Seagrass dynamics in the Bay of Plenty

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Seagrass beds at Tuapiro-Athenree, Tauranga Harbour. Photo credit: PhotoOblique

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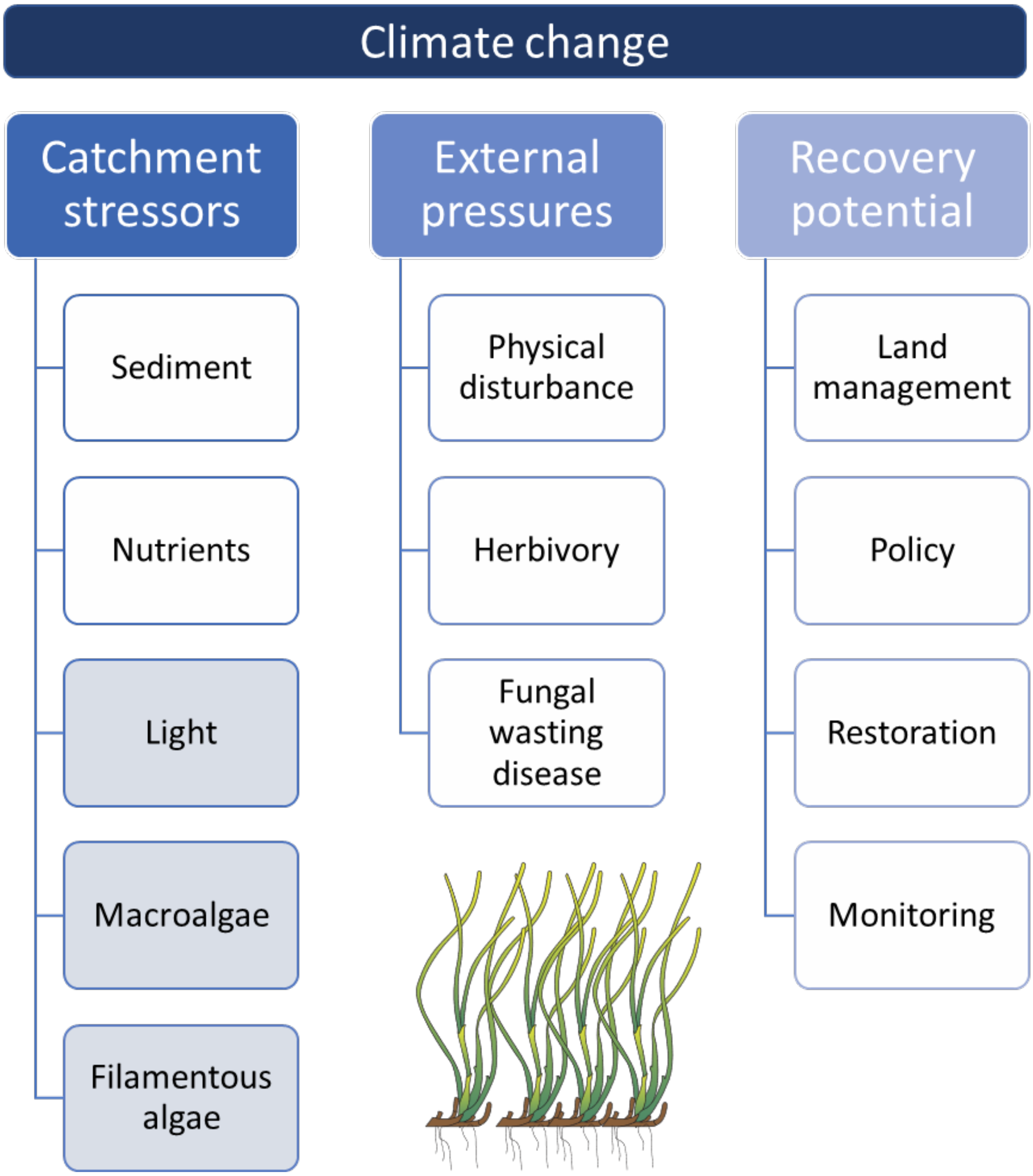
Executive Summary/ Whakarāpopototanga Matua

Seagrass is a flowering marine plant in sheltered coastal and estuary ecosystems. It provides numerous ecosystem services, including habitat, food and nursery areas for a range of fish species, supporting increased biodiversity. The New Zealand Coastal Policy Statement (2010) requires councils to protect indigenous biodiversity in the coastal environment, and avoid, remedy or mitigate adverse effects on habitats in the marine environment, including seagrass. This is implemented at the regional level within the Bay of Plenty Regional Council (BOPRC) Regional Coastal Environment Plan through objectives, policies and rules.

Significant loss of seagrass extent has been documented across all major Bay of Plenty estuaries over the past 100 years. These previous declines have been attributed to sediment and nutrient enrichment of the estuaries caused by changes in land use and loss of fresh and saltwater wetlands. Recent assessment indicates some recovery of seagrass extent in Tauranga Harbour (including significant areas of subtidal seagrass), whilst declines continue across the smaller estuaries of the Bay of Plenty, including Ōhiwa Harbour, Maketū and Waihī Estuaries. Recovery of seagrass appears limited in the smaller estuaries, in particular where the historical damage of mud or eutrophication remains. Active restoration initiatives will need to be investigated to support the return of seagrass habitats in estuaries such as Waihī and Maketū. Management of catchment stressors is required to retain existing seagrass and provide for better water and sediment quality, for restoration of seagrass to degraded habitat. The key stressors requiring management include sediment, nutrients, and the interactive effects of eutrophication (macroalgae growth and limited light availability). This is essential to ensure seagrass has the resilience to withstand external pressures, such as herbivory, fungal wasting disease and disturbances. Updated and revised calculations of black swan grazing pressure on seagrass biomass shows it may be exceeding carrying capacity, particularly at more localised scales due to swan aggregations, which remove a higher biomass and slow the recovery of seagrass.

A range of stressors can impact on seagrass health and extent, and this report summarises the current knowledge of these stressors with reference to the Bay of Plenty, and discusses options for management to reduce the impact of these stressors and plan for future seagrass recovery. We include a summary of possible research gaps that can inform future management of these important marine habitats.



Schematic of catchment and external pressures on seagrass, and the core components required to support seagrass recovery.

Contents/Rārangi Upoko

1 Introduction/Kupu Whakataki

Seagrasses are true flowering plants (Angiospermae) with stems, leaves, roots and flowers, which are specialised to grow rooted and submerged in estuarine and coastal environments (Turner & Schwarz 2006). Seagrass is primarily found in intertidal and shallow subtidal regions of bays, estuaries and coastal waters from subantarctic to tropical coasts (McKenzie et al. 2020). They can range from small patches to expansive meadows covering many hectares. In New Zealand there is one species of seagrass, *Zostera muelleri* (Jones et al. 2008), commonly referred to as eelgrass, karepō, nana and rimurehia (Figure 1).

Figure 1 Left: seagrass bed in Tuapiro Estuary, Tauranga Harbour. Right: Seagrass rhizome and blades.

Z. muelleri is a native species with a national threat status of 'at risk-declining' according to the New Zealand's Threat Level Classification System, managed by the Department of Conservation (DOC) (de Lange et al. 2018). Across New Zealand there are three key habitats where seagrass is generally distributed: soft intertidal sediments (predominant habitat), subtidal sediments (historically larger cover), and within patches of soft intertidal sediments, on rocky intertidal flats (in isolated locations throughout New Zealand, e.g., Taranaki, Hawkes Bay, Gisborne, Kaikoura (Turner & Schwarz 2006). Historically, there may also have been seagrass in soft sediment patches associated with subtidal rocky reefs.

Zostera muelleri appears to primarily reproduce by vegetative propagation (Inglis 2003), although more recently, examples of seagrass flowering have been discovered, including in Tauranga Moana (Clark & Berthelsen 2021; Dos Santos & Matheson 2016; Zabarte-Maeztu et al. 2021a). Seagrass flowers are cryptic, and it is not certain whether the prevalence of flowers has increased, or surveillance has been more effective.

Seagrass meadows are an important component of the estuarine ecosystem, providing ecosystem services including habitat, food, and nursery areas for a range of fish and macroinvertebrates, which increases biodiversity (Morrison et al., 2014). Seagrasses also trap sediments and stabilise the seafloor, remove nutrients (contributing to nutrient cycling), whilst providing a carbon sink removing CO_2 and sequestering carbon within their sediments (Unsworth et al., 2019).

Broad-scale mapping by regional authorities has identified significant losses of seagrass beds throughout New Zealand (Park 1999, Inglis 2003, Ha et al., 2021). The direct causes of such changes have not been determined and may vary in different locations, but are likely to involve compounding impacts of sedimentation following the conversion of land from native forest to agricultural and urban land use (Morrison et al. 2014, Our Marine Environment 2022), sediment nutrient enrichment, reduced light availability, marine heat waves and physical disturbance (e.g., loss from land reclamation, dredging, anchoring and moorings), waterfowl grazing and disease (Orth et al., 2006, Roca et al., 2016, Unsworth et al., 2019, Zabarte-Maetzu et al., 2021c).

The New Zealand Coastal Policy Statement (2010) requires councils: "*To protect indigenous biological diversity in the coastal environment*" and specifically to *"avoid adverse effects of activities on: indigenous ecosystems and vegetation types that are threatened in the coastal environment" and to* "*avoid significant adverse effects and avoid, remedy or mitigate other adverse effects of activities on: (iii) indigenous ecosystems and habitats that are only found in the coastal environment and are particularly vulnerable to modification including.... eelgrass....*" (Policy 11; page 16). Each Council's Regional Coastal Plan is required to give effect to the Coastal Policy Statement, through identification of issues, objectives, and policies to support the health of the coastal environment. The BOPRC Regional Coastal Environment Plan came into effect in 2019. Seagrass loss is identified as a key issue (Issue 8) and a specific objective has been included to take action to prevent further loss (Objective 4): *"Prevent the further loss of the quality and extent of rare and threatened habitats in the coastal environment of the region. These include ….seagrass beds…"* (page 19).

1.1 Report purpose/Taki

This report aims to provide an up-to-date summary of the extensive research advancements on seagrass state and pressures in the Bay of Plenty, with a focus on the current state and trends of seagrass extent within Moana a Toi estuaries, and utilisation of new technology to map seagrass distribution. Critical discussion of each of the key stressors to seagrass health in the context of the Bay of Plenty is provided. Finally, the report provides recommendations on possible management priorities and research gaps to preserve and enhance seagrass ecosystems.

2 State and trends of Moana a Toi seagrass

2.1 Bay of Plenty Regional Council State of the Environment monitoring

Aerial imagery has been utilised by BOPRC to map historical and current distributions of seagrass, detailed in previous reports (Park 1999, 2014, 2016).

In Tauranga Harbour, the earliest suitable aerial photography available for mapping of seagrass across the whole harbour was 1959, which was combined with a subsequent survey in 1996 to identify a 34% loss over that period (Park 1999). At a finer habitat resolution, the highest losses were for subtidal seagrass beds with a 90% loss across the entire Tauranga Harbour, followed by intertidal seagrass within the enclosed and sheltered sub-estuaries of the harbour (Park 1999). The rate of seagrass loss in sub-estuaries correlated well with increased suspended sediment loading coming from the catchments, indicating sedimentation and/or light stress was the cause of decline (Park 1999). A further 6.5% loss of seagrass in Tauranga Harbour occurred between 1996 and 2011 (Park 2016). Tauranga Harbour mapping was updated in 2022, which identified recovery of seagrass past the 1996 coverage, up to a current extent of 3740 ha (Figure 2A). This represents a 25% recovery from seagrass bed demographics in 2011. A large area of the seagrass recovery included subtidal seagrass, increasing from 30 ha in 2011, to 659 ha in 2022 (Figure 3). The most significant seagrass recovery occurred in the open harbour areas to the north and south of the mid-harbour (Tirohanga Point) mangroves (Figure 3). Some regional seagrass losses were however observed, including Blue Gum Bay along the edge of Matakana Island (76 ha), Tuapiro Athenree (8 ha), Tuapiro Estuary (6 ha), and Ongare (5 ha).

Ōhiwa Harbour also showed declines in seagrass between 1945 to 1996 of 33 ha, followed by a slight decline, then increase to 2011 (Figure 2B) (Park 2016). These losses primarily occurred in the sheltered upper regions where a number of catchment inflows to the harbour occur (Figure 3) (Park 2016). Further losses occurred through to 2022, with 66 ha of seagrass now remaining in Ōhiwa Harbour (Figure 3). Similar to the historical losses of seagrass in Tauranga Harbour, the seagrass retreated into intertidal habitats away from the subtidal channels, indicating sedimentation/turbidity stress likely to be the primary cause of loss.

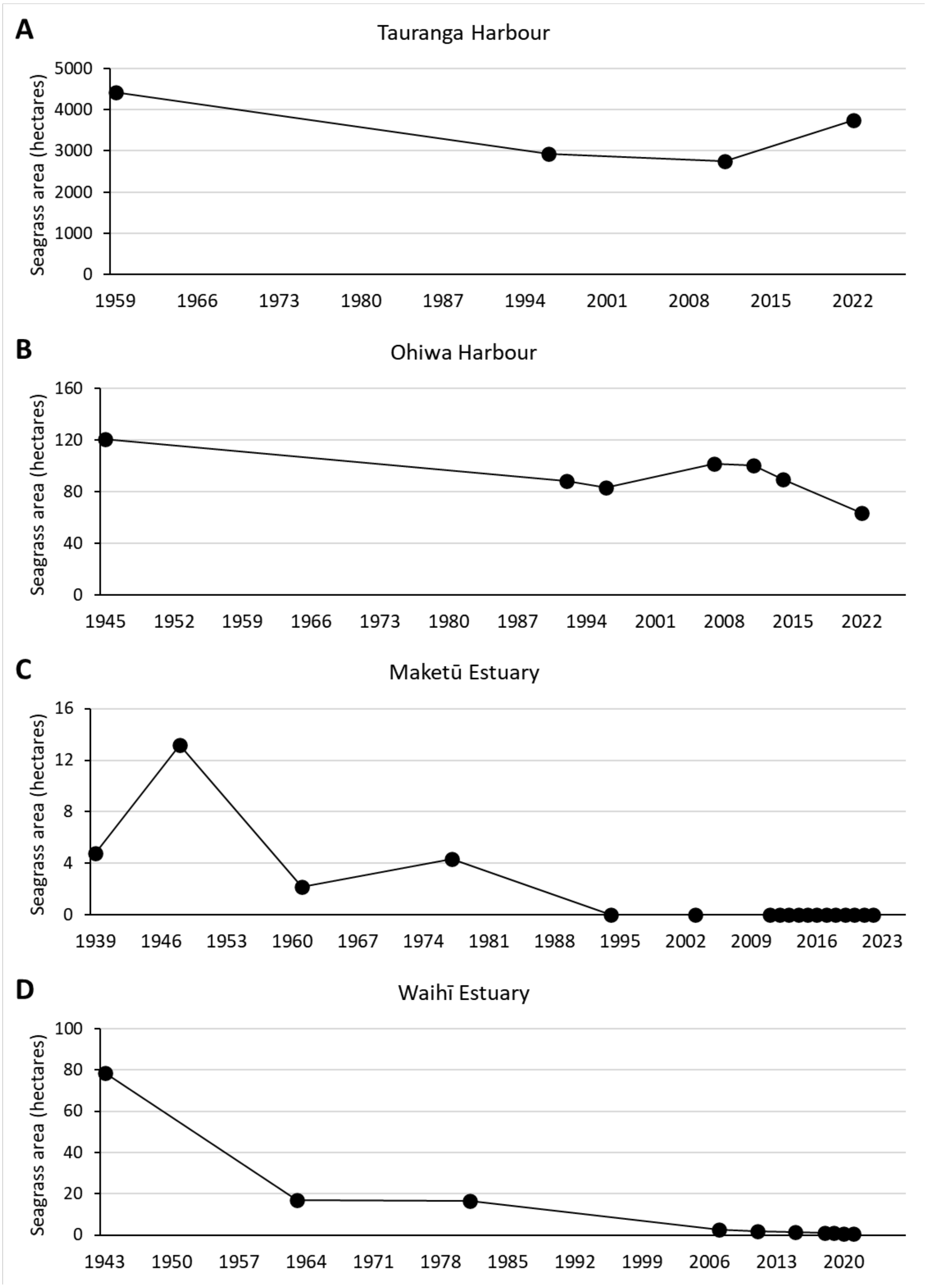


Figure 2 *Change in seagrass aerial extent (hectares) mapped from aerial imagery in Tauranga Harbour (A), Ōhiwa Harbour (B), Maketū Estuary (C) and Waihī Estuary (D).*

