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Community-specific “desired” states for seagrasses through cycles of loss and recovery

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ABSTRACT

Seagrass habitats provide critical ecosystem services, yet there is ongoing concern over mounting pressures and continuing degradation. Defining a desired state for these habitats is a key step in implementing appropriate management but is often difficult given the challenges of available data and an evaluation of where to set benchmarks. We use more than 20 years of historical seagrass biomass data (1995–2018) for the diverse seagrass communities of Australia’s Great Barrier Reef World Heritage Area (GBRWHA) to develop desired state benchmarks. Desired state for seagrass biomass was estimated for 25 of 36 previously defined seagrass communities 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. We identify a historical, decadal-scale cycle of decline with recovery to desired state in coastal intertidal communities. In contrast a number of the estuary and coastal subtidal communities have not recovered to desired state biomass. Understanding a historical context is critically important for setting benchmarks and making informed management decisions on the present state of seagrass in the GBRWHA. The approach we have developed is scalable for monitoring, management and assessment of pressures for other management areas and for other jurisdictions. Our results guide conservation planning through prioritization of the at-risk seagrass communities that are continuing to fall below their desired state.

1. Introduction

There is continuing concern over the exploitation and degradation of marine ecosystems (Dunic et al., 2021; Halpern et al., 2019; Turschwell et al., 2021). 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 and 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 (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., 2016b; Levin and 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;
- Varying environmental covariates;
- The difficulty in separating long-term trends from short-term disturbance-recovery cycles;
- Poor accounting for the effect of species interactions and changes in species composition; and

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- The difficulty in defining the diversity of habitats and assemblages at relevant scales, and selecting appropriate seagrass community 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 ecosystems are one of the most globally extensive and productive in shallow coastal and marine waters worldwide (Jayathilake and Costello, 2018; Unsworth et al., 2019). The ecosystem services seagrasses provide make them one of the world's most valuable marine ecosystems (Costanza et al., 2014). These services include providing coastal protection, food and shelter for faunal communities including fish, crustaceans, green turtle and dugong, biogeochemical cycling, improved water quality, removal of bacterial pathogens, and a globally significant carbon sink (Coles et al., 1993; de los Santos et al., 2020; Fourqurean et al., 2012; Hemminga and Duarte, 2000; Lamb et al., 2017; Orth et al., 2020; Scott et al., 2018).

Seagrass diversity globally includes 72 species in six families (den Hartog and Kuo, 2006; Green and Short, 2003). Seagrass species 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 (Carter et al., 2021b; Coles et al., 2009; Grech and Coles, 2010; Jayathilake and Costello, 2018; McKenzie et al., 2020b). Seagrass community diversity and the environmental conditions that dictate the niche each community occupies complicates the assessment of 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 and 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 and Amado Filho, 1999; Petus et al., 2014; Rasheed et al., 2014; Short and 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. There is a national imperative to report on the state of the Great Barrier Reef World Heritage Area (GBRWHA), including condition and trends in seagrass meadows. Reporting at this whole-of-reef scale potentially obscures some of the complexities of change in seagrass communities, and in the data available to reliably assess and evaluate them. Assessments need to be presented in ways that accommodate scale and ecological complexity without being convoluted or impractical.

In this paper we define desired state for the extensive and diverse seagrass habitats in Australia'sGBRWHA and adjacent estuaries, including for 25 of 36 identified seagrass communities (Carter et al., 2021b). These communities are diverse in species mix, spatial extent, and in the complex range of environmental conditions that define community boundaries (Carter et al., 2021b). 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, is available as data at the scale of the GBRWHA, 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 seagrass 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 for other global seagrass regions and other habitats where similar historical data is available.

2. Methods

2.1. Study area

Australia's Great Barrier Reef is one of the world's most extensive coral reef structures. The Great Barrier Reef was proclaimed a Marine Park (GBRMP) by the Australian Federal government in 1975 (*Great Barrier Reef Marine Park Act 1975*) and inscribed as a World Heritage Area (GBRWHA) in 1981 in recognition of the reef's globally outstanding and biodiverse marine ecosystem (Fig. 1). The GBRWHA includes 2500 km of coastline, more than 2500 individual reefs, and over 900 islands that protect an extensive shallow inter-reef lagoon. 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 includes the adjacent estuaries along the mainland Australian coast (Fig. 1).

2.2. Seagrass communities

The GBRWHA contains large and diverse seagrass meadows that extend from tropical to sub-tropical waters, with recent community analysis identifying 36 distinct seagrass community types (Table 1; Carter et al., 2021b). 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 >0.2; see Carter et al., 2021b), using site data that excluded a period of significant environmental impact (2009–2012) in the southern two-thirds of the GBRWHA, and restricted to data collected during the seagrass growing season (August–January). Within that, intertidal and subtidal community types were defined for three water bodies (estuary, coastal, and reef) (Fig. 1). Sites were classed as intertidal if they fell within Bishop-Taylor et al.'s (2019) intertidal extent model, 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 occur 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* (abbreviated to *Z. capricorni*) (Carter et al., 2021a). The thirty-six 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.

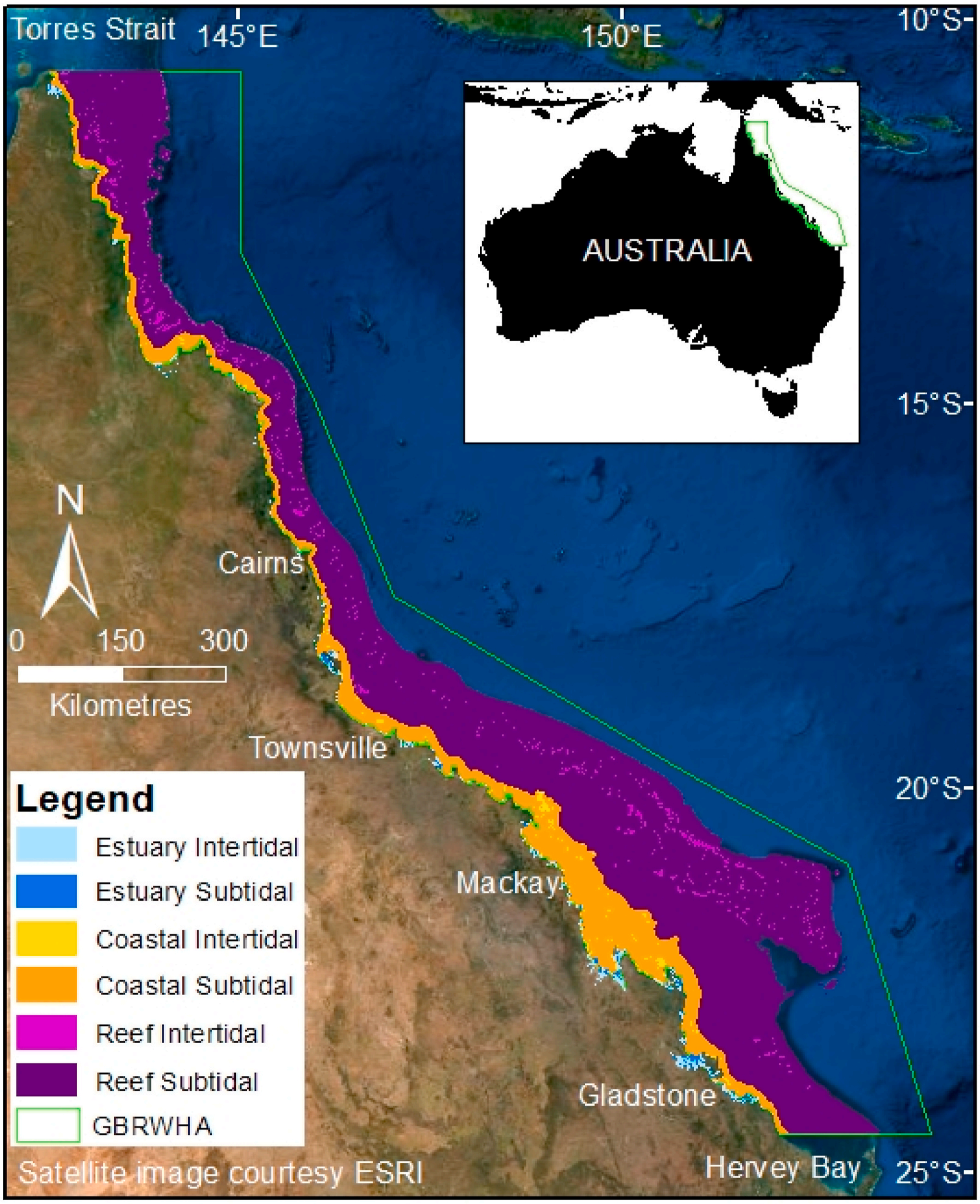

Fig. 1. Great Barrier Reef World Heritage Area (GBRWHA) and extent of six modelled areas: Estuary intertidal, estuary subtidal, coastal intertidal, coastal subtidal, reef intertidal, and reef subtidal.

