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Assessing threats and rehabilitation opportunities for mangrove-saltmarsh blue carbon ecosystems

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ABSTRACT

Mangroves and salt marshes are vital intertidal ecosystems that provide key ecosystem services, including shoreline stabilization, carbon storage, and biodiversity support. This study assessed the current condition and threats to valued tidal wetland mangrove and saltmarsh ecosystems along the southern Great Barrier Reef (GBR) coastline, Queensland, Australia. The study also identified potential sites of rehabilitation potentially suitable for funding under Australia's blue carbon sequestration methodology. In 2023, an aerial shoreline survey was conducted between Gladstone and Cairns, revealing widespread impacts from climate-related stressors, such as shoreline erosion, expanding depositional banks and storm damage, along with anthropogenic stressors, including altered hydrology and agricultural encroachment. Fifty-two potential rehabilitation parcels totalling 17,255 ha were identified, with rehabilitation opportunities categorized as saltmarsh-dominated, mangrove-dominated, or mixed systems. Rehabilitation suitability was evaluated also using tidal barrier mapping, potential inundation extent, and trends in the normalized difference vegetation index (NDVI) of adjacent mangroves. The results revealed the importance of local stressors, the need for integrating stakeholder involvement, and the implementation of long-term environmental monitoring to bolster effective rehabilitation. This study provides a framework and baseline for selection and prioritizing rehabilitation efforts for improving the resilience and functionality of tidal wetlands in the GBR region.

1. Introduction

Mangroves and saltmarsh are vital intertidal wetland ecosystems that provide a wide range of ecosystem services critical to coastal well-being and resilience. These include shoreline stabilization by trapping sediments, reducing erosion, and mitigating storm surges (Barbier, 2015; Sandilyan and Kathiresan, 2015); improving water quality by filtering pollutants (Reis et al., 2017); and supporting biodiversity and habitat through their role as nurseries and food supplies for marine species (Carrasquilla-Henao and Juanes, 2017). Additionally, these ecosystems play a significant role in carbon storage (blue carbon) and greenhouse gas reduction, contributing to global climate change mitigation (Rogers et al., 2019). Their ecological functions underpin commercial fisheries, coastal protection, and cultural heritage, emphasizing the need for their conservation and rehabilitation (Barbier, 2015; zu Ermgassen et al., 2020).

Globally, mangroves and saltmarshes face comparable pressures, with widespread losses recorded in Southeast Asia, the Caribbean, South India and Sri Lanka, and West Africa due to aquaculture, timber harvesting, coastal development, and altered hydrology (Ashton, 2022; Campbell et al., 2022; Goldberg et al., 2020). As sea levels rise and coastlines erode, maintaining mangrove forests becomes increasingly important to protect coastal land-based activities from inundation and storm damage. However, habitat degradation, biodiversity loss, simplified structures, and altered ecosystem processes weaken their ability to mitigate, respond to and recover from climate impacts while protecting coastal zone areas (Cahoon et al., 2021). Despite their importance, mangroves remain vulnerable to destruction and degradation because of poor catchment and coastal management. Approximately 62 % of global mangrove loss between 2000 and 2016 resulted from land conversion for aquaculture and agriculture (Goldberg et al., 2020).

In Australia's Great Barrier Reef (GBR) region, mangroves and saltmarsh are legally protected in Queensland but still face threats from altered water flow, catchment land clearing and harvesting, urban development, pollution, rising sea levels and climate change (Canning and Waltham, 2021). Coastal tidal wetland ecosystems play a critical role in the GBR's sustainability by filtering sediment and contaminants

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from terrestrial runoff, preserving water clarity essential for coral and seagrass health. Mangroves' complex root systems anchor coastal soils, preventing erosion and minimizing sediment deposition on coral reefs and seagrass meadows (Duke and Larkum, 2019; Duke and Wolanski, 2000; Wolanski and Duke, 2002), which is vital for the survival of these nearshore coastal marine ecosystems. Moreover, mangrove ecosystems act as nurseries for marine species that later migrate to nearshore reefs and seagrasses, bolstering biodiversity and ensuring a resilient food web (Brown et al., 2020; Rogers and Mumby, 2019; Sambrook et al., 2019). The presence of mangroves and saltmarsh is, therefore, integral to maintaining both the structural, functional and biological integrity of the GBR. While managing climate change may be difficult at the local level, reducing local stressors and restoring these ecosystems can strengthen their resilience to climate impacts and help sustain their ecosystem services (Jakovac et al., 2020; Sunkur et al., 2023).

Like many places around the world, mangrove and tidal saltmarsh rehabilitation projects in Australia have encountered several key obstacles, including environmental factors such as altered hydrology and poor water quality, technical issues like inappropriate site selection and insufficient rehabilitation expertise. Additional issues included social barriers with limited stakeholder engagement and conflicts over land tenure (Saunders et al., 2024; Stewart-Sinclair et al., 2020). Economic and political obstacles also hinder progress, with rehabilitation projects often constrained by short-term funding and delays in policy implementation or approvals. These constraints are not unique to Australia; similar ecological, technical, and governance barriers have been reported from large-scale mangrove rehabilitation efforts in Asia, Africa, and the Americas (Bell-James et al., 2025; Dale et al., 2014; Zimmer et al., 2022). Clearly there is value in both cross-regional learning and context-specific approaches. Addressing these barriers requires integrated approaches combining sound technical planning, capacity building, stakeholder involvement and sustainable funding mechanisms to support long-term rehabilitation success (Saunders et al., 2024; Stewart-Sinclair et al., 2020).

Recently, the Australian Government introduced a voluntary funding mechanism, via the Emissions Reduction Fund (renamed to the Australian Carbon Credit Unit (ACCU) Scheme), for coastal wetland projects aimed at increasing carbon abatement through rehabilitation via tidal reintroduction (The 'Australian Blue Carbon Method' hereinafter; Lovelock et al., 2023; Nong and Siriwardana, 2017). Eligible projects can earn tradeable carbon credits, which can be sold to the government or private investors seeking carbon offsets. In addition to tidal reintroduction, projects must also demonstrate localised management of potential threats likely to undermine rehabilitation outcomes, such as managing fire, feral animals and weeds (Lovelock et al., 2023; Nong and Siriwardana, 2017).

While there are several approaches for rehabilitating mangrove and saltmarsh ecosystems beyond tidal reintroduction, such as altering sediment deposition, supplemental planting or via removal of limiting stressors, these methods are not currently recognised by the scheme, limiting the applicability and practicality of the funding mechanism (Kelleway et al., 2020). Nonetheless, identifying a short list of potential rehabilitation sites coupled with a summary list of local threats likely to impede rehabilitation success would be useful to organisations aiming to invest in blue carbon project initiatives (Howie et al., 2024; Rogers et al., 2023).

While some valuable first steps towards identifying future blue carbon rehabilitation sites have been made in the Great Barrier Reef region, the methodologies used to-date have largely relied on desktop assessments of satellite imagery, combined with limited field investigations (Hagger et al., 2022; Rowland et al., 2023). Such assessments mostly only have the capacity to identify local threats likely to affect specific rehabilitation sites.

