Waikato Regional Council Technical Report 2025/35

Subtidal seagrass surveys at Slipper Island / Whakahau and Great Mercury Island / Ahuahu: 2025

ISSN 2230-4363 (Online)
waikatoregion.govt.nz

Prepared by	Dan Crossett, Dana Clark, Lisa Floerl, Courtney Moir (Cawthron Institute)
For	Waikato Regional Council Private Bag 3038 Waikato Mail Centre HAMILTON 3240
Publication date	December 2025
Document ID	33589575

	Name	Date
Peer Reviewer	Nicola Salmond Michael Townsend	November 2025 November 2025
Approving Manager	Mike Scarsbrook	November 2025

CAWTHRON

Subtidal seagrass surveys at Slipper Island / Whakahau and Great Mercury Island / Ahuahu: 2025

Cawthron Report 4160

World-class science
for a better future

REVIEWED BY: Anna Berthelsen

APPROVED FOR RELEASE BY:
Grant Hopkins

PROJECT NUMBER: 19035

ISSUE DATE: 27 June 2025

RECOMMENDED CITATION: Crossett D, Clark D, Floerl L, Moir C. 2025. Subtidal seagrass surveys at Slipper Island / Whakahau and Great Mercury Island / Ahuahu: 2025. Nelson: Cawthron Institute. Cawthron Report 4160. Prepared for Waikato Regional Council.

Subtidal seagrass surveys at Slipper Island / Whakahau and Great Mercury Island / Ahuahu: 2025

Dan Crossett, Dana Clark, Lisa Floerl, Courtney Moir

Prepared for Waikato Regional Council

CAWTHRON

Contents

Executive summary

Subtidal seagrass meadows are ecologically valuable coastal habitats that offer numerous ecosystem services, particularly as nursery areas for juvenile fish. In Aotearoa New Zealand, these meadows are now rare and largely restricted to offshore islands. In 2025, Waikato Regional Council (WRC) commissioned the Cawthron Institute (Cawthron) to survey subtidal seagrass meadows at South Bay (Slipper Island / Whakahau) and Huruhi Harbour (Great Mercury Island / Ahuahu). The aim was to evaluate how meadow extent and ecological conditions had changed since the previous survey in 2019. We also carried out qualitative surveys of seagrass meadows at Home Bay (Slipper Island) and Parapara Bay (Great Mercury Island).

Surveys took place over 3 days in March 2025, with favourable sea conditions and relatively good in-water visibility. The seagrass meadow at South Bay covered an area of 0.18 km^2, slightly smaller than the estimated cover of 0.19 km^2 in 2019, but still six times larger than that recorded in 2004 and more than double the area estimated in 1973. However, the 2004 and 1973 surveys likely underestimated the true extent of the meadow. At Huruhi Harbour, meadow extent increased to 0.10 km^2 in 2025, up from 0.09 km^2 estimated in 2019 and 0.07 km^2 in 2004 but significantly less than the 0.52 km^2 reported in 1975. However, the 1975 survey likely overestimated meadow extent in Huruhi Harbour. Most of the 2025 expansion occurred toward the southeastern edge of the meadow, where seagrass extended further out of the harbour. Seagrass meadows at Home Bay and Parapara Bay appeared to have declined in extent since 2019.

Seagrass meadows at South Bay and Huruhi Harbour supported diverse marine fauna, including numerous fish species. Large schools of juvenile fish were particularly abundant at South Bay compared to observations in 2019.

In 2025, above-ground biomass was higher at South Bay (112 $gDW{\cdot}m^{-2}$) compared to Huruhi Harbour (53 $gDW{\cdot}m^{-2}$), while seagrass cover and leaf length were similar between the two locations. Average seagrass cover ranged from 26–50%, with maximum values exceeding 75% at both sites. Average leaf length was 234 mm at South Bay and 254 mm at Huruhi Harbour. In contrast, the 2019 survey showed significantly higher seagrass cover, leaf length, and above-ground biomass at South Bay compared to Huruhi Harbour. When depth was accounted for, seagrass condition indicators were higher at South Bay than Huruhi Harbour in 2025.

In 2025, very little macrofauna was observed in either meadow, which was similar to 2019. Epiphyte / sediment cover at South Bay was low (about 10%, primarily epiphytic algae) but increased slightly in deeper areas compared with 2019. However, at Huruhi Harbour, there was a significant increase in epiphyte / sediment (dominated by fine sediment) cover from 1% in 2019 to nearly 40% coverage in 2025.

At South Bay, fungal wasting disease prevalence was similar between 2019 and 2025 at around 35%, and the severity remained at less than 1% coverage. Huruhi Harbour exhibited a significant increase in the disease, with prevalence more than doubling to 78%, and severity increasing from less than 1% to 10% coverage.

Overall, we conclude that the condition of the South Bay seagrass meadow has remained relatively stable since 2004, and extent is similar to 2019, aside from some retreat noted near Transect 5. In contrast, the Huruhi Harbour meadow has shown progressive improvement since 2004, with increases in meadow extent, seagrass cover, above-ground biomass and leaf length. However, these positive trends were accompanied by increased levels of epiphyte / sediment cover and a concerning increase in fungal wasting disease since 2019. Both meadows still show signs anthropogenic impact, similar to 2019, particularly from vessel propeller or anchor scarring and swing moorings.

Ongoing monitoring is essential to detect early signs of change and enable timely management interventions where needed. To achieve this, we recommend that surveys at South Bay and Huruhi Harbour are conducted every 3–5 years (i.e. by 2028–30), and ideally in March to capture potential vessel impacts and ensure comparability with 2025 data. High-quality aerial or satellite images, from within 12 months of field surveys, should be investigated for ground-truthing *in situ* assessments. We recommend continued monitoring of all key indicators of seagrass condition and stress. We also recommend scaling up visual biomass estimates to more accurately estimate above-ground biomass across the meadows. Continued monitoring of the meadows at Home and Parapara Bays using high-quality aerial or satellite imagery validated with field assessments is advised.

Protective measures should be considered for South Bay and Huruhi Harbour, including implementing restrictions or raising public awareness to limit damage from vessel anchoring, swing moorings, propeller scarring and dredging.

We suggest that an important next step for Aotearoa New Zealand is to find and monitor other subtidal seagrass meadows that still exist around our coastline to ensure these critical coastal habitats are protected and managed appropriately.

1. Introduction

Aotearoa New Zealand has only one species of seagrass, *Nanozostera muelleri* (previously classified as *Zostera muelleri*, *Z. capricorni* or *Z. novaezelandiae*; Sullivan and Short 2023). This species is classified as 'At Risk-Declining' under the Department of Conversation's threat classification system (de Lange et al. 2017). Seagrass meadows are recognised as ecosystems of high ecological value and regarded as one of the most valuable coastal habitats globally due to the ecological services they provide (Costanza et al. 1997). They are highly productive systems that support broader coastal ecosystems through the net export of organic material (Hailes 2006) and contribute to approximately 15% of the net global CO_2 uptake by marine biota (Duarte and Chiscano 1999). Seagrass meadows also act as a sink for terrestrially derived nutrients (Short 1987) and stimulate nutrient cycling (Pellikaan and Nienhuis 1988). Their rhizomes and roots stabilise the sediment, while their three-dimensional canopy promotes sediment deposition, contributing to improved water quality (Fonseca 1996; Heiss et al. 2000).

The structural complexity provided by seagrass meadows, in what is often an otherwise homogenous, soft-sediment environment, plays a significant role in shaping the diversity, abundance and spatial distribution of associated flora and fauna (Henriques 1980; Turner et al. 1999; van Houte-Howes et al. 2004). For example, a 2004 survey of subtidal seagrass meadows at Slipper Island / Whakahau (hereafter referred to as Slipper Island), near the Coromandel, recorded twice as many macroinvertebrate taxa and more than three times the number of individuals compared with adjacent bare sediments (Schwarz et al. 2006). Subtidal seagrass meadows also serve as nursery habitats and nocturnal resting grounds for fish (Morrison et al. 2014a; Stewart 2015; Morrison and Francis 2001). In the same 2004 survey, 25 fish species were recorded within subtidal seagrass meadows at Slipper Island and Great Mercury Island / Ahuahu (hereafter referred to as Great Mercury Island), including juveniles of several commercial species (Schwarz et al. 2006). These meadows supported high abundances of exquisite gobies (*Favonigobius exquisitus*), sand gobies (*Pomatoschistus minutus*), juvenile yellow-eye mullet (*Aldrichetta forsteri*), snapper (*Pagrus auratus*) and pipefish. Notably, juvenile snapper densities in the seagrass meadows at Great Mercury Island were the highest recorded in any habitat in Aotearoa New Zealand.

Unfortunately, seagrass meadows have declined in extent worldwide (Short and Wyllie-Echeverria 1996), and Aotearoa New Zealand is no exception (Inglis 2003; Berthelsen et al. 2024). Between the 1920s and 1970s, substantial losses of seagrass were recorded in estuaries and harbours around Whangārei, Auckland, Whangamatā, Tauranga and Christchurch (Inglis 2003). Subtidal meadows have been particularly affected; for example, between 1959 and 1996, Tauranga Harbour experienced a 90% loss of its subtidal seagrass (Park 1999), indicating that environmental conditions over that period had become less favourable for the permanently submerged seagrass (Inglis 2003). Today, subtidal seagrass meadows in Aotearoa New Zealand are primarily restricted to offshore islands (Turner and Schwarz 2006a; Morrison et al. 2023).

Seagrass loss is often attributed to declines in water clarity and quality resulting from human activities. In particular, increased sediment and nutrient loads can degrade the underwater light environment through increased turbidity and stimulating the growth of phytoplankton, macroalgae and epiphytes (Short and Wyllie-Echeverria 1996). Seagrass meadows are also vulnerable to the release of toxic compounds into coastal waters, such as those from oil spills and industrial discharges, and direct mechanical damage from activities such as dredging, coastal development and anchoring (Short and

Wyllie-Echeverria 1996). Additional stressors include severe storms, overgrazing and / or competition from natural or introduced species and fungal wasting disease (Matheson et al. 2009). Fungal wasting disease, caused by the marine slime mould *Labyrinthula zosterae,* is thought to be responsible for the catastrophic die-off of *Zostera marina* meadows along the Atlantic coasts of North America and Europe during the 1930s (Ralph and Short 2002). *Labyrinthula* was first detected in Aotearoa New Zealand in the 1960s and may have contributed to widespread seagrass losses observed in several harbours during this period (Armiger 1964). Since then, the pathogen has been found in seagrass populations throughout Aotearoa New Zealand (Armiger 1965; Woods and Schiel 1997; Ramage and Schiel 1999; Gillespie et al. 2012a, 2012b, 2012c; Berthelsen et al. 2016; Šunde et al. 2017; Clark and Crossett 2019). Blooms of *Labyrinthula* may occur when environmental conditions are favourable for their growth (low light, warm temperatures, high salinity; Ralph and Short 2002) and seagrass is particularly susceptible when it is stressed by anthropogenic impacts or other environmental pressures (Turner and Schwarz 2006a). In the Thames estuary in the United Kingdom, research has demonstrated that exposure to elevated nitrate concentrations and herbicides increases the susceptibility of *Z. marina* to *Labyrinthula* infection, supporting the hypothesis that disease outbreaks may be linked to increased agricultural run-off and chemical use (Hughes et al. 2018).

Given the vulnerability of seagrass meadows to environmental change, effective management of these habitats depends on the collection of accurate information on their distribution and condition (McKenzie et al. 2001; Turner and Schwarz 2006a). The Waikato Region has four known areas of subtidal seagrass: Huruhi Harbour and Parapara Bay (Great Mercury Island), and South and Stingray Bays (Slipper Island) The Huruhi Harbour and South Bay meadows were surveyed in the 1970s (Grace and Whitten 1974; Grace and Grace 1976) and again in 2004 (Schwarz et al. 2006) and 2019 (Clark and Crossett 2019). The Parapara Bay meadow was surveyed in the 1970s (Grace and Grace 1976) and in 2019 (Clark and Crossett 2019), while the Stingray Bay meadow was first surveyed in 2019 (Clark and Crossett 2019).

In 2019, Cawthron Institute (Cawthron) conducted a survey of the Great Mercury and Slipper Island seagrass meadows for Waikato Regional Council (WRC) to map the extent and assess the ecological condition of subtidal seagrass meadows in Huruhi Harbour and South Bay. Seagrass extent in nearby Parapara and Stingray Bays was also recorded. Following recommendations by Clark and Crossett (2019) to resurvey the meadows every 3–5 years, WRC commissioned Cawthron to repeat the survey in 2025 (the focus of this report), aiming to determine whether the extent or ecological condition of the meadows had changed since 2019.

This report presents a comprehensive resurvey of the seagrass meadows at South Bay and Huruhi Harbour, as well as qualitative surveys of seagrass meadows at Home and Parapara Bays. Surveys were designed to facilitate comparison with previous surveys, and methods are also consistent with guidance for councils on seagrass monitoring (Shanahan et al. 2023). At South Bay and Huruhi Harbour, we mapped meadow extent and assessed key indicators of seagrass condition (seagrass cover, leaf length, above-ground biomass; Duarte and Kirkman 2001; Shanahan et al. 2023), alongside indicators of stress (cover of macroalgae, epiphytes and fine sediment, and the severity and prevalence of fungal wasting disease; Shanahan et al. 2023). Results were compared with those from the 2019 survey, and where appropriate, with earlier surveys from 2004 and 1975–76, to understand how these subtidal meadows have changed over time. The report provides valuable information to support the protection and management of subtidal seagrass in the Waikato Region.

2. Methods

2.1 Study areas

Slipper and Great Mercury Islands are situated off the eastern coast of the Coromandel Peninsula, Aotearoa New Zealand (Figure 1). Slipper Island, located 8 km southeast of Pauanui, is largely occupied by a private resort. Great Mercury Island, part of the Mercury Islands group located 35 km north of Whitianga, is privately-owned and pest-free. In 2025, surveys were conducted earlier in the year (March) compared to previous surveys in 2004 and 2019, which took place in May–June. This timing shift was intentional, as WRC wanted to understand whether anthropogenic impacts – particularly vessel-related damage – could be detected immediately after the busy summer season for recreational boating. South Bay, Stingray Bay and Home Bay (Slipper Island) were surveyed on 13 and 15 March 2025, and Huruhi Harbour and Parapara Bay (Great Mercury Island) were surveyed on 14 March 2025. Tidal ranges during the survey period were classified as spring tides, ranging from 1.36 m and 1.58 m (NIWA 2025).

Great Mercury Island is under a controlled area notice with Biosecurity New Zealand, so extra care was taken to ensure protocols were followed to prevent the spread of invasive *Caulerpa* species. During our field surveys, divers monitored for invasive *Caulerpa* species and inspected our vessel, anchor, chain and gear before leaving the field site. If invasive *Caulerpa* was found, we planned to use the 'bag it, bin it' procedure. We were also ready to decontaminate all diving gear with 1% TriGene detergent and clean the vessel, anchor and anchor-chain with 200 ppm bleach based on Cawthron's biosecurity protocol.

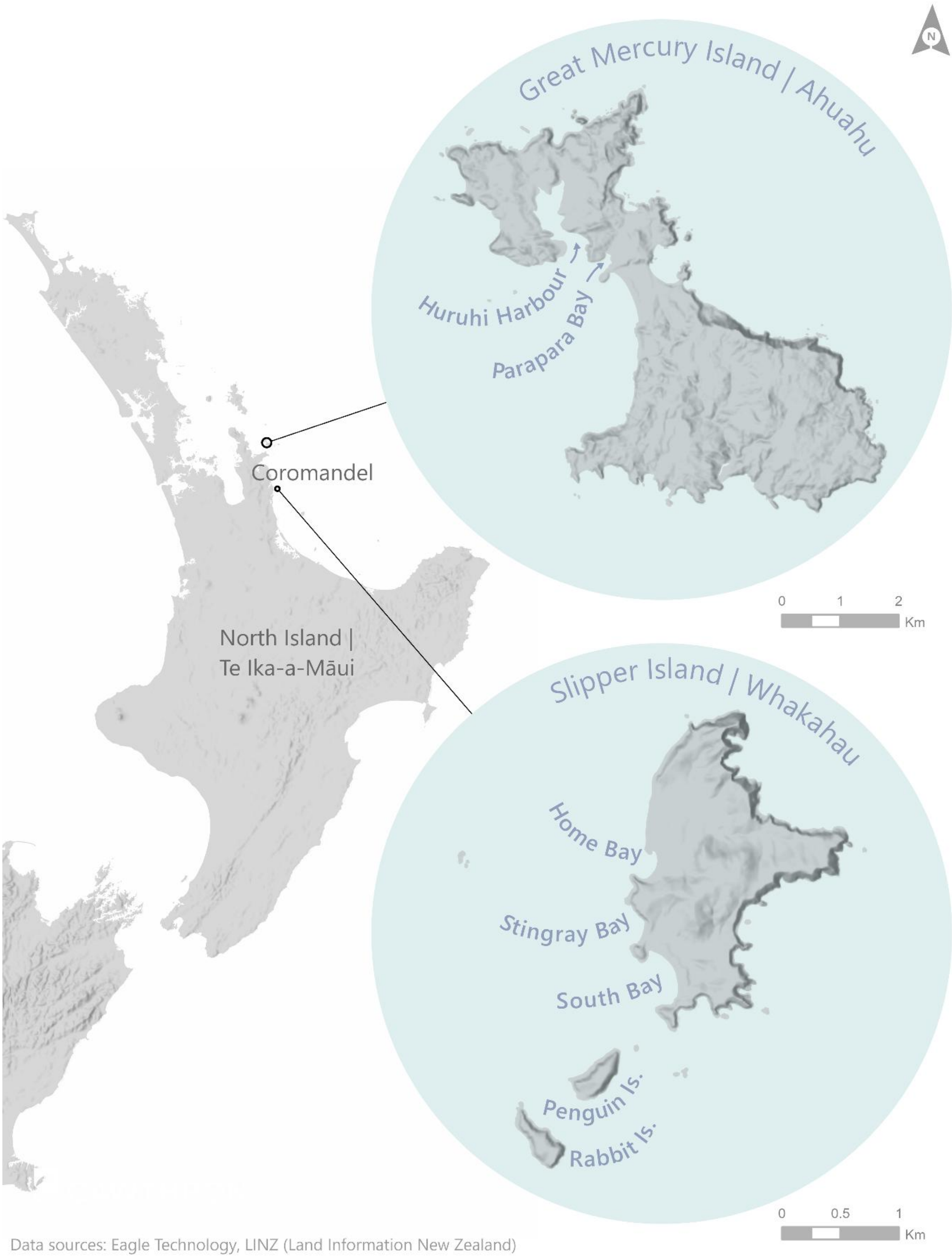


Figure 1. Map of Great Mercury Island / Ahuahu (top circle zoom-in) and Slipper Island / Whakahau (bottom circle zoom-in), which are offshore from the Coromandel Peninsula in the North Island of Aotearoa New Zealand. Seagrass meadows were surveyed in Huruhi Harbour and Parapara Bay, Great Mercury Island / Ahuahu, and Home, Stingray and South Bays, Slipper Island / Whakahau.

2.2 Mapping seagrass extent

To help delineate the 2025 seagrass extent, recent aerial photographs of the study sites were obtained from the Land Information New Zealand (LINZ) Data Service (2023–24[1]; Figure 2). The most recent imagery was imported into Field Maps, an ArcGIS Online web application, along with previous ground-truthing point data and seagrass extent polygons. These data were used as a reference during fieldwork to ensure ground-truthing data were collected around meadow boundaries. Data points and observations of habitat types and transitions were recorded using the Field Maps app and an additional GPS unit.

Seagrass extent mapped from aerial imagery was ground-truthed in the field by observers from the boat where water clarity permitted, or by snorkellers in deeper or more turbid areas. Following previous survey methodology (Schwarz et al. 2006; Clark and Crossett 2019), seagrass meadow boundaries in 2025 were determined as the point where seagrass cover exceeded 5%. As noted by Schwarz et al. (2006), this threshold likely underestimates the potential niche available for seagrass growth, since some plants will extend beyond this boundary, and the method does not account for bare patches within the meadow. To further investigate changes in meadow extent and assess potential anthropogenic impacts over time, aerial imagery from 2016–19 and 2021–24 was also reviewed (Appendix 5).

Seagrass meadows in South Bay and Huruhi Harbour were also mapped using side-scan sonar deployed from an autonomous operated vehicle (AOV; BlueRobotics BlueBoat[2]), and these data were compared with results obtained from ground-truthed aerial imagery. The BlueBoat is operated using BlueOS software with a sonar view accessory. Flight plans are created using QGroundControl software, which provides a GPS link to the autonomous vessel for synchronised mapping and sonar data overlay. Side-scan sonar data were processed using Sonar View and analysed in ReefMaster.[3] Survey parameters included a speed of 1 m/s, sonar frequency of 450 kHz, a side-scan width of 40 m, and transect spacing of 25 m (allowing a 10 m crossover between swarths on either side of the boat).

Seagrass extent was also qualitatively estimated through ground-truthing of recent aerial images in three additional bays – Stingray Bay and Home Bay (north of South Bay, Slipper Island) and Parapara Bay (east of Huruhi Harbour, Great Mercury Island). However, these areas were not mapped as extensively as South Bay and Huruhi Harbour, as this was beyond the scope of the study.

2.3 Determining seagrass condition

Survey locations and sampling design

In 2025, six temporary,100 m long transects were laid within the seagrass meadows in South Bay and Huruhi Harbour, positioned as close as possible to 2019 GPS start and end points (see Appendix 1 for coordinates). At South Bay, the seagrass meadow was stratified into three depth strata, with two

[1] Waikato 0.3m Rural Aerial Photos (2023-2024) | LINZ Data Service
[2] https://bluerobotics.com/product-category/boat/
[3] https://reefmaster.com.au/

transects surveyed in each zone (Figure 2). Following recommendations from the 2019 survey (Clark and Crossett 2019), a seventh transect was added at South Bay to better assess seagrass condition in the 2–4 m depth range (T7 in Figure 2). The seagrass meadow at Huruhi Harbour was narrower, so single transects were spread evenly across its depth gradient (Figure 2).