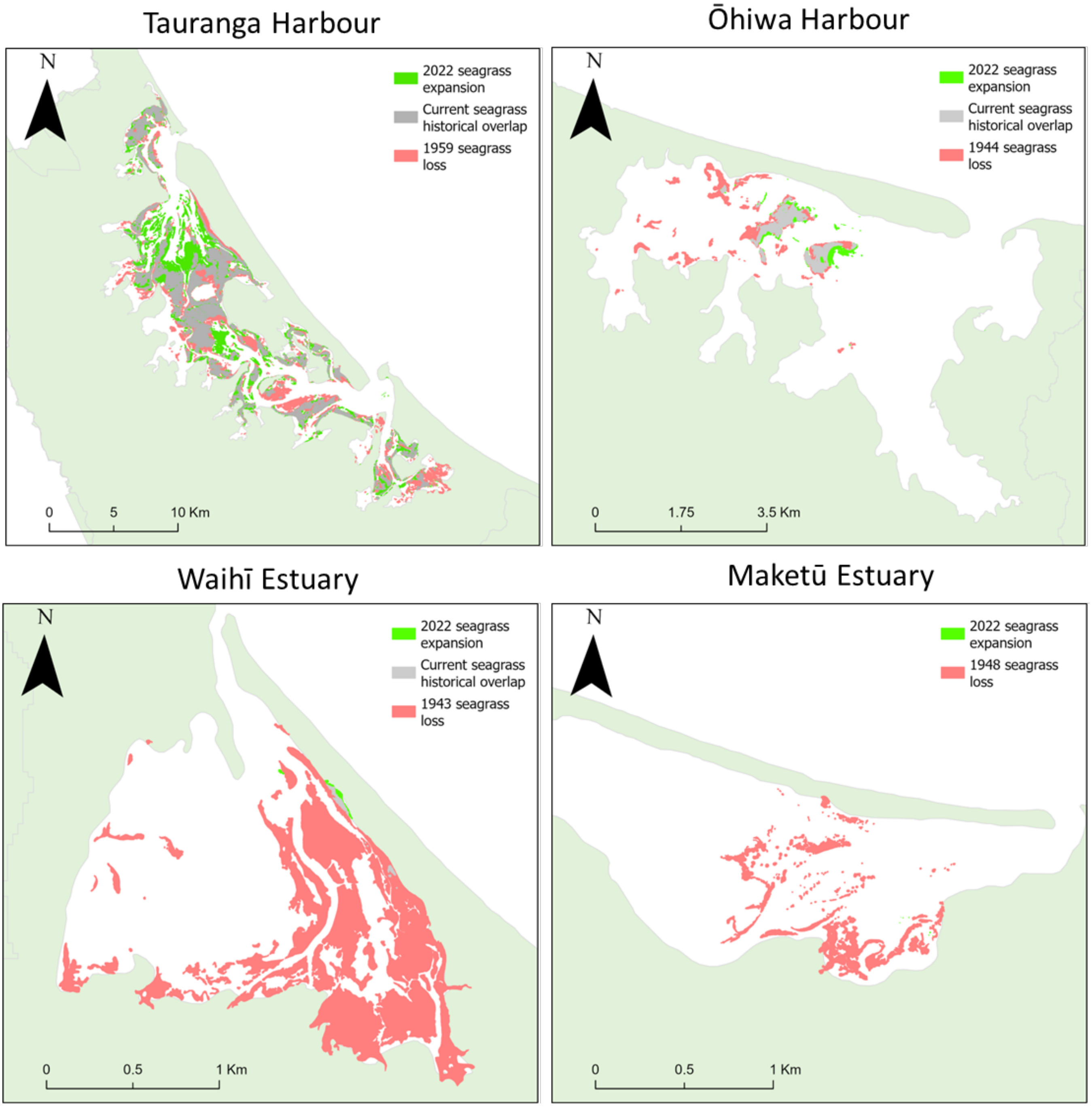


Figure 3 *Change in seagrass extent from historical (~1940-50s) to current day (2022) in Bay of Plenty estuaries mapped by aerial imagery. Green = most recent expansion, Grey = current seagrass coverage that overlaps the historical coverage, Red = areas of seagrass loss between historical and current.*

Seagrass extents in Maketū Estuary were high through to 1961 (2.2 ha), followed by a major decline by 2011 (to 0.0004 ha) and remaining low to the present day (Figure 2C, Figure 3) (Park 2014). From 1975, nuisance macroalgae (predominantly *Agarophyton, Ulva* and blue-green algae) in the estuary began to increase markedly, reaching 40% cover in 2010 across the intertidal habitat (Park 2018a). Maketū Estuary has been subjected to major changes in the salinity regime, with the Kaituna River being diverted out to sea through the Te Tumu Cut. The early increases in seagrass cover relate to a period where the Kaituna River was flowing through the estuary, lowering the salinity. Seagrass declines since 1977 are likely associated with declining water quality, due to lack of freshwater flushing and drainage of adjacent wetlands (Park 2018a). Since the Kaituna River Re-Diversion Project in 2020, which diverted water from the Te Tumu Cut back into the estuary, there has been an improvement in ecological health (Park 2022), including the re-establishment of a small patch of seagrass (0.0058 ha, or 58 m^2).

The extent of seagrass in Waihī Estuary declined significantly from 1943 to 2011, following extensive modification of stream flows to the estuary and drainage of large wetlands in 1926 (Park 2018a). The seagrass appeared to stabilise between 1963 and 1981, before declining again from the 1980's onwards, associated with declining catchment water quality (Park 2018a). The current seagrass extent is stable at around 0.86 ha in 2021 (Figure 2D), a minute proportion of historical cover (Figure 3).

Relative to size, Waihī Estuary historically held the highest seagrass percentage cover of all the estuaries prior to catchment and wetland degradation (Figure 4). Relative to size, Tauranga Harbour hosts the greatest coverage of seagrass and appears to have some resilience and capacity for recovery, compared to the smaller estuaries of Waihī and Maketū where seagrass is functionally lost from the system. Ōhiwa Harbour has shown some historical resilience and recovery of seagrass.

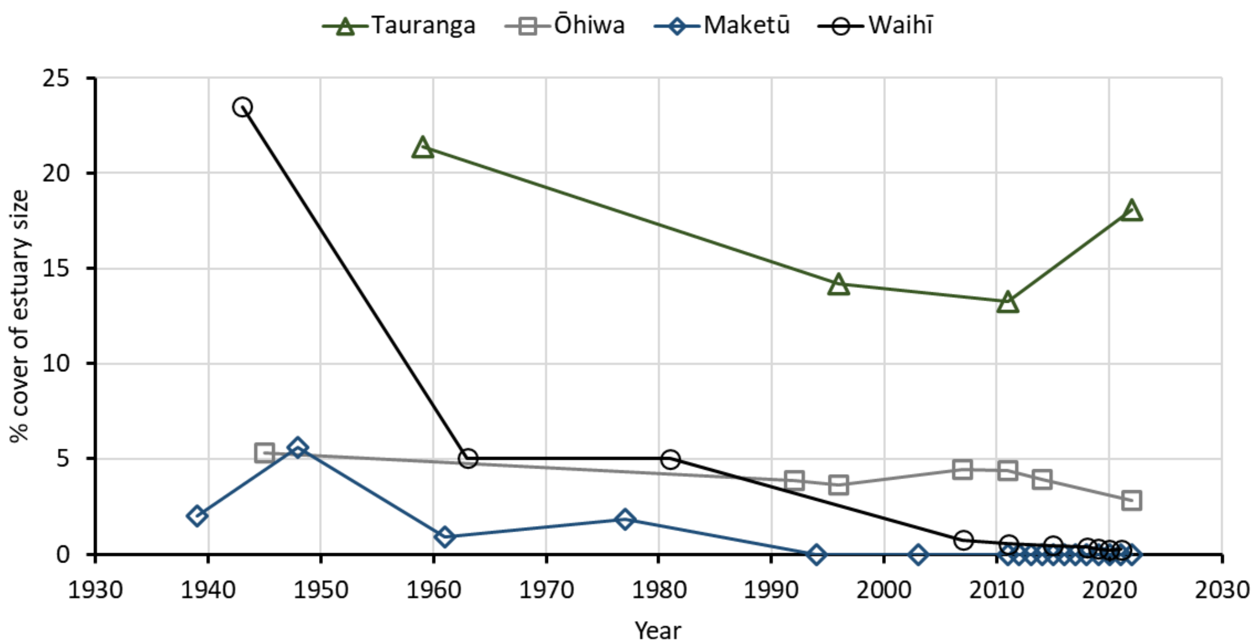


Figure 4 *Seagrass extent as a percentage of estuary size over time mapped from aerial imagery. Green triangles = Tauranga Harbour, Grey squares = Ōhiwa Harbour, Blue diamonds = Maketū Estuary, Black circles = Waihī Estuary.*

3 Impacts of stressors on seagrass

Seagrasses are vulnerable to various abiotic stressors, including light, sedimentation, nutrient levels (indirect effect through algal blooms), physical disturbances, and the effects of climate change (including increased frequencies and duration of marine heat waves). Tolerance ranges have been established for some of these stressors at the individual level, which can indicate when seagrass communities may be at risk of decline. However, in natural environments, these stressors often co-occur and interact with each other, leading to persistent interactions, that can result in cumulative impacts on seagrass ecosystems (Stockbridge et al, 2020).

Seagrasses tend to use a strategy of space occupation and have traits that allow them to flourish in oligotrophic conditions where they can accumulate nutrients and biomass over time. Such traits include a high allocation of resources to vegetative growth and reproduction, and their branching rhizomes establish a resilient framework for dense meadows to develop when conditions are good, and create active, nutrient regenerating rhizospheres. Vertical extension of ramets, which can be senesced under adverse conditions, allows further resilience against sedimentation and nepheloid layers, and the ability of roots to remain oxic via gas-permeable tissues allows survival in reducing, muddy sediments (Zabarte-Maetzu et al 2021b). Conversely, seagrass is less able to exploit eutrophic conditions suitable for rapid growth, where they cannot compete with fast-growing weedy macroalgae such as *Ulva, Agarophyton, Cladophora* and *Chaetomorpha.*

Most photosynthetic organisms' respond to stressors through physiological changes such as decreased photosynthetic rates and net carbon exchange which leads to reduced growth and biomass. Quantifying the effects of these stressor interactions becomes challenging when two or more stressors are present concurrently. Over time, these stress responses can lead to decline in seagrass extent (Leonard et al. 2020), and the decline in extent is not necessarily linear or gradual.

3.1 Sedimentation

Terrestrial sediments naturally erode into rivers and estuaries, but in New Zealand, erosion rates are particularly high due to the geomorphic conditions (steep slopes, tectonic activity, volcanism) and high frequency, and intensity of rainfall (Hicks et al. 2011). Terrestrial sediments are washed into estuarine environments through runoff from the land, from river and stream channel erosion and landslides (Thrush et al. 2004). Marked increases in sediment loading to estuaries coincided with catchment deforestation in the mid-1800s (Ministry for the Environment and Stats NZ, 2019; Swales et al. 2002; Morrison et al. 2009). The further development of farming and forestry in estuary catchments added further sediment into estuaries (Parliamentary Commissioner for the Environment 2020). The rate of sedimentation in many New Zealand regions are orders of magnitude higher in recent times compared to pre-European settlement (Jones et al. 2022, Swales et al. 2002; Handley et al. 2017). The accumulation of fine sediments in estuaries is highly variable and driven by tidal processes, wind-driven currents and waves, freshwater flow and the local morphology of the estuary (de Ruiter et al. 2019), as has been reported recently for the sub-estuaries in Tauranga Harbour (Park et al. 2022).

There are a number of direct and indirect impacts of sedimentation on seagrass, including increased turbidity (discussed below in the 'Light' section), smothering/burial, changing sediment conditions, eutrophication (discussed in the 'Nutrient' section), and sediment anoxia and sulphide production (Zabarte-Maetzu et al 2021c).

Fine sediments (particles <63 μm) depositing to the seafloor work their way into the sediment, restricting solute transport (such as nutrients and oxygen) by reducing substrate porosity. Limited solute diffusion driven by fine sediment and organic matter accumulation can exacerbate oxygen reduction, compounded by increased microbial oxygen demand, leading to sediment anoxia and increased concentration of sulphides and mud associated nutrients (Zabarte-Maeztu et al. 2021c; Burkholder et al. 2007). Sediment contamination by heavy metals can also be highly correlated to mud content due to the adsorption of metals to fine sediments and/or organic material (Green et al. 2001).

Fine sediments may also settle on seagrasses, smothering leaf surfaces and inhibiting photosynthesis via shading, and in extreme cases, result in complete burial of plants (Zabarte-Maeztu et al. 2021c; Cabaco, et al. 2008; Brodersen et al. 2017). In particular, smaller seagrass species such as *Z. muelleri* found in New Zealand are vulnerable to burial through the acute effects of large deposits (such as associated with large storm events) (Zabarte-Maeztu et al. 2021c; Benham et al. 2019; Sorensen 2020). There are limited experimental studies that examine the effects of chronic accumulation of mud on seagrass beds. Seagrass growing in muddier substrates have lower rhizome growth and below ground biomass (Zabarte-Maeztu et al. 2021b), increased leaf width and above: below ground biomass ratio (Flowers, in prep.), which indicates that high phenotypic plasticity increases resilience of seagrass to extreme burial events (Sorensen, 2020), and sustains primary production in muddier substrate (Flowers, in prep.). Multi-stressor effects of mud are strong determinants of seagrass growth and persistence, often interacting with nutrient enrichment such as organic matter (Zabarte-Maeztu et al. 2020, 2021c) or irradiance (Zabarte-Maeztu et al. 2021b) further complicating the seagrass response. Research by Flowers (in prep) indicates that seagrass can maintain primary production in the face of some increase in sediment mud content, by maintaining above-ground biomass (at the expense of below-ground biomass) with increasing mud content, to maintain high rates of photosynthesis. Conversely, once seagrass is lost from the system, unvegetated habitats lose photosynthetic efficiency with increasing mud content (Flowers, in prep.).

