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Integrating outcomes of IUCN red list of ecosystems assessments for connected coastal wetlands

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ABSTRACT

Human-induced habitat clearing and pollution are leading drivers of biodiversity loss. Ecosystem assessments are required to identify ecosystems at risk of collapse, but they should account for cross-system linkages and dynamics where necessary. This is particularly true for coastal wetlands (e.g. seagrass, mangroves and saltmarsh), which exhibit high ecological connectivity and have individually suffered global declines over the last century. We use the coastal wetlands of Moreton Bay, Queensland, Australia, as a model system to examine how integrating outcomes of multiple, simultaneously conducted, ecosystem assessments can assist in identifying appropriate management and conservation strategies. We simultaneously conducted separate assessments of seagrass, mangrove and saltmarsh ecosystems against the IUCN Red List of Ecosystems criteria. Despite substantial human population growth in the region, seagrass and mangroves were assessed as *Least Concern*. Mangroves were found to be rapidly encroaching on saltmarsh. This process, together with past clearing, were the major drivers behind saltmarsh being assessed as *Endangered*. Given the importance of connectivity among these connected ecosystems, collapse in any one ecosystem can have seascape-wide consequences, highlighting the benefit of conducting multi-ecosystem assessments. Consequently, a fully integrated assessment of the coastal wetlands as a single entity would miss key processes, such as mangrove encroachment, potentially underestimating overall risks. Our study highlights the plight of saltmarsh and the value of conducting simultaneous RLE assessments for multiple ecosystems comprising seascapes. We recommend that connectivity be accounted for explicitly in assessments of other connected, high-risk ecosystems.

1. Introduction

Habitat degradation is a global crisis that reduces the many benefits humans derive from ecosystems. Loss of habitat via direct human interference and reductions in habitat quality is one of the leading causes of species and population declines worldwide (Maxwell et al. 2016). While habitat loss causes immediate and evident conservation problems, the effects of habitat degradation arise more subtly, making them more difficult to detect and quantify (Lönnstedt et al. 2014). Habitat change has diverse and significant impacts to ecosystems (Sievers et al. 2018) and the services they provide (Costanza et al. 2014). Quantifying and evaluating habitat change and its impacts on ecosystem function is thus complex, yet vital for informing management and conservation decisions.

Evaluating habitat loss and degradation ultimately requires repeatable and standardised techniques. Efforts like the Ocean Health Index (Halpern et al. 2012) and the Wetland Extent Trends Index (Dixon et al. 2016) provide some indication of the health status of coastal wetlands, although with a primary focus on ecosystem services and changes in extent, respectively. The Red List of Ecosystems (RLE) developed by the International Union for Conservation of Nature (IUCN) was designed to account for a range of ecologically relevant

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aspects of ecosystem condition (Keith et al. 2013). Analogous to the IUCN Red List of Threatened Species™, which categorises species extinction risk, the RLE provides a framework to assess the risk of collapse of ecosystems. The RLE methodology has been rigorously tested both theoretically (Keith et al. 2013, Murray et al. 2017) and empirically (Bland et al. 2017b, Ferrer-Paris et al. 2019) and has been applied to > 2,500 ecosystems across 100 countries (Bland et al. 2019). As application of the RLE gains momentum, so are the practical conservation and management outcomes resulting from these assessments (Bland et al. 2019).

Previous RLE assessments have typically assessed single ecosystems in isolation (e.g. Bland et al. 2017b, Marshall et al. 2018), but there can be benefits in integrating the outcomes of assessments conducted for multiple, connected ecosystems. Connected ecosystems exchange resources (e.g. sediment, nutrient and organic matter) and provide food and habitat to species that may migrate among ecosystems (Sheaves 2009, Boström et al. 2011). Further, the habitat-forming species that define connected ecosystems can even be direct competitors for space and resources (Cavanaugh et al. 2019).

These concepts are particularly relevant for many aquatic ecosystems, which are often connected, performing functions and providing services that rely on the movement of organisms, matter and energy among connected ecosystems (Hyndes et al. 2014, Olds et al. 2016). For example, vegetated coastal wetlands (e.g. mangroves, saltmarsh and seagrass) exist alongside one another in habitat mosaics, often forming communities that are highly connected through biotic and abiotic processes (McKee & Rooth 2008, Sheaves 2009, Olds et al. 2016). Changes in the status of one coastal wetland ecosystem can impact the others, so integrating outcomes of multiple ecosystem assessments can be important for accuracy, completeness and maximising conservation outcomes in coastal environments (Olds et al. 2016, Mahoney & Bishop 2017).

Here, we apply the RLE framework to the connected coastal wetlands of Moreton Bay, Queensland, Australia. The Moreton Bay region supports the highest density and fastest-growing human population in Queensland (Treasury 2018) and is both ecologically and economically important. In this region, seagrass meadows, mangrove forests and saltmarshes form critical transition zones between the land and sea, and provide important ecosystem services, including climate regulation through carbon storage (Alongi 2012), coastal protection (Silliman et al. 2019), and the provision of habitat for megafauna (Sievers et al. 2019) and fisheries species (Weinstein et al. 2000, Carrasquilla-Henao and Juanes, 2017). In applying the RLE framework across the coastal wetland seascape in Moreton Bay, we aimed to: (i) assess the risk of collapse of ecosystems across the seascape, (ii) assess the utility of the RLE framework for assessing and managing interlinked ecosystems, and (iii) address data deficiency in connected systems. This study provides a blueprint for conducting multi-ecosystem assessments of other connected ecosystems on land, in the sea and at their interface.

2. Materials and methods

We applied the RLE criteria according to IUCN guidelines (see Bland et al. 2017a; Table S1) to assess the risk of collapse of coastal wetlands, that encompass saltmarsh, mangroves, and seagrass ecosystems, in Moreton Bay, Australia. We assessed trends and status in ecosystems under four of the five criteria (A through D). Like many RLE assessments (see Keith et al. 2013), we did not assess criterion E as this requires a sophisticated quantitative analysis to assess the future risk of ecosystem collapse (analogous to a Population Viability Analysis for species) that was not possible given the available data and expertise. Within these criteria, ecosystems are assessed at several levels of risk of ecosystem collapse, with levels akin to those popularised by the Red List of Threatened Species™ (*Critically endangered, Endangered, Vulnerable, Near Threatened, Least Concern,* and *Data Deficient*; Table S1; www.iucnredlist.org). Following the standardised framework set out by the IUCN, the final risk level assigned to the ecosystem is the most severe category assigned to any one sub-criteria (i.e. the one-out-all-out principle; Bland et al. 2019).

Criterion A assesses changes in ecosystem extent, where ecosystems with greater losses in area are considered to be at higher risk of collapse. Criterion B assesses ecosystems against thresholds of distribution size (i.e. total area as opposed to the measure of change addressed by criterion A) to identify ecosystems at risk of spatially explicit threats (Murray et al. 2017). Criterion C assesses environmental degradation by physical or abiotic processes (such as declining water quality). Criterion D assesses disruption of biotic processes or interactions (such as trends in characteristic species). Criteria C and D require estimation of the relative severity of decline in key ecosystem indicators, which is then combined with information on the proportion of the ecosystem affected to determine the risk category (Table S1). Criteria A, C and D are assessed over three time frames: the past 50 years (sub-criterion 1), the next 50 years (sub-criterion 2), and since the pre-industrial era at a nominal date of 1750 (sub-criterion 3).

The IUCN RLE also requires detailed information on the ecosystem under assessment, including its spatial distribution, the abiotic and biotic environment, the ecosystem processes and the key threats acting upon the ecosystem. For brevity, we provide an overview of these below, and a more comprehensive description in the Supplementary appendices.

2.1. Ecosystem description

Moreton Bay is a marine and estuarine embayment in southeast Queensland, Australia (27° 20′S; 152° 10′E) that covers approximately 1,500 km^2 (Fig. 1). Moreton Bay is a Ramsar-listed wetland (site no. 631) and to protect its significant ecological resources, a network of marine protected areas was established in 1994 and expanded and rezoned in 2009 (Department of National Parks, 2015). The extent of each ecosystem was defined as all occurrences of the vegetative, habitat-forming species (e.g. mangrove trees, saltmarsh plants, and seagrasses).

