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Final Report

Seagrass communities of the Great Barrier Reef and their desired state: Applications for spatial planning and management

Alex Carter, Rob Coles, Michael Rasheed and Catherine J. Collier

JAMES COOK UNIVERSITY AUSTRALIA

Seagrass communities of the Great Barrier Reef and their desired state: Applications for spatial planning and management

Alex Carter[1], Rob Coles[1], Michael Rasheed[1] and Catherine J. Collier[1]

[1] TropWATER, James Cook University

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A case study for seagrass meadows in the Burdekin region

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Cover photographs: (front) Reef-associated seagrass community with soft coral at Green Island. Photo by Dieter Tracey; (back) Dugong feeding trails in an intertidal meadow on Cape York Peninsula. Photo by Alex Carter.

This report is available for download from the NESP Tropical Water Quality Hub website:
http://www.nesptropical.edu.au

CONTENTS

LIST OF TABLES

LIST OF FIGURES

ACRONYMS

AIMS....................Australian Institute of Marine Science
BOM.....................Bureau of Meteorology
PARb ....Benthic Photosynthetically Active Radiation
CI ....Confidence interval
CI ....Coastal Intertidal
CR........................*Cymodocea rotundata*
CRAN...................Comprehensive R Archive Network
CRC ....Cooperative Research Centre
CS........................*Cymodocea serrulata*
CS........................Coastal Subtidal
CSIRO..................Commonwealth Scientific Industrial Research Organisation
CVRE ....Cross Validated Relative Error
EA........................*Enhalus acoroides*
EI ....Estuary Intertidal
ERT......................Ecologically relevant target
ES ....Estuary Subtidal
ESRI.....................Environmental Systems Research Institute
GIS.......................Geographic Information System
GBRMP................Great Barrier Reef Marine Park
GBRWHA............Great Barrier Reef World Heritage Area
GLM ....Generalized linear models
GPS ....Geographical Positioning System
HC........................*Halophila capricorni*
HD........................*Halophila decipiens*
HO ....*Halophila ovalis*
HS........................*Halophila spinulosa*
HT ....*Halophila tricostata*
HU........................*Halodule uninervis*
IDW......................Inverse Distance Weighted
IMOS....................Integrated Marine Observing System
ITEM ....Intertidal Extent Model
JCU......................James Cook University
MODIS ....Moderate Resolution Imaging Spectroradiometer
MSL ....Mean Sea Level
MRT ....Multivariate regression trees
NESP ....National Environmental Science Program
PAR ....Photosynthetically Active Radiation
PSU......................Practical Salinity Unit
RADAR ....Radio Detection And Ranging
RF ....Random Forest
RI ....Reef Intertidal
RIMReP ....Reef Integrated Monitoring and Reporting Program
RRRC...................Reef and Rainforest Research Centre Limited
RS........................Reef Subtidal
SI ....*Syringodium isoetifolium*
TH........................*Thalassia hemprichii*

TropWATER........Centre for Tropical Water and Aquatic Ecosystem Research
TWQ.....................Tropical Water Quality
ZC........................*Zostera muelleri* subsp. *Capricorni*

ABBREVIATIONS

°C..degrees celsius
gDW m^{-2}......................................grams dry weight per square metre
Km ....kilometres
m...metre
ms^{-1}..metres per second
m^{-2} s^{-1}..per square metre per second
mol photons m^{-2} d^{-1} ....moles of photons per square metre per day
n..number of samples

GLOSSARY

Benthic Photosynthetically Active Radiation (PARb)

An estimate of the quantum of photosynthetically active radiation reaching the benthos based on a remote sensing algorithm.

Colonising

A seagrass life-history strategy with traits including fast shoot turnover and time to sexual reproduction, low physiological resistance (e.g. to low light events), and an ability to rapidly recover from disturbances from seeds in a seed bank, and from lateral expansion and shoot production.

Condition/state

Relative quantities of characteristics of the seagrass such as biomass and spatial extent.

Confidence intervals

A range of plausible values for an unknown parameter. Most commonly, and throughout this report, a 95% or a 99% confidence interval is used.

Data set

A compilation of spatial data collected from 1984 to 2018 during seagrass surveys in the Great Barrier Reef World Heritage Area and adjacent estuaries.

Desired state

Desired state is an aspirational goal for reporting on ecological health and for guiding management decisions.

Dry season

The period of the year when the least rainfall and river discharge occurs. The exact time-period can be defined in various ways, but in this report it refers to May to October.

eAtlas

An online portal for open-access environmental research, maps and data for tropical Australia.

Ecologically relevant target

A target such as for river load reductions to lagoonal Great Barrier Reef waters that support healthy habitats in what we define as a ‘desired state’.

eReefs

A coupled hydrodynamic-biogeochemical model, and an application of the CSIRO Environmental Modelling Suite.

Growing season

The period of the year when seagrass typically grows the fastest and reaches the highest levels of extent and biomass for the year. The precise period is defined in different ways depending on the application, and also on the location but is typically in the range of August to January in the Great Barrier Reef.

Indicator

A measurable quality of the ecological or environmental system. Sometimes used synonymously with 'metric'.

Intertidal

The area where the seabed is within the tidal range.

Metric

A measurable quality of the ecological or environmental system. Sometimes used synonymously with 'indicator'.

Opportunistic

A seagrass life-history strategy with intermediate (between colonising and persistent) and adaptable traits including shoot turnover and time to sexual reproduction, high physiological resistance (e.g. to low light events), and a poor ability to recover from disturbances due to limited or no seed bank and slow rates of lateral expansion and shoot production.

Persistent

A seagrass life-history strategy with traits including slow shoot turnover and time to sexual reproduction, high physiological resistance (e.g. to low light events), and a poor ability to recover from disturbances due to limited or no seed bank and slow rates of lateral expansion and shoot production.

R

A free software environment for statistical computing and graphics.

Resilience

The capacity to provide ecological services in the future, based on being able to retain condition and function in the face of disturbances.

Senescent season

The period of the year when seagrass growth is slowest and seagrass reaches the lowest levels of extent and biomass for the year. The precise period is defined in different ways depending on the application, and also on the location but is typically in the range of February to July in the Great Barrier Reef.

Subtidal

The area where the seabed is below the lowest tide.

Suspended particulate matter

The suspended matter in the water column of estuarine and marine waters comprised of fine mineral particles, organic matter, living organisms such as bacteria and plankton, and other particles. SPM is deleterious to marine organisms and ecosystems because it can stick to organisms, and contribute to reductions in water clarity.

Water clarity

Describes how far light can travel through the water column and is affected by suspended particulate matter.

Wet season

The period of the year when most of the rainfall and river discharge occurs. The exact time-period can be defined in various ways, but in this report it refers to November to April.

ACKNOWLEDGEMENTS

This project is supported with funding from the Australian Government's National Environmental Science Program and in-kind support from James Cook University's (JCU) Centre for Tropical Water and Aquatic Ecosystem Research (TropWATER).

We would like to thank the many JCU TropWATER and Queensland Government (Fisheries) staff who have contributed to data collection during seagrass surveys and monitoring programs since the mid-1980s. This data provides the foundation for all analyses in this report.

Many organisations provided funding for seagrass surveys and monitoring programs included in this project. These include Ports North, Gladstone Ports Corporation, CSIRO, Maritime Safety Queensland/ Department of Transport and Main Roads, Australian Maritime Safety Authority, North Queensland Bulk Ports, Port of Townsville, Trinity Inlet Management Plan, Trinity Inlet Waterways, Fisheries Research Development Corporation, CRC Reef Research Centre, Queensland Department of Agriculture and Fisheries, Great Barrier Reef Marine Park Authority, and the Mackay-Whitsunday-Isaac Healthy Rivers to Reef Partnership.

The eReefs model simulations used in our analysis were produced as part of the eReefs project (https://research.csiro.au/ereefs/), a collaboration between the Science Industry Endowment Fund, the Commonwealth Scientific Industrial Research Organisation (CSIRO), the Australian Institute of Marine Science (AIMS), the Bureau of Meteorology (BOM), and the Great Barrier Reef Foundation, with support from BHP Billiton Mitsubishi Alliance, the Australian and Queensland governments, and with observations obtained through the Integrated Marine Observing System.

Dr Jon Brodie was a fervent supporter of Reef conservation. He identified the need for ecologically relevant targets (ERTs) for seagrass meadows, and was pivotal in securing the funding for project NESP TWQ Hub Project 3.2.1, which led to NESP TWQ Hub Project 5.4. This report builds on Jon's legacy and is motivated by an aspiration to compile evidence needed to underpin ERTs.

EXECUTIVE SUMMARY

The research program reported here evolved from an interest in developing ecologically relevant target criteria that, if met, correspond to desired ecological outcomes (e.g. desired state) for the Great Barrier Reef World Heritage Area (GBRWHA) and to achieving the over-arching objective of the Great Barrier Reef Marine Park Authority's Long-term Sustainability Plan.

The objective of the original National Environment Science Program (NESP) Tropical Water Quality Hub (TWQ) Project 3.2.1 *Deriving ecologically relevant load targets to meet desired ecosystem condition for the Great Barrier Reef: a case study for seagrass meadows in the Burdekin region* was to examine relationships between catchment inputs of sediment and seagrass desired state, and to compare these against the 2018 Water Quality Improvement Plan's ecological targets. This objective was met using a case study in Cleveland Bay based on sediment loads from the Burdekin River and other smaller catchments that discharge into the bay (Collier et al., 2020).

The techniques developed in the Cleveland Bay case study are used in the present report at the scale of the whole GBRWHA for NESP TWQ Hub Project 5.4. To achieve this we followed three steps: (1) a consolidation and verification of seagrass data at the GBRWHA scale, (2) an analysis of the distribution of GBRWHA seagrass habitat and communities, and (3) an estimation of a desired state target for communities with sufficient data.

To achieve step 1, we compiled and standardised 35 years of seagrass survey data in a spatial database, including 81,387 georeferenced data points. Twelve seagrass species were recorded, the deepest of which (*Halophila spinulosa*) was found at 76 m. This database is a valuable resource that provides coastal managers, researchers and the global marine community with a long-term spatial resource describing seagrass populations from the mid-1980s against which to benchmark change.

For step 2, we identified 88,331 km^2 of potential seagrass habitat within the GBRWHA; 1,111 km^2 in estuaries, 16,276 km^2 in coastal areas, and 70,934 km^2 in reef areas. Thirty-six seagrass community types were defined by species assemblages. The environmental conditions that structure the location and extent of these communities included depth, tidal exposure, latitude, current speed, benthic light, proportion of mud, water type, water temperature, salinity, and wind speed. Environmental parameters interact with the topography of the reef and changes in the coastal plain, its watersheds, and its development with latitude. We describe seagrass distributions and communities that are shaped by multiple combinations of these environmental complexities and how that may influence marine spatial planning and environmental protection initiatives (Chapter 3).

For step 3, we used more than 20 years of historical data (1995-2018) on seagrass biomass for the diverse seagrass communities of the GBRWHA to develop desired state benchmarks. Of the 36 seagrass communities, desired state was identified for 25 of them, with the remainder having insufficient data. Desired state varied by more than one order of magnitude between community types, and was influenced by the mix of species in the communities and the range of environmental conditions that define community boundaries. We identified a historical, decadal-scale cycle of decline and recovery. Recovery to desired state has occurred for coastal intertidal communities following the most recent declines in 2008 - 2012. A number of the estuarine and coastal subtidal communities have not recovered to desired state biomass in recent years (Chapter 4).

This body of work provides a huge step forward in our understanding of the complexities of GBRWHA seagrass communities. We discuss the relevance of these research outputs to future marine spatial planning and management. This includes zoning in “representative areas”, hierarchical monitoring design (e.g. RIMReP), and the setting of ecologically relevant sediment load targets for desired state (e.g. Lambert et al., 2019). The updated seagrass data, seagrass distribution, community classification and desired state targets provides important new information for incorporation into marine spatial planning and management that is discussed in Chapter 5. These applications include:

- Future assessments of non-reef habitats within the GBRWHA and GBRMP.
- Assessing how risk and spatial protection intersect with seagrass communities and the role they play in protecting seagrass, e.g. Queensland State and Commonwealth marine parks, Fish Habitat Areas, Dugong Protected Areas, Port Exclusion Zones.
- Expanding our spatial analysis to areas ecologically connected but outside of the GBRWHA such as Torres Strait, the Gulf of Carpentaria, and Fraser Island coast, where we already have seagrass data.
- Designing a hierarchical seagrass monitoring design with coarse scales (intertidal, subtidal, estuary, coast, reef) and fine scales (36 communities). We have identified significant knowledge gaps that should guide future monitoring efforts (e.g. RIMReP and Queensland Land and Sea Ranger Program), including a lack of consistent and recent data for reef seagrass communities.
- We identified communities where data is deficient, such as in estuaries where important seagrass communities have potential exposure to multiple threats for which more consistent environmental data would be valuable.
- Identifying potential restoration sites.

Our work has highlighted the critical role of historical data in understanding spatial complexity and for making informed management decisions on the current state of seagrass in the GBRWHA. Our approach can be adapted for monitoring, management and assessment of pressures at other relevant scales and jurisdictions. Our results guide conservation planning through prioritisation of at-risk communities that are continuing to fail to attain desired state.

1.0 INTRODUCTION

Seagrasses form one of the most extensive and important marine coastal habitats in the world, with a diversity of 72 species in six families. They frequently co-occur as mixes of species or communities (den Hartog & Kuo, 2006; Green & Short, 2003; Jayathilake & Costello, 2018; Short et al., 2011). Seagrasses grow in a diverse range of locations, including estuaries, coastal bays, lagoons, reef-tops and open seas, intertidal through to deep subtidal; in tropical and temperate regions; and across gradients in water temperature, salinity, desiccation, bottom current stress, light and water quality (Coles et al., 2009; Jayathilake & Costello, 2018; McKenzie et al., 2020b). The critical ecosystem services seagrass communities provide make them one of the most valuable marine ecosystems on the planet (Costanza et al., 2014).

Seagrasses are one of the key ecological communities in the Great Barrier Reef World Heritage Area (GBRWHA), with extensive areas of seagrass habitat within the reef's lagoon and adjacent estuaries (Coles et al., 2015; Great Barrier Reef Marine Park Authority & Queensland Government, 2015). The ecosystem services these seagrass communities provide include substrate stabilization and improvements in water quality by filtering organic matter and microbes from the water, baffling wave and tidal energy which reduces suspended particulate matter and improving water clarity (Bainbridge et al., 2018; Lamb et al., 2017; Lewis et al., 2018; Nordlund et al., 2016). As a major benthic primary producer in the reef ecosystem, seagrasses recycle nitrogen and produce and protect carbon sinks, with benefits to the global carbon cycle (Fourqurean et al., 2012; Lavery et al., 2013; Pendleton et al., 2012; York et al., 2018), and to local water chemistry (Unsworth et al., 2012). Seagrass meadows play a critical role as food and shelter for fish and crustaceans caught by recreational, traditional and commercial fishers (Hayes et al., 2020). They provide essential food for dugongs (*Dugong dugon*) and green sea turtles (*Chelonia mydas*) (Kelkar et al., 2013; Marsh et al., 2011; Scott et al., 2018; Scott et al., 2020; Tol et al., 2016).

Zoning in the Great Barrier Reef Marine Park (GBRMP) to protect biodiversity including seagrasses and to regulate human activities has been in place since 1981, when the region became the world's first coral reef ecosystem to achieve World Heritage Area status. Early protection focused on fishing, i.e. no-take zones and habitat protection. Zoning was updated in 2004 after a lengthy process of data assessment and using a more sophisticated modelling approach (Fernandes et al., 2009). Key to the rezoning was an expert-based "Delphic consensus" approach that identified 30 reef and 40 non-reef bioregions, the protection of which was assessed against 10 biophysical principals (Fernandes et al., 2009). These principals typically directed protection of habitats along practical management lines, such as management areas being larger rather than smaller, and replicated along the length of the GBRWHA to reduce the risk of complete habitat loss. While effective for some metrics (Dobbs et al., 2008), the rezoning identified only five bioregions where seagrass was a key element (http://www.gbrmpa.gov.au/__data/assets/pdf_file/0011/17300/nonreef-bioregions-in-the-gbrmp-and-gbrwh.pdf).

Analysis of the complexity of seagrass communities requires updating, particularly for coastal waters and estuaries, with the latter largely excluded from the GBRMP and GBRWHA marine protection zoning. Estuaries and rivers adjacent to the GBRWHA are small by international standards, but their flow and sediment load variability in a monsoon-influenced coastline makes them both key attributes of the GBRWHA and sources of environmental forcing (Bainbridge et al., 2018; Lambert et al., 2019).

A series of intense tropical cyclones with associated high rainfall and flooding severely reduced seagrass biomass and extent in parts of the southern two-thirds of the GBRWHA between 2009 and 2012 (Coles et al., 2015; Collier et al., 2012; McKenna et al., 2015; Rasheed et al., 2014), and was implicated in increased stranding and mortality of marine turtles (Flint et al., 2015; Flint et al., 2017) and dugong (Flint & Limpus, 2013; Wooldridge, 2017). These events focused global concern on the resilience of coastal ecosystems to environmental disturbance, particularly in a warming climate (Brodie & Pearson, 2016; Coles et al., 2015; York et al., 2017). They highlighted the broad scales over which seagrass meadows can be impacted, an issue that is important for management responses to address and is germane to the concept of risk and replication in park planning and zoning. Catchment-derived pollutants, particularly sediment loads, were linked to those seagrass losses and this highlighted the catastrophic consequences for seagrass of declines in water quality and available light (Coles et al., 2015; Collier et al., 2012; McKenna et al., 2015; McKenzie et al., 2012; Petus et al., 2014; Rasheed et al., 2014; Schaffelke et al., 2017; Waterhouse et al., 2017). Seagrasses in the region also are vulnerable to local disturbances such as those associated with ports and coastal developments (Grech et al., 2011; York et al., 2015). The challenge for researchers providing advice to management at these scales is exacerbated by the high bar set by the reef management authority's objective to "maintain diversity of species and ecological habitats in at least a good condition and with a stable to improving trend" (Great Barrier Reef Marine Park Authority & Queensland Government, 2015).

The research program reported here evolved from an interest in developing ecologically relevant target criteria that, if met, correspond to desired ecological outcomes for the GBRWHA (e.g. desired state) and to achieving the over-arching objective of the Great Barrier Reef Marine Park Authorities' Long-term Sustainability Plan. We use the techniques developed in a case study in Cleveland Bay (Collier et al., 2020) to set desired state targets at the scale of the GBRWHA. To achieve this we followed three steps outlined in this report:

1. Compile seagrass spatial data collected within the GBRWHA into a standardized form with site-specific spatial and temporal information, and to make this data available for the global marine research community on eAtlas (Chapter 2).
2. Use the seagrass data synthesis created in Chapter 2 to better understand seagrass and seagrass community structuring at the GBR-scale by: (1) defining potential seagrass habitat for the GBRWHA; (2) classifying the diversity of seagrass communities within seagrass habitat; and (3) determining the environmental conditions that allow for the presence of seagrass habitat and influence the composition of seagrass communities (Chapter 3).
3. Define desired state for the extensive and diverse seagrass habitats in the GBRWHA and adjacent estuaries for the 36 identified seagrass communities identified in Chapter 3 (Chapter 4).

We discuss the relevance of these research outputs to future marine spatial planning and management.

2.0 SYNTHESIZING 35 YEARS OF SEAGRASS SPATIAL POINT DATA

2.1 Pre-amble

This chapter is based on an article that has been submitted for publication as a data article: Carter A.B., McKenna S.A., Rasheed M.A., Collier C., McKenzie L., Pitcher R., and Coles R. (In Review) Synthesizing 35 years of seagrass spatial data from the Great Barrier Reef World Heritage Area, Queensland, Australia. *Limnology & Oceanography Letters*.

The data synthesis is an outcome of successive investment through NESP TWQ Hub Project 3.1, NESP TWQ Hub Project 3.2.1 and this NESP TWQ Hub Project 5.4. The data underpins the community classification (Chapter 3) and desired state (Chapter 4) analysis. The data are available on the eAtlas data portal: https://doi.org/10.25909/y1yk-9w85

2.2 Executive summary

We compiled and standardised 35 years of data in a spatial database, including 81,387 data points with georeferenced seagrass and species presence/absence, depth, dominant sediment type, and collection date. We include records collected under commercial contract being made available for the first time here. Twelve seagrass species were recorded. The deepest seagrass was found at 76 m. Our database is a valuable resource that provides coastal managers and the global marine community with a long-term spatial resource describing seagrass populations from the mid-1980s against which to benchmark change.

2.3 Background and objectives

Key to addressing the challenges marine ecosystems face around the world is access to data for analysis and comparison at appropriate spatial and temporal scales in a user-friendly format. Such data can be used for describing the present condition of ecosystems; understanding long-term trends while accounting for short-term impact-recovery cycles; defining the desired state of the diversity of habitats; establishing ecologically relevant targets that can be used to maintain resilience; and implementing appropriate management frameworks that maintain resilience (Brodie et al., 2017; Collier et al., 2020; Hallett et al., 2016b; Levin & Möllmann, 2015; O'Brien et al., 2017; York et al., 2017). To this end, there is an increasing use of Geographic Information Systems (GIS) to record, synthesize, and analyse data and to benchmark previous states to inform research, conservation, ecosystem-based management, and marine spatial planning (St. Martin, 2004; St. Martin & Hall-Arber, 2008).

Within the GBRWHA, seagrass research extends back to the 1970s (Birch & Birch, 1984) but data collection with a major spatial/mapping focus did not commence until the mid-1980s. Mapping projects since that time range from surveys quantifying seabed benthic cover across the entire GBRWHA funded by a range of government agencies, to those collected under industry contracts for specific areas and where covenants on their use and availability may be in force (Table 1).

We compiled seagrass spatial data collected during surveys within the GBRWHA and adjacent estuaries into a standardized form with site-specific spatial and temporal information (Figure 1). We revisited, evaluated, simplified, standardized and corrected individual records, including those from two to three decades ago by drawing on the knowledge of one of our authors (Coles) who led the early seagrass data collection and mapping programs. Our objective was to provide this extensive seagrass data set, along with an interactive website, as a tool for the global marine research community to interrogate species distributions and to benchmark trends through time in this iconic World Heritage Area.

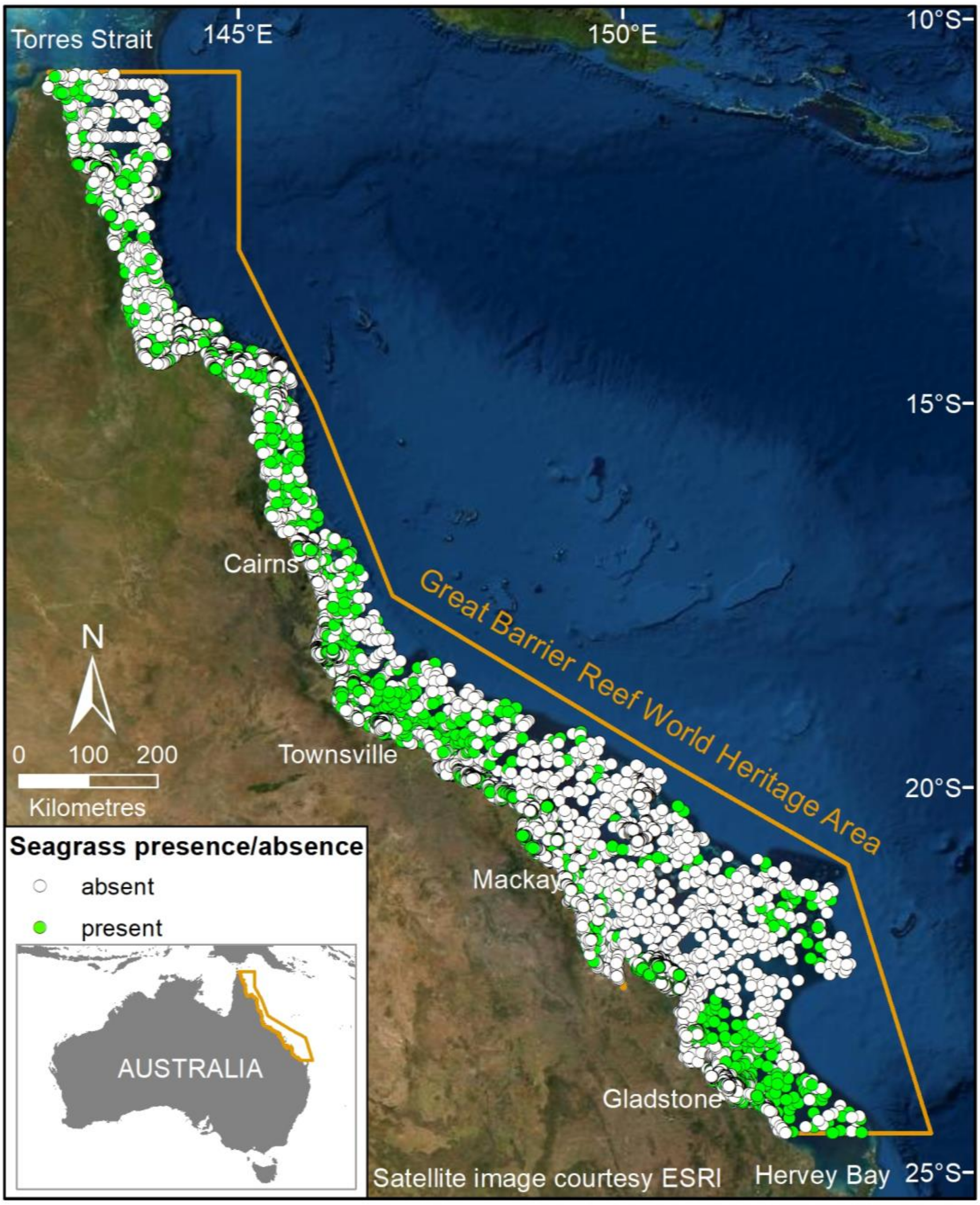

Figure 1: Seagrass presence and absence at individual sampling sites across the Great Barrier Reef World Heritage Area (orange boundary). Satellite image courtesy: ESRI

Table 1: Survey purpose and location of spatial data used in seagrass data compilation, 1984-2018.