In the Tauranga Harbour, the majority of seagrass beds have been lost from the sub-estuaries, due to increased catchment sediment loading, and 'muddification' of the seafloor in these areas (Park, 1999) (Figure 5). Similar patterns are evident in the other estuaries of the Bay of Plenty (Figure 5). Tauranga Harbour broadscale surveys in 1992 identified a negative relationship between mean seagrass percentage cover and the percentage mud in surface sediments (Park & Donald 1994). The relationship identified that seagrass was unlikely to be present once the mud content reaches a level of approximately 13% (Park & Donald 1994), although in other New Zealand studies, this has been identified as being slightly higher (up to 35% mud content) (Zabarte-Maeztu et al. 2020; Flowers in prep.).

The highest seagrass losses in Tauranga Harbour have occurred within sub-estuaries with strong links to high catchment sediment loads (Park 2016), and proportion of mud extent in the estuary (Crawshaw & Park, in prep). In many of these estuaries, we have also seen an expansion of mangroves out into the modified muddy estuary floor (Park 2004). Similar trends are seen around New Zealand (Swales et al. 2021), also coinciding with the intensification of agriculture and urban development, and resultant declines in freshwater quality, suggesting a potential synergy with nutrient enrichment also spurring mangrove expansion (Jones et al. 2022). The majority of sub-estuaries in Tauranga Harbour are now characterised by a high sediment mud content extent (Crawshaw et al. 2022), where fine sediments are accumulating and changing the sediment composition.

Historical sedimentation rates (6470-7100 years prior to 1940) in Tauranga Harbour were around 0.05 mm/year, based on 14C isotope ageing of cockle shells (Huirama et al. 2021), in contrast, current sedimentation rates (mid 1950s-2020) are between 2.6-15.1 mm/yr, indicating a significant change in sedimentation loading to Tauranga Harbour (Park et al. 2022). All other estuaries in the Bay of Plenty have shown a decrease in seagrass cover, associated with extensive catchment modification and a mix of increased sediment/and nutrients. In addition to smaller, chronic loading of sediment, larger rainfall or catchment slip events can introduce significant slugs of sediment to the estuary (Figure 5C). In Waihī Estuary, following a year of repeated high rainfall events, the annual sedimentation at numerous sites was found to be around 26 mm, a significant jump from the normal annual deposition of ~2 mm-6 mm. Similar events have been recorded in Ōhiwa Harbour, where one rainfall event in 2014 accounted for 93% of the sediment load to the harbour that year (Park 2022).

Catchment sediment load reductions are proposed under BOPRC's implementation of the National Policy Statement for Freshwater Management (NPS-FM) (Park et al. 2022), in conjunction with ongoing non-regulatory interventions with land management, will help reduce ongoing pressure on seagrass from chronic and acute sedimentation. However, a legacy of past land use remains in Bay of Plenty estuaries, and habitat is poor in many regions due to the extent of soft mud (Park et al. 2022). The estuary sediments may take decades to respond to sediment reductions, if at all, depending on the bioturbation capacity, flushing, colonisation by mangrove (Park et al. 2022), and interactions with sea level rise (see climate change section). Additional actions may be required to support increased flushing of these estuaries, such as in Maketū Estuary where the increase of freshwater flows from the Kaituna River to the estuary has resulted in some improvements in sediment quality in the upper estuary.

Figure 5 A *Seagrass bed impacted by high mud content. B: Deep mud in a historical seagrass bed location in Waihi Estuary. C: Large sediment plumes into Maketū and Waihī estuaries on 10 April 2023 following a rainfall event (Planet Imagery).*

3.2 Light

Light is one of the most important and complex factors that can reduce or enhance the metabolism of plants. Various natural events, including seasonal fluctuations, extreme climatic events, and depth, can affect light interactions in marine environments. Nevertheless, urbanisation and alterations in coastal catchments have introduced additional light-related stressors for seagrass ecosystems, such as increased nutrient load resulting in opportunistic algae blooms that can smother and shade seagrass, or higher turbidity (caused by suspended sediment) that can increase the attenuation of light in the water column. These conditions can lead to light limitation, which, if substantial and persistent, may result in a significant decline in seagrass abundance (Schwarz, 2004; Zabarte-Maeztu et al., 2021c).

Seagrass requires high light levels to maintain healthy photosynthetic activity, typically around 25% incident radiation (Morrison et al., 2014; Matheson, 2022). In estuarine environments, riverine sediments from rainfall events and fine terrigenous sediments from anthropogenic development are considered the main sources of elevated turbidity in these ecosystems, and their negative impact on light penetration through the water column can limit benthic photosynthesis (Drylie et al., 2018). Many seagrass declines have been linked to elevated water column turbidity and reduced irradiance (e.g. Moksnes et al. 2017).

The quantity and quality of light can generate morphological and physiological changes. Under light-limited conditions, seagrasses may experience a decrease in photosynthesis rate, to which they may respond through morphological adaptations, such as increased canopy height and broader, thinner, and smoother leaves, as well as decreased shoot density (Flowers et al. 2023). Intertidal seagrasses can mitigate high turbidity by increasingly relying on primary productivity during emersion at low tide (Drylie et al., 2018). Intertidal areas are now important refuges of primary production in turbid estuaries, particularly at higher tidal elevation. Although seagrasses can adapt to decreased light levels, overall, they show reduced biomass, shoot density and growth rate (Bulmer et al. 2016; Zabarte-Maeztu et al., 2021c).

Quantifying the light climate experienced by seagrasses is challenging. Not only does incident irradiance vary through time, but so does water depth, turbidity and light attenuation properties, typically all on different time bases (e.g., Zimmerman, 2006). Other related variables include temperature and the shape of a seagrass plant which changes from upright when immersed to collapsed when dry. For this reason, attempts to establish a mechanistic approach to quantifying the role of light in controlling seagrass distribution, and developing a management response that seeks, for example, to establish a turbidity regime to achieve a specific light level, is difficult and very site specific. An empirical approach is more often used, when the observed depth limit of seagrass is assumed to be set by a minimum light flux (over a defined period), and the light at this depth is taken to be the minimum light dose over that defined period to ensure survival. Using this minimum light dose approach, subtidal Z. muelleri during summer was interpreted as requiring 4.91 mol m-2 d-1 (Bulmer et al. 2016), whilst a second study identified 7.3 mol m-2 d-1 resulted in significant above-ground biomass accrual and could form the basis of a minimum summer light dose requirement (Matheson 2022). Saturation irradiance values (the level at which growth is maximal) measured in Tauranga Harbour seagrass beds identified the median value ranged between 16.6–17.3 mol m-2 d-1 (Flowers et al. 2023), much higher than the minimum light dose requirement. Decreases in PAR below a range of 26–35 mol m-2 d-1 resulted in decreases in gross primary productivity (Flowers et al. 2023), which has been previously identified as a critical threshold for estuary functioning, correlated to shifts in the structure and functioning of intertidal sandflats (Thrush et al. 2021, Gammal et al. 2022). This highlights the importance of maintaining high water clarity, as there are significant flow-on effects from reduced primary productivity on the marine ecosystem.

In the Tauranga and Ōhiwa Harbours, seagrass losses are related to various water quality factors (Park, 2016). Light stress, in combination with increased sediment muddiness, has impacted some seagrass communities in the region. For example, seagrass sampled at some locations in Tauranga Harbour displayed wider and thicker leaves and a highly negative $\delta^{13}C$ isotope value indicative of light-limited growth (Crawshaw, 2020). There is limited information around the light availability in Bay of Plenty estuaries, however, a new project under the Sustainable Seas National Science Challenge is investigating the use of satellite imagery to map spatial stressor footprints, such as turbidity or light attenuation (Shao et al. 2023a). A combination of the use of satellite imagery and *in situ* monitoring of the light environment will provide an understanding of the current light conditions for seagrass. This could be developed into a light monitoring framework, used to identify critical riverine inputs causing low light conditions, enabling strategic land management intervention.

In summary, light requirements for seagrass in New Zealand have been established through various studies and indicate the current light environment (and ability for seagrass to offset photosynthetic requirements during emergence periods) is suitable for seagrass in some estuaries. A focus on reducing catchment sediment loading will be important to maintaining or improving the light environment in estuaries to support healthy seagrass growth and production.

3.3 Nutrients

As with all plant species, nutrients such as nitrogen and phosphorus are critical components to support healthy seagrass growth (Lapointe et al. 2004; Udy & Dennison, 1997). In coastal environments, nitrogen availability is important in regulating primary production (Herbet, 1999), because they are often nitrogen limited (Howarth, 1988). Nutrient loading to estuaries is increasing due to agricultural intensification (Heggie & Savage, 2010), and from other human based sources (sewage effluent/stormwater outfalls/amenity fertilisation) (Turner & Schwarz 2006) resulting in eutrophication, or nutrient enrichment of estuaries. Seagrasses obtain nutrients from both the water column and from sediment porewater (Short, 1987; Romero et al. 2006) and thus both waterborne and sediment-sequestered nutrients are available.

Nutrient loading effects on seagrass have been described as a self-accelerating process, with multiple indirect effects compounding on seagrass health (Figure 6) (Burkholder et al. 2007; Duarte 1995). Nutrient loading to seagrass habitats can result in reduced water clarity through excessive growth of opportunistic phytoplankton, and proliferation of epiphytic or macroalgal species, all causing reductions in light reaching the leaves of seagrasses (Neckles et al. 1993, McGlathery 1995), resulting in loss or degradation (Burkholder et al. 2007). Over extended periods of time, increased nutrient enrichment of water can lead to the elevation of nutrients within the sediment porewater (Worm, 2000). Observational studies suggest that seagrass growth is negatively correlated with increased nitrogen loading (Hauxwell et al. 2003, Lapointe et al. 1994), and conditions can become toxic to seagrass if nutrient levels persist for extended periods of time, including ammonium and sulphide accumulation that are typically related to sediment anoxia under enriched conditions (Burkholder et al. 2007). Experimental porewater nitrogen loading studies indicate seagrass resilience to increasing sediment nitrogen enrichment is dependent on seagrass biomass, and resilience may be lost between 140 and 285 g DW m^{-2} (Gladstone-Gallagher et al. 2018).

The existing sediment grain size can also be important, where seagrass in sandy sediments with a low above-ground biomass, have been shown to have a lower resistance to nutrient enrichment (via ammonium toxicity) compared to higher biomass seagrasses in muddier sediments (Li et al. 2019).

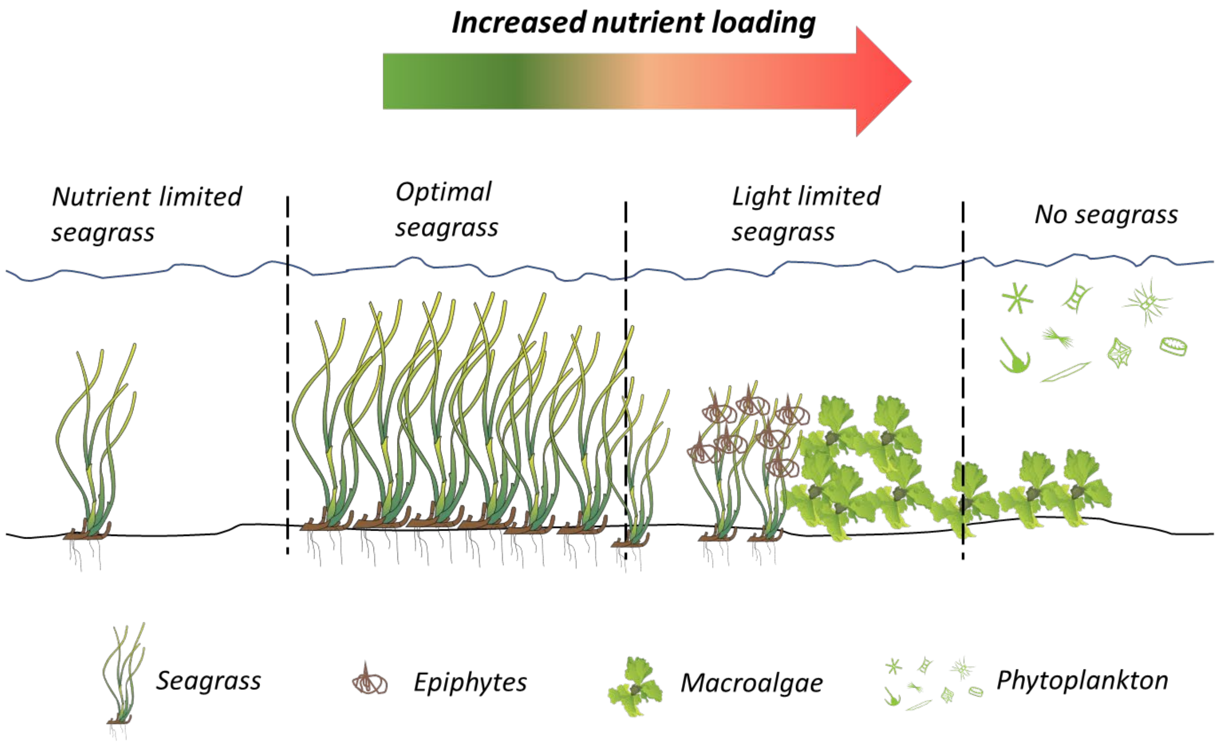


Figure 6 Conceptual model (cartoon) of the effect of increased nutrient loading on seagrasses. Based on Kennish 2009.

Seagrasses have low nitrogen loading tolerances, with a range of work showing a threshold catchment N loading rate above which seagrass does not grow of 100 kg N ha^{-1} y^{-1} (27.4 mg m^{-2} d^{-1}) (Schallenberg et al., 2017, Park, 2018), and another New Zealand study indicated that below 50 mg m^{-2} d^{-1} seagrass response becomes unpredictable (Robertson & Savage 2021). Above this threshold limit, seagrass is sparse and macroalgae and/or phytoplankton communities dominate (Robertson & Savage 2021). However, in the Bay of Plenty, TN loading thresholds summarised in Park (2018) show loss of seagrass can be observed at nitrogen loading rates as low at 10 mg m^{-2} d^{-1}, and this may be a multiple stressor effect of increasing fine sediments in the estuaries.

In the Bay of Plenty, each of our estuaries have been assessed for eutrophication susceptibility using the New Zealand Estuarine Trophic Index, with results ranging from B to D Grade. Tauranga Harbour overall was a B Grade, with a moderate susceptibility to nutrient enrichment, however, many of its sub-estuaries were in the D Grade, indicating a very high eutrophication risk and, in these estuaries, high areas of seagrass loss has occurred, likely due to the combination of sediment mud content and nutrient loading (Crawshaw et al. 2022).

Ōhiwa Harbour has similarly been graded in the B Band. Maketū and Waihī estuaries are graded in the D Band, with a very high risk of eutrophication. Modelling in Waihī Estuary identifies that nutrient reductions alone are not sufficient to increase seagrass abundance, with a combination of light availability (driven by grain size, i.e., mud), sediment grain size, and competition from macroalgae being important components to consider (Chakravarthy et al. 2021).

In summary, nutrient inputs to the estuary are likely to have impacts on seagrass health but is also likely to be interacting (often synergistically) with other stressors (e.g., nuisance macroalgae blooms, sedimentation) which may worsen impacts. Flow-on effects of nutrient enrichment (such as growth of nuisance algae and increased presence of parasitic epiphytes and other fouling species), are discussed more in the sections below.

Bay of Plenty Regional Council's implementation of the NPS-FM is proposing significant nutrient reductions to two estuaries severely impacted by eutrophication (Maketū and Waihī estuaries) to support the potential return of seagrass and improve estuary health. In other harbours, such as Tauranga and Ōhiwa, best practice guidelines will be implemented to support a reduction in total nutrient loads, with nutrient limits for rivers likely providing significant improvements in the estuary nutrient conditions.