Most of Moreton Bay's mangrove communities occur in sheltered estuaries, with the distribution largely driven by hydrological and geomorphological processes. Of these, tidal inundation is one of the most important as different mangrove species display different growth optima and tolerances with respect to inundation, salinity, sedimentation and nutrients (see review by Krauss et al. 2008). The interaction between sea level rise (SLR) and sedimentation can cause significant habitat-shifting if there is sufficient sediment supply for vertical accretion, whereby mangroves can encroach into other vegetative communities such as saltmarsh as sea levels rise and coastal retreat occurs (Saintilan et al. 2014). Moreton Bay mangrove communities consist of seven species (plus the mangrove fern *Acrostichum speciosum*). The most abundant and widespread is the grey mangrove (*Avicennia marina*) (Lovelock et al. 2019). This species forms forests up to 15 m tall on the seaward edges and scrub < 2 m tall in the high intertidal zone, where it mixes with saltmarshes (Lovelock et al. 2019). Other mangrove species present are the black mangrove (*Lumnitzera racemosa*), milky mangrove (*Excoecaria agallocha*), orange mangrove (*Bruguiera gymnorrhiza*), red mangrove (*Rhizophora stylosa*), river mangrove (*Aegiceras corniculatum*) and yellow mangrove (*Ceriops tagal*). These mangroves provide critical habitat, food and breeding areas for a range of biota such as epiphytes on pneumatophores, invertebrates, fish, and birds (Lovelock et al. 2019).

The Moreton Bay saltmarshes are intertidal communities dominated by salt-tolerant flowering plants, primarily low shrubs, herbs and grasses (Saintilan 2009, Accad et al. 2016). Interactions between hydrology and geomorphology control the degree of marine and fluvial sedimentation, the development of intertidal flats, and ultimately the conditions suitable for saltmarsh establishment (Saintilan & Rogers 2013). Within Moreton Bay, tidal inundation is one of the important factors affecting the distribution of saltmarsh due to a combination of

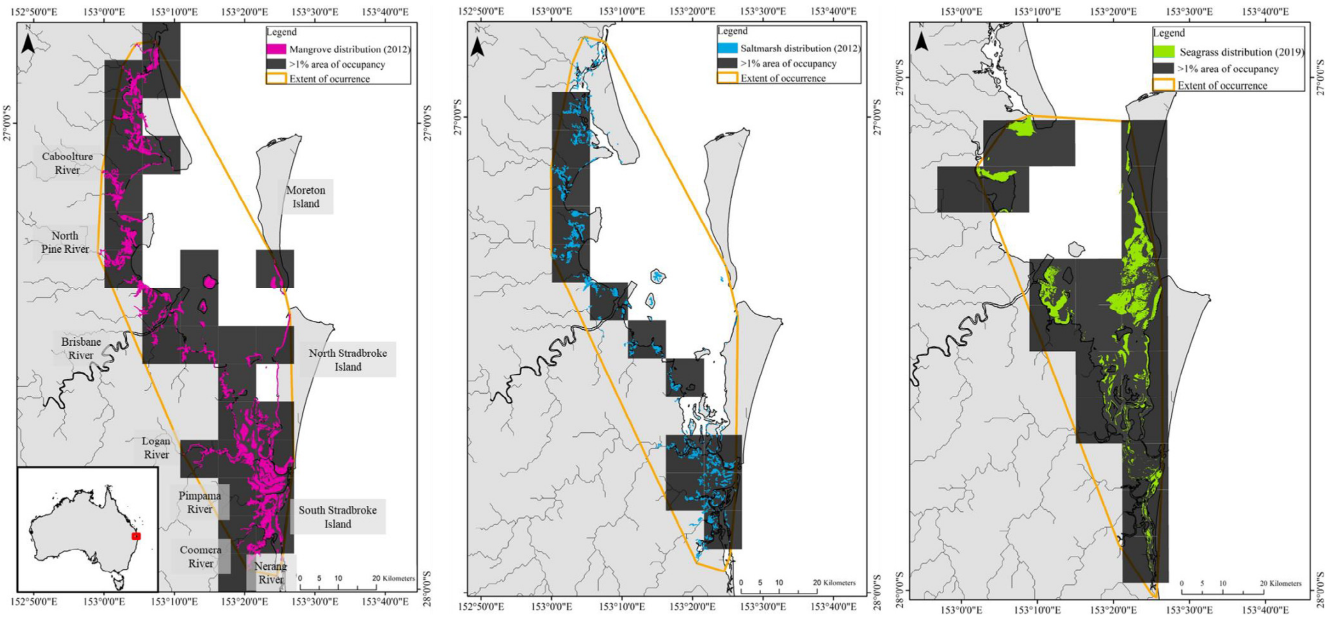

Fig. 1. Mapped distribution of the mangrove (left), saltmarsh (centre) and seagrass (right) ecosystems of Moreton Bay, Queensland, Australia. Mangrove map also shows the seven key rivers, the three key islands, the minimum convex polygon enclosing all occurrences of ecosystems (orange line), and all occupied 10-km^2 grid cells (dark grey). Mangrove and saltmarsh extent from the Queensland Department of Environment and Science (wetlandinfo.des.qld.gov.au/wetlands/resources/tools/mangroves-moreton-bay.html). Seagrass extent from Healthy Land and Water (seqcatchments.maps.arcgis.com/apps/webappviewer3d/index.html?id = 4c9c69cf54d34f499c478ae97b82e44c). Ecosystem distributions (i.e. polygons) are expanded to aid visualisation.

vulnerability to prolonged inundation and being outcompeted by mangroves in the lower and mid-intertidal regions (Saintilan & Rogers 2013). There are 20 saltmarsh species in the high intertidal zone of Moreton Bay, dominated by halophytic grass (*Sporobolus virginicus*) and succulent herbs (*Sarcocornia quinqueflora*, *Suaeda arbusculoides* and *S. australis*) (Accad et al. 2016, Lovelock et al. 2019). Like mangroves, these saltmarshes provide important habitat, food and breeding areas for a range of animal taxa, with the most conspicuous birds, fish and invertebrates (Connolly 2009, Spencer et al. 2009).

The seagrass communities of Moreton Bay are composed of eight species that occur both sub- and inter-tidally (Maxwell et al. 2019). These form extensive meadows in shallow, soft sediment areas where they slow water movement, stabilise sediments, and consequently promote the settlement of silt and reduce turbidity (Maxwell et al. 2015). Water depth, nutrient levels, sediment composition, water clarity and wave action are important constraints on seagrass distribution, and complex ecological feedback loops operate within seagrass ecosystems (Saunders et al. 2013, Maxwell et al. 2017). *Zostera muelleri* is the most common species, occurring in 70–80% of meadows (Figure S3; Roelfsema et al. 2009, Roelfsema et al. 2013). Other species are *Cymodocea serrulata*, *Halodule uninervis*, *Halophila ovalis*, *Halophila decipiens*, *Halophila spinulosa, Halophila minor* and *Syringodium isoetifolium.* The seagrass meadows support diverse communities of epiphytic algae, sessile and motile invertebrates, and fish (Maxwell et al. 2019, Olds et al. 2019). These seagrasses are also a critical food resource for charismatic marine megafauna such as dugongs (*Dugong dugon*) and green turtles (*Chelonia mydas*), which in turn play important ecological roles in seagrass ecosystems (Chilvers et al. 2005, Lanyon 2019, Sievers et al. 2019).

All three ecosystems provide vital resources for a suite of commercially and recreationally important fisheries species and bird species (Tibbetts & Connolly 1998, Johnson 1999, Spencer et al. 2009, Olds et al. 2019). For example, the migratory wading birds of international significance, feed and roost in the mangroves and saltmarsh, and saltmarsh can act as refuges during times of drought for many breeding shorebird species (Milton 2003, Laegdsgaard 2006, Spencer et al. 2009). Intertidal and shallow subtidal habitats are particularly important to wading birds, as their short legs do not allow them to colonise deep waters (Tavares et al. 2015). These groups perform important and diverse ecological functions within coastal wetlands (e.g. herbivory, predation, scavenging, and nutrient storage and transport) (Polis et al. 2004, Poore et al. 2012, Tavares et al. 2019). Importantly, since many of these animals are temporary residents, transient or exhibit ontogenetic habitat shifts, the function of each ecosystem is often contingent on connectivity with the others, and degradation or collapse of any one the ecosystems would likely have flow-on impacts across the seascape.