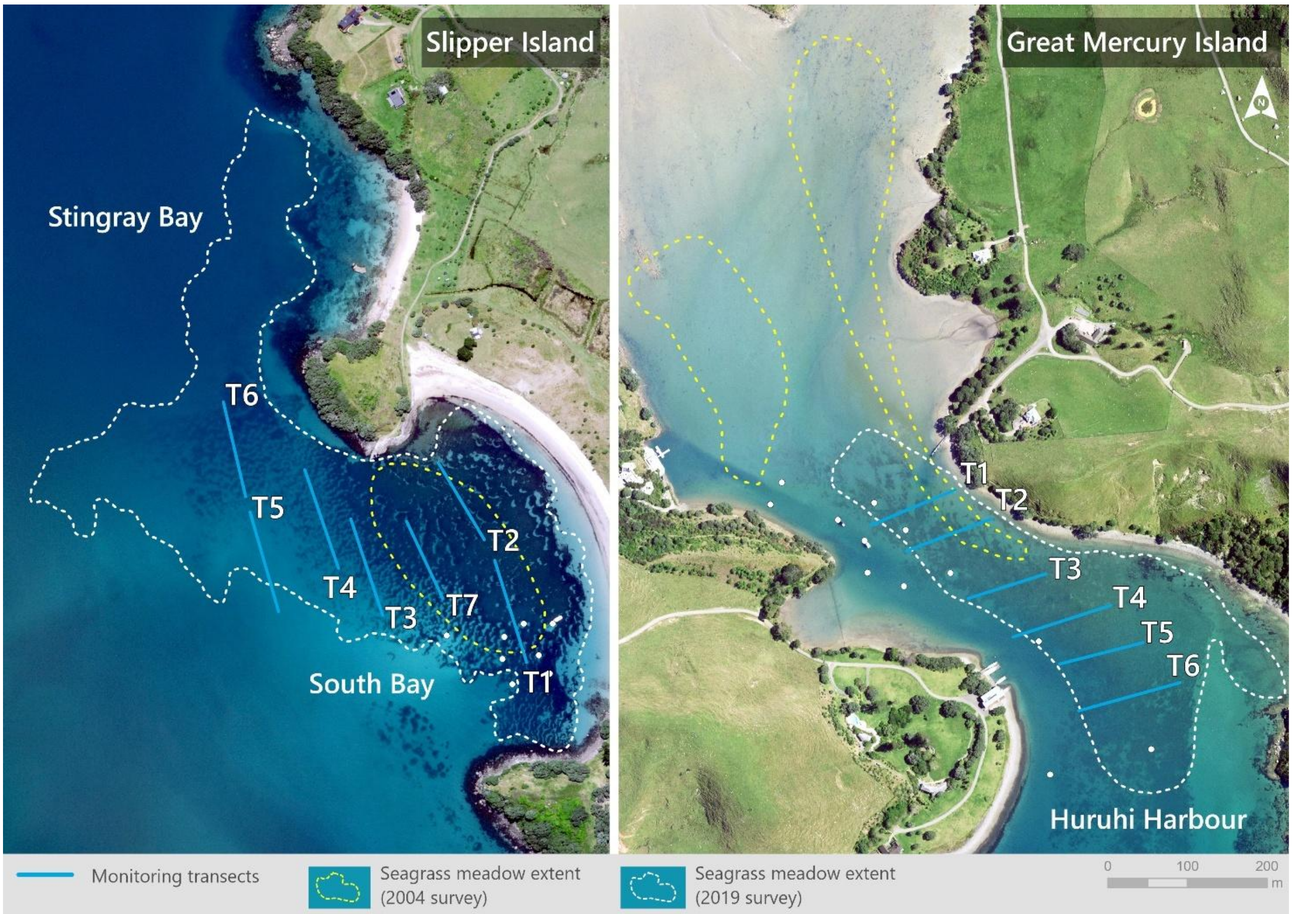


Figure 2. Transect locations surveyed at (A) South Bay, Slipper Island / Whakahau and (B) Huruhi Harbour, Great Mercury Island / Ahuahu. Seagrass extent estimated during the 2004 survey is indicated by the yellow dashed-line polygons, and the 2019 survey is indicated by the white dashed-line polygons. Numbering indicates the start of each transect. Note that a seventh transect (T7) was added and surveyed in 2025 to assess seagrass condition in the 2–4 m depth range. Data sourced from 'Waikato 0.3m Rural Aerial Photos (2023–2024)' (LINZ Data Service).

Seagrass condition and stress indicators

Key parameters that indicate the ecological condition of seagrass (seagrass cover, leaf length and above-ground biomass; Duarte and Kirkman 2001; Shanahan et al. 2023) were quantified at fixed points along each transect by scuba divers. Cover of macroalgae and epiphytes and visual signs of the presence and severity of fungal wasting disease were also recorded as indicators of stress.

Seagrass cover

Cover of seagrass was estimated within a 0.25 m^2 quadrat at 5 m intervals along each transect. Following Schwarz et al. (2006), cover was estimated *in situ* using the Braun-Blanquet cover scale (Braun-Blanquet 1932), an internationally recognised method that reduces observer bias. This technique classifies percent cover into five categories: 1 = 1–5%, 2 = 6–25%, 3 = 26–50%, 4 = 51–75%, 5 > 75% (see Appendix 2). Concurrent photo-quadrats were also collected to provide a permanent record, which can be more accurately quantified later if required.

Seagrass leaf length

Canopy height reflects the structural role of seagrass meadows, including their habitat and refuge functions. Leaf length, a measure of canopy height, was estimated within an approximate 0.02 m^2 quadrat at 10 m intervals along each transect. Within each quadrat, we measured the maximum height of 10 randomly selected seagrass blades. Maximum height was used to ensure comparability with the 2004 survey.

Seagrass biomass

The structural role of seagrass depends largely on the amount of vegetative material it develops above and below ground. Biomass is a useful monitoring metric because it responds quickly and detectably to environmental disturbances (Duarte and Kirkman 2001). However, collecting biomass samples is destructive, particularly for below-ground biomass, which requires removal of rhizomes and roots.

Due to the sensitive nature of subtidal seagrass meadows, we carried out small-scale sampling of above-ground biomass only. The above-ground portion of the biomass is a more responsive indicator of disturbance than below-ground biomass and collection is less destructive, with bare patches recolonised within a few months (Duarte and Kirkman 2001). During the 2019 and 2025 surveys, we tested whether a visual estimate could be used as a proxy for above-ground biomass as a non-destructive and rapid method for future sampling. Three different visual measures were tested: (1) visual assessment of above-ground biomass estimated in the field (described below); (2) seagrass cover estimated from photos using the Braun-Blanquet scale (described above); and (3) seagrass cover estimated from photos using a dots-on-rocks (DOR) approach (Meese and Tomich 1992). The DOR method was carried out using the ImageJ[4] application, where presence or absence of seagrass was recorded across a grid of points (48 in 2025 and 30 in 2019; Figure 3).

[4] https://imagej.net/ij/

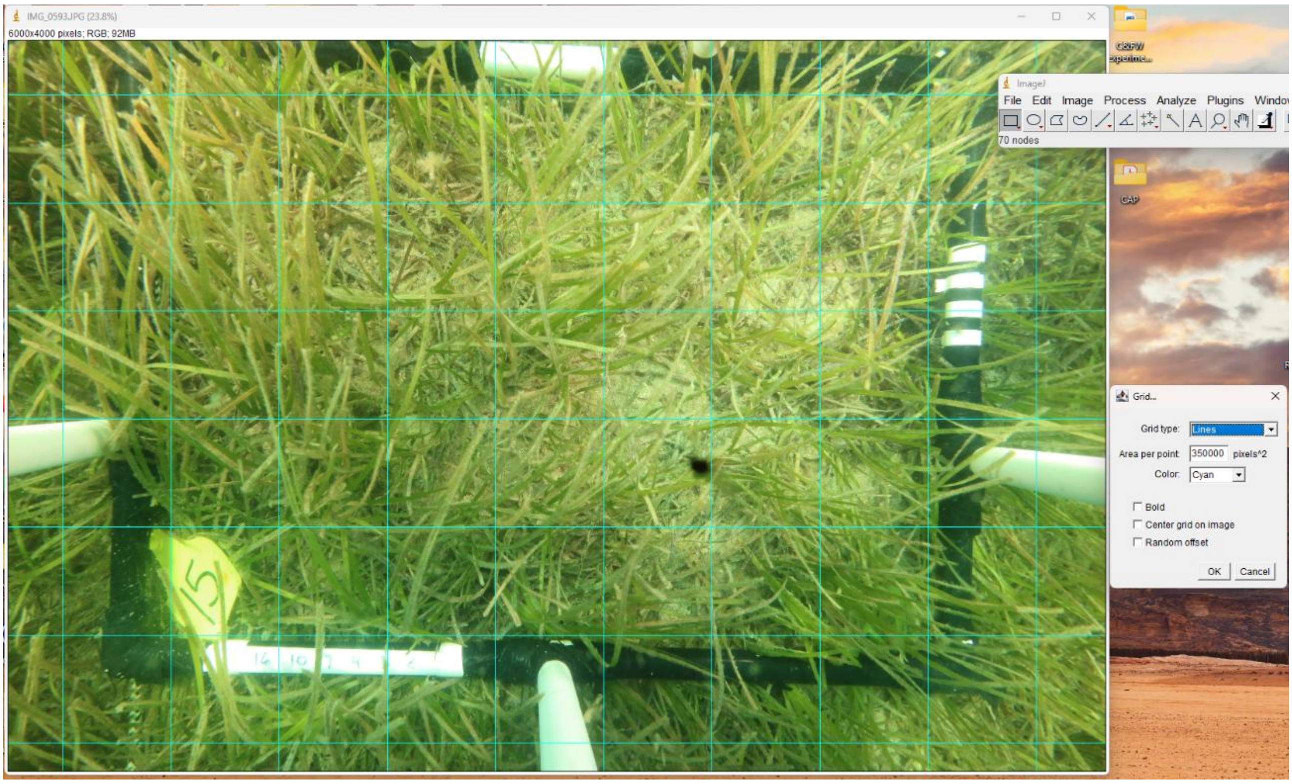


Figure 3. Image showing how seagrass percent cover was calculated using the Grid tool in the ImageJ application. A 6 × 8 grid was overlaid on a photo-quadrat (0.25 m^2) image and the number of intersections with seagrass was divided by 48 (total intersection points) and multiplied by 100.

In 2019, we developed a set of standard ranks for the visual assessment of biomass. Although this method is likely to be less accurate than quantitative harvesting techniques, it allows more samples to be taken, ensuring that a representative area is assessed across the seagrass meadow. Following the methods of Mellors (1991), five reference quadrats were selected to represent a scale against which the above-ground biomass in each sample was compared. To develop the reference scale, quadrats were placed in different areas of the seagrass meadow, ranging from an area which was visually determined to have the highest biomass (rank 5) to an area deemed to have the lowest biomass (rank 1). Ranks 2 to 4 were placed in areas midway along this visual biomass gradient. Rank 0 was used for quadrats containing no seagrass. Reference and sample quadrats were photographed so they can be referred to for future sampling (Appendix 3, photos sourced from the 2019 surveys).

During the 2025 transect surveys, photos were taken of the above-ground biomass in two small (0.0225 m^2) quadrats per transect. Above-ground biomass in each quadrat was then estimated using the visual rank system developed in 2019 (Appendix 3) and harvested for quantitative analyses. In the laboratory, seagrass material was separated from the sediment and thoroughly rinsed through a 1 mm sieve to ensure the removal of attached sediment and invertebrates. Following Schwarz et al. (2006), seagrass material was oven-dried at 80 °C for 48 hours, then the dried samples were transferred to a desiccator, and once cool, were weighed on a balance.

Harvested dry weight biomass values were then compared to visual biomass estimates and seagrass cover estimated from the small quadrat photos using the Braun-Blanquet and the DOR methods.

Macroalgae cover
Macroalgae can shade seagrass and limit its growth, making monitoring of macroalgal cover essential as an early warning indicator of ecological stress (Kirkman 1996). Cover of macroalgae was estimated at 5 m intervals along each transect using a 0.25 m^2 quadrat and the Braun-Blanquet cover scale (Braun-Blanquet 1932), as described earlier.

Epiphyte / sediment cover
Like macroalgae, epiphytes can shade seagrass and reduce light availability, meaning their abundance is a useful indicator of eutrophication within seagrass meadows. To assess seagrass cover, a semi-quantitative scale (Appendix 4) was used to estimate the cover of epiphytes on 10 randomly selected seagrass blades within an approximate 0.02 m^2 quadrat, placed at 10 m intervals along each transect. Fine sediments, often trapped by epiphytes and contributing to light attenuation, were assessed concurrently. Because it was difficult to distinguish between sediment and epiphytes in the field, they were recorded together as a combined metric at both sites.

Fungal wasting disease
Fungal wasting disease is characterised by patches of darkened seagrass leaves (Burdick et al. 1993), with histological examination of leaves confirming the link with *Labyrinthula* cells (Berthelsen et al. 2016). Examination of histological slides in 2019 confirmed the presence of *Labyrinthula zosterae* cells at both South Bay and Huruhi Harbour (Clark and Crossett 2019). To assess disease prevalence and severity, observations were made within an approximate 0.02 m^2 quadrat placed at 10 m intervals along each transect. In each quadrat, 10 seagrass blades were randomly selected and visually assessed for signs of infection. Disease severity was ranked using the Wasting Index Key (Appendix 4; Burdick et al. 1993), which scores the extent of darkened leaf patches.

2.4 Additional information

Photo-quadrats (0.25 m^2) were taken at 5 m intervals along each transect, concurrent with our *in situ* field estimates of seagrass condition. These photographs provide a permanent visual record of seagrass cover and above-ground biomass, as well as macroalgae and epiphytes cover and the presence of fungal wasting disease. In addition to photo-quadrats, video footage and depth information (reported as depths relative to mean sea level [MSL]) were also collected. Impact from anchor or propeller scarring and swing moorings was estimated by drawing polygons around visibly damaged areas (for example, bare areas in seagrass meadows surrounding swing moorings) in ArcGIS Online web application based on the most recent high-quality aerial imagery. At South Bay, divers also estimated the area of disturbance from two swing moorings by measuring the distance from the mooring block to the edge of the seagrass in four cardinal directions. Observations of incidental fauna encountered within the seagrass meadows were also recorded to provide additional ecological context. Particular attention was given to detecting the presence of invasive *Caulerpa* species during field surveys.

2.5 Statistical analyses

To assess potential change in key seagrass condition indicators against year (2019 and 2025) and depth, we fitted a series of generalised linear models (GLMs) or ordinal regressions (for ranked response variables) for each site. The main scope of this study was to assess change in seagrass condition between 2019 and 2025; however, given that depth is a key factor influencing seagrass meadow characteristics (Duarte and Kirkman 2001), we included depth as an independent and interactive term with year in all models. Replicate measurements for leaf length were averaged at the quadrat level (10 blades per quadrat). Replicate measurements of epiphyte / sediment cover and fungal wasting disease severity were not averaged, and 'transect' was added as a random effect to the model to account for these replicates coming from the same transect. In these models, each of the seagrass indicators (seagrass cover, leaf length, above-ground biomass, epiphyte / sediment cover, fungal wasting disease severity, fungal wasting disease prevalence) were analysed as a dependent variable. GLMs were fitted using appropriate error families based on the distribution of the response variable (where residuals indicated that the assumption of normality was violated, alternative distributions for the error were fitted). Year was included as a fixed effect in all models, along with the interaction between 'Depth' and 'Year' to assess whether depth-related trends varied between 2019 and 2025. 'Site' was not included as a fixed effect in the models.

To assess the effectiveness of the visual biomass assessment techniques, we fitted generalised linear mixed models (GLMM) with a gamma error distribution to compare quantitatively harvested above-ground dry weights with three visual measures: (1) visual biomass rank, (2) seagrass cover estimated using the Braun-Blanquet scale, and (3) seagrass cover assessed using a DOR method. We included Year and its interaction with the visual scores to assess whether the relationship between visual and quantitative measures of biomass changed over time. For the visual biomass rank, 'Year' also served as a proxy for observer, since different observers conducted assessments in different years. We also added 'Year' as a random effect in the model, to account for the non-independence of the data from one year to the next. We also estimated and compare the trends for the two different years, in each model. We did this by constructing a reference grid of the predicted trends and averaging them over the predictors in the grid. This allowed for a direct comparison of how each visual assessment method performed across time.

All analyses were carried out using R (version 3.5.3; R Core Team 2019). The 'emmeans' package was used to estimate and compare the trends of the GLMM models between years (Lenth 2024).

3. Results

3.1 Seagrass extent

Slipper Island

In 2025, the seagrass meadow at South Bay (Slipper Island) covered an area of approximately 0.18 km^2, slightly smaller than the estimated cover of 0.19 km^2 in 2019 (Figure 4). However, the 2025 extent remained six times larger than the 0.03 km^2 area estimated in 2004 and more than double the 0.07 km^2 area estimated in 1973 (Appendix A5.1). As in 2019, seagrass remained relatively dense across most of the meadow in 2025, becoming increasingly patchy toward the edges and with depth. Maximum depth of the South Bay seagrass meadow in 2025 was 8.4 m (MSL), compared to 7.9 m (MSL) in 2019, and 5–6 m (MSL) in earlier surveys.

In 2025, seagrass was again observed to extend north into Stingray Bay, covering a total of 0.032 km^2, similar to the extent in 2019 but more extensive than seagrass recorded in Stingray Bay in 1973. Like the 2019 survey, the deeper extent of seagrass in Stingray Bay was difficult to determine due to poor water clarity and difficulty distinguishing the seaward boundary from aerial imagery. As in previous surveys (1973 and 2019), the headlands of South Bay and the inner region of Stingray Bay were primarily comprised of boulder-dominated habitats covered with *Carpophyllum* seaweed. These areas appear as darker sections adjacent to land in the aerial imagery (Figure 4). However, in 2025 we identified a series of previously undocumented patchy seagrass beds along the southern end of South Bay, further expanding the known spatial complexity of the meadow in this area (Figure 4).

Comparison of aerial imagery at three time points between 2016 and 2024 revealed fluctuations in seagrass extent at South and Stingray Bays (Appendix A5.2), as well as at Home Bay (Appendix A5.3). However, seagrass extent was not quantitatively assessed from these images. These observed changes highlight the importance of ground-truthing recent aerial imagery whenever possible to ensure accurate assessments of seagrass meadow extent. Aerial imagery also revealed anthropogenic impacts on these seagrass meadows. For example, aerial images from 2021–24 showed numerous boats anchored over the seagrass meadow in South and Home Bays (Appendices A5.2 and A5.3). In 2025, divers qualitatively assessed seagrass around the pier at Home Bay (Appendix A5.3) finding only patches of seagrass on sand among cobble and boulder habitats. Darker areas in the aerial images corresponded to rocky reef covered with seaweed, including *Ecklonia radiata* and *Carpophyllum* spp. (Appendix A5.3).

Seagrass meadows were also mapped using side-scan sonar. Although we did not conduct quantitative comparisons between ground-truthed estimates of seagrass extent based on aerial imagery and those derived from side-scan sonar, the overall extent seemed broadly similar between the methods (Figure 5). Side-scan sonar appeared to more effectively capture narrow ‘arms’ or small extensions of the seagrass meadows, particularly at the meadow fringes and in deeper, offshore areas compared to ground-truthed aerial imagery (Figure 5).

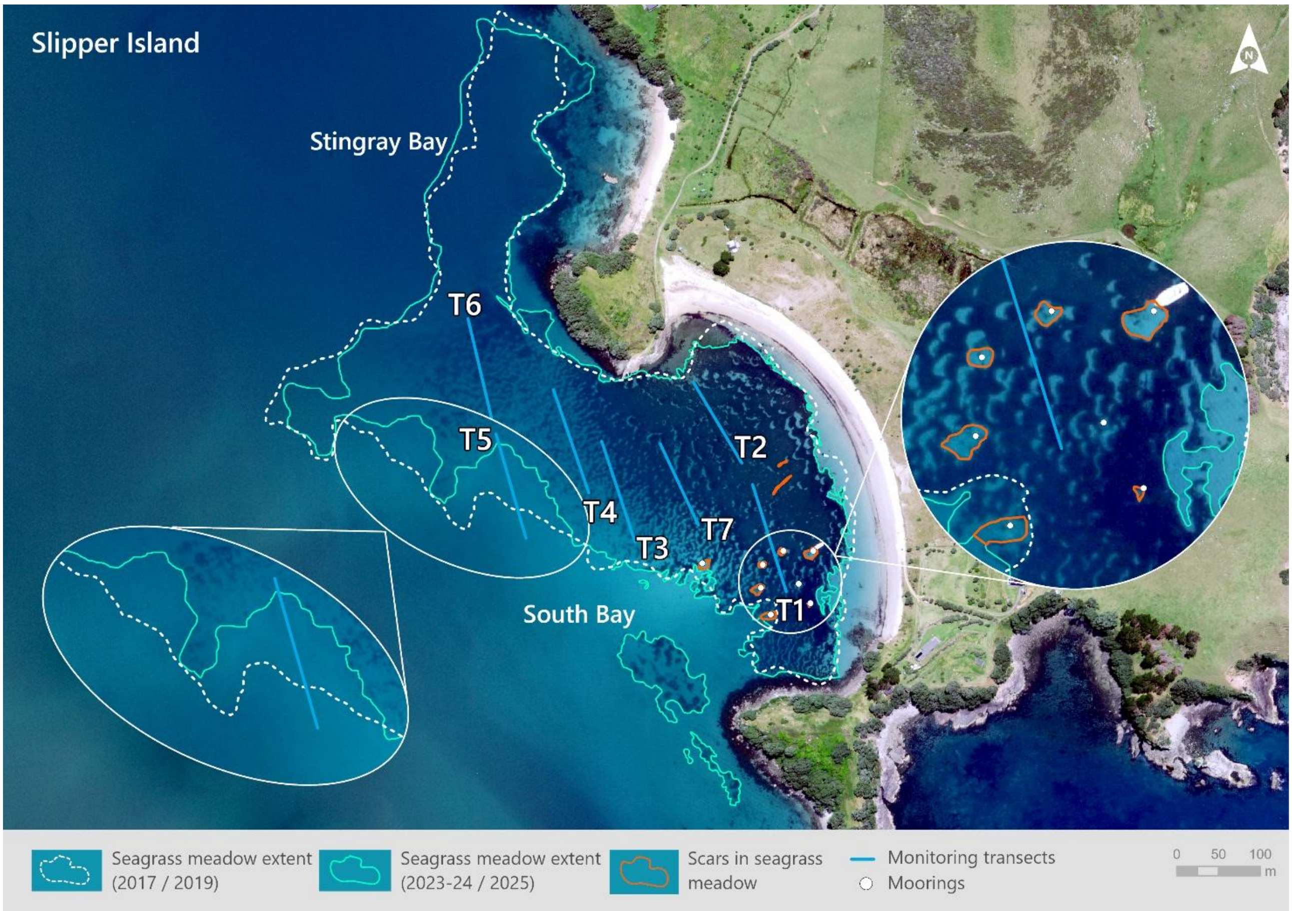


Figure 4. Extent of the seagrass meadow at South Bay (south) and Stingray Bay (north), Slipper Island / Whakahau. Estimates from the 2025 survey are represented by solid green polygons (ground-truthed from 2023–24 aerial images). Estimates for 2019 are represented by white dashed-line polygons (ground-truthed from 2017 aerial images) and solid blue lines represent survey transects. White dots represent mooring locations, and solid brown polygons represent scars or large holes in the seagrass meadow. Note that scars or holes were only described where obvious from aerial images. Zoomed map inserts show where seagrass extent has declined between 2019 and 2025 (bottom left insert) and where aerial images show obvious negative impact on seagrass meadows (right insert). Data sourced from 'Waikato 0.3m Rural Aerial Photos (2023–2024)' (LINZ Data Service). Data available at: https://data.waikatoregion.govt.nz:8443/ords/f?p=140:12:0::NO::P12_METADATA_ID:11077

Figure 5. Imagery of side-scan sonar of South Bay (south), and southern section of Stingray Bay (north), Slipper Island / Whakahau, provided by BlueBoat, with estimates of seagrass meadow extent from the 2025 survey (solid green polygons). Note that the texture change of side-scan imagery matches the boundary estimated by ground-truthing extent of seagrass from aerial imagery.

Great Mercury Island

In 2025, seagrass extent in Huruhi Harbour was approximately 0.10 km^2, a slight increase from the 0.09 km^2 recorded in 2019 (Clark and Crossett 2019). Compared to 2019, improved sea conditions, in-water visibility and aerial imagery in 2025 allowed for more accurate estimation of deeper areas; however, aerial imagery through time suggests there has been a true expansion of seagrass between 2019 and 2025 (Figure 6, Appendix A5.5). As in 2019, the 2025 survey showed a similar northern distribution of seagrass into the upper reaches of the harbour, with small patches extending towards both shores, but no seagrass in intertidal areas. Despite the 2025 expansion, the seagrass meadow remains much smaller than estimated in 1975, where it was reported to cover the entire harbour, including intertidal areas, with an estimated area of 0.52 km^2 and a maximum depth of approximately 5 m (Appendix A5.4; Grace and Grace 1976). Coverage was substantially reduced to 0.07 km^2 by 2004, although it still occupied intertidal areas (Schwarz et al. 2006).

Aerial imagery revealed changes in seagrass meadow extent at both Huruhi Harbour (Appendix A5.5) and Parapara Bay (Appendix A5.6) through time. The imagery also aligned with our *in situ* observations in 2019 and 2025. For example, during the qualitative survey of Parapara Bay in 2025, only small patches of seagrass were observed, consistent with the limited extent visible in aerial images from 2021–24 (Appendix A5.6). In contrast, both diver observations and aerial imagery from 2019 indicate more extensive seagrass cover at that time (Appendix A5.6).

The northern (inner reach) and southern (harbour entrance) seagrass meadow at Huruhi Harbour was mapped using side-scan sonar. However, side-scan sonar did not clearly reveal seagrass in Huruhi Harbour when compared to aerial imagery and *in situ* dive assessments (Figure 7).