3.4 Competition with other algae/eutrophication effects

3.4.1 Filamentous algae

The growth of epiphytes (i.e., filamentous algae) on seagrass is an indication of excess nutrient inputs (Neckles et al. 1993; Nelson 2017). Filamentous algae grow directly on the seagrass leaves, causing shading and a reduction in light available for photosynthesis, resulting in an eventual decline in seagrass cover (Short et al. 1995, Duarte 1995, Nelson 2017).

Epiphyte growth on seagrass have been observed at numerous sites across Tauranga Harbour (Crawshaw 2020), including large blooms of the filamentous algae *Cladophora sp*. (Figure 7). Although no direct evidence was collected, further visits to these sites indicated a temporary loss of seagrass in the areas where high epiphyte coverage had been present. It is likely that filamentous algae are also impacting seagrass in other estuaries across the Bay of Plenty.

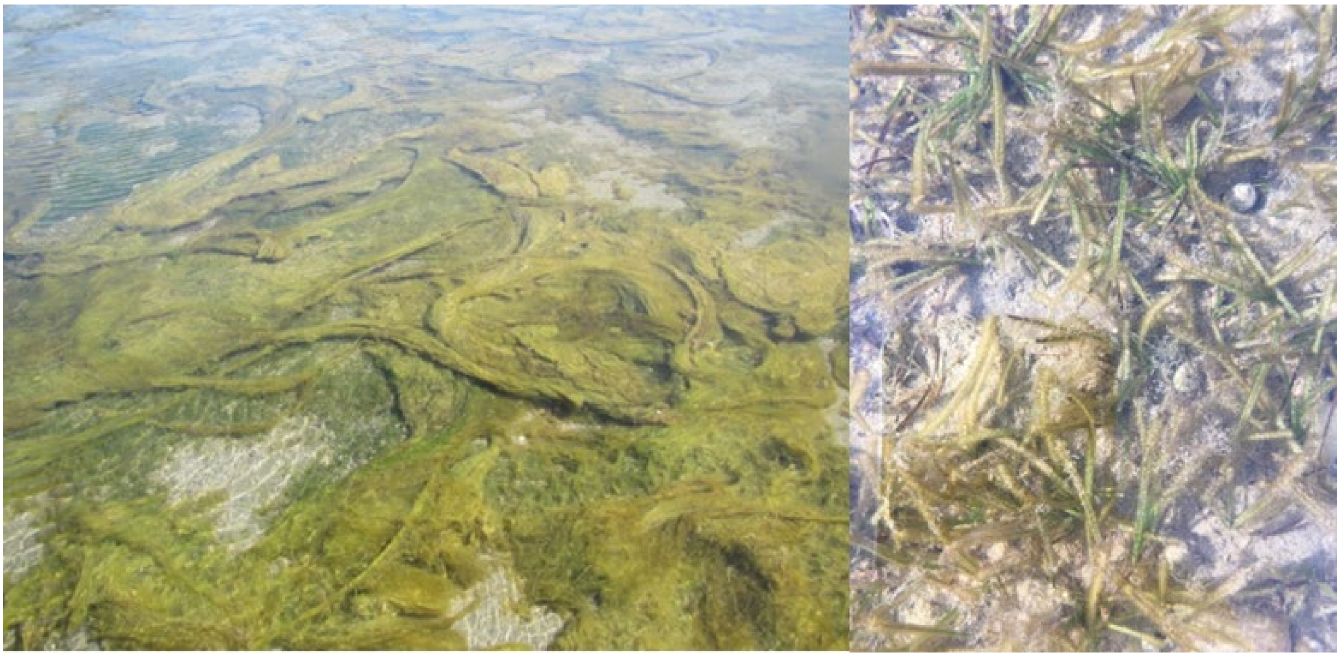


Figure 7 Left: A large bloom of the green filamentous algae Cladophora sp. over a seagrass bed at Bowentown, northern Tauranga Harbour 2018. Right: Typical filamentous algae growth on seagrass in Tauranga Harbour. (Photos: Bay of Plenty Regional Council).

A similar filamentous algae occurrence was observed in Pāuatahanui Inlet, Porirua Harbour, identified as *Chaetomorpha lingustica*, where the algae formed a dense entangled structure over the seagrass (Zabarte-Maeztu et al. 2022). The presence of *C. lingustica* resulted in a 50% reduction in seagrass cover within monitored plots (Zabarte-Maeztu et al. 2022). It is hypothesised that a combination of high light irradiance, high daily maximum temperatures and excess nutrients may facilitate the blooms of *C. lingustica,* consistent with other blooming algae species (Hallegraeff 2003).

There is currently a lack of understanding of how widespread problematic epiphyte growth is in the Bay of Plenty. A new seagrass monitoring framework has been developed by the Coastal Special Interest Group (Shanahan et al. 2023), and a seagrass monitoring programme established in the Bay of Plenty under the Natural Environment Regional Monitoring Network (NERMN). This programme includes annual monitoring of numerous seagrass sites to further our understanding of the prevalence and intensity of filamentous algae growth on seagrass in the Bay of Plenty.

3.4.2 Nuisance macroalgae

One of the most common symptoms of eutrophication in coastal ecosystems is the blooming of macroalgae (Valiela et al. 1997). Blooming macroalgae can have a range of detrimental effects to coastal ecosystems, such as noxious odours, reduction in sediment oxygen conditions, displacement of seagrass and reduced benthic biodiversity by physical smothering and light reduction, fish kills, and nutrient release, through the breakdown of decaying organic matter (Norkko & Bonsdorff, 1996, Teichberg et al., 2010, Niemand, 2018). Coastal eutrophication is largely driven by increased nutrient loads, particularly nitrogen (Howarth 2008). Two of the primary blooming algae in New Zealand estuaries are sea lettuce (*Ulva* spp.) and *Agarophyton* (formerly *Gracilaria*) (Stevens et al. 2022, Robertson et al. 2016).

Tauranga Harbour has historically experienced problematic growth of sea lettuce (*Ulva* spp.) across the intertidal flats (Park 2011), although the intensity of the blooms has decreased since monitoring began in the early 90s (Crawshaw 2021). The main regions where *Ulva* has bloomed in the past are also locations of large seagrass beds (Otūmoetai, Ongare Point, Grace Road). Sea lettuce blooms of low coverage or thickness appear to have minimal impacts on seagrass beds, due in part by the highly mobile nature of sea lettuce, being
redistributed by wind and waves during each tidal cycle. Large amounts of sea lettuce can, however, lead to the fragmentation and degradation of seagrass beds, either directly by smothering the seagrass, or by shading it from sunlight (Frankenstein 2000). The primary impacts to seagrass beds occur with dense accumulations of sea lettuce that limit the amount of light reaching the seagrass beds, cause anoxia and associated toxic conditions, which has been observed frequently along the upper tidal margins of Tauranga Harbour during large sea lettuce bloom years (Figure 8).

Figure 8 Dead seagrass from sea lettuce smothering at Otūmoetai, Tauranga Harbour in 2017.

In estuaries such as Maketū and Waihī, growth of *Agarophyton* has had severe impacts on the estuary ecology and seagrass habitats (Park 2018a), associated with declining water quality and loss of extensive saltmarsh and wetland ecosystems. *Agarophyton* is also present in low biomass in Tauranga and Ōhiwa Harbour where it is expanding. *Agarophyton* can form dense beds with the algae entrained into the sediments (Figure 9 left), which promotes the settling of sediments. Over time, the algae can reach a biomass limit and decay, leading to sediment anoxia and growth of sulphur reducing bacteria, leaving behind an anoxic sediment soup limiting regrowth of anything in the area (Figure 9 right) (Stevens et al. 2022). This creates a legacy effect of eutrophication that can take a long time to recover from.

Figure 9 Left: Dense beds of Agarophyton in Waihī Estuary 2020. Right: Severe anoxia from decaying Agarophyton in Waihī Estuary 2016.

The extensive impacts of both sea lettuce and *Agarophyton* growth in Bay of Plenty estuaries indicate nutrient enrichment is a key stressor to seagrass beds (and overall estuary health). Freshwater nutrient limit setting under the NPS-FM will go some way to address current nutrient loads, however, a legacy effect will remain, in particular in areas where expansive macroalgae beds exist. Macroalgae remediation options were considered in a modelling project for Waihī Estuary (DHI 2022), which identified that catchment nutrient reductions, combined with a one-off active macroalgae removal across the estuary would result in a decrease over time in macroalgae cover, and also a return and increase of seagrass growth (up to 39% after three years). Active estuary restoration may be required in highly degraded estuaries following land management interventions, to support the recovery of key habitats such as seagrass beds, and overcome the legacy effects of eutrophication.

3.4.3 Neptune's necklace (*Hormosira banksii*)

An estuarine ecotype of the fucoid alga Neptune's necklace (*Hormosira banksii*) has been observed growing on top of seagrass beds in a number of locations in Tauranga Harbour (Waipu Bay, Wairoa River/Matua, and Athenree) (Crawshaw & Shalier 2023 in review) (Figure 10). *Hormosira banksii* is native to New Zealand and south-eastern Australia and is most commonly observed on rocky shores (Adams 2007; Womersley 1967, 1981), and (until now) there are no reported cases of it growing amongst seagrass, although examples exist of establishment in temperate Australian mangrove forests (Bishop et al. 2013). *Hormosira banksii* is an ecosystem engineer, providing complex three-dimensional structure similar to that which seagrass provides, and is often associated with increased mollusc abundance and species richness (Bellgrove et al. 2017; Coleman et al. 2018; Bishop et al. 2013). Existing structure in estuaries (such as seagrass) may act to trap the free-living algae, and through vegetative growth and reproduction (fragmentation), allowing various species to increase in density and create dense colonies, as reported in Australian mangrove pneumatophores (Bishop et al. 2013; King 1981).

Figure 10 Top: Dense Hormosira banksii accumulations on seagrass beds at Athenree sandflats, Tauranga Harbour (December 2022). Photos: Bay of Plenty Regional Council. Bottom: Aerial imagery of Waipu Bay in 2021 with dense accumulations of Hormosira banksii evident as brown-orange areas. Photo credit: PhotoOblique.

Hormosira banksii has been recorded in Tauranga Harbour at least once during an ecological survey (Park & Donald 1994), but these populations were since thought to have declined in extent (S. Park pers. Comm). Examination of aerial imagery and field surveys have identified there is high current coverage of the algae, with a distinct presence on seagrass beds. It creates dense mats in places and hosts a range of marine biodiversity.

In Waipu Bay, distributions of *Hormosira banksii* were evident as early as 1963 and appear to be increasing over time with proportionally higher cover in the last few years, compared to the historical imagery that was available (in part possibly due to better aerial imagery) (Crawshaw & Shailer, in review). Of the surveys conducted so far, the median wet weight cover of *Hormosira banksii* was 242 g/m^2, with a maximum recorded wet weight of 14,000 g/m^2. In lower densities, *Hormosira banksii* appears to have minimal impact on seagrass or sediment health, however, at higher densities it appears to smother the seagrass, resulting in anoxic patches that in some instances were large enough to be observed in the aerial imagery.

The seagrass beds appear to act as a trapping structure to allow small pieces of *Hormosira banksii* to become established and grow/spread, with the areas affected appearing to be primarily limited to shallow intertidal areas near the two entrances to Tauranga Harbour. Research indicates this species is unlikely to be an indicator of nutrient enrichment, but rather an indicator of good water quality (Bellgrove et al. 2017, Coleman et al. 2018, Doblin & Clayton 1995), and represents a unique assemblage within New Zealand. Further research is required to understand the impacts on harbour ecology and drivers of the increased presence and cover, however, at current coverage and densities the two species appear to be co-existing.

3.5 Anthropogenic physical disturbances

Activities in the marine environment such as the construction of hard physical structures on the seafloor (e.g., breakwaters, docks, pipelines), land reclamation, moorings and anchoring sites, dredging and coastal developments are recognised as disturbances leading to immediate seagrass loss (Turner & Schwarz 2006). In the Bay of Plenty, many of these activities are now captured under the Regional Coastal Environmental Plan and require the management of effects under the Resource Management Act (RMA) (avoid, remedy, mitigate) of such activities on or near seagrass beds.

Ongoing physical disturbances in the marine environment such as vehicles driving over seagrass can result in long term loss of seagrass, such as is visible in Figure 11A at Otūmoetai, 60 days following the vehicle travelling over the seagrass. In areas where boat access has been provided onto the estuary, there is often a clear loss of seagrass in the area that the vehicle traffic regularly travels over to reach the channel (Figure 11B, C). In some instances, it is evident from aerial imagery that vehicles use the boat launch to then access other areas of the estuary, leaving large scars throughout seagrass (e.g., Ongare Point). Monitoring in Nelson/Marlborough estuaries indicate that damaged seagrass can take several seasons to regenerate, and repeated disturbances resulting in long term damage or loss (Gillespie et al. 2011; Sunde et al. 2017).

Figure 11 Physical impacts on seagrass in Tauranga Harbour. A: Tyre tracks through seagrass 60 days following movement. B: Permanent tracks through seagrass from boat launches and stormwater discharge points. C: Vehicle tracks through seagrass and boats moored in seagrass beds. D: Subtidal seagrass regrowing around swing moorings.

With the rise of boating activities across Tauranga and Ōhiwa harbours, there is increasing pressure to provide moorings for boats. There are currently ~450 swing moorings located across Ōhiwa and Tauranga harbours, with designated mooring zones identified in the Regional Coastal Environment Plan. Moorings are sold from owner to owner, and new moorings are rarely established. The majority of these are swing moorings, where an anchor point holds a chain, which is sufficiently long for the tidal rise and fall. Conventional swing moorings can impact the seafloor surrounding the anchor, via the area swept by the chain (Unsworth et al. 2017). Repeated scouring can clear a circular mooring scar in the seabed, which is particularly evident in regions with subtidal seagrass (Clark & Crossett 2019; Morrisey et al. 2018). Where anchorages are not provided, single mooring anchors are used, and the mooring chain and anchor can be dragged along the seabed with changes in tidal movements and wind directions (Morrisey et al. 2018; Turner & Schwarz 2006). Chain scour can loosen sediment, making them more vulnerable to erosion and alteration of texture by water movements (Morrisey et al. 2018).

The mooring zones are generally in channels where no subtidal seagrass is present, with the exception of Te Puna Estuary where subtidal seagrass beds have begun re-establishing around the moorings (Figure 11D). With the ongoing rise of boat usage and vehicle access to launch boats across Tauranga Harbour, there may be an opportunity to focus some investigations into how we can best manage these activities to minimise impacts on seagrass beds. Where existing moorings are located within seagrass habitats, there are more environmentally friendly mooring designs available to minimise scouring impacts (Morrisey et al. 2018). Outreach could be undertaken to the boating community to raise awareness around the significance of seagrass habitats, with communications to anchor outside of seagrass where possible, or designation of “safe anchoring zones” or anchoring guidelines to minimise impacts. Additionally, guidance could be provided at boat launching zones requesting vehicles only travel within designated pathways and minimise movements outside of the launching area.

3.6 Seagrass wasting disease

Fungal wasting disease manifests as patches of darkened seagrass leaves and has been confirmed to be linked with the protozoan *Labyrinthula zosterae* (Berthelsen et al. 2016; Burdick et al. 1993) (Figure 12). This disease has spread globally and been linked to seagrass losses internationally (Lorus & Milne, 1951; Ziegler et al. 1961; Ralph & Short 2002). This disease is likely to naturally occur in seagrass meadows, but outbreaks can occur with low light, warm temperatures and lower salinity, and seagrass can be more susceptible when stressed (Matheson 2009, Hughes et al. 2018, Groner et al. 2021). It has been recorded thus far in Nelson, Marlborough and the Coromandel (Berthelsen et al. 2016, Sunde et al. 2017; Clark & Crossett 2019).