A more broadscale assessment is required to identify and evaluate the range of potential threats using an aerial version of the boat-based shoreline assessment method described by Mackenzie et al. (2016). The aerial survey methodology was distinguished by its acquisition of an archival database of geopositioned, oblique, high-resolution, overlapping still imagery taken along continuous sections of coastline (Duke et al., 2022a).

With prior aerial surveys, researchers were able to identify a diversity of dominant threats, like the El Nino-related mass dieback of mangroves along more than 1000 km of the Gulf of Carpentaria (Duke et al., 2022a); damage caused by severe tropical cyclones (Duke et al., 2024), the influences of sea level rise (Duke and Mackenzie, 2018), extreme flooding, feral pigs and scrub fires (Duke et al., 2020). Knowledge of such relatively local threats from large sections of coastline provides data that better informs rehabilitation proponents by helping identify likely project sites and their prioritization for maximizing future rehabilitation outcomes while accommodating site constraints and the availability of project resources.

2. Methods

2.1. Sea level and rainfall trends

Sea level and rainfall records were sourced from the Bureau of Meteorology (BOM), which operates numerous gauges along the GBR coast (Bureau of Meteorology, 2025a, b). For this study, we selected four long-term stations (indicated by BOM station code) at major centres to provide indicative trends: mean monthly sea level at Cairns (59060), Townsville (59250), Mackay (59510) and Gladstone (59750), and annual rainfall at Cairns (31011), Townsville (32040), Mackay (33119) and Gladstone (39123). Trends were assessed for each dataset between 1985 and 2023 using linear regression in R 4.3.3 (R Core Team, 2024).

2.2. Coastline surveys

An aerial survey of mangroves and saltmarsh from Gladstone to Cairns was conducted in May 2023. Photographs were mostly taken from a Robinson 44 helicopter, providing a manoeuvrable platform for identifying and scoring specific shoreline habitat indicators (Table 1). These indicators linked to threats such as rising temperatures, sea-level rise, pollution, and reclamation. Continuous georeferenced photographs (~20–30 % overlap) were captured from a helicopter flying at ~30 m altitude and ~100 m from the shoreline, at speeds of 70–100 km/h. This approach has successfully assessed large sections of northern Australia's coastline, from Gladstone to Broome (Canning and Duke, 2023; Duke et al., 2010). Using these images and field observations, shoreline condition and threats were evaluated across 16 sub-regions (Fig. 1).

For the shoreline assessed in each sub-region, a suite of ecological indicators of direct human-related and climate-related stressors was developed, applied and reported upon in multiple regional surveys (Duke and Canning, 2024b; Duke and Mackenzie, 2018; Duke et al., 2021, 2022b). We considered human stressors to be those where alterations have arisen from local human activities, including the introduction of feral species, whereas climate stressors (of which humans influence globally) are those arising from broader climate change and extreme weather events. For the current study, these indicators were rated in terms of extent and severity (Tables 1 and 2), with the product of these two ratings, normalized out of 5, indicating an overall stressor score ranging from 0 to 5. For example, across a sub-region terrestrial retreat may have been observed across ~40 % of the length of coastline, resulting in an extent rating of 3, with recovery unlikely to occur, resulting in a severity rating of 5, with the product of the extent and severity ratings, normalized by 5, giving rise to an overall impact score of 3 for terrestrial retreat. For each sub-region, an overall score was calculated for both the human-related stressors and the climate-related stressors by summing the severity ratings for the indicators within each category. Extent was quantified as the proportion of the total mapped shoreline length in each sub-region assessed (Figs. 1 and 2), rather than by fixed segments (e.g., every 100 m). This ensured consistency and

Table 1
Human-related stressors (except climate change) and climatic stressors (including climate change) and visual indicators appraised during coastal surveys had been developed, applied and reported upon in multiple regional surveys (Duke and Canning, 2024b; Duke and Mackenzie, 2018; Duke et al., 2021, 2022b).

Category	Stressor	Visual Indicator	Impact
Human-drivers	Structure Loss	Presence of constructed features such as rock walls, wharves, ramps, and roads.	Loss of natural habitat, reduced ecological function, and disruption of natural hydrology.
	Direct Loss	Absence of trees due to direct removal, including clearing or landfill.	Habitat loss, reduced ecosystem services such as shoreline protection, and diminished biodiversity.
	Altered Hydrology	Alteration of tidal hydrology due to tidal bunds, drains, or impounded areas.	Disrupted water flow, increased water pooling, mangrove dieback, and reduced ecosystem resilience.
	Encroachment	Absence of a buffer between coastal ecosystems and human settlement, or loss due to conversion along tributaries for development.	Loss of ecosystem services, habitat fragmentation, and increased vulnerability to coastal storms and flooding.
	Access Tracks	Observable damage from vehicle or foot tracks.	Soil compaction, damage to vegetation, altered drainage patterns, and potential habitat fragmentation.
	Stock Impact	Observable impact from livestock presence or tracks, such as from cattle, horses, and goats.	Vegetation damage, soil compaction, reduction in mangrove recruitment, and ecosystem degradation.
	Feral Pig Damage	Direct presence of feral pigs, tracks, wallows, and diggings.	Disturbance of vegetation and soil, lack of mangrove recruitment, habitat degradation, and erosion.
	Pollutant Impact	Presence of oil spills, scum, dumpsites, or vegetation dieback from pollutants like herbicide runoff.	Water contamination, reduced vegetation health, potential fish and wildlife mortality, and long-term soil toxicity.
	Nutrient Excess	Pervasive algal growth or enhanced vegetation growth compared to neighbouring areas.	Destabilization of plant roots, tree toppling, smothering of roots by algae, and reduced ecosystem stability.
	Fire Scorch	Presence of fire damage to vegetation.	Destruction of habitat, increased vulnerability to erosion, and loss of ecosystem services such as carbon sequestration.
	Weed Smother	Presence of invasive weeds overtaking native vegetation.	Replacement of native species, habitat loss, increased fire risk, and changes to ecological processes.
Climate stressors	Storm Damage	Broken trees, defoliated areas, and general forest damage from high winds or cyclones.	Habitat destruction, increased vulnerability to erosion, loss of biodiversity, and long-term recovery challenges.
	Shore Erosion	Fallen trees along shorelines and steep banks where land has eroded.	Loss of habitat, shoreline retreat, reduced protection from storms, and increased sedimentation affecting downstream ecosystems.
	Root Burial	Roots covered in sediment or dieback with signs of sediment deposition.	Mangrove dieback, loss of breathing roots, reduced plant stability, and long-term habitat loss.
	Inner Fringe Collapse	Seaward mangrove fringe excessively submerged, observed as canopy gaps or irregular dieback.	Habitat loss, reduced shoreline protection, increased vulnerability to sea-level rise and storm surges.
	Channel Bank Erosion	Fallen trees along tidal channels and steep eroded banks.	Loss of mangrove habitat, increased sedimentation in tidal channels, and destabilization of banks.
	Pan Scouring	Sheet erosion in salt pans or missing saltmarshes.	Loss of saltmarsh habitat, increased sediment transport into tidal systems, and reduced habitat quality for estuarine species.
	Ecotone Negative Shift	Lines and patches of dead mangrove vegetation along upper saltmarsh transition zones.	Loss of mangrove habitat, expansion of saltmarsh areas, and changes to wetland zonation.
	Ecotone Positive Shift	Young mangrove vegetation encroaching upon saltmarsh or terrestrial ecosystems.	Gain of mangrove habitat, loss of saltmarsh or terrestrial vegetation, and shifts in ecosystem composition.
	Depositional Gain	Presence of young trees on recently expanded sediment banks.	Gain of mangrove habitat, but long recovery time for full ecosystem services.
	Terrestrial Retreat	Dieback of terrestrial vegetation due to excessive submergence and saltwater intrusion.	Loss of terrestrial vegetation, potential expansion of mangroves, and increased erosion of upland areas.
	Light Gaps	Circular gaps in canopy cover, possibly from lightning strikes or dieback.	Natural forest regeneration process, though increased storm frequency may fragment ecosystems.
	Altered Hydrology (Natural)	Dead or dying trees accompanied by a naturally formed impoundment.	Disruption of natural water flows, loss of mangrove trees, and reduced ecosystem function.

