9.3 Mangrove forest extent and quality

Author, affiliation: Andrew Swales (NIWA), Carolyn Lundquist (NIWA)

Citation for this chapter: Swales, A., Lundquist, C. (2024). Mangrove forest extent and quality. *In:* Lohrer, D., et al. *Information Stocktakes of Fifty-Five Environmental Attributes across Air, Soil, Terrestrial, Freshwater, Estuaries and Coastal Waters Domains*. Prepared by NIWA, Manaaki Whenua Landare Research, Cawthron Institute, and Environet Limited for the Ministry for the Environment. NIWA report no. 2024216HN (project MFE24203, June 2024). [https://environment.govt.nz/publications/information-stocktakes-of-fifty-five-environmental-attributes]

State of knowledge of the "Mangrove Habitat Extent and Quality" attribute: Good / established but incomplete – general agreement, but limited data/studies

Our overall assessment of the state of knowledge for the attribute Mangrove Forest Extent and Quality is good/established but incomplete. There is generally good information on the broadscale spatial extent and distribution of mangrove forest in New Zealand's northern estuaries. Information sources include aerial photographic surveys extending back to the 1930s, with progressive improvements over time (i.e., black & white colour, scale, accuracy). Present-day satellite coverage (e.g., ESA Sentinel) provides high-resolution multi-spectral products for mapping mangrove forest extent and some aspects of attribute quality. LiDAR coverage of intertidal habitats has also steadily improved in frequency, resolution, and accuracy. Remote sensing may not, however, capture incremental changes/low density mangrove stands on forest fringes nor adequately capture the recent movement of mangrove into saltmarsh habitat. The quantity and frequency of on-the-ground monitoring of mangrove forest characteristics and the quality thereof varies between regions, as do the variables measured. Research interest in NZ mangrove forest systems has grown substantially since the 1970s and particularly over the last 20 years. The knowledge generated by this research encompasses the biophysical and social sciences and research and applied studies on coastal wetland blue carbon and biodiversity, which has gathered pace within the last decade. Understanding of the drivers of NZ mangrove forest development and ecosystem function are good, although much of this work has been conducted at a handful of locations. Knowledge gaps remain in aspects of mangrove forest characteristics/quality. Understanding of the future resilience of NZ mangrove forests to climate warming and relative sea level rise across the range of environmental setting where they occur is at an early stage.

Part A—Attribute and method

A1. How does the attribute relate to ecological integrity or human health?

Mangrove habitat expansion in New Zealand's northern estuaries from the mid-1800s onwards has largely occurred due to the expansion and vertical accretion of (unvegetated) intertidal flat habitat, suitable for mangrove colonisation. This accelerated estuary infilling process is driven by catchment sediment loading associated with large-scale catchment deforestation and conversion to pastoral agriculture and more recent land-use intensification (e.g., production forestry). In this sense, the general pattern of mangrove habitat expansion in many estuaries is a symptom of excessive soil erosion during the historical era [e.g., reviews (Horstman et al. 2018; Morrisey et al. 2010; Swales et al. 2020b)]. Although monitoring changes in the extent of mangrove habitat is readily amenable to remote sensing, the range of mangrove in New Zealand, limited to the upper North Island, means that mangrove habitat extent is not a suitable indicator of ecological quality at a national scale. Below we describe the key processes driving mangrove habitat expansion in New Zealand's northern estuaries and their ecological values.

Due to their physiology, mangroves require regular tidal exposure to the air with a maximum hydroperiod (i.e., duration and frequency of inundation) that coincides with mean sea level. NZ mangrove forests are composed of monospecific stands of the grey mangrove (Mānawa, *Avicennia marina* var. *australasica*) and occur south to Kawhia Harbour on the west coast and to Ōhiwa Harbour on the east coast (i.e., 38.1°S). Their national distribution has been controlled by low winter air temperatures, frost frequency, biogeography and oceanography that limit mangrove propagule dispersal [e.g., de Lange & de Lange (1994)] and the limited availability and relative remoteness of suitable estuarine environments south of their present range. Although climate warming may change their potential range (e.g., landward migration facilitated by sea level rise), biogeographic limitations to their southern expansion remain.

In general, the expansion of mangrove habitat in New Zealand's northern estuaries has coincided with adverse effects on the ecological integrity of estuaries. This has largely been driven by the delivery of mud (i.e., particle size < 62.5 micron) associated with increased catchment soil erosion in the historical era. In many estuaries, increased fine sediment loads from catchments as well as ongoing impacts today of legacy fine sediment have resulted in order of magnitude increases in sedimentation rates, shift from sand- to mud-dominated systems and reduction in the areal extent of subtidal habitats. Ecosystem degradation has been characterised by loss of plants and animals sensitive to increased water turbidity, reduced light levels, and smothering by fine sediment deposition (e.g., seagrass meadows, filter-feeding bivalves [Bainbridge et al. 2018; Booth 2019; Inglis 2003; Thrush et al. 2004; Zabarte-Maeztu et al. 2021]). The substrate elevation in many mangrove forests is also close to the upper limit of the present tidal frame (i.e., Mean High Water Spring) so that hydroperiods are short and the opportunity for estuarine fish to utilise mangrove forest is limited. Although many bird species make extensive use of mangrove habitat for roosting, feeding or breeding, there are no New Zealand birds that are exclusively found in mangroves (reviewed in Morrisey et al. 2010). Even in fine-sediment degraded estuaries, mangroves do provide important ecosystem services which include sequestration of fine sediment and associated stormwater contaminants (e.g., heavy metals and carbon [Bulmer et al. 2020; Swales et al. 2002]). Mangrove forests may also reduce the risk of coastal erosion and inundation associated with storm tides (Gijsman et al. 2021; Swales et al. 2015). Mangrove forests provide long-term carbon sequestration, serving as important carbon sinks, and mitigating climate change by reducing overall greenhouse gas

(GHG) emissions (Bulmer et al. 2020; Ross et al. 2024; Suyadi et al. 2020) and there is increasing interest in restoration of mangrove forests as a climate mitigation strategy (Stewart-Sinclair et al. 2024).

In a small number of estuaries that have not experienced large fine-sediment loadings, the ecological integrity of these systems may be enhanced by mangrove forests. Estuaries with relatively small catchments and/or retain areas of indigenous forest landcover fall into this category. These systems may include subtidal and intertidal seagrass meadows and mangrove habitat on sandy substrates (e.g., Rangaunu, Whangateau).

A2. What is the evidence of impact on (a) ecological integrity or (b) human health? What is the spatial extent and magnitude of degradation?

There is substantial weight of evidence that mangrove habitat expansion during the historical era is associated with adverse impacts on the ecological integrity of New Zealand's northern estuaries. This association is coincidental rather than causal, as the evidence indicates that mangrove habitat expansion is symptomatic of excessive fine sediment loading and sedimentation in NZ estuaries (Morrisey et al. 2010; Suyadi et al. 2019; Swales et al. 2020b) in comparison to pre-deforestation/baseline conditions (Suyadi et al. 2019).

A3. What has been the pace and trajectory of change in this attribute, and what do we expect in the future 10 - 30 years under the status quo? Are impacts reversible or irreversible (within a generation)?