Ad hoc surveys of seagrass in Tauranga Harbour have identified darkened patches on leaves, however, no histological testing has been completed to date to confirm the presence of fungal wasting disease. This type of analysis will be investigated if darkened seagrass blades are identified during future seagrass surveys.

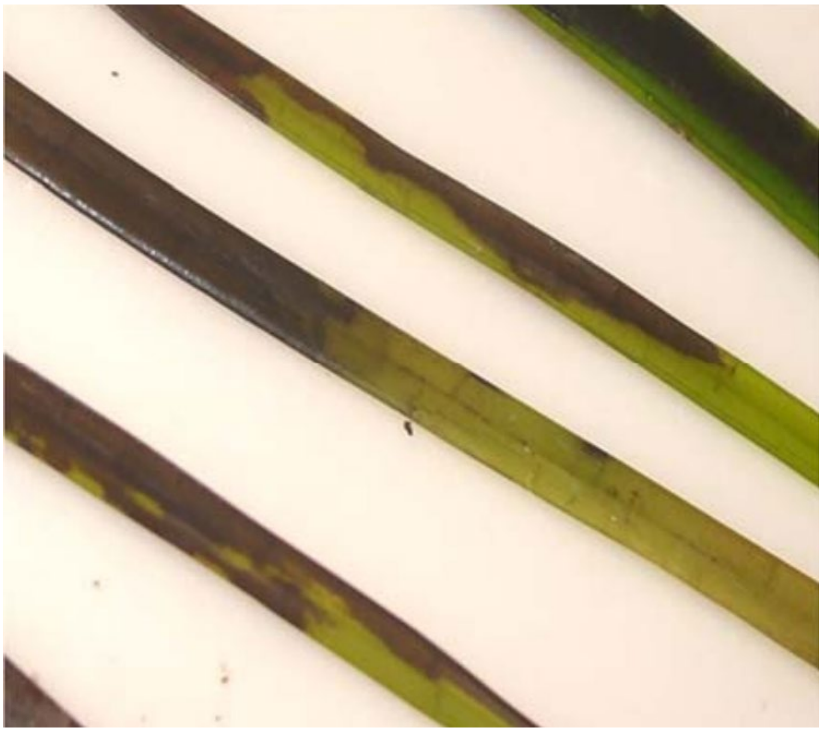


Figure 12 Blackened seagrass blades indicating L. zosterae. Source: Berthelsen et al. (2016).

3.7 Climate change

Climate change is the most widespread anthropogenic threat to marine ecosystems (Unsworth et al. 2019). Climate change could place additional pressure on seagrass habitats through exposure to more frequent and intense storm events, poorer water quality (increased sediment runoff), extreme temperatures and a loss of suitable habitat due to coastal squeeze (Unsworth et al. 2019, Turner & Schwarz 2006; Short & Neckles 1999). The changing climate is an additional factor outside of an already degraded habitat subject to multiple stressors.

High temperatures can result in thermal stress to seagrass resulting in a reduction of leaf and shoot density, above-ground biomass and leaf senescence (York et al. 2013). Recent marine heatwaves in 2022 could contribute to seagrass losses, as bleached seagrass has been reported washing up on shore (Mattheson, per comms), and the predicted increasing frequency and intensity of marine heatwaves (Behrens et al. 2022) could threaten seagrass meadows (Zimmerman 2021). In addition, as this species of seagrass is primarily intertidal, it will also exhibit stress from increasing air temperatures (Rasheed & Unsworth 2011, Marba et al. 2022). *Zostera muelleri* has an upper temperature tolerance of ~30°C (York et al. 2013, Collier et al. 2011), therefore, future warm weather events (particularly if they occur during midday low tides) may further stress intertidal seagrass beds.

Rising sea levels may lead to longer submergence periods, in some instances moving intertidal beds to permanent subtidal habitat. Sea level rise predictions for the Bay of Plenty under RCP8.5 median project 0.52 m of sea level rise by 2070, and 1.25 m by 2130 (Pearce et al. 2019). Sea level rise will squeeze intertidal zones against hard structures or natural topography in modified estuaries, slowly elevating the low tide mark (Borchert et al. 2018; Rogers et al. 2019). A recent paper focusing on Tauranga Harbour highlighted that squeezing of intertidal areas by sea level rise will likely alter the distribution of key habitats and communities, such as shellfish beds (Rullens et al. 2022), due to changes in salinity, biomorphodynamics (interactions of biology, morphology and physical forcing) and geomorphology, leading to changing submerged/emerged periods for intertidal sediments. The ability for seagrass to adapt from an intertidal to subtidal environment will be dependent on light availability, and modelling suggests that seagrass productivity will increase with longer submergence periods if water clarity is good (Flowers et al. 2023; Shao et al. 2023a), however, this requires management of the ongoing stressors of sedimentation and elevated turbidity in coastal environments (Thrush et al. 2004).

At a regional level, there are few options available to slow the rate of climate change, however, there are management options available to help mitigate the potential effects and prepare for a changing climate. National climate risk assessments (Ministry for the Environment 2020) highlighted intertidal habitats as having the most significant risk in the natural environment, and adaptive management is required at regional scales. Habitat protection policies could be designed to incorporate projected future distributions of seagrass meadows, such as potential shifts in suitable habitat due to sea level rise (Unsworth et al. 2019), and policies to improve and maintain water quality (Zimmerman 2021). Councils can plan for the landward migration of estuarine habitats, and can use non-regulatory means, such as incentives, to implement managed retreat policies in the absence of national direction (Rullens et al. 2022), which BOPRC is currently investigating for saltmarsh habitat restoration (Crawshaw & Fox, 2022). SLR predictions and land use layers can be utilised to identify areas where land retirement could occur, creating areas for natural migration of habitats in the long term (Crawshaw & Fox, 2022).

Environmental targets for water quality to support seagrass (e.g., light requirements) should be developed and allow for cumulative impacts and ecological feedbacks (Maxwell et al. 2017). In some estuaries (i.e., Waihī Estuary) modelling for the NPS-FM has incorporated seagrass environmental requirements (light, nutrients and sediment grain size) in the development of policy limits and targets for the catchment. Internationally, improving water transparency and limiting nutrient loading has been important for recovery of seagrass (Zimmerman 2021). Catchment sediment loads are modelled to increase with climate change in the Bay of Plenty, even under best catchment management practice scenarios (Park et al. 2022). Addressing sedimentation issues to improve water clarity and the light climate for seagrass will be essential, and restoration of habitats such as subtidal shellfish reefs (such as horse mussels) may support increased water clarity (Rullens et al. 2022).

3.8 Herbivory by swans and geese

Herbivory of seagrass is wide ranging, from smaller crustaceans and gastropods, to urchins, fish, waterfowl, marine reptiles and mammals (Valentine & Heck 2021), highlighting the importance of this habitat in the marine food web. As much as half (or more) of net above-ground production of seagrasses is consumed by herbivory, however, expansive seagrass beds prevail regardless of this intense grazing pressure (Heck & Valentine 2006, Valentine & Heck 2021). Some studies suggest that there could be compensatory or resistance mechanisms to minimise the impact of grazing by the seagrass (Valentine & Heck 2021) or indirect effects (such as availability of alternative resources (macroalgae), increased predation, and population level competition for resources)(Buñuel et al. 2023).

In the Bay of Plenty, the rise of swan herbivory on seagrasses in and surrounding estuaries has been an ongoing topic of contention. This section focuses on herbivory by two large waterfowl that frequent Bay of Plenty estuaries, black swan and Canadian geese, and the potential impacts on seagrass based on previous research.

3.9 Existing research on swan/geese impacts on seagrass

3.9.1 Black swans (*Cygnus atratus*)

Australian black swans (*Cygnus atratus*) were introduced to both the North and South Islands of New Zealand in large numbers (over 100) prior to 1870 with further releases later on (Miers & Williams, 1969). In the South Island, the first recorded introduction was by the Nelson Acclimatisation Society in, or just before 1864 and seventy birds were released in the Southland and Otago Acclimatisation Society districts prior to 1870. Introductions were made in the North Island from 1864 onwards. Prior to the extensive introduction of the Australian black swan, New Zealand did have a native black swan (Pouwa) distinct from but descended from the Australian *C. atratus* (Rawlence *et al*. 2017). A DNA and morphometric analysis by Rawlence *et al*. (2017) show this swan (*C. sumnerensis)* was a larger (67% heavier) semi-terrestrial bird with shorter wings that was likely on a path to flightlessness. The Poūwa was hunted to extinction by the mid-fifteenth century in the north and South Island and the mid-seventeeth century on Chatham Islands with bones found in Maori midden deposits (Sagar *et al*. 1995).

It has been postulated that around the time of the extensive releases of Australian black swan, that they self-introduced (Kirk 1895, Miers & Williams 1969). Although it would be a remarkable coincidence, it is possible as Rawlence *et al*. (2017) found some evidence to suggest “vagrant” *C. atratus* were present in New Zealand shortly after the extinction of *C. sumnerensis* but failed to establish. The hypothesis of self-introduction in the late 19th Century (Miers & Williams, 1969) postulates that they could have arrived in 1867 at the same time as the Australian white eye (*Aythya australis*) arrived. However, large numbers of *C. atratus* had already been introduced and established by this time. Due to the extensive evolutionary divergence of the native Pouwa from *C. atratus*, Rawlence *et al*. (2017) make the point that the introduction of the Australian black swan is not an ecological replacement and in the anthropogenically modified landscape, they could be an unwanted pest.

The black swan is classified as native resident non-threatened under the New Zealand Threat Classification System (Robertson et al. 2021). The national black swan (*C. atratus*) population appears to have been stable over the last 20 years (McDougall et al. 2023). About 60,000 swan were counted nationally in 2022 (McDougall et al. 2023) which is on par with 1979–1981 counts (Williams 1980, 1981) and less than counts prior to the 1968 Wahine storm (Williams 1980) when the population on Lake Ellesmere/Te Waihora alone was 70,000 (Marchant and Higgins 1990, Heather and Robertson 1996) while Williams (1979) gives a range of 40,000–80,000 during the 1950's and 1960's. Current management of swan populations in the Bay of Plenty and Rotorua Lakes (Eastern Fish and Game A1 management unit) is by relaxed season conditions, with a 16-week season and no daily limit (McDougal 2022b). Because of the black swans' delayed maturity and small clutch size, they cannot withstand heavy hunting (Williams 1981), therefore, careful harvest management is required.

The collapse of the black swan population on Lake Ellesmere/Te Waihora after the Wahine storm likely corresponds to the failure of the tall growing aquatic macrophytes that were devastated during the storm and subsequently failed to recover, presenting a loss of the population's primary food source. In contemporary times, eutrophication and siltation of some key lakes and lagoons has seen a collapse of macrophyte beds resulting in swan vacating these habitats e.g., Lake Waikare (McDowall 1994), and Lake Whangape (D. Klee pers com.), in the Waikato, and Whakaki Lagoon, Wairoa region (M. McDougall pers com.). There is a strong correlation between the decline in the Whangapae swan population and the increase in population size on Tauranga Harbour (r=0.58 p= 0.0012; McDougall 2022). Further, it is likely that some of the Waikato birds now frequent the Northwestern harbours (D. Klee pers com.).

Swans are omnivorous, foraging predominately on a large variety of aquatic plant species, (Marchant and Higgins 1990) including estuarine species such as, sea lettuce and seagrass (Sagar et al. 1995), and may forage on grass when submerged macrophyte food supplies are reduced (Williams 1979, Marchant and Higgins 1990). In Bay of Plenty estuaries, swan have been regularly noted as feeding on benthic macrofauna (Park pers com.). Monitoring of the swan shows that some populations (such as Tauranga Harbour) have high seasonal distributions (McDougall 2022) as they migrate to moult and nest. Williams (1980) reports swan banded on the Rotorua lakes being resighted at Lake Wairarapa, Fairwell Spit, and as far south as Otago. Band recovery data shows dispersal from the Waikato lakes to, amongst other places, Kaipara Harbour, Rotorua Lakes, and Tauranga Harbour (Williams 1977). It is unknown if these movement patterns are still relevant today following the collapse of aquatic macrophyte beds in many of the key Waikato lakes.

Black swans forage for seagrass primarily, when the seagrass is covered by a shallow layer of water (Dos Santos et al. 2012) and flocks track the rise and fall of the tide. Grazing creates circular de-vegetated patches, with 92% of shoots, 25% of roots, and 99% of rhizomes removed (Dos Santos et al. 2012). This intense grazing creates gaps/hollows that may increase hydrodynamic disturbance and erosion (Eklof et al. 2015), leading to sediment destabilisation and greater erosion potential (Dahl et al. 2021). Seagrass biomass recovers slowly from swan grazing, as this requires regrowth of the rhizomes into the denuded areas and can take years to recover from heavy grazing events (Dos Santos et al. 2013). The average seagrass consumption rate by swans at two sites in Tauranga Harbour was estimated based on faecal analysis at 394 g dry mass per swan per day (Dos Santos et al. 2012). These authors noted that grazing pressure varied between locations within the harbour and estimated that in 2009-2012, swan grazing consumed 15% of the average annual seagrass biomass in Tauranga Harbour. The method used to estimate grazing did not consider seagrass uprooted but not consumed.

3.9.2 Canada Geese (Branta canadensis)

Canada geese (*Branta canadensis*) were first introduced to Wellington in the 1870's but failed to establish (McDowall 1994). A later introduction in 1905 were propagated and then distributed throughout the country (McDowall 1994). Currently, management of this species is not conducted by DOC or Fish and Game, and Canada geese are unprotected wildlife.

In New Zealand, Canada geese are year-round resident birds (White, 1986), found on pasture near lakes, rivers and estuaries (Smith, 2019). Canada geese are primarily pasture feeders (Batt 1992, Williams et al. 2006) but also utilise other plants such as submerged macrophytes or seagrass (Ferries, 2021, Kollars et al. 2017). Isotope analysis of Canada geese feeding on north-western harbours (e.g. Kawhia and Raglan) showed that pasture was the dominant food contributing to 75–93% to all three tissue types (plasma, red blood cells and feathers) (Ferries 2021). However, Ferries (2021) did not examine the impact of geese on *Zostera sp*. over the summer months when he postulated, that they would be more likely to feed on *Zostera* due to pasture being less palatable at this time. Geese remove seagrass from the sediment by dabbing, grubbing and trampling behaviour (Ferries 2021). These behaviours uproot shoots, and quantities of such shoots are often seen left behind in areas grazed by geese (Figure 13).

Canada geese numbers have increased significantly since the removal of the species from the 1st Schedule of the Wildlife Act 1957, and now occupy areas where previously there were none, however, no counts are actively being conducted in the Bay of Plenty (Matthew McDougall, pers. comms). While Canada geese were classified as a game bird (1^{st} Schedule of the Wildlife Act 1957) hunting was confined to the winter months which meant they were less wary than currently (i.e., they are not being shot at all the time). A significant proportion of the Eastern Fish & Game region population was shot every year (mean annual harvest in the Eastern Region >2,000; Fish and Game unpub. data), and the birds were confined to a relatively stable number of sub-populations (McDougall 2010). Uncontrolled hunting has resulted in the sub-populations fracturing into many smaller flocks which have subsequently increased in size, resulting in what appears to be a huge increase in total population across the Waikato and Bay of Plenty (M McDougall and D. Klee pers com.).