reproducibility across regions, with severity then scored using the criteria in Table 1.

2.3. Potential rehabilitation sites

While surveying the condition and threats along the coastline, sites that appeared suitable for rehabilitation under the Australian Blue Carbon Method were identified and screened further for rehabilitation suitability and prioritization. Further screening including assessment of the potential tidal inundation area, identifying tidal barrier locations, anticipated rehabilitation endpoint ecosystem as either saltmarsh or mangrove, long-term changes in mangrove health where adjacent to tidal barriers (Duke et al., 1998), along with the land tenure status and any regulatory assessment triggers (as for May 2023). Together, this information is useful for rehabilitation agencies as they appraise and prioritise sites for further rehabilitation suitability.

The likely extent of tidal inundation following the removal of tidal barriers was mapped manually by using contours at the same elevation as the landward edge of nearby unconstrained tidal wetlands, informed by the Queensland highest astronomical tide (HAT) mapping (State of Queensland, 2013). Barriers, such as tidal gates or earth bund walls, were identified either through direct observation from surveys or inferred from digital elevation models (DEMs). These models revealed raised ridges in the flow paths of depressions and waterways, indicating potential barrier locations. DEMs for each parcel were sourced from the latest 1 m resolution LiDAR imagery available through the Queensland Government (State of Queensland, 2021), and flow paths identified (minimum hydrological response unit size of 1 ha) using QGIS and SWAT+ (Bieger et al., 2017; Dile et al., 2016; QGIS.org, 2020).

Endpoint ecosystem composition was coarsely assigned as being either mangrove-dominated, saltmarsh-dominated, or a mangrove-saltmarsh mix (Duke et al., 2019). Designations were estimated by determining the highest elevation of nearby seaward mangrove zones and comparing with elevations observed within each potential rehabilitation site. Designation was also informed by relationships between long-term regional rainfall in proportion with mangrove to saltmarsh-saltpan extent from Duke et al. (2019). As reported, rainfall influences the relative areas of mangrove and saltmarsh–saltpan in tropical tidal wetlands, with drier areas having a greater proportion of saltmarsh–saltpan, and wetter areas with relatively larger mangrove areas. To further examine the role of rainfall in estimating the proportional occupation of mangroves in the upper-tidal zone, regression

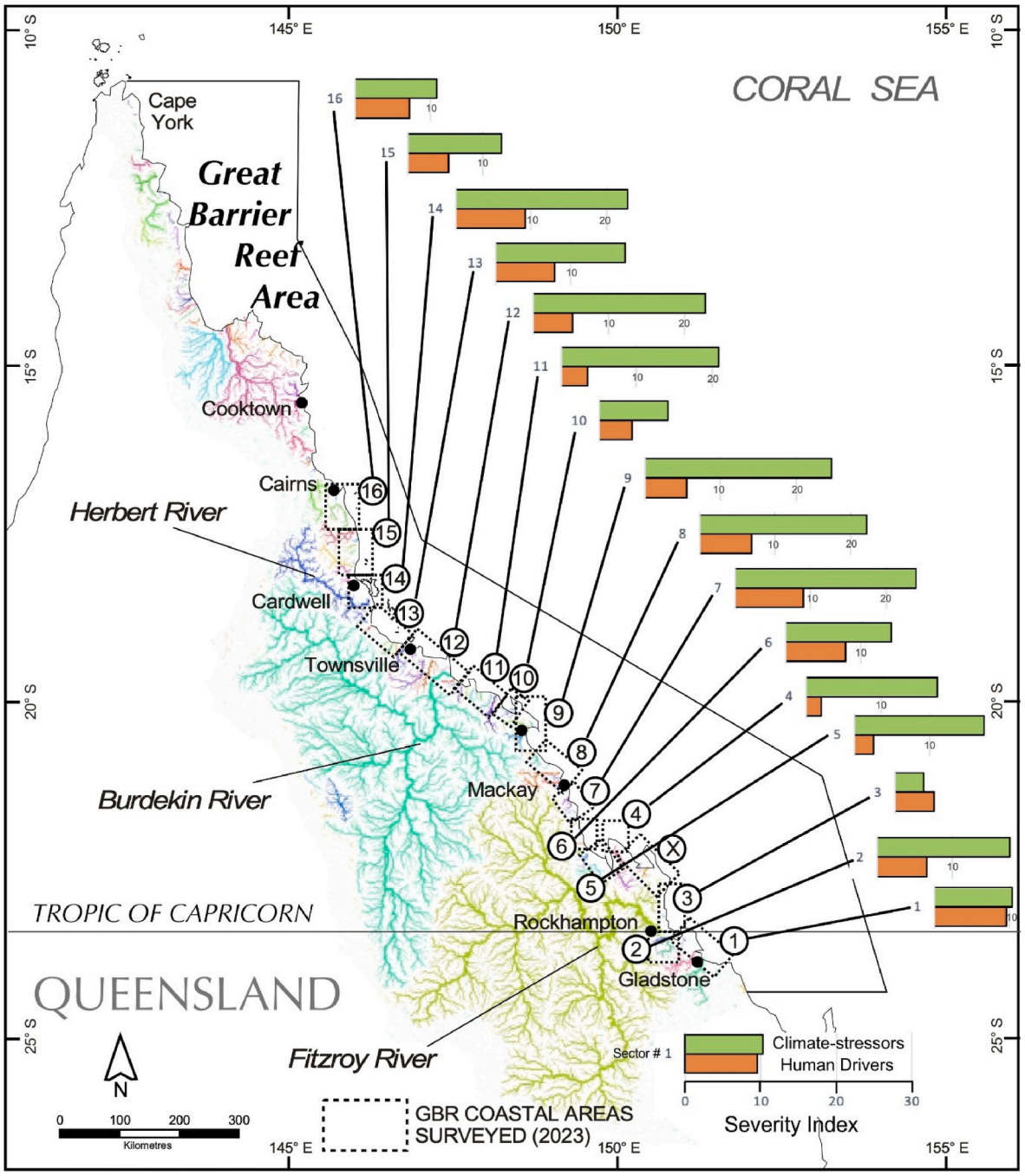

Fig. 1. The sum of severity scores for climate-stressors and human drivers of change for 16 sub-regions of the coastline between Rodd's Bay and Cairns (see Fig. 3), surveyed using oblique imaging in May 2023. Waterways in different colours indicate different catchments. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

analysis was used to assess the relationship between the ratio of the mangrove–saltmarsh ecotone (MSE) elevation to highest astronomical tide (HAT) elevation at potential rehabilitation sites and the long-term mean annual rainfall (mm) as predicted by the baseline WorldClim v2 dataset (Fick and Hijmans, 2017).

