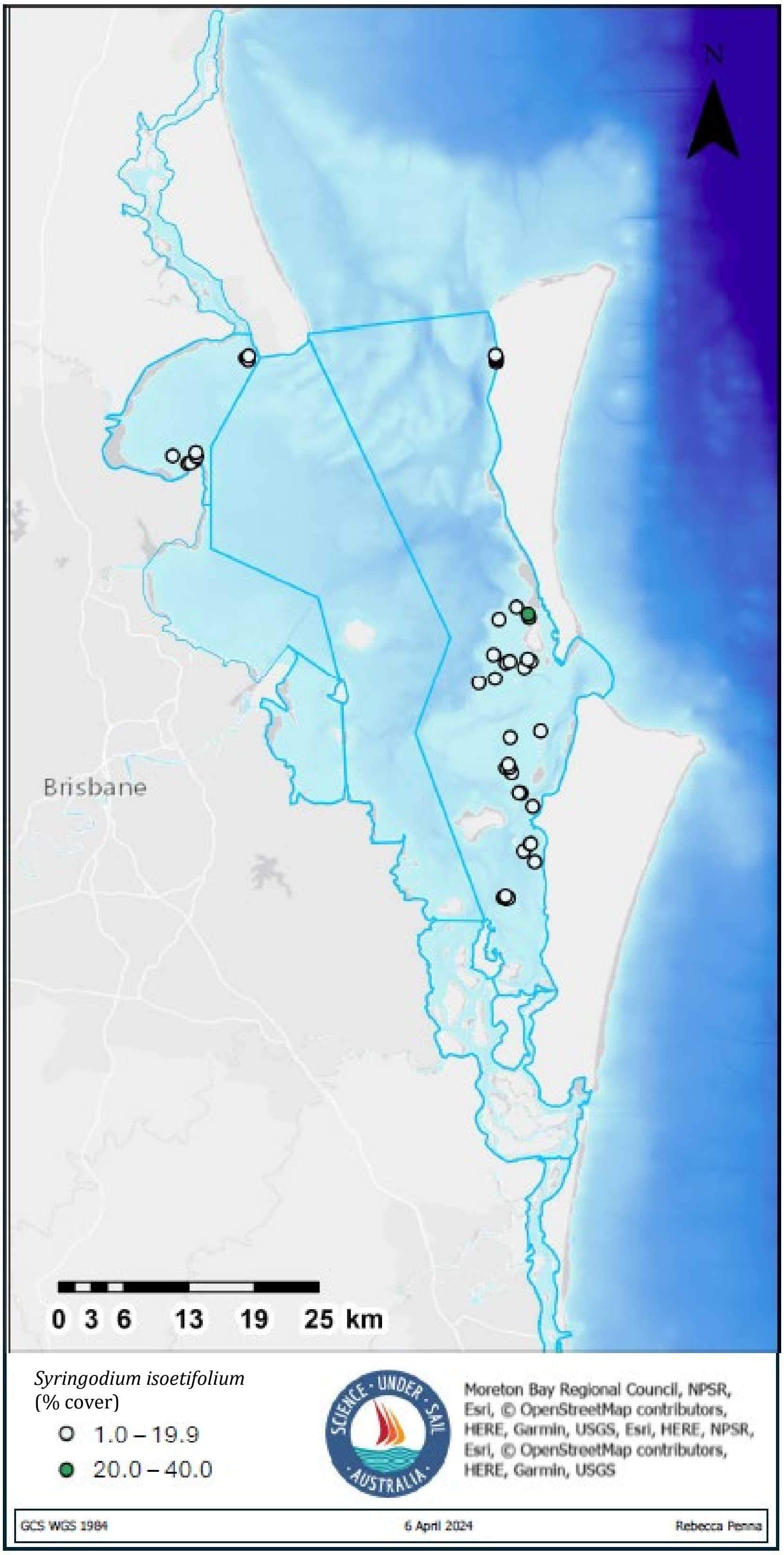

Figure A2 1. Sites were *Syringodium isoetifolium* was present and its % cover in MBMP in 2023/24.