Survey purpose/ Data location	Year/s	Reference
(1) 1980s GBRWHA-scale coastal surveys		
Cape York to Cairns	1984, 1985	(Coles et al., 1985)
Cairns to Bowen	1987	(Coles et al., 1992)
Bowen to Water Park Point	1987	(Coles, 1987)
Water Park Point to Hervey Bay	1988	(Coles et al., 1990)
(2) GBRWHA-scale deep-water surveys		
GBRWHA Deep-Water	1994-1999	(Coles et al., 2009)
GBRWHA Seabed Biodiversity	2003-2005	(Pitcher et al., 2007)
(3) Oil spill response atlas (OSRA) intertidal surveys		
Princess Charlotte Bay to Cape Flattery	2011-2014	(Carter et al., 2013; Carter et al., 2012; Carter & Rasheed, 2014; Carter & Rasheed, 2015)
Hydrographers Passage	2003	(Rasheed et al., 2006)
Margaret Bay	2001	(Rasheed et al., 2005)
(4) Targeted seagrass mapping surveys		
Bustard Bay	2009	(Taylor et al., 2010)
Cape Flattery	1996	(Ayling et al., 1997)
Clairview	2017-2018	(Carter & Rasheed, 2019)
Clump Point	1997	(Roder et al., 1998)
Dugong Protection Area	1999	(Coles et al., 2002)
Dunk Island to Cleveland Bay	1996	Unpublished data
Edgecumbe Bay	2008	(Coles et al., 2007)
Green Island	1997, 2003	(McKenzie & Lee Long, 1996; McKenzie et al., 2014b)
Lizard Island	1995	(McKenzie et al., 1997)
Low Isles	1997	(McKenzie et al., 2016)
Lucinda to Bowling Green Bay	2007	(Coles et al., 2007)
Oyster Point	1995-1998	(Lee Long et al., 2001)
Shoalwater Bay	1996	(Lee Long et al., 1996a)
Whitsunday Islands	1999-2000	(Campbell et al., 2002)
(5) Queensland ports seagrass long-term monitoring surveys		
Cairns	1993, 1996, 2000-2018	(Lee Long et al., 1996b; Rasheed & Roelofs, 1996; Rasheed et al., 2019)
Gladstone	2002-2018	(Chartrand et al., 2019)
Mackay and Hay Point	2001-2018	(Rasheed et al., 2001; York & Rasheed, 2019)
Abbot Point	2005-2018	(McKenna et al., 2019)
Mourilyan Harbour	1993-2018	(Wells et al., 2019)
Townsville	2007-2018	(Bryant et al., 2019)

2.4 Data description

This data set is a compilation of spatial data from seagrass surveys in the GBRWHA from 1984 to 2018. Data were collected for five major survey purposes: (1) an original project that mapped all coastal seagrass to ~15 m depth in the 1980s; (2) extensive cross-shelf subtidal surveys in the early to mid-1990s and again in 2003-2005; (3) mapping of intertidal areas as part of an oil spill response atlas, commencing in 2001; (4) smaller and more targeted mapping projects such as for Dugong Protected Area surveys; and (5) a comprehensive series of mapping and monitoring projects for Queensland ports that in some cases extend back more than 20 years (Table 1). In total, the data set has 81,387 data points that can be viewed interactively through eAtlas or downloaded.

Mapping data for historic records (1980s) were transcribed from original logged and mapped data based on coastal topography, dead reckoning fixes and RADAR estimations (McKenzie et al., 2014a). More recent data (1990s onwards) is GPS located. A range of site descriptions and contextual information is contained in original trip reports and publications (Table 1). Details for each survey site include: latitude and longitude, depth in metres below mean sea level (MSL), overall seagrass presence/absence, individual seagrass species presence/absence, dominant sediment type, survey month and year, survey name, and sampling method. Seagrass data is limited to the extent of the GBRWHA, with the exception that estuarine seagrass data that extended west into State of Queensland waters is included. Seagrass distributions generated from modelled data (Coles et al., 2009; Grech & Coles, 2010) are not included in this data set.

The twelve seagrass species included in this data set are: *Cymodocea rotundata* (Ascherson & Schweinfurth, 1870), *Cymodocea serrulata* ((R.Brown) Ascherson & Magnus 1870), *Enhalus acoroides* ((Linnaeus f.) Royle, 1839), *Halophila capricorni* (Larkum, 1995), *Halophila decipiens* (Ostenfeld, 1902), *Halophila ovalis* ((R.Brown) J. D. Hooker, 1858), *Halophila spinulosa* ((R.Brown) Ascherson, 1875), *Halophila tricostata* (Greenway), *Halodule uninervis* ((Forsskål) Ascherson, 1882), *Syringodium isoetifolium* ((Ascherson) Dandy, 1939), *Thalassia hemprichii* ((Ehrenberg) Ascherson, 1871), and *Zostera muelleri* subsp. *capricorni* ((Ascherson) S. W. L. Jacobs, 2006).

Data, metadata and the interactive website are available at eAtlas at https://doi.org/10.25909/y1yk-9w85 (Carter et al., 2020). We intend this data to be used as a stand-alone product or integrated with other publically available biophysical data sets and models (e.g. https://ereefs.org.au/ereefs) to explain distributions and change. We include and make available data not previously available to the public.

2.5 Methods

All spatial data were converted to point shapefiles with the same coordinate system (GDA94), then compiled into a single point shapefile using ArcMap (ArcGIS version 10.7 Redlands, CA: Environmental Systems Research Institute, ESRI). We include 12 seagrass species that were identified using *in situ* observations (Figure 2). Species names have been updated to meet recent taxonomic changes and to ensure consistency in species names in the compilation.

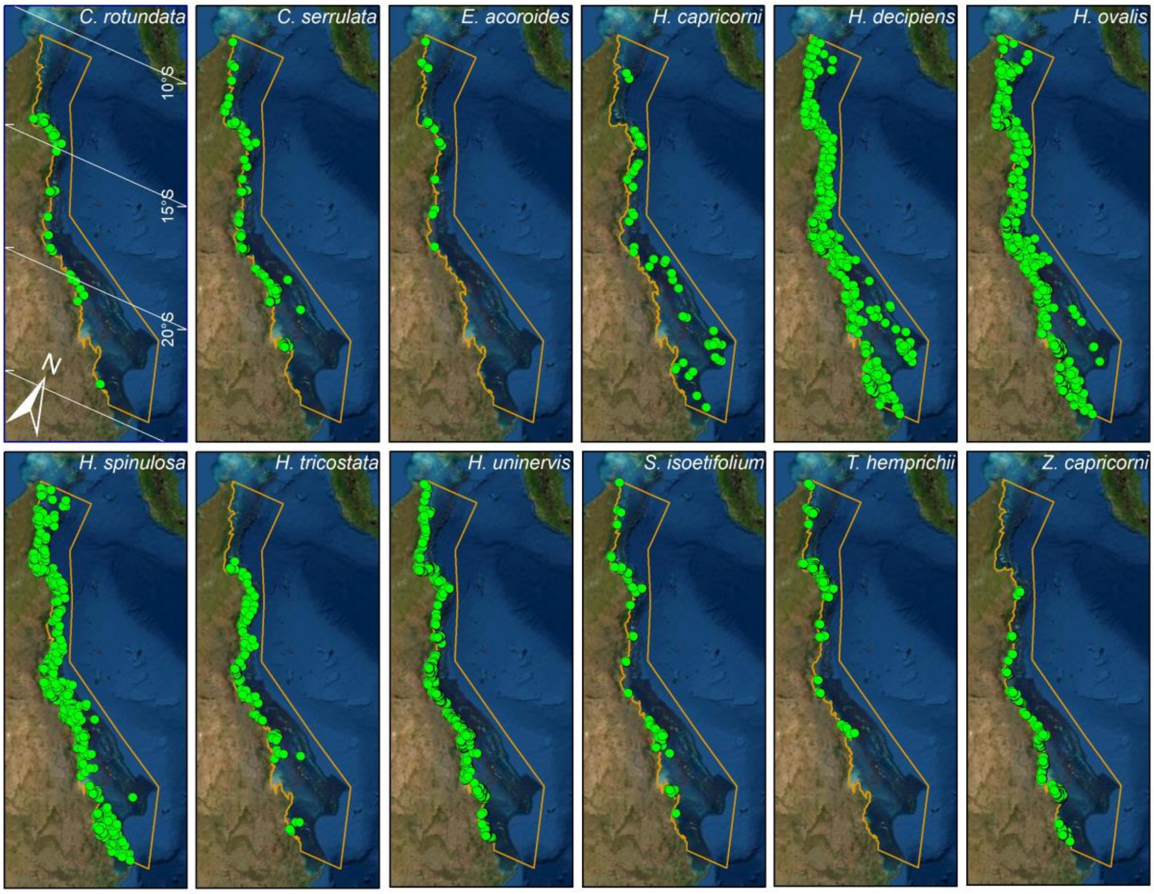

Figure 2: Distribution of 12 seagrass species in our data set (green dots) throughout the Great Barrier Reef World Heritage Area (orange boundary) observed in the site data synthesis, 1984-2018. Sites were surveyed to a depth of 117 m but seagrass presence was not recorded deeper than 76 m. Satellite image courtesy: ESRI.

The data were collected using a variety of survey methods. These include walking and diving; recorded video transects from towed cameras; observations from helicopters in low hover; trawl and net samples; and van Veen grab samples. These sampling methods to study, describe and monitor seagrass meadows were implemented by the JCU TropWATER Seagrass Group (the Marine Ecology Group at Queensland Fisheries prior to 2013) and CSIRO, and tailored to the location and habitat surveyed, and are described in detail in the relevant publications for each data set provided in Table 1. For long-term monitoring data sets, the most recent report is referenced. In this compilation we have updated and standardized the terms used to describe survey methods. We have only included spatial datasets where the primary purpose was mapping and that had sufficient metadata available for interpretation.

Sediment type in the original data sets were based on grain size analysis or deck descriptions. For consistency, in this compilation we include only the most dominant sediment type (mud, sand, shell, gravel, rock, rubble), removed descriptors such as “fine”, “very fine”, “coarse”, etc., and replaced redundant terms, e.g. “mud” and “silt” are termed “mud”.

Depth (m below MSL) for each subtidal site was extracted from the gbr30 data set (Beaman, 2017). Depth for intertidal sites was recorded as 0 m MSL.

2.6 Technical validation

The original seagrass data comes from a variety of surveys conducted for different purposes. Only two projects, the Great Barrier Reef Deep-Water and Seabed Biodiversity (Table 1), were sampled in a systematic way across the entire GBRWHA. In the 2009-2018 period, survey coverage is relatively small as the focus of most surveys shifted from large-scale baseline mapping to smaller-scale annual long-term monitoring, particularly in ports (Figure 3). For early data (1980s and 1990s), each data point was reviewed and compared with original trip logs and recollections of trip participants.

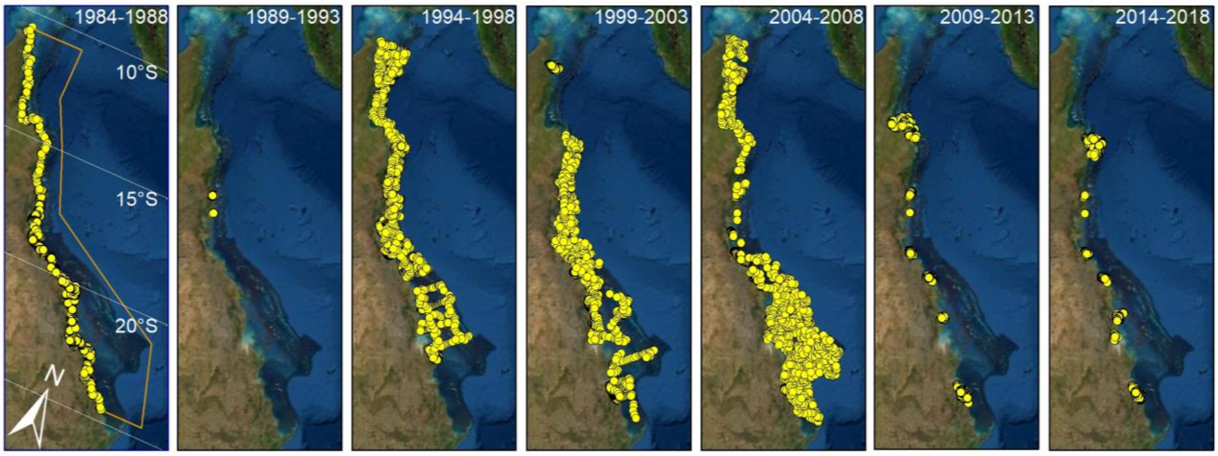

Figure 3: Distribution of sampling sites (yellow dots) throughout the Great Barrier Reef World Heritage Area (orange boundary) in 5-year increments, 1984-2018. Satellite image courtesy: ESRI

Most data (80%) comes from the austral growing season (August – January). Seagrasses were recorded at around 40% of sites; however, many surveys were targeted at known meadows (e.g. ports long-term monitoring annual surveys of designated monitoring meadows) so sites were not always randomly assigned to include areas unlikely to have seagrass. Sites include depths down to 123 m; the deepest recorded seagrass was *H. spinulosa* at 76 m. Only species of the genus *Halophila* were found in sites below 40 metres.

There is an earlier version of the seagrass site data on eAtlas which includes similar information which is still available (1984-2014; https://eatlas.org.au/nesp-twq-1/gbr-seagrass-mapping-3-1). As with all data sets with large temporal and spatial coverage there are survey-specific limitations and we recommend contacting the JCU data custodians when using this data to ensure those limitations are understood.

2.7 Data use and recommendations for reuse

Worldwide, the management and conservation of marine ecosystems requires accurate spatial data at a scale that matches human activities and impacts (Halpern et al., 2008; Hughes et al., 2005; Lagabrielle et al., 2018; Visconti et al., 2013). The synthesis of large spatial distribution data sets provides a valuable tool that can be used to inform marine spatial planning, ecosystem based management, research, and education (Halpern et al., 2008). A key strategy to assist this is to ensure we validate and share data that has been collected over the years (Rajabifard et al., 2005). This project makes publicly available one of the world's most comprehensive seagrass data sets. Importantly, we include site data from previously unreleased industry-funded surveys. Also important for users of this data is we include location information not just for sites that were surveyed and seagrass recorded, but also location information where seagrass was absent.

Spatial data is an increasingly important tool in the assessment and management of the marine environment (Hughes et al., 2005; Rajabifard et al., 2005; St. Martin & Hall-Arber, 2008). The immediate scientific value of this project and its approach already has been widely demonstrated, with earlier subsets of this synthesis used to answer a range of key ecological questions including: probability modelling of seagrass distributions in the GBRHWA's deep-water lagoon (Coles et al., 2009) and inshore region (Grech & Coles, 2010); seagrass risk exposure modelling (Grech et al., 2012; Grech et al., 2011); propagule distribution (Grech et al., 2016); connectivity among meadows (Grech et al., 2018; Tol et al., 2017); understanding changes in seagrass meadow health using MODIS imagery (Petus et al., 2014); and defining the desired state of seagrass communities in the Townsville region (Collier et al., 2020; Lambert et al., 2019). We now make available the data behind these analyses and data updated to 2018 for the global community to use and compare.

3.0 SEAGRASS HABITAT AND COMMUNITY CLASSIFICATION

3.1 Pre-amble

This chapter is based on a manuscript that has been submitted for publication:
Carter A.B., Collier C., Lawrence E., Rasheed M.A., Robson, B. and Coles R. (Submitted) Defining seagrass habitat and community diversity in the Great Barrier Reef lagoon – A machine learning approach for spatial planning and management, Queensland, Australia. *Diversity and Distributions*.

This analysis is an outcome of investment through NESP TWQ Hub Projects 3.1 and 3.2.1 and this NESP TWQ Hub Project 5.4. The seagrass probability model and community classification uses the seagrass data synthesis (Chapter 2) and underpins the desired state analysis (Chapter 4).

Seagrass models are available on the eAtlas data portal.
Seagrass habitat: https://eatlas.org.au/data/uuid/108ee868-4fb1-4e5f-ae57-5d65198384cc
Seagrass communities: https://eatlas.org.au/data/uuid/313183fe-de3a-4874-bcba-d13d4ae4ecbc

3.2 Executive Summary

In this analysis we identify 88,331 km^2 of potential seagrass habitat; 1,111 km^2 in estuaries, 16,276 km^2 in coastal areas, and 70,934 km^2 in reef areas. Thirty-six seagrass community types were defined by species assemblages. The environmental conditions that structure the location and extent of these communities included depth, tidal exposure, latitude, current speed, benthic light, proportion of mud, water type, water temperature, salinity, and wind speed. These environmental parameters interact with the topography of the reef lagoon and changes in the coastal plain, its watersheds, and with latitude. The influence of the coast diminishes with distance off-shore, changes in terrestrial sediment fluxes, and with depth. We describe seagrass distributions and communities that are shaped by multiple combinations of these environmental conditions and how that may influence marine spatial planning and environmental protection initiatives.

3.3 Introduction

Coastal marine habitats are some of the most at-risk ecosystems in the world (Halpern et al., 2008). Proximity to land-based anthropogenic activities exposes these habitats to threats from multiple stressors (Wilson et al., 2007). The scale and complexity of marine habitats and the high cost of sampling them means the data used to inform management is often less precise than for equivalent terrestrial systems (Carr et al., 2003). Unlike terrestrial locations, comprehensive aerial and satellite full coverage mapping is restrained by light attenuation and turbidity in the water column. This is often compounded by significant gaps in the data available on important covariates such as risk and threats, asymmetry in ecological connectivity, a lack of long-term historical data, enormous variations in scale, and poorly documented temporal cycles of impacts and recovery (Beger et al., 2010; Brodie & Waterhouse, 2012; Coles et al., 2015). For some coastal and marine habitats, it is difficult to detect even large changes in their status and distribution with current levels of monitoring consistency and spatial coverage.

Maintaining the resilience of important coastal marine habitats at large regional scales presents challenges for scientists and managers because of their enormous extent and inherent spatial and temporal variability. These challenges include describing ecosystem condition, understanding long-term trends while evaluating short-term impact-recovery cycles, defining desired state for the diversity of habitats and communities or assemblages within a habitat, and determining ecologically relevant targets (e.g. river load targets) (Brodie et al., 2017; Kroon, 2012) that will maintain healthy habitats (e.g. desired state) (Collier et al., 2020; Fernandes et al., 2019; Lambert et al., 2019). Compounding these challenges is the difficulty of advising on and implementing appropriate management frameworks to maintain resilience within multiple priority habitats, and time and investment constraints typically faced by marine management agencies.

Spatial data and visualization techniques are increasingly important in addressing the challenge of understanding and communicating options for managing large and complex coastal marine habitats. Habitat and community maps are a frequently used spatial tool that visualise and evaluate the association of species and communities of interest with key environmental drivers likely to affect those communities (Greene et al., 2007). The spatial representation of habitats and communities, and the ability to capture the range of environmental features that support biological diversity, provide the foundation for large-scale spatial assessments of where habitats and communities are likely located (Greene et al., 2007), levels of connectivity (Grech et al., 2018; Grech et al., 2016), understanding spatial and temporal change (Young & Carr, 2015), and defining desired state (Collier et al., 2020). These spatial representations are a critical component of marine spatial planning, particularly in resolving spatial conflicts, incorporating indigenous knowledge and aspirations into the planning process, defining management units such as marine protected areas, and designing representative monitoring programs (Diggon et al., 2019; Foley et al., 2010; Kenchington & Day, 2011; Noble et al., 2019).

Four broad classifications have been applied to describe seagrasses in the GBRWHA: estuarine, coastal, deep-water (subtidal), and reef - based on which dominant environmental factor is limiting - terrigenous runoff, physical disturbance, low light, and low nutrients, respectively (Carruthers et al., 2002; Coles et al., 2015; Waycott et al., 2005). Seagrass communities within these categories are diverse and complex (Collier et al., 2020). Previous GBRWHA-scale seagrass models have focussed on overall seagrass distribution or on the distribution of single species. These models were limited by data availability to specific regions (e.g. coastal, deep-water); and/or were at a spatial scale (e.g. $\geq$1 km grids) too large to capture the smaller-scale (metres) areas of seagrass such as narrow intertidal bands within estuaries; or were part of much larger modelling projects that excluded the possibility of using detailed GBRWHA-specific environmental data (Coles et al., 2009; Grech & Coles, 2010; Jayathilake & Costello, 2018).

Our seagrass data consolidation (Chapter 2) combined with greatly improved environmental data and models provide an opportunity to model seagrass habitat in much more detail than previously possible and, for the first time, to model seagrass community types throughout the GBRWHA. This recent data includes high resolution GBRWHA-wide models of environmental conditions that influence seagrass distribution. These include models for depth (Beaman, 2017), tidal exposure (Bishop-Taylor et al., 2019), hydrodynamics (Steven et al., 2019), benthic light (Baird et al., 2016; Baird et al., 2020) and sediment (Baird et al., 2020; Margvelashvili et al., 2018).

Our objective was to use this new information to better understand seagrass and seagrass community structuring at the Great Barrier Reef scale by: (1) defining potential seagrass habitat for the GBRWHA; (2) classifying the diversity of seagrass communities within seagrass habitat; and (3) determining the environmental conditions that allow for the presence of seagrass habitat and influence the composition of seagrass communities. This analysis will allow informed decisions at an appropriate spatial scale in marine spatial planning, management, monitoring, evaluating and mitigating risk, and restoration.

3.4 Methods

3.4.1 Study area

The Great Barrier Reef is one the world's most extensive coral reef structures, an environment home to a globally outstanding and biodiverse marine ecosystem. The GBRWHA covers an area of around 350,000 square kilometres, including 2,500 kilometres of coastline and a shelf that extends up to 250 kilometres offshore. Extensive seagrass meadows stretch along intertidal banks and reef-tops, and extend from coastal estuaries to offshore inter-reef waters. Meadows within the GBRWHA range from tropical (10°S) to subtropical (~25°S) (Coles et al., 2015), and also extend north and south of GBRHWA boundaries into Torres Strait (Carter et al., 2014) and south-east Queensland (Lee Long et al., 1993; Maxwell et al., 2019). Our study area covers coastal and reef areas in the continental shelf region of the GBRWHA where depth below mean sea level is generally <100 m, and the adjacent estuaries along the mainland Australian coast (Figure 4).

The enormity of the GBRWHA means there is large variation in geography, topography, and environmental conditions (Hopley et al., 2007; Wolanski, 1994). The northern GBRWHA (< ~16°S) is characterised by a narrow shelf, shallow inter-reef waters (<30m), elongate reefs, warmer water temperatures, high benthic light, low current speed, and low salinity (Figure 4). The central GBRWHA (~16 and 20°S) is characterised by lower reef density, intermediate inter-reef depths (> 40 m), low current speed, low salinity, and low wind speed (Figure 4). The southern GBRWHA (> ~20°S) is characterised by high reef density situated in deep water (down to 140 m) across a wide continental shelf, high salinity, high current speed, cooler water and lower mud content in the sediment (Hopley et al., 2007; Wolanski, 1994). There are also major regional differences along the coast adjacent to the reef lagoon in climate, land type, and land use, e.g. tropical and subtropical; wet and dry tropics; pristine, sugar cane or cattle-dominated catchments (Hopley, 1986; Waycott et al., 2005). Adding to this complexity is a coastal mountain range that in the northern GBRWHA runs close to the coast with mostly small watersheds and short rivers compared with the central and southern GBRWHA. The human population is concentrated along coastal communities of the central and southern coast. Threats and risk to coastal seagrass integrate these broad trends (Grech et al., 2011; Rasheed et al., 2007).