Aerial photographic surveys since the late 1930s indicate the rate of mangrove habitat in New Zealand's northern estuaries expansion has averaged 4.1% yr^{-1} (range 0.2–20.2% yr^{-1}) (Morrisey et al. 2010; Suyadi et al. 2019). This expansion of NZ mangrove habitat has occurred at twice the rate as observed in the temperate *Avicennia* forests of southeastern Australia (average 2.1% yr^{-1}, range 0.7%–9.1% yr^{-1}, see review (Morrisey et al. 2010)). There is also evidence that the main phase of mangrove habitat expansion in some estuaries occurred at an early stage, during the period of catchment deforestation in the mid–late-1800s, and forest development was largely complete by the early-1900s [e.g., Kaipara Harbour, see review (Swales et al. 2020b)]. Mangrove habitat loss has also occurred over the last century due to land reclamation for ports and urban development, agriculture, road and causeway construction and landfills. Historical changes prior to the 1930s (both losses and gains) in the extent of New Zealand's mangrove forests have not been accurately quantified because substantial habitat change occurred prior to systematic aerial photographic surveys.

Comparison of the Landcover Database (Ver. 5, 2021) mangrove layers for 1996 and 2018 suggests that there has been virtually no measurable change in mangrove habitat in since the mid-1990s (i.e., 28,204 ha (1996), 28,172 ha (2018) (Hicks et al. 2019). However, there are several potential factors that suggest the LCDB mapping may not provide a reliable evaluation of recent changes in mangrove habitat extent. The LCDB Ver. 5 does not include more recent coastal vegetation mapping as part of updates to regional coastal plans or spatial planning processes (e.g., Northland Regional Coastal Plan; Tai Timu Tai Pari Hauraki Gulf Marine Spatial Plan [Waikato Regional Council 2017], or from mangrove removal activities which have been substantial in some harbours [Bulmer et al. 2017; Lundquist et al. 2012; Lundquist et al. 2014]). Further, broadscale metrics (i.e., hectare as per LCDB) are unlikely to adequately detect small scale changes in mangrove habitat boundaries, including increasing density of canopy cover on expanding forest fringes that cumulatively account for substantial expansion (see, e.g., Suyadi et al. 2018a, 2019). Mangrove seedling recruitment events resulting in large increases in habitat extent (i.e., tens of metre) are also infrequent, largely due to physical controls on the success of seedling establishment. This in turn is determined by the local wind-wave climate in estuaries and timing of propagule fall during the spring–neap tidal cycle. Large recruitment events depend on propagule fall coinciding with an extended period of calm weather (i.e., 1–2 weeks) and a declining spring to neap high tide level, when bed disturbance by wave action is minimal (Balke et al. 2015; Gijsman et al. 2024). Such is the case in the southern Firth of Thames, where large-scale seedling recruitment has not occurred since the mid-1990s (Swales et al. 2015). The detection of changes will depend on the spatial resolution of the base data for mapping (i.e., aerial photography, LiDAR, satellite). The LCDB mapping is also unlikely to have captured the gradual colonisation of saltmarsh habitat by mangroves that has been occurring, as described below.

Observations and anecdotal evidence also suggest, however, that mangroves have been colonising estuarine saltmarsh habitats over the last decade or so, which may not have been adequately captured by aerial surveys/remote sensing. Mangrove colonisation of saltmarsh has been observed by the authors in many estuaries, for example Whangateau Harbour, Tauranga Harbour and the southern Firth of Thames, and is no doubt occurring in other locations. In southeastern Australia, mangrove habitat expansion (*Avicennia* spp.) into saltmarshes is well documented. This habitat expansion has been attributed to climate warming and resulting southern extension of temperature thresholds coincident with SLR, although complicated by limitations on mangrove propagule dispersal, among other factors (Saintilan et al. 2014). This process also appears to be enhanced by meteorological events that provide windows of opportunity for mangrove propagule transport and establishment into saltmarsh habitat, including annual to decadal scale ENSO-cycles that influence sea level on the coasts during periods of onshore wind and/or lower than average atmospheric pressure (Swales et al. 2015; Swales et al. 2020b). Extreme storm tides and elevating sea level also provide a key mechanism for mangrove colonisation of saltmarsh habitat. For example, a storm tide in the southern Firth of Thames that occurred January 2018 (largest event since 1938) introduced vast numbers of propagules into a glasswort marsh (*Salicornia quinqueflora*) and resulted in the rapid loss of supratidal saltmarsh ribbonwood, most likely due to storm-induced salinisation (Swales et al. 2023). Both saltmarsh habitats pre-dated the development of the mangrove forest from the early 1960s. These climate-warming related processes will ultimately result the displacement of saltmarsh and mangrove dominance on temperate shorelines, including in the northern estuaries of New Zealand.

Future changes in mangrove habitat extent over the next 20–30 years will be influenced by sea level rise (SLR) associated with climate warming and physical barriers (i.e., natural or anthropogenic) to landward migration. To persist into the future, mangroves must maintain their elevation in the upper intertidal zone, above MSL. This can be achieved by one of two main mechanisms: (1) landward migration “upslope” and/or (2) vertical accretion of mineral sediment (from rivers) or organic sediment produced by the mangrove forests themselves (McBride et al. 2016; Swales et al., 2020a). In NZ mangrove systems, sedimentation is dominated by river-borne mineral sediment, with much smaller contributions from organic sediment several produced by the trees themselves. Over the next 20–30 years mangrove habitat migration into saltmarsh is likely to accelerate, with episodic storm tides perturbing the system and resulting in step changes in coastal wetland habitats. Section D3 describes the potential longer-term impacts of climate change.

A4-(i) What monitoring is currently done and how is it reported? (e.g., is there a standard, and how consistently is it used, who is monitoring for what purpose)? Is there a consensus on the most appropriate measurement method?

Monitoring can occur at a range of scales, from monitoring of boundaries and expansion/contraction of individual mangrove forests using hand-held GPS (see techniques described in Swales et al. 2011) to broad scale habitat mapping using satellite remote sensing. Ground-truthed surveys by regional councils vary in frequency, from opportunistic responses to required updates for coastal policy statements (e.g., Northland Regional Plan, 2020) to scheduled updates.

Waikato Regional Council routinely measures the extent (but not quality) of mangrove habitats through two different projects. These projects use different methods and are undertaken for different purposes, which are mapping the extent of mangrove habitat for (1) a subset of Waikato estuaries as component of an environmental indicators [Extent of coastal habitats | Waikato Regional Council]; and (2) more detailed habitat surveys conducted every 10 years (plan to increase frequency to ~ 4 years subject to funding) [Intertidal habitat mapping for ecosystem goods and services: Waikato estuaries | Waikato Regional Council]. The mapping includes the mangrove/saltmarsh ecotone where these habitats are difficult to separate (e.g., Whangapoua Harbour) (Source: Dr Steve Hunt, WRC).

Bay of Plenty Regional Council map estuarine wetlands (including mangrove) along with freshwater wetlands using aerial photography, LiDAR/DEM and obliques. Wetland classes mapped follow Johnson and Gerbeaux (2004). Landcare Research is currently developing standards/guidelines for mapping extent of wetlands for MfE. Methods for freshwater and estuarine wetlands should align if possible. Mangrove and saltmarsh condition is also monitored (SOE) at ~150 vegetation plots established in five estuaries. Methods are adapted from those developed for freshwater wetlands (Clarkson et al. 2004). There is no standard for this type of monitoring that BoPRC are aware of (S. Dean, BOPRC, pers. comm.). Baseline data has been collected and the first survey will likely occur in 2024/2025. No data analysis has been undertaken to date. BoPRC are also undertaking other projects relevant to the mangrove and saltmarsh attributes:

- Long running study comparing mangrove sites to sites where mangroves have been removed – data on macrofauna, sediment quality and epifauna – data not yet analysed.
- Mangrove encroachment into saltmarsh: mapping was done in around 2011 but has not been repeated.
- Blue carbon coring and emissions work in saltmarsh and areas being restored into saltmarsh.
- Rod Surface Elevation Tables installed in saltmarsh and mangrove habitats in Athenree Estuary and Nukuhou Inlet, Ohiwa Harbour.
- The Coastal SIG also have an Envirolink-funded project to develop a tool for using satellite imagery to map coastal wetland habitats. (Source: Ms Shay Dean, BoPRC).