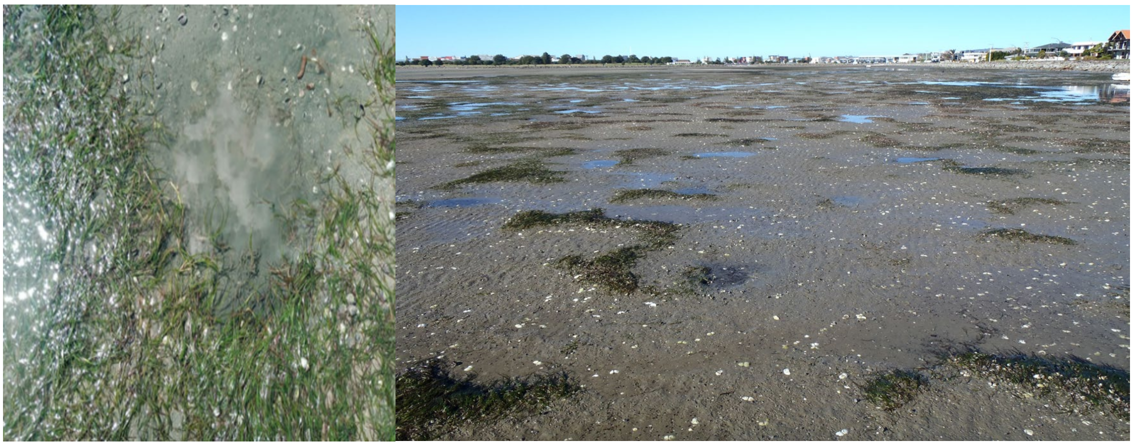


Figure 13 Left: An area recently grazed by Canada geese includes faeces disturbed sediment and uprooted seagrass plants. Right: Swans grazing at Otūmoetai 2012 with >70% of seagrass removed.

3.9.3 Swan and geese numbers and trends

Aerial counts of black swan are conducted annually for population monitoring and setting the game bird season regulation (McDougall 2022a), with swan count data in the Bay of Plenty going back to 1991. The following information is extracted from McDougall 2022a and replotted using data from Fish and Game. The black swan population in Tauranga Harbour has been steadily increasing since 1991 (Figure 14A), with a linear regression, indicating the January population increasing by approximately 150 birds per year (McDougall 2022a). Populations in Tauranga Harbour decrease over the autumn and winter as breeding birds move away to nest (breeding does not occur in any of New Zealand's marine habitats; Williams 1980).

Ōhiwa Harbour also hosts a population of swans, which were counted in 1992 but then excluded from the counts due to the low numbers until counts were reinstated in 2018. It is now showing an increase in numbers (Figure 14B). Maketū and Waihī estuaries also host a small population of swans, that has varied over time, with slightly increasing in some recent years in Maketū Estuary and remaining consistent in Waihī Estuary (Figure 14C,D).

Although populations are increasing in Ōhiwa and Tauranga Harbour, over the wider Bay of Plenty – Rotorua lakes (Eastern Fish and Game A1 management unit) the swan numbers have remained stable over the past 20 years (McDougal 2022b). The increasing swan population in Tauranga Harbour and Ōhiwa is most likely due to the collapse of the macrophyte beds in many of the Waikato Lakes, with swans relocating to find food (McDougal 2022a).

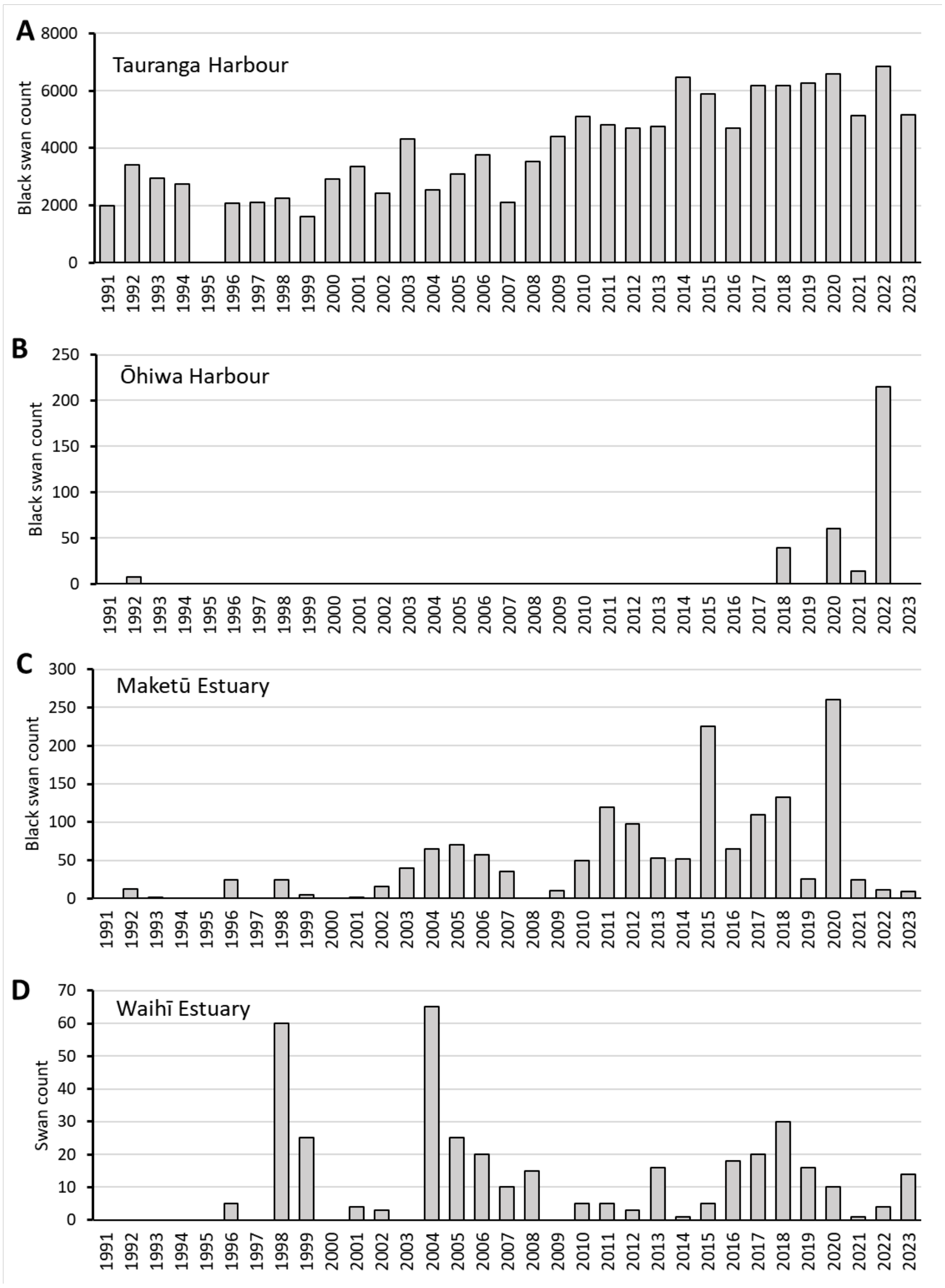


Figure 14 *Black swan aerial counts in January in Bay of Plenty Estuaries between 1991 and 2023 (data provided by Fish & Game, reported in McDougal 2022a).*

3.10 Ecological effects of swan and geese on seagrass

3.10.1 Biodiversity

Due to the total removal of seagrass by swans from sizable patches or more extensive areas at Otūmoetai, the associated physical disturbance and changes to habitat complexity and stability can reduce biodiversity. In Tauranga Harbour, a rarely reported species community association occurs with coralline turf living extensively amongst the seagrass in areas near the harbour entrances. At Otūmoetai, this species association was common in the mid to lower seagrass beds on the intertidal flats in the early 1990's (Park & Donald, 1994). However, following extensive grazing and disturbance of these seagrass beds by swans from the late 1990's onwards, the occurrence of coralline turf in the seagrass beds has declined from 2.6% cover to 0.12% in 2012 (Morrison 2012).

3.10.2 Current estuary estimates of swan grazing pressure

Previous research has estimated the swan grazing pressure in Tauranga Harbour (Dos Santos 2012) to provide an approximate estimate of how much seagrass a colony of swans is able to consume over a yearly basis. The black swan annual grazing pressure (Sg) has previously been expressed as a proportion (%) of the standing seagrass biomass and estimated using the following equation from Dos Santos (2012):

$$Sg = \frac{CR\ x\ n\ x\ t}{Sb * a} \text{ x } 100$$

Where CR is the seagrass consumption rate (394 g DM per swan per day), “n” is the annual swan count across the estuary, “t” is days in the year, “Sb” is the seagrass biomass (g m^{-2}), and “a” is the current aerial extent of intertidal seagrass (m^2). The Dos Santos (2012) study reported an average seagrass grazing pressure by swans of 15% of the standing seagrass biomass, utilising a consumption rate of 394 g DM per swan per day (based on faecal outputs). This feeding rate doesn’t take into account the total loss of seagrass biomass, as it is only based on the seagrass consumed which is predominately the rhizomes (99% removal) and these are a crucial structural growth component of the plants. Taking into account the total biomass of seagrass lost, but not all consumed, the rate based on the work of Dos Santos (2012) is 657 g DM per swan per day. The calculation also uses an annual average which does not separate out the seasonality of swan populations and seagrass biomass, which changes significantly over the course of the year. Utilising the annualised method for assessing swan grazing pressure, the 2022 estimate is 36% of annual intertidal seagrass biomass removed by swan grazing (Table 1), compared to 26% in 2012. This is likely an over-estimate, given the high swan number estimated across the year, therefore, we also look at a range of seasonal measures to provide a more likely scenario across seasons.

Using seasonal 2022 swan counts from Fish & Game (January, April, and August), and seasonal seagrass biomass measurements from Dos Santos (2012), the seasonal black swan grazing pressure was calculated across three time periods. In Tauranga Harbour, the resulting 2022 black swan grazing pressure ranged from a peak in summer of 19%, 6% in Autumn and down to a winter low of 3% (Table 1). Overall, this would represent an annual grazing pressure of 28% of the standing seagrass biomass. The Dos Santos (2012) report indicated that seagrass had a relatively low tolerance for biomass removal, in the range of 19%-20% annual biomass removal, before causing decline of the seagrass biomass the subsequent growing season. Mateo et al. (2006) showed that seagrass persisted when herbivory was <10%, therefore, a conservative approach would be to consider that an herbivory threshold may fall within the range of 10%–20% grazing removal of total seagrass biomass.

To calculate the daily seagrass removal rate (g DM m^{-2} d^{-1}), the average seagrass impact rate per swan (657 g DM $swan^{-1}$ d^{-1}, discussed above) was multiplied by the annual number of swans in the harbour, divided by the seagrass aerial intertidal extent (km^2). This was then compared to the above-ground seagrass production rates calculated by Turner and Schwarz 2006 (1.1 to 2 g DM m^{-2} d^{-1}), which provides a rough indication of how much of the seagrass daily above-ground production is consumed by swans. In 2022, this ranges from 1%-2% in winter, up to a peak of 7%-13% in summer (Table 1), which indicates that the productivity of seagrass is marginal for supporting the grazing pressure exerted by the swans in summer, if spread evenly over the whole harbour.

This analysis of grazing pressure by black swan in Tauranga Harbour in terms of harbour-wide averages for both annual and daily intertidal seagrass biomass removal rates, indicates that the carrying capacity may be, or close to being exceeded. When black swan aggregate into localised areas as noted by Dos Santos (2012), the higher levels of biomass removal may have a greater impact on the recovery dynamics of seagrass. This has been recorded in the past at Otūmoetai (i.e., Figure 13) with significant loss of seagrass and slow recovery times (2.4-4 years) (Dos Santos et al. 2013).

In Ōhiwa Harbour, seagrass extent is continuing to show declines (likely due to sedimentation pressure), whilst there appears to be an increasing number of swans in the harbour. Current grazing pressure is estimated to range from a peak in summer of 29% to a low in winter of 4% (Table 1). Annually, this could represent a grazing pressure of up to 40% and more likely to be exceeding carrying capacity.

For the smaller estuaries such as Waihī and Maketū estuaries, low seagrass cover provides limited food resources for swans, and there are correspondingly low densities of swan in these estuaries. However, the swan grazing poses significant pressure on the seagrass in Maketū in particular, where small patches are regrowing following the re-diversion project, and swans are removing these returning patches as they appear.

The seagrass in estuaries across the Bay of Plenty appear to be playing an important role in supporting the existing swan populations in the region, with birds moving from the Rotorua Lakes and Waikato area to the coast. Although both Tauranga and Ōhiwa harbours have high seagrass extents, the calculations presented above, although improved from the original estimates, still do not accurately take into account the full dynamics of seasonality and swan behaviour (aggregation) impacts on the standing biomass at localised scales and hence potential reduction of seagrass bed extent across whole harbours. This includes loss of the rarer coralline turf and seagrass community association in areas such as Otūmoetai. In the smaller estuaries, there is evidence that swans are limiting the ability for seagrass communities to naturally re-establish following management intervention (the Kaituna River Re-diversion Project), and measures may be needed to reduce the grazing impacts. In Maketū Estuary, due to the very small areas of seagrass installation of mesh to protect the seagrass will be trialled.

Overall, the calculations indicate that waterfowl grazing is having a measurable impact on seagrass beds and requires consideration alongside the multitude of other pressures facing seagrass habitats in understanding seagrass health. Averaging growth and consumption across the vegetated areas of the estuaries does, however, fail to fully capture the potential destructive impact of intense, concentrated grazing in sensitive areas. At present, there is insufficient information to provide any estimate of the potential impact of Canada geese grazing on seagrass, but this will only add to the pressure on seagrass. The seagrass grazing estimates provided above should be considered conservative in terms of a probable under-estimate of impacts, as any reduction in productivity/growth rates and re-establishment of seagrass beds in a fast changing marine climate, deleterious to seagrass ecology, have not been taken into account.

3.10.3 Research gaps for herbivory

There remain a number of unknowns about swan and geese dynamics as they relate to seagrass management. Below are a number of research opportunities to further understand the role of herbivory on seagrass beds.

- Are swan and geese populations resident or migratory, or a mix of both? This project could focus on tagging of swans and geese to determine movements.

- Swan number in the management areas are not increasing, however, birds are migrating away from the Rotorua Lakes regions to the coasts to find food. Are the records sufficient to determine if there are correlations between the inland areas where swans have declined and the loss of freshwater macrophytes?
- How do swans and geese graze in the estuaries, and what controls their movements, therefore, the potential for local impacts versus system wide impacts? This could be addressed by determining the distribution of swans within estuaries or sub estuaries, perhaps using a network of time lapse cameras, and relating this to variables that may control location chose.
- What are the feeding dynamics of geese on seagrass? Observations by Fish & Game tend to indicate they spend time moving around through the seagrass, but preferentially feed in lowland pasture where the fibre content is higher. Analysis of faecal material using DNA markers for a range of species may provide a better picture of food of the two species.
- What role do marine megafauna (e.g., stingrays/eagle rays) play in creating/enlarging devegetated patches in seagrass?
- How does seagrass recover following grazing and how long does this take? Is cover reduction (as measured in surveys) impacted equally to biomass grazing? Does more concentrated and/or larger bare patch creation result in changes to recovery and growth the following seasons and reduce overall intertidal extents?