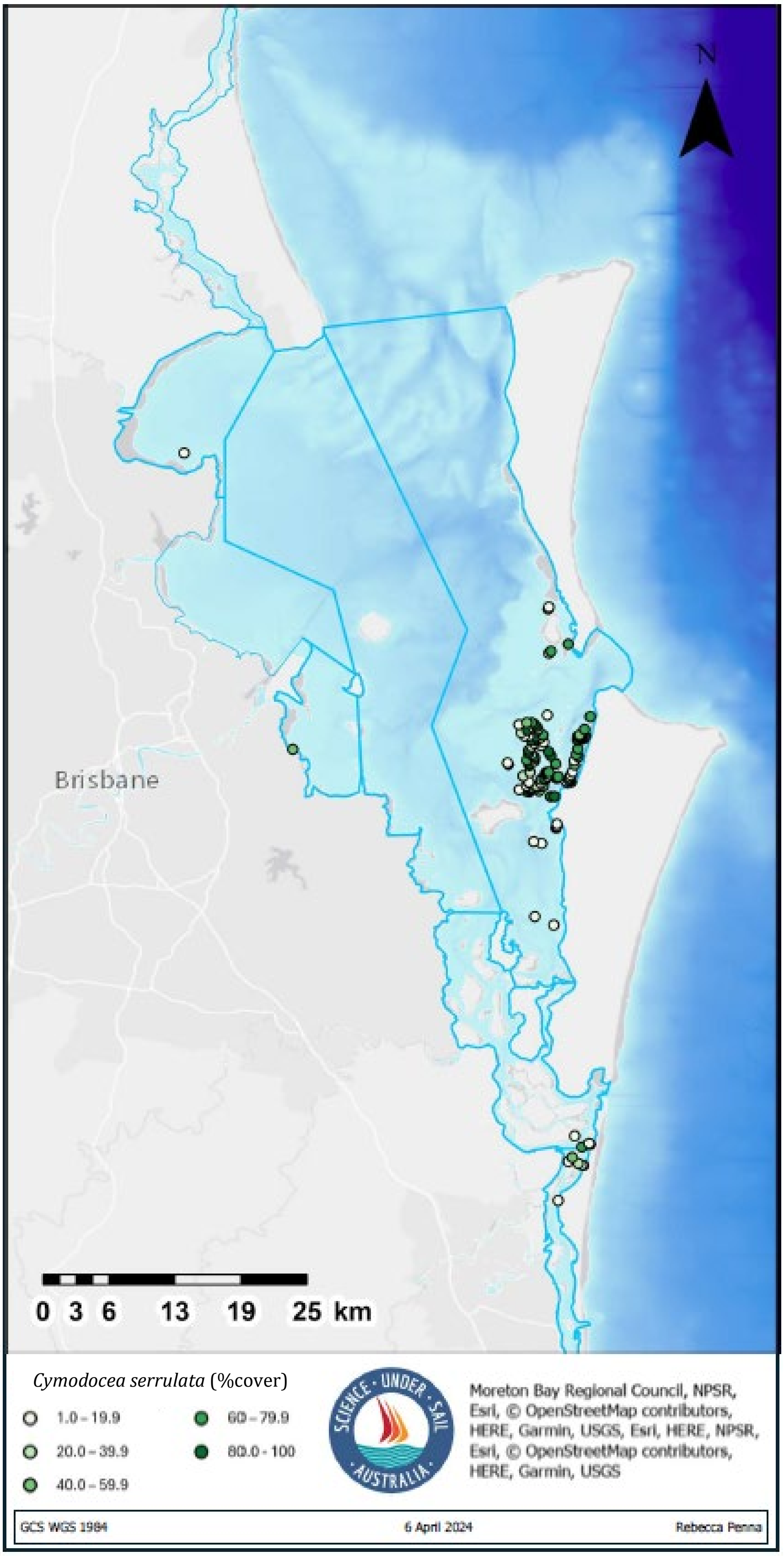

Figure A2 2. Sites were *Cymodocea serrulata* was present and its % cover in MBMP in 2023/24.