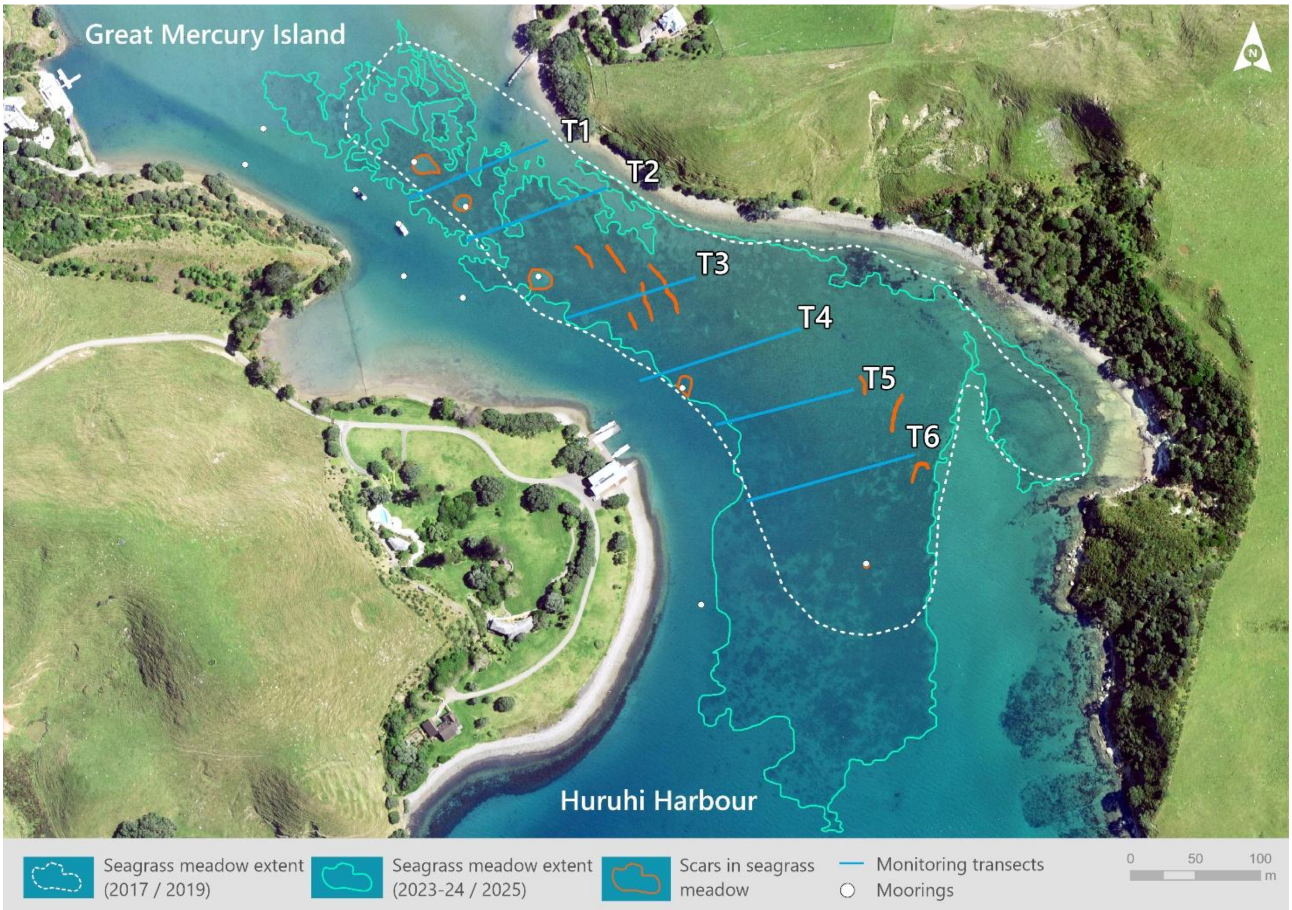


Figure 6. Extent of the seagrass meadow at Huruhi Harbour, Great Mercury Island / Ahuahu. Estimates from the 2025 survey are represented by solid green polygons (ground-truthed from 2023–24 aerial images) and estimates for 2019 are represented by white dashed-line polygons (ground-truthed from 2017 aerial images). Solid blue lines represent survey transects. White dots represent mooring locations, and solid brown polygons represent scars or large holes in the seagrass meadow. Note that scars or holes were only described where obvious from aerial images. Data sourced from 'Waikato 0.3m Rural Aerial Photos (2023–2024)' (LINZ Data Service). Data available at: https://data.waikatoregion.govt.nz:8443/ords/f?p=140:12:0::NO::P12_METADATA_ID:11077

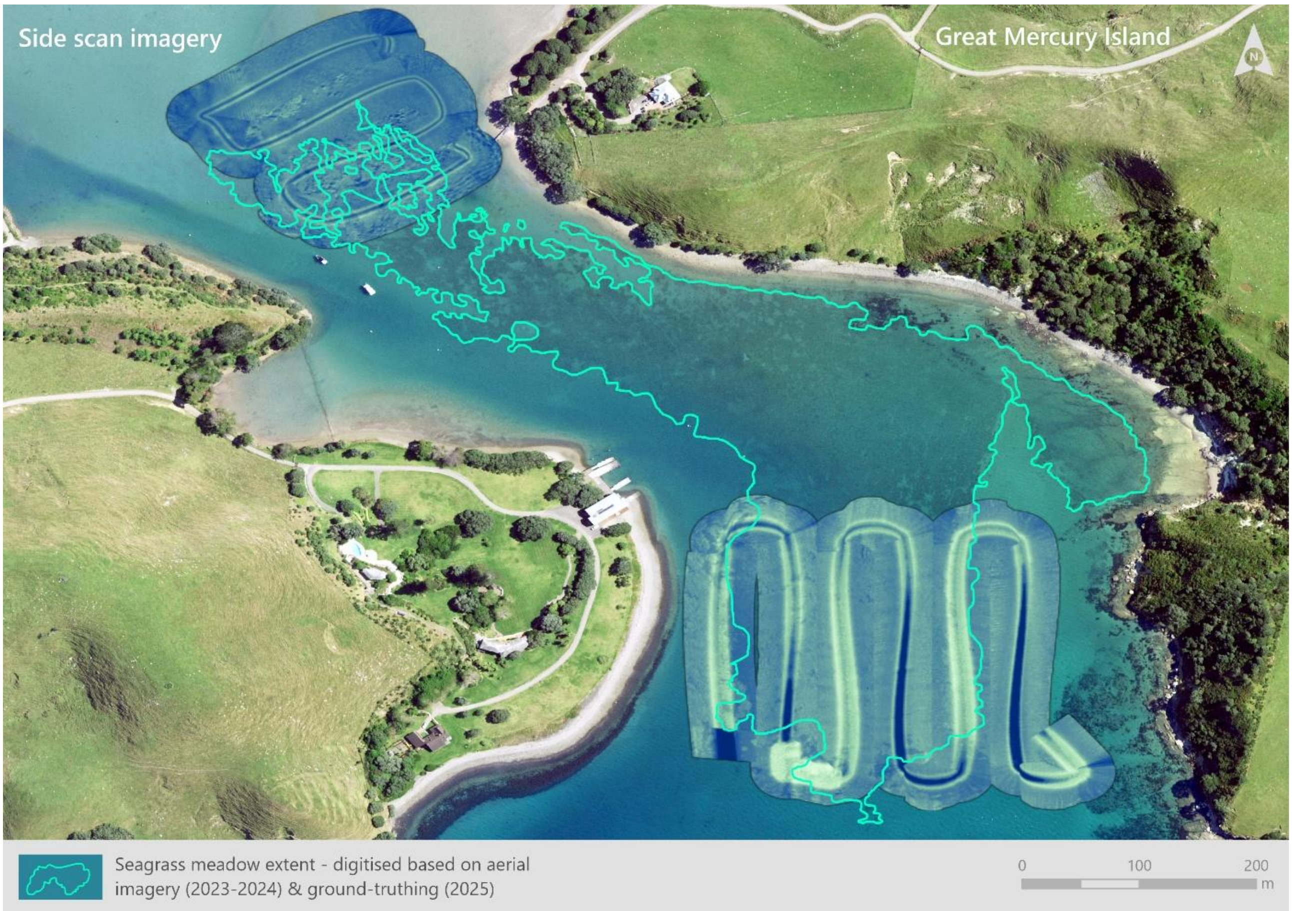


Figure 7. Imagery from side-scan sonar of northern and southern extent of seagrass meadow in Huruhi Harbour, Great Mercury Island/ Ahuahu, provided by BlueBoat. Estimates of seagrass meadow extent from the 2025 survey are displayed with solid green polygons.

3.2 Seagrass condition

Indicators of seagrass condition

In 2025, average seagrass cover and leaf length were similar between the South Bay and Huruhi Harbour, with average seagrass cover ranging from 26–50%[5] and maximum values exceeding 75% at both sites (Figures 8 and 9). Average leaf length was 234 mm (± 2.8 SE) at South Bay and 254 mm (± 2.4 SE) at Huruhi Harbour (Figure 10). In contrast, above-ground biomass was more than twice as high at South Bay (112 gDW·m^{-2} ± 22 SE) compared to Huruhi Harbour in 2025 (52.9 gDW·m^{-2} ± 5.1 SE; (Figure 9). In 2019, seagrass cover, above-ground biomass and average leaf length were all significantly higher at South Bay compared to Huruhi Harbour (Clark and Crossett 2019).

[5] Seagrass Braun-Blanquet rank 3.7 (± 0.2 SE) at South Bay and 3.9 (± 0.1 SE at Huruhi Harbour) in 2025.

Figure 8. Seagrass meadows at South Bay (Transects 1 – top left and 6 – top right), Slipper Island / Whakahau, and Huruhi Harbour (Transects 2 – bottom left and 6 – bottom right), Great Mercury Island / Ahuahu.

However, given the consistent, significant negative relationship between seagrass cover, biomass and leaf length and depth at South Bay in both years (Tables 1–3; Figures 9 and 10), it is more informative to compare values between meadows at equivalent depths, rather than using meadow-wide averages. For example, at 2–3 m depth in 2025, all seagrass condition indicators were higher at South Bay than Huruhi Harbour. Specifically, seagrass cover was 51–75%[6] at South Bay versus 26–50% at Huruhi Harbour, biomass was 226 $gDW \cdot m^{-2}$ versus 50 $gDW \cdot m^{-2}$, and average leaf length was 229 mm versus 216 mm.

At South Bay, seagrass cover, biomass and leaf length were not significantly different between year (2019 and 2025). In contrast, at Huruhi Harbour, there was no significant relationship between these seagrass condition indicators and depth. However, seagrass cover and leaf length were significantly greater in 2025 compared to 2019 (Tables 1 and 3; Figures 9 and 10). Average seagrass cover increased from 6–25%[7] to 26–50%[8] over this period and average leaf length increased from 139 mm (± 6.9 SE) in

[6] Seagrass Braun-Blanquet rank 4.9 at South Bay and 3.9 at Huruhi Harbour in 2025.
[7] Seagrass Braun-Blanquet rank average 2.3 (± 0.1 SE) at Huruhi Harbour in 2019.
[8] Seagrass Braun-Blanquet rank average 3.9 (± 0.1 SE) at Huruhi Harbour in 2025.

2019 to 254 mm (± 2.4 SE) in 2025. Above-ground biomass was not significantly different between the two years, averaging 50–53 $gDW \cdot m^{-2}$ (Table 2, Figure 9).

Comparisons of seagrass above-ground biomass and leaf length over two decades (2004: Schwarz et al. 2006; 2019: Clark and Crossett 2025; and 2025: this study) reveal interesting trends. At South Bay, average above-ground biomass declined from 155 $gDW \cdot m^{-2}$ (± 27 SE) in 2004, to 118 $gDW \cdot m^{-2}$ (± 30 SE) in 2019, and a similar value of 112 $gDW \cdot m^{-2}$ (22 ± SE) in 2025. Average leaf length at South Bay decreased from 288 mm (±17 SE) in 2004 to 215 mm (± 9.1 SE) in 2019, with a moderate increase to 234 mm (± 2.8 SE) in 2025. However, when comparisons are restricted to shallower areas, average above-ground biomass was higher in 2019 and 2025 compared to 2004, and average leaf length was comparable (Clark and Crossett 2019). In contrast, average above-ground biomass at Huruhi Harbour increased over time from 36 $gDW \cdot m^{-2}$ (± 4.3 SE) in 2004 to 50 $gDW \cdot m^{-2}$ (± 10 SE) in 2019 and a similar value of to 52.9 $gDW \cdot m^{-2}$ (5.1 ± SE) in 2025. Average leaf length at this site increased from 78 mm (± 3.0 SE) in 2004 to 139 mm (± 6.9 SE) in 2019 and reached 234 mm (± 2.8 SE) in 2025.

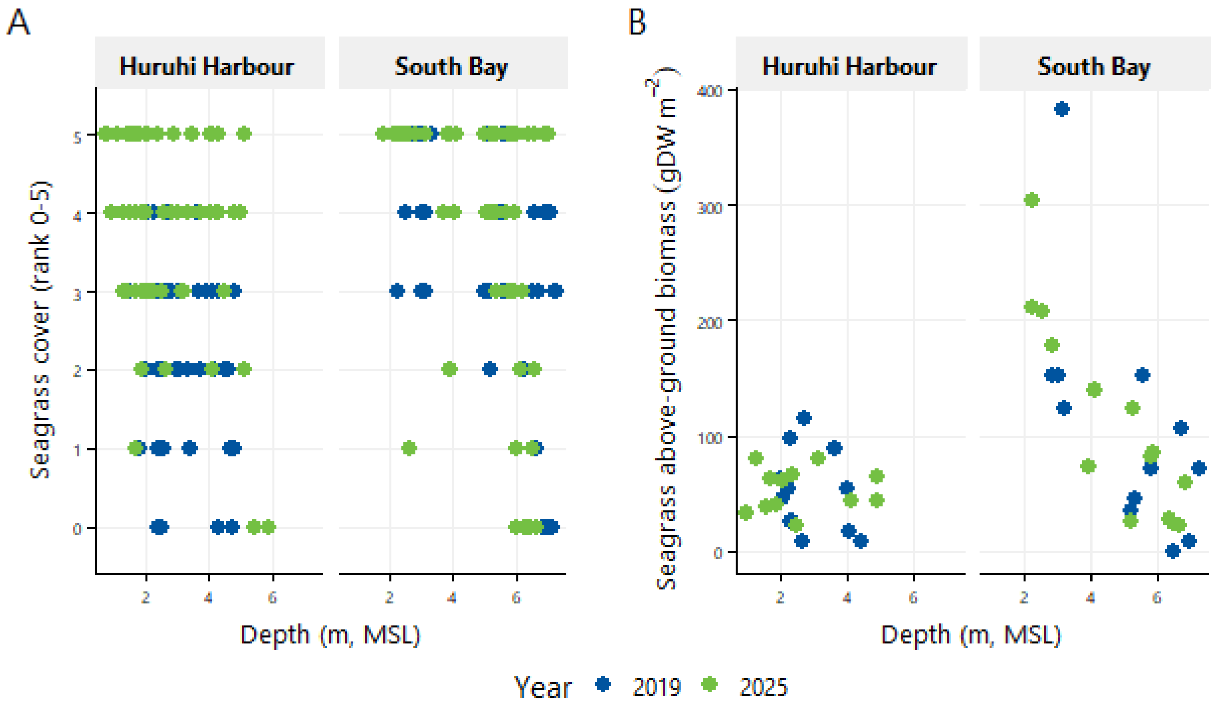


Figure 9. Seagrass condition and stress indicators regressed against depth at South Bay, Slipper Island / Whakahau, and Huruhi Harbour, Great Mercury Island / Ahuahu, at two time points: 2019 (blue dots) and 2025 (green dots). Indicators include (A) seagrass cover rank (Braun-Blanquet scale with rank to percentage cover: rank 0 [0%]; rank 1 [1–5%]; rank 2 [6–25%]; rank 3 [26–50%]; rank 4 [51–71%]; rank 5 [> 75%]), and (B) above-ground biomass (gDW·m^{-2}).

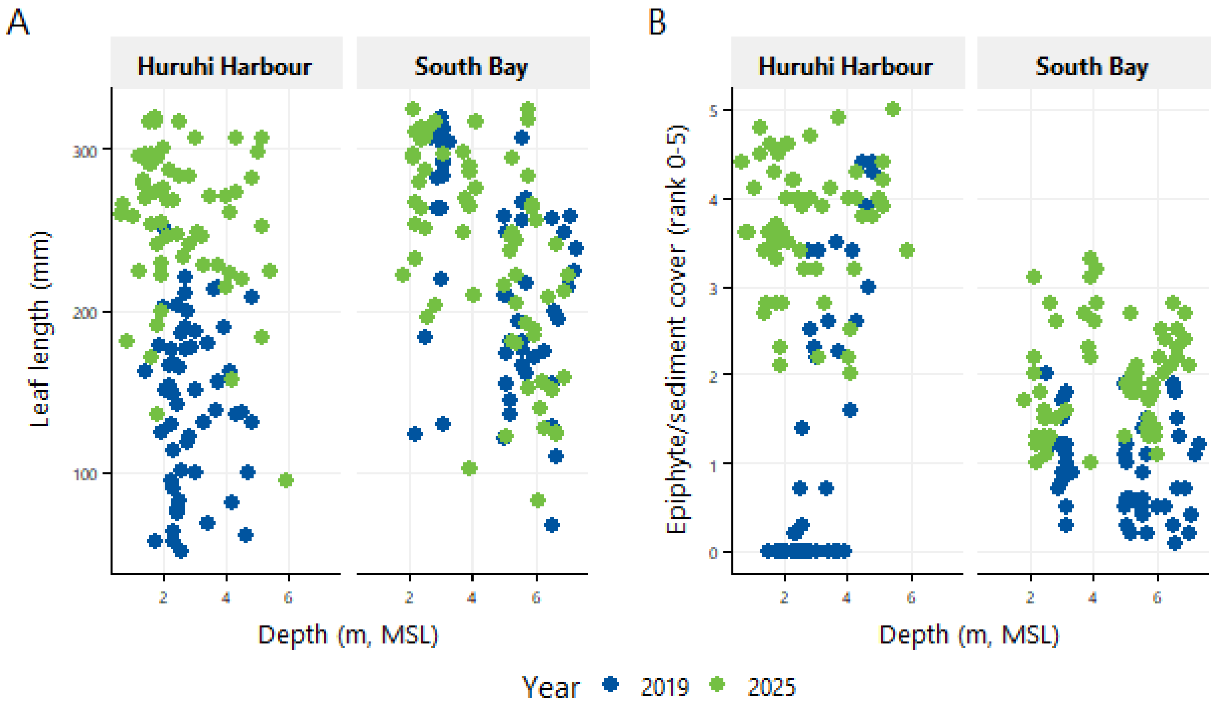


Figure 10. Seagrass condition and stress indicators regressed against depth at South Bay, Slipper Island / Whakahau, and Huruhi Harbour, Great Mercury Island / Ahuahu, at two time points: 2019 (blue dots) and 2025 (green dots). Indicators include (A) leaf length (mm), and (B) epiphyte / sediment cover rank (0 = 0%; 1 = 1%; 2 = 10%; 3 = 20%; 4 = 40%; 5 = 80%). Replicates for both indicators are a mean across 10 seagrass blades.

Table 1. Results from comparison of seagrass cover (Braun-Blanquet scale) against sampling year (2025 and 2019), depth and the interaction of both for South Bay, Slipper Island / Whakahau and Huruhi Harbour, Great Mercury Island / Ahuahu. A cumulative link mixed model fitted with the Laplace approximation was used for comparison at each site. Significant values (p < 0.05) are in bold.

Predictors	Huruhi Harbour			South Bay		
	Estimates	CI	*p*	Estimates	CI	*p*
Year [2025]	11.20	1.57–83.39	**0.019**	13.01	0.83–292.88	0.084
Depth	0.74	0.44–1.25	0.266	0.53	0.38–0.72	**< 0.001**
Year [2025] × Depth	0.98	0.51–1.87	0.945	0.68	0.39–1.14	0.160

Table 2. Results from comparison of above-ground biomass (gDW·m^{-2}) against sampling year (2025 and 2019), depth and the interaction of both for South Bay, Slipper Island / Whakahau and Huruhi Harbour, Great Mercury Island / Ahuahu. A linear model was used for comparison at each site. Significant values (p < 0.05) are in bold.

Predictors	Huruhi Harbour			South Bay		
	Estimates	Std. error	*p*	Estimates	Std. error	*p*
(Intercept)	78.1	30.09	**0.017**	311.2	61.813	**< 0.001**
Year [2025]	-26.3	35.41	0.466	10.85	80.160	0.894
Depth	-9.60	10.02	0.350	-39.7	11.546	**0.002**
Year [2025] × Depth	10.0	11.88	0.410	-4.72	15.391	0.762

Table 3. Results from comparison of leaf length (mm) against sampling year (2025 and 2019), depth and the interaction of both for South Bay, Slipper Island / Whakahau and Huruhi Harbour, Great Mercury Island / Ahuahu. A linear model was used for comparison at each site. Replicates are a mean across 10 seagrass blades and significant values (p < 0.05) are in bold.

Predictors	Huruhi Harbour			South Bay		
	Estimates	Std. error	*p*	Estimates	Std. error	*p*
(Intercept)	140.36	23.72	**< 0.001**	317.23	25.19	**< 0.001**
Year [2025]	134.26	27.56	**< 0.001**	12.98	32.53	0.690
Depth	0.56	7.82	0.943	-19.45	4.97	**< 0.001**
Year [2025] × Depth	-8.25	9.11	0.365	-1.65	6.58	0.802

Indicators of seagrass stress

Macroalgae cover was uncommon in both seagrass meadows in 2025, which was similar to results from 2019. Occasionally drift macroalgae, such as *Ecklonia radiata,* was found along transects. However, in 2019, turfing coralline algae was observed throughout Transect 5 and Transect 6 in South Bay. In 2025, turfing coralline algae was not present at either of these transects, but Transect 5 is also the area where we saw the most obvious loss of seagrass in South Bay, and likely associated coralline species.

In 2019, epiphyte / sediment cover was low at both sites, averaging about 1% (mean rank 0.9–1.1), but displayed opposite depth trends (Clark and Crossett 2019). Cover declined with depth at South Bay and increased with depth at Huruhi Harbour. In 2025, epiphyte / sediment cover at South Bay remained low

(10%; mean rank 1.96 ± 0.04 SE; Figure 10 and see Appendices A6.1–A6.2), but the depth trend reversed slightly. There was no change in epiphyte / sediment cover in shallow areas but slightly more cover in deeper areas (Table 4; Figure 10). In contrast, Huruhi Harbour showed a significant increase in epiphyte / sediment cover in 2025 (40%; mean rank 3.7 ± 0.04 SE), particularly at shallow depths, and the effect of depth was reversed, with lower epiphytes / sediment cover in deeper areas (Table 4; Figure 10 and see Appendices A6.3–A6.4).

In 2019, fungal wasting disease was present at both sites with similar prevalence (35–38%) and low severity (< 1%; Clark and Crossett 2019). In 2025, average disease prevalence at South Bay was similar to 2019 at 36% (± 2.0 SE) and average disease severity remained below < 1% cover on average (0.87 ± 0.05 SE). In 2019, disease prevalence and severity decreased with depth at South Bay, while in 2025, this depth pattern had reversed or diminished (Figure 11, Tables 5 and 6). At Huruhi Harbour, disease prevalence increased significantly in 2025 compared to 2019, with an average prevalence of 78% (± 1.7 SE; Figure 11 and Table 5). Average severity also increased from 2019 to approaching 10% in 2025 (mean rank 1.73 ± 0.05 SE). There was also a noticeable increase in disease severity at shallower depths in 2025 compared to 2019, but no clear depth trend for disease prevalence was observed (Figure 11, Tables 5 and 6).

Table 4. Results from comparison of epiphyte / sediment cover (0–5 scale) against sampling year (2025 and 2019), depth and the interaction of both for South Bay, Slipper Island / Whakahau and Huruhi Harbour, Great Mercury Island / Ahuahu. A cumulative link mixed model fitted with the Laplace approximation (distance as random effect) was used for comparison at each site. Replicates are a mean across 10 seagrass blades and significant values (p < 0.05) are in bold.