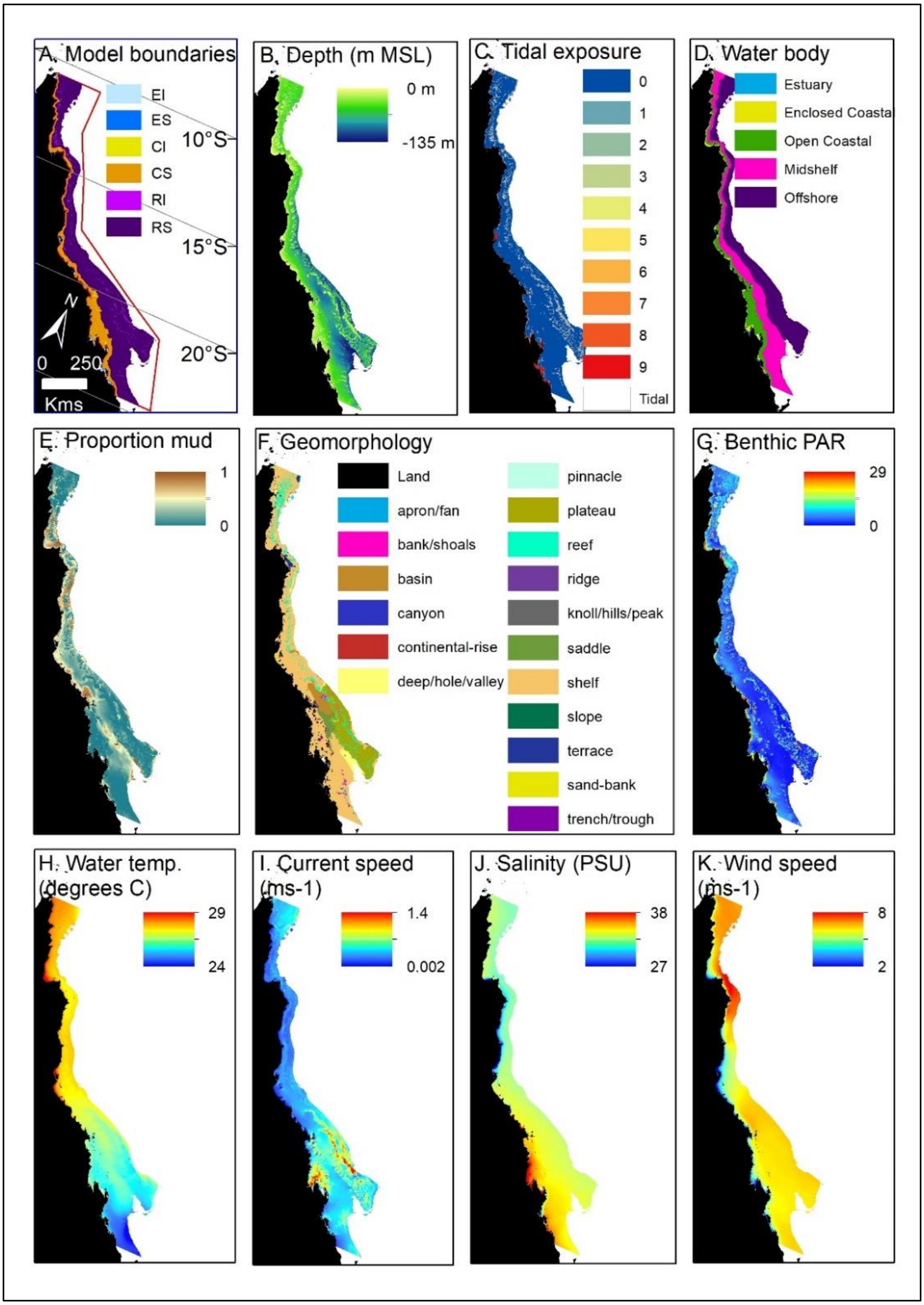

Figure 4: (A) Seagrass model boundaries (EI, estuary intertidal; ES, estuary subtidal; CI, coastal intertidal; CS, coastal subtidal; RI, reef intertidal; RS, reef subtidal). Red outline is the Great Barrier Reef World Heritage Area. (B-K) environmental variables used in random forest models to predict potential seagrass habitat, and multivariate regression trees to predict seagrass community types. Data sources listed in Section 3.4.4. See Appendix 1 for a large version of (A).

3.4.2 Seagrass data

Seagrass presence/absence data comes from a synthesis of seagrass surveys collected throughout the GBRWHA and adjacent estuaries between 1984 and 2018 (Chapter 2).

Long-term monitoring programs in ports quantified the impact of a series of intense tropical cyclones with high rainfall and flooding that severely reduced seagrass presence and altered species composition along the southern two-thirds of the GBRWHA between 2009 and 2012. Recovery has been variable among locations (Coles et al., 2015; Collier et al., 2012; McKenna et al., 2015; Petus et al., 2014; Rasheed et al., 2014). Previous seagrass community analysis demonstrates species assemblages during and after major disturbance events are disproportionally dominated by colonising species, leading to an overly simplistic community classification relative to seagrass diversity present under "average" conditions (Collier et al., 2020). Because of this, ports long-term monitoring data was excluded from our analysis if overall seagrass condition was classed as poor or very poor according to the annual report card produced for each of those locations (McKenna et al., 2020; Reason et al., 2020; Smith et al., 2020; Van De Wetering et al., 2020a; Van De Wetering et al., 2020b; York & Rasheed, 2020). This avoided defining seagrass community types based on data that overwhelmingly represented a significant environmental impact, rather than average environmental conditions (e.g. sediment type, depth). This process was applied only to ports data because these were the only locations in the central and southern GBRWHA where sampling occurred during 2009-2012.

Data was also restricted to the seagrass growing season (August-January; approx. 80% of sites) to reduce the likelihood of including times and sites in the analysis where seagrass was absent due to the seasonal and ephemeral nature of some species. This is particularly important for deep-water *Halophila* communities, which may be present only as a seed bank through the colder months of the year (Chartrand et al., 2018; York et al., 2015).

3.4.3 Models and environmental predictors

We used six models: estuary intertidal, estuary subtidal, coast intertidal, coast subtidal, reef intertidal and reef subtidal (Figure 4A; Appendix 1). This separation was used as it accounted for variation in availability of environmental data (e.g. lack of environmental data for estuaries), variation in seagrass sampling history and intensity (e.g. a gradient in sampling intensity that decreases with distance from the Australian mainland coast, and with depth), and well-established general differences in seagrass species distributions (e.g. intertidal versus subtidal species) (Coles et al., 2009; Collier et al., 2020; Rasheed et al., 2014).

For each site we used the following spatial data to quantify the average environmental conditions at each site and, to assign each site to one of the six models:

- Depth (metres below mean sea level; subtidal sites only) – *numeric data*. Depth below mean sea level (MSL) data for subtidal sites only was extracted from the gbr30 (30m pixel resolution) raster (Figure 4B) (Beaman, 2017). Deep waters extending east of the continental shelf and beyond the historical seagrass data set were excluded (< approx. -100 m). Depth was not included as a variable in intertidal models.
- Tidal exposure – *categorical data*. Relative tidal exposure obtained from the Intertidal Extent Model (ITEM version 2.0) with categorical bands 1-9, where 1 = exposed 0-

10% of the time, 9 = exposed 80-100% of the time, and 0 = areas of water across the observed tidal range (Figure 4C) (Bishop-Taylor et al., 2019; Geoscience Australia, 2017).

- Intertidal/subtidal – categorical data. Used to define and to separate data for the intertidal and subtidal models. Sites were classed as intertidal if they fell within the ITEM (intertidal extent model) bands 1-9, or were classed as tidal regions of reefs or shoals within Queensland maritime waters (© State of Queensland (Department of Natural Resources, Mines and Energy) 2019), even where ITEM = 0. This allowed for the inclusion of sites particularly on intertidal reef-tops known from helicopter field surveys to expose during spring tides but that were not defined as intertidal by the ITEM model (Figure 4C). These intertidal sites were classed as 0 for tidal exposure. All other sites were classed as subtidal.
- Water type – *categorical data*. Sites were classed as estuarine if they were within the Queensland coastal waterways geomorphic habitat mapping estuary boundary (Dyall et al., 2004). Non-estuarine water types were classified according to the Marine Water Bodies definitions (version 2_4; Data courtesy of the Great Barrier Reef Marine Park Authority): enclosed coastal, open coastal, mid-shelf and offshore (Figure 4D). Values for sites outside these layers were estimated using the nearest water type polygon. These water types were also grouped more broadly to define the estuary, coast (enclosed and open coastal) and reef (mid-shelf and offshore) models.
- Sediment – *numeric data*. For coastal and reef sites the proportion of mud for each site was extracted from the eReefs 1 km grid hydrodynamic model, available at: https://research.csiro.au/ereefs/models/model-outputs/access-to-raw-model-output/ (see also Baird et al., 2020; Margvelashvili et al., 2018). Values were based on an arbitrary date of 30 January 2018, which incorporates modelled sediment movement leading up to that date (Figure 4E). For enclosed and open coastal sites outside the raster extent, proportion of mud was estimated by inverse distance weighted (IDW) interpolation. For estuarine seagrass, sediment was classified as the dominant sediment type from field descriptions (Chapter 2) (categorical data) because the proportion mud raster excluded most estuarine seagrass data and interpolation was not appropriate or the 1 km grid size was too coarse for application within the narrow tidal bands of estuaries.
- Benthic geomorphology – *categorical data*. Sites were categorised by geomorphic (benthic) features as defined by the Geomorphic Features of the Australian Margin (Figure 4F) (Heap & Harris, 2008). Benthic geomorphology was excluded from estuary analysis.
- Benthic light – *numeric data*. This was extracted from the eReefs 1 km grid biogeochemical and optical model (v924) “EpiPAR_sg” variable, representing benthic photosynthetically active radiation (PARb) above the seagrass canopy in mol photons $m^{-2}d^{-1}$. These data are available from: http://dapds00.nci.org.au/thredds/catalog/fx3/gbr1_bgc_924/catalog.html (see also Baird et al., 2016; Baird et al., 2020). Values are an aggregation of daily benthic light from 2003 to 2019. Data is available for both wet and dry season; we used the dry season data because the two data sets were highly correlated (Figure 4G). Values for coastal sites outside the raster extent were estimated by IDW interpolation. Benthic light was excluded from the estuary analysis.

- Water temperature, mean current speed, and salinity - *numeric data*. This was extracted from the eReefs 1km grid hydrodynamic model representing water temperature (°C; Figure 4H), mean current speed (ms^{-1}; Figure 4I), and salinity (Practical Salinity Unit, PSU; Figure 4J) at -2.35 m depth below mean sea level, available at: https://data.aims.ereefs.org.au/thredds/fileServer/derived-download/gbr1_2.0/all-one/all-one.nc (Steven et al., 2019). Values for each data set are an aggregation of daily data from 12/2014 to 03/2019, which is then aggregated to monthly data and averaged over the year. Values for coastal sites outside the raster extent were estimated by IDW interpolation. These were all excluded from estuary analysis.
- Wind speed - *numeric data*. This was extracted from the eReefs 1km grid hydrodynamic model representing wind speed (ms^{-1}), available at: https://data.aims.ereefs.org.au/thredds/fileServer/derived-download/gbr1_2.0/all-one/all-one.nc. Wind speeds used by the eReefs models are derived from the Australian Bureau of Meteorology's ACCESS data products (Bureau of Meteorology, 2020a; Soldatenko et al., 2018; Steven et al., 2019) (Figure 4K). Values used here are an aggregation of daily data from 12/2014 to 03/2019, which is then aggregated to monthly data and averaged over the year. Values for coastal sites outside the raster extent were estimated by IDW interpolation. Wind speed was excluded from estuary analysis because of the effects of local topography that we could not account for.
- Latitude – *numeric data*. Environmental data useful to model estuarine seagrass distribution was limited, so latitude was included in the estuary models as a proxy for the north-south gradient in environmental conditions that was evident in the non-estuarine environmental data sets described above.

3.4.4 Statistical analysis

We conducted a two-step analysis to (1) define potential seagrass habitat, then (2) classify seagrass communities within that habitat. To define potential seagrass habitat, we used the machine learning technique random forest (RF) to examine the probability of seagrass occurrence irrespective of species. The RF method is a non-parametric tree-based analysis that generates multiple classification or regression trees, each calibrated on a bootstrap sample of the original data using a subset of the predictor variables, with the model prediction calculated as the average value over the predictions of all the trees in the forest (Breiman, 2001). The accuracy of the RF model depends on the predictive power of each tree and the correlation between trees (Breiman, 2001).

Random forest models were implemented using the *randomForest* package (Liaw & Wiener, 2002) in R version 4.0.2 (R Core Team, 2020). For each RF model, seagrass presence/absence (1/0) data was randomly partitioned into training (80% of data set) and testing (remaining 20%) datasets (Table 1). For each model, we set the number of classification trees (ntree) to 500. The optimal number of predictor variables to be randomly selected at each node (mtry) was determined by tuning each model (Table 2). The importance of predictor variables was assessed using the mean decrease in accuracy. Variables included in each model were plotted using the *plotmo* package (Milborrow, 2020) where, for each plot, the background variables are held fixed at their median values (calculated from the training data). Each model was validated using a confusion matrix derived from the independent validation (test) data, using the *caret* package in R (Kuhn 2020). A confusion matrix shows agreement and disagreement in a table format, with predicted values forming the matrix columns and observed values forming the rows. From this matrix we calculated the total accuracy (i.e., percentage of sites correctly classified) and accuracy for each class (present/absent).

To avoid the issue of multicollinearity of environmental variables in our models we calculated variance inflation factors (VIFs) for all environmental variables. Highly correlated variables (VIF >3) were removed prior to analysis: tidal range (collinear with water temperature) was not included in any model; beyond that, collinearity and the variables excluded differed among models. Variables available in the RF models were:

(1) $RF_{(estuary,\ intertidal)}$ ~ Tidal exposure + Latitude + Sediment
(2) $RF_{(estuary,\ subtidal)}$ ~ Depth + Latitude + Sediment
(3) $RF_{(coast,\ intertidal)}$ ~ Current speed + Tidal exposure + PARb + Proportion mud + Salinity + Water temperature + Water type + Wind speed
(4) $RF_{(coast,\ subtidal)}$ ~ Current speed + Depth + Geomorphology + PARb + Proportion mud + Salinity + Water temperature + Water type + Wind speed
(5) $RF_{(reef,\ intertidal)}$ ~ Tidal exposure + Geomorphology + PARb + Proportion mud + Water temperature + Water type + Wind speed
(6) $RF_{(reef,\ subtidal)}$ ~ Current speed + Depth + PARb + Proportion mud + Water temperature + Water type + Wind speed

Table 2: Random Forest (RF) and Multivariate Regression Tree (MRT) model specifications for estuarine, coastal and reef intertidal and subtidal areas in the Great Barrier Reef World Heritage Area and adjacent estuaries. Total number of sites used in each model (split between 80% for model training and 20% for testing), total sites used in each model, and the optimal number of predictor variables that were randomly selected at each node in RF models (mtry).

Model name	RF models		MRT models
	Number of sites	mtry	Number of sites
Estuary Intertidal	4962	2	4347
Estuary Subtidal	6426	1	5420
Coast Intertidal	5328	2	3895
Coast Subtidal	16,073	3	10,151
Reef Intertidal	2569	2	1292
Reef Subtidal	2695	2	1258
Total	38,053	-	26,363

The six RF models were used to generate rasters of seagrass predicted probability across the entire GBRWHA. We created this by predicting each model onto a raster stack of data corresponding to the same predictors included in each model using the *raster* package in R (Hijmans, 2020). Raster data sets within each stack were predicted to the 30m resolution of the depth model (Beaman, 2017) using the *sf* package in R (Pebesma, 2018). We defined potential seagrass habitat as regions where the RF models predicted a probability $\geq$0.2. This threshold was determined by expert evaluation of visualisations of the data; it captured the mapped extent of seagrass and excluded extensive regions where seagrass has never been recorded. Remaining regions (probability $\leq$0.2) were classed as unlikely seagrass habitat.

Our second analysis defined seagrass communities within potential seagrass habitat using multivariate regression trees (MRTs) in the R package *mvpart* (De'ath, 2004) (available in archive form on CRAN at https://cran.r-project.org). MRTs are a constrained analysis that repeatedly splits the assembled data, in this case a matrix of presence/absence data for each species as the response variable for each model, into groups that represent a distinct community composition defined by threshold values of associated environmental variables (De'ath, 2002). Using species presence/absence from each site resulted in the community type being defined based on the frequency of occurrence of each species. For each MRT we used the same environmental predictors as for the RF models. We excluded sites from unlikely seagrass habitat to allow the six MRT models to identify patterns in seagrass species presence without being overwhelmed by zeros due to seagrass absence (Table 2). As the aim was to cluster the sites spatially, we did not include 'year' as a factor in the model. Instead, we aimed to categorise where each seagrass species is found, on average, through time.

We selected the best MRT for each habitat model using the cross-validated relative error (CVRE). The CVRE represents the capacity of the tree to predict community composition for new sites. Calculation of the CVRE is based on a repeated random sub-sampling cross-validation, where number of cross-validations (xval) can be specified and controls the proportional allocation of sites to training and testing (evaluation) sets and this is repeated 10 times, where each time data are randomly allocated to train and test groups. We designated 80% of our data for model training and 20% for testing. The CVRE is the average test error over the chosen number of cross-validations. We repeated the cross-validation 100 times to stabilise variability in CVRE estimates due to the random cross-validation; the *mvpart* package then estimates the mean CVRE, where 0 indicates perfect prediction and >=1 indicates no predictive power. The depth (number of splits) in the trees was selected by finding that depth that fitted the best predictive tree in the cross-validation.

All maps were created in ArcMap 10.8 (ESRI, Redlands, CA). The area of each seagrass probability level from the RF analysis, and each seagrass community from the MRT analysis, was determined by multiplying the pixel size (900m^2) by the total number of pixels for each category of interest in each raster of the modelled predictions for seagrass probability and community type.

3.5 Results

3.5.1 Seagrass habitat

We identified approximately 88,321 km^2 of potential seagrass habitat (probability of seagrass present $\geq$0.2 calculated as the average value over the predictions of all the trees in the forest) in the GBRWHA (Figure 5). This includes 1,111 km^2 of potential seagrass habitat in estuaries, 16,276 km^2 in coastal areas, and 70,934 km^2 in reef areas (Table 3). The performance of RF models varied; estuary subtidal and intertidal models were the least accurate (72 and 73% overall accuracy, respectively) and reef subtidal and intertidal models were the most accurate (81 and 84% overall accuracy, respectively) (Table 4).

The importance of different environmental variables in predicting seagrass presence/absence differed among the six RF models (Table 5). In subtidal areas, depth was the most important environmental condition in estuaries and coasts, and the second most important after benthic light (PARb) in reef areas. The least important environmental condition for predicting seagrass habitat in subtidal coastal and reef areas was water type, and dominant sediment type in estuaries.

In intertidal areas of estuaries, relative tidal exposure was the most important environmental condition for predicting seagrass presence/absence; in contrast, on reefs tidal exposure was the least important and water temperature was most important. For coastal intertidal areas wind speed was most important, followed by water temperature, salinity, then tidal exposure and benthic light (Table 5).

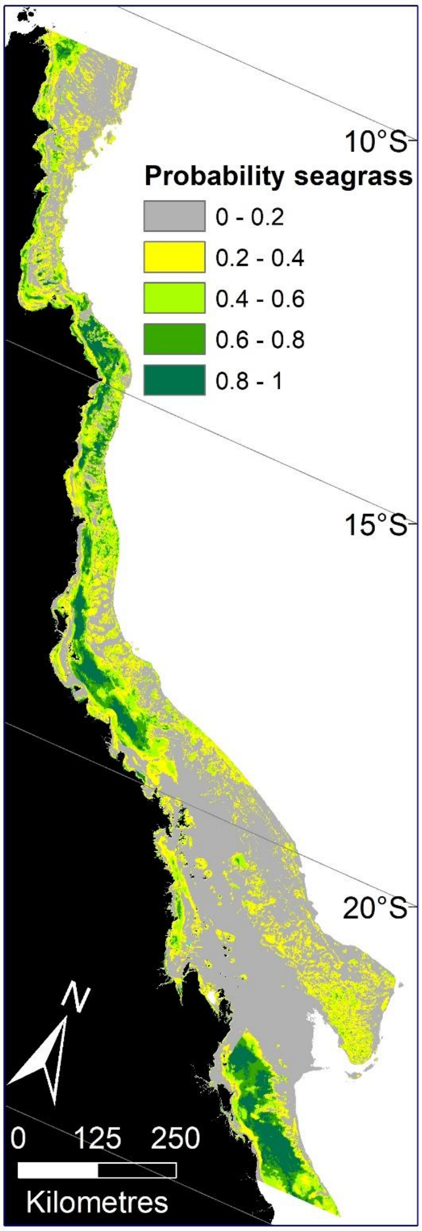

Figure 5: Predicted probability of seagrass presence across the Great Barrier Reef World Heritage Area and adjacent estuaries based on six Random Forest models. Potential seagrass habitat classed as probability >0.2 (calculated as the average value over the predictions of all the trees in the forest).

Table 3: Potential seagrass habitat (km^2) for each probability class across the Great Barrier Reef World Heritage Area and adjacent estuaries based on six Random Forest models.

Probability of seagrass	Model						
	Estuary		Coast		Reef		
	Intertidal	Subtidal	Intertidal	Subtidal	Intertidal	Subtidal	**Total**
<0.2	125	473	124	17,829	3070	79,306	**100,927**
0.2 - <0.4	99	203	319	9466	820	29,893	**40,800**
0.4 - <0.6	196	58	323	4006	594	16,419	**21,596**
0.6 - <0.8	116	49	110	1487	269	12,075	**14,106**
≥0.8	197	193	56	509	141	10,723	**11,819**

Table 4: Random Forest confusion matrices and performance using testing data for six models. Columns show predicted values (P) and rows show observed (O) values. Accuracy of model predictions for each class of seagrass absent, present, and overall accuracy.

RF Model	Confusion Matrix			Accuracy (%)
Estuary Intertidal		Absent (O)	Present (O)	
	Absent (P)	239	95	72% (absent)
	Present (P)	185	464	72% (present)
Estuary Subtidal		Absent (O)	Present (O)	
	Absent (P)	380	156	71% (absent)
	Present (P)	194	536	73% (present)
Coast Intertidal		Absent (O)	Present (O)	
	Absent (P)	307	108	74% (absent)
	Present (P)	129	516	80% (present)
Coast Subtidal		Absent (O)	Present (O)	
	Absent (P)	1310	387	77% (absent)
	Present (P)	391	1092	74% (present)
Reef Intertidal		Absent (O)	Present (O)	
	Absent (P)	257	33	89% (absent)
	Present (P)	50	166	77% (present)
Reef Subtidal		Absent (O)	Present (O)	
	Absent (P)	294	62	83% (absent)
	Present (P)	39	132	77% (present)

Table 5: Importance of environmental variables for each Random Forest model. Values are the mean decrease of accuracy in predictions on the out-of-bag samples when a given variable is excluded from the model. The most important variable is in bold. "-" indicates variable not included in model.

Environmental variable	Model					
	Estuary		Coast		Reef	
	Intertidal	Subtidal	Intertidal	Subtidal	Intertidal	Subtidal
Current speed	-	-	51	94	-	41
Depth	-	**215**	-	**159**	-	54
Geomorphology	-	-	-	32	27	-
Latitude	104	162	-	-	-	-
PARb	-	-	63	94	28	**57**
Proportion mud	-	-	54	92	41	49
Salinity	-	-	69	103	-	-
Sediment type	70	92	-	-	-	-
Tidal exposure	**108**	-	63	-	21	-
Water temperature	-	-	70	98	**51**	53
Water type	-	-	24	23	28	31
Wind speed	-	-	**71**	103	40	42

The relationship between each environmental predictor and the probability of seagrass being present varied among the models (Figure 6). In subtidal areas, in estuaries the probability of seagrass presence declines dramatically in the first 5 m to $p<0.2$, in coasts the probability of seagrass presence reduces over the first 10 m and then stabilises at $p\sim0.35$, while in reefs the probability of seagrass increases between 0 and 40 m depth, then declines sharply between 40 and 60 m (Figure 6). Proportion of mud in the sediment also had a varied relationship with the probability of seagrass – in coastal and reef intertidal areas there was a positive relationship between the proportion of mud in the sediment and probability of seagrass, while in coastal and reef subtidal areas it was a negative relationship (Figure 6). In reef areas there was a greater probability of seagrass in subtidal than intertidal areas, while in coastal areas the probability of seagrass was greater in the intertidal zone (Figure 6).

There were distinct environmental thresholds identified by some models. In reef areas, the long-term annual average temperature of 27°C was a threshold where, above that temperature, seagrass probability decreased in intertidal areas but increased in subtidal areas (Figure 6). In both intertidal and subtidal coastal areas, the probability of seagrass increased with water temperature >26°C, then declined once waters were >28°C (Figure 6). The probability of seagrass was always greatest where current speeds were lowest and salinity was >34 PSU. Latitude had a strong effect on the probability of intertidal and subtidal estuarine seagrass, which was most likely to be present within the latitudinal range of 18 and 24°S (Figure 6).