2.2. Indicators of ecosystem decline

To estimate risk, we need to define the endpoint of ecosystem decline (i.e. the point at which an ecosystem is considered collapsed). Within the RLE, “an ecosystem is collapsed when it is virtually certain that its defining biotic or abiotic features are lost from all occurrences, and the characteristic native biota are no longer sustained” (Bland et al. 2016). Here we define collapse as the complete loss of the habitat forming vegetation or the characteristic biota.

There are several key threatening processes that can lead to such changes for these ecosystems in Moreton Bay. For example, land reclamation for urban areas and agriculture (particularly grazing), environmental contamination, increased commercial and recreational use of the coastline and waterways, and the construction of major shipping, airport and industrial hubs have all affected Moreton Bay’s coastal wetlands directly (e.g. clearing) and indirectly (e.g. pollution and eutrophication) (Pantus & Dennison 2005, Gibbes et al. 2014, Accad et al. 2016, Coates-Marnane et al. 2016, Lockington et al. 2017). Nutrients and pollution can affect fauna directly, increase turbidity which reduces light penetration and seagrass photosynthesis, and can reduce levels of dissolved oxygen (DO) in water (e.g. eutrophication causing algal blooms, which reduces DO; Bricker et al. 2008). When DO becomes too low, aquatic animals such as fish and macroinvertebrates suffer a range of sub-lethal and lethal effects (Vaquer-Sunyer & Duarte 2008).

Future climate change will also likely influence these ecosystems via sea-level rise (SLR), increased storm intensity and frequency,

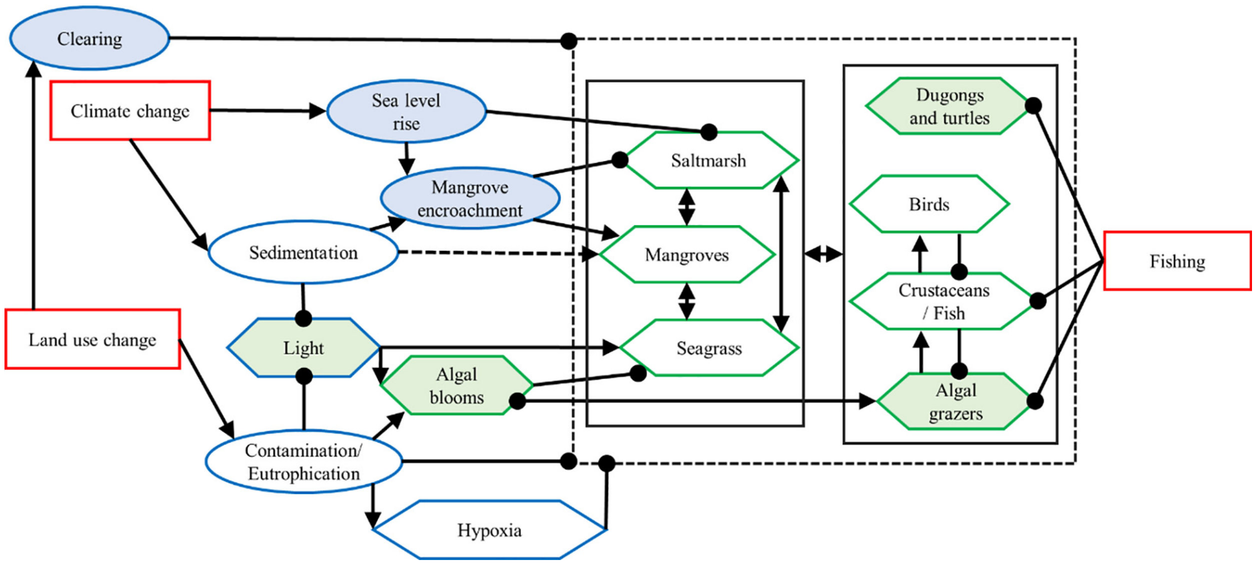

Fig. 2. Conceptual model of key threats and key processes (both abiotic and biotic) relevant to the risk assessment for the mangroves, saltmarsh and seagrass of Moreton Bay, Australia. Only the most influential threats have been shown. Red boxes represent threats, blue ellipses represent abiotic processes, blue hexagons represent the abiotic environment, green ellipses represent biotic processes, and green hexagons represent biotic components. Those coloured blue primarily influence mangrove and saltmarsh, whilst those coloured green primarily influence seagrass. The dashed box represents the seascape, and the solid boxes the key floral and faunal groups under assessment. Pointed arrowheads indicate positive effects and rounded arrowheads indicate negative effects. The dashed arrow indicates the context-dependent effect of sedimentation, which can positively or negatively affect these ecosystems (see supplementary appendices for greater detail).

alterations to precipitation, and higher temperatures (Eslami-Andargoli et al. 2009, Sasmito et al. 2016, Unsworth et al. 2018). For saltmarsh in particular, SLR causes previously suitable areas to become unsuitable, and forces mangroves to migrate landward where they often outcompete and encroach into saltmarsh communities (Saintilan & Williams 1999, Traill et al. 2011, Accad et al. 2016). This threat is exacerbated when landward migration is impeded by development that hardens the shore. Therefore, although similar threats influence all three ecosystems, they can respond differently. Using expert elicitation, we conceptualise and summarise the most pressing threats to each ecosystem in conceptual diagram (Fig. 2).

We collated available data on the most relevant indicators of collapse based on this conceptual diagram (Fig. 2; Table 1). For criterion C, collapse is assumed to occur when conditions within the ecosystem are no longer suitable to support the characteristic biota. To assess criterion C, we had sufficient data to assess SLR (all ecosystems), DO levels (mangrove and seagrass), and light availability (using secchi depth as a proxy; seagrass; Table 1). While shoreline hardening (mangroves and saltmarsh) and contaminant levels (all ecosystems) were also considered as suitable indicators (see Appendices), we did not find sufficient data for these latter two, and they are not discussed further. Since some mangroves are very high in the intertidal (and some seagrasses are intertidal), and only rarely visited by the animals most affected by DO (e.g. fish), we could have used a lower percentage for extent for our assessment (and, thus, needed a higher severity to assign a threatened status). However, since our risk adverse scenario (i.e. 100% extent affected) did not lead to a threatened status, we did not attempt to quantify the proportion of the mangrove and seagrass ecosystems potentially affected by reductions in DO. We consider additional indicators of habitat degradation, such as changes in structural habitat attributes (e.g. density, fragmentation), net primary productivity, plant health metrics and plant reproductive rates important for assessments of the likelihood of ecosystem collapse. However, these are not routinely monitored even in a relatively well-studied area like Moreton Bay. Therefore, we cannot assess these indicators using the RLE framework.

For criterion D, collapse is assumed to occur when the abundance of the ecologically or economically important functional groups within ecosystems decline to zero (Table 1). Specifically, we looked at population trends in birds (mangroves and saltmarsh), fisheries species (all ecosystems) and megafauna (dugongs and green turtles; seagrass; Table 1). We also used rates of mangrove encroachment into saltmarsh as an indicator of ecosystem collapse for saltmarsh under criterion D (Table 1).

2.3. Ecosystem assessment

Here we briefly describe the key data used to assess the three ecosystems under the RLE framework for criteria A through D (Table 1). We provide detailed information about these data, information about peripheral datasets (that are informative but not necessarily important for the assignment of a status), indicators of potential relevance for which we do not have sufficient data, and extended description of the results in the Supplementary appendices. Note that for all indicators for which we defined collapse as 0% extent remaining (see Table 1), we also repeated analyses for the more conservative threshold of collapse at 10% remaining. The assignment of threat status did not change, so we do not present those results.