Given that the health of mangroves adjacent to potential rehabilitation sites could influence rehabilitation efficacy through propagule availability, the long-term trend in the normalized difference vegetation index (NDVI) of mangroves was assessed at sites adjacent to tidal barriers (Table S1). For sites with multiple tidal barriers near mangroves, multiple NDVI points were evaluated, with one point assigned per barrier approximately 100 m downstream from the centre of the barrier. Sites without adjacent mangroves were excluded from the assessment. NDVI, a well-established indicator of vegetation health, quantitatively measures canopy density and photosynthetic activity. Mathematically, the NDVI is the ratio of the difference between the near-infrared (NIR) and red (R) spectral reflectance to their sum.

$$NDVI = (NIR - R) / (NIR + R)$$

Where.

- NIR is the reflectance in the near-infrared region, which healthy vegetation strongly reflects.

Table 2
Estuarine survey image point assessment criteria and classification for tidal wetland habitat and environmental condition descriptors. This scoring methodology was developed by Duke and Mackenzie (2018).

Extent of impact on tidal wetland area	The extent of tidal wetlands impacted determined by the proportional area showing impact	Impact extent 0. Non-mangrove 1. 1–10 %. Relatively unaffected. 2. 10–30 % – Minor. 3. 30 %–60 % – Moderate. 4.60 %–90 % – Severe. 5. >90 % - Extreme.	*Extent score* – extent of impact (% of coastline length) where tidal wetlands occur.
Severity of impact on tidal wetland area	The severity of tidal wetlands impacted as determined by the degradation state observed.	Impact severity 1. No impact 2. Minimal – present but no observable effect 3. Minor – recovery within less than one year 4. Moderate – recovery over 1–2 years 5. Major – recovery over 2–10 years 6. Severe – recovery unlikely – collapse/replace	*Severity score* – severity of impact to tidal wetland.

- R is the reflectance in the red region, which healthy vegetation strongly absorbs due to chlorophyll.

NDVI values, range from −1 to 1, with higher NDVI values indicating healthier vegetation. Declining trends could signal less favourable conditions for rehabilitation, such as reduced propagule dispersal or other stressors affecting mangrove recruitment, while stable or increasing trends would support rehabilitation efforts.

Maximum monthly NDVI values were derived at points using cloud-free Tier 1 imagery from the U.S. Geological Survey Landsat Program (30 m resolution), spanning March 1987 to December 2023, from Landsat 5, 7, and 8 (WRS Paths 93, 94, and 95; Rows 72, 73, and 74) (Loveland and Dwyer, 2012). Data were accessed and processed using Google Earth Engine (Gorelick et al., 2017). Trends were detected using the 'beast' function in the Rbeast package v0.9.4 in R 4.3.3 (R Core Team, 2024; Zhao et al., 2022), using default values and seasonal patterns modelled as an annual harmonic curve, which implements the methods described in Zhao et al. (2022). The annual harmonic is necessary given the seasonality of mangrove canopies. Rbeast calculates the probability of a trend by decomposing a time series into its trend, seasonal, and noise components using a Bayesian framework (Zhao et al., 2022). This probability is derived by assessing how much of the variability in the time series is attributed to a consistent trend. The mean probability of a negative trend was calculated for each site and interpreted using the IPCC Likelihood Scale: probabilities above 0.66 indicate a trend is 'likely,' above 0.9 as 'very likely,' and above 0.99 as 'virtually certain' (Mastrandrea et al., 2010).

3. Results

3.1. Sea level and rainfall

Between 1985 and May 2023, tide-gauge records indicated sea level rise along the GBR coast, with mean trends ranging from about 2.1 mm yr^{-1} in Cairns ($R^2 = 0.07$, $p < 0.001$) to 3.8 mm yr^{-1} in Gladstone ($R^2 = 0.23$, $p < 0.001$), and intermediate rates of 3.7 mm yr^{-1} in Townsville and 3.4 mm yr^{-1} in Mackay (Fig. 2). Rates of sea level rise in the Gladstone area were identified as increasing notably in the last decade (Duke and Canning, 2024a). Annual rainfall showed no long-term trends across the GBR, with weak and non-significant slopes ranging from a slight increase in Townsville (+10.4 mm yr^{-1}, $p = 0.19$) and Cairns (+3.4 mm yr^{-1}, $p = 0.64$) to small decreases in Mackay (−4.2 mm yr^{-1}, $p = 0.62$) and Gladstone (−2.0 mm yr^{-1}, $p = 0.63$; Fig. 2).

3.2. Coastline threats

Overall, almost the entire coastline between Rodds Bay (Gladstone) and Cairns showed extensive impacts from climatic stressors (Fig. 1). Most notably, these impacts were dominated by shoreline erosion, bank erosion, and terrestrial retreat – all three are associated with rising sea levels, concordant with climate change (Fig. 3). In addition to sea level rise impacts, parts of the coastline were severely impacted by storm damage, particularly between Saddleback Island and Bowen and between Taylors Beach and Cardwell (also see Duke et al., 2024). Within the human drivers, bank erosion and depositional gain were most notable, with a high level of bank erosion observed in the Fitzroy River Estuary (Fig. 3).

3.3. Potential rehabilitation sites

Overall, 52 potential rehabilitation parcels were identified across 28 clustered locations, totalling approximately 17,255 ha with parcel area ranging from 2 ha to 4,350 ha (Fig. 4). Tidal bund walls were the primary saltwater exclusion structures at all sites south of Ingham, while sites between Ingham and Cairns used both bund walls and tidal gates. No negative trends in NDVI were considered likely at any of the long-term NDVI locations assessing mangroves directly adjacent to barriers (Table S1).

Within the upper intertidal zone (between mean sea level and highest astronomical tide level), northern sites are anticipated to be dominated by seaward fringing mangroves with greater saltmarsh contributions in southern areas (Fig. 4). Across climate zones, the mangrove-saltmarsh ecotone elevations in the subtropics (zones 1–7) ranged from 1.0 to 3.5 m (median = 2.8 m), in the dry tropics (zones 7–12) the elevations ranged 1.1–2.0 m (median = 1.4 m), and in the wet tropics (zones 13–16) elevations ranged 1.6–1.8 m (median = 1.6 m). Furthermore, the elevation of the mangrove-saltmarsh ecotone (MSE) relative to the elevation of the highest astronomical tide (HAT) was strongly correlated with mean annual rainfall (quadratic regression: $R^2 = 0.81$, $F_{2,50} = 108.6$, $P < 0.01$).