- 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 (Fig. 2; Carter et al., 2021b).

2.3. Biomass data

Seagrass above-ground biomass was determined using visual estimates of above-ground biomass, a widely-used, non-destructive metric often applied to time-series analysis (Aragones and Marsh, 1999; Rasheed, 1999, 2004) and assessments of meadow-scale change (McKenna et al., 2015; Rasheed and Unsworth, 2011). 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 (Supplementary Table S1).

For each of these surveys, sites with an area of 5 m radius, were haphazardly allocated to ensure good spatial coverage. Above-ground biomass was assessed visually within three replicate quadrats (50 × 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} (g DW m^{-2}). Following each survey, the visual assessment is

Table 1
Seagrass communities in the Great Barrier Reef World Heritage Area and adjacent estuaries, including predicted area, geographic range, and characteristic species (listed in order from most to least frequent). Species abbreviations: CR, *Cymodocea rotundata*; CS, *Cymodocea serrulata;* HD, *Halophila decipiens*; HO, *Halophila ovalis*; HS, *Halophila spinulosa*; HT, *Halophila tricostata*; HU, *Halodule uninervis*; SI, *Syringodium isoetifolium*; TH, *Thalassia hemprichii*; ZC, *Zostera capricorni.* More detailed community description available at: https://doi.org/10.1038/s41598-021-01471-4 (Carter et al., 2021b).

Community	Predicted area (km^2)	Geographic range	Most common species
Estuary Intertidal 1	288	Northern to southern extent GBRWHA	ZC/HO
Estuary Intertidal 2	5	South of Bingil Bay to southern end Hinchinbrook Island	HU
Estuary Intertidal 3	77	Southern end Hinchinbrook Island to northern tip Curtis Island	ZC/HU/HO
Estuary Intertidal 4	3	Northern extent of GBRWHA to Bingil Bay	ZC/HO
Estuary Intertidal 5	7	Northern tip Curtis Island to southern extent GBRWHA	ZC/HO
Estuary Intertidal 6	4	South of Mourilyan Harbour to Townsville	HU
Estuary Intertidal 7	156	South of Townsville to Shoalwater Bay	ZC
Estuary Intertidal 8	5	Northern extent of GBRWHA to Mourilyan Harbour	ZC
Estuary Intertidal 9	39	South of Shoalwater to southern extent GBRWHA	ZC/HO
Estuary Subtidal 1	182	Northern to southern extent GBRWHA	HO/HD
Estuary Subtidal 2	96	Hinchinbrook Island to Gladstone	HD/HU/HO/ZC
Estuary Subtidal 3	122	Hinchinbrook Island to Gladstone	HO/ZC
Estuary Subtidal 4	36	Northern Hinchinbrook Island and the upper reaches of Trinity Inlet	HO/HD
Estuary Subtidal 5	38	Cairns to northern extent of GBRWHA	ZC/HO/HU/CS
Estuary Subtidal 6	16	Central and northern Hinchinbrook Island	HO
Coastal Intertidal 1	141	Whitsunday Islands to southern extent GBRWHA	ZC/HO
Coastal Intertidal 2/3	296	Northern to southern extent GBRWHA	ZC/HU/HO
Coastal Intertidal 4	178	Northern to southern extent GBRWHA	TH/HO/HU
Coastal Intertidal 5	39	Townsville to southern extent GBRWHA	HO/HU/ZC
Coastal Intertidal 6	154	Whitsunday Islands to southern extent GBRWHA	HU/HO/CS
Coastal Subtidal 1	7589	Northern to southern extent GBRWHA	HD
Coastal Subtidal 2	4575	Northern to southern extent GBRWHA	HU/HO/HS/HD
Coastal Subtidal 3	68	Northern to southern extent GBRWHA	HO/ZC/HU
Coastal Subtidal 4	161	Northern to southern extent GBRWHA	HU
Coastal Subtidal 5	2938	Northern extent GBRWHA to Whitsunday Islands	HU/HO/HS/CS
Coastal Subtidal 6	62	Northern to southern extent GBRWHA	ZC/HU/HO/CS
Coastal Subtidal 7	75	Northern to southern extent GBRWHA	HU/HO/CS
Reef Intertidal 1	318	Northern to southern extent GBRWHA	TH/HO
Reef Intertidal 2	887	Northern to southern extent GBRWHA	TH
Reef Intertidal 3	608	Northern to southern extent GBRWHA	TH/CR
Reef Intertidal 4/5	10	Clusters of reefs in Cairns and Princess Charlotte Bay regions	HU/TH/CR

Table 1 (*continued*)

Community	Predicted area (km^2)	Geographic range	Most common species
Reef Subtidal 1	19,434	Northern extent GBRWHA to Princess Charlotte Bay; Bloomfield to Palm Island Group	HD/HS/HT/HO
Reef Subtidal 2	49,052	Princess Charlotte Bay to Bloomfield; Palm Island Group to southern extent GBRWHA	HS/HD/HO
Reef Subtidal 3	623	Northern to southern extent GBRWHA	HU/CR/HO/SI/CS

calibrated for each individual observer against harvested biomass samples (Mellors, 1991).

We used the community models developed in Carter et al. (2021b) to predict community type using seagrass species presence/absence data for each survey site (Fig. 3). 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. Communities were combined where they had very similar biomass and species or one was represented by only a very small area. This resulted in reef intertidal communities RI4 and RI5 being combined 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 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 temporal analysis due to the increased sample size.

2.4. 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) using the *mgcv* package (Wood, 2014, 2017). All statistical analyses were conducted using R version 4.0.2 (R Core Team, 2020). The only covariate included in the model was year (categorical covariate) as the available environmental covariates were previously used to determine seagrass communities, and we were interested only in identifying years of maximum biomass. 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 data due to low certainty in these estimates. All plots were created using the *ggplot* package (Wickham, 2016).