Predictors	Huruhi Harbour			South Bay		
	Estimates	Z value	*p*	Estimates	Z value	*p*
Year [2025]	11.62	20.50	**< 0.001**	0.065	0.193	0.847
Depth	2.39	16.53	**< 0.001**	-0.254	-4.807	**< 0.001**
Year [2025] × Depth	-2.51	-16.23	**< 0.001**	0.381	5.561	**< 0.001**

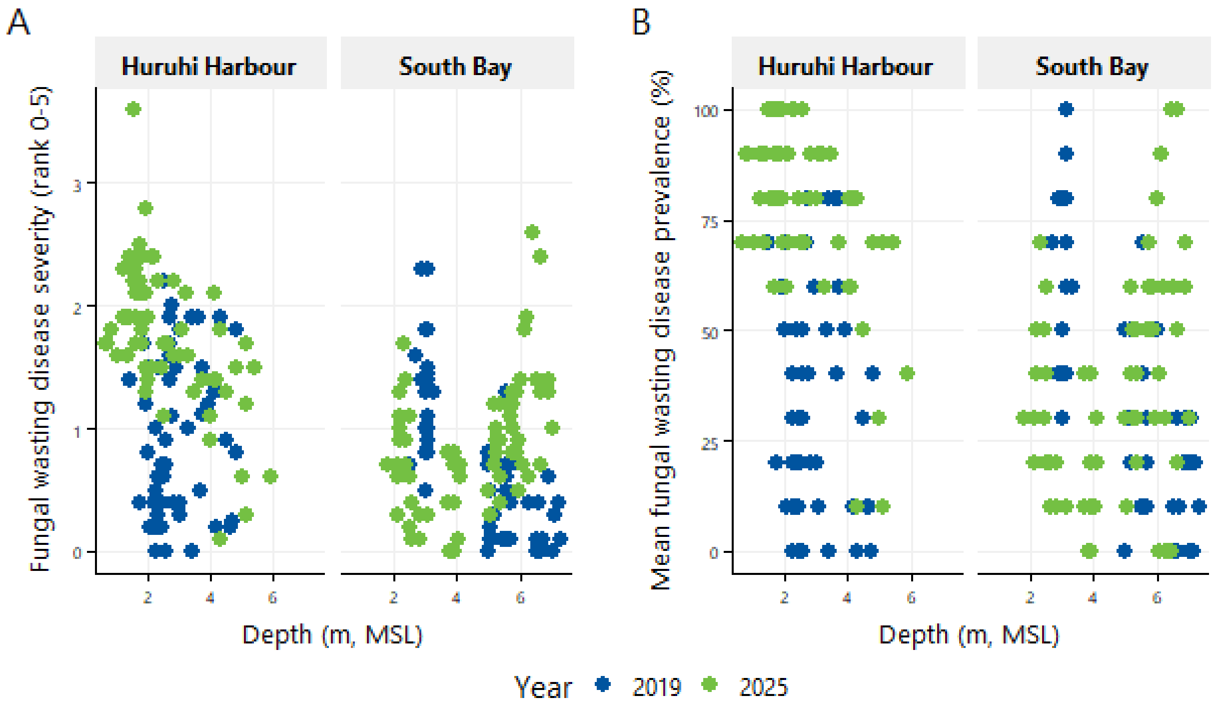


Figure 11. Seagrass condition and stress indicators regressed against depth at South Bay, Slipper Island / Whakahau and Huruhi Harbour, Great Mercury Island / Ahuahu, at two time points: 2019 (blue dots) and 2025 (green dots). Indicators include (A) fungal wasting disease severity (wasting index rank, where 0 = 0%; 1 = 1%; 2 = 10%; 3 = 20%; 4 = 40%; 5 = 80%) and (B) fungal wasting disease prevalence (%). Replicates for both indicators are a mean across 10 seagrass blades.

Table 5. Results from comparison of average fungal wasting disease prevalence (%) against sampling year (2025 and 2019), depth and the interaction of both for South Bay, Slipper Island / Whakahau and Huruhi Harbour, Great Mercury Island / Ahuahu. A generalised linear model, binomial error family was used for comparison at each site. Replicates are a mean across 10 seagrass blades and significant values (p < 0.05) are in bold.

Predictors	Huruhi Harbour			South Bay		
	Odds ratios	CI	*p*	Odds ratios	CI	*p*
(Intercept)	0.53	0.08 – 3.44	0.501	7.95	1.31 – 55.85	**0.029**
Year [2025]	25.85	2.39 – 313.97	**0.008**	0.04	0.00 – 0.39	**0.007**
Depth	1.05	0.56 – 1.93	0.875	0.57	0.38 – 0.82	**0.004**
Year [2025] × Depth	0.60	0.28 – 1.28	0.179	2.00	1.25 – 3.32	**0.005**

Table 6. Results from comparison of fungal wasting disease severity (wasting index, 0–5 scale) against sampling year (2025 and 2019), depth and the interaction of both for South Bay, Slipper Island / Whakahau and Huruhi Harbour, Great Mercury Island / Ahuahu. A cumulative link mixed model fitted with the Laplace approximation (distance as random effect) was used for comparison at each site. Replicates are a mean across 10 seagrass blades and significant values (p < 0.05) are in bold.

Predictors	Huruhi Harbour			South Bay		
	Estimates	CI	*p*	Estimates	CI	*p*
Year [2025]	2.79	7.738	**< 0.001**	-3.46	-9.200	**< 0.001**
Depth	0.23	2.206	**0.0274**	-0.53	-9.235	**< 0.001**
Year [2025] × Depth	-0.54	-4.616	**< 0.001**	0.79	10.212	**< 0.001**

3.3 Visual biomass assessment

The visual biomass assessment techniques were evaluated using 50 above-ground biomass samples collected between 2019 and 2025 from both meadows (Huruhi Harbour and South Bay). Above-ground biomass values ranged from 0 to 382 $gDW \cdot m^{-2}$ (Figures 9 and 12). Of the three methods tested, the visual biomass rank showed the strongest correlation with quantitatively measured above-ground biomass, with a marginal R^2 value of 0.85 (Figure 12, Table 7A). In comparison, Braun-Blanquet cover and DOR method had lower marginal R^2 values of 0.78 and 0.76, respectively. The visual biomass rank method was also the most consistent across years, with the year of sampling / observer having no significant effect on the relationship between visual ranks and quantitative biomass, nor altering overall trends (Tables 7A and 7B). In contrast, biomass estimates for both the Braun-Blanquet and DOR methods were significantly influenced by year and its interaction with biomass (Table 7A), with notable changes in relationship trends across the years (Table 7B). Generally, all three visual assessment methods aligned more closely with quantitative biomass at lower biomass values (Figure 12). However, as biomass increased, variation in visual estimates also increased, resulting in reduced accuracy and reliability in visual assessments at higher biomass levels.

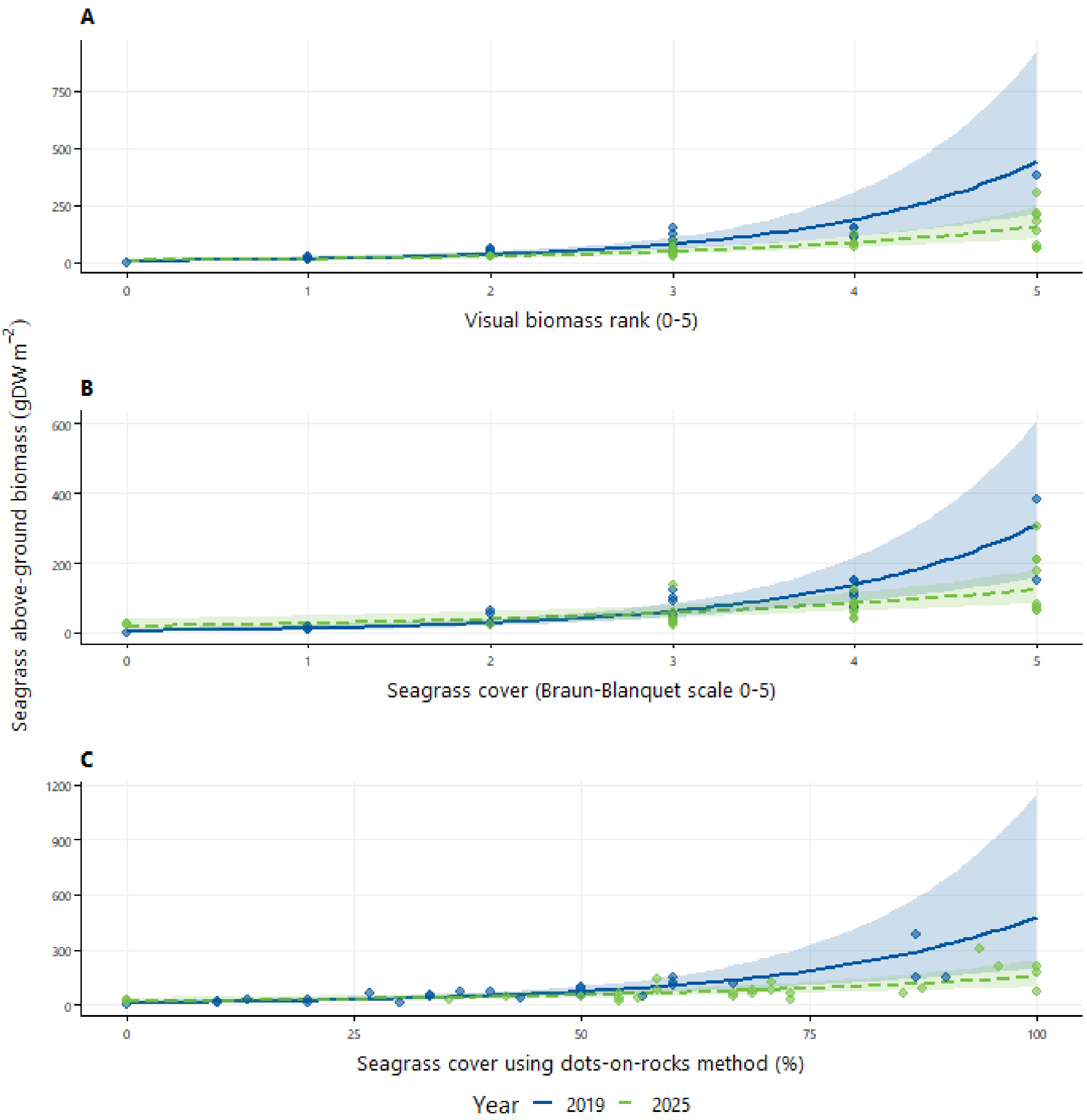


Figure 12. Quantitatively harvested above-ground biomass (gDW·m^{-2}) compared with (A) visual biomass rank (0–5 scale) and (B) seagrass cover estimated using the Braun-Blanquet scale (0–5) and (C) seagrass cover estimated using dots-on-rocks method (%), *n* = 50. Data from both Slipper Island / Whakahau and Great Mercury Island / Ahuahu are combined and displayed at two time points: 2019 (blue dots) and 2025 (green dots). The lines represent the estimated values from the generalised mixed effect models (with a gamma family error distribution, and year as a random effect, to account for the potential not independence of data). The shaded area shows the 95% confidence interval around the predicted line. Because the gamma family only allow strictly positive values, we transformed the data adding 0.000001 to all the above-ground biomass data. This addition is less that the measurement error of the data.

Table 7. (A) Results from comparison using generalised linear mixed models (GLMM) of quantitatively harvested above-ground biomass ($gDW \cdot m^{-2}$) with three visual measures: visual biomass rank (0–5 scale), seagrass cover estimated using the Braun-Blanquet scale (0–5) and seagrass cover estimated using a dots-on-rocks method (%). We included Year (2025 and 2019) and its interaction with the visual scores to assess whether the relationship between visual and quantitative measures of biomass changed over time. We also added 'Year' as a random effect in the model to account for the non-independence of the data from one year to the next. (B) We also estimated and compare the trends for the two different years (2025 and 2019) in each model. We did this by constructing a reference grid of the predicted trends and averaging them over the predictors in the grid. Data from Slipper Island / Whakahau and Great Mercury Island / Ahuahu were combined for models and significant values ($p < 0.05$) are in bold.

A	Visual biomass rank (0–5)			Seagrass cover (0–5)			Seagrass cover (%)		
Predictors	**Estimates**	**CI**	***p***	**Estimates**	**CI**	***p***	**Estimates**	**CI**	***p***
(Intercept)	5.97	3.18 – 11.20	**< 0.001**	5.19	2.63 – 10.24	**< 0.001**	10.43	5.72 – 19.01	**< 0.001**
Year [2025]	1.28	0.40 – 4.10	0.678	3.54	1.28 – 9.81	**0.016**	1.76	0.68 – 4.58	0.239
Visual biomass rank (0–5)	2.33	1.86 – 2.90	**< 0.001**	–	–	–	–	–	–
Year : Visual biomass rank (0–5)	0.79	0.56 – 1.11	0.166	–	–	–	–	–	–
Seagrass cover (0–5)	–	–	–	2.27	1.82 – 2.83	**< 0.001**	–	–	–
Year : Seagrass cover (0–5)	–	–	–	0.65	0.48 – 0.86	**0.004**	–	–	–
Seagrass cover (%)	–	–	–	–	–	–	1.04	1.03 – 1.05	**< 0.001**
Year [2025] : Seagrass cover (%)	–	–	–	–	–	–	0.98	0.97 – 1.00	**0.046**
Marginal R^2	0.845			0.780			0.764		

B					
Predictors	**Contrast**	**Estimate**	**SE**	**Z ratio**	***p* value**
Visual biomass rank (0–5)	**Year 2019 – Year 2025**	0.238	0.169	1.407	0.159
Seagrass cover (Braun-Blanquet scale 0–5)	**Year 2019 – Year 2025**	0.438	0.144	3.022	**0.002**
Seagrass cover (%)	**Year 2019 – Year 2025**	0.015	0.007	2.049	**0.040**

3.4 Other observations

As observed in 2019, areas of anthropogenic disturbance were present at in the South Bay and Huruhi Harbour seagrass meadows in 2025. Most evident were the obvious impacts of swing moorings on the seagrass meadows, which scour the area around them of seagrass (Figure 13). At South Bay, divers estimated the disturbance areas of two swing moorings to be approximately 56.7 m^2 and 201 m^2, respectively. Depressions in the meadow caused by swing moorings can be deep, more than 1 m in some places, often accumulating dense drifts of seaweed (Figure 13). The seagrass meadow at South Bay contains approximately eight swing moorings, while there are approximately four in the meadow in Huruhi Harbour. In 2025, we estimated approximately 700 m^2 and 600 m^2 of damage from swing moorings at Huruhi Harbour and South Bay, respectively. Anchor and propeller scars were also evident at both meadows, but particularly in shallower parts of South Bay (Figure 13). Anchor and propellor scars were typically 2–3 m long, and we estimated 200 m^2 and 100 m^2 of damage from these at Huruhi Harbour and South Bay, respectively. No invasive *Caulerpa* seaweed was observed in 2025.

In 2025, large schools of juvenile fish were observed at South Bay, largely comprised of koheru / scad (*Decapterus koheru*) and / or hautere / jack mackerel (*Trachurus novaezelandiae*) (Figure 14). Juvenile and adult spotties (*Notolabrus celidotus*), trevally (*Pseudocaranx dentex*), snapper (*Pagrus auratus*), goatfish (*Upeneichthys lineatus*), various gobi and triplefins, pilchards (*Sardinops sagax*), hermit crabs (*Pagurus* sp.), comb sea stars (*Astropecten polyacanthus*), purple fanworm (*Branchiomma* sp.) and various whelks were also seen in the seagrass at South Bay. At Huruhi Harbour, fewer fish were observed; however, it should be noted that the in-water visibility was considerably poorer at this site. Numerous eagle rays (*Myliobatis tenuicaudatus*) were observed in the seagrass meadows at Huruhi Harbour as well as juvenile and adult spotties, snapper, kingfish (*Seriola lalandi lalandi*), triplefins, goatfish, gobies, hermit crabs, comb sea stars, parchment tubeworms (*Chaetopterus* sp.) and various whelks. Sponges and ascidians were also observed on submerged logs or branches.

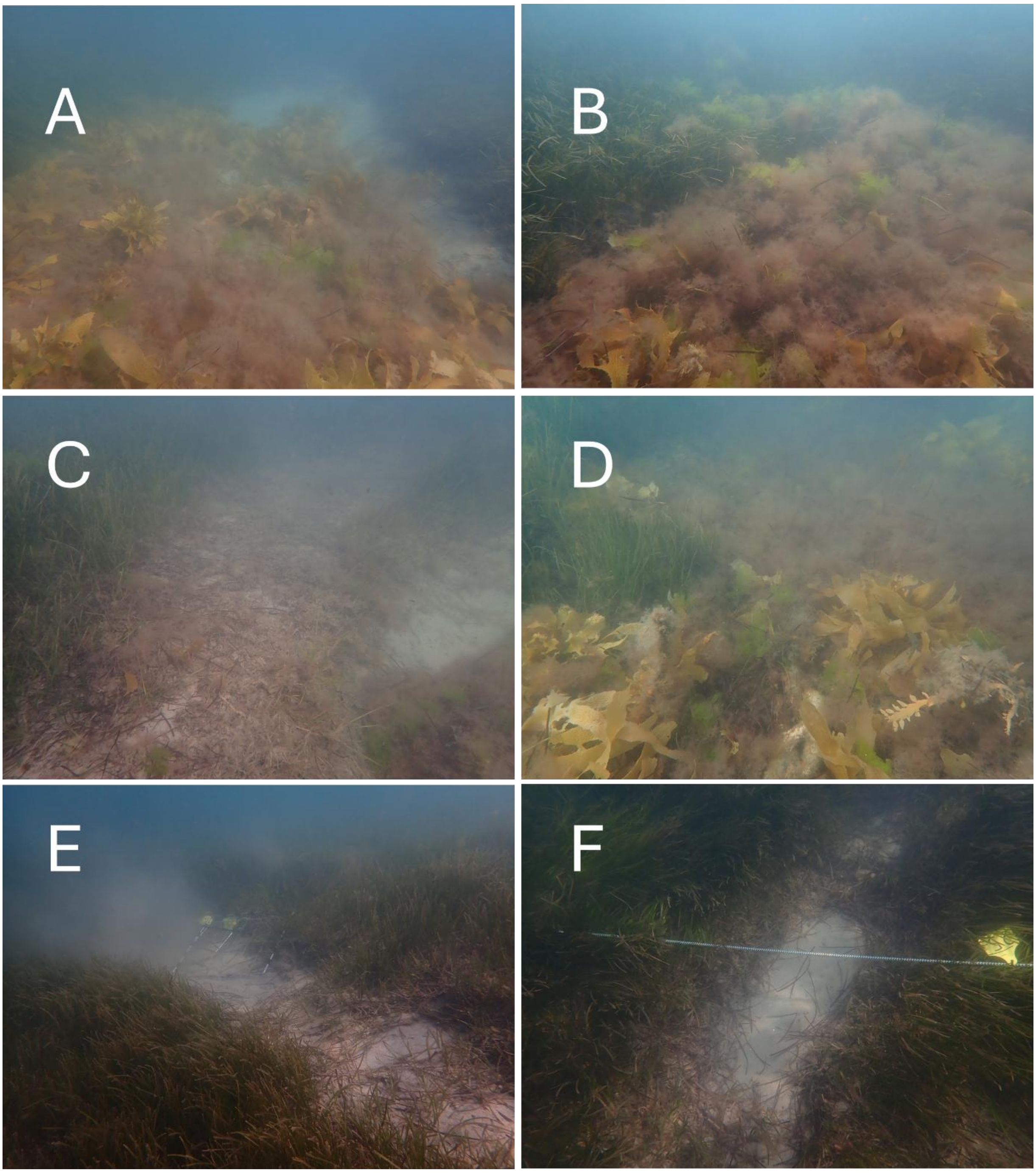


Figure 13. Examples of swing mooring impact (A–D) and anchor or propeller scarring (E–F) in seagrass meadow at South Bay, Slipper Island / Whakahau, in 2025. Note the dense accumulation of drift seaweed in images A, B and D due to large hollows created from swing mooring chain.

Figure 14. Schools of koheru / scad and / or hautere / jack mackerel at South Bay, Slipper Island / Whakahau, in 2025.

4. Discussion

4.1 Seagrass extent

At 0.18 km^2, the seagrass meadow at South Bay (Slipper Island) was found to be slightly smaller in 2025 than the 0.19 km^2 estimated in 2019; however, but the meadow remained six times larger than the area estimated in 2004 (Schwarz et al. 2006) and more than twice the size estimated in 1973 (Grace and Whitten 1974). The spatial extent of seagrass meadows can naturally vary year to year due to multiple environmental factors such as light availability, seawater temperature and nutrient levels (Olesen and Sand-Jensen 1994; Ismail 2001; Spalding et al. 2003; Turner and Schwarz 2006a). Some variation may also be attributed to the margin of error in mapping methods, which was much higher in the 1973 and 2004 surveys and likely underestimated true extent (see Clark and Crossett for further discussion). Additionally, the boundaries of both meadows were mapped in greater detail in the 2025 survey compared with 2019.

Between 2019 and 2025, we observed the greatest change in meadow extent around Transect 5 (approximately 5 m depth, see circle in Figure 4), where seagrass had retreated from south to north. This change was confirmed through aerial imagery. During the field surveys in 2025, we were approached by a local resident of Slipper Island, who described this retreat but also reported notable expansion of seagrass on the northern end of the meadow, extending into deeper waters and into Stingray Bay. This reported northern expansion was not observed in aerial imagery or diver surveys in 2025 (see Figure 4), possibly because it lies at depths beyond the effective range of these methods. Seagrass was generally dense across most of the meadow, becoming patchier near the edges and with depth. We also identified a new patch of seagrass on the southern end of South Bay. As in 2019, we benefited from favourable field conditions and enhanced confidence in the accuracy of both aerial imagery ground-truthing and *in situ* dive surveys.

At Huruhi Harbour (Great Mercury Island), we found a nearly 10% increase in seagrass meadow extent in 2025 compared to 2019 (0.10 km^2 vs 0.09 km^2), and over a 30% increase relative to 2004 estimates (Schwarz et al. 2006). Most of this expansion occurred towards the southeastern edge of the meadow, where seagrass extended further out of the harbour (Figure 6). Although improved field conditions in 2025 (i.e. better in-water visibility, no wind) enabled more accurate mapping of deeper areas, this southeastern expansion was also evident in aerial imagery, supporting the conclusion that it reflects a genuine increase in meadow extent (Appendix A5.5). Despite this recent expansion, seagrass extent in 2025 was greatly reduced compared to that reported in 1975 (Grace and Grace 1976), when seagrass was documented throughout the entire harbour, including intertidal areas. While the 1975 estimate was likely exaggerated – based on only five sampling stations (see Clark and Crossett 2019 for further discussion) – intertidal seagrass was still present in 2004, suggesting a continued retreated from the upper reaches of Huruhi Harbour, where no seagrass was observed in either 2019 or 2025.

When comparing methods for delineating seagrass extent, we found mixed results when using an AOV equipped with side-scan sonar to map seagrass extent. At South Bay, the side-scan sonar provided more detail on the spatial extent of the seagrass meadow compared to aerial imagery. With further refinement, this technology could provide greater mapping accuracy and reduce resource costs for future mapping (Buscombe 2017; Greene et al. 2018). However, the methods we used need to be tested further before they can be reliably applied in Aotearoa New Zealand, as side-scan sonar was less effective at Huruhi Harbour. This is likely because the seagrass at this site was less dense than at South Bay. Moreover, the higher fine sediment cover on the seagrass blades may have reduced the contrast needed to distinguish seagrass from the surrounding muddy substrate.

In addition to mapping the seagrass meadows at South Bay and Huruhi Harbour, we also carried out qualitative assessments of seagrass extent at Home Bay (Slipper Island) and Parapara Bay (Great Mercury Island). At Home Bay, only small patches of seagrass were observed growing on sand interspersed with cobble and boulders near the pier. At Parapara Bay, seagrass extent in 2025 appeared markedly reduced compared to 2019, with only a few small patches observed during the 2025 survey.