4. Discussion

4.1. Threats to mangroves and saltmarsh

Across the southern Great Barrier Reef (GBR) region, climatic threats dominated and were widespread, mostly indicated by shoreline erosion, bank erosion, shifts in ecotone position, and terrestrial retreat caused by rising sea levels and damage from cyclone coastal crossings. The types of abrupt damage inflicted by tropical cyclones on mangrove forests include canopy defoliation, limb breakage, stem breakage, below ground root exposure, above-ground root burial, and tree death (Krauss and Osland, 2020). For the GBR coast, the most severe cyclone damage was observed between Taylor's Beach and Cardwell, near Hinchinbrook

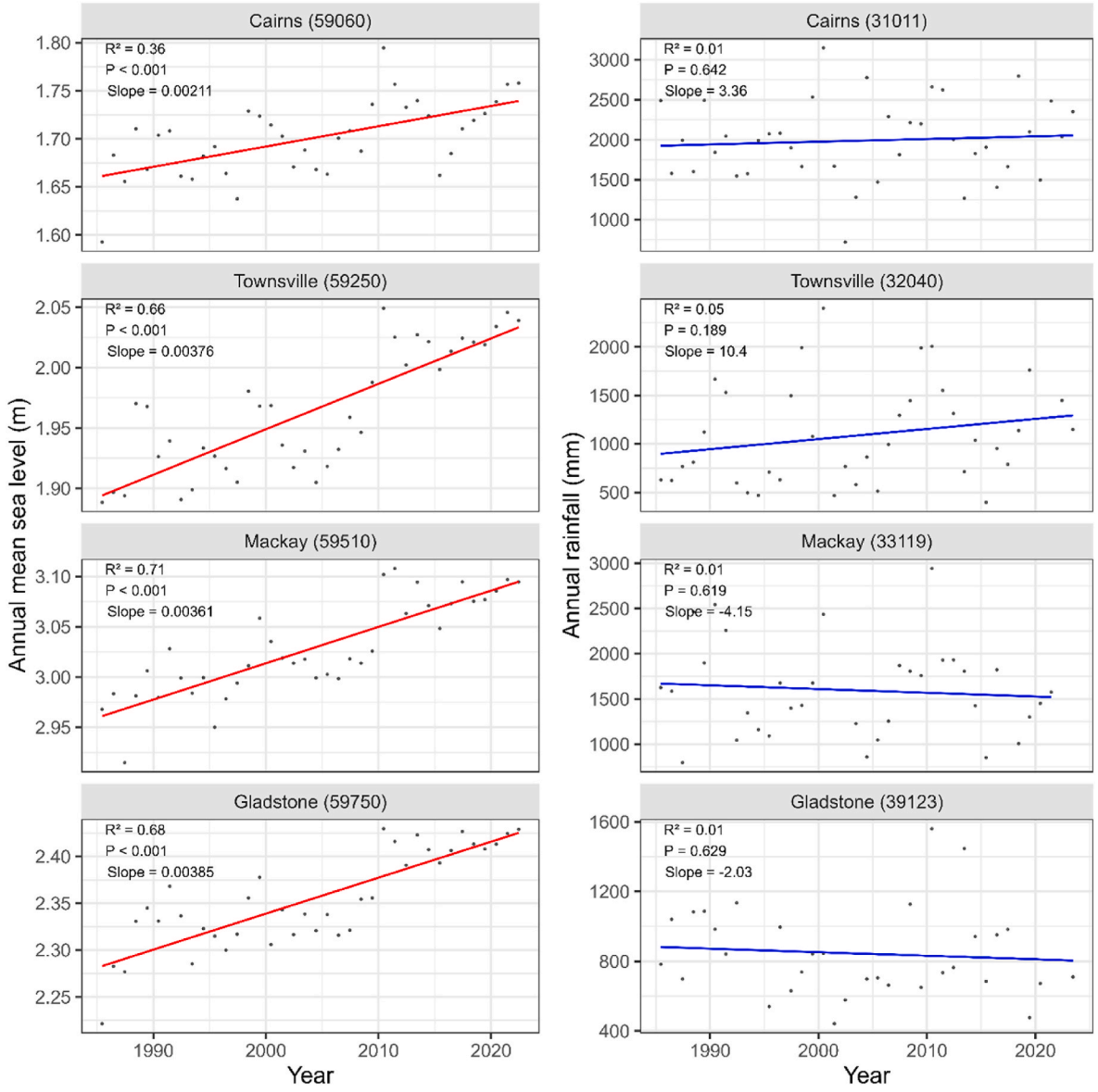

Fig. 2. Long-term trends in annual mean sea level (left panels) and annual rainfall (right panels) for Cairns, Townsville, Mackay and Gladstone, 1985–2023, with fitted linear regressions overlaid (red for sea level, blue for rainfall). (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

Island, caused by the category 5 Severe Tropical Cyclone Yasi in 2011 (Duke et al., 2024). This was followed by severe cyclonic damage between Saddleback Island and Bowen, between Hay Point and St Helen's Bay, and between Cardwell and Innisfail, caused by cyclones Aivu (category 3 in 1989), Debbie (category 4 in 2017), and Larry (category 4 in 2006) respectively. These impacts were directly observed in our 2023 aerial survey, with large areas of canopy loss, flattened trees and limited regrowth still evident. Our field-based observations were later supported by areas with slow or little recovery in canopy cover using long-term Landsat NDVI analysis by Duke et al. (2024). It is alarming that the more severely damaged areas showed little recovery, despite instances with substantial time since impact and impacted areas having healthy mangroves nearby that can supply propagules. Sediment subsidence from organic matter degradation and erosion may shift areas below the mangrove habitable zone. Alternatively, fallen trees might hinder propagule establishment, with frequent smaller climatic disturbances disrupting juvenile tree recovery. Further assessment of elevational changes in poorly recovering areas and regular temporal monitoring is warranted to appraise the impacts of smaller scale disturbances. Comparable climate-related impacts have been reported globally, including storm-driven mangrove loss in the Caribbean and Gulf of Mexico (Carranza Ortiz et al., 2018; Fickert, 2018; Serrano-Rubio et al., 2020; Tomiczek et al., 2020) and widespread shoreline retreat in Southeast Asia where sea-level rise interacts with coastal development pressures (DasGupta and Shaw, 2013; Lovelock et al., 2015; Ong and Ellison, 2021). Addressing these threats will require coordinated national climate policies to mitigate rising sea levels and support rehabilitation projects that enhance ecosystem resilience to storm impacts and erosion. Rehabilitation initiatives could focus on immediate planting mangroves in areas prone to erosion and storm surges to stabilize shorelines and reduce sediment smothering.