In the Auckland Region (Source: Grant Lawrence, AC), fine scale wetland-change mapping was conducted for the 2010-11 and 2017 period. Between these surveys, mangrove habitat expansion typically occurred seaward or sideways of existing mangrove habitat onto unvegetated intertidal flat (forming sparse or dense monocultures depending on the location). There are some examples of landward expansion, although these are typically onto unvegetated intertidal flat. At Puhinui and Pollen Island, where substantial areas of saltmarsh occur, mapping based on 2010-11, 2017 and 2023 imagery, shows a complex mosaic where Saltmarsh and Mangrove intergrade. This ecotone presently appears to be stable with no obvious signs of mangrove encroaching on/or displacing saltmarsh over the last decade.

Northland Regional Council mapped saltmarsh and mangrove habitat in 2020. This mapping is summarised in 19 worksheets covering the various harbours and estuaries (https://www.nrc.govt.nz/resource-library-summary/research-and-reports/saltmarsh-and-mangroves/), with mapping methods reported at https://www.nrc.govt.nz/media/5ynp3hea/northland-intertidal-vegetation-mapping-methodology-2020-2.pdf, and the GIS layers available at: https://localmaps.nrc.govt.nz/localmapsviewer/?map=55bdd943767a493587323fc025b1335c

More recently, NRC have mapped all of Northland's wetlands, including saltmarsh and mangrove, using a slightly different mapping method. The 2024 mangrove and saltmarsh layer was generated by combining 2014-2016 imagery and 2019 LiDAR data. These new map products will be released in 2024. (Source: Mr Richard Griffiths, Resource Scientist -Coastal).

Analysis of satellite remote sensing has also been opportunistic, including council-funded analyses, academic theses, and a recent MBIE-funded project on blue carbon that describes a national mapping exercise of mangroves, saltmarsh, and seagrass; Bulmer et al. 2024). There is no consistency between councils on coastal wetland typologies, with diversities of mangrove categories from differences based on morphology (scrub v. fringe forest) to mixed communities (i.e., ecotones) such as saltmarsh or seagrass within mangroves, or for differences in mangrove density and patchiness (see Suyadi et al. (2018a, 2019) for mangrove forest landscape characteristics). Mangrove density, height, and characteristics of patches (e.g., width) have strong influence on ecosystem services provided (Horstman et al. 2014; Suyadi et al. 2018a).

A4-(ii) Are there any implementation issues such as accessing privately owned land to collect repeat samples for regulatory informing purposes?

Primary monitoring of mangrove forest extent can occur via satellite remote sensing, thus there are no access issues, unless ground-truthing is required. Much of the information on mangrove habitat quality, associated with ecological and environmental characteristics (e.g., macrofaunal communities, sedimentation rates, sediment properties etc.) does require field surveys. In the Bay of Plenty, some sites have been excluded from the BoPRC mangrove and saltmarsh SOE monitoring programme due to a lack of landowner permissions.

A4-(iii) What are the costs associated with monitoring the attribute? This includes up-front costs to set up for monitoring (e.g., purchase of equipment) and on-going operational costs (e.g., analysis of samples).

Quantitative techniques for extracting hyperspectral signatures indicating presence of mangroves from satellite remote sensing have been developed based on current satellite technology (Sentinel-2 images). Analytical techniques were available for aerial photographs from prior analyses (Suyadi et al. 2018b; Swales et al. 2009). Sentinel-2 satellite images are open source, but operational costs (technical analyses) would be required to update mangrove maps on a regular basis. Timeframes of

5–10-year intervals are likely suitable to quantify changes in extent. During these longer timeframes, satellite technology may change, requiring development of new techniques, though global communities regularly provide open-source code for mangrove image analysis.

The costs associated with monitoring the mangrove-forest quality attribute in any given estuary will depend on a range of factors - the number and type of parameters measured, spatial density of measurements, temporal frequency, logistics, etc. Monitoring costs, consequently, could range from 10s to 100s thousands of dollars per year. BoPRC's condition/SOE monitoring for saltmarsh and mangrove habitat has an estimated cost of $50,000/year over three years. No data analysis has been undertaken to date. NRC (Richard Griffiths) comments, "*Depends on what you are going to monitor. Mapping their extent is relatively straightforward but time consuming (even using remote sensing methods still requires a lot of manual checking and QA*. *Northland has a big coastline and a lot of mangroves, so the cost* (*time*) *of mapping the extent depends on these factors* (*length of coastline + amount of mangroves*). *Ecological monitoring of mangrove habitat as a whole could be expensive (macroinvertebrates) or cheap* (*sediment quality, eDNA, birds*) *depending on what you did"*.

A5. Are there examples of this being monitored by Iwi/Māori? If so, by who and how?

There are no formal examples acknowledged by the authors of iwi monitoring mangrove habitats as far as we are aware. Mangrove extent, and more so expansion, has long been part of formal and informal wānanga and discussions in hapū, iwi, public and academic forums (Maxwell 2018; LeHeron et al 2022), thus leading to specific modules being highlighted in estuary monitoring toolkits as co-developed by NIWA (see below).

Mangrove habitats can be challenging environments to work in, particularly in muddy substrates. In some situations, there are also health and safety considerations when working in soft mud. An estuarine monitoring toolkit, Ngā Waihotanga Iho, was developed by NIWA (Swales et al. 2011) to provide potential guidelines of how whānau, hapū and iwi members may want to utilise tools to measure environmental changes in their estuaries. One of the modules, includes estuarine plant and habitat modules that provide methods to monitor a range of parameters including change in habitat extent.

The rapidly growing interest in blue carbon and coastal wetland restoration and enhancement of biodiversity are being driven by hapū and iwi, and or driven by other organisations who acknowledge the role of Indigenous Peoples efforts in protecting whenua and moana. For instance, NGOs, including The Nature Conservancy and Conservation International are partnering with hapū and iwi Māori, including the internationally recognised Hinemoana Halo, which includes supporting coastal wetland and seagrass restoration (Conservation International, 2024). Within Aotearoa, DOC has also recommended best practices to align with hapū and iwi Māori towards Blue Carbon initiatives that include wetland/estuary environments in Aotearoa (Kettles et al 2024). This includes a recent study that maps the current extent of blue carbon habitats alongside the following iwi Ngāti Porou, Te Whānau a Apanui, Ngāti Wai, Te Rarawa, Ngāti Ruanui, Ngā Rauru, Ngāi Tahu and Ngāti Kuri (Bulmer et al. 2023).

A6. Are there known correlations or relationships between this attribute and other attribute(s), and what are the nature of these relationships?

As noted in Sections A1 and A2, there are associations between mangrove habitat extent and substrate muddiness and saltmarsh quality attributes. Mangrove habitat extent in many estuaries coincides with intertidal flat areas above MSL elevation. These upper intertidal, typically muddy, environments have been substantially increased in extent in many estuaries due to soil erosion and elevated sediment loads from catchments during the historical era. Mangrove habitat expansion is symptomatic of excessive fine sediment loading and sedimentation in NZ estuaries [see reviews (Horstman et al. 2018; Morrisey et al. 2010; Swales et al. 2020b)] in comparison to pre-deforestation/baseline conditions (Hicks et al. 2019). Saltmarsh extent and quality is likely to progressively decline due to mangrove habitat colonisation of saltmarsh habitat, facilitated by climate warming and SLR.