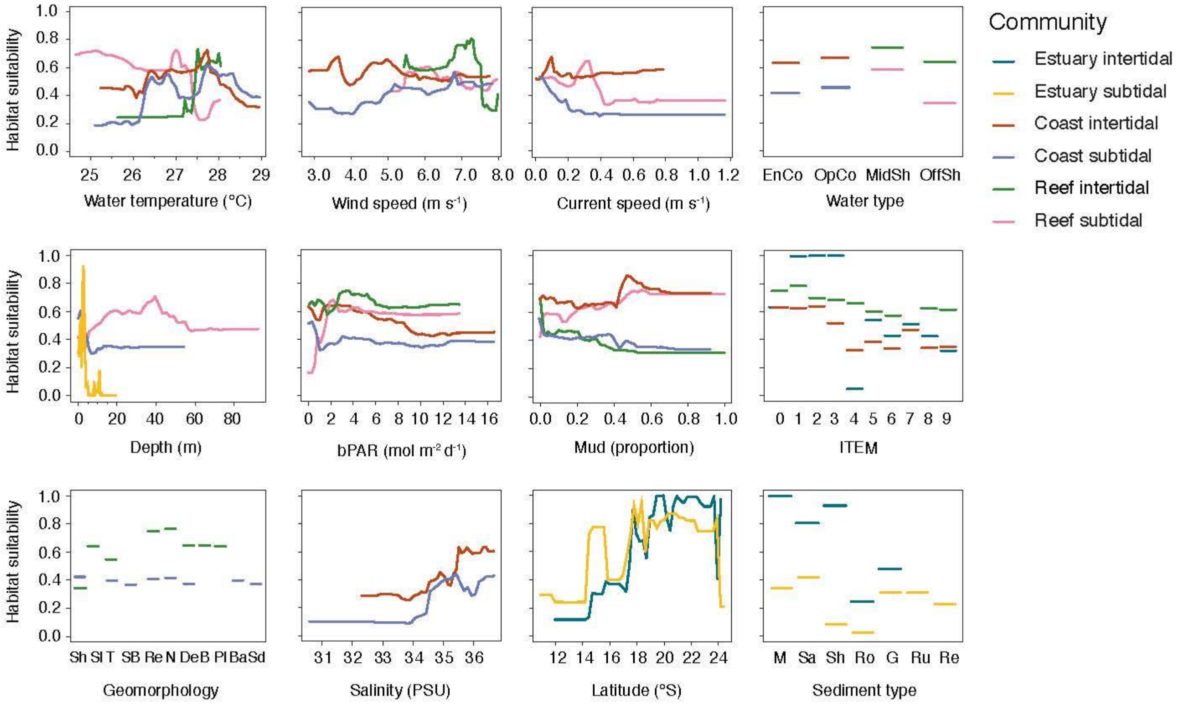

Figure 6: Partial plots of variable importance from six Random Forest models. Abbreviations for factor levels are: Water type (EnCo, enclosed coastal; OpCo, open coastal; MidSh, mid-shelf; OffSh, offshore); Geomorphology (Sh, shelf; Sl, slope; T, terrace; SB, sand bank; Re, reef; N, N/A beyond the extent of the layer or on land; De, deep hole or valley; B, basin; Pl, plateau; Ba, basin; Sd, saddle); and Sediment (M, mud; Sa, sand; Sh, shell; Ro, rock; Ru, rubble; Re, reef).

3.5.2 Seagrass communities

Within regions of potential seagrass habitat, we identified 36 seagrass community types defined by their distinct species assemblages (Figure 7 and Figure 8, Table 6). The importance of environmental conditions in structuring the location and spatial extent of these communities also was diverse, and included depth, tidal exposure, latitude, current speed, benthic light, proportion of mud in the sediment, water type, water temperature, salinity, and wind speed (Figure 9, Figure 10, Figure 11). Estuaries contain communities with the smallest spatial extent in the GBRWHA; five of the communities had a predicted total area between 4 and 7 km^2 (Table 6). Estuary communities were predicted by variations in relative tidal exposure, depth and latitude, but not the dominant sediment type (Figure 9). Hinchinbrook Island in the central GBRWHA was identified as an area of high community diversity and a transition zone between communities for both intertidal and subtidal estuarine communities (Figure 7 and Figure 9E; Table 6). Coastal communities occur in a highly dynamic transition zone between estuaries and reefs, were predicted by the greatest variety of environmental variables, and were the only area where all 12 seagrass species were present (Figure 10). Reef communities have a distinct species composition. Species such as *T. hemprichii, C. rotundata*, and *S. isoetifolium* often dominate intertidal and shallow subtidal reef communities, while species found in estuarine and coastal areas (*E. acoroides*, *Z. muelleri* subsp. *capricorni*) are not present.

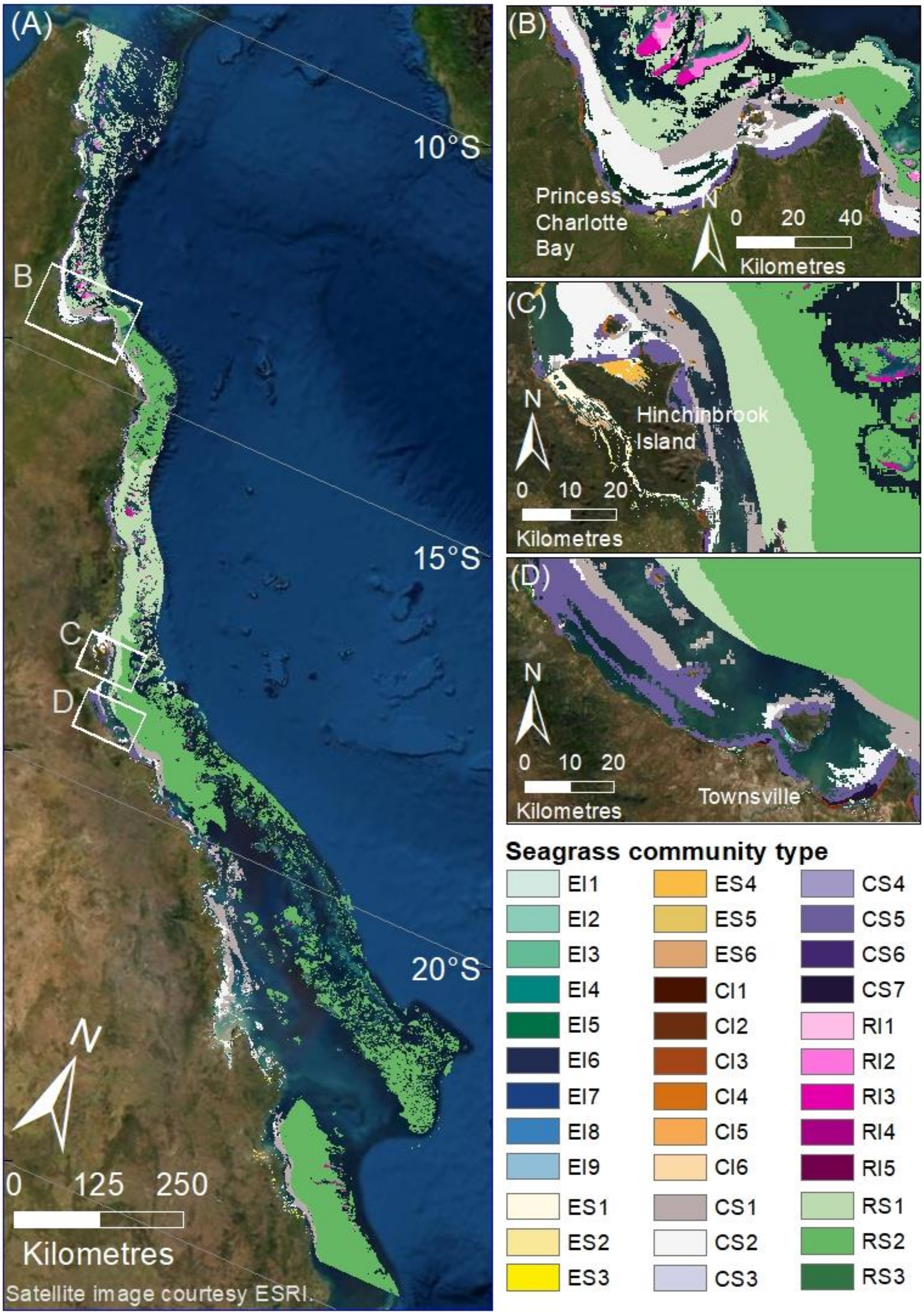

Figure 7: (A) Thirty-six seagrass communities predicted for the Great Barrier Reef World Heritage Area and adjacent estuaries: estuary intertidal (EI1-EI9), estuary subtidal (ES1-ES6), coastal intertidal (CI1-CI6), coastal subtidal (CS1-CS7), reef intertidal (RI1-RI5), and reef subtidal (RS1-RS3) communities. (B-D) Finer-scale maps demonstrating predicted boundaries between communities at select locations.

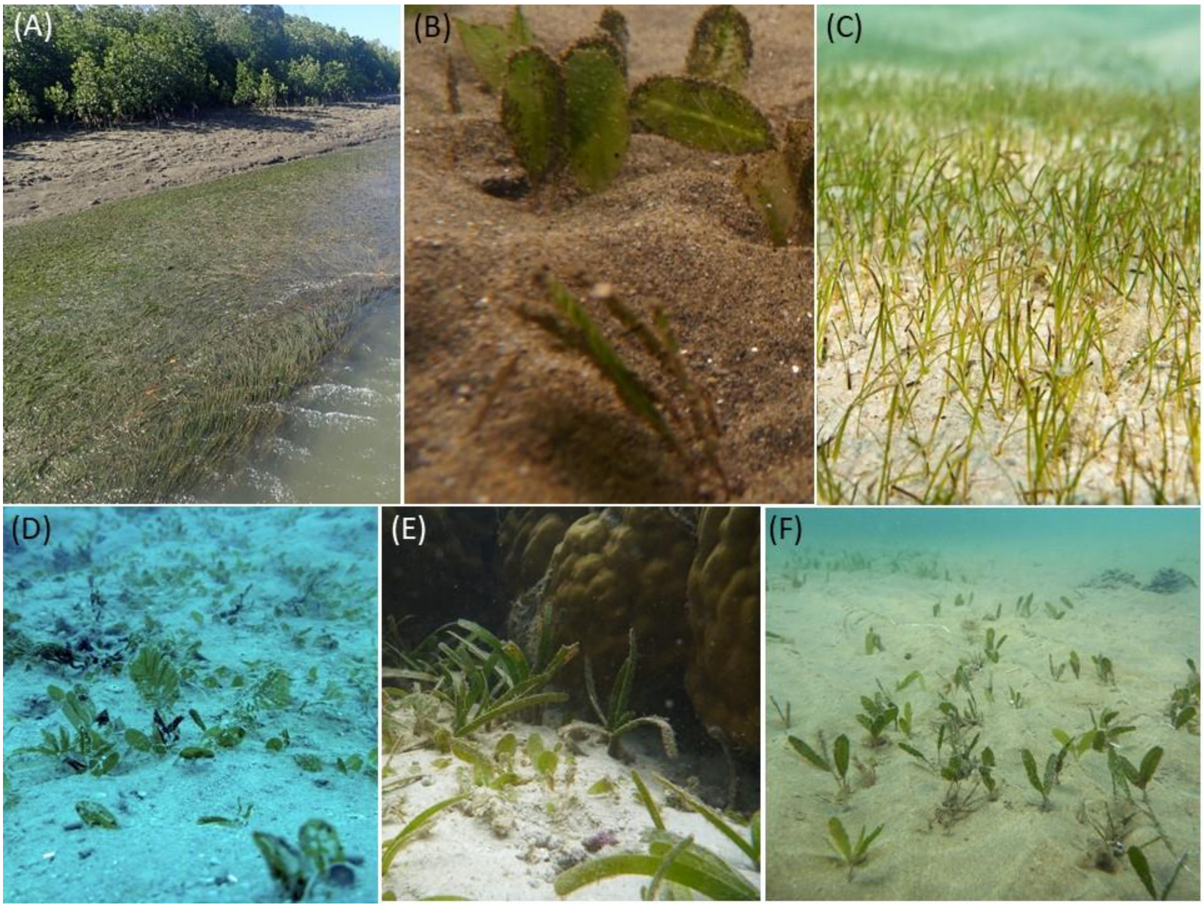

Figure 8: Common seagrass communities in the Great Barrier Reef World Heritage Area and adjacent estuaries: (A) estuary intertidal *Z. muelleri* subsp. *capricorni*, (B) estuary subtidal *H. ovalis* dominated, (C) coastal intertidal *H. uninervis* dominated, (D) coastal subtidal *H. ovalis* and *H. spinulosa*, (E) reef intertidal *T. hemprichii* and *H. ovalis*, and (F) reef subtidal *H. decipiens*.

Table 6: Seagrass communities in the Great Barrier Reef World Heritage Area and adjacent estuaries, including predicted area and geographic range. See Figures 7, 9, 10 and 11 for locations.

Community	Predicted area (km^2)	Geographic range
Estuary Intertidal 1	288	Northern to southern extent GBRWHA
Estuary Intertidal 2	5	South of Bingil Bay to southern end Hinchinbrook Island
Estuary Intertidal 3	77	Southern end Hinchinbrook Island to northern tip Curtis Island
Estuary Intertidal 4	3	Northern extent of GBRWHA to Bingil Bay
Estuary Intertidal 5	7	Northern tip Curtis Island to southern extent GBRWHA
Estuary Intertidal 6	4	South of Mourilyan Harbour to Townsville
Estuary Intertidal 7	156	South of Townsville to Shoalwater Bay
Estuary Intertidal 8	5	Northern extent of GBRWHA to Mourilyan Harbour
Estuary Intertidal 9	39	South of Shoalwater to southern extent GBRWHA
Estuary Subtidal 1	182	Northern to southern extent GBRWHA
Estuary Subtidal 2	96	Hinchinbrook Island to Gladstone
Estuary Subtidal 3	122	Hinchinbrook Island to Gladstone
Estuary Subtidal 4	36	Northern Hinchinbrook Island and the upper reaches of Trinity Inlet
Estuary Subtidal 5	38	Cairns to northern extent of GBRWHA
Estuary Subtidal 6	16	Central and northern Hinchinbrook Island
Coastal Intertidal 1	141	Whitsunday Islands to southern extent GBRWHA
Coastal Intertidal 2	91	Northern to southern extent GBRWHA
Coastal Intertidal 3	205	Northern to southern extent GBRWHA
Coastal Intertidal 4	178	Northern to southern extent GBRWHA
Coastal Intertidal 5	39	Townsville to southern extent GBRWHA
Coastal Intertidal 6	154	Whitsunday Islands to southern extent GBRWHA
Coastal Subtidal 1	7589	Northern to southern extent GBRWHA
Coastal Subtidal 2	4575	Northern to southern extent GBRWHA
Coastal Subtidal 3	68	Northern to southern extent GBRWHA
Coastal Subtidal 4	161	Northern to southern extent GBRWHA
Coastal Subtidal 5	2938	Northern extent GBRWHA to Whitsunday Islands
Coastal Subtidal 6	62	Northern to southern extent GBRWHA
Coastal Subtidal 7	75	Northern to southern extent GBRWHA
Reef Intertidal 1	318	Northern to southern extent GBRWHA
Reef Intertidal 2	887	Northern to southern extent GBRWHA

Reef Intertidal 3	608	Northern to southern extent GBRWHA
Reef Intertidal 4	9	Small reef patches northern to southern extent GBRWHA
Reef Intertidal 5	1	Small reef patches in Cairns and Princess Charlotte Bay regions
Reef Subtidal 1	19,434	Northern extent GBRWHA to Princess Charlotte Bay; Bloomfield to Palm Island Group
Reef Subtidal 2	49,052	Princess Charlotte Bay to Bloomfield; Palm Island Group to southern extent GBRWHA
Reef Subtidal 3	623	Northern to southern extent GBRWHA

3.5.3 Estuary intertidal

Nine seagrass communities were predicted in the estuarine intertidal model (Figure 7 and Figure 9, Table 6). *Z. muelleri* subsp. *capricorni*, *H. uninervis* and *H. ovalis* were found in nearly all of these communities but with varying frequencies of occurrence (Figure 8A and Figure 9). The most extensive estuarine intertidal community EI1 was predicted to cover a total ~288 km^2 throughout the GBRWHA in areas and associated with extremely infrequent (ITEM = 0) and medium to high tidal exposure (ITEM = 4-9). The remaining estuarine intertidal communities were predicted to occur in distinct latitudinal bands. Four intertidal communities occurred where tidal exposure was very low (ITEM = 1): the *Z. muelleri* subsp. *capricorni* dominated community EI4 in the northern GBRWHA, the *H. uninervis* dominated community EI2 between Bingil Bay and Hinchinbrook Island (17.81 – 18.46°S), the mixed species community EI3 between Hinchinbrook Island and northern Curtis Island (23.57°S), and the *H. ovalis* and *Z. muelleri* subsp. *capricorni* dominated community EI5 from Curtis Island south (Figure 7 and Figure 9F; Table 6). An additional four intertidal communities were predicted where tidal exposure was low (ITEM = 2-3): the *Z. muelleri* subsp. *capricorni* dominated community EI8 north of Mourilyan Harbour, the *H. uninervis* dominated community EI6 between Mourilyan Harbour and Townsville (17.62 – 19.28°S), the extensive *Z. muelleri* subsp. *capricorni* dominated community EI7 between Townsville and Shoalwater Bay (156 km^2), and the *Z. muelleri* subsp. *capricorni* and *H. uninervis* dominated community EI9 south of Shoalwater (19.28°S) (Figure 7 and Figure 9; Table 6).

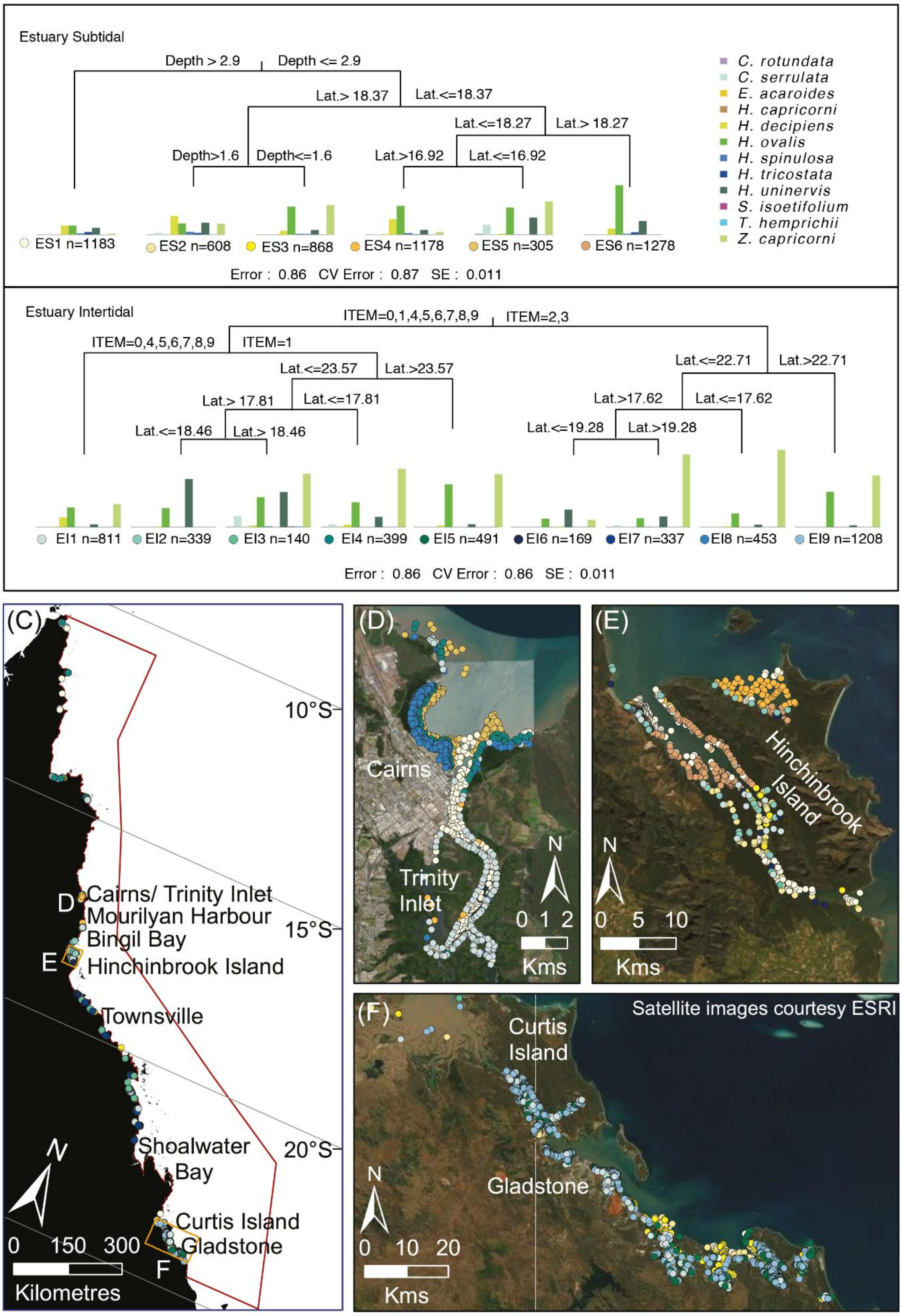

Figure 9: Multivariate regression tree (MRT) and seagrass communities classified for estuaries using species presence/absence data for (A) subtidal sites and (B) intertidal sites. The number (n) below each community is the count of observations that fall into that community. The histogram shows the frequency of occurrence for each species in that community with the height of the bar representing the frequency that each species was observed in that assemblage. The coloured dots represent unique communities for coast intertidal (EI) 1-9, and coast subtidal (ES) 1-6. The CV Error is the cross-validated relative error. (C) The spatial distribution of communities across the Great Barrier Reef World Heritage Area (red border), and (D-F) finer-scale maps of communities at select locations.

3.5.4 Estuary subtidal

The estuary subtidal model predicted six seagrass communities (Figure 7 and Figure 9; Table 6). Community ES1 is the most extensive, predicted to cover a total subtidal area of 182 km^2 in depths below 2.9 m MSL. This is the only estuarine subtidal community predicted to occur throughout the GBRWHA, and it is dominated in equal parts by *H. ovalis* and *H. decipiens* with no *Z. muelleri* subsp. *capricorni* (Figure 8B and Figure 9). The remaining subtidal communities occur in depths shallower than 2.9 m (Figure 9E). Between Hinchinbrook Island (18.37°S) and Gladstone, community ES2 is predicted in the intermediate depth range 1.6 - 2.9 m MSL and has a species mix similar to the deep estuarine community ES1 but with the addition of *Z. muelleri* subsp. *capricorni*, while the *Z. muelleri* subsp. *capricorni* and *H. ovalis* community ES3 is predicted in the 0 - 1.6 m MSL depth range. From Hinchinbrook Island north, subtidal communities were predicted to occur in distinct latitudinal bands similar to intertidal communities: the small *H. ovalis* community ES6 (16 km^2) between central and northern Hinchinbrook Island (18.37 - 18.27°S), the *H. ovalis*/ *H. decipiens* community between northern Hinchinbrook Island and Trinity Inlet, and the mixed species community ES5 north of Trinity Inlet (16.92°S) (Figure 7 and Figure 9, Table 6).

3.5.5 Coastal intertidal

The coastal intertidal model predicted communities separated by variations in water type, water temperature, salinity and tidal exposure (Figure 10). Three communities were predicted within the enclosed coastal water type: in cooler (<26.4 °C) southern GBRWHA waters the *H. ovalis* and *Z. muelleri* subsp. *capricorni* dominated community CI1; in warmer waters the *Z. muelleri* subsp. *capricorni* community CI2 where tidal exposure is low (ITEM = 2-3) and the more speciose community CI3 where tidal exposure is very low (ITEM = 0-1) or intermediate to high (ITEM = 4-9) (Figure 7 and Figure 10, Table 6). Three communities were also predicted within the open coastal water type. Community CI4 is predicted to occur throughout the GBRWHA where salinity is <35.4 PSU and, unusually for coastal communities, this speciose community has relatively high frequency of *T. hemprichii* and *C. rotundata* usually associated with intertidal reef communities. Communities CI5 and CI6 were predicted to occur in regions of high salinity between Townsville and the Keppel Islands: Community CI5 in areas of low (ITEM = 2-3) and high (ITEM = >5) tidal exposure and community CI6 areas of very low (ITEM = 0-1) and intermediate (ITEM = 4) tidal exposure (Figure 7, Figure 8C and Figure 10, Table 6).

3.5.6 Coastal subtidal

The coastal subtidal model predicted communities separated by variations in current speed, depth, and the proportion of mud in the sediment (Figure 10). Four communities were associated with very low current speeds (<0.11 ms^{-1}): the *H. uninervis* dominated community CS4 in areas where almost no mud (proportion mud <0.005) is present in the sediment, and the more diverse communities CS5, CS6 and CS7 when some mud is present. Community CS5 is the largest of these low current communities (2938 km^2) and predicted at depths >2 m MSL from the Whitsunday Islands north (Figure 7), with 10 species recorded. Communities CS6 and CS7 are predicted to occur throughout the GBRWHA at depths <2.0 m: CS6 where the proportion of mud is low to moderate (0.005 – 0.38), and CS7 where the proportion of mud is >0.38 and the frequency of *H. uninervis* and *H. spinulosa* is greater than in community CS6 (Figure 10).