Regarding the human drivers, altered hydrology from tidal bund walls, constructed barriers used to prevent tidal flows into certain areas, often for agriculture or freshwater retention, were primarily dominant in the drier southern areas (south of Townsville), whereas agricultural encroachment was more prevalent in the wet tropics (between Townsville and Cairns). Queensland's tidal bund walls were primarily constructed circa 1950s to retain freshwater and raise ponded pastures to

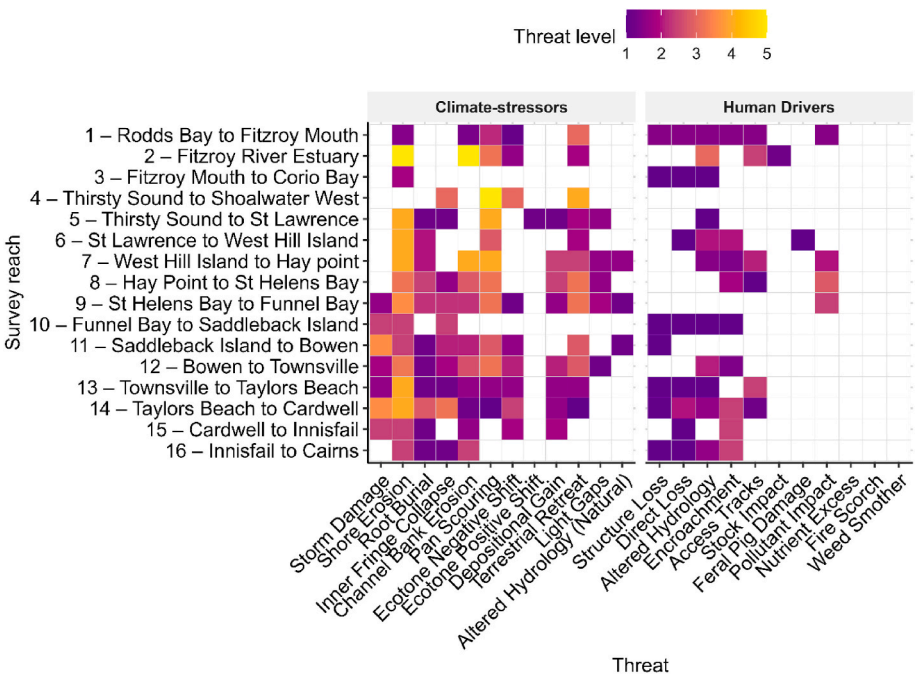

Fig. 3. The threat severity scores for individual threats identified for 16 sub-regions of the coastline between Rodd's Bay and Cairns (Fig. 1), surveyed using oblique imaging in May 2023.

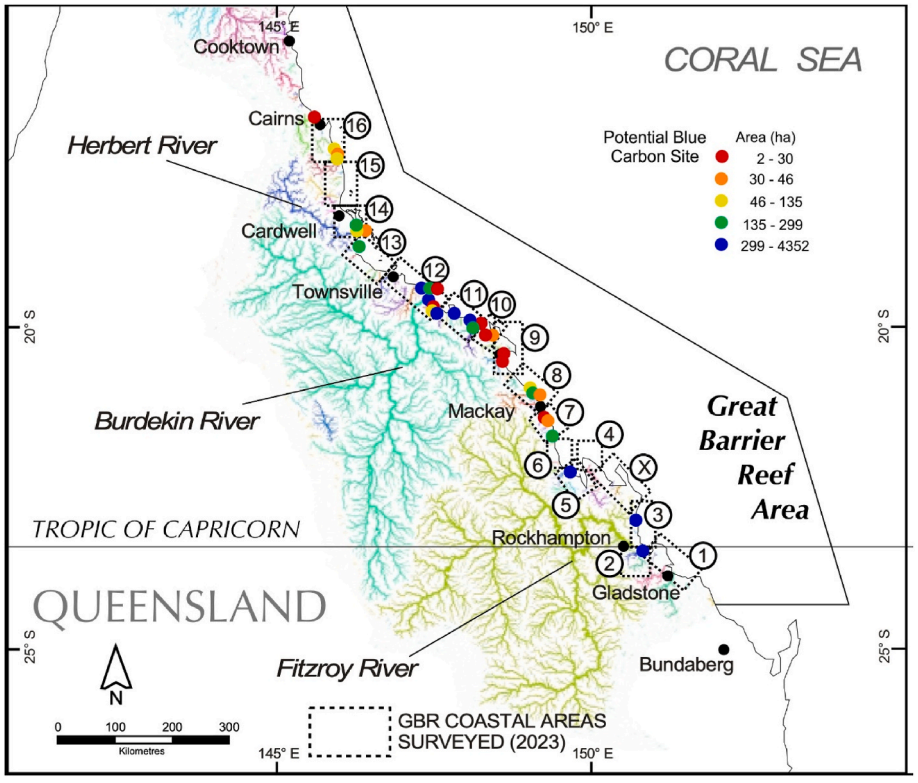

Fig. 4. The locations and approximate area (Ha) of potential blue carbon rehabilitation sites identified 16 sub-regions of the coastline between Rodd's Bay and Cairns. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

supplement late dry season forage when terrestrial pastures lack growth. Salt marsh damage from 4WD vehicles, often resulting in considerable erosion, was most prevalent near the southern and drier urban areas (e.g., near Rockhampton and Mackay). Pollutant impacts, suspected to be herbicide runoff, were largely observed within the mouths of the Fitzroy River, Pioneer River, Rocky Dam creek and Murray Creek. To mitigate these impacts, the increased uptake of sustainable agricultural practices, such as buffer zones and controlled runoff management, are essential. Rehabilitation efforts should also include targeted hydrological rehabilitation strategies to address disruptions caused by bund walls and

improve water flow in impacted areas. Vehicle access to salt marshes would need to be limited if salt marshes are to recover and reduce pan scouring and hydrological disruptions from wheel ruts. In the areas of most severe urban and agricultural encroachment, long term land use planning and incentives are needed to facilitate a managed retreat to allow mangroves to shift landward as sea levels rise (Leo et al., 2019; O'Donnell, 2022). These threats highlight the need for a comprehensive coastal management strategy that emphasizes the protection of existing ecosystems while prioritizing rehabilitation efforts to address current impacts and strengthen the long-term resilience of mangrove and saltmarsh habitats against evolving climatic and anthropogenic pressures (Powell et al., 2019).

4.2. Opportunities and challenges in rehabilitation

The identification of 52 potential rehabilitation parcels, totalling 17,255 ha, highlights the many rehabilitation opportunities along the GBR coastline. The range of parcel sizes, from 2 ha to 4,350 ha, further supports projects at a range of scales. The assessment's use of high-resolution elevation data and NDVI trends provided detailed mapping of tidal barriers and endpoint ecosystems. NDVI trends are a useful indicator of canopy condition and recovery potential, helping identify sites where adjacent mangroves remain healthy enough to supply propagules, but they do not capture below-ground stressors such as sediment condition or hydrological alteration, which must also be considered when assessing rehabilitation feasibility. Northern parcels were identified as mangrove-dominated, while southern sites were more likely to support mixed mangrove-saltmarsh systems, reflecting regional climatic differences in long-term rainfall (Duke et al., 2019). Broadly, sites with higher long-term rainfall are likely to have proportionally greater mangrove occupation of the upper-tidal zone than sites with comparatively lower rainfall, and this strong pattern was observed across mangroves adjacent to potential sites (Fig. 5).Sea water intrusion modelling that accounts for changes in long-term rainfall, freshwater runoff and sea level rise with climate change could be used to further improve estimates of potential coverage by mangrove and saltmarsh and how this might change over time (Karim et al., 2021; Rowland et al., 2023). Additionally, long-term rehabilitation success could be affected by unassessed factors like sedimentation or propagule availability (Van der Stocken et al., 2019; Zimmer et al., 2022). In particular, sediment characteristics such as particle size, stability, and geochemistry strongly influence propagule establishment and long-term accretion dynamics (He et al., 2022; Le Minor et al., 2019; van Hespen et al., 2022). These properties may also shift over time, for example where bund walls trap fine sediments and associated contaminants from upstream erosion, reinforcing the need for site-specific sediment assessments as part of rehabilitation feasibility studies. Other local factors, including hydrodynamic energy, substrate type, and biotic and abiotic stressors such as predation, salinity, nutrient availability, and flooding regime, can also act as important filters on mangrove establishment and should be incorporated into site-specific feasibility assessments (Van der Stocken et al., 2019).