*Table 1 Estimates of swan grazing pressure in intertidal seagrass beds of Tauranga and Ōhiwa harbours. Tauranga Harbour 2012 estimates by Dos Santos and 2022 estimates for Tauranga Harbour, using the total loss of seagrass from swan grazing (657 g DM per swan d^{-1}) and using an annualised swan number of 5341 (0.78 of 6847 swan count in January) as per Dos Santos 2012. Seasonal variability in swan grazing pressure (as a % of the seagrass biomass) on seagrass in Tauranga and Ōhiwa harbours based on current seagrass extent (expressed over a third of the year, 121.7 days). Seasonal swan numbers were provided by Fish and Game. Seasonal seagrass biomass was estimated from Dos Santos (2012). *Based on an average seasonal swan proportion of 0.17 (winter) or 0.68 (autumn) of the summer count based on the last eight years Fish and Game data.*

Location	Year	Seagrass extent (ha)	Swan number	Seagrass biomass (g m^{-2})	Swan grazing pressure (%)	Daily seagrass removal rate (g DM m^{-2} d^{-1})	Average % of daily seagrass production removed
Tauranga	2012	2933	3700	117	26	0.083	4-8
	2022	3081	5341	117	36	0.114	6-10
	2022 - Summer	3081	6847	92.5	19	0.146	7-13
	2022 - Autumn	3081	4880	227.5	6	0.104	5-9
	2022 - Winter	3081	1164*	110	3	0.025	1-2
Ōhiwa	2022 - Summer	63.7	210	92.5	29	0.216	11-20
	2022 - Autumn	63.7	143*	227.5	8	0.147	7-13
	2022 - Winter	63.7	35*	110	4	0.036	2-3

3.11 Future technology to advance monitoring seagrass extent

3.11.1 Using satellites and machine learning to map seagrass

Introduction

In recent decades, use of satellite remote sensing for collection of information about the earth's surface has grown due to the significant improvements in quality and availability of information provided by a variety of active and passive sensors on orbiting satellites (Devi et al. 2015). The strengths of satellite remote sensing lie in the spatial and temporal scale of data collection; most platforms have orbits designed to cover the globe and return times of a few days are typical. The challenges of remote sensing have traditionally included the low resolution of images (pixel size), the high cost of imagery, the need for good observing conditions for optical sensors, and converting sensed data into usable information.

The challenges are being addressed. Ready access to useful data is being met by the provision of freely available, high-resolution images from international space agencies. For example, the European Space Agency provides open access to data from the Multispectral Imagers (MSI) carried on the Sentinel-2 satellites, that deliver up to 10 m resolution imagery across 13 spectral bands in the visible to short wave infrared part of the electromagnetic spectrum. There are currently two satellites in the Sentinel 2 fleet, and two more are due to launch over the next few years. Frequent overpasses by multiple imagers to some extent also mitigates against the need for good observing conditions (usually this means cloud free) but cannot completely eliminate this issue.

"Classification" is the term used to describe the conversion of remote sensed data into information on the desired surface characteristic. Remote sensing classification falls into two basic techniques: object-based and pixel-based. The object-based technique uses models that 'look' across the image and compares the similarity in shape and size of distinct objects, and groups them into classes of objects (Blaschke, 2010; Roelfsema et al., 2014; Urbański et al., 2009). Pixel-based approaches use algorithms that analyse spectral information of individual pixels within an image and classify them into statistically defined classes (the unsupervised method) or classes defined by the user (the supervised method). In work at Tauranga Moana, classification was to bare, sparse and densely covered with seagrasses using ground truthed data.

Unsupervised techniques have no requirement of prior ground truth data, but understanding how defined objects and classes relate to real features requires post-definition inspection of field sites or images. Supervised classification needs information from field surveys as input for the training phase, where the algorithm is presented with a set of ground control points where the target score (e.g., present-absent, high-low-zero, biomass etc) has been measured and then uses these points to develop a method for prediction of that score from the remote sensing data, and applies this to the full area. A second, independent series of observed points is then used to validate the predictions.

Together, the combination of readily available multispectral satellite imagery with moderately high resolution, tools for image correction and powerful classification models have enhanced the potential for the use of satellite remote sensing for environmental monitoring. With these systems in place, it is becoming increasingly possible to target a wide range of ecosystems, and to exploit the potential for broad area coverage with frequent return.

Remote sensing in Tauranga Moana

Satellite remote sensing of seagrass in Tauranga Moana was investigated in 2021. This research used a supervised, pixel-based approach to explore prediction of cover (presence-absence and high-low-zero cover estimation) and the estimation of blue carbon biomass. Training and validation of the models used an extensive ground truthing campaign in which cover, and biomass were measured around the harbour. The success of this initial mapping exercise led to an attempt to hindcast the evolution of the current distribution of seagrass in Tauranga Harbour (Ha et al., 2021). For historical continuity, Landsat provides a longer record than Sentinel, and in this study images from February or March in 1990, 2003, 2011, 2014 and 2019 were used with a range of ML algorithms, some of which had been developed very recently.

Figure 15 shows how between 1990 and 2001, the distribution of seagrasses appeared to shift, with declines in the northern and southern end and an increase in the central region. To examine this in more detail, maps of the change in seagrass presence-absence over two periods, 1990 to 2011 and 1990 to 2019 were derived (Figure 16) and further highlight the early loss of seagrass in the northern, expansion in the central area in 2011, followed by central decline to the present day. The bulk of recent decline is seen as due to loss of seagrass in the central portion of the harbour. The overlap between aerial photographic estimates (Park 2011) and satellite in 2011 allowed a direct comparison of the two methods, and the satellite estimate of 2380 ha of seagrass is similar to that of 2744 from photography.

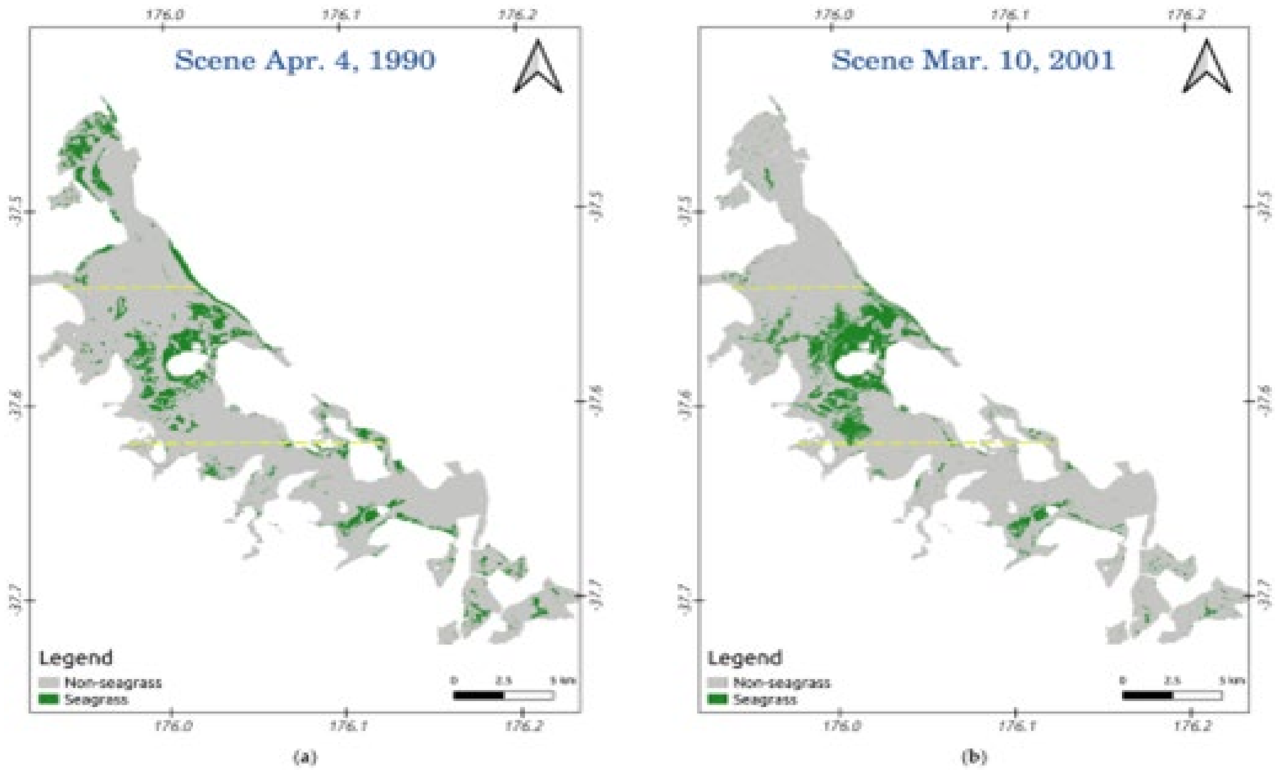


Figure 15 Examples of seagrass distribution in the years 1990 and 2001, using CatBoost machine learning with Landsat imagery. The dotted yellow lines indicate arbitrarily defined breaks into northern, central and southern harbour.

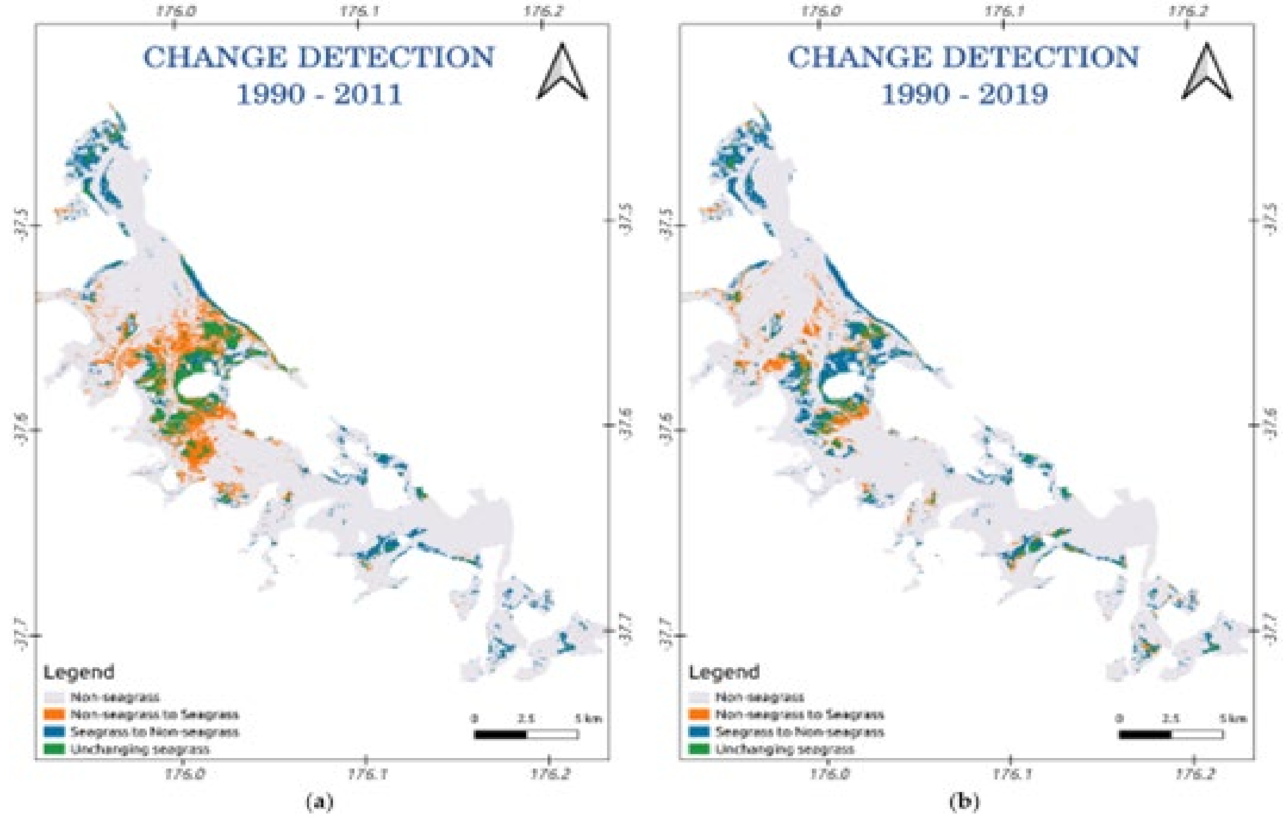


Figure 16 Seagrass change detection between 1990–2011 and 1990–2019.

Use of higher resolution imagery and ever advancing techniques for enhancing analytical precision for satellite remote sensing continues to increase the accuracy with which seagrass cover and density can be resolved from space. The latest research uses 3 m resolution PlanetScope data, obtained under an educational licence, and coupled this with a superpixel approach to sharpen boundary resolution. Superpixel is a technique that gathers pixels with similar spectral and shape features through the process of segmentation and rasterisation, which helps to reduce the number of image pixels, improve the spectral character of the target, and sharpen the seagrass meadow shape, to allow discrimination of seagrass from other classes (Stutz et al. 2018). When applied to Tauranga Harbour, this approach increased accuracy by ~10% (Ha et al 2023). The improved satellite pixelation, coupled to the enhanced analytical pipeline, allows a higher resolution map to be developed (Figure 17), including the classification between dense and sparse seagrass.

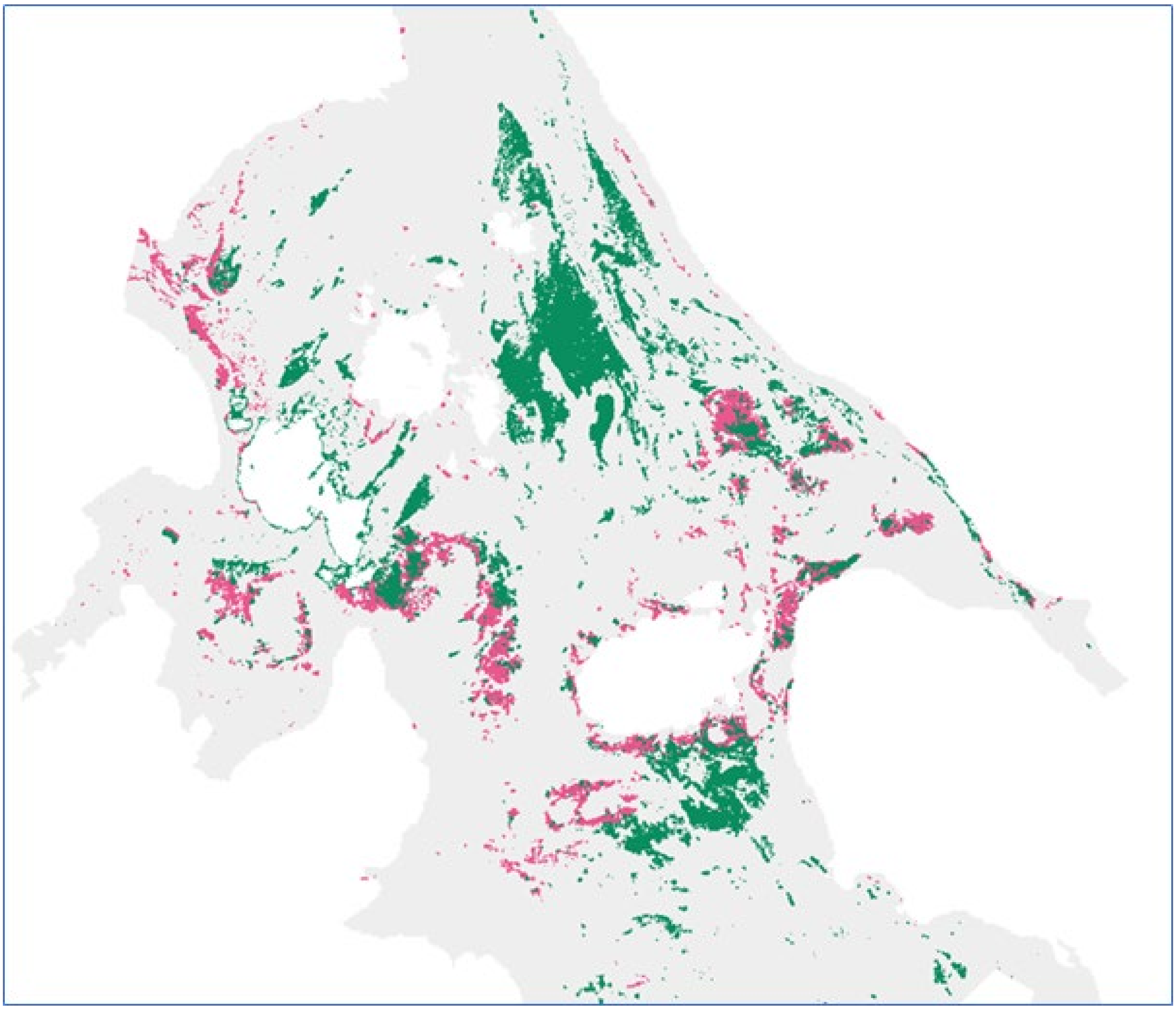


Figure 17 Excerpt from seagrass distribution map for central Tauranga Moana derived from very high-resolution satellite data (PlanetScope) and further enhanced using superpixel techniques (Ha et al., 2023). On this image, green represents dense seagrass, red sparse seagrass.