Part B—Current state and allocation options

B1. What is the current state of the attribute?

The current state of mangrove habitat extent is well understood in northern New Zealand's estuaries where mangroves occur, and is included in the national land cover map LCDB (Manaaki Whenua Landcare Research 2019); monitoring occurs somewhat regularly by regional councils. Although the LCDB suggests that there has been virtually no measurable change in mangrove habitat extent since the mid-1990s [latest estimate (2018) 28,172 ha (Manaaki Whenua Landcare Research 2019)], this is unlikely to be accurate for the reasons described in Section A3.

B2. Are there known natural reference states described for New Zealand that could inform management or allocation options?

Natural reference states have not been described for New Zealand mangrove habitat extent and quality, as far as we are aware. Historical records and sediment core studies indicate that mangroves substantially increased their distribution in northern estuaries as a direct result of increased fine sediment loads and rapid accretion of intertidal flat habitat suitable for colonisation. This process was triggered by large-scale catchment deforestation and increased soil erosion (Hicks et al. 2019; Morrisey et al. 2010). In many estuaries, this disturbance was manifest by order of magnitude increases in sedimentation rates, shift from sand- to mud-dominated systems and reduction in the areal extent of subtidal habitats (Morrisey et al. 2010; Swales et al. 2015; Swales et al. 2002; Thrush et al. 2004).

A natural reference state for NZ mangrove habitat would therefore include some or all of the following characteristics:

- Sand or muddy-sand substrate with a generally low terrigenous mud input during the historical era.
- Low sediment accumulation rates (SAR) over decadal time scales. SAR approach pre-deforestation values (e.g., < 1.2 mm yr^{-1}; Swales et al. 2020a).
- Low concentrations of major stormwater contaminants (e.g., heavy metals; Swales et al. 2002), organic compounds associated with fossil fuel), and nutrients (Schwarz 2002; Thomson et al. 2024).

- Low concentrations of suspended fine sediment in the water column to enable penetration of photosynthetically available radiation (PAR), sufficient to support subtidal seagrass habitat.
- Coastal wetland sequence, with increasing elevation, consisting of (all/some of): subtidal and intertidal seagrass habitat, mangrove habitat, saltmarsh habitat and supratidal habitat (e.g., saltmarsh ribbonwood, manuka, kahikatea).
- Presence of diverse benthic fauna in mangrove dominated by mud burrowers e.g., species of crabs and shrimp that have some sensitivity to mud deposition (Ellis et al. 2004; Ellis et al. 2015).
- Mangrove habitat and creeks utilised by fish species that are sensitive to fine sediment. For example, the foraging success and health of juvenile snapper are adversely affected by elevated levels of suspended fine sediment (Lowe et al. 2015; Swales et al. 2016).

Estuaries, or parts thereof, that may approach the reference state include the Rangaunu Harbour and Wairakau Creek, Whangaroa Harbour (Northland), southern Whangateau Harbour (Auckland), and the Purangi River (Coromandel). These systems share some/all biophysical characteristics – relatively small catchment sediment loading during the historical era that reflective catchment attributes (size, steepness, erodibility, climate, indigenous forest landcover etc) and/or estuary biophysical characteristics that favour fine-sediment export to the sea. The Rangaunu Harbour, for example, has a relatively small catchment, some 80% of which is composed of lowland sand dune country. The saltmarsh, mangrove and subtidal seagrass habitats fringing the south-west shoreline of the harbour are also largely isolated from the Awanui River's fine-sediment load, with river plumes transported along the harbour's northern shore to the sea.

B3. Are there any existing numeric or narrative bands described for this attribute? Are there any levels used in other jurisdictions that could inform bands? (e.g., US EPA, Biodiversity Convention, ANZECC, Regional Council set limit)

There are no known numeric bands for New Zealand mangroves. Spatial metrics to assess condition via satellite remote sensing have been developed globally to inform mangrove forest patch characteristics (Hai et al. 2022) with respect to patchiness and fragmentation, and could be applied in New Zealand. Other more comprehensive indices of mangrove forest quality have also been developed globally, including ecological and environmental characteristics (e.g., macrofaunal communities, turbidity), as well as social attributes (Ibrahim et al. 2019), reflecting mangrove use and economic value.

B4. Are there any known thresholds or tipping points that relate to specific effects on ecological integrity or human health?

There are no known thresholds or tipping points for mangrove forests; however, key environmental characteristics such as hydroperiod (inundation time), the potential effects of sea level rise, and barriers to shoreward expansion are key drivers of future distributions of mangrove forests.

B5. Are there lag times and legacy effects? What are the nature of these and how do they impact state and trend assessment? Furthermore, are there any naturally occurring processes, including long-term cycles, that may influence the state and trend assessments?

There are several important temporal lags/legacy effects that have influenced mangrove habitat extent in estuaries:

- Sediment delivery from catchments.
- Vertical accretion of intertidal flats to elevation (i.e., mean sea level) within the hydroperiod tolerance for *Avicenna*.
- Mangrove seedling recruitment - windows of opportunity.

The historical pattern and pace of estuary infilling and creation and expansion of intertidal habitat potentially suitable for mangrove colonisation and forest development has varied depending on catchment and estuary characteristics. A non-linear relationship between catchment sediment loads, tidal prism volume and the area of intertidal habitat above MSL suitable for mangrove colonisation has been determined for a range of estuary types in the Auckland Region (Section B5). The model shows that drowned river valley estuaries typically have relatively larger areas on intertidal flats suitable for mangrove than do estuarine embayments (Swales et al. 2020b).

<u>Sediment delivery from catchments:</u> New Zealand studies show that annual sediment loads are dominated by fine sediment (mud [≤ 62.5 microns] and sand [62.5–2000 microns]) transported during storm events (e.g., Basher & Dymond 2013; Hicks et al. 2000; Hughes et al. 2012) that dominate sedimentation in marine receiving environments. Most catchments in northern New Zealand, where mangroves occur, are relatively small (i.e., < 1000 km^2), so time lags in sediment delivery to estuaries and coastal marine systems will mainly depend on sediment transport characteristics. Time lags for sediment delivery from the catchments to estuaries varies markedly between sand and mud. Mud is readily maintained in suspension in rivers due to relatively low settling velocities (i.e., 0.1–2 mm s^{-1}; Lamb et al. 2020) so that a large fraction of the mud will be delivered to estuaries during floods (i.e., hours to days), unless retained in a catchment sediment sink during over-bank flow conditions (e.g., flood plain, vegetated areas). Fine sand has much higher settling velocities (up to ~30 cm s^{-1}) so that transport rates are typically much lower than for mud. Thus, the bulk of the mud load is likely to be transported along the entire length of a river channel network to estuaries during the course of a flood event.

<u>Vertical accretion of intertidal flats:</u> Mangroves may colonise intertidal flats once they become ecologically suitable, by vertically accreting above MSL elevation, so that hydroperiod is within the physiological tolerance for *Avicenna* spp. The rate at which this vertical accretion and formation of intertidal flats occurs is dictated by the catchment sediment load, spatial dimensions of the receiving estuary (i.e., fetch and depth), sub-environment and hydrodynamic properties that dictate sediment transport. Historical sediment accumulation rates (SAR) measured on tidal flats in ~30 estuaries (mainly upper North Island) have averaged 3.2 mm yr^{-1} (range: 2–5.2 mm yr^{-1}; summarised in Huirama et al. 2021), equivalent to ~ 30 cm of accretion over 100 years. SAR are typically much higher near catchment outlets, tidal creeks and where sediment loading has been excessive. In tidal creeks, SAR of 10 mm yr^{-1} are not uncommon, and these environments hold some of the oldest mangrove forests. In the Firth of Thames, the intertidal flats accreted rapidly (SAR ~ 20 mm yr^{-1}) so that large areas became suitable for colonisation by the early 1960s. This process has been sustained by legacy sediment that has accumulated in the southern Firth that has accumulated in the southern Firth over the last ~170 years or so. In summary, the time lag for intertidal flat formation to the MSL threshold will vary widely, but typically over decades to centuries.