4.2 Seagrass condition

Similar to the 2004 and 2019 surveys, the seagrass meadow at South Bay (Slipper Island) was larger (nearly triple and double, respectively) than the meadow at Huruhi Harbour (Great Mercury Island) and extended to greater depths. The coarse sandy sediments at South Bay are less prone to resuspension than the finer sediments at Huruhi Harbour, allowing seagrass to grow deeper. At South Bay, depth had the greatest influence on seagrass condition, with seagrass cover, above-ground biomass and leaf length all greater at shallower depths. When accounting for depth, none of these indicators showed significant differences between 2019 and 2025, leaf length was comparable to 2004 and biomass appears to have increased since 2004 (Clark and Crossett 2019). However, our biomass estimates are based on a limited number of samples and should therefore be interpreted with caution. Epiphyte / sediment cover also declined with depth and remained low in both 2019 and 2025, although in 2025 there was a slight increase in deeper areas. Fine sediment was rarely observed on seagrass blades at this site; instead, there was more epiphytic algal growth, which is favoured by higher light availability in shallower waters.

At first glance − and similar to observations from the 2004 and 2019 surveys − the lush seagrass meadows at South Bay appeared healthier than those at Huruhi Harbour. This impression was supported by the greater in-water visibility and large schools of juvenile fish observed at South Bay. Quantitative analysis of above-ground biomass supported these observations, with biomass at South Bay nearly twice that of Huruhi Harbour, aligning with earlier reports. However, in 2025 – and contrary to the patterns observed in 2019 and 2004 – average leaf length was greater at Huruhi Harbour, and seagrass cover was similar between the two meadows. Notably, average leaf length at Huruhi Harbour in 2025 nearly doubled compared to 2019 and was four times greater than in 2004. Additionally, there has been a steady increase in above-ground biomass in Huruhi Harbour from 2004 to 2025, and between 2019 and 2025 seagrass has extended into deeper waters. These changes all suggest the meadow condition is improving at this site.

Nonetheless, these apparent trends should be interpreted cautiously, as they are based on only three survey points over two decades. Comparisons between meadows are further complicated by differences in survey design: transects at Huruhi Harbour were located in shallower waters (< 0.5 m in 2004 and 1.2–5.3 m in 2019 and 2025) than those at South Bay (2.2–7.6 m) and seagrass condition indicators all declined with depth at South Bay. At equivalent depths, seagrass condition indicators were higher at South Bay than Huruhi Harbour. In contrast, the lack of a relationship between these variables and depth at Huruhi Harbour may reflect the limited depth gradient at that site. Despite the absence of depth-related patterns in 2019 and 2025, it is possible that the very shallow depths of the Huruhi Harbour transects in 2004 may have contributed to shorter leaf lengths recorded in that survey.

Seagrass characteristics are known to fluctuate seasonally due to variations in photosynthetically available radiation, seawater temperature and nutrient availability (Turner and Schwarz 2006a). For example, lower above-ground biomass is often observed during winter months (Ramage and Schiel 1999; Ismail 2001; Turner and Schwarz 2006b), while longer leaf lengths have been reported in late summer / autumn or winter (Ismail 2001; Turner and Schwarz 2006b). Habitat factors, such as sediment characteristics and exposure, also have a significant impact on seagrass health (Ramage and Schiel 1999; Schwarz et al. 2006; Clark and Berthelsen 2021). The earlier timing of the 2025 survey (March[9]) compared to those in 2019 and 2004 (May–June) may have contributed to the observed improvements in seagrass condition at Huruhi Harbour. However, similar seasonal differences were not observed at South Bay, suggesting that the improvements at Huruhi Harbour may reflect genuine ecological change rather than seasonal variability alone. It is more likely that interannual environmental fluctuations, such as rainfall, frequency of large storm events, wind direction (Turner et al. 1999; Unsworth et al. 2019; Tang and Hadibarata 2022) or land management practices (Short and Wyllie-Echeverria 1996; Turner and Schwarz 2006a) had a greater influence on changes in seagrass condition at Huruhi Harbour between survey years.

At Huruhi Harbour, we suspect that sediment characteristics play a key role in determining seagrass condition. The fine, muddy sediments are easily resuspended, increasing water turbidity and reducing light availability, thus limiting the depth of the seagrass meadow. High loads of terrigenous fine sediment accumulation, often attributed to poor land management, can smother seagrass (Clark and Berthelsen 2021; Zabarte-Maeztu et al. 2021) and may have contributed to the loss of intertidal seagrass in the upper reaches of Huruhi Harbour. For example, there is also evidence of this in other estuaries along the Coromandel Peninsula (i.e., Wharekawa and Te Kouma; personal communication from Mike Townsend). Despite the increase in leaf length, seagrass cover and above-ground biomass, epiphyte / sediment cover increased by nearly fourfold at Huruhi Harbour in 2025 compared to 2019. This may appear contradictory but could be explained by recent environmental conditions. For example, water clarity may have been above average for Huruhi Harbour over the seagrass growing season (September–March) due to low rainfall and storm activity, allowing seagrass to thrive. Fine sediment may have accumulated shortly before the 2025 surveys, potentially following a storm event and subsequent period of calm weather. The fact that epiphyte / sediment cover did not increase with depth at Huruhi Harbour in 2025, as in 2019, may also confirm this because it is likely that fine sediments would resuspend in the shallows and be deposited into deeper areas due to strong tidal currents. It is

[9] Note that this was done because WRC wanted to see if anthropogenic impacts (i.e. anchor or propeller scarring) were more obvious shortly after summer, as there is generally a high volume of boat traffic in these areas during this period.

also possible that the longer leaf lengths at this site were a response to the increased sediment, as seagrass may lengthen their leaves under low light conditions (Shanahan et al. 2023).

Fungal wasting disease was present in both meadows in 2025. At South Bay, fungal wasting disease prevalence was similar between 2019 and 2025 at around 35%, and the severity remained at less than 1% coverage. Huruhi Harbour exhibited a significant increase in the disease, with prevalence more than doubling to 78%, and severity increasing from less than 1% coverage to 10%. These findings are concerning, particularly at Huruhi Harbour, as fungal wasting disease can become lethal to seagrass at 25% coverage (Durako and Kuss 1994; Ralph and Short 2002). *Labyrithula*, the bacteria responsible for fungal wasting disease, tends to proliferate during periods of low light, warm temperatures and high salinity (Ralph and Short 2002), and increased agriculture run-off (Hughes et al. 2018). It is difficult to identify which specific factors may have contributed to the increase in fungal wasting disease. At Huruhi Harbour, this is further complicated by the simultaneous improvement in seagrass condition indicators (seagrass cover, above-ground biomass and leaf length) at this site. One plausible explanation is that an increase in nutrient input from agricultural run-off, coinciding with a period of relatively calm weather, may have contributed to both elevated fungal wasting disease and seagrass growth. These findings underscore the importance of considering a suite of indicators when evaluating the condition of a seagrass meadow. While some metrics may suggest recovery or resilience, others – such as disease prevalence – may point to emerging stressors with the potential to undermine ecosystem health over time.

At both meadows, there was evidence of anthropogenic impact, particularly from propeller or anchor scarring and swing moorings. Similar impacts were observed during the 2019 surveys (Clark and Crossett 2019). In 2025, using recent aerial imagery and targeted *in situ* dive assessments, we estimated approximately 700 m^2 of damage from swing moorings and 200 m^2 from anchor / propeller scarring in Huruhi Harbour. At South Bay, estimated damage was slightly lower, with around 600 m^2 attributed to swing moorings and 100 m^2 to anchor or propeller activity. If combining estimated mooring and scarring, damage would be around 1% loss of seagrass at Huruhi Harbour and approximately 0.35% loss of seagrass at South Bay. However, we note this is likely an underestimate, for example, many of the swing mooring impact zones in South Bay were covered by dense accumulations of drift macroalgae, which likely reduced the accuracy of aerial image-based assessments by making it difficult to distinguish these areas from the adjacent seagrass meadow.

Overall, we conclude that the condition of the South Bay seagrass meadow at Slipper Island has remained relatively stable since the 2019 and 2004 surveys. In contrast, the Huruhi Harbour meadow at Great Mercury Island has shown progressive improvement in condition since 2004, with increases in seagrass cover, above-ground biomass and leaf length. However, these positive trends were accompanied by increased levels of epiphyte / sediment cover and a concerning increase in fungal wasting disease compared to the 2019 survey.

4.3 Visual biomass assessment

In 2019 and 2025, we trialled three visual biomass assessment techniques as non-destructive and rapid methods to estimate above-ground biomass in seagrass meadows. Similar to 2019, the visual biomass rank method had the strongest correlation with quantitative sampling when data from both years and sites were combined. In 2025, the coefficient of determination was 0.85, higher than in 2019 (0.75 R^2; Clark and Crossett 2019), and within the range of other studies (0.65–0.96 R^2; Mellors 1991). The other two methods also performed well when assessing data from both years together. Seagrass cover estimated using the Braun-Blanquet scale (0–5 rank) and DOR method were also both suitable proxies for quantitative sampling of above-ground biomass with R^2 values of 0.78 and 0.76, respectively.

The visual biomass rank method remained the most reliable for estimating above-ground biomass, as it was unaffected by the year of sampling / observer and produced consistent trends across years. However, the visual biomass rank method requires the most effort in the field, compared to the other two methods, which can be estimated from photos after the field work is completed.

All visual assessment methods showed better agreement with quantitative sampling lower biomass values. As biomass increased, the variation between visual estimates and quantitative estimates also increased, reducing accuracy at higher biomass levels.

4.4 Recommendations for future monitoring and management

Monitoring

Protecting the few known subtidal seagrass meadows in Aotearoa New Zealand (such as those in WRC's region) is critical due to the many ecosystems services they provide. The seagrass meadows at Slipper and Great Mercury Islands are among the few known subtidal seagrass habitats in Aotearoa New Zealand and have been shown to play an important role in supporting biodiversity and fish populations (Schwarz et al. 2006). Ongoing monitoring is essential to detect early signs of change and enable timely management interventions. To support this, it is important to collect consistent seagrass condition data through time (i.e. this report; Schwarz et al. 2006; Clark and Crossett 2019) and maintain access to high-quality aerial imagery. Consistent with national recommendations for council-led seagrass monitoring (Shanahan et al. 2023), Huruhi Harbour and South Bay could serve as sentinel sites, providing early indications of ecological change within the region. We recommend for surveys to be conducted at both meadows every 3–5 years (i.e. by 2028–30), ideally in March, to capture potential vessel impacts and ensure comparability with 2025 data. More frequent monitoring could provide better insights into the influence of interannual variation in weather patterns or anthropogenic impacts, and possibly allow for earlier detection of ecological change; however, this would require greater investment by WRC.

For future surveys, we recommend continued monitoring of key indicators of seagrass condition (seagrass cover, leaf length, above-ground biomass) as well as indicators of stress (macroalgae cover, epiphyte / sediment cover, and the severity and prevalence of fungal wasting disease). We also recommend acquiring high-quality aerial photographs taken within 12 months of the ground-truthing surveys and suggest that high resolution satellite imagery is explored in the future. Although potentially more costly, satellite imagery may offer more accurate and timely representations of seagrass meadow extent, as it can be captured closer in time to ground-truthing surveys. The use of AOVs equipped with side-scan sonar and video capability (e.g. BlueBoat[10],[11]) may be useful for future surveys. However, further refinement of this technology is necessary, and all data collected should be ground-truthed using high-quality aerial or satellite imagery and *in situ* assessments to ensure accuracy. As recommended by Clark and Crossett (2019), physical parameters that are important for seagrass growth and survival (e.g. light, turbidity, depth, sediment characteristics, nutrient levels, temperature, storm events) could be included in future monitoring programmes so that changes in seagrass condition can be interpreted (Turner and Schwarz 2006a; Clark and Berthelsen 2021).

We now have confidence that the visual biomass ranks reliably reflect quantitatively measured above-ground biomass and are not significantly influenced by the year of survey / observer. Going forward, we recommend scaling up the sampling effort from two to 10 quadrats per transect (i.e. every 10 metres)[12] to better capture spatial variation in above-ground biomass across each meadow. The existing small quadrat size (0.0225 m^2) should continue to be used to ensure consistency when applying the visual ranking system. For each quadrat, above-ground biomass should be assessed *in situ* using the visual biomass rank method and a photo taken as a reference. We also recommend continuing to collect a limited number of above-ground biomass samples (e.g. 12 per meadow using 0.0225 m^2 quadrats) in future surveys. These data can be added to the 2019 and 2025 survey results, increasing the number of replicates in the seagrass cover-biomass regression and allowing differences between years or observers to be evaluated. In addition, it would be prudent to continue to estimate above-ground biomass using one of the seagrass cover measures (Braun-Blanquet or DOR methods), as both provided strong correlations with biomass and are less prone to observer bias. If assessments are made by different researchers at the next survey, it is essential that the visual estimates are tested again.

Although the extent and condition of the seagrass meadow at South Bay were similar to 2019, and improvements were observed at Huruhi Harbour, qualitative assessments at Home Bay (Slipper Island) and Parapara Bay (Great Mercury Island) suggest declines in meadow extent in these sites. We recommend continuing to monitor these meadows using high-quality aerial or satellite imagery, ideally ground-truthed at similar intervals, consistent with the surveys at South Bay and Huruhi Harbour. An important next step for Aotearoa New Zealand, and also recommended by Shanahan et al. (2023), is to identify and monitor other subtidal seagrass meadows that still exist around our coastline. This could involve scanning high-quality aerial or satellite imagery of offshore islands or areas where subtidal seagrass historically occurred. In addition, habitat suitability modelling based on known presence / absence data and environmental variables (i.e. Floerl et al. 2021; Bennett et al. 2022; Schattschneider and Floerl 2022; Shao et al. 2024) may help prioritise locations for further investigation. These locations can then be investigated using aerial or satellite imagery and validated through field surveys.

[10] Mapping seagrass with the BlueBoat, Omniscan SideScan, and the Washington DNR - General Discussion / Research - Blue Robotics Community Forums
[11] Payload bracket and Shallow water Habitat Mapping - Blue Robotics Vehicles / BlueBoat - Blue Robotics Community Forums
[12] A power analysis could be used to determine an optimal number of quadrats pert transect.

Conservation

Once seagrass meadows are lost, the locations that once supported them may become unsuitable for seagrass recovery, as key feedback mechanisms that maintain the necessary environmental conditions are often disrupted (Turner and Schwarz 2006a). Seagrass restoration efforts are typically expensive and laborious and have historically shown varied success, although new techniques are currently being developed (Berthelsen et al. 2024). Therefore, priority should be placed on the protection and conservation of existing seagrass meadows, supported by ongoing monitoring to track trends in distribution, extent and condition (Turner and Schwarz 2006a; Morrison et al. 2014b). High vessel traffic areas, such as within sheltered areas of Huruhi Harbour and South Bay, are vulnerable to anthropogenic impact and can become hotspots for invasive species (e.g. exotic *Caulerpa* species). Thus, protective measures should be considered for these areas, including implementing restrictions or raising public awareness to limit damage from vessel anchoring, swing moorings, propeller scarring and dredging, and the spread of invasive species. Conservation measures should also be linked in with initiatives such as Revitalising the Gulf[13] and the Hauraki Gulf / Tīkapa Moana Marine Protection Bill[14] to assist with management of these increasing rare, but highly valuable, subtidal habitats.

[13] Revitalising the Gulf: Government action on the Sea Change Plan: Our work
[14] New marine protections in the Hauraki Gulf

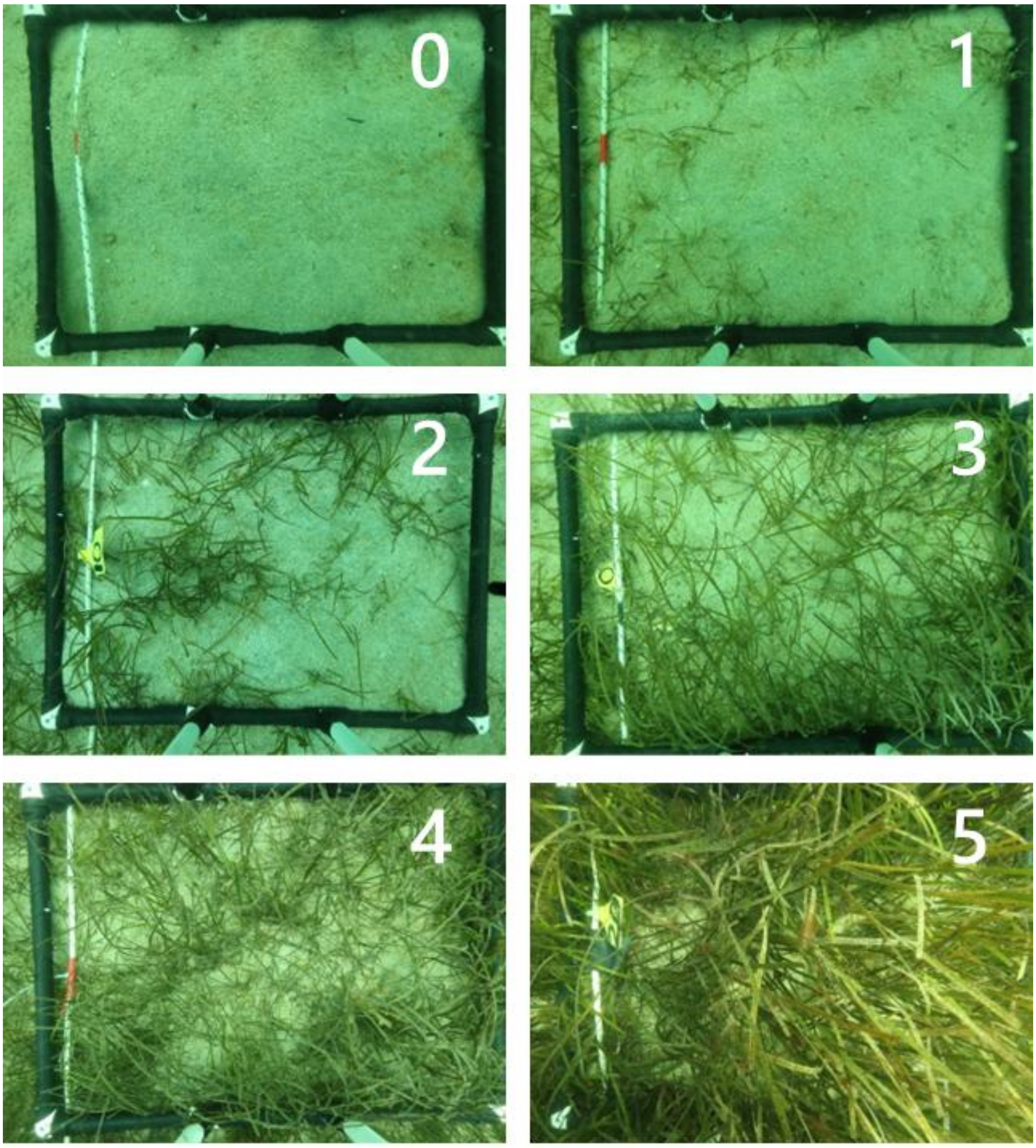
5. Appendices

Appendix 1. Transect locations with depth ranges from the 2019 and 2025 surveys

Transect	Depth range (m, MSL)		Start		End	
	2019	2025	NZTME	NZTMN	NZTME	NZTMN
Slipper T1	2.9–3.5	2.9–3.7	1861255	5894592	1861214	5894721
Slipper T2	2.2–3.1	2.0–2.6	1861201	5894747	1861143	5894843
Slipper T3	4.9–5.3	4.4–4.9	1861074	5894652	1861033	5894773
Slipper T4	5.5–5.8	4.4–5.2	1861018	5894711	1860976	5894835
Slipper T5	6.0–7.2	6.0–6.6	1860907	5894781	1860943	5894656
Slipper T6	6.5–7.6	6.3–7.3	1860873	5894920	1860901	5894801
Slipper T7	NA	3.6–4.0	1861150	5894673	1861104	5894770
Great Mercury T1	1.2–2.7	1.3–2.5	1848055	5945990	1847949	5945945
Great Mercury T2	1.3–2.7	1.0–2.1	1848101	5945953	1847993	5945913
Great Mercury T3	1.9–3.0	1.5–2.7	1848169	5945885	1848071	5945853
Great Mercury T4	2.1–4.8	1.5–3.1	1848251	5945845	1848128	5945805
Great Mercury T5	2.7–5.3	2.2–5.1	1848291	5945798	1848186	5945771
Great Mercury T6	3.4–4.8	3.1–4.2	1848338	5945748	1848210	5945712

Appendix 2. Seagrass cover classes

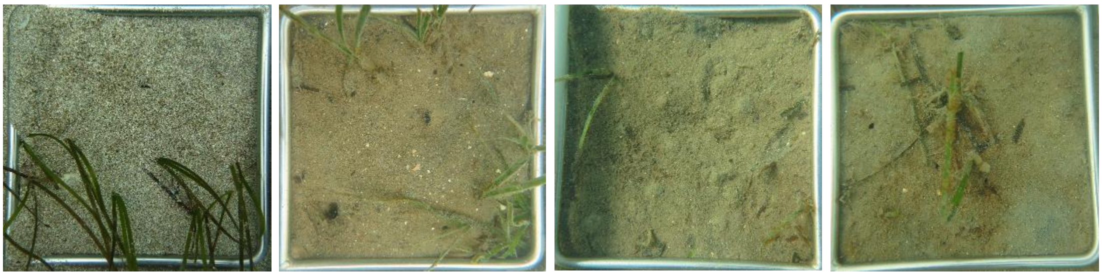
Photographs representing the Braun-Blanquet seagrass cover classes used in this survey. Cover classes are shown in the upper right corner of each image. cover class 0 (0%); cover class 1 (1–5%); cover class 2 (6–25%); cover class 3 (26–50%); cover class 4 (51–71%); cover class 5 (> 75%).



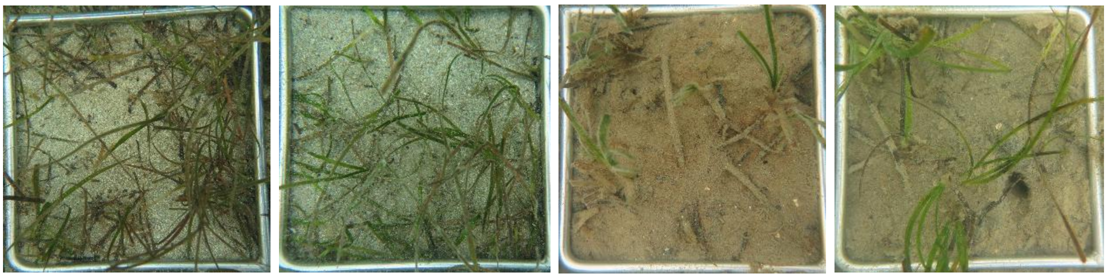
Appendix 3. Reference scale for visual biomass estimates

Photographs representing the range of visual biomass ranks used to estimate seagrass biomass in this survey. The visual biomass rank system was developed using photographs collected from South Bay and Huruhi Harbour in 2019. Note that quadrats used for the visual biomass estimates are 0.0225 m^2.