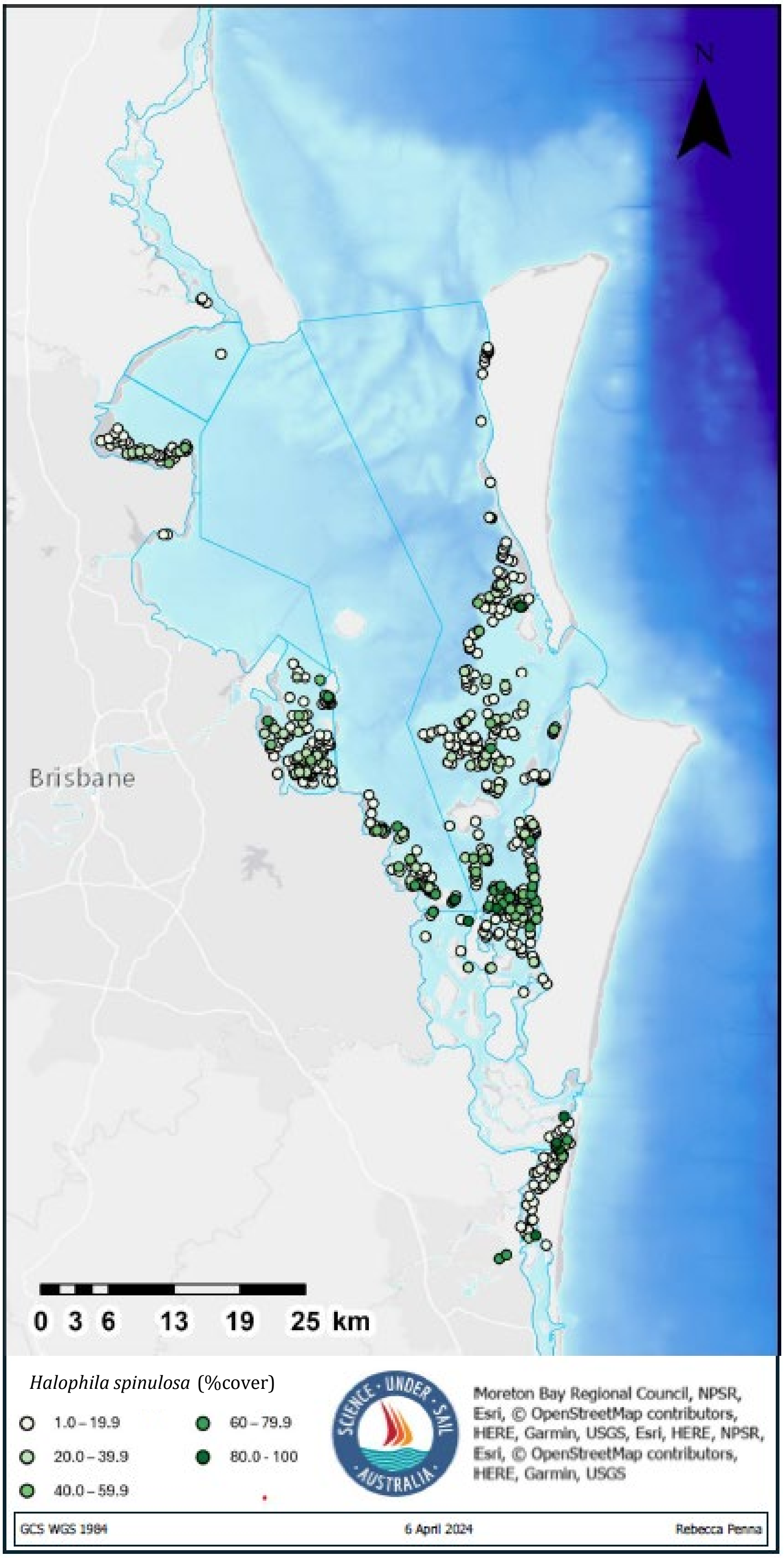

Figure A2 3. Sites were *Halophila spinulosa* was present and its % cover in MBMP in 2023/24.