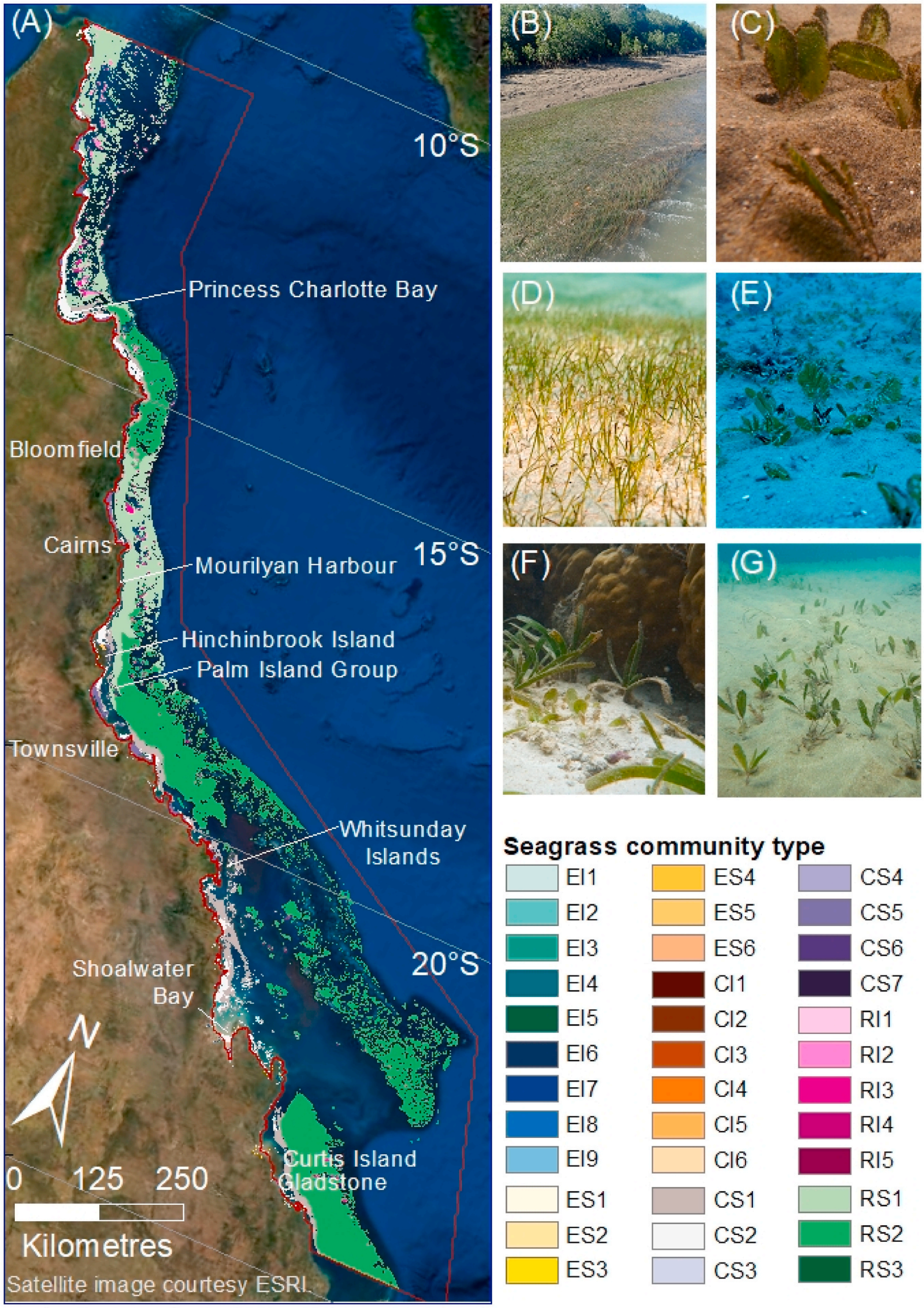

Fig. 2. (A) Distribution of thirty-six seagrass communities classified within potential seagrass habitat (probability of seagrass >0.2) for the Great Barrier Reef World Heritage Area. Estuary intertidal (EI1-EI9), estuary subtidal (ES1-ES6), coastal intertidal (CI1–CI6), coastal subtidal (CS1-CS7), reef intertidal (RI1-RI5), and reef subtidal (RS1-RS3). Common dominant species in the six seagrass habitats include: (B) estuary intertidal *Z. capricorni*, (C) estuary subtidal *H. ovalis*, (D) coastal intertidal *H. uninervis*, (E) coastal subtidal *H. ovalis* and *H. spinulosa*, (F) reef intertidal *T. hemprichii* and *H. ovalis*, and (G) reef subtidal *H. decipiens*. GBRWHA boundary in red. Photos courtesy D. Tracey and TropWATER JCU staff. The seagrass communities model is available as an interactive map at: https://doi.org/10.26274/NRE6-YS16. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

2.5. Reporting against biomass desired state

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

- *Desired state is met* with a high level of confidence if the mean biomass exceeds desired state and its upper CI.
- *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.

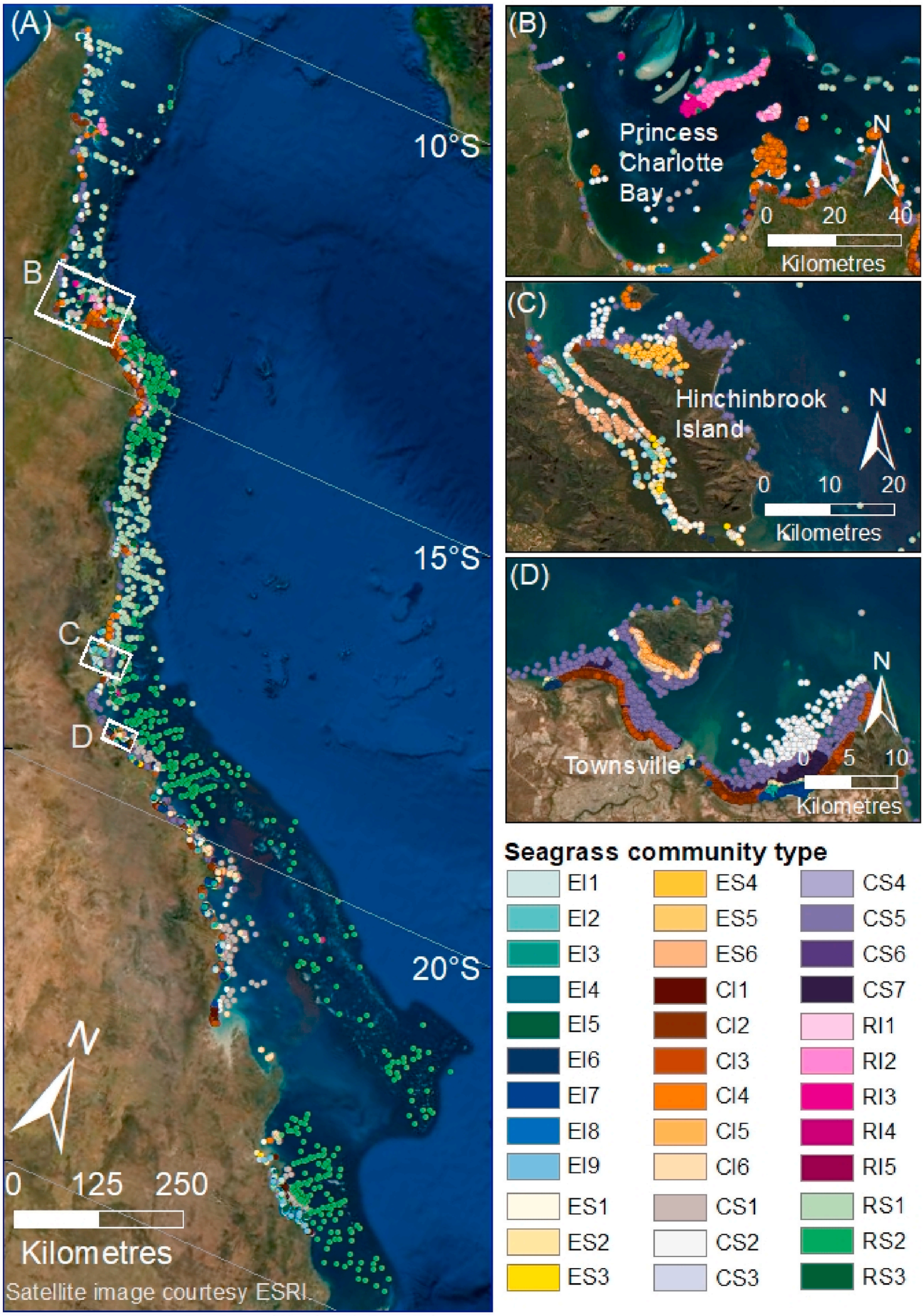

Fig. 3. (A) Seagrass site data used to define biomass desired state for thirty-six seagrass communities on the Great Barrier Reef: estuary intertidal (EI1-EI9), estuary subtidal (ES1-ES6), coastal intertidal (CI1–CI6), coastal subtidal (CS1-CS7), reef intertidal (RI1-RI5), and reef subtidal (RS1-RS3). (B–D) More detailed maps demonstrate the complex mix of different communities within relatively small areas (Carter et al., 2021b).

- *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.
- *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.

3. Results

Based on the available data we were able to establish desired state for 11 of the 15 estuary seagrass communities, all 12 of the coastal seagrass communities, and 2 out of 7 reef seagrass communities (Table 2). 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 of the mean biomass, 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 between different communities (Table 2).

Biomass reductions below desired state occurred in all estuary 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, 2020). Recovery to desired state has not occurred in most of those communities; some recovered and attained desired state in 2017 and 2018 and most displayed increasing trends in recent years. Desired state and trends in biomass are described in further detail below according to habitat type.

3.1. *Estuary seagrass communities*

The desired state of estuary seagrass community biomass was variable among communities. Within a community, the confidence intervals were narrow and often had a number of years (3–10 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. capricorni* was overwhelmingly the dominant species (EI7 and EI8), compared with communities where *Z. capricorni* was still dominant but lower biomass species such as *H. uninervis* and *H. ovalis* also frequently occurred (EI9 and EI4; Fig. 4; Tables 1 and 2). The most extensive estuary 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 (Fig. 2; Table 1). The highest biomass community EI7 was the second largest community, predicted to cover a total 156 km^2 between Townsville and Shoalwater Bay (Figs. 2 and 5; Tables 1 and 2). There was insufficient data to identify desired state for communities EI2, EI3, and EI6.

In estuary subtidal communities, desired state biomass for community ES5 was the greatest of any estuary 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 (Fig. 4; Table 2). Like many estuary communities ES5 was dominated by *Z. capricorni*, but with relatively higher frequencies of *H. uninervis* and the high biomass species *C. serrulata* compared with other subtidal communities (Table 1). Desired state was considerably lower in the remaining estuary subtidal communities dominated by the low biomass *Halophila* species (mean biomass <5 gDW m^{-2}; Fig. 4; Tables 1 and 2). The most extensive community was the *H. ovalis* and *H. decipiens* dominated community ES1, which was predicted to cover ~182 km^2 of estuary waters deeper than 2.9 m between the northern and southern extent of the GBRWHA (Fig. 2; Table 1). There was insufficient data to identify desired state for community ES6.

Three estuary intertidal communities and four subtidal communities had data that extended back to the mid-1990s (Fig. 4). Of these, there was 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

Table 2
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
		357	n/a	n/a	n/a

(*continued on next page*)

Table 2 (*continued*)

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
Reef Intertidal 3	<5 years of adequate data				
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

biomass remained very low until 2015. Despite small increases in biomass in all estuary 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 (Fig. 4).