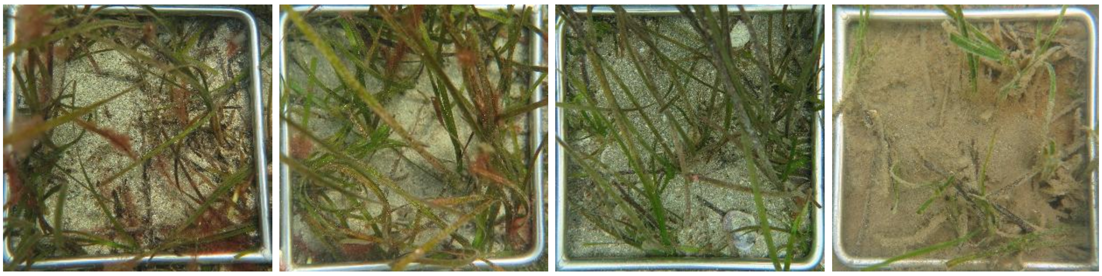
Rank 1



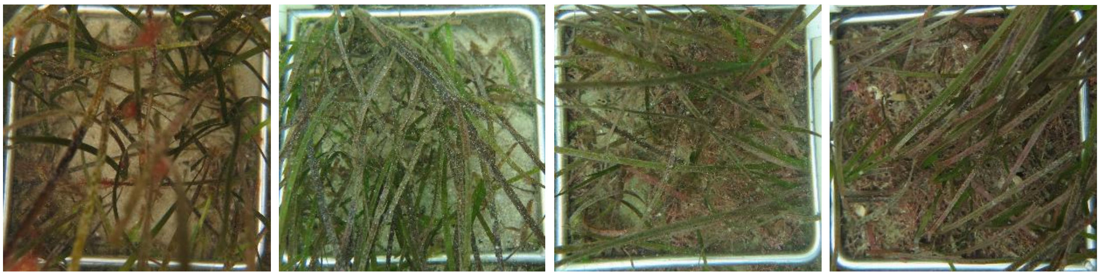
Rank 2



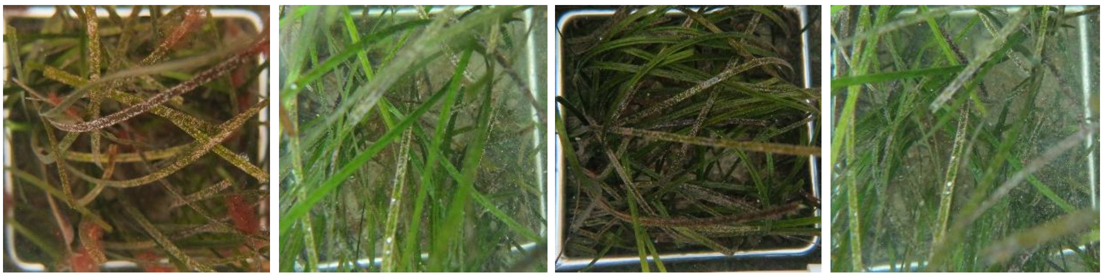
Rank 3



Rank 4



Rank 5



Appendix 4. Semi-quantitative scale for estimating epiphyte cover and severity of fungal wasting disease

The Wasting Index Method was developed by Burdick et al. (1993) as a rapid visual determination of the amount of necrotic tissue on seagrass shoots infected with fungal wasting disease (*Labyrinthula*). We used a semi-quantitative ranking system, corresponding to the percentage of disease cover in each class of the Wasting Index Key (Figure A 4.1), to estimate percentage cover of both fungal wasting disease and epiphyte cover. Figure A4.2 show examples of suspected fungal wasting disease on seagrass blades *in situ*.

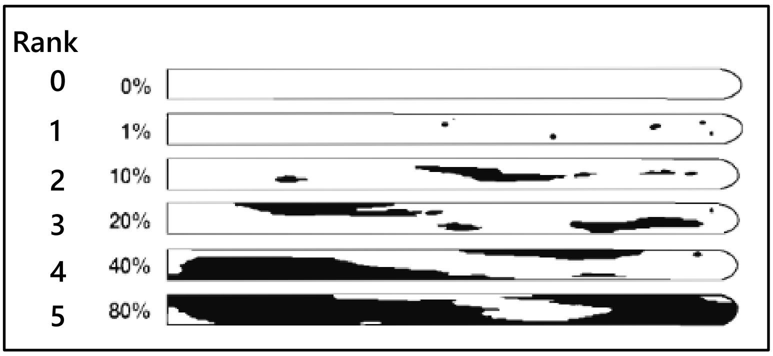


Figure A4.1. Ranks corresponding to the Wasting Index Key developed by Burdick et al. (1993).

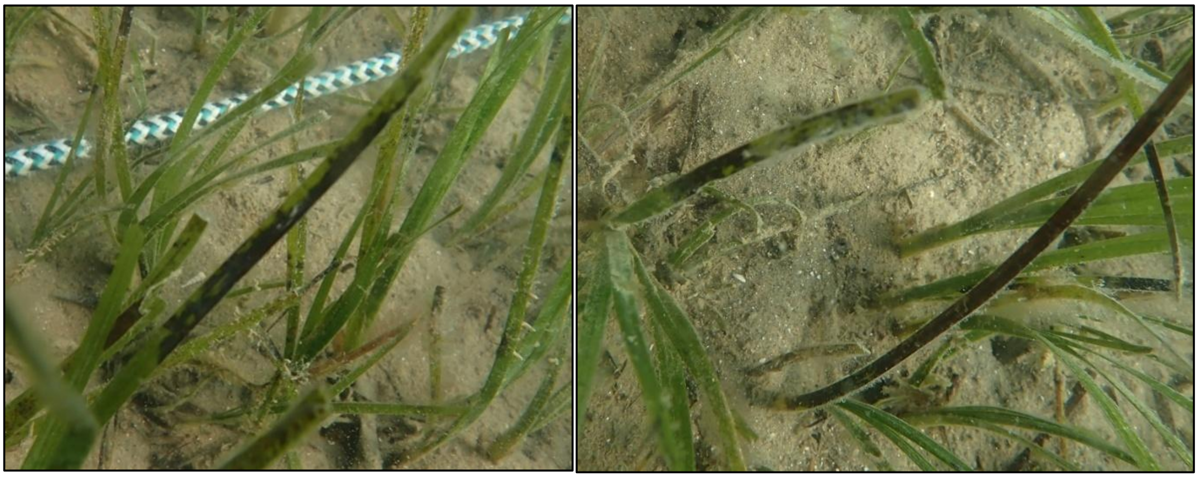


Figure A4.2. Photographs which show examples of suspected fungal wasting disease on seagrass blades *in situ*. Note the dark blotches covering seagrass blades in the centre of both photographs.

Appendix 5. Seagrass extent and aerial imagery through time

A5.1 Extent of the seagrass meadow at South Bay (south) and Stingray Bay (north), Slipper Island

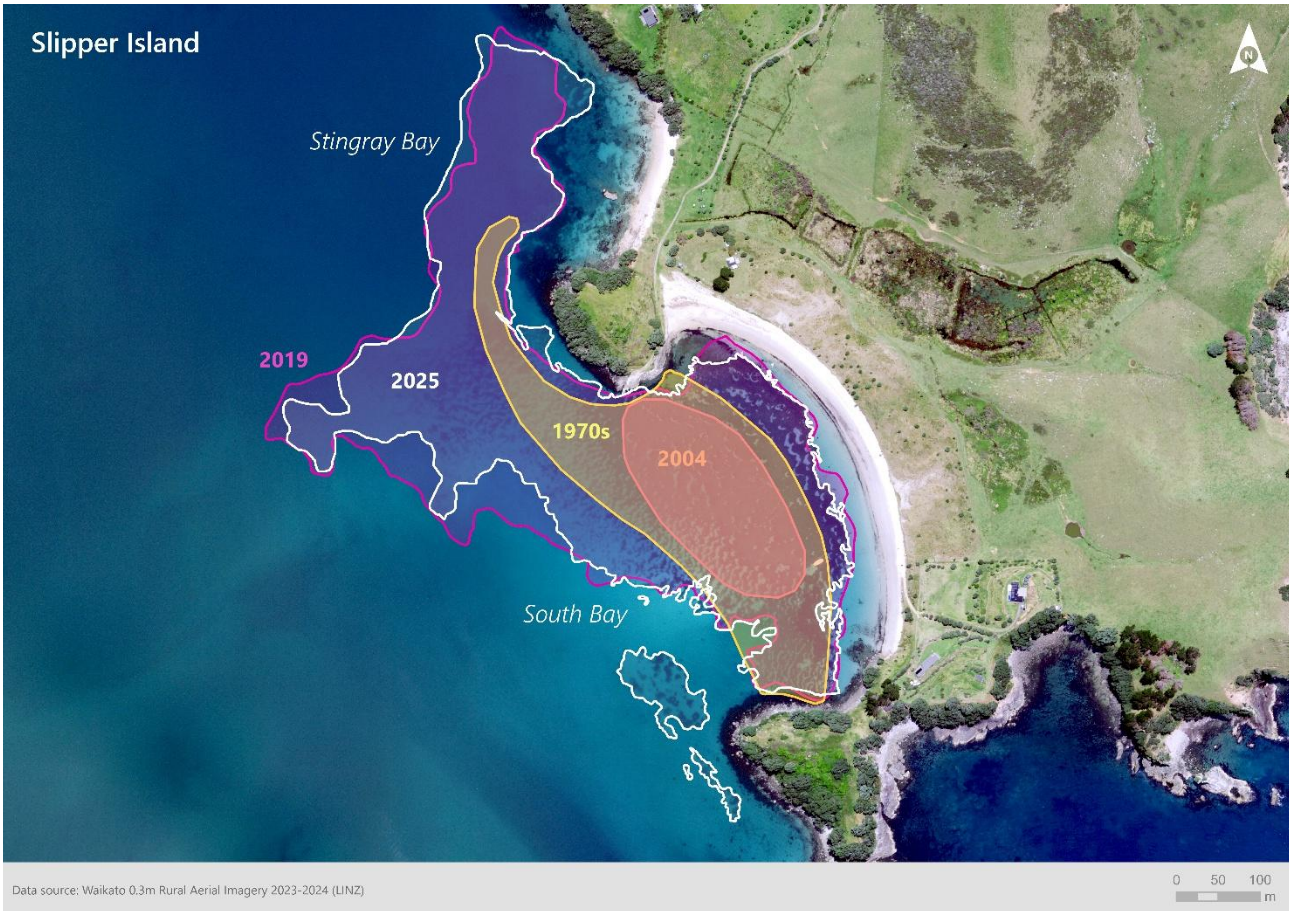


Figure A5.1. Extent of the seagrass meadow at South Bay (south) and Stingray Bay (north), Slipper Island / Whakahau, estimated in 2025 (white polygon), 2019 (pink polygon), 2004 (orange polygon) and 1973 (yellow polygon). Aerial image from 2023–24 and sourced from 'Waikato 0.3m Rural Aerial Photos' (LINZ Data Service).

A5.2 Demonstration of seagrass meadow extent over time, and between aerial images for South Bay and Stingray Bay, Slipper Island

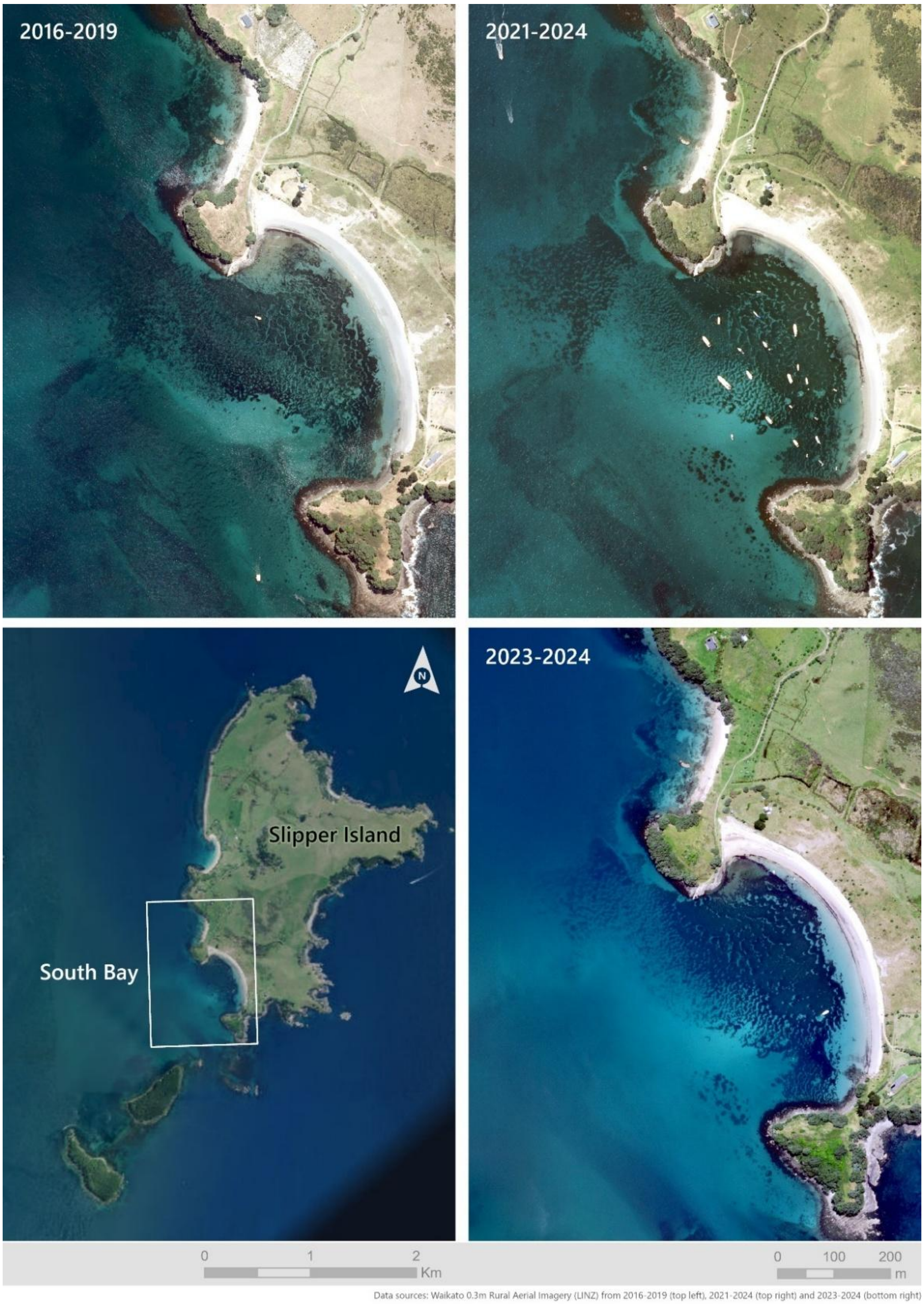


Figure A5.2. Demonstration of seagrass meadow extent over time, and between aerial images for South Bay and Stingray Bay, Slipper Island / Whakahau. Clockwise, from bottom left image: map of Slipper Island / Whakahau with box that represents South and Stingray Bays; aerial image from 2016–19; aerial image from 2021–24; and aerial image from 2023–24 (this image was used to for ground-truthing). Note the number of vessel (white objects) in the 2021–24 image. Data was sourced from 'Waikato 0.3m Rural Aerial Photos' (LINZ Data Service).

A5.3 Demonstration of seagrass meadow extent over time, and between aerial images for Home Bay, Slipper Island

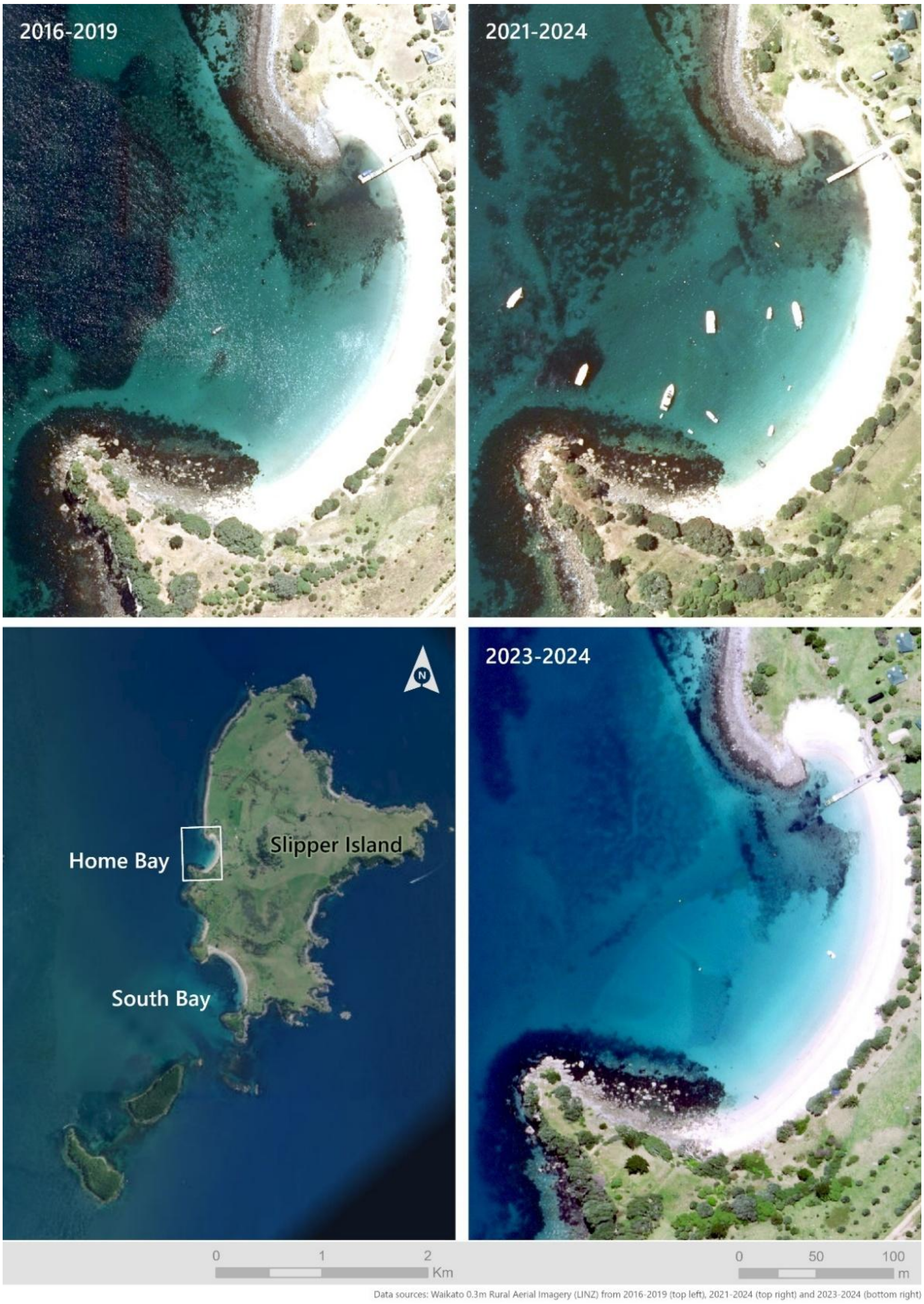


Figure A5.3. Demonstration of seagrass meadow extent over time, and between aerial images for Home Bay, Slipper Island / Whakahau. Clockwise, from bottom left image: map of Slipper Island / Whakahau with box that represents Home Bay; aerial image from 2016–19; aerial image from 2021–24; and aerial image from 2023–24 (this image was used to for ground-truthing). Note the number of vessel (white objects) in the 2021–24 image. Data sourced from 'Waikato 0.3m Rural Aerial Photos' (LINZ Data Service).

A5.4 Extent of the seagrass meadows at Huruhi Harbour (north) and Parapara Bay (east), Great Mercury Island

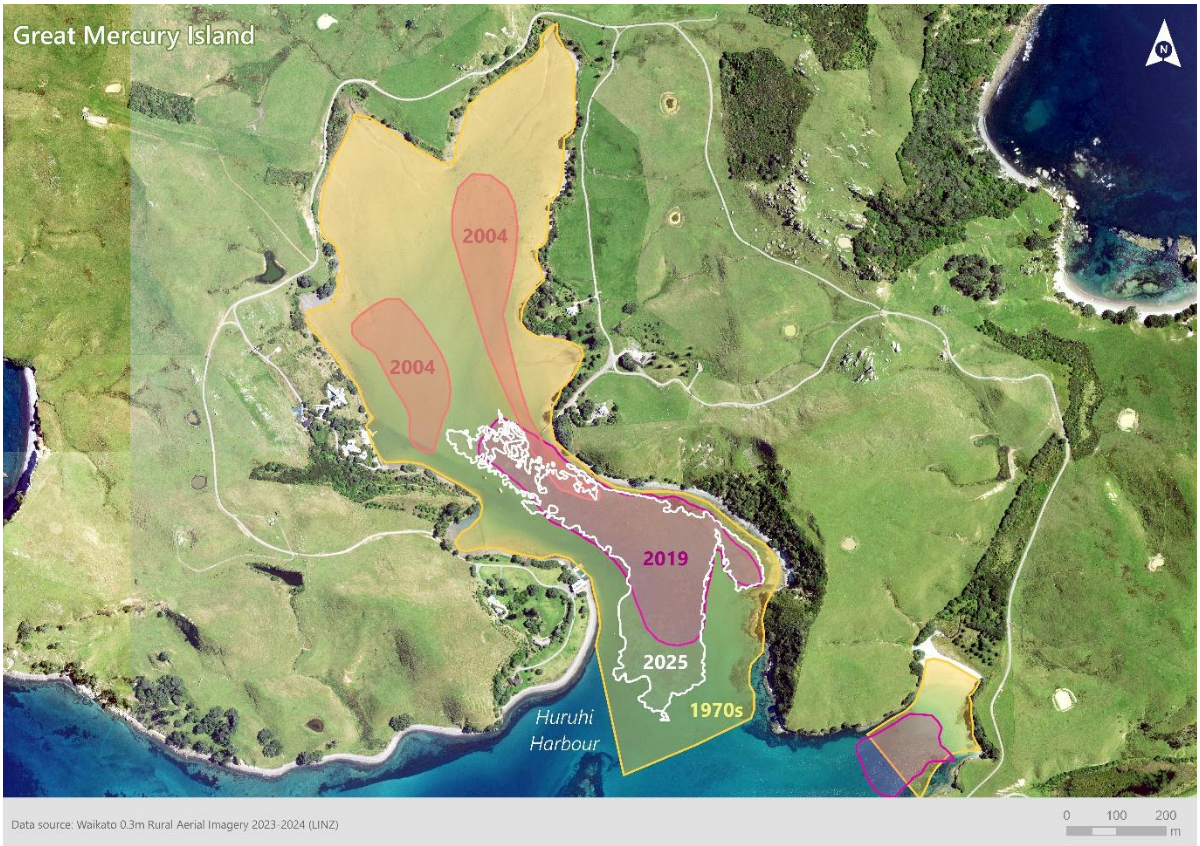


Figure A5.4. Extent of the seagrass meadows at Huruhi Harbour (north) and Parapara Bay (east), Great Mercury Island / Ahuahu, estimated in 2025 (white polygon), 2019 (pink polygon), 2004 (orange polygon) and 1973 (yellow polygon). Aerial image from 2023–24 and data sourced from 'Waikato 0.3m Rural Aerial Photos' (LINZ Data Service).

A5.5 Demonstration of seagrass meadow extent over time, and between aerial images for Huruhi Harbour, Great Mercury Island

Figure A5.5. Demonstration of seagrass meadow extent over time, and between aerial images for Huruhi Harbour, Great Mercury Island / Ahuahu. Clockwise, from bottom left image: map of Great Mercury Island / Ahuahu with box that represents Huruhi Harbour; aerial image from 2016–19; aerial image from 2021–24; and aerial image from 2023–24 (this image was used to for ground-truthing). Data sourced from 'Waikato 0.3m Rural Aerial Photos' (LINZ Data Service).

A5.6 Demonstration of seagrass meadow extent over time, and between aerial images for Parapara Bay, Great Mercury Island

Figure A5.6. Demonstration of seagrass meadow extent over time, and between aerial images for Parapara Bay, Great Mercury Island / Ahuahu. Clockwise, from bottom left image: map of Great Mercury Island / Ahuahu with box that represents Parapara Bay; aerial image from 2016–19; aerial image from 2021–24; and aerial image from 2023–24 (this image was used to for ground-truthing). Data sourced from 'Waikato 0.3m Rural Aerial Photos' (LINZ Data Service).