Mangrove seedling recruitment: events resulting in large increases in habitat extent (i.e., tens of metre) are infrequent, largely due to physical controls on the success of propagule anchoring and seedling establishment. This recruitment process is determined by the local wind-wave climate in estuaries and timing of propagule fall during the spring–neap tidal cycle. Large recruitment events depend on propagule fall coinciding with an extended period of calm weather (i.e., 1–2 weeks) and a declining spring to neap high tide level, when bed disturbance by wave action is minimal (Balke et al. 2015; Gijsman et al. 2024). Such is the case in the southern Firth of Thames, where large-scale seedling recruitment has not occurred since the mid-1990s (Swales et al. 2015). In tidal creeks and small estuaries with limited wave fetch (e.g., 100s m – km), colonisation will largely depend on tidal flat accretion and propagule production and dispersal.

Strong physical controls on seedling recruitment mean that mangrove habitat extent typically does not match the area of intertidal flat that is potentially suitable for colonisation. This is demonstrated in the east-coast estuaries of the Auckland Region, where mangroves occupied only 58% (i.e., 27 km^2) of the total intertidal flat area above MSL elevation (Swales et al. 2009). The proportion of suitable intertidal flat occupied varied from 22 to 75% and the lower elevation of mangrove trees and seedlings averaged 0.35 m above MSL. The Waitemata Harbour accounted for 32% of the 2009 mangrove habitat, and in the central harbour area had not substantially increased since the 1950s. The largest percentage increases in mangrove habitat extent occurred in the smallest estuaries with high-tide areas less than 1.5 km^2 (i.e., Okura, Orewa, Waiwera).

These considerations suggest that mangrove habitat extent in most estuaries is unlikely to ever attain its potential extent (i.e., current intertidal flat > MSL) due to physical constraints on seedling recruitment. This effect will be exacerbated in the future by sea level rise, as mangrove (and saltmarsh) habitat will progressively migrate landward where submerging low-lying land is available or will be reduced in extent by coastal narrowing and coastal squeeze in the extreme upper intertidal zone. In this later scenario, mangrove will replace saltmarsh, and mangrove habitat would only persist where sediment supplied by rivers was sufficient for these forest remnants to keep pace with SLR.

B6. What tikanga Māori and mātauranga Māori could inform bands or allocation options? How? For example, by contributing to defining minimally disturbed conditions, or unacceptable degradation.

Mātauranga Māori with respect to mangrove forests is context-dependent, with ecosystem-dependent species associated with mangrove forests varying between locales. Examples include use of mangroves as wood, as dye materials, as habitat for kaimoana including oysters (including non-native oyster species), and for mangrove tidal creeks as fishing locales for tuna. Therefore, understanding bands would be best done alongside whānau, hapū and iwi.

Part C—Management levers and context

C1. What is the relationship between the state of the environment and stresses on that state? Can this relationship be quantified?

Mangrove habitat expansion in New Zealand's northern estuaries from the mid-1800s onwards has largely occurred due to accelerated estuary infilling. This has occurred due to elevated sediment loading associated with large-scale catchment deforestation and conversion to pastoral agriculture and more recent land-use intensification. In this sense, the general pattern of mangrove habitat expansion in many estuaries is a symptom of excessive soil erosion and environmental degradation of New Zealand's northern estuaries during the historical era [e.g., reviews (Morrisey et al. 2010; Swales et al. 2020b)]. The proportion of the suitable intertidal habitat is substantially larger than present mangrove habitat extent (McBride et al. 2016). Models have been developed to understand the interplay of factors controlling seedling recruitment on wave-exposed intertidal flats (Balke et al. 2015; Gijsman et al. 2024), although this has not been mapped at estuary scale.

Looking forward, sea level rise will likely drive changes in mangrove habitat extent, with landward/upslope migration to maintain their position above MSL. The relatively wide range of elevation (i.e., MSL to Highest Astronomical Tide [HAT]) that mangrove occupies, relatively large tidal ranges, as well as generally sediment-rich estuarine systems means that New Zealand mangrove forests are unlikely to be lost to inundation during this century (Lovelock et al. 2015a). It should be noted that mangrove habitat is already expanding into saltmarsh habitat and will highly likely displace saltmarsh without management interventions. For both mangrove and saltmarsh, interventions that remove built physical barriers to migration, as lowland areas are inundated, will mitigate the likelihood of habitat loss and generate opportunities for restoration of freshwater–estuarine wetland ecosystems. The pattern and pace of potential changes in coastal wetland habitats under SLR and exploring management interventions are major research objectives of the Future Coasts Aotearoa programme.

C2. Are there interventions/mechanisms being used to affect this attribute? What evidence is there to show that they are/are not being implemented and being effective?

C2-(i). Local government driven

C2-(ii). Central government driven

C2-(iii). Iwi/hapū driven

C2-(iv). NGO, community driven

C2-(v). Internationally driven

Mangrove forests continue to decline across most of the world, particularly in tropical habitats. Aotearoa is an exception, where mangrove trends are of expanding temperate mangroves (Morrisey et al. 2010).

National and international examples of interventions are provided by hapū and iwi. Recent interest in mangrove forest restoration as a climate adaptation/blue carbon mitigation strategy, in partnership with hapū and iwi, includes more recent mapping of existing and potential mangrove habitats (Bulmer et al. 2023; Bulmer et al. 2024). International support of Indigenous partnership by NGOs in blue carbon climate mitigation potential (Conservation International 2024), and funding of mapping/monitoring exercises, with focus on carbon sequestration values of mangrove forests (see Stewart-Sinclair et al. 2024).

The rapidly growing interest in blue carbon and coastal wetland restoration and enhancement of biodiversity are being driven by hapū and iwi, and or driven by other organisations who acknowledge the role of Indigenous Peoples efforts in protecting whenua and moana. For instance, NGOs, including The Nature Conservancy and Conservation International are partnering with hapū and iwi Māori, including the internationally recognised Hinemoana Halo, which includes supporting coastal wetland and seagrass restoration (Conservation International, 2024). Within Aotearoa, DOC has also recommended best practices to align with hapū and iwi Māori towards Blue Carbon initiatives that include wetland/estuary environments in Aotearoa (Kettles et al 2024). This includes a recent study that maps the current extent of blue carbon habitats alongside the following iwi Ngāti Porou, Te Whānau a Apanui, Ngāti Wai, Te Rarawa, Ngāti Ruanui, Ngā Rauru, Ngāi Tahu and Ngāti Kuri (Bulmer et al. 2023).