2.3.1. Decline in distribution – criterion a

To calculate changes in saltmarsh and mangrove extent over the past 50 years (sub-criterion A1), we used maps from Dowling (1986), Dowling and Stephens (2001) and Accad et al. (2016). Combined, the dataset contains changes in extent throughout Moreton Bay from 1955 to 2012. We also assessed declines of each mangrove species, as well as the two key saltmarsh groups from this dataset: the succulent shrubland/open-succulent shrubland (comprising *Sarcocornia* spp., *Suaeda arbusculoides*, *S. australis*) and the grassland/closed-grassland (comprising almost exclusively *Sporobolus virginicus*). To calculate changes in seagrass extent, we used bay-wide maps of seagrass distribution from 2004, 2011 and 2019 (Roelfsema et al. 2009, Roelfsema et al. 2015, Maxwell et al. 2019). We also used maps restricted to the eastern side of the bay for 1972, 1982, 1992, 2002 and 2010 (Lyons et al. 2015). To assess future risk of collapse (sub criterion A2), we linearly extrapolate from current rates of decline for the three ecosystems (Bland et al. 2017a). There are obvious assumptions and uncertainties involved in this extrapolation, and we consequently interpret these in the context of

Table 1

Description of the indicators used to assess the risk of collapse for the mangroves, saltmarshes and seagrass ecosystems of Moreton Bay, including information on the collapse thresholds used and a brief description of the data sources used. Note that other indicators were attempted, but sufficient data to elicit a status did not exist. SLR: seal-level rise.

Criterion	Ecosystem	Indicator	Collapse threshold	Data
Decline in distribution (A)				
	Mangrove & saltmarsh	Change in extent	0% extent remaining	Extent estimates from 1955, 1997 and 2012 (Accad et al. 2016).
	Seagrass	Change in extent	0% extent remaining	Extent estimates from 2004, 2011 and 2019 (Healthy Land and Water; Roelfsema et al. 2009, Roelfsema et al. 2015).
Restricted geographic distribution (B)				
	Mangrove, saltmarsh & seagrass	Current distribution	0% extent remaining	Most recent maps (2012 and 2019) from criterion A
Environmental degradation (C)				
	Mangrove	Dissolved oxygen levels	4.6 and 2.0 mg/L (Vaquer-Sunyer and Duarte 2008)	Measurements at 91 sites 2000–2018 (hlw.org.au)
		SLR	0% extent remaining	Modelling of mangrove extent under SLR (Traill et al. 2011; Beumer et al. 2012; Mills et al. 2016)
	Saltmarsh	SLR	0% extent remaining*	Modelling of saltmarsh extent under SLR (Traill et al. 2011)
	Seagrass	Dissolved oxygen levels	4.6 and 2.0 mg/L (Vaquer-Sunyer and Duarte 2008)	Measurements at 32 sites 2000–2018 (hlw.org.au)
		SLR	0% extent remaining	Modelling of seagrass extent under SLR (Saunders et al. 2013)
		Light levels	Qualitatively by looking at trends in average and sites below a range of thresholds	Secchi depth measurements at 32 sites around the bay from 2000 to 2018 from Healthy Land & Water (hlw.org.au)
Disruption of biotic processes or interactions (D)				
	Mangrove, saltmarsh & seagrass	Abundance of wetland-associated fisheries species	Population declines to 0	Department of Fisheries (Queensland) stock assessments and catch numbers (fish.gov.au/Reports/Species; Thurstan et al. 2019)
	Mangrove & saltmarsh	Abundance of wetland-associated birds	Population declines to 0	Queensland Wader Study Group assessments (Milton 2017)
	Saltmarsh	Mangrove encroachment	0% extent remaining	Estimates of mangrove encroachment (Accad et al. 2016)
	Seagrass	Abundance of seagrass-associated megafauna	Population declines to 0	Dugong and green turtle population estimates (Chilvers et al. 2005; Daley et al. 2008; Meager et al. 2013; Sobtzick et al. 2017)

* Modelling predicted 2% of saltmarsh remaining, so we deemed this close enough to complete loss given variability in estimate (see Traill et al. 2011).

current environmental protections, using expert elicitation, and provide plausible bounds of confidence when necessary. More sophisticated methods to predict future extent under different future scenarios are presented under sub-criterion C2.

2.3.2. Restricted geographic distribution – criterion B

To quantify extent of occurrence (sub-criterion B1), we calculated the area of a minimum convex polygon enclosing all mapped occurrences of the ecosystem. To quantify the area of occupancy (sub-criterion B2), we summed the number of 10 × 10 km grid cells that contained the ecosystem, excluding grid cells that contained patches accounting for < 1% of the grid cell area (Bland et al., 2016). This was done to prevent inflation of the area of occupancy estimate due to many small, dispersed patches which may not substantially offset risks or may constitute mapping errors. To assign a status based on these two sub-criterion, an ecosystem must meet the thresholds that delineate threat categories, as well as at least one of three further sub-criteria that distinguish restricted ecosystems at appreciable risk of collapse from those that persist over long periods within small stable ranges (Keith et al. 2013; Table S1). Hence, the number of threat-affected locations (sub-criterion B3) were estimated for the most significant threats likely to cause collapse over a short time period (~20 years; Bland et al. 2017a).

2.3.3. Environmental degradation – criterion C

For mangroves and seagrass, we used data on dissolved oxygen (DO) levels collected by Healthy Land and Water between 2000 and 2018 as part of the Ecosystem Health Monitoring Program (sub-criterion C1; Table 1). Sampling sites were located throughout the distribution of ecosystems around the bay (Fig. 3), so we considered them to be representative of all sites within Moreton Bay (i.e. 100% extent for the assessment criteria; Table S1). We fitted generalised linear mixed models for site-specific annual means with year fitted as a fixed effect and survey site fitted as a random factor. Since we cannot be sure whether or not temporal trends in DO during the monitoring period also occurred prior to monitoring, we used model estimates to estimate initial values for both 1968 (i.e. extrapolating to 50 years in the past; trends have been occurring preceding monitoring) and 2000 (i.e. first year data are available; assume first year of data is similar to 50 years ago), and used these values to calculate plausible ranges for relative severities based on relevant thresholds of collapse. Our thresholds of collapse were 4.6 mg/L and 2.0 mg/L (Table 1). Oxygen levels below 2.0 mg/L result in significant mortality and sublethal responses of animals, while waters with an oxygen concentration of 4.6 mg/L is expect to maintain populations for most species except the 10% most sensitive (data from 206 species spanning the full taxonomic range of benthic metazoans; Vaquer-Sunyer & Duarte 2008).

For seagrass, we also used secchi depth as a proxy for light levels reaching seagrass (sub-criterion C1), where greater values indicate a greater potential for light to penetrate water. Estimating collapse thresholds for seagrass based on secchi depths is problematic given the issues in converting secchi depths into meaningful light levels (e.g. mol m^{-2} d^{-1}, % surface irradiance) and the large variability in light thresholds among species and locations (Erftemeijer and Lewis, 2006). Instead, we visualise overall trends in mean secchi depth over the last 19 years and trends for the proportion of sites with mean secchi depths below particular depths (i.e. a range of possible thresholds).

To assess risk of collapse from future SLR for all ecosystems (sub-criterion C2), we relied on published models that quantify how much ecosystem is likely to exist under future SLR scenarios and used values for SLR that lead to complete loss as our collapse threshold (Table 1). For mangroves and seagrass, no modelled SLR scenario (see Traill et al. 2011, Beumer et al. 2012, Saunders et al. 2013, Mills et al. 2016, Runting et al. 2018) led to estimated declines in extent of > 30% (even under the most extreme scenarios; see appendix A and C, respectively), so no relative severity calculations were conducted (as these ecosystems satisfy the criteria for a status of *Least Concern*). For saltmarsh, a 1.8 m rise in sea level led to a predicted 98% reduction in saltmarsh extent (Traill et al. 2011), which we considered close enough to 0% remaining to use 1.8 m as our collapse threshold. To estimate predicted SLR in Moreton Bay in 50 years time, we relied on the equations from Vermeer and Rahmstorf (2009) for scenario B1 (Eq. (1))

$$\text{Rate of SLR at } y = 0.1824y - 362.89, \quad (1)$$

where y = year. This equation considers the melting of ice sheets and glaciers and assumes that there is some degree of mitigation of CO_2 emissions (also see Saunders et al. 2013). This equation provides accurate estimates based on current rates of SLR in Moreton Bay (Lovelock et al. 2011). Using this equation and summing predicted annual SLR over the next 50 years, we get a total SLR of 57.4 cm which we use as our future value for the calculation of relative severity (Keith et al. 2013; Table S1).