Appendix 6. Photo-quadrat images

A6.1 Photo-quadrat series of seagrass at 25 m on six transects at South Bay, Slipper Island, from 2025 and 2019

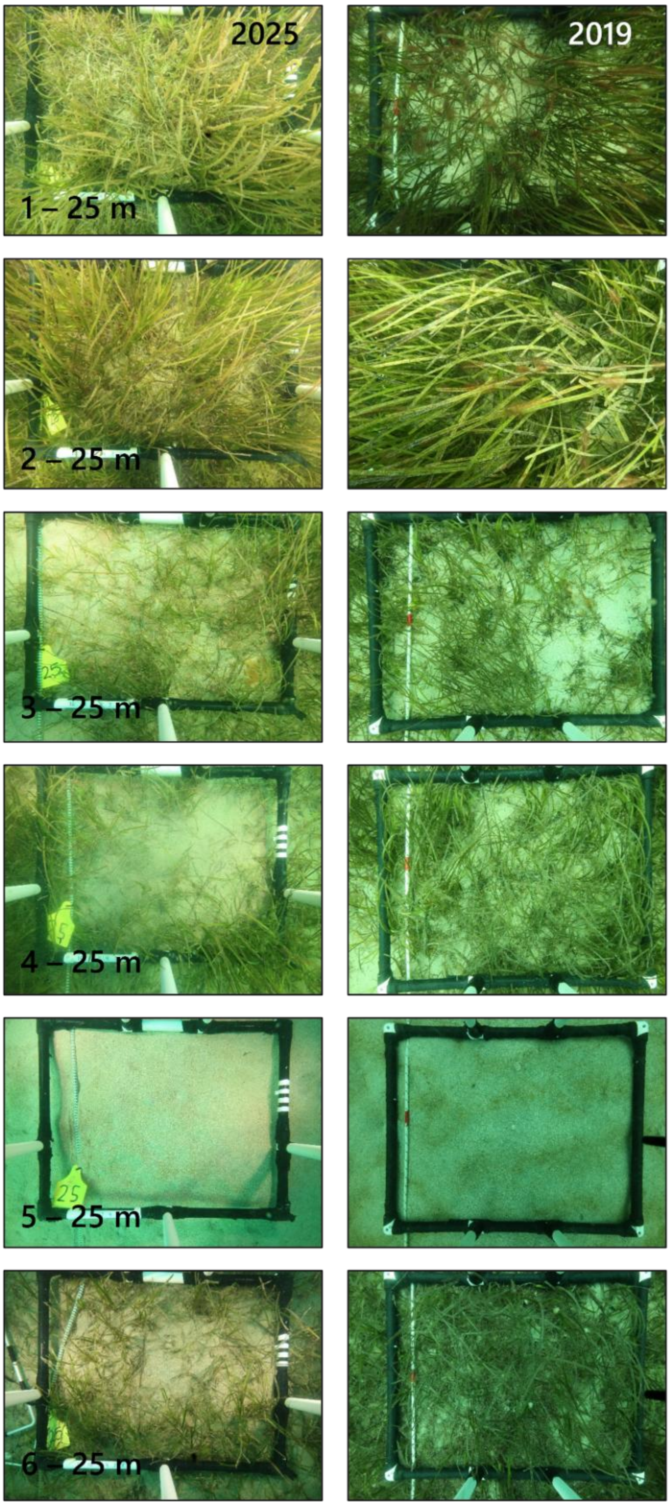


Figure A6.1. Photo-quadrat series of seagrass at 25 m on six transects at South Bay, Slipper Island / Whakahau, from 2025 (left column of images) and 2019 (right column of images). Images relate to subtidal transects 1–6, from top to bottom.

A6.2 Photo-quadrat series of seagrass at 75 m on six transects at South Bay, Slipper Island, from 2025 and 2019

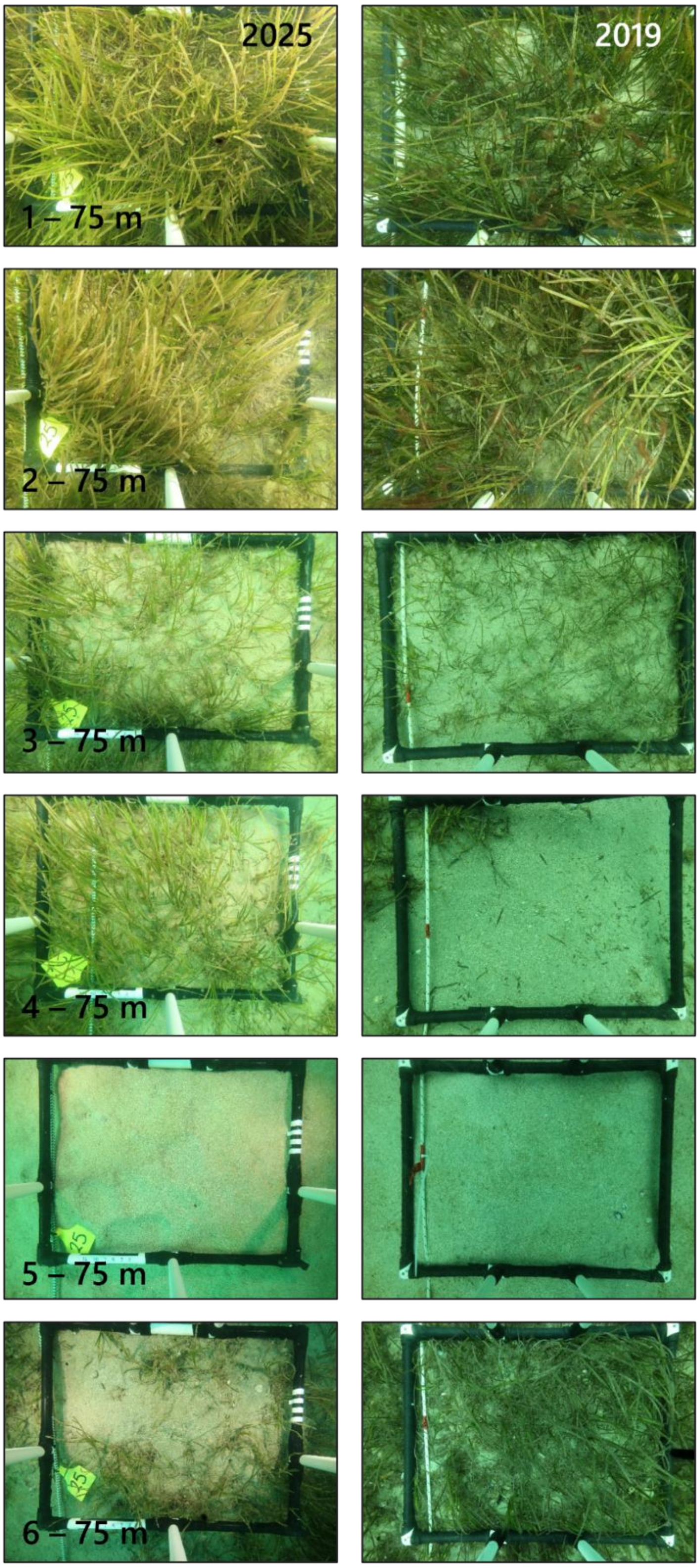


Figure A6.2. Photo-quadrat series of seagrass at 75 m on six transects at South Bay, Slipper Island / Whakahau, from 2025 (left column of images) and 2019 (right column of images). Images relate to subtidal transects 1–6, from top to bottom.

A6.3 Photo-quadrat series of seagrass at 25 m on six transects at Huruhi Harbour, Great Mercury Island, from 2025 and 2019

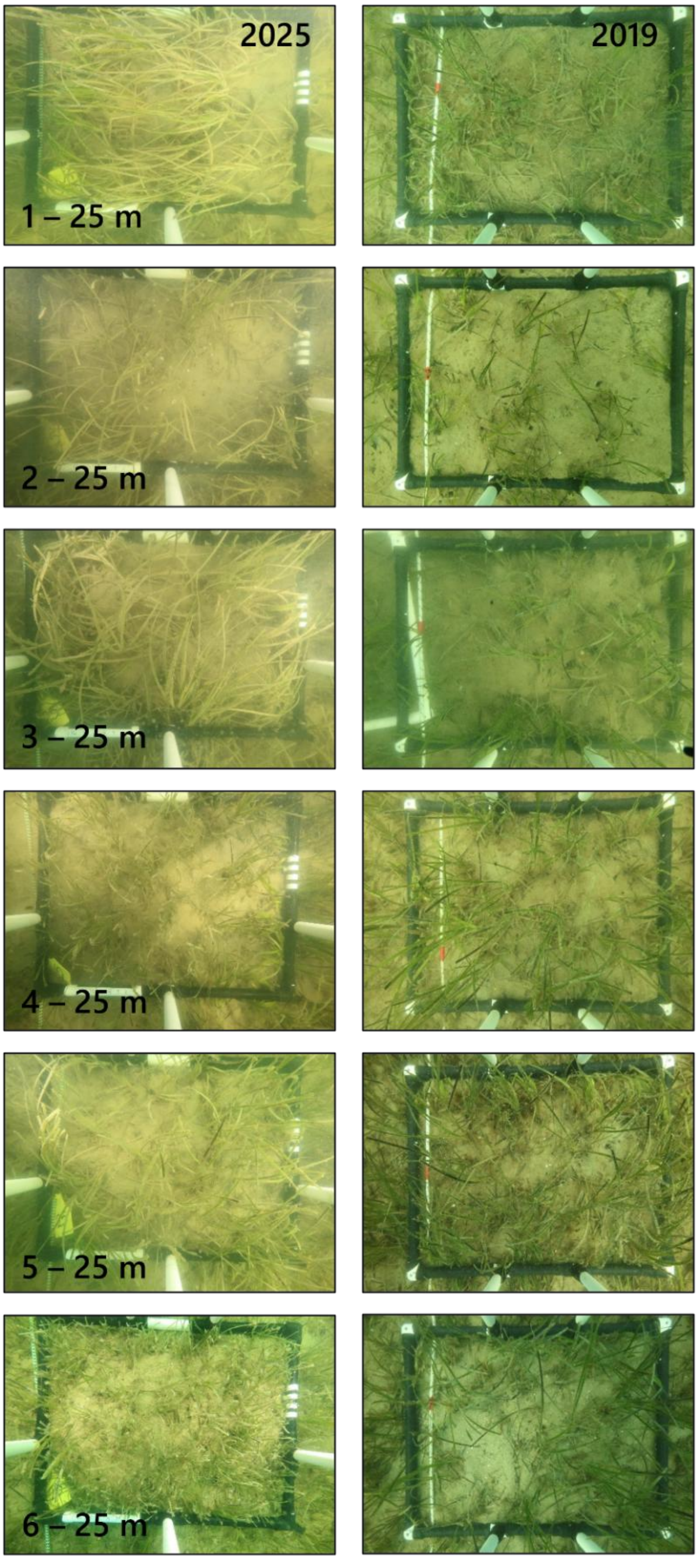


Figure A6.3. Photo-quadrat series of seagrass at 25 m on six transects at Huruhi Harbour, Great Mercury Island / Ahuahu, from 2025 (left column of images) and 2019 (right column of images). Images relate to subtidal transects 1–6, from top to bottom.

A6.4 Photo-quadrat series of seagrass at 75 m on six transects at Huruhi Harbour, Great Mercury Island, from 2025 and 2019

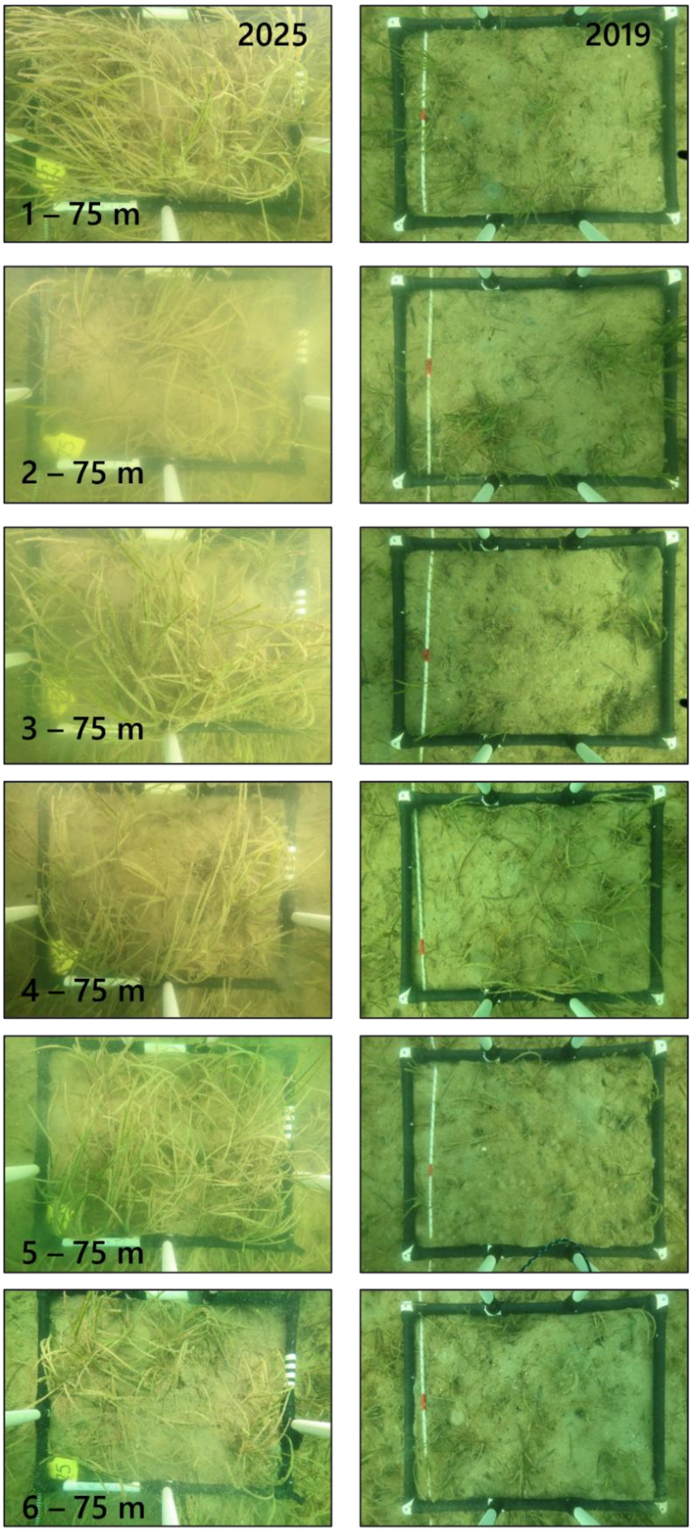


Figure A6.4. Photo-quadrat series of seagrass at 75 m on six transects at Huruhi Harbour, Great Mercury Island / Ahuahu, from 2025 (left column of images) and 2019 (right column of images). Images relate to subtidal transects 1–6, from top to bottom.

Appendix 7. Raw data from South Bay and Huruhi Harbour 2025 subtidal seagrass surveys

Meadow	Transect	Distance (m)	Depth (m, MSL)	Seagrass cover (rank 1–5)	Mean canopy height (mm)	Biomass (gDW m^{-2})	Visual biomass (rank 1-5)	Macroalgal cover (rank 1-5)	Mean epiphyte cover (rank 1–5)	Mean fungal wasting disease severity (rank 1–5)	Fungal wasting disease prevalence (%)
South Bay	T7	5	4.03	5				0			
South Bay	T7	10	3.93	2	102			0	1	0	0
South Bay	T7	15	4.03	3				0			
South Bay	T7	20	3.93	5	289.5			0	2.2	0.8	40
South Bay	T7	25	4.13	3		138.7	5	2			
South Bay	T7	30	4.13	5	316			0	3.2	0.6	20
South Bay	T7	35	4.13	5				0			
South Bay	T7	40	4.13	5	275			0	2.8	0.7	30
South Bay	T7	45	4.03	4				0			
South Bay	T7	50	4.03	4	209			0	2.6	0.1	10
South Bay	T7	55	4.03	5				0			
South Bay	T7	60	3.93	5	264.5			0	3.1	0.7	20
South Bay	T7	65	3.83	5				0			
South Bay	T7	70	3.93	5	285			0	3.3	0.4	10
South Bay	T7	75	3.93	5		73.3	5	0			
South Bay	T7	80	3.83	5	269			0	2.3	0	0
South Bay	T7	85	3.83	4				0			
South Bay	T7	90	3.73	4	248			0	2.7	0.4	10
South Bay	T7	95	3.73	4				0			
South Bay	T7	100	3.73	4	297.5			0	2.7	0.8	40
South Bay	T6	5	7.03	5				0			
South Bay	T6	10	7.03	5	221.5			0	2.1	1	30
South Bay	T6	15	6.93	5				0			

Meadow	Transect	Distance (m)	Depth (m, MSL)	Seagrass cover (rank 1–5)	Mean canopy height (mm)	Biomass (gDW m^{-2})	Visual biomass (rank 1-5)	Macroalgal cover (rank 1-5)	Mean epiphyte cover (rank 1–5)	Mean fungal wasting disease severity (rank 1–5)	Fungal wasting disease prevalence (%)
South Bay	T6	20	6.93	5	159			0	2.7	1.4	60
South Bay	T6	25	6.83	3		59.1	4	0			
South Bay	T6	30	6.93	5	212			0	2.4	1.3	70
South Bay	T6	35	6.73	5				0			
South Bay	T6	40	6.63	5	241			0	2.2	0.7	20
South Bay	T6	45	6.53	5				0			
South Bay	T6	50	6.63	2	123			0	2.3	1.3	50
South Bay	T6	55	6.43	5				0			
South Bay	T6	60	6.53	1	151			0	2.8	1.4	60
South Bay	T6	65	6.53	3				0			
South Bay	T6	70	6.43	5	208			0	2.1	2.6	100
South Bay	T6	75	6.43	2		23.6	2	0			
South Bay	T6	80	6.23	3	156.5			0	2	1.9	60
South Bay	T6	85	6.13	2				0			
South Bay	T6	90	6.13	2	139.5			0	2.5	1.8	90
South Bay	T6	95	6.23	3				0			
South Bay	T6	100	6.03	5	255			0	1.1	1.4	80
South Bay	T5	5	6.06	3				0			
South Bay	T5	10	6.06	1	82			0	2.2	0.7	40
South Bay	T5	15	6.06	0				0			
South Bay	T5	20	6.06	0	NA			0	NA	NA	NA
South Bay	T5	25	6.06	0				0			
South Bay	T5	30	6.06	0	NA			0	NA	NA	NA
South Bay	T5	35	6.16	0				0			
South Bay	T5	40	6.26	0	127			0	2.4	0.6	30

Meadow	Transect	Distance (m)	Depth (m, MSL)	Seagrass cover (rank 1–5)	Mean canopy height (mm)	Biomass (gDW m^{-2})	Visual biomass (rank 1-5)	Macroalgal cover (rank 1-5)	Mean epiphyte cover (rank 1–5)	Mean fungal wasting disease severity (rank 1–5)	Fungal wasting disease prevalence (%)
South Bay	T5	45	6.36	0		28	3	0			
South Bay	T5	50	6.46	0	NA			0	NA	NA	NA
South Bay	T5	55	6.26	0				0			
South Bay	T5	60	6.26	0	NA			0	NA	NA	NA
South Bay	T5	65	6.46	0				0			
South Bay	T5	70	6.36	0	NA			0	NA	NA	NA
South Bay	T5	75	6.46	0				0			
South Bay	T5	80	6.46	0	NA			0	NA	NA	NA
South Bay	T5	85	6.46	0				0			
South Bay	T5	90	6.46	0	NA			0	NA	NA	NA
South Bay	T5	95	6.36	0				0			
South Bay	T5	100	6.66	0	124.5	23.1		0	2.5	2.4	100
South Bay	T4	5	5.88	4				0			
South Bay	T4	10	5.78	3	152.5			0	1.3	1.3	60
South Bay	T4	15	5.88	1				0			
South Bay	T4	20	5.78	5	324			0	1.3	0.8	60
South Bay	T4	25	5.78	4		82.2	4	0			
South Bay	T4	30	5.88	5	263.5			0	1.9	0.8	50
South Bay	T4	35	5.78	3				0			
South Bay	T4	40	5.88	3	265.5			0	1.8	1.3	60
South Bay	T4	45	5.78	3				0			
South Bay	T4	50	5.78	5	282.5			0	1.7	1	50
South Bay	T4	55	5.78	4				0			
South Bay	T4	60	5.78	5	318			0	1.5	1.2	70
South Bay	T4	65	5.78	5				0			

Meadow	Transect	Distance (m)	Depth (m, MSL)	Seagrass cover (rank 1–5)	Mean canopy height (mm)	Biomass (gDW m^{-2})	Visual biomass (rank 1-5)	Macroalgal cover (rank 1-5)	Mean epiphyte cover (rank 1–5)	Mean fungal wasting disease severity (rank 1–5)	Fungal wasting disease prevalence (%)
South Bay	T4	70	5.68	4	192.5			0	1.4	1.1	50
South Bay	T4	75	5.88	4		86.2	4	0			
South Bay	T4	80	5.98	3	188.5			0	1.3	0.5	30
South Bay	T4	85	5.98	4				0			
South Bay	T4	90	5.98	4	184.5			0	1.4	0.9	30
South Bay	T4	95	5.78	3				0			
South Bay	T4	100	5.18	5	246			0	1.8	0.7	40
South Bay	T3	5	4.98	1				0			
South Bay	T3	10	4.98	5	215.5			0	1.3	0.5	30
South Bay	T3	15	5.18	5				0			
South Bay	T3	20	5.08	4	122			0	1.9	0.3	10
South Bay	T3	25	5.18	3		25.3	3	0			
South Bay	T3	30	5.18	4	237			0	2.7	1.2	60
South Bay	T3	35	5.18	5				0			
South Bay	T3	40	5.28	5	248.5			0	1.8	0.8	30
South Bay	T3	45	5.18	4				0			
South Bay	T3	50	5.38	5	222.5			0	2.1	0.4	20
South Bay	T3	55	5.18	5				0			
South Bay	T3	60	5.38	5	205			0	2	0.9	30
South Bay	T3	65	5.48	5				0			
South Bay	T3	70	5.28	5	294			0	2	1	50
South Bay	T3	75	5.28	4		123.6	5	0			
South Bay	T3	80	5.28	4	181			0	2	0.9	50
South Bay	T3	85	5.18	5				0			
South Bay	T3	90	5.38	4	243			0	1.9	1.2	50

Meadow	Transect	Distance (m)	Depth (m, MSL)	Seagrass cover (rank 1–5)	Mean canopy height (mm)	Biomass (gDW m^{-2})	Visual biomass (rank 1-5)	Macroalgal cover (rank 1-5)	Mean epiphyte cover (rank 1–5)	Mean fungal wasting disease severity (rank 1–5)	Fungal wasting disease prevalence (%)
South Bay	T3	95	5.28	2				2			
South Bay	T3	100	5.38	3	179			0	1.8	0.6	40
South Bay	T2	5	2.32	5				0			
South Bay	T2	10	2.22	5	266.5			0	1	1	50
South Bay	T2	15	2.42	5				0			
South Bay	T2	20	2.22	5	252.5			0	1.2	1.3	40
South Bay	T2	25	2.22	5		211.6	5	0			
South Bay	T2	30	2.22	5	310.5			0	1.2	1.1	40
South Bay	T2	35	2.32	5				0			
South Bay	T2	40	2.42	5	263			0	1.6	0.9	30
South Bay	T2	45	2.22	5				0			
South Bay	T2	50	2.22	5	231.5			0	1.3	0.9	40
South Bay	T2	55	2.32	5				0			
South Bay	T2	60	2.32	5	279.5			0	1.8	1.7	70
South Bay	T2	65	2.22	5				0			
South Bay	T2	70	2.12	5	294			0	3.1	0.6	30
South Bay	T2	75	2.22	5		304.4	5	0			
South Bay	T2	80	2.12	5	324			0	2.2	0.7	30
South Bay	T2	85	2.22	5				0			
South Bay	T2	90	2.12	5	297			0	2	0.3	20
South Bay	T2	95	1.92	5				0			
South Bay	T2	100	1.82	5	222.5			0	1.7	0.7	30
South Bay	T1	5	3.22	5				0			
South Bay	T1	10	3.12	5	297			0	1.6	0.3	10
South Bay	T1	15	2.92	4				0			

Meadow	Transect	Distance (m)	Depth (m, MSL)	Seagrass cover (rank 1–5)	Mean canopy height (mm)	Biomass (gDW m^{-2})	Visual biomass (rank 1-5)	Macroalgal cover (rank 1-5)	Mean epiphyte cover (rank 1–5)	Mean fungal wasting disease severity (rank 1–5)	Fungal wasting disease prevalence (%)
South Bay	T1	20	2.82	5	203			0	1.5	0.1	10
South Bay	T1	25	2.82	5		178.2	5	0			
South Bay	T1	30	2.82	5	317			0	2.6	0.3	20
South Bay	T1	35	2.62	5				0			
South Bay	T1	40	2.62	5	312			0	2.8	0.1	10
South Bay	T1	45	2.62	5				0			
South Bay	T1	50	2.62	1	196			0	1.3	0.4	20
South Bay	T1	55	2.52	5				0			
South Bay	T1	60	2.52	5	287			0	1.2	1.1	60
South Bay	T1	65	2.42	5				0			
South Bay	T1	70	2.52	5	250.5			0	1.1	0.2	20
South Bay	T1	75	2.52	5		208.4	5	0			
South Bay	T1	80	2.52	5	306			0	1.5	0.6	40
South Bay	T1	85	2.52	5				0			
South Bay	T1	90	2.42	5	312.5			0	1.5	1.4	50
South Bay	T1	95	2.42	5				0			
South Bay	T1	100	2.42	5	305			0	1.3	0.7	30
Huruhi	T1	5	0.77	4				0			
Huruhi	T1	10	0.87	4	180.5			0	3.6	1.8	90
Huruhi	T1	15	0.87	5				0			
Huruhi	T1	20	1.07	5	257.5			0	4.1	1.6	70
Huruhi	T1	25	1.27	5		80.4	4	0			
Huruhi	T1	30	1.47	5	269.5			0	2.8	1.9	90
Huruhi	T1	35	1.87	5				0			
Huruhi	T1	40	1.87	4	293.5			0	3.5	1.8	80