Within the four northern regional authorities where mangrove occur (i.e., Northland, Auckland, Waikato and Bay of Plenty), specific sections of regional coastal plans are dedicated to mangrove management. These sections include rules and guidelines to inform consent activities with respect to mangrove removals. Mangrove removals occur regularly to support infrastructure (i.e., access, transit, power lines etc.), whereas more restrictive mangrove removal guidance has been implemented for removals related to other values such as recreation, accessibility and viewscape. Mangrove removal applications have declined significantly following large consent processes (e.g., Tauranga, Whangamata, Tairua) in the 2010s, and review of limited recovery of most mangrove removal locations (though see Whangamata for exceptions within an adaptive management framework) (Bulmer et al. 2017; Lundquist et al. 2012; Lundquist et al. 2014). Central government interventions with respect to mangroves have been primarily in assessing expansion in Ramsar sites (Miranda, Firth of Thames) where expansion was perceived as a threat to shorebird habitat.

Part D—Impact analysis

D1. What would be the environmental/human health impacts of not managing this attribute?

Left unmanaged, it is anticipated that mangrove expansion rates within northern New Zealand may continue to increase in areas with ongoing sediment supply and estuarine infilling. However, for many other areas, particularly in narrow tidal creeks, the intertidal area that is suitable for mangrove colonisation based on inundation frequency is already colonised, and little additional expansion is likely to occur. As noted earlier, mangrove expansion is correlated with higher levels of land-based sediment supply, which are typically correlated with degraded estuaries. Other impacts are more likely to be related to mental health and human wellbeing, as mangrove expansion is associated with reduced coastal access and recreational access due to muddier sediments and vegetation blocking access to sandflats and to boat launch sites.

D2. Where and on who would the economic impacts likely be felt? (e.g., Horticulture in Hawke's Bay, Electricity generation, Housing availability and supply in Auckland)

Economic impacts of mangroves are anticipated to occur with the potential for voluntary (or government supported) credit schemes for mangrove restoration to support climate mitigation. This opportunity is currently being explored, with key barriers of land tenure and regulatory context requiring further consideration (Stewart-Sinclair et al. 2024).

D3. How will this attribute be affected by climate change? What will that require in terms of management response to mitigate this?

Sea level rise (SLR) due to climate warming will have a major impact on mangrove habitat extent. Around the fringes of some estuaries, natural topographic features will prevent appreciable landward migration of mangrove (and saltmarsh) habitat. This scenario is referred to as coastal narrowing (Swales et al. 2020a). The shorelines of many NZ estuaries in lowland catchments dominated by agriculture and urban centres and their hinterland have commonly been modified by built infrastructure, including stop banks, roads, railway lines, reclamations for port, urban and industrial development. These built barriers, unless breached or removed will prevent natural adaption to SLR through habitat migration, which is referred to as coastal squeeze (Swales et al. 2020a). Under this condition, mangrove forests will require a sufficient supply of sediment to vertically accrete to keep pace with SLR. The southern Firth of Thames is a notable example of this process, with the mangrove forest tidal platform presently being some two metres higher than farmland inside the stopbank (Swales et al. 2015) due to the high availability of fine sediment supplied by rivers during the historical period (i.e., legacy sediment) and today.

The extent to which SLR will result in changes in mangrove habitat extent will also depend on the local geological and geomorphic setting. Key processes include the rate of vertical land motion (VLM, subsidence or uplift) and sediment supply rate. Relative, or local, SLR (i.e., RSLR) incorporates VLM (i.e., with the regional SLR trend). Again, the southern Firth of Thames provides pertinent example, where subsidence of a deep sedimentary basin is occurring at ~ 8 mm yr^{-1}, resulting in RSLR of ~10 mm yr^{-1} in the mangrove forest (Swales et al. 2016), some five-fold higher than the regional rate of SLR. Despite this extreme RSLR, the mangrove forest today occupies ~11 km^2 of upper intertidal flat and accumulated a ~ 2 m thick deposit of fine sediment. The forest continues to expand seaward today. This rapid habitat expansion and persistence of the mangrove forest has been possible due to the supply of contemporary and legacy sediment delivered to the Firth by rivers that maintain the intertidal flat elevation above MSL (Hicks et al. 2019; Swales et al. 2015). This relationship between catchment sediment loads, tidal prism volume and the area of intertidal habitat above MSL suitable for mangrove colonisation has been determined for a range of estuary types in the Auckland Region. A simple exponential model ($r^2 = 0.69$, $P < 0.001$) also shows that drowned river valley estuaries typical have relatively larger areas suitable for mangrove habitat than so estuarine embayments (Swales et al. 2020a).

As observed elsewhere in the Indo-Pacific (Lovelock et al. 2015b), mangrove habitat will likely persist in many of our northern estuaries beyond 2100, where landward migration is possible and/or sediment supply from catchments is sufficient to keep pace with RSLR. Rod Surface Elevation Tables (RSET) (Cahoon et al. 2002)) are used globally as a primary tool to monitor the elevation trajectories of mangrove forests and saltmarshes relative to rising sea levels, e.g., (Lovelock et al. 2015b; Webb et al. 2013). In New Zealand, RSET have been used by NIWA and Waikato Regional Council for research and monitoring mangrove forest elevation trajectories in the southern Firth of Thames since 2007 (Swales et al. 2016; Swales et al. 2019). The network is now being expanded in the MBIE-funded *Future Coasts Aotearoa* Endeavor Programme, in partnership with Regional Councils and DOC, so far including sites in the Bay of Plenty, Auckland and Canterbury Regions.

The natural range of NZ mangrove habitat has historically been controlled by number of physiological and biogeographic factors. These include low winter air temperatures, frost frequency, biogeography and oceanography that limit mangrove propagule dispersal [e.g., (de Lange & de Lange 1994)] and the limited availability and relative remoteness of suitable estuarine environments south of their present range. Although climate warming may change their potential local extent (e.g., landward migration facilitated by sea level rise), biogeographic limitations to their southern expansion remain.

References:

Bainbridge, Z., S. Lewis, R. Bartley, K. Fabricius, C. Collier, J. Waterhouse, A. Garzon-Garcia, B. Robson, J. Burton, A. Wenger, and J. Brodie. 2018. Fine sediment and particulate organic matter: A review and case study on ridge-to-reef transport, transformations, fates, and impacts on marine ecosystems. Marine Pollution Bulletin 135:1205-1220.

Balke, T., A. Swales, C. E. Lovelock, P. M. J. Herman, and T. J. Bouma. 2015. Limits to seaward expansion of mangroves: Translating physical disturbance mechanisms into seedling survival gradients. Journal of Experimental Marine Biology and Ecology 467:16-25.

Basher, L., and J. R. Dymond. 2013. Erosion processes and their control in New Zealand.

Booth, J. D. 2019. Recent (post-1930) changes in the extent of subtidal seagrass (Zostera muelleri) beds of the eastern Bay of Islands, New Zealand. New Zealand Journal of Marine and Freshwater Research 53:113-127.

Bulmer, R. H., F. Stephenson, H. F. E. Jones, M. Townsend, J. R. Hillman, L. Schwendenmann, and C. J. Lundquist. 2020. Blue carbon stocks and cross-habitat subsidies. Frontiers in Marine Science 7.

Bulmer, R. H., M. Lewis, E. O'Donnell, and C. J. Lundquist. 2017. Assessing mangrove clearance methods to minimise adverse impacts and maximise the potential to achieve restoration objectives. New Zealand Journal of Marine and Freshwater Research 51:110-126.

Bulmer, R. H., P. J. Stewart-Sinclair, O. Lam-Gordillo, S. Mangan, L. Schwendenmann, and C. J. Lundquist. 2024. Spatial variability in blue carbon habitats in Aotearoa New Zealand – opportunities for conservation and restoration. Restoration Ecology. e14225. https://doi.org/10.1111/rec.14225

Bulmer, R., P. J. Stewart-Sinclair, S. Mangan, and O. Lam-Gordillo. 2023. Hinemoana Halo blue carbon mapping and restoration opportunity analysis. Page 49. NIWA client report prepared for Conservation International.