2.3.4. Disruption of biotic processes or interactions – criterion D

To assess criterion D, we used monitoring data for economically and/or ecologically important functional groups that are associated with or reliant on the ecosystem (Table 1). Dugong and green turtle abundance (seagrass assessment; sub-criteria D1 and D3) were extracted from all available published sources (Chaloupka & Limpus 2001, Chilvers et al. 2005, Daley et al. 2008, Meager et al. 2013, Sobtzick et al. 2017), fisheries catch numbers and species status for species with known associations with wetlands (all ecosystems; sub-criteria D1 and D3) were extracted from stock reports (https://www.

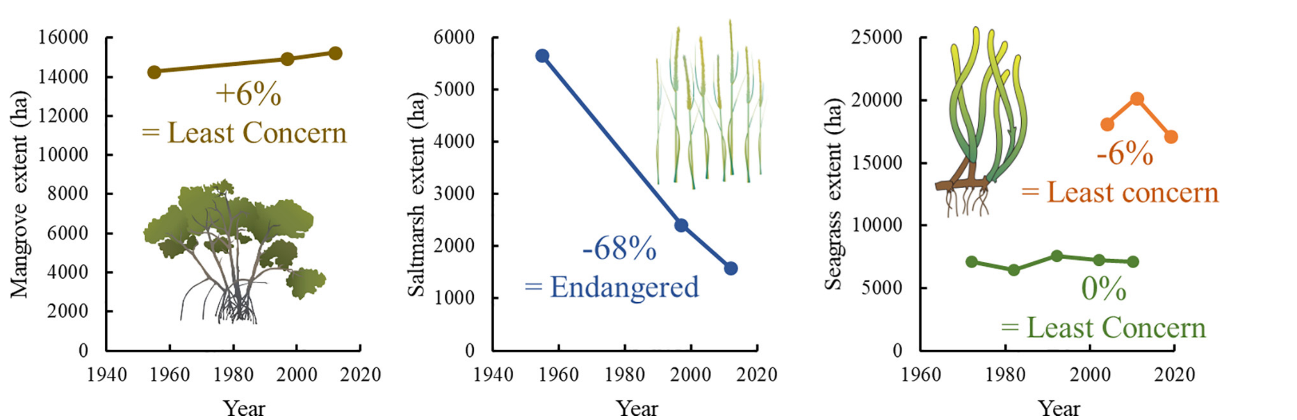

Fig. 3. Changes in extent of mangroves (left), saltmarsh (centre) and seagrass (right) used for the calculation of ecosystem status under sub-criterion A1. For seagrass, orange data is for bay-wide estimates, while green data is for an eastern subset of the bay. Due to the mapping techniques used, no bounded estimates of uncertainty are available. Habitat images from ian.umces.edu.

fish.gov.au/Reports/Species) and published studies (Thurstan et al. 2019), and population trends for birds (mangroves and saltmarsh assessments; sub-criterion D1) were retrieved from surveys of the Port of Brisbane from 2003 to 2016 (Milton 2017). For all, collapse was assumed to occur when populations decline to zero (Table 1). For saltmarsh, we also assessed encroachment by mangroves, using a collapse threshold of 100% encroachment (i.e. 0% saltmarsh extent remaining) based on the data within Accad et al. (2016) (sub-criterion D1; Table 1).

For all criterion C and D indicators, we calculated relative severity based on IUCN RLE guidelines (Eq. (2); Bland et al. 2017a):

$$100 \times (\text{observed or predicted decline})/(\text{maximum decline})], \quad (2)$$

where observed or predicted decline is (Eq. (3)):

$$\text{Initial value–present or future value} \quad (3)$$

and maximum decline (Eq. (4)) is: initial value – collapse value.

Relative severity thus measures the proportional progress of an ecosystem on a trajectory to collapse and is compared to thresholds that delineate threat categories to assign a status (Table S1).

3. Results

3.1. Decline in distribution – criterion A

The area of mangroves in Moreton Bay increased by 6% over the last 50 years (i.e. 14,273–15,231 ha between 1955 and 2012; Fig. 3), and no species declined by > 30% during this period (Accad et al. 2016). Mangroves were, therefore, assessed as *Least Concern* for sub-criterion A1 (< 30% loss; Table 2). By contrast, the area of saltmarsh declined by 69% over the last 50 years (i.e. 5,656–1,580 ha between 1955 and 2012; note we use absolute rate of decline, i.e. linear, as the data suggests this; Fig. 3; Accad et al. 2016). Declines for succulent shrublands and grasslands were similar to the overall saltmarsh ecosystem. Saltmarsh was, therefore, assessed as *Endangered* for sub-criterion A1 (50–80% decline; Table 2). The area of seagrass decreased by 6% over the last 15 years (i.e. 18,111–17,149 ha between 2004 and 2019; Fig. 3). There was, however, no decline in the extent of seagrass in the eastern bay over a period of 38 years (Fig. 3; Lyons et al. 2015). Given the range of mapping techniques used, estimates among years must be compared with caution (see caveats in Appendix C), however, the data suggest seagrass has not declined by > 30%, so seagrass is assessed as *Least Concern* for sub-criterion A1.

Extrapolating 50 years into the future based on current rates of extent change, both mangroves and seagrass meet the criteria for *Least Concern* for sub-criterion A2 (< 30% loss; Table 2). By contrast, saltmarsh extent is predicted to decline by 67 to 100% over the next 50 years, dependent on whether losses occur proportionately (proportional rate of decline) or linearly (absolute rate of decline), respectively. Although direct clearing and land-conversion remain a threat, we expect these to slow in the future and the rate of saltmarsh decline has slightly reduced over the past 20 years (Fig. 3; Accad et al. 2016, Rogers et al. 2016). Therefore, we assess saltmarsh as likely to be *Endangered* because continued losses are anticipated (e.g. from mangrove encroachment), but because of considerable uncertainty around this estimate, we assign a plausible range of *Vulnerable* (30–50% decline) to *Endangered* (50–80% decline) under sub-criterion A2 (Table 2).

Humans have likely contributed to changes in the extent of both mangroves and saltmarsh within Moreton Bay since European settlement, with moderate losses associated with land-reclamation for industrial and port development, river channelling, and the construction of pathways, while minor losses were associated with impoundments, oil spills, vegetation clearing and sedimentation (Duke et al. 2003). There is also evidence that seagrass has been lost from some areas of Moreton Bay (e.g. Bramble Bay in the 1940 s; Dennison & Abal 1999), but seagrass recovery has also occurred in some of these locations

Table 2
Application of the IUCN Red List of Ecosystems criteria for the coastal wetlands of Moreton Bay, Australia. DD, Data Deficient; LC, Least Concern; VU, Vulnerable; EN, Endangered; NE, Not Evaluated. Categories in brackets indicate plausible bounds of status for each sub-criterion. See Table S2 for individual threat status assignments for each indicator within criteria C and D. The overall ecosystem status is defined as the most severe category assigned to any one sub-criteria (i.e. the one-out-all-out principle; Bland et al. 2019).

Criterion	Declining distribution (A)	Restricted distribution (B)	Environmental degradation (C)	Biotic disruption (D)	Quantitative risk analysis (E)	Overall ecosystem status
Mangroves						
Sub-criterion 1 (past 50 years)	LC	LC	LC	LC	NE	**LC**
Sub-criterion 2 (next 50 years)	LC	LC	LC	DD		
Sub-criterion 3 (since 1750)	DD	LC	DD	DD		
Saltmarsh						
Sub-criterion 1 (past 50 years)	EN	EN	DD	VU(LC-VU)	NE	**EN**
Sub-criterion 2 (next 50 years)	EN(VU-EN)	EN	VU	DD		
Sub-criterion 3 (since 1750)	DD	LC	DD	DD		
Seagrass						
Sub-criterion 1 (past 50 years)	LC	LC	LC	LC	NE	**LC**
Sub-criterion 2 (next 50 years)	LC	LC	LC	DD		
Sub-criterion 3 (since 1750)	DD	LC	DD	LC		

(Maxwell et al. 2019). There are, however, no suitable quantitative estimates of the historic extent of these ecosystems in Moreton Bay, so all are assessed as *Data Deficient* under sub-criterion A3 (Table 2).