Meadow	Transect	Distance (m)	Depth (m, MSL)	Seagrass cover (rank 1–5)	Mean canopy height (mm)	Biomass (gDW m^{-2})	Visual biomass (rank 1-5)	Macroalgal cover (rank 1-5)	Mean epiphyte cover (rank 1–5)	Mean fungal wasting disease severity (rank 1–5)	Fungal wasting disease prevalence (%)
Huruhi	T1	45	1.97	5				0			
Huruhi	T1	50	1.77	5	273			0	3.7	2.5	80
Huruhi	T1	55	1.67	4				0			
Huruhi	T1	60	1.67	1	170.5			0	4.3	1.9	80
Huruhi	T1	65	1.67	3				0			
Huruhi	T1	70	1.57	5	290.5			0	3.4	2.2	80
Huruhi	T1	75	1.67	5		62.2	5	0			
Huruhi	T1	80	1.67	4	296			0	3.5	1.7	60
Huruhi	T1	85	1.77	5				0			
Huruhi	T1	90	1.77	5	319			0	2.8	2.2	80
Huruhi	T1	95	1.77	5				0			
Huruhi	T1	100	1.77	5	316.5			0	3.3	2.1	90
Huruhi	T2	5	0.58	5				0			
Huruhi	T2	10	0.68	5	259			0	4.4	1.7	70
Huruhi	T2	15	0.78	3				0			
Huruhi	T2	20	0.78	5	265			0	3.6	1.7	90
Huruhi	T2	25	0.98	3		33.3	4	0			
Huruhi	T2	30	1.28	3	224			0	4.8	2.3	90
Huruhi	T2	35	1.28	1				0			
Huruhi	T2	40	1.28	4	295			0	4.5	1.9	80
Huruhi	T2	45	1.28	5				0			
Huruhi	T2	50	1.38	5	278.5			0	3.4	1.9	90
Huruhi	T2	55	1.38	3				0			
Huruhi	T2	60	1.38	3	280.5			0	2.7	1.6	70
Huruhi	T2	65	1.48	1				0			

Meadow	Transect	Distance (m)	Depth (m, MSL)	Seagrass cover (rank 1–5)	Mean canopy height (mm)	Biomass (gDW m^{-2})	Visual biomass (rank 1-5)	Macroalgal cover (rank 1-5)	Mean epiphyte cover (rank 1–5)	Mean fungal wasting disease severity (rank 1–5)	Fungal wasting disease prevalence (%)
Huruhi	T2	70	1.48	4	296.5			0	3.6	2.4	100
Huruhi	T2	75	1.58	3		38.2	3	0			
Huruhi	T2	80	1.58	5	316.5			0	4.6	3.6	100
Huruhi	T2	85	1.58	4				0			
Huruhi	T2	90	1.68	5	290			0	4.3	2.3	90
Huruhi	T2	95	1.48	5				0			
Huruhi	T2	100	1.68	3	252.5			0	4.3	2.1	100
Huruhi	T3	5	1.65	3				0			
Huruhi	T3	10	1.85	2	240			0	4.5	2.1	100
Huruhi	T3	15	1.85	3				0			
Huruhi	T3	20	1.95	3	254.5			0	3.6	1.5	70
Huruhi	T3	25	2.05	4		60.9	5	0			
Huruhi	T3	30	2.05	3	275.5			0	4.6	1.9	90
Huruhi	T3	35	2.05	5				0			
Huruhi	T3	40	2.05	5	300			0	4	1.4	60
Huruhi	T3	45	2.05	4				0			
Huruhi	T3	50	2.15	3	245			0	4.6	2.4	90
Huruhi	T3	55	1.95	5				0			
Huruhi	T3	60	2.25	3	286.5			0	4.2	2.4	100
Huruhi	T3	65	2.35	4				0			
Huruhi	T3	70	2.35	5	268.5			0	4.2	2.2	100
Huruhi	T3	75	2.35	5		65.8	5	0			
Huruhi	T3	80	2.55	4	316.5			0	3.4	1.1	70
Huruhi	T3	85	2.55	5				0			
Huruhi	T3	90	2.65	2	233.5			0	3.2	1.7	70

Meadow	Transect	Distance (m)	Depth (m, MSL)	Seagrass cover (rank 1–5)	Mean canopy height (mm)	Biomass (gDW m^{-2})	Visual biomass (rank 1-5)	Macroalgal cover (rank 1-5)	Mean epiphyte cover (rank 1–5)	Mean fungal wasting disease severity (rank 1–5)	Fungal wasting disease prevalence (%)
Huruhi	T3	95	2.65	5				0			
Huruhi	T3	100	2.85	4	282.5			0	4	2.2	80
Huruhi	T4	5	1.87	3				0			
Huruhi	T4	10	1.87	4	190.5			0	2.1	2.4	100
Huruhi	T4	15	1.87	3				0			
Huruhi	T4	20	1.87	3	136			0	2.3	1.7	90
Huruhi	T4	25	1.87	4		40.9	4	0			
Huruhi	T4	30	1.97	4	199.5			0	3.6	2.8	100
Huruhi	T4	35	1.87	4				0			
Huruhi	T4	40	1.97	3	222.5			0	2.8	1.3	80
Huruhi	T4	45	1.97	5				0			
Huruhi	T4	50	1.97	5	229.5			0	3.6	2.1	100
Huruhi	T4	55	1.97	5				0			
Huruhi	T4	60	2.17	3	268.5			0	3.5	1.5	70
Huruhi	T4	65	2.27	3				0			
Huruhi	T4	70	2.47	3	246.5			0	4	1.5	80
Huruhi	T4	75	2.47	3		22.7	3	0			
Huruhi	T4	80	2.57	4	283.5			0	3.9	1.7	100
Huruhi	T4	85	2.67	4				0			
Huruhi	T4	90	2.87	5	240.5			0	4.7	1.6	90
Huruhi	T4	95	3.07	5				0			
Huruhi	T4	100	3.47	5	271			0	4.1	1.3	90
Huruhi	T5	5	3.02	4				0			
Huruhi	T5	10	3.12	3	248.5			0	2.2	1.8	90
Huruhi	T5	15	3.02	5				0			

Meadow	Transect	Distance (m)	Depth (m, MSL)	Seagrass cover (rank 1–5)	Mean canopy height (mm)	Biomass (gDW m^{-2})	Visual biomass (rank 1-5)	Macroalgal cover (rank 1-5)	Mean epiphyte cover (rank 1–5)	Mean fungal wasting disease severity (rank 1–5)	Fungal wasting disease prevalence (%)
Huruhi	T5	20	3.02	4	306.5			0	3.2	1.6	80
Huruhi	T5	25	3.12	4		79.6	4	0			
Huruhi	T5	30	3.22	3	245			0	3.9	2.1	90
Huruhi	T5	35	3.32	4				0			
Huruhi	T5	40	3.32	4	228.5			0	2.8	1.6	60
Huruhi	T5	45	3.62	5				0			
Huruhi	T5	50	3.72	4	228.5			0	4.9	1.4	70
Huruhi	T5	55	3.72	5				0			
Huruhi	T5	60	4.02	5	270.5			0	4	0.9	60
Huruhi	T5	65	4.12	5				0			
Huruhi	T5	70	4.32	5	306.5			0	4.3	0.1	10
Huruhi	T5	75	4.92	4		64.4	4	0			
Huruhi	T5	80	5.12	2	183			0	4.4	0.3	10
Huruhi	T5	85	5.12	2				0			
Huruhi	T5	90	5.42	0	224			0	5	1.5	70
Huruhi	T5	95	5.52	0				0			
Huruhi	T5	100	5.92	0	95.5			0	3.4	0.6	40
Huruhi	T6	5	4.02	4				0			
Huruhi	T6	10	4.12	2	223			0	2.5	1.4	60
Huruhi	T6	15	4.12	4				0			
Huruhi	T6	20	4.02	4	214.5			0	2.2	1.1	80
Huruhi	T6	25	4.12	4		42.7	4	0			
Huruhi	T6	30	4.12	5	260.5			0	2	2.1	80
Huruhi	T6	35	4.12	5				0			
Huruhi	T6	40	4.22	4	157.5			0	3.2	1.4	80

Meadow	Transect	Distance (m)	Depth (m, MSL)	Seagrass cover (rank 1–5)	Mean canopy height (mm)	Biomass (gDW m^{-2})	Visual biomass (rank 1-5)	Macroalgal cover (rank 1-5)	Mean epiphyte cover (rank 1–5)	Mean fungal wasting disease severity (rank 1–5)	Fungal wasting disease prevalence (%)
Huruhi	T6	45	4.22	5				0			
Huruhi	T6	50	4.32	4	273			1	4	1.8	80
Huruhi	T6	55	4.42	4				0			
Huruhi	T6	60	4.52	3	219.5			0	3.8	1.3	50
Huruhi	T6	65	4.62	4				0			
Huruhi	T6	70	4.82	4	282			0	3.8	1.5	70
Huruhi	T6	75	4.92	3		43.1	3	0			
Huruhi	T6	80	5.02	4	297.5			0	4	0.6	30
Huruhi	T6	85	5.02	4				0			
Huruhi	T6	90	5.12	5	306.5			0	3.9	1.2	70
Huruhi	T6	95	5.12	5				0			
Huruhi	T6	100	5.12	5	252			0	4.2	1.7	70

6. Acknowledgements

Thank you to Jamie Wilson and Bay Boat Company for vessel and skipper assistance. Thanks to Paula Casanovas for helping with interpretation of data and statistical analysis, Allison Brownlee for assistance with mapping and figure creation, and Nick Hearn for technical support. Thanks to Anna Berthelsen and Louisa Fisher for reviewing this report. Thanks also to Mike Townsend from WRC who was our main point of contact.

7. References

Armiger LC. 1964. An occurrence of *Labyrinthula* in New Zealand *Zostera*. New Zealand Journal of Botany 2(1):3–9.

Armiger LC. 1965. A contribution to the autecology of *Zostera* [master's thesis]. Auckland: University of Auckland.

Bennett H, Smeaton M, McGrath E, Newcombe E, Floerl L, Major R, Casanovas P. 2022. Assessment of seabed effects associated with farming salmon offshore of northern Stewart Island / Rakiura. Nelson: Cawthron Institute. Cawthron Report No. 3315B. Prepared for Ngai Tahu Seafood Resources.

Berthelsen A, Clement D, Gillespie P. 2016. Shakespeare Bay estuary monitoring 2016. Nelson: Cawthron Institute. Cawthron Report No. 2833. Prepared for Marlborough District Council.

Berthelsen A, Crossett D, Ho M, Brettle E, Clark D. 2024. A blueprint for seed-based seagrass restoration in Aotearoa New Zealand. Nelson: Cawthron Institute. Cawthron Report 4011. Prepared for Westpac NZ Government Innovation Fund.

Braun-Blanquet J. 1932. Plant sociology: the study of plant communities. Translated and edited by GD Fuller, HS Conard. London: McGraw-Hill.

Burdick DM, Short FT, Wolf J. 1993. An index to assess and monitor the progression of wasting disease in eelgrass *Zostera marina*. Marine Ecology Progress Series. 94:83–90.

Buscombe D. 2017. Shallow water benthic imaging and substrate characterization using recreational-grade sidescan-sonar. Environmental Modelling & Software. 89:1–18.

Clark D, Berthelsen A. 2021. Review of the potential for low impact seagrass restoration in Aotearoa New Zealand. Nelson: Cawthron Institute. Cawthron Report No. 3697. Prepared for Nelson City Council.

Clark D, Crossett D. 2019. Subtidal seagrass surveys at Slipper and Great Mercury Islands. Nelson: Cawthron Institute. Cawthron Report No. 3347. Prepared for Waikato Regional Council.

Costanza R, d'Arge R, de Groot R, Farber S, Grasso M, Hannon B, Limburg K, Naeem S, O'Neill RV, Paruelo J, et al. 1997. The value of the world's ecosystem services and natural capital. Nature. 387:253–260.

de Lange PJ, Gosden J, Courtney SP, Fergus AJ, Barkla JW, Beadel SM, Champion PD, Hindmarsh-Walls R, Makan T, Michel P. 2024. Conservation status of vascular plants in Aotearoa New Zealand, 2023. Wellington: Department of Conservation. New Zealand Threat Classification Series 43.

Duarte CM, Chiscano CL. 1999. Seagrass biomass and production: a reassessment. Aquatic Botany. 65:159–174.

Duarte CM, Kirkman H. 2001. Methods for the measurement of seagrass abundance and depth distribution. In: Short FT, Coles RG, editors. Global seagrass research methods. Amsterdam: Elsevier. p 141–153.

Durako M, Kuss KM. 1994. Effects of *Labyrinthula* infection on the photosynthetic capacity of *Thalassia testudinum*. Bulletin of Marine Science. 54(3):727–732.

Floerl O, Atalah J, Bugnot AB, Chandler M, Dafforn KA, Floerl L, Zaiko A, Major R. 2021. A global model to forecast coastal hardening and mitigate associated socioecological risks. Nature Sustainability, 4(12):1060–1067.

Fonseca MS. 1996. The role of seagrasses in nearshore sedimentary processes: a review. In: Nordstrom KF, Roman CT, editors. Estuarine shores: evolution, environmental and human alterations. New York: John Wiley. p. 261–286.

Gillespie P, Clement D, Asher R. 2012a. Baseline mapping of selected intertidal habitats within Grove Arm, Queen Charlotte Sound. Nelson: Cawthron Institute. Cawthron Report No. 2133. Prepared for Marlborough District Council.

Gillespie P, Clement D, Clark DE, Asher R. 2012c. Nelson Haven fine-scale benthic baseline 2012. Nelson: Cawthron Institute. Cawthron Report No. 2209. Prepared for Nelson City Council.

Gillespie P, Forrest R, Clark DE, Asher R. 2012b. Coastal effects of the Nelson (Bell Island) regional sewerage discharge: benthic monitoring survey 2011. Nelson: Cawthron Institute. Cawthron Report No. 2068. Prepared for Nelson Regional Sewerage Business Unit.

Grace RV, Whitten RF. 1974. Benthic communities west of Slipper Island, north-eastern New Zealand. Tane 20:5–20.

Grace RV, Grace AB. 1976. Benthic communities west of Great Mercury Island, north-eastern New Zealand. Tane. 22:85–101.

Greene A, Rahman AF, Kline R, Rahman MS. 2018. Side scan sonar: a cost-efficient alternative method for measuring seagrass cover in shallow environments. Estuarine, Coastal and Shelf Science. 207:250–258.

Hailes SF. 2006. Contribution of seagrass (*Zostera muelleri*) to estuarine food webs revealed by carbon and nitrogen stable isotope analysis [master's thesis]. Hamilton: University of Waikato.

Heiss WM, Smith AM, Probert PK. 2000. Influence of the small intertidal seagrass *Zostera novazelandica* on linear water flow and sediment texture. New Zealand Journal of Marine and Freshwater Research. 34(4):689–694.

Henriques PR. 1980. Faunal community structure of eight soft shore, intertidal habitats in the Manukau Harbour. New Zealand Journal of Ecology. 3:97–103.

Hughes RG, Potouroglou M, Ziauddin Z, Nicholls JC. 2018. Seagrass wasting disease: nitrate enrichment and exposure to a herbicide (Diuron) increases susceptibility of *Zostera marina* to infection. Marine Pollution Bulletin. 134:94–98.

Inglis GJ. 2003. The seagrasses of New Zealand. In: Green EP, Short FT, editors. World atlas of seagrasses. Berkley (CA): University of California Press. p. 148–157.

Ismail N. 2001. Ecology of eelgrass, *Zostera novazelandica* Setchell, in Otago Harbour, Dunedin, New Zealand [PhD thesis]. Dunedin: University of Otago.

Kirkman H. 1996. Baseline and monitoring methods for seagrass meadows. Journal of Environmental Management. 47:191–201.

Lenth R. 2024. emmeans: estimated marginal means, aka least-squares means. R package, version 1.10.1. https://CRAN.R-project.org/package=emmeans

Matheson F, Dos Santos V, Inglis G, Pilditch C, Reed J, Morrison M, Lundquist C, Van Houte-Howes K, Hailes S, Hewitt J. 2009. New Zealand seagrass general information guide. Hamilton: NIWA. NIWA Information Series No. 72.

McKenzie LJ, Finkbeiner MA, Kirkamn H. 2001. Methods for mapping seagrass distribution. In: Short FT, Coles RG, editors. Global seagrass research methods. Amsterdam: Elsevier. p. 101–121.

Meese RJ, Tomich PA. 1992. Dots on rocks: a comparison of percentage cover methods. Journal of Experimental Marine Biology and Ecology. 165:59–73.

Mellors JE. 1991. An evaluation of a rapid visual technique for estimating seagrass biomass. Aquatic Botany. 42(1):67–73.

Morrison MA, Elliot S, Hughes A, Kainamu A, Williams E, Lowe M, Lohrer D, Needham H, Semadeni-Davies A. 2023. Land-based effects on coastal fisheries and kaimoana and their habitats – a review. Wellington: Ministry for Primary Industries. New Zealand Aquatic Environment and Biodiversity Report No. 309.

Morrison M, Francis M. 2001. In search of small snapper. Seafood New Zealand. 9:21–22.

Morrison MA, Jones EG, Consalvey M, Berkenbusch K. 2014a. Linking marine fisheries species to biogenic habitats in New Zealand: a review and synthesis of knowledge. Wellington: Ministry for Primary Industries.

Morrison MA, Lowe ML, Grant CM, Smith PJ, Carbines GD, Reed J, Bury SJ, Brown J. 2014b. Seagrass meadows as biodiversity and productivity hotspots. Wellington: Ministry for Primary Industries. New Zealand Aquatic Environment and Biodiversity Report No. 137.

NIWA 2025. Tide forecaster. [accessed 1 May 2025]. Wellington: NIWA. https://niwa.co.nz/coasts/tide-forecaster

Olesen B, Sand-Jensen K. 1994. Patch dynamics of eelgrass *Zostera marina*. Marine Ecology Progress Series. 106:147–156.

Park SG. 1999. Changes in abundance of seagrass (*Zostera* spp.) in Tauranga Harbour from 1959–1996. Whakatane: Environment Bay of Plenty. Environmental Report 99/30.

Pellikaan GC, Nienhuis PH. 1988. Nutrient uptake and release during growth and decomposition of eelgrass, *Zostera marina* L., and its effects on the nutrient dynamics of Lake Grevelingen. Aquatic Botany. 30(3):189–214.

R Core Team 2019. R: a language and environment for statistical computing. Vienna: R Foundation for Statistical Computing.

Ralph PJ, Short FT. 2002. Impact of the wasting disease pathogen, *Labyrinthula zosterae*, on the photobiology of eelgrass *Zostera marina*. Marine Ecology Progress Series. 226:265–271.

Ramage DL, Schiel DR. 1999. Patch dynamics and response to disturbance of the seagrass *Zostera novazelandica* on intertidal platforms in southern New Zealand. Marine Ecology Progress Series. 189:275–288.

Schattschneider J, Floerl L. 2022. Evaluation of suitability of satellite and aerial imagery as data sources for dune vegetation classification, mapping, and monitoring. Cawthron Advice Letter 2244 dated 14 June 2022. Prepared for Hawke's Bays Regional Council.

Schwarz A-M, Morrison M, Hawes I, Halliday J. 2006. Physical and biological characteristics of a rare marine habitat: sub-tidal seagrass beds of offshore islands. Wellington: Department of Conservation. Science for Conservation 269.

Shanahan B, Crawshaw J, Squires K, McElroy T, Griffiths R, Wade O. 2023. Guidance on council seagrass monitoring. Coastal Special Interest Group (CSIG) internal report. [accessed 11 April 2024]. https://www.envirolink.govt.nz/assets/Envirolink/2322-HBRC268-Guidance-on-council-seagrass-monitoring.pdf

Shao Z, Bryan KR, Lehmann MK, Flowers GJL, Pilditch CA. 2024. Scaling up benthic primary productivity estimates in a large intertidal estuary using remote sensing. Science of The Total Environment. 906:167389.

Short FT. 1987. Effects of sediment nutrients on seagrasses: literature review and mesocosm experiment. Aquatic Botany. 27(1):41–57.

Short FT, Wyllie-Echeverria S. 1996. Natural and human-induced disturbance of seagrasses. Environmental Conservation. 23(1):17–27.

Spalding M, Taylor M, Ravilious C, Short FT, Green E. 2003. The distribution and status of seagrasses. In: Green EP, Short FT, editors. World atlas of seagrasses. Berkley (CA): University of California Press. p. 5–26.

Stewart AL. 2015. 148.10 *Stigmatopora nigra* Kaup, 1856. In: Roberts CD, Stewart AL, Struthers CD, editors. The fishes of New Zealand. Vol. 3. Wellington: Te Papa Press. p. 1062.

Šunde C, Berthelsen A, Sinner J, Gillespie PA, Stringer K, Floerl L. 2017. Impacts of vehicle access at Delaware (Wakapuaka) Inlet. Nelson: Cawthron Institute. Cawthron Report No. 3015. Prepared for Nelson City Council.

Sullivan BK, Short FT. 2023. Taxomonic revisions of Zosteraceae (*Zostera, Nanozostera, Heterozostera* and *Phyllospadix*). Aquatic Botany. 187:103636.

Tang KHD, Hadibarata T. 2022. Seagrass meadows under the changing climate: a review of the impacts of climate stressors. Research in Ecology. 4(1):27–36.

Turner SJ, Hewitt JE, Wilkinson MR, Morrisey DJ, Thrush SF, Cummings VJ, Funnell G. 1999. Seagrass patches and landscapes: the influence of wind-wave dynamics and hierarchical arrangements of spatial structure on macrofaunal seagrass communities. Estuaries 22(4):1016–1032.

Turner SJ, Schwarz A-M. 2006a. Management and conservation of seagrass in New Zealand: an introduction. Wellington: Department of Conservation. Science for Conservation 264.

Turner SJ, Schwarz A-M. 2006b. Biomass development and photosynthetic potential of intertidal *Zostera capricorni* in New Zealand estuaries. Aquatic Botany. 85(1):53–64.

Unsworth RKF, McKenzie LJ, Collier CJ, Cullen-Unsworth LC, Duarte CM, Eklöf JS, Jarvis JC, Jones BL, Nordlund LM. 2019. Global challenges for seagrass conservation. Ambio. 48(8):801–815.

van Houte-Howes KSS, Turner SJ, Pilditch CA. 2004. Spatial differences in macroinvertebrate communities in intertidal seagrass habitats and unvegetated sediment in three New Zealand estuaries. Estuaries. 27(6):945–957.

Woods CMC, Schiel DR. 1997. Use of seagrass *Zostera novazelandica* (Setchell, 1933) as habitat and food by the crab *Macrophthalmus hirtipes* (Heller, 1862) (Brachyura: Ocypodidae) on rocky intertidal platforms in southern New Zealand. Journal of Experimental Marine Biology and Ecology. 214:49–65.

Zabarte-Maeztu I, Matheson FE, Manley-Harris M, Davies-Colley RJ, Hawes I. 2021. Fine sediment effects on seagrasses: a global review, quantitative synthesis and multi-stressor model. Marine Environmental Research. 171:105480.

World-class science for a better future.

CAWTHRON

98 Halifax Street East, Nelson 7010 | Private Bag 2, Nelson 7042 | New Zealand | Ph. +64 3 548 2319

cawthron.org.nz