In Australia, interest in removing tidal barriers as a method to facilitate rehabilitation is growing in popularity, largely attributed to the recent introduction of carbon payments via the ACCU Scheme, which incentivizes projects that enhance 'blue carbon', carbon sequestered in coastal and marine ecosystems like mangroves, saltmarsh, and seagrasses where tidal barriers are permanently removed. Comparable blue carbon frameworks are emerging internationally, including in the United States, Kenya and Indonesia, linking carbon markets with tidal wetland rehabilitation (Needelman et al., 2018). Beyond market-based mechanisms, East African initiatives highlight how restoration outcomes depend strongly on governance and community engagement. In Mozambique, community-based management combined with replanting has reversed localised degradation by creating alternative livelihoods and local enforcement norms, though weak institutional capacity has limited sustained impact (Macamo et al., 2024). In Madagascar, integrating traditional ecological knowledge with scientific monitoring has achieved high propagule survival, demonstrating the value of local stewardship where ecological and cultural knowledge align (Ravaoarinorotsihoarana et al., 2023). Across West-Central Africa, multi-level governance approaches have slowed mangrove loss but continue to be undermined by competing land uses and insufficient regulation, underscoring the challenge of scaling up restoration in developing contexts (Feka and Morrison, 2017; Quenum et al., 2024). These examples illustrate that technical interventions, such as barrier

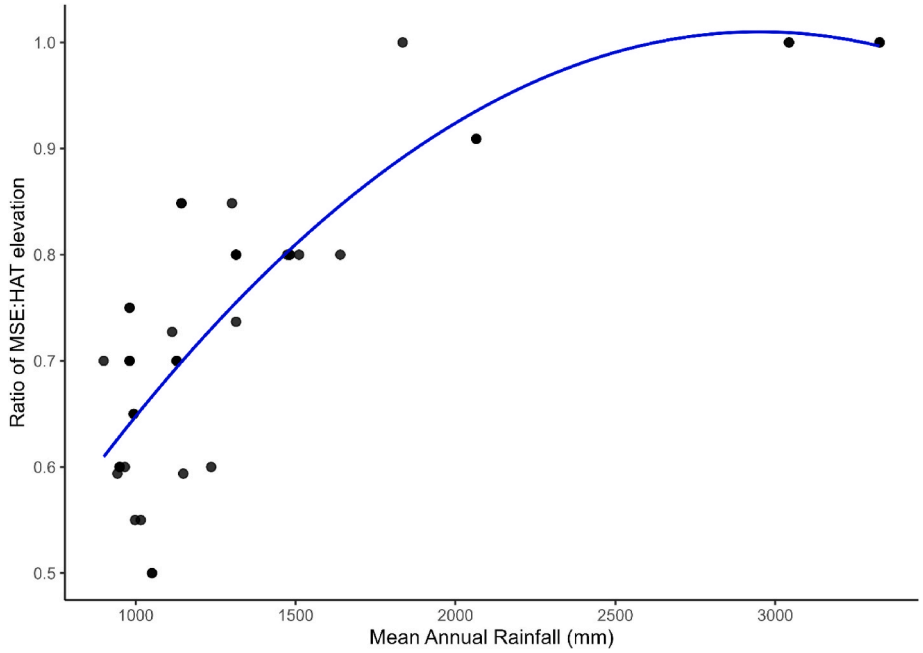

Fig. 5. Ratio of mangrove–saltmarsh ecotone (MSE) elevation to highest astronomical tide (HAT) elevation at potential rehabilitation sites, plotted against mean annual rainfall (mm) with a fitted quadratic regression ($R^2 = 0.81$, $F_{2,50} = 108.6$, $P < 0.01$).

removal, must be coupled with supportive governance, cultural legitimacy, and sustained monitoring if they are to achieve durable ecological outcomes. Removing tidal barriers can restore tidal inundation, but ecosystem recovery is neither immediate nor guaranteed (Su et al., 2021; Zimmer et al., 2022), often requiring decades longer than the 25-year blue carbon accounting period and there are few examples of success with adequate long-term monitoring (Duke and Wolanski, 2000; Gatt et al., 2022; Twomey et al., 2024).

The poor recovery of damage from Cyclone Aivu (1989) observed in this study, for example, highlights how even where tidal activity is unrestricted recovery can be challenging. Like many dam removal projects, the removal of tidal bund walls is likely to face additional challenges due to the mobilisation of accumulated sediments from land runoff, which may deposit downstream and release otherwise stored carbon from the sediment (e.g., Ritchie et al., 2018; Rubin et al., 2023). Sediment erosion during tidal bund removal risks smothering downstream ecosystems and exposing acid sulfate soils, causing acidic discharge. For example, over 1.5 m of sediment behind bunds in Trinity Inlet was resuspended post-breach, with mud transported along the coastline for six months annually due to trade winds and tidal activity (Wolanski and Duke, 2002). Furthermore, tidal bunds can significantly alter sediment geochemistry by changing oxygen availability and the chemical forms of elements, leading to conditions that influence nutrient cycling and potential release of toxic element, as observed in Trinity Inlet, where reintroducing seawater caused a three-phase shift: sulfur-dominated conditions initially sequestered trace elements, drainage and oxidation led to severe acidification and leaching of elements like Al and Zn, and eventual seawater inundation restored reducing conditions, promoting arsenic and chromium enrichment (Keene et al., 2010). Comparable challenges have been reported from tidal wetland rehabilitation, particularly in the US and Europe, where riverine barrier removal often mobilised fine sediments and contaminants (Bednarek, 2001; Boardman and Foster, 2023; Foley et al., 2017). To mitigate these risks, it is essential to assess sediment characteristics and volumes before bund removal. Incremental removal - gradually dismantling tidal barriers to avoid abrupt sediment shifts - synchronized with mangrove colonization can help minimize disturbances and ensure more stable and effective ecosystem rehabilitation outcomes.