3.2. Coastal seagrass communities

Desired state biomass was similar for intertidal coastal seagrass communities CI1, CI4, and CI5 (mean biomass range 7.8–9.6 gDW m^{-2}; Fig. 5; Table 2) despite considerable variation in their dominant species and distribution (Fig. 2; Table 1). Desired state biomass was much higher for communities CI2/3 and CI6 (mean biomass 21.5–22.1 gDW m^{-2}; Fig. 5; Table 2). 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. capricorni* and *H. ovalis* (Fig. 2; Table 1).

Coastal subtidal communities were spatially dominated by three large, adjacent communities in waters deeper than 1.6 m MSL (Carter et al., 2021b): 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 (Fig. 2; Table 1). Desired state biomass was the same for the two shallow subtidal communities CS2 and CS5 (mean biomass 10.7–11.3 gDW m^{-2}; Fig. 5; Table 2), likely due to the similar species mix in these communities that included *H. uninervis*, *H. ovalis*, *H. spinulosa*, *H. decipiens*, *C. serrulata* (Table 1). Desired state biomass in the deep subtidal community CS1 was much lower (desired state range:

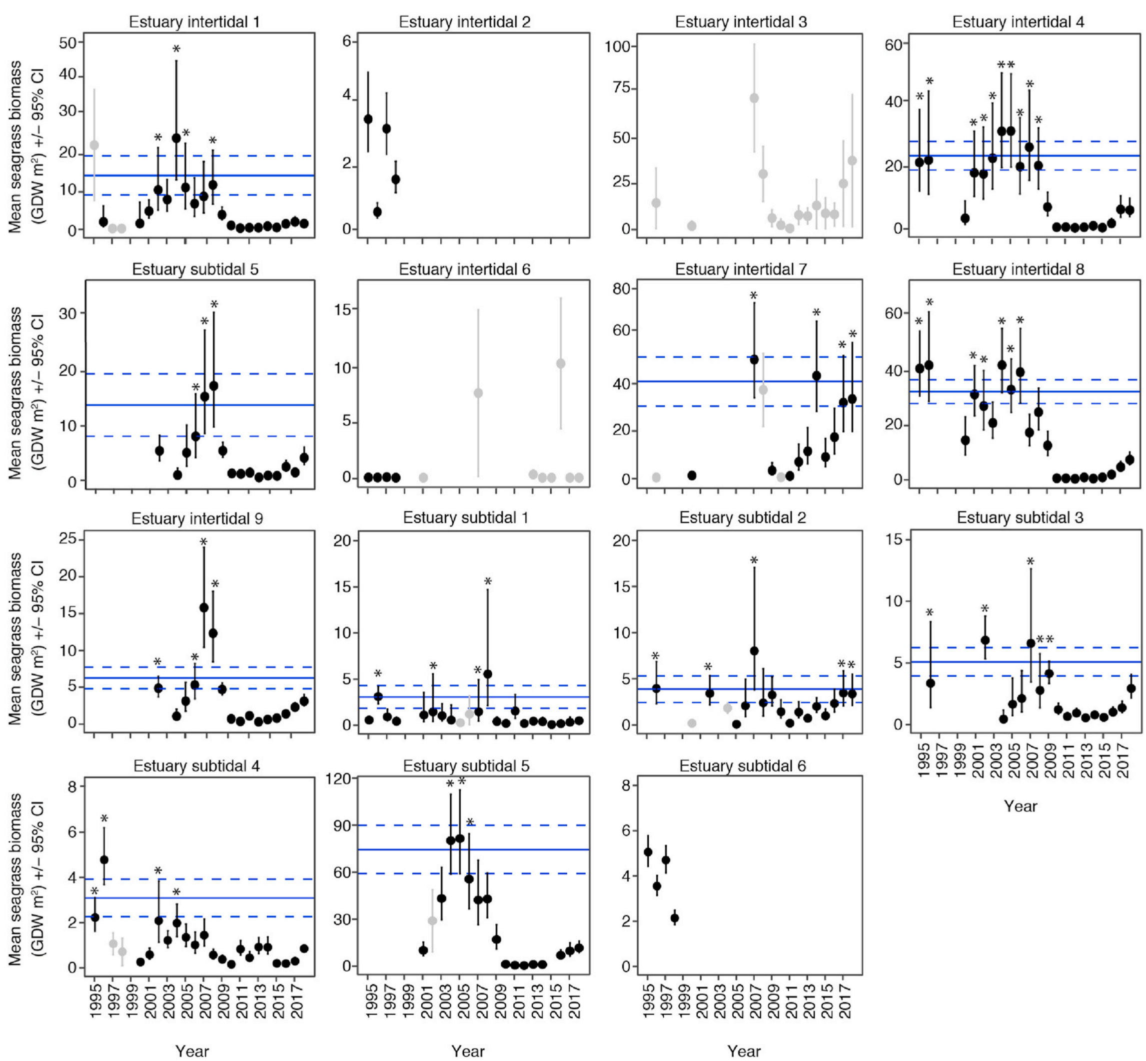

Fig. 4. Annual mean above-ground biomass (gDW m^{-2} ±95% CI) for estuary intertidal and subtidal seagrass communities, 1995–2018. Seagrass above-ground biomass desired state (solid blue line) with upper and lower 99% CIs (dashed blue lines). Asterisks indicate reference years used to set desired state. 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. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

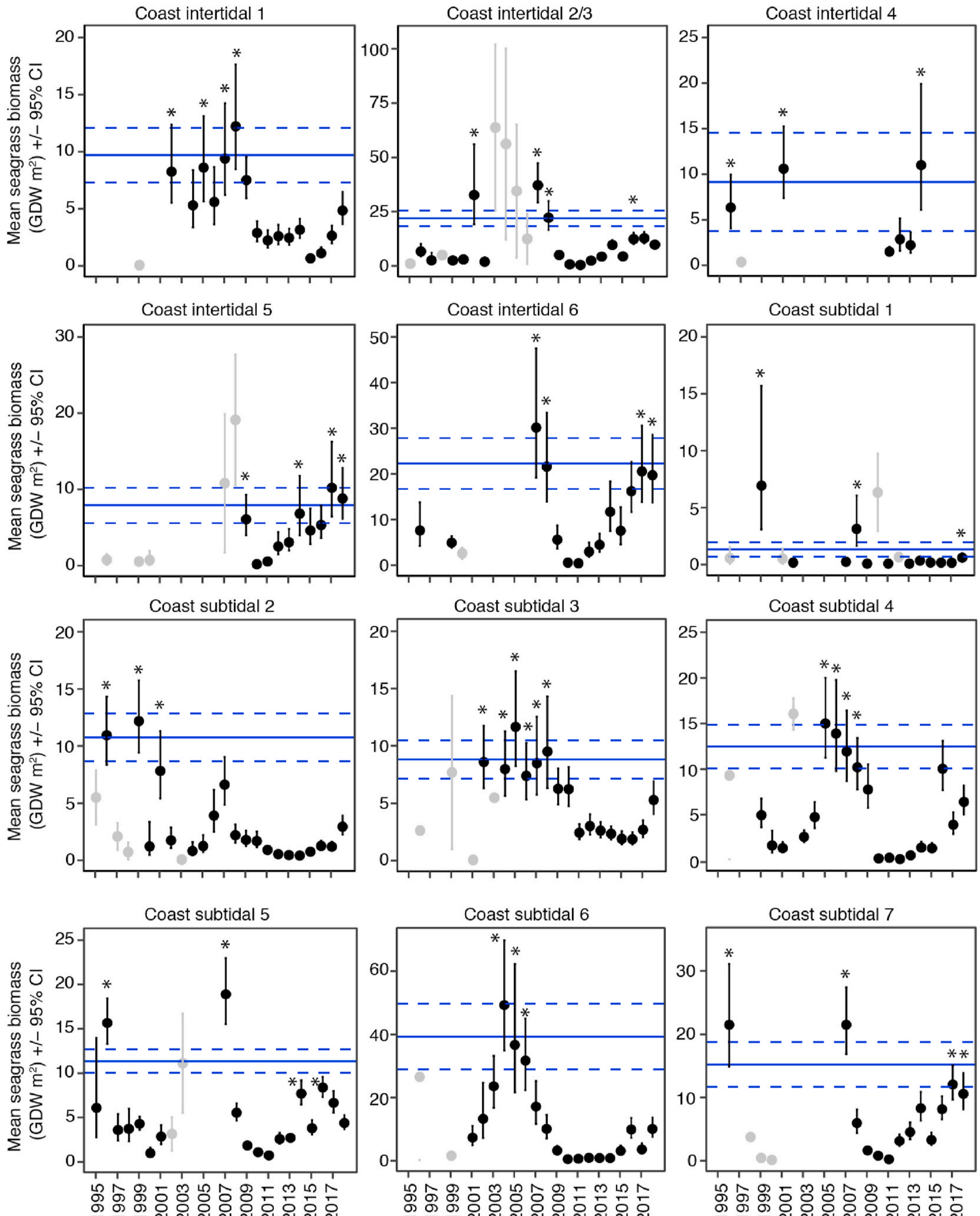

Fig. 5. Annual mean above-ground biomass (±95% CI) for coastal intertidal and subtidal seagrass communities, 1995–2018. Seagrass above-ground biomass desired state (solid blue line) with upper and lower 99% CIs (dashed blue lines). Asterisks indicate reference years used to set desired state. Years with values in grey were not included in desired state analyses due to low sample size (n < 15) in that community. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

0.6–1.9 gDW m^{-2}; Fig. 5; Table 2) due to the dominance of the low-biomass species *H. decipiens* (Table 1). 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; Fig. 2; Table 1). Desired state biomass varied greatly among these communities, from 7.1 to 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. capricorni* dominated CS6 (Fig. 5; Table 2).