3.2. Restricted geographic distribution – criterion B

The minimum convex polygon for the ecosystems were 3,225 km^2 for mangroves, 3,002 km^2 for saltmarsh, and 2,281 km^2 for seagrass (Fig. 1). Although some threats continue to affect mangroves and seagrass (see criterion C), we do not have strong evidence to suggest declines in distribution, environmental quality or biotic interactions will likely occur within the next 20 years. Mangroves and seagrass are, therefore, assessed as *Least Concern* for sub-criterion B1. Given current (and predicted continuing) rates of decline in saltmarsh, and the fact that saltmarsh occurs at fewer than five threat-based locations, it is assessed as *Endangered* for sub-criterion B1 (i.e. < 20,000 km^2, but > 2,000 km^2; Table 2; Table S1).

There were 28 (mangrove), 14 (saltmarsh) and 20 (seagrass) 10-km^2 grid cells that contain the ecosystems (Fig. 1). As for sub-criterion B1, saltmarsh is the only ecosystem that satisfies both criteria and is assessed as *Endangered* for sub-criterion B2 (i.e. < 20, but above 2; Table 2).

We consider that Moreton Bay is a single location for threatening processes because the most pressing stressors (e.g. SLR, urbanisation) threaten all three ecosystems at all locations in the bay, and because the bay is managed as a single marine park. However, given current management and protection of saltmarsh (Rogers et al. 2016) we believe the key threats will not *likely* lead to Critically Endangered or Collapsed status within a short time period (~20 years), and therefore assesses each ecosystem as *Least Concern* for sub-criterion B3 (Table 2).

3.3. Environmental degradation – criterion C

Our assessment of current decline examined changes in dissolved oxygen (DO) levels for both mangroves and seagrasses, and changes in secchi depth for seagrass (Table 1). DO levels have slightly decreased over the last 18 years across Moreton Bay, but are still well above our derived thresholds for ecosystem collapse, and calculations of relative severity put these ecosystems at *Least Concern* (Fig. 4). Although there are issues with using secchi depth to estimate light levels reaching seagrass (see Materials and Methods), we nevertheless show that mean secchi depth has not changed over the last 19 years, and the proportion of sites with low light levels (i.e. shallow secchi depths) has not increased (Appendix C). Therefore, we assess both mangroves and seagrass as *Least Concern* for sub-criterion C1 (Table 2).

As discussed (see section 2.3.3), no modelled future SLR led to > 30% extent declines for mangroves or seagrass, so these ecosystems are assessed as *Least Concern* for sub-criterion A2 (< 30% loss; Table 2). For saltmarsh, we calculated relative severity using range standardisation (Keith et al. 2013), a current value of 0 m (i.e. current sea-level), a future value of 0.574 m, and a collapse threshold of 1.8 m (see section 2.3.3. for details). This produced a severity of 32%, so saltmarsh is assessed as *Vulnerable* for sub-criterion C2 (between 30 and 50%; Table 2).

An assessment of historical changes was not possible as there are no robust, quantitative data that can be used to describe historic changes in relevant abiotic variables for Moreton Bay. Thus, these ecosystems are all assessed as *Data Deficient* for sub-criterion C3 (Table 2).

3.4. Disruption of biotic processes or interactions – criterion D

A substantial area of saltmarsh has been lost from the region due to the establishment and encroachment of mangroves (Lovelock et al. 2011). Mangrove encroachment led to a loss of ~ 3,000 ha (34%) of saltmarsh and claypan habitat between 1955 and 2012 (Accad et al. 2016). Applying range standardisation and using a collapse threshold of 100% mangrove encroachment and an initial value of 0% at the start of the 50 years, relative severity is calculated to be 34%. Since small amounts of saltmarsh encroached into mangroves and other vegetation types, and small amounts of saltmarsh were encroached by other vegetation types during this period (Accad et al. 2016), we assess saltmarsh under this indicator as *Vulnerable*, with a plausible range of *Least Concern* to *Vulnerable* (Table 2).

Population counts for waterbird species associated with mangroves and saltmarsh were steady between 2003 and 2016 (Milton 2017). Of slight concern is the migratory eastern curlew (*Numenius madagascariensis*), whose population nominally declined during this period (Milton 2017). However, there is substantial variability in counts among years for this species, overall trends were not statistically significant (based on analysis in Milton 2017), and there is high potential for apparent trends to be indicative of natural variation rather than long-term population declines. As a group, there is strong evidence to suggest populations have not declined by > 30%. Therefore, we assess sub-criterion D1 for mangrove and saltmarsh birds as *Least Concern* (Table 2).

In terms of fisheries species, there is little evidence to suggest catch rates have declined since the 1940 s for species which use these three ecosystems (Thurstan et al. 2019). Recent fisheries stock reports also state that population trends and catch rates of these wetland-associated species are stable and sustainable. Therefore, we assess sub-criterion D1 for mangroves, saltmarsh and seagrass fisheries species as *Least Concern* (Table 2). Although historical catch data suggest little change in the populations of fisheries species associated with coastal wetlands since the 1940 s (Thurstan et al. 2019), we do not consider this sufficient to assign a status for any of the three ecosystems, so sub-criterion D3 is assessed as *Data Deficient* for fisheries species (Table S1).

The Moreton Bay dugong population was estimated to be between 118 and 307 individuals in the 1970 s, 453 in 2005, 696 in 2011 and 601 in 2016 (Chilvers et al. 2005, Meager et al. 2013, Sobtzick et al. 2017). There is also evidence of a large and increasing population of sea turtles (primarily green turtles) within the bay (> 25,000 individuals in 2016; Sobtzick et al. 2017). Monitoring from a large portion of Queensland suggests that green turtle populations steadily increased between 1985 and 1992 following commercial harvest, which affected the population up until 1959 (Chaloupka & Limpus 2001, Limpus & Fien 2009). The number of female green turtles nesting at beaches nearby in the southern Great Barrier Reef remained stable between 1967 and 2004 (Limpus & Fien 2009). It is not known how many of these turtles originated in Moreton Bay, but stability in the nesting populations implies no long-term downward trend in associated foraging areas. Given this information, dugong and sea turtle populations are unlikely to have declined by > 30% over the past 50 years, so we assess seagrass as *Least Concern* for sub-criterion D1 (Table 2).

Historical accounts suggest a decline in dugong abundance in southern Queensland in the 1800 s due to commercial dugong harvesting (Daley et al. 2008). Estimates from informal boating surveys suggest that there were 300–400 individuals in Moreton Bay at the end of the 1800 s, and from 1963 to 1980, anecdotal reports suggest the Moreton Bay dugong population was stable or rebuilding (Meager et al. 2013). Overall, Daley et al. (2008) suggest that commercial dugong harvesting probably had an impact on Moreton Bay population, but we have little evidence to suggest that population abundance now is significantly different from historic abundances. Green turtles were also exploited for meat and shells within Moreton Bay from the mid- to late-1800 s, but populations were reported to be high and demand was easily met (Daley et al. 2008). In 1968, all of Queensland's turtle species became protected and harvesting ceased. Taken together, sub-criterion D3 for seagrass megafauna is assessed as *Least Concern* (less than a 50% decline; Table 2).