While terrestrial rehabilitation projects often leverage tree planting, successful mangrove rehabilitation projects using propagule are rare given the low survival rate. A review of 74 restoration sites in Southeast Asia found an average propagule survival rate of 20 %, with a median of just 10 % (Wodehouse and Rayment, 2019). Furthermore, 66 % of these projects recorded survival rates below 20 %. Projects in the mid-mangrove zone achieved relatively higher success, averaging a 30 % survival rate, whereas efforts on mudflats—accounting for 32 % of all attempts—had a strikingly low survival rate of only 1.4 %. These low outcomes have largely been attributed to mismatched species and site conditions, particularly at high and low elevations. For example, *Rhizophora* spp. were used in 65 % of cases, including on mudflats where their natural establishment is highly unlikely. Many projects prioritized meeting area or planting quotas over ensuring plant survival, often targeting large, undisturbed areas. Conversely, sites with even minimal natural regeneration consistently reported above-average success, suggesting that only about 16 % of the planting efforts were genuinely necessary. Before investing in propagule planting, it may be beneficial to first focus on addressing threats that could inhibit colonization and monitor natural recovery for 1–2 years before recruiting more interventionist strategies (Bosire et al., 2008; Lewis et al., 2019; Zimmer et al., 2022).

4.3. Key considerations for rehabilitation planning

When planning mangrove rehabilitation projects, it is essential to assess local and regional threats, natural site conditions, and potential trade-offs to ensure long-term success. Key considerations include identifying change drivers, damage severity, and future risks, and determining if they can be managed locally (Lewis et al., 2019; Zimmer et al., 2022). Sites with evidence of natural mangrove regeneration may require less intervention, whereas areas lacking regeneration might indicate a greater need for rehabilitation. The site's elevation and inundation regime must align with mangrove growth requirements for specific zones, as unsuitable conditions, such as mudflats or standing water, often result in poor establishment. In this study, the mangrove–saltmarsh ecotone was clustered at ~1.6 m in the wet tropics (range 1.6–1.8 m), more variable in the dry tropics (median 1.4 m, range 1.1–2.0 m), and higher in the subtropics (median 2.8 m, range 1.0–3.5 m). Mangroves are most likely to establish between mean sea level and these upper elevation limits. Within this zone, species show distinct vertical zonation, for example, *Avicennia marina* often dominates the landward fringe and saltmarsh edge, while *Rhizophora* and *Bruguiera* species are typically restricted to lower intertidal areas with regular tidal flushing. Low-energy, silty substrates with regular inundation favour propagule anchoring and seedling survival, whereas exposed sandy or permanently ponded areas generally result in poor establishment. Rehabilitation sites should also be free of large debris that could pose risks during tidal movements, and social or environmental obstacles, such as grazing, boat activity, or harmful practices, must be mitigated to support young mangrove growth (Lewis et al., 2019; Zimmer et al., 2022). Rehabilitation must consider the ecological value of artificial habitats (Canning and Waltham, 2021), like the Tedlands complex, which supports diverse bird species, barramundi, and the threatened southern purple-spotted gudgeon (Canning et al., 2021a; Sabatino, 2020). Efforts to restore tidal conditions in such areas should weigh the potential loss of biodiversity against the benefits of blue carbon abatement.

Once rehabilitation sites and objectives have been established, the rehabilitation process involves evaluating site-specific conditions, addressing degradation drivers, and implementing strategies to rehabilitate tidal wetland ecosystems with a focus on long-term resilience (Lewis et al., 2019; Zimmer et al., 2022). Effective rehabilitation requires assessing current and active threats to tidal wetlands and ensuring strategies are in place to manage them, as unmitigated threats can undermine success. Human drivers can often be managed locally, but addressing climate-related drivers necessitates broader interventions, including national climate policies. Rehabilitation efforts must account for ongoing climate change impacts by selecting species suited to future conditions, allowing for landward migration, planting at elevations that accommodate sea-level rise, and establishing vegetative buffers to mitigate runoff and fire impacts from adjacent agricultural and urban areas (Akram et al., 2023; Gilman et al., 2008).

Successful rehabilitation requires the active participation and buy-in of local stakeholders, including landowners, local governments, and community members (Moraes, 2019; Saunders et al., 2024). Early engagement fosters a sense of ownership and ensures that cultural, social, and economic considerations are integrated into rehabilitation planning. Providing training and resources for stakeholders to participate in monitoring and maintenance not only enhances rehabilitation outcomes but also builds local capacity for sustainable ecosystem management. Additionally, involving stakeholders in decisions about rehabilitation priorities can help balance ecological goals with community needs, such as income generation and coastal protection.

To maximize the impact of rehabilitation efforts, a structured framework for prioritizing sites is recommended (Howie et al., 2024; Piccolo et al., 2024; Rogers et al., 2023). This framework should evaluate ecological urgency, such as areas most affected by climatic and anthropogenic threats, alongside the feasibility and potential for recovery and the socio-economic benefits of rehabilitation. This requires consideration of biophysical, social, governance, logistical, and resource factors to ensure site selection aligns with both ecological priorities and socio-economic objectives, maximizing the likelihood of successful rehabilitation outcomes. Piccolo et al. (2024) highlighted the

importance of integrating these multiple feasibility factors into the prioritization of rehabilitation actions, showing how cyclone risk and land tenure along Queensland's coastline could affect rehabilitation feasibility. Prioritizing sites with the highest potential for achieving long-term ecosystem resilience and community benefits ensures the efficient allocation of resources and maximizes rehabilitation outcomes.

Long-term monitoring is crucial for evaluating outcomes and ensuring that rehabilitation achieves its ecological, social, and functional objectives, which may take decades to materialize. Success criteria should be clearly defined at the outset, with measurable benchmarks tailored to the project's goals, such as timber production, biodiversity enhancement, or fisheries support. Given a widespread lack of adequate mangrove rehabilitation monitoring, The Mangrove Restoration Tracker Tool (Gatt et al., 2024), could be used to guide monitoring and record outcomes. Regular monitoring, particularly during the first five years, allows for adaptive management to address challenges as they arise. Adaptive management frameworks should be integrated into rehabilitation efforts, enabling managers to respond to emerging threats such as sedimentation, propagule failure, or shifts in climate conditions (Peirson et al., 2015).

Ensuring long-term success also requires sustained funding mechanisms, proactive stakeholder engagement, and integration with supportive policy frameworks (Canning et al., 2021b; Vanderklift et al., 2019). Building local capacity through training and participation in monitoring fosters stewardship and ensures ongoing management. Rehabilitation plans should incorporate future climate projections, including sea-level rise and storm intensity, to maintain site resilience over time. Addressing the threats to mangroves and saltmarshes in the southern GBR region requires coordinated efforts that balance ecological priorities with socio-economic considerations. By aligning rehabilitation strategies with broader climate adaptation policies and securing long-term support, these initiatives can safeguard the ecological, cultural, and economic value of tidal wetlands in the face of growing environmental pressures.

CRediT authorship contribution statement

Adam D. Canning: Writing – review & editing, Writing – original draft, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Norman C. Duke:** Writing – review & editing, Supervision, Project administration, Methodology, Investigation, Formal analysis, Data curation, Conceptualization.

Declaration of competing interest

The authors declare the following financial interests/personal relationships which may be considered as potential competing interests: Adam Canning reports financial support was provided by Greening Australia Ltd. If there are other authors, they 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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Data availability

Data will be made available on request.

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