Coastal communities experienced a period of peak biomass and years where desired state was often met over a much larger time frame than for estuary communities — between 2001 and 2009 for intertidal communities and 2001 and 2008 for subtidal communities (Fig. 5). 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 estuary 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 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 estuary 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 (Fig. 5).

3.3. Reef seagrass communities

Desired state estimates were limited to two reef seagrass 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 estuary and coastal communities (mean biomass <5 gDW m^{-2}; Fig. 6; Table 2). Communities RI1 and RI2 were found throughout the GBRWHA and dominated by the common reef-top species *T. hemprichii* (Table 1; Fig. 2). 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 1; Fig. 2). 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) (Fig. 2; Table 1). The limited data for these deep reef communities indicates much lower biomass than RS3, with mean annual biomass ranging from 0 to 8 gDW m^{-2} in the *H. decipiens* dominated community RS1, and 0.1–20 gDW m^{-2} in the mixed species community RS2 (Fig. 6; Tables 1 and 2).

The limited data available for reef communities indicates annual biomass is relatively stable (Fig. 6). However, no biomass data were collected for any reef communities between 2004 and 2010 (inclusive), so we are unsure if the trends seen for many estuary 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 estuary or coastal communities did. This indicates the dramatic biomass declines estuary 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 (Fig. 6).

4. 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 vary among seagrass communities

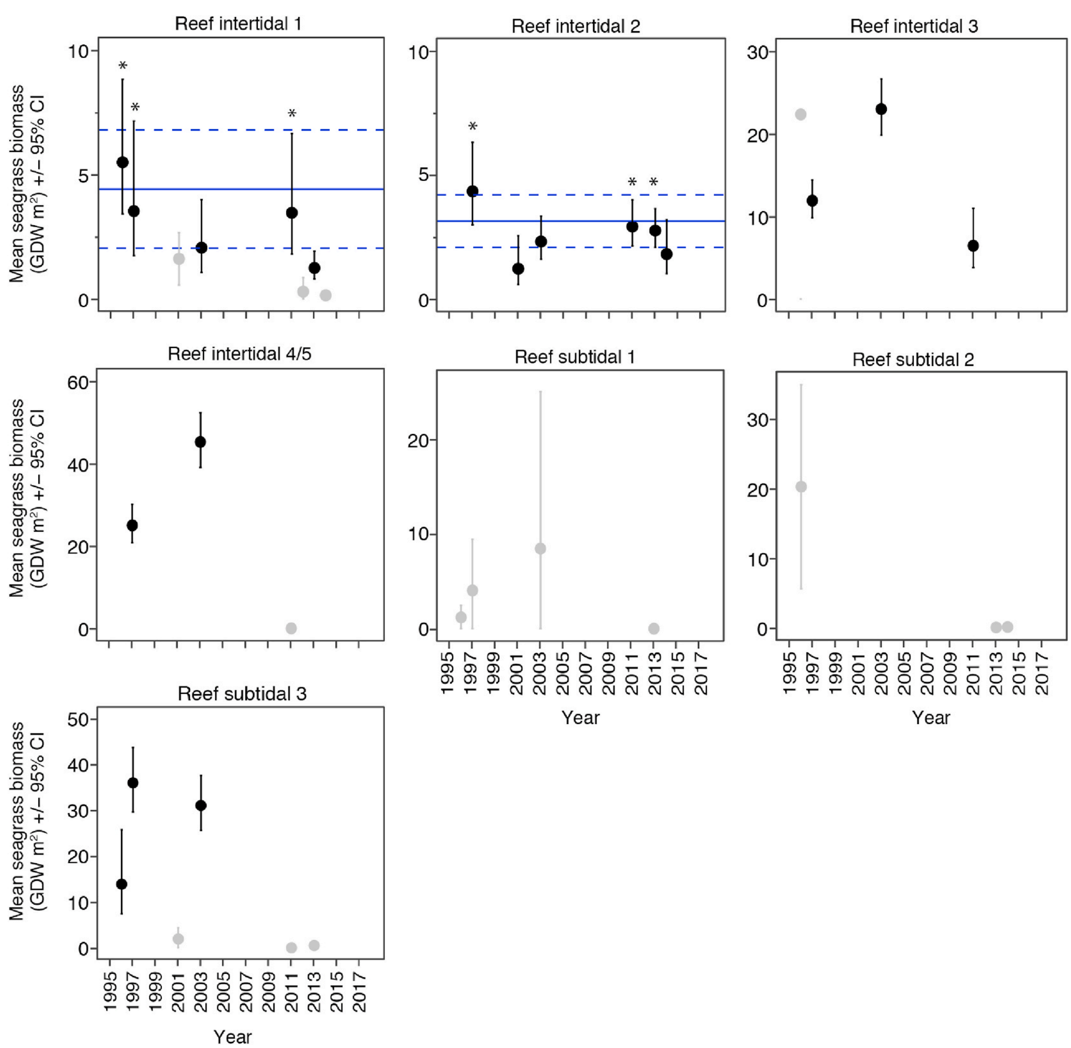

Fig. 6. Annual mean above-ground biomass (±95% CI) for reef intertidal and subtidal seagrass communities, 1995–2018. Seagrass above-ground biomass desired state (solid blue line) with upper and lower 99% CIs (dashed blue lines). Asterisks indicate reference years used to set desired state. Years with 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. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

by over an order of magnitude when expressed as biomass, because of the differences in species and the environmental setting for each community. Trajectories and trends vary among seagrass communities, as does contemporary biomass relative to desired state. Most estuary and coastal communities have not reached desired state in recent years. We demonstrate how targets can be established 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 assessments of seagrass desired state and when identifying critical information gaps.

4.1. 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 such as industrial and port development, coastal urbanisation and infrastructure, land clearing and climate change have been described, and guide management advice and plans (Grech et al., 2012; Griffiths et al., 2020; Tulloch et al., 2020; Turschwell et al., 2021). However, understanding 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). While our analysis does not resolve and quantify pressures, desired state can be used as a justification to do so. For example, the influence of rivers adjacent to Cleveland Bay in the central GBRWHA were correlated to seagrass biomass condition indicators, and sediment load targets to meet desired state have been identified (Lambert et al., 2021).

4.2. Considerations and limitations

Over-arching frameworks can be applied when assigning targets (e.g. Samhouri et al., 2012); however, there are always unique circumstances and challenges when they are applied to 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. There are historic and geographical biases in data in spatially large assessments that influence trends (Dunic et al., 2021). Our approach makes it possible to convey these trends in a tangible manner in a large, dynamic and diverse region. Our approach can be 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. Our approach can also be revised as data becomes available, and our assessment identifies the critical information gaps that need to be filled.

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, incorporate the desired state of additional 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; 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.

4.3. Seagrass communities of the GBRWHA

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 currently data deficient. These communities are also dominated by 'opportunistic' species, which are sensitive to, and rapidly decline when under pressure (Kilminster et al., 2015; Turschwell et al., 2021).

Where possible, we set a desired state biomass target for each community that 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 "facilitates adaptive management for the Reef that is effective, efficient and evolving" is met (Commonwealth of Australia, 2018). 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 to sample biomass across each community's spatial extent.

4.4. Reef seagrass communities

There were gaps in the historical data for most reef seagrass communities that limited our ability to determine desired state with any level of confidence. 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 (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 seagrass 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 because their extensive distribution, ephemeral nature, and remote location have presented challenges for existing monitoring programs.

4.5. Coastal seagrass communities

The diversity of coastal seagrass 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., 2011, 2012; 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 (Carter et al., 2021b).

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 recover to desired state following declines starting around 2008. 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 and 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, 2020). 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 (CI5 and CI6) recovering to desired state in recent years located in open coastal areas. Subtidal communities in coastal waters have not recovered. These inshore regions of the GBRWHA are turbid and there are legacy effects of river loads (Fabricius et al., 2014, 2016; Margvelashvili et al., 2018) making some communities especially vulnerable to water quality decline. In some circumstances shifts in seagrass species and communities may take long periods to recover (Birch and Birch, 1984).