Cahoon, D. R., J. C. Lynch, B. C. Perez, B. Segura, R. D. Holland, C. Stelly, G. Stephenson, and P. F. Hensel. 2002. High-Precision Measurements of Wetland Sediment Elevation: II. The Rod Surface Elevation Table. Journal of Sedimentary Research 72:734-739.

Carolyn J. Lundquist, June Logie, Paula Blackett, Erena Le Heron, Shaun Awatere & Jade Hyslop. 2022. A socio-ecological appraisal of perceived risks associated with mangrove (Mānawa) management in Aotearoa New Zealand, New Zealand Journal of Marine and Freshwater Research, 56:3, 447-465, DOI: 10.1080/00288330.2022.2097270

Clarkson, B. R., B. K. Sorrell, P. N. Reeves, P. D. Champion, T. R. Partridge, and B. D. Clarkson 2004. Handbook for Monitoring Wetland Condition. Coordinated Monitoring of New Zealand Wetlands. A Ministry for the Environment Sustainable Management Fund Project (5105).. [online], Landcare Research, Hamilton, New Zealand National Institute of Water and Atmospheric Research, and the University of Waikato., New Zealand. Available at: https://datastore.landcareresearch.co.nz/is/dataset/handbook-for-monitoring-wetland-condition.

Conservation International, 2024. Hinemoana Halo, Ocean Initiative. Hinemoana ki uta, Hinemoana ki tai, restoring our connection to the oceans, from our high seas to our coastal wates. Report, 16p.

de Lange, W. P., and P. J. de Lange. 1994. An appraisal of factors controlling the latitudinal distribution of mangrove (Avicannia marine var. resinifera) in New Zealand. Journal of Coastal Research 10:539-548.

Ellis, J. I., J. E. Hewitt, D. Clark, C. Taiapa, M. Patterson, J. Sinner, D. Hardy, and S. F. Thrush. 2015. Assessing ecological community health in coastal estuarine systems impacted by multiple stressors. Journal of Experimental Marine Biology and Ecology 473:176-187.

Ellis, J., P. Nicholls, R. Craggs, D. Hofstra, and J. Hewitt. 2004. Effects of terrigenous sedimentation on mangrove physiology and associated macrobenthic communities. Marine Ecology Progress Series 270:71-82.

Gijsman, R., E. M. Horstman, A. Swales, T. Balke, P. W. J. M. Willemsen, D. van der Wal, and K. M. Wijnberg. 2024. Biophysical Modeling of Mangrove Seedling Establishment and Survival Across an Elevation Gradient With Forest Zones. Journal of Geophysical Research: Earth Surface 129:e2024JF007664.

Gijsman, R., E. M. Horstman, D. van der Wal, D. A. Friess, A. Swales, and K. M. Wijnberg. 2021. Nature-Based Engineering: A Review on Reducing Coastal Flood Risk With Mangroves. Frontiers in Marine Science 8.

Hai, P. M., P. H. Tinh, N. P. Son, T. V. Thuy, N. T. Hong Hanh, S. Sharma, D. T. Hoai, and V. C. Duy. 2022. Mangrove health assessment using spatial metrics and multi-temporal remote sensing data. PLOS ONE 17:e0275928.

Hicks, D. M., B. Gomez, and N. A. Trustrum. 2000. Erosion thresholds and suspended sediment yields, Waipaoa River Basin, New Zealand. Water Resources Research 36:1129-1142.

Hicks, M., A. Semadeni-Davies, A. Haddadchi, U. Shankar, and D. Plew. 2019. Updated sediment load estimator for New Zealand. Page 190, NIWA Client Report MFE18502 for Ministry of the Environment https://environment.govt.nz/assets/Publications/Files/updated-sediment-load-estimator-for-nz.pdf

Horstman, E. M., C. J. Lundquist, K. R. Bryan, R. H. Bulmer, J. C. Mullarney, and D. J. Stokes. 2018. The dynamics of expanding mangroves in New Zealand. Pages 23-51 in C. Makowski, and C. W. Finkl, editors. Threats to Mangrove Forests: Hazards, Vulnerability, and Management. Springer International Publishing.

Horstman, E. M., C. M. Dohmen-Janssen, P. M. F. Narra, N. J. F. van den Berg, M. Siemerink, and S. J. M. H. Hulscher. 2014. Wave attenuation in mangroves: A quantitative approach to field observations. Coastal Engineering 94:47-62.

Hughes, A. O., J. M. Quinn, and L. A. McKergow. 2012. Land use influences on suspended sediment yields and event sediment dynamics within two headwater catchments, Waikato, New Zealand. New Zealand Journal of Marine and Freshwater Research 46:315-333.

Huirama, M., S. A., O. G., and O. R. 2021. Sediment accumulation rates in Waimapu, Waikarea and Tuapiro Estuaries, Tauranga Harbour. NIWA Client Report 2021255HN for Bay of Plenty Regional Council, 75 p.

Ibrahim, F.-H., F. Mohd Yusoff, A. Fitrianto, A. A. Nuruddin, S. Gandaseca, Z. Samdin, N. Kamarudin, S. Nurhidayu, M. R. Kassim, K. R. Hakeem, S. Ibrahim, I. Adnan, A. N. Abdul Ghani, R. P. Parman, S. B. Abd Razak, S. A. Ibrahim, F. Hilaluddin, F. Ramli, and N. H. A.-R. Nik Zaidin. 2019. How to develop a comprehensive Mangrove Quality Index? Methods X 6:1591-1599.

Inglis, G. J. 2003. The seagrasses of New Zealand. Pages 134-143 in E. P. Green, and F. T. Short, editors. World atlas of seagrasses. University of California Press, Berkeley, California.

Johnson, P. N., and P. Gerbeaux. 2004. Wetland Types in New Zealand. Department of Conservation, Wellington 184.

Kettles H, Albot O, Lalor K, Nicol S. 2024. Coastal wetland blue carbon in Aotearoa New Zealand: Current projects, reflections and recommendations for a community of practice. Wellington: The Nature Conservancy Aotearoa New Zealand and Department of Conservation. 35 p. + appendices.

Lamb, M. P., J. de Leeuw, W. W. Fischer, A. J. Moodie, J. G. Venditti, J. A. Nittrouer, D. Haught, and G. Parker. 2020. Mud in rivers transported as flocculated and suspended bed material. Nature Geoscience 13:566-570.

Lovelock, C. E., D. R. Cahoon, D. A. Friess, G. R. Guntenspergen, K. W. Krauss, R. Reef, K. Rogers, M. L. Saunders, F. Sidik, A. Swales, N. Saintilan, L. X. Thuyen, and T. Triet. 2015b. The vulnerability of Indo-Pacific mangrove forests to sea-level rise. Nature 526:559-563.

Lovelock, C. E., M. F. Adame, V. Bennion, M. Hayes, R. Reef, N. Santini, and D. R. Cahoon. 2015a. Sea level and turbidity controls on mangrove soil surface elevation change. Estuarine, Coastal and Shelf Science 153:1-9.

Lowe, M. L., M. A. Morrison, and R. B. Taylor. 2015. Harmful effects of sediment-induced turbidity on juvenile fish in estuaries. Marine Ecology Progress Series 539:241-254.

Lundquist, C. J., S. Hailes, K. Carter, and T. Burgess. 2014. Ecological status of mangrove removal sites in the Auckland Region. Page 97. Prepared by the National Institute of Water and Atmospheric Research for Auckland Council. Auckland Council technical report, TR2014/033, Auckland.