Three coastal subtidal communities were predicted to occur throughout the GBRWHA where current speed was >0.11 ms^{-1}. The predicted area of these communities was much larger than low current communities, and communities were associated with different depths. Community CS3 in shallow subtidal waters (<1.6 m MSL) had a species mix similar to coastal intertidal communities. The large (4575 km^2) community CS2 at intermediate depths (1.6 – 12.6 m MSL) was dominated by *H. uninervis* and *H. ovalis* but with a much greater prevalence of typical subtidal species such as *H. decipiens*, *H. spinulosa*, and *C. serrulata*, and very little *Z. muelleri* subsp. *capricorni*. The deep subtidal community CS1 (>12.6m) had the largest predicted total area (7589 km^2) of all coastal communities. This community was dominated almost entirely by *H. decipiens* and *H. spinulosa*, and was one of the few seagrass communities where *H. tricostata* is present (Figure 7, Figure 8D and Figure 10, Table 6).

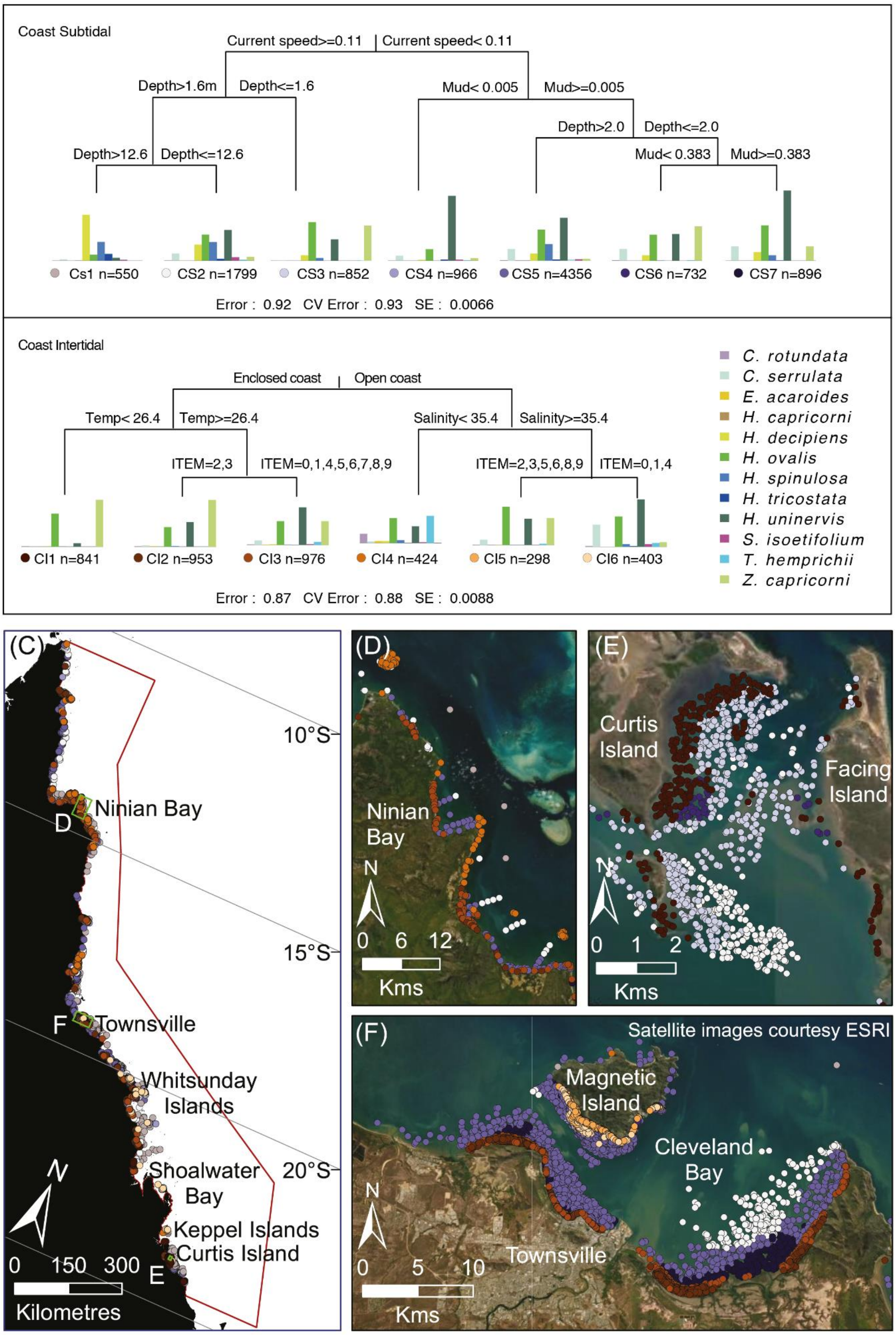

Figure 10: Multivariate regression tree (MRT) and seagrass communities classified for coastal waters using species presence/absence data for (A) subtidal sites and (B) intertidal sites. The number (n) below each community is the count of observations that fall into that community. The histogram shows the frequency of occurrence for each species in that community with the height of the bar representing the frequency that each species was observed in that assemblage. The coloured dots represent unique communities for coast intertidal (CI) 1-6, and coast subtidal (CS) 1-7. The CV Error is the cross-validated relative error. (C) The spatial distribution of communities across the Great Barrier Reef World Heritage Area (red border), and (D-F) finer-scale maps of communities at select locations.

3.5.7 Reef intertidal

Intertidal reef communities were best predicted by a model that included benthic light, proportion mud, and wind speed (Figure 11). Three reef intertidal communities were associated with light levels <13.4 mol photons $m^{-2}d^{-1}$. *T. hemprichii* was the dominant species in all of these communities (Figure 8E). *H. ovalis* occurred in greatest frequency in community RI1, predicted to be most prevalent on fringing reefs around the Palm Island Group in the central GBRWHA and as small patches on reefs north of there when some mud is present in the sediment (Figure 7, Figure 11, Table 6). The large intertidal communities RI2 (887 km^2) and RI3 (608 km^2) were associated with very low mud content: RI2 in the northern GBRWHA where wind speed was high (> 6.8 ms^{-1}) and RI3 throughout the GBRWHA in calmer conditions (Figure 11). Communities RI4 and RI5 were associated with very high light (>13.4 mol photons $m^{-2}d^{-1}$; Figure 11). Both communities were characterised by similar frequencies of the dominant species *T. hemprichii, C. rotundata* and *H. uninervis*, but variations in other species depended on the proportion of mud in the sediment with greater species diversity in community RI4 with the addition of mud. Communities RI4 and RI5 were predicted to occur as small patches on reef tops largely confined to clusters of reefs near Cairns and Princess Charlotte Bay (Figure 7, Table 6).

3.5.8 Reef subtidal

The reef subtidal model predicted three reef communities separated by depth and water temperature (Figure 11). Community RS3 was found at depths <8 m MSL in the transition zone between intertidal and deep subtidal reef communities. This community was predicted to occur as narrow perimeter bands around reefs and islands throughout the GBRWHA, but particularly on reefs between the Palm Island Group and Bloomfield, and on nearshore reefs north of Princess Charlotte Bay (Figure 7 and Figure 11, Table 6). Species composition for RS3 was similar to the intertidal reef communities RI4 and RI5: *C. rotundata*, *H. ovalis* and *H. uninervis* frequently occur, but the dominant intertidal species *T. hemprichii* was replaced by *C. serrulata* and *S. isoetifolium* (Figure 11).

The two largest seagrass communities were associated with reef waters >8 m MSL (Figure 11). Both deep communities were dominated by a mix of *Halophila* species, but the frequency of each species varied with water temperature. Community RS1 (19,434 km^2) was predicted in warmer waters (>27.3°C) north of the Palm Island Group, was dominated by *H. decipiens*, and the relatively rare *H. tricostata* was found in this community more often than in any other. The cooler-water subtidal community RS2 (49,052 km^2) was predicted south of the Palm Island Group and around a cool-water patch in the Lizard Island region of the northern GBRWHA (Figure 7 and Figure 11). Community RS2 is characterised by a more even mix of *Halophila* species: *H. decipiens*, *H. ovalis*, and *H. spinulosa* are equally common, and the rarer species *H. capricorni* is found in this community more often than in any other (Figure 8F and Figure 11, Table 6).

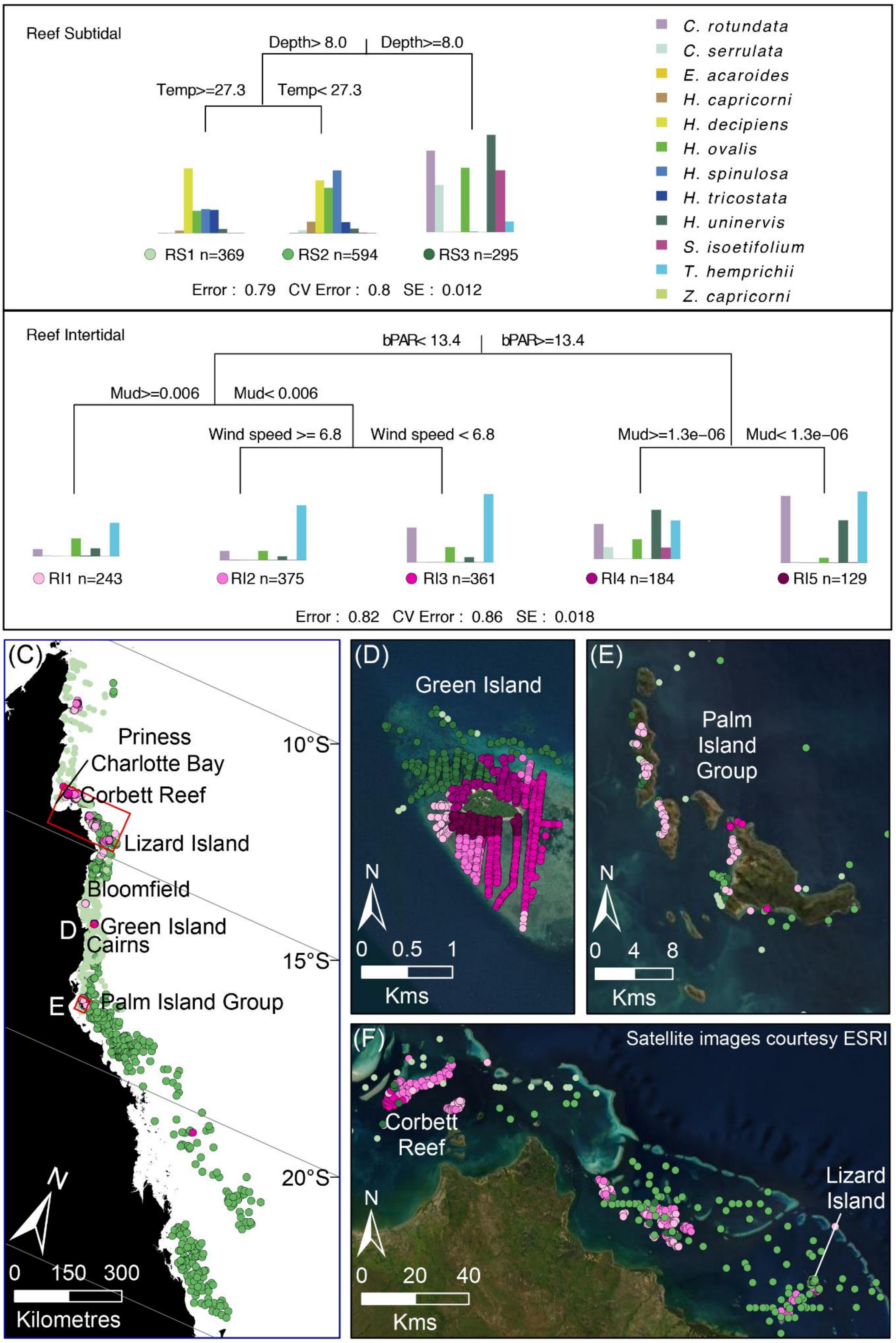

Figure 11: Multivariate regression tree (MRT) and seagrass communities classified for reef waters using species presence/absence data for (A) subtidal sites and (B) intertidal sites. The number (n) below each community is the count of observations that fall into that community. The histogram shows the frequency of occurrence for each species in that community with the height of the bar representing the frequency that each species was observed in that assemblage. The coloured dots represent unique communities for reef intertidal (RI) 1-5, and reef subtidal (RS) 1-3. The CV Error is the cross-validated relative error. (C) The spatial distribution of communities across the Great Barrier Reef World Heritage Area (red border), and (D-F) finer-scale maps of communities at select locations.

3.6 Discussion

We present an approach to define seagrass communities and how they distribute over large spatial scales. Our study area was vast and encompassed a multitude of changing physical and biological conditions and diverse seagrass species. Despite these challenges, and a dataset collected at different times and scales, our application of machine learning techniques provides a statistically valid and transferrable approach for one of the world's most complex seagrass systems. This approach can be adapted for use at other locations to identify the seagrass community types that make up the seagrass biome, addressing the critical gaps in spatial knowledge needed for global seagrass protection (Griffiths et al., 2020; Tulloch et al., 2020; Unsworth et al., 2019). Our seagrass community model provides one of the spatial tools identified as a key information gap for assessing and maintaining resilience. Understanding seagrass community distribution is a critical pre-requisite for assessing resilience and dispersal (Grech et al., 2018; Grech et al., 2016). It is also valuable for deciding whether seagrass restoration is required versus the potential for natural recovery and for identifying suitable donor sites if intervention is warranted (Tan et al., 2020).

The varied environmental conditions that determine seagrass community diversity demonstrate that reporting trends at large scales and with coarse partitions such as "coastal" fail to accurately account for changes at the more precise community level. The advantage of constrained clustering techniques such as the MRTs applied in this study to define community types, is that each cluster defined an assemblage type, but additionally the environmental values defined an associated habitat type for the assemblage. This allowed prediction of assemblage types where the set of environmental values were available but there was no seagrass data. Our analysis provides a basis for management authorities to identify likely seagrass communities within environmental management plans that are inadequately protected or exposed to environmental threats (Tulloch et al., 2020). Our method has a global utility by creating informative models based on data that is scalable and easily available as it requires only presence/absence data for seagrass species, information that combined with location is able to be integrated into citizen science programs.

When the same MRT approach was applied at a smaller scale in Cleveland Bay, near Townsville in the central GBR, a greater number of communities were identified overall (11 communities compared to 7 in this study), because there were more communities in intertidal areas (Collier et al 2020). This was driven largely by the benthic substrate, which was more detailed (categories of substrate noted at each seagrass site) than data available for the GBRWHA (modelled at 1km pixels). In contrast, one less subtidal community was identified. This was because current speed data was not included in that analysis, a variable which differentiated communities in this study. Communities identified from individual locations will amass for the GBR, but may be more suitable for local-scale monitoring and management.

Creating spatially expansive models at the scale of the GBRWHA constrained the analysis to the environmental data also available at that scale, so our models are unlikely to account for smaller-scale localised differences in seagrass communities and their drivers. Our community models demonstrated that very small shifts in depth and tidal exposure can lead to significant shifts in seagrass communities. Depth and tidal exposure were two of the highest resolution environmental data sets we used in our models, but many of the environmental predictors we used represent larger-scale average spatial patterns modelled at a 1km grid scale (e.g. benthic light, wind speed, current speed, water temperature, salinity). This means that variables such as modelled benthic light should be reliable at large-scale seasonal-average spatial patterns, but smaller-scale variations in benthic light that depend on small-scale variations in bathymetric depth and sediment distribution will not be accurately simulated.

Our approach can be readily refined and applied at smaller localised scales with higher resolution or additional data that may better resolve drivers and communities at local scales (Collier et al., 2020). Higher resolution data required for more precise models include the distribution of mud, as well as the amount of siliceous and carbonate benthic sediments, as opposed to a modelled estimate of the proportion of mud. The eReefs sediment model used in our analysis evolves over time but is heavily dependent on initial conditions which are derived from relatively low-resolution observational data, and the proportion of mud is likely to be underestimated in nearshore areas, particularly in the vicinity of river mouths. Large-scale environmental models for the region also rarely extend beyond the GBRWHA boundaries into adjacent estuarine waters. Our estuarine models highlight the diversity of seagrass communities and the paucity of environmental data for estuaries at this scale - our models were limited to just three environmental variables in estuaries but predicted 15 communities. The significant role of latitude in estuarine community divisions highlights how little we know about the complex range of environmental conditions that influence seagrass community structure in estuaries, with latitude acting as a proxy in our models.

Our models predict potential seagrass habitat and seagrass communities under “normal” growing season conditions at the scale of the GBRWHA because we did not include seagrass data collected following large disturbances such as major floods/cyclones or during the seagrass senescent season (February to July). The environmental variables used also represent spatial variations in “typical” conditions, for example the wind speed model does not capture well the impacts of high energy events such as storms. The reality for tropical seagrass meadows is one of significant temporal change driven by a range of natural and anthropogenic impacts at different spatial scales (Adams et al., 2020; Coles et al., 2015; Grech et al., 2012; Lambert et al., 2019). Our models provide a fixed framework based on data collected over a number of years which can now provide a benchmark against which important ecological questions with a temporal component can be addressed at the community scale, including seasonality in growth and distribution, the effect of climate variations such as the El Niño/ La Niña cycles, severe storms and cyclones, recovery and succession, anthropogenic impacts and risk.

Environmental parameters in the GBRWHA interact with the topography of the reef and changes in the extent of the coastal plain, its catchments, and its development along a north-south axis. The influence of the coast also diminishes with distance, changes in sediment, and depth. Seagrass distribution and communities are shaped by multiple combinations of these environmental complexities. Large spatial trends were present; seagrass communities in the northern GBRWHA extend from the coast to the edge of the continental shelf, in the inshore central region bands where no seagrass was present ran parallel to the coast, and in the south there are large inter-reef areas with little or no seagrasses. Environmental parameters shaping the nature of reef lagoon communities is a common theme for more than just seagrass. Spatial and temporal variations in environmental conditions have been found to be important determinants of coral performance and species distribution in the GBRWHA (Canto et al., in review).Temporal changes in water quality variables including benthic light are useful predictors of changes in coral community resilience (Robson et al., 2020; Thompson et al., 2020).

3.6.1 Spatial planning and management applications for the GBRWHA

Our seagrass habitat model and community classification provides an important tool to make informed decisions at an appropriate scale during marine spatial planning, management cycles, monitoring design, threat mitigation, and habitat restoration. Zoning in the Great Barrier Reef Marine Park (GBRMP) to protect biodiversity and regulate human activities has been in place since 1981 when the region became the world's first coral reef ecosystem to achieve world heritage status. The GBRMP was rezoned in 2004 (Fernandes et al., 2009) and while that represented best practice at the time, rezoning identified only five seagrass bioregions where seagrass was a key element (http://www.gbrmpa.gov.au/ data/assets/pdf file/0011/17300/nonreef-bioregions-in-the-gbrmp-and-gbrwh.pdf). We now provide those previously missed details of the complexity of seagrass communities, particularly for coastal waters and estuaries. The detail can applied as is, or adapted to management objectives. Estuaries and rivers adjacent to the GBRWHA are small by international standards, but their flow and sediment load variability in a monsoon-influenced coastline makes them both key attributes of the GBRWHA and sources of environmental forcing (Bainbridge et al., 2018; Lambert et al., 2019).

Our analysis can be used to underpin the better design of GBRMP spatial protection zoning and in the design of monitoring programs to represent the diversity of seagrass habitats. We identified spatial complexity in community types; some extend throughout the GBRWHA while some communities are small and localised. We focus on seagrass habitats but these overlap spatially with other environmental values such as populations of sea turtles and dugong that suggest priority areas for management protection such as the Hinchinbrook Island region where extensive and diverse seagrass communities were predicted. While we provide a framework to understand spatial patterns in seagrass communities it remains open to management authorities to evaluate a level of concern for protection. Some communities have distinct assemblages, while others are differentiated by only slight changes in relative occurrence of species, but identified in our analysis because species and environmental features were different. Sensitivity to environmental threats for community types can also be ameliorated by resilience inferred by connectivity, not included in this model but likely to have an influence at scales of hundreds of kilometres (Grech et al., 2018; Tol et al., 2017).

To design a marine protection system for all seagrass communities these spatial complexities and differing sensitivities to environmental conditions will need to be adapted into a broader marine protection model. We are now able to better evaluate environmental risk to seagrass habitats from natural processes and anthropogenic activity and to assess environmental threats that affect seagrass at a large scale including cyclones and floods (Coles et al., 2015; Collier et al., 2012; McKenna et al., 2015; Petus et al., 2014; Rasheed et al., 2014), climate change (Adams et al., 2020; Collier & Waycott, 2014; Collier et al., 2011), and more localised impacts such as coastal development, dredging, and oil spills (Fraser et al., 2017; Taylor & Rasheed, 2011). Spatial assessments of cumulative anthropogenic risk to seagrasses in the GBRWHA found risks tend to accumulate where ports and coastal development pressures overlay with inputs from coastal catchment runoff (Grech et al., 2011). Our community model provides a tool to identify communities that occur in these risk hotspots and may be vulnerable due to their lack of representation outside of high risk areas.

4.0 SETTING SEAGRASS COMMUNITY-SPECIFIC DESIRED STATE

4.1 Pre-amble

This chapter is based on a manuscript that is being prepared for publication:
Carter A.B., Collier C., Coles R., Lawrence E. and Rasheed M.A. (In prep) Setting community -specific “desired” states for seagrasses through cycles of loss and recovery.

This analysis is an outcome of investment through NESP TWQ Hub Projects 3.1 and 3.2.1 and this NESP TWQ Hub Project 5.4. The desired state analysis uses the method developed for Cleveland Bay seagrass (NESP TWQ Hub Project 3.2.1) and applies this to the seagrass communities throughout the GBRWHA described in Chapter 3.

Seagrass community models are available on the eAtlas data portal:
https://eatlas.org.au/data/uuid/313183fe-de3a-4874-bcba-d13d4ae4ecbc

4.2 Executive Summary

Marine habitats including seagrasses provide critical ecosystem services, yet there is ongoing concern over mounting pressures and continued habitat degradation. Defining a desired state for these habitats is a key step in implementing appropriate management, but is often a difficult proposition, given the challenges of available data and an understanding of where to set benchmarks. We used more than 20 years of historical data (1995-2018) on seagrass biomass for the diverse seagrass communities of Australia’s Great Barrier Reef World Heritage Area (GBRWHA) to develop desired state benchmarks. Of 36 previously defined seagrass communities, desired state was identified for 25 of them, with the remainder having insufficient data. Desired state varied by more than one order of magnitude between community types, and was influenced by the mix of species in the communities and the range of environmental conditions that define community boundaries. We identified a historical, decadal-scale cycle of decline with recovery to desired state in coastal intertidal communities. A number of the estuarine and coastal subtidal communities have not recovered to desired state biomass in recent years. The data were inadequate to identify desired state or assess trends in most of the reef communities. Understanding an historical context is critically important for making informed management decisions on the current state of seagrass in the GBRWHA. The approach can be scaled for monitoring, management and assessment of pressures at other relevant scales and jurisdictions. Our results guide conservation planning through prioritisation of at-risk communities that are continuing to fail to attain desired state.

4.3 Introduction

There is continued concern over the exploitation and degradation of marine ecosystems (Halpern et al., 2008; Jackson et al., 2001). Population growth, coastal development, pollution and other human activities have caused an estimated loss or degradation of 50% of salt marshes, 35% of mangroves, 30% of coral reefs, and as much as 29% of seagrasses worldwide over several decades (Barbier, 2017). These ecosystems provide critical services to global humanity (Barbier, 2017; Costanza et al., 2014), particularly for the population that live near the coast and rely on these habitats for food security.

Protecting and restoring marine ecosystems and ecosystem services requires environmental management and policy to succeed in a complex and uncertain environment that faces multiple pressures (Grech et al., 2011; Head, 2014; Walker & Salt, 2012). Defining what good environmental status looks like, expressed as a target condition or desired state, and knowing when it has been achieved, is critical when deciding whether a management intervention is required to return a system to its desired state (Borja et al., 2013; Hallett et al., 2016a).

Because of their extent and spatial and temporal variability, defining a desired state of marine ecosystems presents enormous challenges for scientists and managers (Collier et al., 2020; Hallett et al., 2016a; Hallett et al., 2016b; Levin & Möllmann, 2015; O'Brien et al., 2017; Pittman et al., 2011; Scott et al., 2018; Thompson et al., 2020). These challenges include:

- Data that is temporally and spatially sparse and not uniformly distributed;
- A limited knowledge of spatially-explicit seascape patterns and the ecological consequences of those patterns;
- The difficulty in separating long-term trends from short-term disturbance-recovery cycles;
- Poor accounting for the effect of species interactions; and
- The difficulty in defining the diversity of habitats and assemblages at relevant scales, and selecting appropriate indicators and metrics.

Overcoming these challenges is important because conservation and management decisions must be made regardless of the integrity of the information available (Kuhnert et al., 2010).