4.6. *Estuary seagrass 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 (Carter et al., 2021b), therefore the biomass data for each of these communities tends to come from the same spatially constrained (latitudinal) areas in each sampling event. A better spread of data would improve our analysis at the GBRWHA wide scale. Lack of recovery to desired state in these communities runs in 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 (Carter et al., 2021b). Therefore, there is more uncertainty in how pressures influence the spatial distribution of estuary communities and changes in them over time relative to coastal and reef seagrass communities.

5. Conclusions

The GBRWHA and adjacent estuaries support diverse seagrass community assemblages, with wide-ranging desired states when measured as biomass. Individual community assessments are necessary when assessing seagrass condition and trends across the region. A number of the coastal subtidal and estuary 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 represents a massive step forward in understanding the complexities inherent 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. 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 deteriorating catchments, it is more important than ever to identify and work towards an attainable desired state for seagrass communities.

Credit author statement

Alex Carter: Conceptualization; Data curation; Formal analysis; Investigation; Methodology; Validation; Visualization; Writing – original draft, review & editing. Rob Coles: Investigation; Writing – review & editing. Catherine Collier: Conceptualization; Funding acquisition; Investigation; Methodology; Project administration; Supervision; Visualization; Writing – original draft, review & editing. Emma Lawrence: Formal analysis; Funding acquisition; Investigation; Methodology; Writing – review & editing. Michael Rasheed: Investigation; Writing – review & editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jenvman.2022.115059.

References

Aragones, L., Marsh, H., 1999. Impact of Dugong grazing and turtle cropping on tropical seagrass communities. Pac. Conserv. Biol. 5, 277–288. https://doi.org/10.1071/PC000277.

Bainbridge, Z., Lewis, S., Bartley, R., Fabricius, K., Collier, C., Waterhouse, J., Garzon-Garcia, A., Robson, B., Burton, J., Wenger, A., Brodie, J., 2018. Fine sediment and particulate organic matter: a review and case study on ridge-to-reef transport, transformations, fates, and impacts on marine ecosystems. Mar. Pollut. Bull. 135, 1205–1220. https://doi.org/10.1016/j.marpolbul.2018.08.002.

Barbier, E.B., 2017. Marine ecosystem services. Curr. Biol. 27, R431–R510. https://doi.org/10.1016/j.cub.2017.03.020.

Birch, W., Birch, M., 1984. Succession and pattern of tropical intertidal seagrasses in Cockle Bay, Queensland, Australia: a decade of observations. Aquat. Bot. 19, 343–367. https://doi.org/10.1016/0304-3770(84)90048-2.

Bishop-Taylor, R., Sagar, S., Lymburner, L., Beaman, R., 2019. Between the tides: modelling the elevation of Australia's exposed intertidal zone at continental scale. Estuarine. Coast Shelf Sci. 223, 115–128. https://doi.org/10.1016/j.ecss.2019.03.006.

Borja, A., Elliott, M., Andersen, J.H., Cardoso, A.C., Carstensen, J., Ferreira, J.G., Heiskanen, A.-S., Marques, J.C., Neto, J.M., Teixeira, H., 2013. Good environmental status of marine ecosystems: what is it and how do we know when we have attained it? Mar. Pollut. Bull. 76, 16–27. https://doi.org/10.1016/j.marpolbul.2013.08.042.

Bureau of Meteorology, 2020. La Niña – Detailed Australian Analysis (Australia).

Carmen, B., Krause-Jensen, D., Alcoverro, T., Marbà, N., Duarte, C.M., Van Katwijk, M. M., Pérez, M., Romero, J., Sánchez-Lizaso, J.L., Roca, G., 2019. Recent trend reversal for declining European seagrass meadows. Nat. Commun. 10, 1–8. https://doi.org/10.1038/s41467-019-11340-4.

Carter, A., McKenna, S., Rasheed, M., Collier, C., McKenzie, L., Pitcher, R., Coles, R., 2021a. Synthesizing 35 years of seagrass spatial data from the Great barrier reef world heritage area, Queensland, Australia. Limnol. Oceanogr. Lett. 1–11. https://doi.org/10.1002/lol2.10193.

Carter, A.B., Collier, C., Lawrence, E., Rasheed, M.A., Robson, B., Coles, R., 2021b. A spatial analysis of seagrass habitat and community diversity in the Great Barrier Reef World Heritage Area. Sci. Rep. 11 https://doi.org/10.1038/s41598-021-01471-4.

Coles, R., McKenzie, L., De'ath, G., Roelofs, A., Long, W.L., 2009. Spatial distribution of deepwater seagrass in the inter-reef lagoon of the Great barrier reef world heritage area. Mar. Ecol. Prog. Ser. 392, 57–68. https://doi.org/10.3354/meps08197.

Coles, R.G., Lee Long, W.J., Watson, R.A., Derbyshire, K.J., 1993. Distribution of seagrasses, and their fish and penaeid prawn communities, in Cairns Harbour, a tropical estuary, Northern Queensland, Australia. Mar. Freshw. Res. 44, 193–210. https://doi.org/10.1071/MF9930193.

Coles, R.G., Rasheed, M.A., McKenzie, L.J., Grech, A., York, P.H., Sheaves, M., McKenna, S., Bryant, C., 2015. The Great barrier reef world heritage area seagrasses: managing this iconic Australian ecosystem resource for the future. Estuarine. Coast Shelf Sci. 153, A1–A12. https://doi.org/10.1016/j.ecss.2014.07.020.

Collier, C.J., Carter, A.B., Rasheed, M.A., McKenzie, L., Udy, J., Coles, R., Brodie, J., Waycott, M., O'Brien, K.R., Saunders, M., Adams, M., Martin, K., Honchin, C., Petus, C., Lawrence, E., 2020. An evidence-based approach for setting desired state in a complex Great Barrier Reef seagrass ecosystem: a case study from Cleveland Bay. Environ. Sustain. Indicat. 7, 100042. https://doi.org/10.1016/j.indic.2020.100042.

Collier, C.J., Waycott, M., McKenzie, L.J., 2012. Light thresholds derived from seagrass loss in the coastal zone of the northern Great Barrier Reef, Australia. Ecol. Indicat. 23, 211–219. https://doi.org/10.1016/j.ecolind.2012.04.005.

Commonwealth of Australia, 2018. Reef 2050 Long-Term Sustainability Plan - July 2018.

Costanza, R., de Groot, R., Sutton, P., Van der Ploeg, S., Anderson, S.J., Kubiszewski, I., Farber, S., Turner, K., 2014. Changes in the global value of ecosystem services. Global Environ. Change 26, 152–158. https://doi.org/10.1016/j.gloenvcha.2014.04.002.

Creed, J.C., Amado Filho, G.M., 1999. Disturbance and recovery of the macroflora of a seagrass (*Halodule wrightii* Ascherson) meadow in the Abrolhos Marine National Park, Brazil: an experimental evaluation of anchor damage. J. Exp. Mar. Biol. Ecol. 235, 285–306. https://10.1016/S0022-0981(98)00188-9.

de los Santos, C.B., Scott, A., Arias-Ortiz, A., Jones, B., Kennedy, H., Mazarrasa, I., McKenzie, L., Nordlund, L.M., de la Torre-Castro, M.d.l.T., Unsworth, R.K., 2020. Seagrass Ecosystem Services: Assessment and Scale of Benefits. Out of the Blue: the Value of Seagrasses to the Environment and to People, pp. 19–21.

den Hartog, C., Kuo, J., 2006. Taxonomy and Biogeography of Seagrasses, Seagrasses: Biology, Ecology and Conservation. Springer, Netherlands, pp. 1–23.

Dunic, J.C., Brown, C.J., Connolly, R.M., Turschwell, M.P., Côté, I.M., 2021. Long-term declines and recovery of meadow area across the world's seagrass bioregions. Global Change Biol. 27, 4096–4109. https://doi.org/10.1111/gcb.15684.

Fabricius, K., Logan, M., Weeks, S., Brodie, J., 2014. The effects of river run-off on water clarity across the central Great Barrier Reef. Mar. Pollut. Bull. 84, 191–200. https://doi.org/10.1016/j.marpolbul.2014.05.012.

Fabricius, K., Logan, M., Weeks, S., Lewis, S., Brodie, J., 2016. Changes in water clarity in response to river discharges on the Great Barrier Reef continental shelf: 2002–2013. Estuarine. Coast Shelf Sci. 173, A1–A15. https://doi.org/10.1016/j.ecss.2016.03.001.

Fourqurean, J.W., Duarte, C.M., Kennedy, H., Marbà, N., Holmer, M., Mateo, M.A., Apostolaki, E.T., Kendrick, G.A., Krause-Jensen, D., McGlathery, K.J., 2012. Seagrass ecosystems as a globally significant carbon stock. Nat. Geosci. 5, 505–509. https://doi.org/10.1038/ngeo1477.