The use of satellites to map seagrass continues to be enhanced, with research by Shao et al. (2023b) additionally mapping the percentage cover of seagrass and estimates of gross primary productivity for both seagrass and microphytobenthos (Figure 18) (using supervised classification with random forest, and artificial neural network regression). The accuracy of this method was 0.96, creating highly accurate assessments of percentage cover across the harbour.

The speed of development of satellite technology now enables its usage in Regional Council monitoring programmes. In particular, the high frequency (although slightly lower resolution) data can enable more frequent monitoring including assessments of seagrass seasonality and event impact monitoring, and estuary wide ecosystem service estimates, which were historically not accessible due to costs of assessing and processing aerial imagery. Many of these new methods are being produced with open-source codes and methods to allow uptake by users with basic coding abilities.

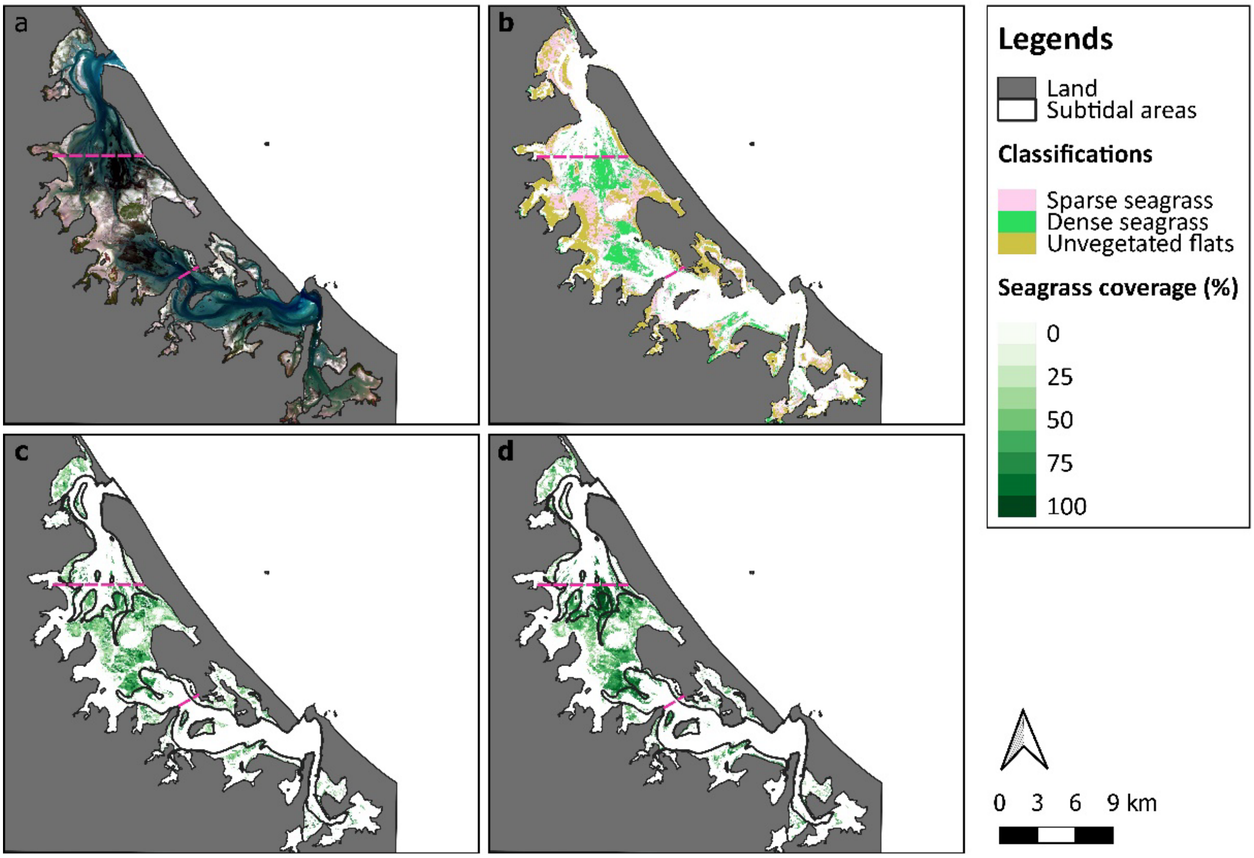


Figure 18 The maps of (a) original satellite data from 24 February 2022, at low tide; (b) spatial distribution of dense seagrass, sparse seagrass and unvegetated flats after classification using random forest; and seasonal variations in seagrass coverage in winter (c) and summer (d) after regression using artificial neural network. The data were derived from images taken on 3 August 2021 (c) and 24 February 2022 (b & d). The magenta dashed lines arbitrarily divide the harbour into three parts: northern, middle, and southern. From Shao et al. 2023b.

4 Discussion/Matapakitanga

4.1 Recommendations for seagrass management

As demonstrated through this report, seagrass habitats are subjected to a multitude of different stressors (Figure 19), derived from land use practices and the resultant catchment contaminants to water (sediment and nutrient inputs) leading to issues with light, macroalgae and filamentous algae blooms, to external additional stressors, including physical disturbances, herbivory and fungal wasting disease. All of these stressors will be subject to the interactive effects of a changing climate. Seagrass is an “ecological engineer” species, insofar as the three-dimensional habitat that it creates supports high local biodiversity, including juvenile fish, and creates conditions that support high water clarity and water quality. To ensure the longevity of these habitats into the future, forward focused strategic direction is required to monitor, protect and where required, restore these important estuarine habitats.

The largest historical impact to seagrass in the Bay of Plenty is from sedimentation and nutrient enrichment to our estuaries, from historical and current land use practices. In many instances, we have removed the important wetland buffer systems around the estuaries and converted the land to pastoral farming, directing high nitrogen and phosphorus loads to the estuary, leading to eutrophication. This has resulted in high growth and coverage of nuisance algae species, which compete for space with seagrass and alter the surrounding sediment environment. Highly mobile sediments from the catchment continues to produce large sediment plumes which deposit into estuaries, reducing light available for seagrass and creating deep muddy sediments devoid of life. These two catchment pressures have resulted in significant seagrass habitat loss across the Bay of Plenty and reduced the resilience of seagrass to recover from additional external pressures (such as bird herbivory, or resilience to disease). The road to recovery for seagrass relies on a number of intertwining management options to recreate a favourable habitat, starting with reduction of nutrients and sediment. Implementation of the NPS-FM aims to significantly reduce these stressors but is reliant on strong policy, community agreeance and the ability to enable land use change in high-risk regions.

Restoration of coastal habitats such as saltmarsh, mangroves and freshwater wetlands will form an important buffer system between the land and the coast, supporting the uptake of nutrients and sediments. There are numerous projects underway across the Bay of Plenty undertaking saltmarsh restoration through rewetting of pasture, and strategic direction is required to ensure this can be done at the scale required for estuary protection (Crawshaw & Fox, 2022). Broadscale maps have been developed that identify suitable parcels of land, which are supporting small scale restoration projects, and further work could be undertaken to prioritise regions for restoration. Some realignment (or reducing the straightening) of inflows to the estuaries will further support increased areas for sediment deposition and capture before it reaches the estuary. Mangroves will restore to natural extents, if activities to limit them are stopped.

Waterfowl grazing has direct (consumption) and indirect (disturbance) impacts on seagrass beds, but is highly variable in space and time, resulting in higher more visible impacts in certain regions. Over whole harbours grazing calculations indicate that loss of intertidal seagrass may be exceeding capacity to sustain full potential extents of the beds. Given the uneven grazing pressure it becomes even more likely that grazing is having an impact at localised scales as seen at Otūmoetai. This was unable to be examined at smaller localised scales due to a lack of information of waterfowl aggregations and the recovery rates of seagrass from high removal rates. However, the more intense swan grazing pressure at Otūmoetai has been linked with a loss of rare biodiversity associations in that area.

In smaller estuaries such as Maketū, grazing pressure is preventing the recovery of seagrass in suitable habitat and active intervention may be required. This could take the form of creating protective barriers around the seagrass to aid biomass recovery, excluding grazing pressure, but it is not an option for estuary wide management. Active removal of swan or geese has been trialled in the past by a range of agencies, however, it has had mixed impacts given the wide areas the birds migrate over, with new birds filling the area left behind from other regions. Any control measures would need to consider the wider potential impacts on other habitats, be based on further science and this would need to be led by the relevant management agencies and with cross-regional collaboration.

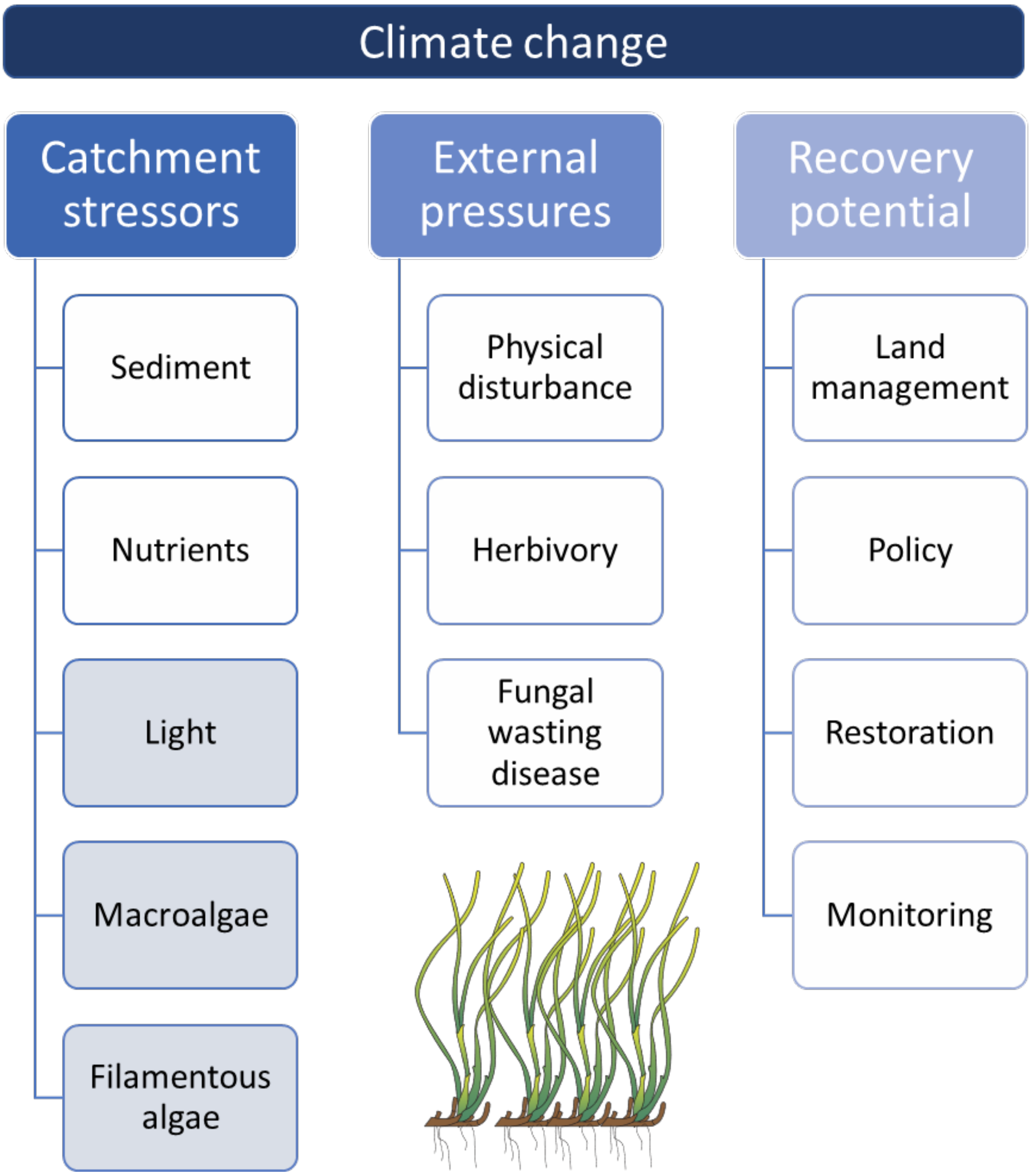


Figure 19 Schematic of catchment and external pressures on seagrass, and the core components required to support seagrass recovery.

There are some positive outcomes for Tauranga Harbour documented in this report, where extents of subtidal seagrass have expanded significantly between the 2022 mapping survey and its immediate predecessor. These subtidal meadows create important habitat for fisheries species, particularly their juvenile stages (Brooke, 2015), may have a higher blue carbon potential than intertidal meadows, and indicates there may have been recent improvements in the environment to support subtidal seagrass production. Further research into these subtidal beds will support our understanding of the additional habitat provisioning and/or ecosystem services they provide alongside the intertidal seagrass habitat.

Monitoring outcomes of policy and land management intervention is required and can be used to track outcomes towards the protection of seagrass habitats. In some locations, it is evident that turning off the taps will not be sufficient, due to the historic damage already caused, and active restoration of seagrass may be required. This needs to involve local community iwi and hapū, and one initiative is already underway in the Waihī Estuary lead by the iwi collective Te Wahapū o Waihī[1], which may provide insights for future programmes across the region. Restoration of seagrass in New Zealand is an active research topic, with a number of techniques being trialled, including seed-based restoration, seedling nurseries and planting of fragments, with the goal to create a blueprint for seagrass restoration[2]. Seagrass restoration cannot occur without improvements in environmental quality, so a holistic approach is required that includes change in land use practices in the up-stream catchments.

Even with the multitude of information available on stressors to seagrass, there remains gaps in our knowledge to understand the changing dynamics of seagrass at large spatial scales. The increase in satellite technology may provide a new method for higher frequency, automated mapping of seagrass. This could increase our temporal and spatial knowledge to the scale of weeks or months, rather than years, and allow further investigation into the drivers of seagrass change. This will support faster investigation of patterns and trends of seagrass extent change – such as the impacts (and recovery dynamics) following localised or large-scale events (large sediment deposition events, marine or terrestrial heatwaves, sea lettuce blooms, compliance investigations).

[1] https://www.waihi-estuary.iwi.nz/

[2] cawthron.org.nz/research/our-projects/seagrass-restoration/

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