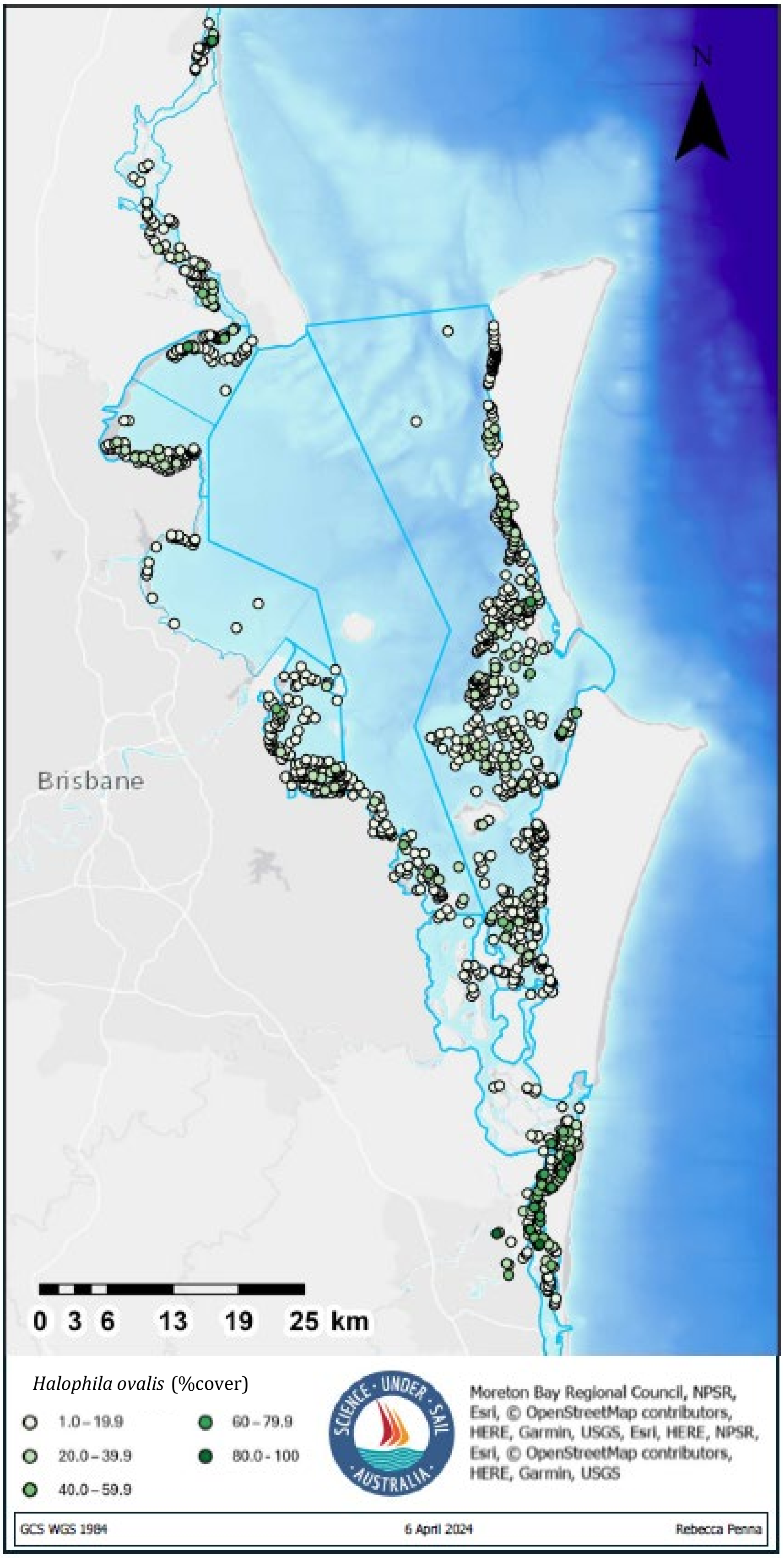

Figure A2 4. Sites were *Halophila ovalis* was present and its % cover in MBMP in 2023/24.