Grech, A., Chartrand-Miller, K., Erftemeijer, P., Fonseca, M., McKenzie, L., Rasheed, M., Taylor, H., Coles, R., 2012. A comparison of threats, vulnerabilities and management approaches in global seagrass bioregions. Environ. Res. Lett. 7, 024006 https://doi.org/10.1088/1748-9326/7/2/024006.

Grech, A., Coles, R., Marsh, H., 2011. A broad-scale assessment of the risk to coastal seagrasses from cumulative threats. Mar. Pol. 35, 560–567. https://doi.org/10.1016/j.marpol.2011.03.003.

Grech, A., Coles, R.G., 2010. An ecosystem-scale predictive model of coastal seagrass distribution. Aquat. Conserv. Mar. Freshw. Ecosyst. 20, 437–444. https://doi.org/10.1002/aqc.1107.

Green, E.P., Short, F.T., 2003. World Atlas of Seagrasses. University of California Press, United States of America.

Griffiths, L.L., Connolly, R.M., Brown, C.J., 2020. Critical gaps in seagrass protection reveal the need to address multiple pressures and cumulative impacts. Ocean Coast Manag. 183 https://doi.org/10.1016/j.ocecoaman.2019.104946.

Hallett, C.S., Valesini, F., Elliott, M., 2016a. A review of Australian approaches for monitoring, assessing and reporting estuarine condition: III. Evaluation against international best practice and recommendations for the future. Environ. Sci. Pol. 66, 282–291. https://doi.org/10.1016/j.envsci.2016.07.015.

Hallett, C.S., Valesini, F., Scanes, P., Crawford, C., Gillanders, B.M., Pope, A., Udy, J., Fortune, J., Townsend, S., Barton, J., 2016b. A review of Australian approaches for monitoring, assessing and reporting estuarine condition: II. State and Territory programs. Environ. Sci. Pol. 66, 270–281. https://doi.org/10.1016/j.envsci.2016.07.013.

Halpern, B.S., Frazier, M., Afflerbach, J., Lowndes, J.S., Micheli, F., O'Hara, C., Scarborough, C., Selkoe, K.A., 2019. Recent pace of change in human impact on the world's ocean. Sci. Rep. 9, 1–8. https://doi.org/10.1038/s41598-019-47201-9.

Head, B.W., 2014. Evidence, uncertainty, and wicked problems in climate change decision making in Australia. Environ. Plann. C Govern. Pol. 32, 663–679. https://doi.org/10.1068/c1240.

Hemminga, M.A., Duarte, C.M., 2000. Seagrass Ecology. Cambridge University Press, Cambridge, United Kingdom.

Jayathilake, D.R.M., Costello, M.J., 2018. A modelled global distribution of the seagrass biome. Biol. Conserv. 226, 120–126. https://doi.org/10.1016/j.biocon.2018.07.009.

Kilminster, K., McMahon, K., Waycott, M., Kendrick, G.A., Scanes, P., McKenzie, L., O'Brien, K.R., Lyons, M., Ferguson, A., Maxwell, P., Glasby, T., Udy, J., 2015. Unravelling complexity in seagrass systems for management: Australia as a microcosm. Sci. Total Environ. 534, 97–109. https://doi.org/10.1016/j.scitotenv.2015.04.061.

Kuhnert, P.M., Martin, T.G., Griffiths, S.P., 2010. A guide to eliciting and using expert knowledge in Bayesian ecological models. Ecol. Lett. 13, 900–914. https://doi.org/10.1111/j.1461-0248.2010.01477.x.

Lamb, J.B., van de Water, J.A.J.M., Bourne, D.G., Altier, C., Hein, M.Y., Fiorenza, E.A., Abu, N., Jompa, J., Harvell, C.D., 2017. Seagrass ecosystems reduce exposure to bacterial pathogens of humans, fishes, and invertebrates. Science 355, 731–733. https://doi.org/10.1126/science.aal1956.

Lambert, V.M., Bainbridge, Z.T., Collier, C., Lewis, S.E., Adams, M.P., Carter, A., Saunders, M.I., Brodie, J., Turner, R.D.R., Rasheed, M.A., O'Brien, K.R., 2021. Connecting targets for catchment sediment loads to ecological outcomes for seagrass using multiple lines of evidence. Mar. Pollut. Bull. 169 https://doi.org/10.1016/j.marpolbul.2021.112494.

Levin, P.S., Möllmann, C., 2015. Marine ecosystem regime shifts: challenges and opportunities for ecosystem-based management. Phil. Trans. Biol. Sci. 370, 20130275. https://doi.org/10.1098/rstb.2013.0275.

Marba, N., Diaz-Almela, E., Duarte, C., 2009. Mediterranean seagrass (*Posidonia oceanica*) loss between 1842 and 2009. Biol. Conserv. 176, 183–190. https://doi.org/10.1016/j.biocon.2014.05.024.

Marbà, N., Duarte, C.M., 2010. Mediterranean warming triggers seagrass (*Posidonia oceanica*) shoot mortality. Global Change Biol. 16, 2366–2375. https://doi.org/10.1111/j.1365-2486.2009.02130.x.

Marbà, N., Krause-Jensen, D., Alcoverro, T., Birk, S., Pedersen, A., Neto, J.M., Orfanidis, S., Garmendia, J.M., Muxika, I., Borja, A., 2013. Diversity of European seagrass indicators: patterns within and across regions. Hydrobiologia 704, 265–278. https://doi.org/10.1007/s10750-012-1403-7.

Margvelashvili, N., Andrewartha, J., Baird, M., Herzfeld, M., Jones, E., Mongin, M., Rizwi, F., Robson, B., Skerratt, J., Wild-Allen, K., 2018. Simulated fate of catchment-derived sediment on the Great Barrier Reef shelf. Mar. Pollut. Bull. 135, 954–962. https://doi.org/10.1016/j.marpolbul.2018.08.018.

Maxwell, P.S., Pitt, K.A., Burfeind, D.D., Olds, A.D., Babcock, R.C., Connolly, R.M., 2014. Phenotypic plasticity promotes persistence following severe events: physiological and morphological responses of seagrass to flooding. J. Ecol. 102, 54–64. https://doi.org/10.1111/1365-2745.12167.

McKenna, S., Chartrand, K., Van De Wetering, C., Wells, J., Carter, A., Rasheed, M., 2020. Port of Townsville Seagrass Monitoring Program: 2019, Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER). James Cook University, Cairns.

McKenna, S.A., Jarvis, J.C., Sankey, T., Reason, C., Coles, R., Rasheed, M.A., 2015. Declines of seagrasses in a tropical harbour, North Queensland, Australia, are not the result of a single event. J. Biosci. 40, 389–398. https://doi.org/10.1007/s12038-015-9516-6.

McKenzie, L.J., Collier, C.J., Langlois, L.A., Yoshida, R.L., Uusitalo, J., Smith, N., Waycott, M., 2020a. Marine Monitoring Program: Annual Report for Inshore Seagrass Monitoring 2018-2019. Report for the Great Barrier Reef Marine Park Authority. Great Barrier Reef Marine Park Authority (Townsville).

McKenzie, L.J., Nordlund, L.M., Jones, B.L., Cullen-Unsworth, L.C., Roelfsema, C., Unsworth, R.K.F., 2020b. The global distribution of seagrass meadows. Environ. Res. Lett. 15, 074041 https://doi.org/10.1088/1748-9326/ab7d06.

McMahon, K., Collier, C., Lavery, P.S., 2013. Identifying robust bioindicators of light stress in seagrasses: a meta-analysis. Ecol. Indicat. 30 https://doi.org/10.1016/j.ecolind.2013.01.030.

Mellors, J.E., 1991. An evaluation of a rapid visual technique for estimating seagrass biomass. Aquat. Bot. 42, 67–73. https://doi.org/10.1016/0304-3770(91)90106-F.

O'Brien, K.R., Waycott, M., Maxwell, P., Kendrick, G.A., Udy, J.W., Ferguson, A.J.P., Kilminster, K., Scanes, P., McKenzie, L.J., McMahon, K., Adams, M.P., Samper-Villarreal, J., Collier, C., Lyons, M., Mumby, P.J., Radke, L., Christianen, M.J.A., Dennison, W.C., 2017. Seagrass ecosystem trajectory depends on the relative timescales of resistance, recovery and disturbance. Mar. Pollut. Bull. 134, 166–176. https://doi.org/10.1016/j.marpolbul.2017.09.006.

Orth, R.J., Carruthers, T.J., Dennison, W.C., Duarte, C.M., Fourqurean, J.W., Heck Jr., K. L., Hughes, A.R., Kendrick, G.A., Kenworthy, W.J., Olyarnik, S., Short, F.T., Waycott, M., Williams, S., 2006. A global crisis for seagrass ecosystems. Bioscience 56, 987–996. https://doi.org/10.1641/0006-3568(2006)56[987. AGCFSE]2.0.CO;2.