Seagrass species frequently occur in distinct assemblages or communities, with varying contributions of colonising, opportunistic and/or persistent seagrass species that may form enduring or transitory meadows (Kilminster et al., 2015). Seagrass community boundaries are the result of variations in environmental conditions, leading to the presence of diverse communities in a range of locations including estuaries, reef-tops, lagoons, open ocean, intertidal to deep subtidal waters (Coles et al., 2009; Grech & Coles, 2010; Jayathilake & Costello, 2018; McKenzie et al., 2020b). Seagrass community diversity and the environmental conditions that dictate the niche each community occupies complicates the application of common metrics such as seagrass biomass when assessing desired state because best-case scenarios differ dramatically between community types and environmental settings (Collier et al., 2020).

The trend in seagrass condition is generally a story of global (Orth et al., 2006; Waycott et al., 2009) and regional decline (Coles et al., 2015; Marba et al., 2009; Marbà & Duarte, 2010; Strydom et al., 2020; Thomson et al., 2015; Unsworth et al., 2018). However, seagrass meadows often exist in cycles of decline and recovery (Carmen et al., 2019; Creed & Amado Filho, 1999; Petus et al., 2014; Rasheed et al., 2014; Short & Wyllie-Echeverria, 1996; York et al., 2015). Teasing apart long-term trends (decadal) from short-term cycles (over several years) and assessing whether a seagrass community requires management intervention because it fails to reach its desired state is essential. This requires a solid definition of desired state for the range of seagrass communities in the assessment area, and an understanding of what environmental conditions determine that community diversity. This knowledge is necessary to provide the foundation for understanding seagrass condition in the context of natural cycles of decline and recovery, and for determining what policy levers are available to improve seagrass condition if required.

In this section we define desired state for the extensive and diverse seagrass habitats in Australia's Great Barrier Reef World Heritage Area (GBRWHA) and adjacent estuaries, including 36 identified seagrass communities (Chapter 3). These communities are diverse in species mix, spatial extent, and in the complex range of environmental conditions that define community boundaries. Defining a desired state of these communities is a key step in implementing appropriate management (Collier et al., 2020), an identified priority for seagrass. Seagrass above-ground biomass was selected as the desired state metric in this study because it is an ecologically-important indicator of seagrass condition, and is sensitive to environmental change and pressures over the spatial-temporal scale of this study (Marbà et al., 2013; McMahon et al., 2013; Petus et al., 2014; Rasheed et al., 2014). In defining desired state targets for GBRWHA communities, we use data collected over decadal scales including periods of decline and recovery to draw out appropriate benchmarks that represent a likely desired outcome for GBRWHA communities. Understanding this historical context is critically important for making informed decisions on the current state of seagrass in the GBRWHA. It is critical also for future management decisions on mitigation (e.g. catchment management) and remediation (e.g. seagrass restoration) following seagrass loss. We highlight gaps in our knowledge of seagrass condition that limit the implementation of effective management for some of the largest and most ecologically important seagrass communities. We present an analysis and approach that can be used to define desired state more generally for other global seagrass regions and other habitats where similar historical data is available.

4.4 Methods

4.4.1 Study area

Our study area covers coastal and reef areas in the continental shelf region of the GBRWHA where mean sea level is generally <100 m, and the adjacent estuaries along the mainland Australian coast (Figure 4A; Appendix 1).

4.4.2 Seagrass communities

The GBRWHA contains large and diverse seagrass meadows that extend from tropical to sub-tropical waters, with recent community analysis identifying 36 seagrass communities (Table 6). These each have uniquely defining environmental conditions and combinations of the twelve species that occur in the GBRWHA. Seagrass communities were classified within an 88,321 km^2 area of potential seagrass habitat (modelled probability of seagrass present $\geq$0.2). Within that, intertidal and subtidal community types were defined for three water bodies (estuary, coastal, and reef) using multivariate regression trees (Figure 4A; Appendix 1). Sites were classed as intertidal if they fell within Bishop-Taylor et al.'s (2019) intertidal extent model bands 1-9, or were classed as tidal regions of reefs or shoals within Queensland maritime waters (© State of Queensland (Department of Natural Resources, Mines and Energy) 2019).

Twelve seagrass species occurred in varying frequencies across these GBRWHA communities: *Cymodocea rotundata*, *Cymodocea serrulata*, *Enhalus acoroides*, *Halophila capricorni*, *Halophila decipiens*, *Halophila ovalis*, *Halophila spinulosa*, *Halophila tricostata*, *Halodule uninervis*, *Syringodium isoetifolium*, *Thalassia hemprichii*, and *Zostera muelleri* subsp. *capricorni*. The 36 seagrass communities were classified based on changes in the frequency of occurrence of these species and combinations of environmental conditions:

- Nine estuary intertidal communities - defined by latitude and tidal exposure.
- Six estuary subtidal communities - defined by latitude and depth.
- Six coastal intertidal communities - defined by distance from the coast, water temperature, tidal exposure and salinity.
- Seven coastal subtidal communities - defined by current speed, depth, and the proportion of mud in the sediment.
- Five reef intertidal communities - defined by benthic light, proportion of mud in the sediment and wind speed.
- Three subtidal reef communities - defined by depth and water temperature (Figure 7).

4.4.3 Biomass data

Seagrass above-ground biomass was estimated using visual estimates, a widely-used, non-destructive method often used in time-series analysis (Aragones & Marsh, 2000; Rasheed, 1999, 2004) and assessments of meadow-scale change (McKenna et al., 2015; Rasheed & Unsworth, 2011). Biomass is routinely measured in a range of seagrass mapping and monitoring programs in the GBRWHA and adjacent estuaries. Using biomass as a common metric allowed us to create a compilation of comparable data from sites surveyed between 1995 and 2018 for analysis.

Seagrass data comes from long-term seagrass mapping and monitoring programs which had four major purposes: (1) cross-shelf subtidal surveys in the mid-1990s and again in 2003-2005; (2) sporadic mapping of intertidal meadows as part of an oil spill response atlas between 2001 and 2014; (3) targeted mapping projects; and (4) frequent (at least annual) and spatially intense mapping and monitoring in six Queensland ports (Table 1).

For each of these surveys, sites with an area of 5m radius were haphazardly allocated to ensure good spatial coverage. Above-ground biomass was assessed visually within three replicate quadrats (50 $\times$ 50 cm) randomly placed within each site. Site biomass was calculated from an average of the three quadrats and scaled up to grams dry weight m^{-2} (gDW m^{-2}). Following each survey, the visual assessment is calibrated for each individual observer against harvested biomass samples (Mellors, 1991).

We used the community models developed in Chapter 3 to predict community type for each survey site (Figure 12). We followed Collier et al.'s (2020) recommendation that community types be re-assessed prior to analysis so that classifications are fit-for-purpose depending on the scale and the desired state indicator used. Where communities had very similar biomass and species, or one was represented by only a very small area they were combined. We therefore combined data from reef intertidal (RI) communities RI4 and RI5 due to their similar biomass and species composition, and because the area of RI5 is just <1 km^2 and RI4 is 9 km^2. We also combined data for coastal intertidal (CI) communities CI2 and CI3 because of the similarity between these adjacent communities in terms of species composition and biomass. Combining these very similar communities let us conduct a more robust analysis due to the increased sample size.

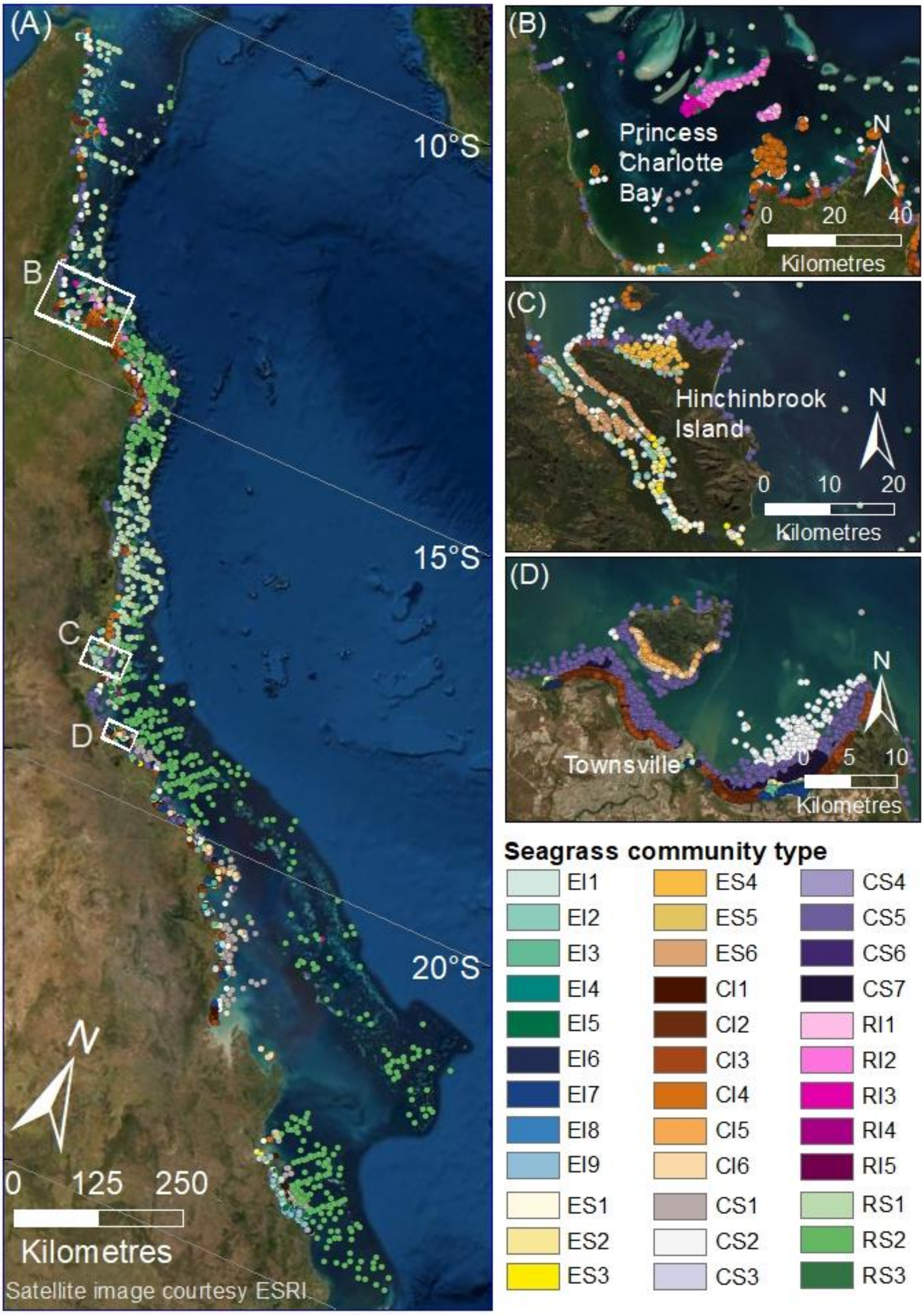

Figure 12: (A) Seagrass site data used to define biomass desired state for thirty-six seagrass communities on the Great Barrier Reef: estuary intertidal (EI1-EI9), estuary subtidal (ES1-ES6), coastal intertidal (CI1-CI6), coastal subtidal (CS1-CS7), reef intertidal (RI1-RI5), and reef subtidal (RS1-RS3). (B-D) More detailed maps demonstrate the complex mix of different communities within relatively small areas.

4.4.4 Statistical analysis

We applied the methods developed by Collier et al (2020) used to define biomass desired state for seagrass communities in Cleveland Bay. For each seagrass community we examined temporal trends in above-ground biomass using Generalized Linear Models (GLMs) fitted with a Tweedie distribution (Tweedie, 1984). The only covariate included in the model was year as the available environmental covariates were previously used to determine seagrass communities, and we were interested only in identifying years of maximum biomass. We estimated uncertainty by calculating the 95% confidence interval (CI) of model predictions for each year. We did not include years with low sample size (number of sites<15) in the analysis due to the high variability and uncertainty in those mean biomass estimates.

We aimed to set ambitious targets. The reference data set used to define biomass desired state for each community therefore only included years when biomass was highest. Specifically, the year where maximum seagrass biomass was present, plus those years where biomass was not significantly different from the maximum year. Significance was determined using Wald post hoc comparisons. In several communities, maximum biomass was significantly different from all other years. Where this occurred, that year was considered an outlier year that was unlikely to represent an achievable desired state, and the reference data set was based on the mean of the four highest biomass years. Four was selected because it is the average number of years used to define desired state for communities without outlier years. Desired state was determined as average above-ground biomass of the reference data for each community, bounded by the 99% confidence intervals. Desired state estimates are not presented for communities with <5 years of adequate data due to low certainty in these estimates. All plots were created using the *ggplot* package in R (Wickham, 2016).

4.4.5 Reporting against biomass desired state

We define desired state as an aspirational target against which to assess future annual growing season condition as per Collier et al. (2020). Setting seagrass desired state for the GBRWHA required an approach designed for this large management area and the diverse and dynamic nature of the seagrass meadows.

The definition of desired state provides a benchmark (Collier et al., 2020), where:

- *Desired state is met* with a high level of confidence placed in that assessment if the mean biomass exceeds desired state and its upper CI (Figure 13a).
- *Desired state is not met* with a high level of confidence if the mean biomass is lower than the lower CI of the desired state (Figure 13b).
- *Desired state is met with moderate confidence* when: 1. the mean biomass of a community is above the upper CI of desired state but the CI overlaps with desired state range; or 2. when the mean biomass of a community is within the desired state range (Figure 13c).
- *Desired state is not met with moderate confidence* when the mean biomass is lower than the desired state range, but the upper biomass CI falls within the desired state range (Figure 13d).

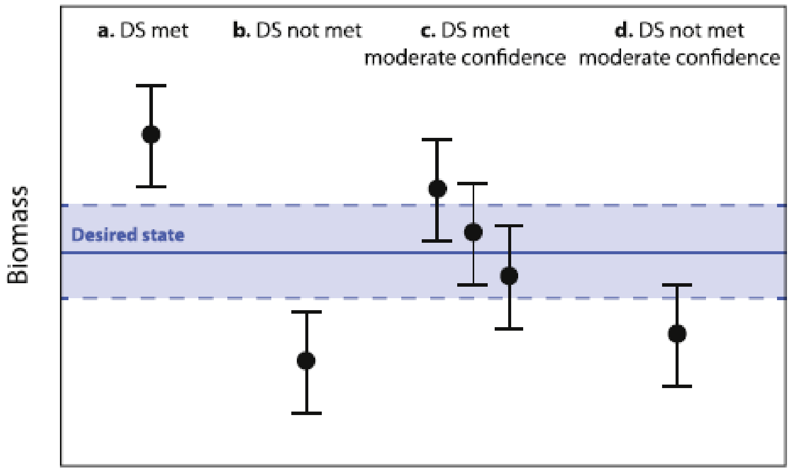

Figure 13: Interpretation of whether desired state (DS) is met for above-ground biomass (mean ± 99% confidence intervals). Figure reproduced with permission from Collier et al. (2020).

4.5 Results

Based on the available data we were able to establish desired state for 11 of the 15 estuarine communities, all 12 of the coastal communities, and only 2 out of 7 reef communities (Table 7). The maximum biomass values were from periods roughly a decade apart: 1995 – 1997, 2004 – 2008, and in 2017. The reference years used to define desired state were even further scattered through time and included all years except 2000, 2010, 2012 and 2015. Desired state is expressed as a range within 99% confidence intervals, but for simplicity the mean is also given. Desired state was influenced by the community species composition, which is in turn affected by environmental setting and, as such, desired state biomass varied by more than an order of magnitude among communities (Table 7).

Biomass reductions below desired state occurred in all estuarine and coastal communities at some time over the 24 years, but most notably beginning in some communities in 2008 and all communities by 2010 corresponding with an extended period of La Niña climate conditions that affected the entire GBRWHA region (Bureau of Meteorology, 2020b). Recovery to desired state has not occurred in most of those communities, although some recovered and attained desired state in 2017 and 2018. Desired state and trends in biomass are described in further detail below according to habitat type.

Table 7: Seagrass community, desired state reference years, number of sites in analysis, number of years in analysis, and above-ground biomass desired state (mean with 99% confidence intervals). n/a = desired state is not estimated for this community because either no years had n≥15 biomass records, or because there were <5 years of adequate survey data.

Community	Desired state reference years (max. year **bold**)	Number of sites	Number of years in analysis	Desired state biomass (gDW m^{-2})	
				Mean	99% CI
Estuary Intertidal 1	2002, **2004**, 2005, 2008	1976	20	14.2	8.9, 19.5
Estuary Intertidal 2	<5 years of adequate data	334	n/a	n/a	n/a
Estuary Intertidal 3	No years with n≥15 sites	114	n/a	n/a	n/a
Estuary Intertidal 4	1995, 1996, 2001, 2002, 2003, **2004**, 2005, 2006, 2007, 2008	1003	21	23.5	18.8, 28.1
Estuary Intertidal 5	2006, 2007, **2008**	1407	16	13.7	7.9, 19.4
Estuary Intertidal 6	<5 years of adequate data	165	n/a	n/a	n/a
Estuary Intertidal 7	**2007**, 2014, 2017, 2018	268	11	40.7	30.3, 51.0
Estuary Intertidal 8	1995, 1996, 2001, 2002, **2004**, 2005, 2006	1181	21	32.3	27.9, 36.6
Estuary Intertidal 9	2002, **2007**, 2008, 2009	3936	16	6.2	4.7, 7.7
Estuary Subtidal 1	1996, 2002, 2007, **2008**	1785	20	3.0	1.8, 4.2
Estuary Subtidal 2	1996, 2002, **2007**, 2017, 2018	1405	16	3.8	2.4, 5.2
Estuary Subtidal 3	1996, 2002, **2007**, 2008, 2009	2629	17	5.0	3.9, 6.2
Estuary Subtidal 4	1995, **1996**, 2002, 2004	3406	21	3.1	2.2, 3.9
Estuary Subtidal 5	2004, **2005**, 2006	592	16	74.1	58.7, 89.4
Estuary Subtidal 6	<5 years of adequate data	1264	n/a	n/a	n/a
Coastal Intertidal 1	2002, 2005, 2007, **2008**	2434	16	9.6	7.3, 12.0
Coastal Intertidal 2/3	2001, **2007**, 2008, 2017	2526	18	21.5	17.9, 25.2
Coastal Intertidal 4	1996, 2001, **2014**	408	6	9.1	3.7, 14.4
Coastal Intertidal 5	2009, 2014, **2017**, 2018	420	10	7.8	5.5, 10.1
Coastal Intertidal 6	**2007**, 2008, 2017, 2018	499	14	22.1	16.6, 27.7
Coastal Subtidal 1	**1999**, 2008, 2014, 2018	829	12	1.3	0.6, 1.9
Coastal Subtidal 2	1996, **1999**, 2001	2580	20	10.7	8.6, 12.8
Coastal Subtidal 3	2002, 2004, **2005**, 2006, 2007, 2008	2600	16	8.8	7.1, 10.4

Coastal Subtidal 4	**2005**, 2006, 2007, 2008	1531	19	12.4	10.0, 14.8
Coastal Subtidal 5	1996, **2007**, 2014, 2016	4824	19	11.3	9.9, 12.6
Coastal Subtidal 6	**2004**, 2005, 2006	1335	18	39.1	28.7, 49.4
Coastal Subtidal 7	1996, **2007**, 2017, 2018	1157	13	15.1	11.5, 18.6
Reef Intertidal 1	**1996**, 1997, 2011	224	5	4.4	2.0, 6.8
Reef Intertidal 2	**1997**, 2011, 2013	340	6	3.1	2.1, 4.2
Reef Intertidal 3	<5 years of adequate data	357	n/a	n/a	n/a
Reef Intertidal 4/5	<5 years of adequate data	313	n/a	n/a	n/a
Reef Subtidal 1	No years with n≥15 sites	33	n/a	n/a	n/a
Reef Subtidal 2	No years with n≥15 sites	53	n/a	n/a	n/a
Reef Subtidal 3	<5 years of adequate data	286	n/a	n/a	n/a

4.5.1 Estuarine communities

The desired state of estuarine community biomass was variable among communities. Within a community, the confidence intervals were narrow and often had a large number of years (3 to 10 years) contributing to desired state. For intertidal communities, desired state was greatest (mean biomass >32.3 –40.7 gDW m^{-2}) in communities where *Z. muelleri* subsp. *capricorni* was overwhelmingly the dominant species (EI7 and EI8), compared with communities where *Z. muelleri* subsp. *capricorni* was still dominant but lower biomass species such as *H. uninervis* and *H. ovalis* also frequently occurred (EI9 and EI4; Figure 14, Table 6, Table 7). The most extensive estuarine intertidal community was EI1 (desired state range: 8.9 - 19.5 gDW m^{-2}) which was predicted to cover a total 288 km^2 from the northern to southern extent of the GBRWHA (Figure 7, Table 6). The highest biomass community EI7 was the second largest community, predicted to cover a total 156 km^2 between Townsville and Shoalwater Bay (Figure 7, Figure 14, Table 6, Table 7). There was insufficient data to identify desired state for communities EI2, EI3, and EI6.

In estuarine subtidal communities, desired state biomass for community ES5 was the greatest of any estuarine community (desired state range: 58.7 – 89.4 gDW m^{-2}) due to a period of extremely high biomass in this community between 2004 and 2007 (Figure 14, Table 7). Like many estuarine communities ES5 was dominated by *Z. muelleri* subsp. *capricorni*, but with relatively higher frequencies of *H. uninervis* and the high biomass species *C. serrulata* compared with other subtidal communities (Table 6). Desired state was considerably lower in the remaining estuarine subtidal communities dominated by the low biomass *Halophila* species (mean biomass < 5 gDW m^{-2}; Figure 14; Table 6, Table 7). The most extensive community was the *H. ovalis* and *H. decipiens* dominated community ES1, which was predicted to cover ~182 km^2 of estuarine waters deeper than 2.9 m between the northern and southern extent of the GBRWHA (Figure 7; Table 6). There was insufficient data to identify desired state for community ES6.

Three estuarine intertidal communities and four subtidal communities had data that extended back to the mid-1990s (Figure 14). Of these, there was support for a biomass peak in 1995-1996 in intertidal communities EI4 and EI8 and in all subtidal communities. All intertidal communities experienced a period between approximately 2004 – 2008 where biomass met desired state most years, followed by a period of decline. This peak also occurred in subtidal communities in the same period but was much shorter – generally only 1-2 years. Desired state was not met in any intertidal communities by 2009, nor any subtidal communities by 2010, and biomass remained very low until 2015. Despite small increases in biomass in all estuarine communities from 2016, only communities EI7 and ES2 have recovered to the extent that biomass again met desired state, which occurred in 2017 and 2018 (Figure 14).

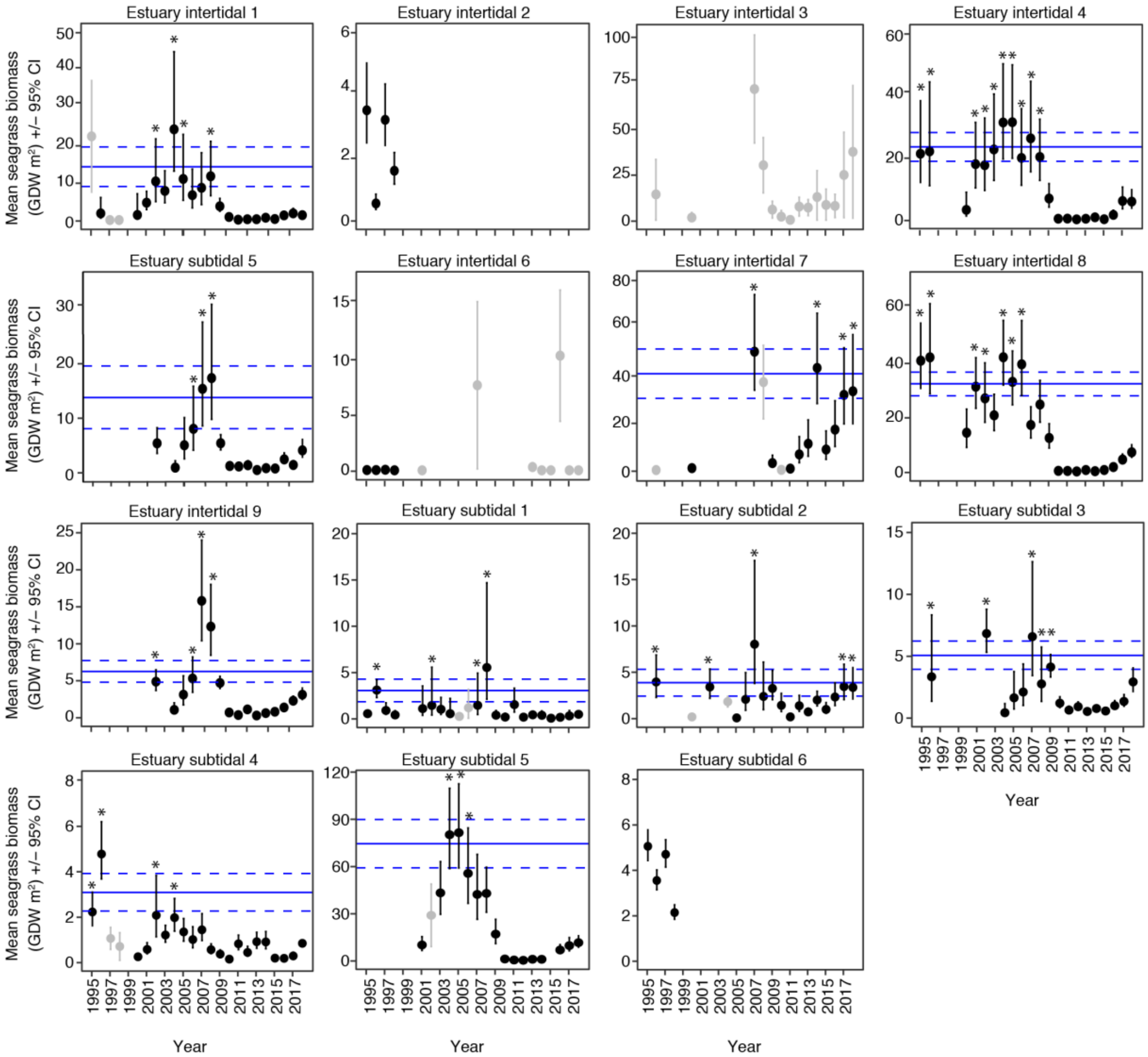

Figure 14: Annual mean above-ground biomass (gDW m^{-2} ±95% CI) for estuarine intertidal and subtidal seagrass communities, 1995 – 2018. Seagrass above-ground biomass desired state (solid blue line) with upper and lower 99% CIs (dashed blue lines). Asterisks indicate reference years used to set desired state. Years with values in grey were not included in desired state analyses due to low sample size (n<15) in that community. Desired state is not presented for communities where there were <5 years of adequate data.