An assessment of historical changes was not possible for mangroves or saltmarsh as there are no suitable data that can be used to sufficiently describe changes in biotic processes or characteristic fauna in Moreton Bay. Therefore, these ecosystems are assessed as *Data Deficient* for sub-

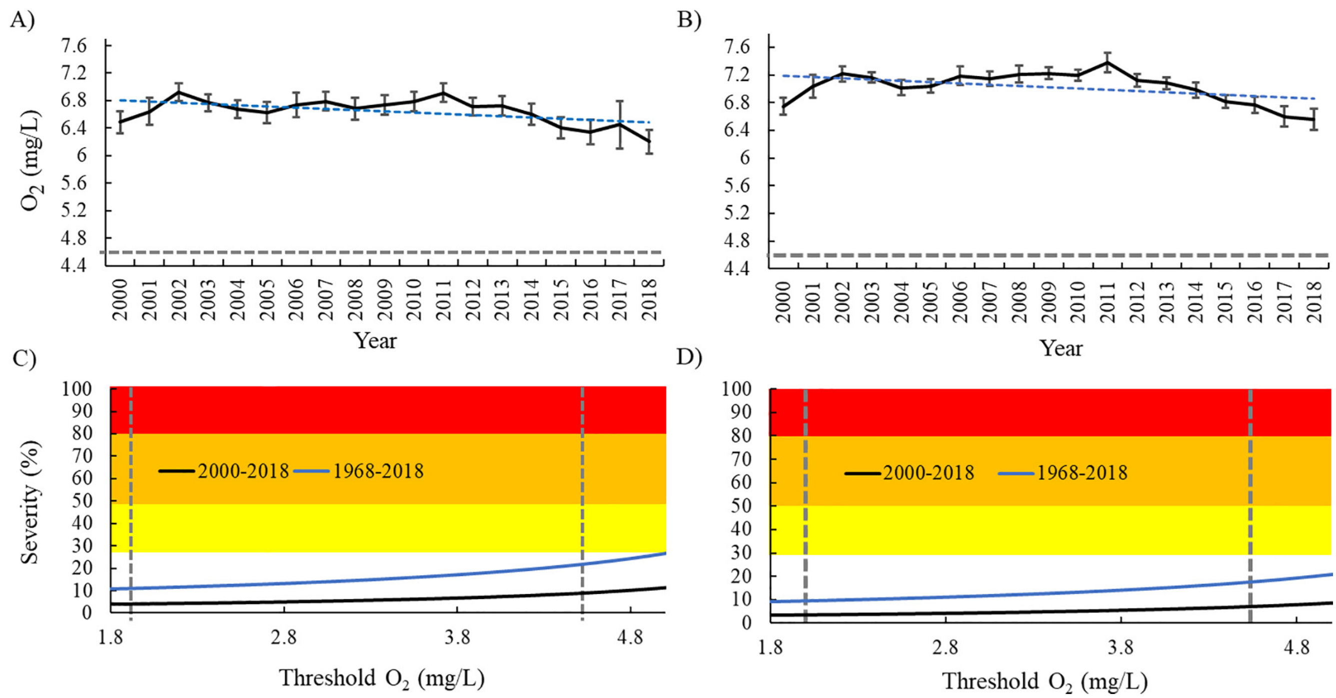

Fig. 4. (A) Mean annual dissolved oxygen levels (with 95% confidence intervals) recorded at 91 sites within 500 m of mangroves (A: left; n = 55 to 91, depending on year) and 32 sites with 100 m of seagrass (B: right) in Moreton Bay. The blue dashed line is the linear trend line ($R^2 = 0.28$) and the horizontal dashed grey line represents the risk adverse threshold level (4.6 mg/L). Severity values were calculated using range standardisation for initial values from 1968 (i.e. extrapolating to 50 years in the past; blue line) or 2000 (i.e. first year data is available; black line), current values from 2018 (i.e. last year data is available), and a range of collapse thresholds for mangroves (C) and seagrass (D). Grey vertical bars represent the chosen thresholds for dissolved oxygen; 4.6 mg/L and 2.0 mg/L (see text for justification). Coloured regions represent different ecosystem statuses based on an extent of 100%: red – critically endangered; orange – endangered; yellow – vulnerable, and; white – non-threatened.

criterion D3 (Table 2).

4. Discussion

Our application of the RLE framework allowed us to assemble current knowledge on the status of, and threats to, the vegetated coastal wetlands of Moreton Bay, Queensland, Australia. By bringing together a diverse assortment of extensive long-term datasets on indicators related to the mangrove, saltmarsh and seagrass ecosystems in this area, we assessed seagrass and mangroves as being of *Least Concern.* However, the saltmarsh ecosystem in the same area is classified as *Endangered,* primarily as a result of past clearing and mangrove encroachment.

4.1. Management of vegetated coastal wetlands in Moreton Bay

The Least Concern status of mangroves and seagrass in Moreton Bay can at least partly be attributed to the ongoing efforts to manage the bay. In the past two decades, coordinated management actions in southeast Queensland have facilitated regional catchment action plans which have led to significant reductions of terrestrial inputs to the bay (Leigh et al. 2013). This has led to improvements in the condition of the bay which has been reflected in annual report card grades (www.reportcard.hlw.org.au). These management actions have contributed to seagrass recovery in areas devoid of seagrass since the 1940 s (Maxwell et al. 2019), and are key driving forces behind the status of mangroves and seagrass in this region.

Despite these management successes, the future of saltmarsh in Moreton Bay looks dire. While we believe that direct clearing of saltmarsh will be minimised due to current legislation and protection measures (see Rogers et al. 2016), other pressures are expected to persist. Ongoing SLR and sediment supply will favour the establishment and expansion of mangroves, suggesting that mangroves will continue to encroach into saltmarsh within Moreton Bay (Lovelock et al. 2011) and in many locations around the world (Rogers et al. 2019). The processes that promote mangrove encroachment are diverse but are largely attributed to changes in temperature, hydroperiod, nutrients, sea level, sedimentation, elevation and salinity (Saintilan et al. 2014). This transition can affect the provision of ecosystem services, function, and food-web structure, for example, by reducing habitat available for fauna requiring open vegetation structure such as migratory birds (Kelleway et al. 2017). Nutrient enrichment (an issue for many of the estuaries of Moreton Bay; Pantus & Dennison 2005) also hastens mangrove encroachment into saltmarsh (Dangremond et al., in press). The ultimate effect of mangrove encroachment will be further exacerbated if saltmarsh is prevented from migrating landward because of the presence of other land types such as urban structures and agricultural fields; a process known as 'coastal squeeze' (Pontee 2013). In Moreton Bay, efforts to minimise eutrophication, sedimentation and coastal squeeze could therefore be made to protect saltmarsh.

4.2. The importance of integrating outcomes for connected ecosystems

The connected nature of coastal wetlands demands consideration of connectivity between habitat types when assessing the risk of collapse for any one ecosystem type. Our results demonstrate how individual ecosystem assessments may miss important processes and provide incomplete or misleading information for policy makers and managers. If we were to rely only on mangroves or seagrass as indicators of coastal ecosystem health, we may mistakenly assume that the seascape is in good condition when, in fact, there is cause for concern based on the connected nature of mangroves and seagrass with saltmarsh (Valiela et al. 2004, Saintilan et al. 2007, Meynecke et al. 2008). Saltmarsh might contribute significant food sources, such as crustacean larvae (Hollingsworth & Connolly 2006), to the fauna that reside within other coastal wetlands. Alternatively, fish associated with mangroves or seagrass might also depend on saltmarsh during some stage of their lives (Pittman & McAlpine 2003). As a result, collapse of saltmarsh and the changing dynamics among ecosystem types can affect the entire seascape often in unpredictable and complex ways that are difficult to measure. For example, 'extinction debt' across the seascape may occur, where local species extinction occurs with a substantial delay following habitat loss or degradation in saltmarsh (Kuussaari et al. 2009). The ecological impacts of past saltmarsh loss for the seascape may by delayed and even ambiguous. Ultimately, the overall integrity of a connected seascape, such as the coastal wetlands examined here, could be jeopardised unless the connected nature of these ecosystems is considered when formulating management and conservation decisions.