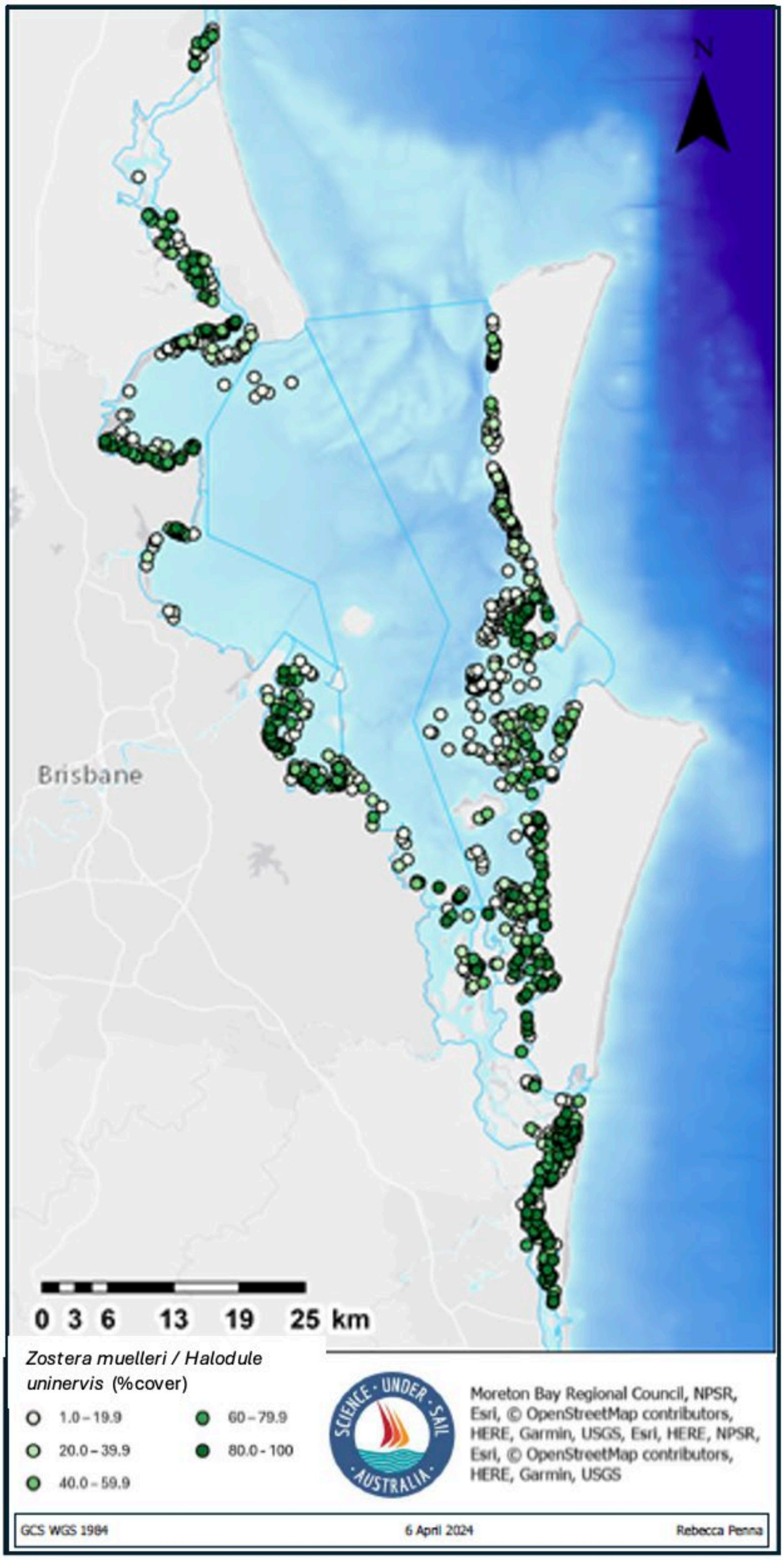

Figure A2 5: Sites were *Zostera muelleri or Halodule uninervis* were present and their combined % cover in MBMP in 2023/24.

Appendix 3

Table A3 1. Summary of habitat data collected at each site

Seagrass	*Zostera muelleri*	% cover
	Halodule uninervis	% cover
	Halophila ovalis	% cover
	Halophila spinulosa	% cover
	Syringodium isoetifolium	% cover
	Cymodocea serrulata	% cover
Macro algae	Filamentous	% cover
	Erect Macroalgae	% cover
	Encrusting	% cover
	Turf Macroalgae	% cover
	Unknown	% cover
Coral	Hard coral	% cover
	Soft Coral	% cover
	Octocoral	% cover
Sediment	Mud	One sediment type was selected
	Sandy Mud	
	Muddy Sand	
	Sand	
	Rock	
	Rock and Mud	
	Rock and Sand	
	Gravel	
Other	Shell Grit Molluscs Sponges Seastars Bioturbation Sand waves	Presence was noted

Appendix 4

Historic data from a 1999 unpublished survey of Moreton Bay *Z. muelleri* meadows (Udy et. al., unpublished).