4.5.2 Coastal communities

Desired state biomass was similar for intertidal coastal communities CI1, CI4, and CI5 (mean biomass range 7.8 – 9.6 gDW m^{-2}; Figure 15; Table 7) despite considerable variation in their dominant species and distribution (Figure 7; Table 6). Desired state biomass was much higher for communities CI2/3 and CI6 (mean biomass 21.5 – 22.1 gDW m^{-2}; Figure 15; Table 7). Community CI2/3 is the most extensive intertidal coastal community (296 km^2); it was found in warm coastal waters throughout the GBRWHA and dominated by *H. uninervis*, *Z. muelleri* subsp. *capricorni* and *H. ovalis* (Figure 7; Table 6).

Coastal subtidal communities were spatially dominated by three large, adjacent communities in waters deeper than 1.6 m MSL: community CS5 (2938 km^2) in low-current near-shore waters, transitioning to CS2 (4575 km^2) in higher-current environments further offshore, and finally to CS1 (7589 km^2) in waters deeper than 12.6 m MSL (Figure 7; Table 6). Desired state biomass was the same for the two shallow subtidal communities CS2 and CS5 (mean biomass 10.7 – 11.3 gDW m^{-2}; Figure 15; Table 7), likely due to the similar species mix in these communities that included *H. uninervis*, *H. ovalis*, *H. spinulosa*, *H. decipiens*, *C. serrulata* (Table 6). Desired state biomass in the deep subtidal community CS1 was much lower (desired state range: 0.6 - 1.9 gDW m^{-2}; Figure 15; Table 7) due to the dominance of the low-biomass species *H. decipiens* (Table 6). Coastal subtidal communities CS3, CS4, CS6 and CS7 are found in small patches throughout the GBRWHA that cover a relatively small total area (total area range: 62 – 161 km^2; Figure 7; Table 6). Desired state biomass varied greatly among these communities, from 7.1 – 10.4 gDW m^{-2} (desired state range) for the *H. ovalis* dominated CS3, to 28.7 – 49.4 gDW m^{-2} for the *Z. muelleri* subsp. *capricorni* dominated CS6 (Figure 15; Table 7).

Coastal communities experienced a period of peak biomass and years where desired state was often met over a much larger time frame than for estuarine communities - between 2001 and 2009 for intertidal communities and 2001 and 2008 for subtidal communities (Figure 15). Like estuaries, there was evidence of another biomass peak in the mid-1990s for most subtidal but not intertidal coastal communities. Biomass declines recorded in estuarine communities occurred at the same time for coastal subtidal communities (2009), but generally occurred one year later for coastal intertidal communities. Biomass in all coastal communities did not meet desired state by 2010. Despite biomass increases in all coastal subtidal communities some time between 2012 and 2018, just two communities have met biomass desired state since the widespread decline - CS4 in 2016 and CS7 in 2017. Signs of recovery were much faster in intertidal communities. Biomass began to increase after 2-3 years in most coastal intertidal communities compared with 6-7 years for estuarine communities. Desired state was met in 2014 in CI4 and CI5, and 2017 for CI6, but has not been met in communities CI1 and CI2/3 despite biomass increases between 2015 and 2018 (Figure 15).

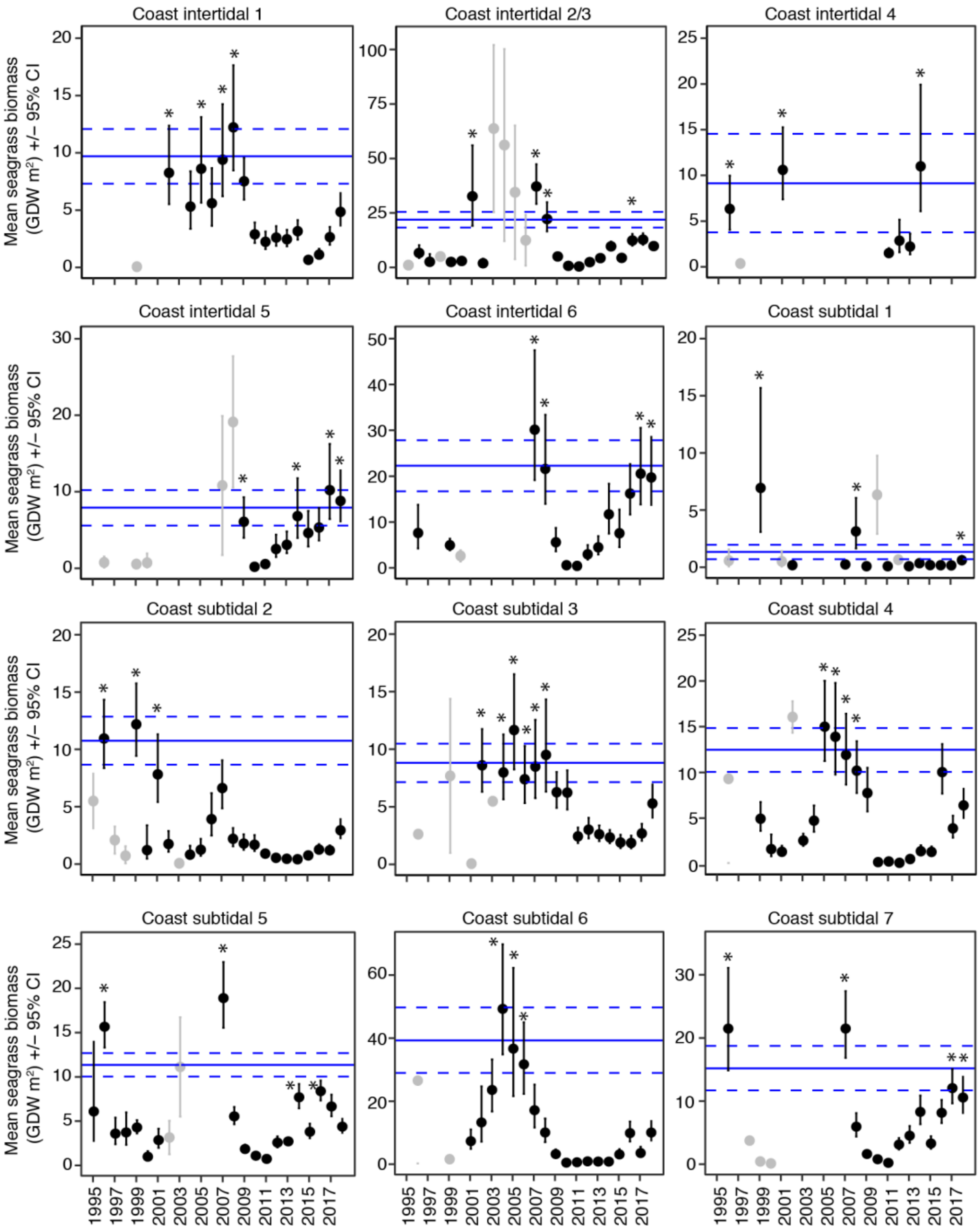

Figure 15: Annual mean above-ground biomass (±95% CI) for coastal intertidal and subtidal seagrass communities, 1995 – 2018. Seagrass above-ground biomass desired state (solid blue line) with upper and lower 99% CIs (dashed blue lines). Asterisks indicate reference years used to set desired state. Years with values in grey were not included in desired state analyses due to low sample size (n<15) in that community.

4.5.3 Reef communities

Desired state estimates were limited to two reef communities (RI1 and RI2) due to the lack of long-term biomass monitoring in other reef areas. For the deeper reef subtidal communities RS1 and RS2 (>8 m MSL) there was insufficient data (i.e. <15 sites) in all of the years sampled, while communities RI3, RI4/5 and RS3 had just 2 - 3 years of adequate data.

Annual biomass in reef intertidal communities RI1 and RI2 was much lower than communities RI3, RI4/5 and RS3, and desired state biomass was relatively low compared with many estuarine and coastal communities (mean biomass <5 gDW m^{-2}; Figure 16; Table 7). Communities RI1 and RI2 were found throughout the GBRWHA and dominated by the common reef-top species *T. hemprichii* (Table 6, Figure 7). Only one reef subtidal community (RS3) had any years (n=3) where sample size was sufficient to estimate mean biomass with confidence. This community represents the highly diverse and high biomass transition zone between intertidal reef-tops and deeper (>8 m MSL) reef communities found throughout the GBRWHA (Table 6, Figure 7). The estimated total area of just 623 km^2 is considerably smaller than the expansive deeper reef communities RS1 (19,434 km^2) and RS2 (49,052 km^2) (Figure 7; Table 6). The limited data for these deep reef communities indicates much lower biomass than RS3, with mean annual biomass ranging from 0 - 8 gDW m^{-2} in the *H. decipiens* dominated community RS1, and 0.1 - 20 gDW m^{-2} in the mixed species community RS2 (Figure 16; Table 6, Table 7).

The limited data available for reef communities indicates annual biomass is relatively stable (Figure 16). Unfortunately no biomass data were collected for any reef communities between 2004 and 2010 (inclusive), so we are unsure if the large biomass peaks that occurred for many estuarine and coastal communities in 2004 - 2007 also occurred on reefs. When data collection resumed for RI1 and RI2 in 2011 both communities met biomass desired state, while in that same year no estuarine or coastal communities did. This indicates the dramatic biomass declines estuarine or coastal communities experienced either did not occur for reef communities or, if they did, reef communities recovered much faster than those closer to land (Figure 16).

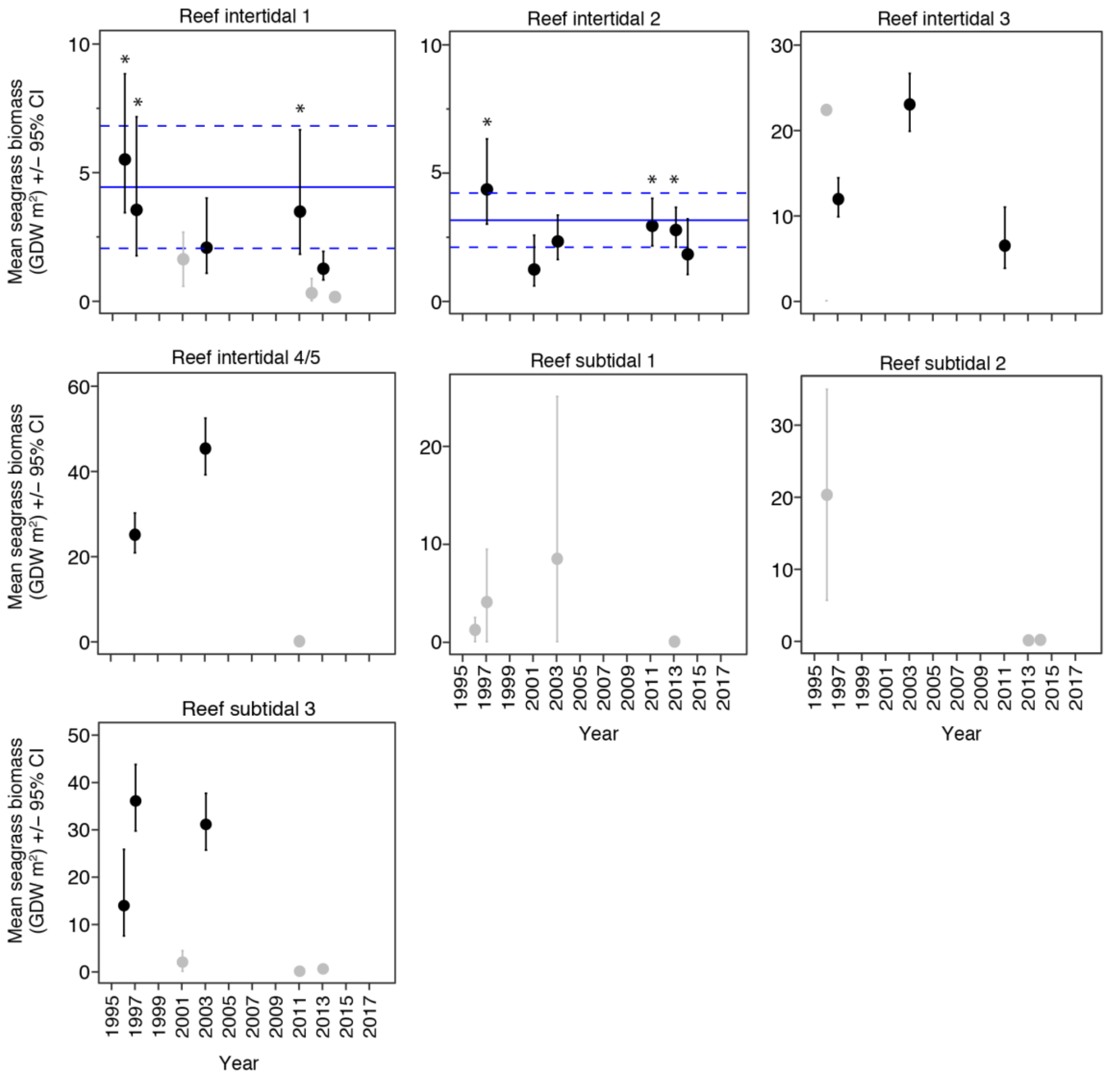

Figure 16: Annual mean above-ground biomass (+95% CI) for reef intertidal and subtidal seagrass communities, 1995 - 2018. Seagrass above-ground biomass desired state (solid blue line) with upper and lower 99% CIs (dashed blue lines). Asterisks indicate reference years used to set desired state. Years with values in grey were not included in desired state analyses due to low sample size (n<15) in that community. Desired state is not presented for communities where there were <5 years of adequate data.

4.6 Discussion

We overcame the challenges of setting desired state in the GBRWHA by using more than two decades of survey data to examine trends in dynamic and diverse seagrass communities and to identify attainable biomass for each. The desired states of seagrass communities vary by over an order of magnitude when expressed as biomass, because of the differences in species present and the environmental setting of each community. Trajectories and trends vary among seagrass communities, as does contemporary biomass relative to desired state. We demonstrate how targets can be conveyed in the face of ecological, spatial and temporal complexity, and contribute towards informed decision-making of this critical habitat in an iconic region. This approach has benefits when making broad assessments of seagrass desired state and when identifying critical information gaps.

4.6.1 Conservation and management applications

Expressing ecosystem condition in terms of management goals underpins assessment and planning for conservation of diversity and ecological function (Borja et al., 2013; Hallett et al., 2016a). Desired state identifies an aspiration for seagrass communities which adheres to principals of objectivity, is based on historical data, and acknowledges uncertainty (Samhouri et al., 2012). Desired state as we have presented it follows the approach outlined in Collier et al (2020), where trends in seagrass biomass are examined and we identify which communities have recovered and which have failed to attain desired state in the past decade. In doing so, we can identify communities that are at risk and may be failing to deliver their ecosystem services.

Our approach allows management activities and interventions to be prioritised based on observation of trends relative to desired state; a necessary task for managing large and complex ecosystems. Risks to marine ecosystems including seagrasses such as industrial and port development, coastal urbanisation and infrastructure, land clearing and climate change have been described, and guide over-arching management plans (Grech et al., 2012; Griffiths et al., 2020; Tulloch et al., 2020). However, the scale and functional relationships of the pressures that cause loss or prevent recovery are also required for more targeted interventions (Samhouri et al., 2012; Virnstein, 1999). Economic trade-offs can also be used to prioritise conservation efforts (Beher et al., 2016). While our analysis did not aim to resolve and quantify pressures, desired state can be applied to do so. For example, the influence of rivers adjacent to Cleveland Bay in the central GBRWHA were correlated to seagrass condition indicators, and sediment load targets to meet desired state were identified (Lambert et al., 2019).

4.6.2 Considerations and limitations

Over-arching frameworks can be applied when assigning targets (e.g. Samhouri et al., 2012); however, there are unique circumstances and challenges when they are applied to new ecosystems and management areas. Our assessment was undertaken across a large management area (350,000 km^2) using biomass data collected from some communities with a geographic range that extends thousands of kilometres. Our approach makes it possible to convey trends in a tangible manner in a large, dynamic and diverse region. It can be easily adapted to incorporate local features specific to individual bays and communities and local assessments of trends in desired state (Collier et al., 2020) to match the scale of the management question where data at that scale is available.

Management questions and jurisdictions also operate at a range of scales which need to be accommodated. Desired state may also need to be refined to include changes to management goals, indicators of seagrass condition and resilience, pressures, and in society's expectations. Indicators of resilience provide insight into the extent to which the habitat can resist future impacts or recover following decline (Collier et al., 2020; Irving et al., 2016; O'Brien et al., 2017). There are additional influences on desired state that relate to the ecosystem services provided by seagrass habitat such as herbivory (Scott et al., 2018) and specific shifts due to environmental stressors (Roca et al., 2016) that would also be beneficial for seagrass management if included in future analysis.

4.6.3 Seagrass communities of the Great Barrier Reef World Heritage Area

The GBRWHA is not a single environmental unit but is made up of many compartments with differing risk profiles and sensitivity to impacts. Threats to the 36 communities are spread unevenly with biases towards coastal and southern locations. Communities at higher risk require greater attention from management authorities as they are likely to preview trends for the wider GBRWHA. Estuaries, where threats to seagrass communities accumulate (Grech et al., 2011), are data deficient.

Where possible, we set a desired state target for each community which represents an achievable goal based on the history of average years for that community's biomass. In doing so we set a benchmark for management authorities of the performance of the framework they have set in place to ensure the outcome "maintain diversity of species and ecological habitats in at least a good condition and with a stable to improving trend" is met. Fundamental to assessing this is adequate data at appropriate scales that is distributed across the identified communities. The seagrass data we used is from historical surveys and ongoing monitoring programs that were not designed to examine long-term seagrass trends at the scale of the GBRWHA. To achieve that, a survey and monitoring program would need to include the spectrum of community types and sample across each community's spatial extent. Ensuring that occurs will be a challenge for the future.

4.6.4 Reef communities

Using historical data resulted in some gaps in our ability to determine desired state. Most reef communities had limited data and we were unable to provide a desired state for them with any level of confidence. The two *Thalassia*-dominated reef communities with enough data (RI1 and RI2) did not exhibit the same biomass decline or recovery that occurred in coastal and estuarine communities. Reef communities have low risk of exposure to discharge from rivers (Bainbridge et al., 2018) and coastal activities (Grech et al., 2011; York et al., 2015). Shallow reef communities are vulnerable to local physical disturbances from cyclones, which can have lasting impacts to habitat substrate, alter feedbacks that maintain substrate, and leave a legacy of decline (McKenzie et al., 2020a). The persistent species common in the shallow reef communities have slow rates of recovery if they decline (Adams et al., 2016; Kilminster et al., 2015; O'Brien et al., 2017), and seagrass communities that are not conditioned with phenotypic plasticity to stressors such as riverine discharge can be more sensitive to them when they occur (Maxwell et al., 2014).

Reef communities have not been a focus of long-term monitoring programs that assess biomass because they fall outside of the high-risk areas when compared to estuary and coastal seagrass (Grech et al., 2011). The most extensive reef subtidal communities, both of which have an area that is almost two orders of magnitude greater area than any other community, have little temporally resolved data for examining trends and assigning desired state. The most recent assessments of reef intertidal communities (RI1 and RI2; 2012-2014) indicate a decline below desired state; however we have low confidence in that assessment due to the low sample size, and because biomass estimates for these communities between 2011 and 2014 are based on reef-top surveys that did not resample the same reefs each year. Subtidal reef communities in particular have not been routinely assessed. These communities' extensive potential distribution, ephemeral nature, and remote location make them difficult to assess for desired state or to design a program that can effectively monitor their status in a cost-effective way.

4.6.5 Coastal communities

The diversity of coastal communities is reflected in the large range in biomass desired states, recent trends and contemporary biomass. Coasts face a range of pressures originating on land and in the coastal zone (Grech et al., 2012; Grech et al., 2011; Rasheed et al., 2014; York et al., 2015). Monitoring efforts are greatest in coastal areas of the GBRWHA (Table 1) so they have the greatest amount of data available to assess desired state, trends and trajectories. Coastal seagrass communities have an assortment of dominant species and species life history strategies (Kilminster et al., 2015) and were classified using different environmental conditions including current speed, mud levels, depth or relative tidal exposure, water temperature and salinity (Chapter 3).

The biomass of coastal communities has fluctuated greatly over the >20 year data set, varying by more than an order of magnitude in all communities. Most have failed to fully recover following declines starting around 2008 although recent trends are generally of increasing biomass. The declines resulted from extreme weather associated with a series of La Niña events, which included high rainfall and river discharge, with high sediment and nutrients loads delivered to reef waters (Coles et al., 2015; Collier et al., 2012; McKenna et al., 2015; Petus et al., 2014; Rasheed et al., 2014). These declines were recorded in all locations where long-term monitoring occurs: Abbot Point (Van De Wetering et al., 2020b), Cairns (Reason et al., 2020), Hay Point (York & Rasheed, 2020), Gladstone (Smith et al., 2020), Mourilyan (Van De Wetering et al., 2020a), and Townsville (McKenna et al., 2020). The 2010-2012 La Niña event brought with it Australia's wettest 24-month period on record, widespread rainfall and flooding throughout Queensland, and several tropical cyclones. This included severe tropical cyclone Yasi which crossed the coast near Hinchinbrook Island as the strongest cyclone to make landfall in Queensland in almost a century (Bureau of Meteorology, 2020b). Declines in biomass also occurred in the period 1997 – 2003. These were not as severe, but monitoring in that period also was less consistent with fewer dedicated long-term monitoring programs.

Biomass in coastal intertidal communities is improving, with two communities recovering to desired state in recent years (CI5 and CI6) located in open coastal areas (Chapter 3). Although the recent trend has been positive, subtidal communities in enclosed coastal waters (Chapter 3) have not recovered to desired state. These inshore regions of the GBRWHA are turbid and there are legacy effects of river loads (Fabricius et al., 2014; Fabricius et al., 2016; Margvelashvili et al., 2018) making some communities especially vulnerable to water quality decline. Shifts in seagrass species and community biomass may take several years to more than a decade to recover (Birch & Birch, 1984; Rasheed et al., 2014).

4.6.6 Estuary communities

Estuary seagrass communities in or adjacent to the GBRWHA cover small areas within inlets, rivers and tidal creeks where they are subject to large event-driven fluctuations. These communities show a consistent trend of decline following 2008, and very poor levels of recovery since. The biomass desired state of these communities varies considerably, ranging from the largest biomass desired state (ES5), to the smallest (EI6), among all of the community types we examined. Only one out of 15 estuary communities have recovered to desired state in recent years. The estuary communities are delimited based on latitude (Chapter 3). The biomass data for each of these communities tends to come from the same spatially constrained (latitudinal) areas in each sampling event and a better spread of data would improve our analysis at the GBRWHA wide scale. Lack of recovery to desired state in these communities runs parallel to those in the adjacent enclosed coastal and subtidal coastal communities.

Estuaries adjacent to the GBRWHA are at the land-sea interface and at a jurisdictional border, making monitoring and management plans difficult to implement. The estuaries are not within the GBRWHA or GBRMP, so fall to state or local authorities to manage. The spatial coverage of biomass data from estuary communities is highly fragmented and mostly limited to estuaries adjacent to ports where there are long-term monitoring programs. Spatially explicit environmental data needed to assign estuary seagrass into community types was not readily available (Chapter 3). Therefore, there is less certainty in predictions of where estuary seagrasses and seagrass communities occur.