Encroachment dynamics and high inter-ecosystem connectivity are certainly not unique to coastal wetlands. For example, forest-savannah-grassland landscapes exist in mosaics that undergo complex encroachment dynamics, most recently with woody plants encroaching into grasslands causing considerable ecological changes (Eldridge et al. 2011, Buitenwerf et al. 2012, Saintilan & Rogers 2015). Beyond connectivity amongst vegetated coastal wetlands, these habitats also display considerable ecological connectivity with coral reefs and mudflats (Laegdsgaard & Johnson 1995, Olds et al. 2012) and even deeper offshore environments, where species that use mangroves and seagrass as juveniles are highly prevalent as adults (Pearson & Stevens 2015). Therefore, utilising an assessment approach that focuses beyond ecosystem types in isolation can be beneficial across a diverse range of ecological systems and situations.

We could have conducted a fully integrated assessment (i.e. where the three habitats were combined and treated as one single ecosystem) instead of, or in addition to, assessing the coastal wetland ecosystems independently and integrating outcomes to gain an overall understanding of the risk of collapse. In our circumstance and given the availability of data, however, this would have provided no additional capacity to deal with data deficiencies. Further, this approach would mask important inter-ecosystem processes. For example, in fully integrated assessment, mangrove encroachment into saltmarsh may have resulted in no net change in habitat extent, whereas in the individual assessments we were able to identify large within-habitat changes that could be interpreted in combination. Therefore, separate assessments conducted and interpreted together are more useful than an integrated assessment here. Given there are important functional differences in ecosystems, the total amount or condition of coastal wetlands is likely less important (or at least not more important) than the amount and condition of each. Whether this is true across highly connected ecosystems elsewhere has yet to be tested.

4.3. Knowledge gaps and considerations for future assessments

The complexity of conducting RLE assessments in coastal systems has been discussed previously (see Mahoney & Bishop 2017, Marshall et al. 2018), and through the assessment process we highlighted several key knowledge gaps for Moreton Bay that are likely common across coastal ecosystems worldwide and that prevent more comprehensive assessments. These were (1) a lack of quantitative, long-term data on indicators of collapse (Rowland et al. 2019), (2) problematic datasets to estimate population trends of fauna (e.g. confounded fisheries catch data), and (3) difficulties in setting quantitative collapse thresholds (Bland et al. 2018).

Although we were able to quantify changes in extent for our three ecosystems, we were unable to comprehensively assess changes in indicators of habitat degradation. Losses of key biota and biodiversity can arise from habitat degradation, which may occur independently of or concurrently with changes in absolute extent. Data on temporal changes in structural metrics such as the density of plants (e.g. seagrass, mangrove trees) or rates of fragmentation could provide important information assessments of the likelihood of ecosystem collapse. For example, increases in rates of fragmentation can lead to a reduction in important ecological functions (Haddad et al. 2015). Further, reductions in seagrass canopy density can decrease benthic species richness (Herkül & Kotta 2009), and increase sediment resuspension, consequently reducing habitat suitability (Carr et al. 2016). Temporal trends in net primary productivity, plant health metrics or plant reproductive rates could similarly inform these risk assessments. Reductions in any of these could be indicative of important environmental changes and overall system degradation. Unfortunately, sufficient temporal data on these metrics does not exist for Moreton Bay. Although we recognise the challenges, we recommend that these metrics begin to be routinely monitored for use in future RLE assessments, and to gain a greater understanding of health the ecosystem more broadly.

Satellite remote sensing is revolutionising the collection of such information (Sarker et al. 2019), providing unprecedented access to high-resolution multi-spectra imagery of the world's coastal marine areas (Vuolo et al. 2016). For instance, Worthington and Spalding (2018) used Landsat time series to assess the change in a range of mangrove vegetation indices (e.g. normalised difference vegetation index; NDVI) to identify degraded areas. Remote sensing can provide accurate data for a new suite of indicators of collapse and will likely form a significant part of future assessments conducted under the RLE framework (Murray et al. 2018). Alternatively, drones may provide an opportunity to monitor some of these indicators across large spatial scales at relatively low cost (Manfreda et al. 2018).

It is often difficult to standardise the data collected for assessing animal population trends through time due to potential confounding, and thus to assess whether surveyed abundances are truly indicative of population trends (i.e. a measure of ecosystem collapse). For example, advanced technologies in fisheries improve efficiencies (i.e. catch per unit effort) and may convolute assessments of population trends (Thurstan et al. 2016), especially when comparing across relatively large time scales as is attempted under the RLE framework. The same is true for monitoring of any species when methodologies or monitoring locations change through time, which can be common when collating population estimates from various sources and over long time periods. Now that the RLE framework has been adopted as a global standard for assessing ecosystems (see Bland et al. 2019), there is great benefit in developing monitoring programs that suit the needs of the RLE.

For several indicators of collapse, quantitative collapse thresholds below which characteristic biota, ecological functions and/or processes are not supported are unavailable in the literature (Bland et al. 2018), which can prevent accurate calculations of relative severity and the assignment of a threat status. Although we have long-term datasets on various water quality parameters (e.g. nitrogen, phosphorus and total suspended solids), we could not use these to assign a status due to a lack of established thresholds. The formulation of thresholds is a critical bottleneck in the RLE assessment process (also see Bland et al. 2018). Large-scale experiments showing how wetland ecosystems can collapse as a result of changes in adjacent systems (Deegan et al. 2012, Coverdale et al. 2014) point to how thresholds might be determined for Moreton Bay.

Some of the issues we have raised here can, at least partly, be overcome by developing a comprehensive quantitative risk model within criterion E. For our system, a dynamic process model that incorporates elements such as SLR, sediment accretion rates, coastal squeeze and inter-ecosystem dynamics (e.g. Runting et al. 2018), would be invaluable for this type of assessment, as would a better understanding of the ways that changes in ecosystem extent and structure affect fauna. Incorporating connectivity among these connected coastal wetlands within these models adds an important perspective to how habitat loss and degradation in any one ecosystem influences risks to the seascape. The RLE (regardless of whether criterion E is applied) provides a framework around which to organise the information gained from such modelling and presents it in a risk- and management-relevant context. For now, the development of such models is beyond the scope of this research paper, but we emphasise that future efforts to assess the risk of collapse would benefit by utilising quantitative predictive models.

5. Conclusions

The IUCN RLE framework provides a standardised tool to assess the risk of ecosystem collapse. Using the coastal wetlands of Moreton Bay as a case study, we estimate the risk of ecosystem collapse from various threats and symptoms of decline and highlight the importance of integrating outcomes for highly connected ecosystems. Through multiple lines of evidence, we show that mangroves and seagrass satisfy the criteria for *Least Concern* while saltmarsh is *Endangered* due to declines in extent from clearing and mangrove encroachment. Given the interconnected nature of these coastal wetlands, collapse in any one ecosystem can have significant flow-on effects in the others. As RLE assessments progress around the world and inform conservation action (Bland et al. 2019), there is need to identify the best way to utilise and tailor assessments to ecosystems of interest. Recognition of the importance of ecological connectivity (through integrating assessment outcomes) can maximise the value of available data, the accuracy of assessments and the potential for successful conservation outcomes. By identifying important indicators that are data deficient, research and monitoring efforts can be tailored to enable more robust future RLE assessments.

CRediT authorship contribution statement

Michael Sievers: Conceptualization, Formal analysis, Writing - original draft, Methodology, Validation, Investigation, Writing - review & editing. **Ryan M. Pearson:** Methodology, Validation, Investigation, Writing - review & editing. **Mischa P. Turschwell:** Conceptualization, Methodology, Validation, Investigation, Writing - review & editing. **Melanie J. Bishop:** Methodology, Validation, Investigation, Writing - review & editing. **Lucie Bland:** Methodology, Validation, Investigation, Writing - review & editing. **Christopher J. Brown:** Conceptualization, Methodology, Validation, Investigation, Writing - review & editing. **Vivitskaia J.D. Tulloch:** Conceptualization, Methodology, Validation, Investigation, Writing - review & editing. **Jodie A. Haig:** Conceptualization, Methodology, Validation, Investigation, Writing - review & editing. **Andrew D. Olds:** Methodology, Validation, Investigation, Writing - review & editing. **Paul S. Maxwell:** Methodology, Validation, Investigation, Writing - review & editing. **Rod M. Connolly:** Conceptualization, Methodology, Validation, 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

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