Orth, R.J., Lefcheck, J.S., McGlathery, K.S., Aoki, L.R., Luckenbach, M.W., Moore, K.A., Oreska, M.P.J., Snyder, R., Wilcox, D.J., Lusk, B., 2020. Restoration of seagrass habitat leads to rapid recovery of coastal ecosystem services. Sci. Adv. 6 https://doi.org/10.1126/sciadv.abc6434.

Petus, C., Collier, C., Devlin, M., Rasheed, M., McKenna, S., 2014. Using MODIS data for understanding changes in seagrass meadow health: a case study in the Great Barrier Reef (Australia). Mar. Environ. Res. 98, 68–85. https://doi.org/10.1016/j.marenvres.2014.03.006.

Pittman, S.J., Kneib, R.T., Simenstad, C.A., 2011. Practicing coastal seascape ecology. Mar. Ecol. Prog. Ser. 427, 187–190. https://doi.org/10.3354/meps09139.

R Core Team, 2020. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria.

Rasheed, M.A., 1999. Recovery of experimentally created gaps within a tropical *Zostera capricorni* (Aschers.) seagrass meadow, Queensland Australia. J. Exp. Mar. Biol. Ecol. 235, 183–200. https://doi.org/10.1016/S0022-0981(98)00158-0.

Rasheed, M.A., 2004. Recovery and succession in a multi-species tropical seagrass meadow following experimental disturbance: the role of sexual and asexual reproduction. J. Exp. Mar. Biol. Ecol. 310, 13–45. https://doi.org/10.1016/j.jembe.2004.03.022.

Rasheed, M.A., McKenna, S.A., Carter, A.B., Coles, R.G., 2014. Contrasting recovery of shallow and deep water seagrass communities following climate associated losses in tropical north Queensland, Australia. Mar. Pollut. Bull. 83, 491–499. https://doi.org/10.1016/j.marpolbul.2014.02.013.

Rasheed, M.A., Unsworth, R.K.F., 2011. Long-term climate-associated dynamics of a tropical seagrass meadow: implications for the future. Mar. Ecol. Prog. Ser. 422, 93–103. https://doi.org/10.3354/meps08925.

Reason, C., McKenna, S., Rasheed, M., 2020. Seagrass Habitat of Cairns Harbour and Trinity Inlet: Cairns Shipping Development Program and Annual Monitoring Report 2019, Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER) Report No. 20/06. James Cook University, Cairns, p. 54.

Roca, G., Alcoverro, T., Krause-Jensen, D., Balsby, T.J., Van Katwijk, M., Marbà, N., Santos, R., Arthur, R., Mascaró, O., Fernández-Torquemada, Y., 2016. Response of seagrass indicators to shifts in environmental stressors: a global review and management synthesis. Ecol. Indicat. 63, 310–323. https://doi.org/10.1016/j.ecolind.2015.12.007.

Samhouri, J.F., Lester, S.E., Selig, E.R., Halpern, B.S., Fogarty, M.J., Longo, C., McLeod, K.L., 2012. Sea sick? Setting targets to assess ocean health and ecosystem services. Ecosphere 3, 1–18. https://doi.org/10.1890/ES11-00366.1.

Scott, A.L., York, P.H., Duncan, C., Macreadie, P.I., Connolly, R.M., Ellis, M.T., Jarvis, J. C., Jinks, K.I., Marsh, H., Rasheed, M.A., 2018. The role of herbivory in structuring tropical seagrass ecosystem service delivery. Front. Plant Sci. 9, 1–10. https://doi.org/10.3389/fpls.2018.00127.

Short, F., Wyllie-Echeverria, S., 1996. Natural and human-induced disturbance of seagrasses. Environ. Conserv. 23, 17–27. https://doi.org/10.1017/S0376892900038212.

Smith, T., Chartrand, K., Wells, J., Carter, A., Rasheed, M., 2020. Seagrasses in Port Curtis and Rodds Bay 2019 Annual Long-Term Monitoring and Whole Port Survey. Centre for Tropical Water & Aquatic Ecosystem Research. James Cook University, Cairns, p. 71 (TropWATER) Publication 20/02.

Strydom, S., Murray, K., Wilson, S., Huntley, B., Rule, M., Heithaus, M., Bessey, C., Kendrick, G.A., Burkholder, D., Fraser, M.W., Zdunic, K., 2020. Too hot to handle: unprecedented seagrass death driven by marine heatwave in a World Heritage Area. Global Change Biol. 26, 3525–3538. https://doi.org/10.1111/gcb.15065.

Thompson, A., Martin, K., Logan, M., 2020. Development of the coral index, a summary of coral reef resilience as a guide for management. J. Environ. Manag. 271, 111038. https://doi.org/10.1016/j.jenvman.2020.111038.

Thomson, J.A., Burkholder, D.A., Heithaus, M.R., Fourqurean, J.W., Fraser, M.W., Statton, J., Kendrick, G.A., 2015. Extreme temperatures, foundation species, and abrupt ecosystem change: an example from an iconic seagrass ecosystem. Global Change Biol. 21, 1463–1474. https://doi.org/10.1111/gcb.12694.

Tulloch, V.J., Turschwell, M.P., Giffin, A.L., Halpern, B.S., Connolly, R., Griffiths, L., Frazer, M., Brown, C.J., 2020. Linking threat maps with management to guide conservation investment. Biol. Conserv. 245, 108527. https://doi.org/10.1016/j.biocon.2020.108527.

Turschwell, M.P., Connolly, R.M., Dunic, J.C., Sievers, M., Buelow, C.A., Pearson, R.M., Tulloch, V.J., Côté, I.M., Unsworth, R.K., Collier, C.J., 2021. Anthropogenic pressures and life history predict trajectories of seagrass meadow extent at a global scale. Proc. Natl. Acad. Sci. Unit. States Am. 118 https://doi.org/10.1073/pnas.2110802118.

Tweedie, M.C., 1984. An Index Which Distinguishes between Some Important Exponential Families, Statistics: Applications and New Directions: Proc. Indian statistical institute golden Jubilee International conference, pp. 579–604.

Unsworth, R.K.F., Ambo-Rappe, R., Jones, B.L., La Nafie, Y.A., Irawan, A., Hernawan, U. E., Moore, A.M., Cullen-Unsworth, L.C., 2018. Indonesia's globally significant seagrass meadows are under widespread threat. Sci. Total Environ. 634, 279–286. https://doi.org/10.1016/j.scitotenv.2018.03.315.

Unsworth, R.K.F., McKenzie, L.J., Collier, C.J., Cullen-Unsworth, L.C., Duarte, C.M., Eklof, J.S., Jarvis, J.C., Jones, B.L., Nordlund, L.M., 2019. Global challenges for seagrass conservation. Ambio 48, 801–815. https://doi.org/10.1007/s13280-018-1115-y.

Van De Wetering, C., Carter, A., Rasheed, M., 2020a. Seagrass Habitat of Mourilyan Harbour: Annual Monitoring Report – 2019, Centre for Tropical Water & Aquatic Ecosystem Research. James Cook University, Cairns, p. 51. JCU Publication 20/09.

Van De Wetering, C., Reason, C., Rasheed, M., Wilkinson, J., York, P., 2020b. Port of Abbot Point Long-Term Seagrass Monitoring Program - 2019, Centre for Tropical Water & Aquatic Ecosystem Research JCU Publication. James Cook University, Cairns, p. 53, 20/12.

Virnstein, R.W., 1999. Seagrass management in Indian river lagoon, Florida: dealing with issues of scale. Pac. Conserv. Biol. 5, 299–305. https://doi.org/10.1071/PC000299.

Walker, B., Salt, D., 2012. Resilience Thinking: Sustaining Ecosystems and People in a Changing World. Island press.

Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A., Fourqurean, J.W., Heck, K.L., Randall Hughes, A., Kendrick, G.A., Kenworthy, W.J., Short, F.T., Williams, S.L., 2009. Accelerating loss of seagrass across the globe threatens coastal ecosystems. Proc. Natl. Acad. Sci. Unit. States Am. 106, 12377–12381.

Wickham, H., 2016. ggplot2: Elegant Graphics for Data Analysis. Springer.

Wood, S., 2014. Package 'mgcv' version 1.8-3.

Wood, S.N., 2017. Generalized Additive Models: an Introduction with R, second ed. Chapman and Hall/CRC.

York, P., Rasheed, M., 2020. Annual Seagrass Monitoring in the Mackay-Hay Point Region – 2019. JCU Centre for Tropical Water & Aquatic Ecosystem Research Publication, James Cook University, Cairns, p. 51.

York, P.H., Carter, A.B., Chartrand, K., Sankey, T., Wells, L., Rasheed, M.A., 2015. Dynamics of a deep-water seagrass population on the Great Barrier Reef: annual occurrence and response to a major dredging program. Sci. Rep. 5, 13167. https://doi.org/10.1038/srep13167.