4.6.7 Conclusions

The GBRWHA and adjacent estuaries support diverse seagrass community assemblages, with wide-ranging desired states when expressed as biomass. Individual community assessments were more appropriate when assessing seagrass condition across the region. A number of the coastal subtidal and estuarine communities have had protracted periods of reduced biomass resulting from previous extreme weather conditions and have not attained their desired state in recent years. Our analysis for the GBRWHA seagrass communities is a step forward in understanding complexities in such a large management area. It challenges the temptation to report a simplified version of seagrass trends addressing the GBRWHA as if it were a single entity. The desired state analysis points to a decadal cycle of loss and recovery for many communities rather than a chronic decline in condition. Predicted increased frequencies of La Niña conditions means the ability for communities to bounce back in the future may be less certain (Rasheed & Unsworth, 2011). Most of the coast and estuary communities are from bay-wide monitoring programs with large gaps in between bays. For reef communities, under-representation in the data set makes it difficult to calculate desired state, contemporary condition, and trends. With mounting pressures on seagrass habitat due to climate change, increasing population on the coastlines, and deteriorated catchments, it is more important than ever to identify and work towards an attainable desired state.

5.0 CONCLUSION

5.1 Summary

The National Environment Science Program Tropical Water Quality Hub was designed to initiate innovative research on biodiversity and climate science to assist decision makers to understand, manage and conserve Australia's unique tropical marine environment. The science program focus is on the Great Barrier Reef but also extends to other important tropical marine systems in northern Australia. The current program has three themes:

1. Improved understanding of the impacts, including cumulative impacts, and pressures on priority freshwater, coastal and marine ecosystems and species.
2. Maximise the resilience of vulnerable species to the impacts of climate change and climate variability by reducing other pressures, including poor water quality.
3. Natural resource management improvements based on sound understanding of the status and long-term trends of priority species and systems.

The TropWATER JCU Seagrass Group's research program (the Marine Ecology Group at Fisheries Queensland prior to 2013) extends back to the mid-1980s. The program has evolved from a focus on large-scale mapping to a comprehensive program that includes research and monitoring of ecosystem processes, biophysical interactions, temporal trends, risks and resilience, economic values, report cards, and ecological connectivity. The common theme has been to provide data, tools and advice that support environmental management decisions. The NESP projects reported here incorporate those themes, and enabled us to review and revisit data collected over three decades. This data had been underused in terms of extracting patterns and trends that could inform our knowledge of how the GBRWHA seagrass communities are structured and how that could influence an understanding of possible trajectories and outcomes for seagrass communities and seagrass management.

Our findings presented here are the culmination of successive NESP TWQ Hub projects over the last five years. These include our first seagrass data synthesis from 1984-2014 (NESP TWQ Hub Project 3.1) (Carter et al., 2016); a synthesis of light thresholds as guidance for environmental managers was also developed (NESP TWQ Hub Project 3.3) (Collier et al., 2016); and defining seagrass communities, desired state targets, and calculating ecologically relevant load targets for Cleveland Bay (NESP TWQ Hub Project 3.2.1) (Bainbridge et al., 2018; Collier et al., 2020; Lambert et al., 2019). In this project we update the seagrass data synthesis to include data collected 2015 to 2018 (Chapter 2); use this data to model potential seagrass distribution and identify 36 seagrass communities (Chapter 3); and apply the method developed for Cleveland Bay to identify a desired state range for seagrass biomass for the majority of seagrass communities throughout the GBRWHA and adjacent estuaries (Chapter 4).

This body of work provides a huge step forward in our understanding of the complexities of GBRWHA seagrass communities. We provide as tools for management quantitative models and methodologies that can be used to describe the biophysical subsets within the GBRWHA and individual communities of seagrass with a level of detail not previously available. The classification of seagrass into these 36 communities provides an elegant way to interpret this complexity. We provide an approach to setting "desired state" for most of these communities.

This information will facilitate management planning, guide and inform the distribution monitoring activity, help develop realistic management targets, and allow a better assessment of the vulnerability of seagrass community types to the risks inherent in climate variability and coastal processes.

5.2 Management application

The habitat suitability, community classification and desired state analysis provides a reef-scale assessment for communicating and assessing spatial and temporal variability, with potential for implementation in a number of management areas. The focus to date has been on generating these data products and tools. Implementation opportunities will be explored over the coming year.

5.2.1 Spatial planning

The GBRMP which overlays most of the GBRWHA was rezoned in 2004, increasing protected areas to approximately 33% of the park. The process is well documented (Fernandes et al., 2009; Kenworthy et al., 2007) and followed an evolution of zoning approaches from protecting coral reefs and managing user conflicts, to accepting the concept of protecting “representative areas” that included both reef and non-reef habitats. That approach identified 40 non-reef bioregions but in doing so highlighted the scarcity of biophysical data at the scale of the GBRWHA, particularly for remote and deeper locations (Kerrigan et al., 2010). The 2004 rezoning was based on the experience and expertise of scientists working in the region using a qualitative approach. The limitations of this approach were recognized in two ways: explicit descriptions of the fuzziness of the bioregion boundaries, and a caveat that recognised that improvement in data or analytical techniques would refine the bio-regionalisation in the future (Kerrigan et al., 2010).

The approach we have taken in this project, by harvesting and verifying all available information and spatially describing communities in a quantitative way, works towards addressing the previously identified data scarcity and the need for more precise analytical techniques for seagrasses. We defined potential seagrass habitat by choosing a threshold of >0.2 probability to exclude from our analysis areas that were very unlikely to have seagrass; but to retain areas where data is limited and seagrass presence is patchy. Our definition of potential seagrass habitat does not redefine seagrass area at the scale of the GBRWHA. We set realistic desired state for most communities, extending the value of our spatial models beyond simple area management. The process identified those communities where data is deficient and needs to be remedied.

The complex jurisdictions inherent in such a large area influence the distribution of the data available. The majority of the park is managed under Australian Commonwealth legislation. The inshore strip is managed by the Queensland government, with smaller jurisdictions such as port authority management and local government authorities also influencing activities and environmental pressures. As many as 65 legislative instruments can influence management approaches in addition to Australia’s’ common law and customary traditional rights (McGrath, 2011). Only a few of these influence day-to-day management and data availability; the majority of seagrass data emanates from Commonwealth and port authority funding and this has led to a lack of data from estuaries. Estuaries are managed by the Queensland government who are also responsible for adjacent urban and land environmental management. While the GBRWHA lacks the enormous estuarine seagrass systems such as America's Chesapeake Bay, it does include estuaries with important seagrass communities that have potential exposure to multiple threats for which more consistent data would be valuable.

5.2.2 Status and trends

A comprehensive and up-to-date understanding of the Great Barrier Reef, its values, the processes that support it, and the pressures that affect it is fundamental to managing the reef and making informed decisions (Commonwealth of Australia, 2018). To achieve this the GBRMPA has initiated the Reef 2050 Integrated Monitoring and Reporting Program (RIMReP) based on a driver, pressure, state, impact and response model (Udy et al., 2019). For this to be effective, key indicators will be required to monitor condition and trend for habitats such as seagrass meadows, along with trend and resilience values and levels of environmental threat. We contend that to achieve a qualitative data stream that will satisfy the objectives of RIMReP and take into account the spatial extent the GBRWHA requires a series of principles that form a compromise between the need for highly precise data and what is realistic at the GBRWHA scale. These principles are:

1. A hierarchical monitoring design where information is collected and linked at spatial and temporal scales matched to the scales of management actions and reporting products.
2. High resolution information is not required everywhere but is required at key locations.
3. Long-term datasets should be maintained.
4. The program needs to provide information for trend and current status analysis as well as information to inform, calibrate and validate models.
5. Monitoring of pressures and values will be co-located wherever possible.
6. Collect the information once and use it many times.

Our habitat suitability, community classification and desired state estimates reported here provide a framework for a hierarchical monitoring design with coarse scales (intertidal, subtidal, estuary, coast, reef) and fine scales (36 communities), with identified gaps in data consistency. It provides information on trend and current status and the data to calibrate and validate models. Combined with previous research outputs (Grech et al., 2011; Rasheed et al., 2007) and the inshore seagrass Marine Monitoring Program (McKenzie et al., 2020a) we now have the ability to advise on those communities where environmental pressures are greatest. The RIMReP annual business plan outlines priorities for the 2020-21 year. Our research can contribute to a number of these, including:

- Finalise a list of prioritised gaps for indicators and information collection.
- Continue collecting critical information about the Reef.

Our research will enable monitoring programs to be refined to complement a more targeted model able to address policy requirements and Outlook reporting.

5.2.3 Targets and objectives

The planning and policy actions of the 2015 Reef 2050 Long-Term Sustainability Plan (Commonwealth of Australia, 2015) set an aspirational action to "adopt an approach of continuous improvement as part of an adaptive management strategy for the World Heritage Area". Achieving this requires a refinement in understanding the complex ecosystems of the GBRWHA and their interactions. We outline a process in this report that for seagrasses will meet the actions in the Sustainability Plan to develop regionally relevant standards for ecosystem health, prioritise communities critical to reef health in each region and to improve mapping, modelling and monitoring for protection and planning (Reef 2050 Action EHA6/7/15 Reef 2050, 2015).

Within that process and budgetary constraints we still struggle to address data needs at the scale and complexity of the GBRWHA. We also remain within the "silo" of a single habitat type, i.e. seagrass, while fully recognising that interactions with other habitat types such as coral, algae, sponge and mussel beds, and interactions with bioturbation and herbivory, add layers of ecosystem complexity not well addressed by existing data sets and analyses.

5.2.4 Outcomes for Traditional Owners

Rangers combine science with Traditional Ecological Knowledge to conserve and care for sea country. Seagrass monitoring is one of the methods that some ranger groups use to track the health of sea country. Our seagrass distribution model and community classification provide valuable information towards this goal by identification of monitoring and conservation priorities through the lens of western science. With the recently announced expansion of the Queensland Land and Sea Ranger Program, there will be an increase in demand for relevant and up-to-date information, which our analysis provides.

5.2.5 Research opportunities

Expanding the spatial extent of models

This project only analyses data from the GBRWHA. The Torres Strait to the north is connected to the World Heritage Area via the Hiri current (Brinkman et al., 2002) and the distances are likely to be within the range of seagrass propagule distribution (Grech et al., 2016). Seagrass meadows of the Fraser Island coast to the south are also likely to be connected to the GBRWHA through water movement and propagule distribution. Both areas are within the range of possible transfer of seeds by biotic movement (Tol et al., 2017). The jurisdiction boundaries bisect connected ecosystems. TropWATER and other agencies have data for these areas and they could be combined and included in a larger spatial composite and analysis to provide a more complete understanding of what is effectively an extensive seagrass biome.

Data sets extending back to the 1980s are also available for the Gulf of Carpentaria but require verifying and consolidating in a similar manner to the data consolidation we present in Chapter 2. With shallow turbid waters seagrasses in the Gulf are likely to be vulnerable to climate change induced weather variability (Rasheed & Unsworth, 2011), similar to those observed for mangrove forests in the region (Duke et al., 2017).

Levels of protection

Various instruments are used to manage environmental protection and use in the GBRWHA. Protection levels include Queensland State government Fish Habitat Areas which protect direct disturbance to habitats, to more broad protection as in the GBRMP no-take zones which protect seagrass from the expansion of bottom trawl fishing grounds. Port authority "Port Exclusion Zones" have no intentional aim to protect habitat but may do so inadvertently by excluding activities such a bottom trawling. How each of 36 seagrass community types we identify intersect with layers of spatial protection, and the effectiveness of that protection for seagrass, is not quantified and should be. Also important is evaluating indirect risks from environmental threats that overlay protected zones and reduce their value, and interact with the complex responses embedded in prescriptive and non-prescriptive protection approaches identified by Coles and Fortes (2001).

Data gaps and age

Monitoring of seagrass meadows at key locations is ongoing but is biased to southern locations and coastal waters. It is 15 years since a broad-scale survey has been conducted of GBRWHA seagrass meadows that includes deeper subtidal and reef waters, and data on seagrass in estuaries is patchy. This issue is widely canvassed in a recent review of seagrass monitoring programs for RIMReP (Udy et al., 2019).

Detailed areal maps providing seascape configuration data would allow for the development of fragmentation metrics and understanding natural spatial patterning. With critical thresholds of fragmentation evident in seagrass meadows, this spatial data may provide a cost-effective monitoring tool. With present methods it is not feasible to create detailed maps at the scale of the GBRWHA with the precision to detect changes such as fragmentation. New technologies of remote sensing, artificial intelligence/ machine learning may provide that opportunity in the future and these approaches should be explored.

Environmental spatial data is at a much lower resolution (1 km) than is required for high resolution spatial modelling, and most environmental models do not extend into estuaries and narrow coastal strips. Our analysis highlights how little we know about the complex range of environmental conditions that influence seagrass community structure in estuaries, with latitude acting as a proxy in our models due to the lack of environmental data.

Understanding variability

Our analysis has demonstrated that there are significant differences in seagrass biomass between high rainfall La Niña events and drier El Niño times. To provide advice to management it is important to understand the implications these cycles have on seagrass state, including lag times and biomass responses for difference communities. These cycles led to incursions below seagrass desired state and, while the result of natural phenomena, have implications for the animals that rely on seagrass meadows for shelter and food, including turtles and dugong.

Seagrasses may form transitory meadows (Kilminster et al., 2015), where seagrass presence and species composition fluctuate over time. For example, some seagrass meadows are annuals and not present in the GBRWHA for considerable parts of the year during the senescent season. Management arrangements for these species requires understanding their location and presence at different times of the year. Our analysis focussed on seagrass data collected during the seagrass growing season, and most seagrass data available was collected during this period. A senescent season analysis was beyond the scope of this report but remains important information for management decisions.

Challenges for the future

Quantifying the complexity of seagrass communities and the environmental conditions that define community boundaries will improve our understanding of when and where it may be appropriate to intervene with restoration after a seagrass meadow has been lost or impacted following an anthropogenic or climate-related event. Recent modelling on the connectivity among meadows (Grech et al., 2018; Grech et al., 2016; Jackson et al., 2020; Tol et al., 2017) suggests connectivity and propagule exchange at least at scales of 100s of kilometres and should lead to natural recovery without assistance. How robust this modelling is for the different community types and locations is less clear, particularly for more isolated estuaries.

How risk intersects with the vulnerability of different seagrass community types can be used to prioritise management activities. The highest risk to seagrass occurs where anthropogenic risks accumulate; these locations are where industrial ports are located and rivers discharge (Grech et al., 2011). Climate change means that risk profile may be changing. Threats from climate change were not a priority among seagrass researchers when surveyed in 2008 (Grech et al., 2012), but recent large-scale loss of seagrass caused by marine heat waves in Western Australia (Arias-Ortiz et al., 2018), and high temperature intolerance in some seagrass species (Adams et al., 2020; Collier et al., 2018; Collier et al., 2017; Collier et al., 2011) raises concerns for seagrass resilience to warming sea temperatures in the GBRWHA. More intense tropical storms and sea–level rise will also increase the level of environmental risk for some seagrass communities. We can now add a further dimension to an assessment of risk by overlaying the spatial coverage of different seagrass communities, which have unique levels of vulnerability because of their species compositions and environmental settings.

Desired state is defined here in terms of biomass. Understanding the processes facilitating recovery and being able to express desired state in terms of resilience is essential for assessing risk, and for stating health. Resilience is complex and can be defined in various ways (Connolly et al., 2020; Connolly et al., 2018; O'Brien et al., 2017). Our challenge over coming years will be to apply the general principals outlined in this report - adhere to ecological understanding of diversity and complexity while expressing resilience in a way that can be communicated and applied.

5.3 Key Messages

- The seagrass communities we identified inhabit complex regions of the estuaries, coasts and reef lagoon and are themselves made up of complex assemblages of species. The GBRWHA seagrasses do not function as a single environmental unit but are made up of many communities with differing risk profiles and sensitivity to impacts. This detail needs to be addressed when reporting trends and for spatial planning.

- The quantitative data stream required to take into account the spatial extent of the GBRWHA seagrass communities is inconsistent in space and time. Some years are poorly represented, data may be clumped spatially, and there is a bias to southern regions and to intertidal and inshore meadows. Our approach for consolidating data, determining potential seagrass habitat, classifying communities, and calculating desired state is scalable to smaller locations but requires data availability to match that scale. Adhering to the principals of RIMReP can lead to a spatially and temporally balanced data stream and a scalable analysis.
- The history of seagrass in the GBRWHA follows decadal-scale cycles of decline and recovery. Declines are driven by large events, the most recent being a period of successive La Niña events between 2008 and 2012. Assessment of pressures and risk should be undertaken in the context of these weather patterns.
- Some seagrass communities are in better condition than others. Despite a trend of increasing biomass in recent years, recovery to biomass desired state has not occurred for most coastal subtidal and estuarine communities.
- Estuarine communities outside of the GBRWHA or GBRMP are poorly represented in the data. There is an urgent need to understand the environmental setting of the estuaries, pressures on them and assess where intervention may be required to facilitate recovery.
- Incorporating metrics of diffuse factors such as resilience, fragmentation, pressures, disease, bioturbation, herbivory, recovery trajectories, connectivity and source/sinks presents a challenge as data are not available at appropriates scales in space in time or not available. Understanding these processes would enable a more comprehensive ecological assessment including risk and potential for recovery. Interdisciplinary and novel approaches are required to make these mainstays of monitoring and management.

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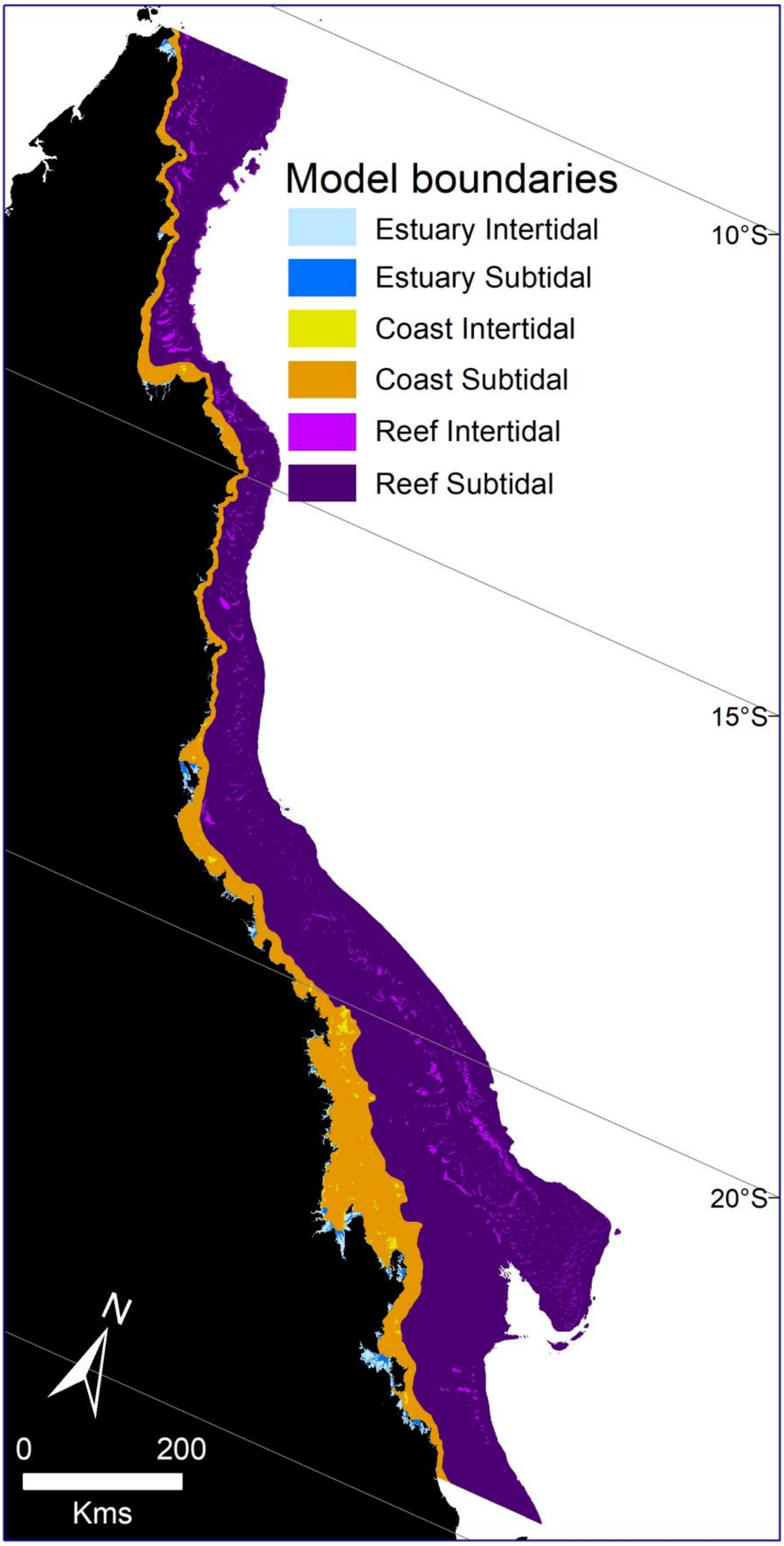
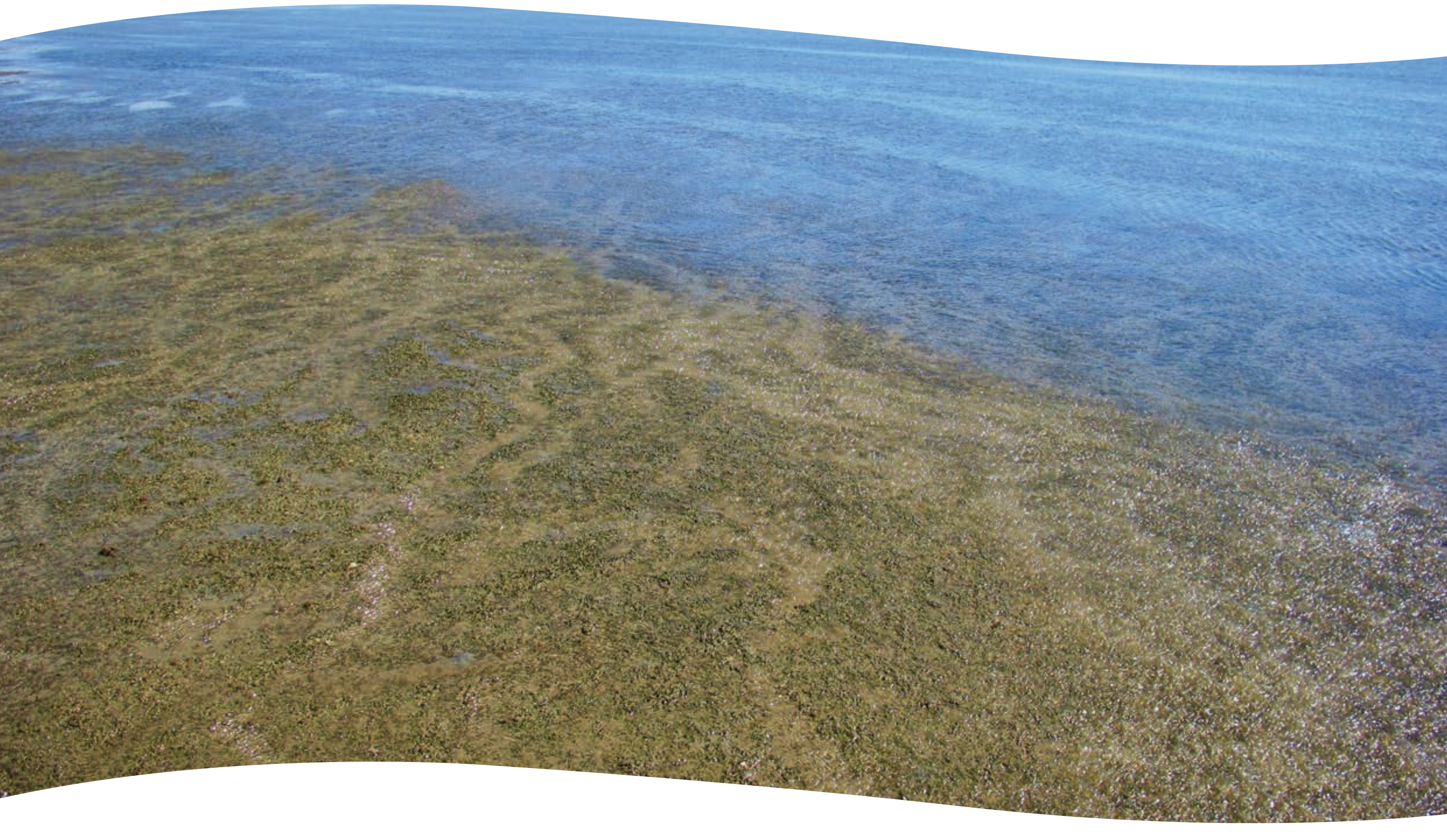
APPENDIX 1: SEAGRASS MODEL BOUNDARIES





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