Lundquist, C., S. Hailes, K. Cartner, K. Carter, and M. Gibbs. 2012. Physical and ecological impacts associated with mangrove removals using in situ mechanical mulching in Tauranga Harbour. Page 106 p., NIWA Technical Report #137. ISSN 1174-2631 (print); ISSN 2253-5985 (online).

Manaaki Whenua Landcare Research. 2019. Land Cover Database Version 5.0, Mainland, New Zealand in M. W. L. Research, editor, https://lris.scinfo.org.nz/layer/104400-lcdb-v50-land-cover-database-version-50-mainland-new-zealand/.

Maxwell 2018 Mangroves are assets to the many but a curse to the few - polarised perceptions in New Zealand. ISME/GLOMIS Electronic Journal (Vol 16) No. 4.

McBride, G., G. Reeve, M. Pritchard, C. Lundquist, A. Daigneault, R. Bell, P. Blackett, A. Swales, S. Wadhwa, A. Tait, and C. Zammit. 2016. The Firth of Thames and Lower Waihou River. Synthesis Report RA2, Coastal Case Study. Climate Changes, Impacts and Implications (CCII) for New Zealand to 2100. . Page 50. MBIE contract C01X1225. .

Morrisey, D. J., A. Swales, S. Dittmann, M. A. Morrison, C. E. Lovelock, and C. M. Beard. 2010. The ecology and management of temperate mangroves. Oceanography and Marine Biology 48:43-160.

Ross, F. W. R., D. E. Clark, O. Albot, A. Berthelsen, R. Bulmer, J. Crawshaw, and P. I. Macreadie. 2024. A preliminary estimate of the contribution of coastal blue carbon to climate change mitigation in New Zealand. New Zealand Journal of Marine and Freshwater Research:1-11.

Saintilan, N., N. C. Wilson, K. Rogers, A. Rajkaran, and K. W. Krauss. 2014. Mangrove expansion and salt marsh decline at mangrove poleward limits. Global Change Biology 20:147-157.

Schwarz, A. 2002. The role of nutrients in contributing to mangrove expansion in NIWA, editor. NIWA Client Report HAM2002-051 prepared for Environment Waikato and the Department of Conservation. NIWA, Hamilton.

Stewart-Sinclair, P. J., R. H. Bulmer, E. Macpherson, and C. J. Lundquist. 2024. Enabling coastal blue carbon in Aotearoa New Zealand: opportunities and challenges. Frontiers in Marine Science 11.

Suyadi, J. Gao, C. J. Lundquist, and L. Schwendenmann. 2018a. Characterizing landscape patterns in changing mangrove ecosystems at high latitudes using spatial metrics. Estuarine, Coastal and Shelf Science 215:1-10.

Suyadi, J. Gao, C. J. Lundquist, and L. Schwendenmann. 2018b. Sources of uncertainty in mapping temperate mangroves and their minimization using innovative methods. International Journal of Remote Sensing 39:17-36.

Suyadi, J. Gao, C. J. Lundquist, and L. Schwendenmann. 2019. Land-based and climatic stressors of mangrove cover change in the Auckland Region, New Zealand. Aquatic Conservation: Marine and Freshwater Ecosystems 29:1466-1483.

Suyadi, J. Gao, C. J. Lundquist, and L. Schwendenmann. 2020. Aboveground carbon stocks in rapidly expanding mangroves in New Zealand: Regional assessment and economic valuation of blue carbon. Estuaries and Coasts 43:1456-1469.

Swales, A., D. F. T. Rickard, R. J. Craggs, M. A. Morrison, C. J. Lundquist, R. Stott, J. J. Fitzgerald, M. J. Ahrens, R. J. Davies-Colley, F. E. Matheson, R. G. Bell, A. C. Hume, M. Felsing, J. E. Hewitt, R. Ovenden, M. E. Crump, and M. A. Smith. 2011. Habitat Mapping Te Mahere Kainga Koiora. Ngā Waihotanga Iho – Estuary Monitoring Toolkit for Iwi. . National Institute of Water & Atmospheric Research Ltd. (NIWA), Wellington, New Zealand.

Swales, A., G. Reeve, D. R. Cahoon, and C. E. Lovelock. 2019. Landscape Evolution of a Fluvial Sediment-Rich Avicennia marina Mangrove Forest: Insights from Seasonal and Inter-annual Surface-Elevation Dynamics. Ecosystems.

Swales, A., M. Pritchard, and G. B. McBride. 2020b. Biogeomorphic evolution and expansion of mangrove forests in fluvial sediment-rich estuarine systems. Pages 3–45 in F. D.A., and F. Sidik, editors. Dynamic Sedimentary Environments of Mangrove Coasts. Elsevier.

Swales, A., P. Denys, V. I. Pickett, and C. E. Lovelock. 2016. Evaluating deep subsidence in a rapidly accreting mangrove forest using GPS monitoring of surface-elevation benchmarks and sedimentary records. . Marine Geology 380:205–218.

Swales, A., R. B. Williamson, L. F. Van Dam, M. J. Stroud, and M. S. McGlone. 2002. Reconstruction of urban stormwater contamination of an estuary using catchment history and sediment profile dating. Estuaries 25:43-56.

Swales, A., R. Bell, and D. Lohrer. 2020a. Estuaries and lowland brackish habitats. in C. Hendtlass, and D. Neale, editors. Coastal Systems and Sea Level Rise, New Zealand Coastal Society Special Publication, pp 55–63. .

Swales, A., R. G. Bell, R. Gorman, J. W. Oldman, A. Altenberger, C. Hart, L. Claydon, S. Wadhwa, and R. Ovenden. 2009. Potential future changes in mangrove-habitat in Auckland's east-coast estuaries in A. R. Council, editor. Technical Report no. 079 June 2009. Auckland Regional Council, Auckland.

Swales, A., S. Hunt, K. R. Carter, and C. E. Lovelock. 2023. Stormy weather: the response and resilience of a rapidly subsiding New Zealand mangrove-forest to an extreme storm-tide event. NZ Coastal Society Annual Conference, Wellington, 22–24 November 2023.

Swales, A., S. J. Bentley, and C. E. Lovelock. 2015. Mangrove-forest evolution in a sediment-rich estuarine system: opportunists or agents of geomorphic change? Earth Surface Processes and Landforms 40:1672-1687.

Thomson, T., J. I. Ellis, M. Fusib, N. Prinza, L. C., S. J. Bury, U. Shankar, S. C. Cary, and C. A. Pilditch. 2024. Effects of catchment land use on temperate mangrove forests. Science for the Total Environment In press.

Thrush, S. F., J. E. Hewitt, V. J. Cummings, J. I. Ellis, C. Hatton, A. Lohrer, and A. Norkko. 2004. Muddy waters: elevating sediment input to coastal and estuarine habitats. Frontiers in Ecology and the Environment 2:299-306.

Waikato Regional Council 2017. Sea Change - Tai Timu Tai Pari: Hauraki Gulf Marine Spatial Plan Waikato Regional Council, Hamilton, New Zealand.

Webb, E. L., D. A. Friess, K. W. Krauss, D. R. Cahoon, G. R. Guntenspergen, and J. Phelps. 2013. A global standard for monitoring coastal wetland vulnerability to accelerated sea-level rise. Nature Climate Change 3:458-465.

Zabarte-Maeztu, I., F. E. Matheson, M. Manley-Harris, R. J. Davies-Colley, and I. Hawes. 2021. Interaction of substrate muddiness and low irradiance on seagrass: A mesocosm study of Zostera muelleri. Aquatic Botany 175:103435.