SALT ECOLOGY

Broad Scale Temporal Changes in Seagrass Extent, Whanganui (Westhaven) Inlet, 1948-2021

Prepared for

Tasman District Council

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Cover photo: Whanganui/Westhaven Inlet looking toward the estuary entrance, March 2021, showing dense seagrass beds next to soft muds.

RECOMMENDED CITATION

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August 2022

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For the environment
Mō te taiao

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SUMMARY

As part of its State of the Environment programme, Tasman District Council (TDC) monitors the ecological condition of significant estuaries in its region. This report describes a broad scale survey of seagrass (*Zostera muelleri*) extent conducted on 26 March 2021 in Whanganui/Westhaven Inlet (hereafter referred to as Whanganui Inlet) and compares results to seagrass extent determined from ~1948 and 2013 aerial photographs, and from previous ground-truthed mapping surveys undertaken in 1990 and 2016.

The approach is based on the methods described in the National Estuary Monitoring Protocol (NEMP) and comprises mapping and classifying intertidal seagrass and underlying substrate using aerial photography, detailed ground-truthing, and digital mapping using Geographic Information System (GIS) technology.

KEY FINDINGS

In March 2021, seagrass was extensive (229ha), comprising 12% of the 1954ha intertidal area, with high cover (>50%) extending across 182.8ha (9.4%) of the intertidal flats. The largest seagrass beds were on the southern side of the main drainage channel in the eastern arm of the inlet, often raised slightly higher than the surrounding unvegetated sediment. Most seagrass (78%) was growing in sandy substrate (<10% mud content), 18% was growing in muddy-sand (10-25% mud), and only 4% was growing in sediment with a mud content >25%.

Many parts of the estuary were found to have seagrass beds that had recently died or were in a very poor condition. Whereas healthy seagrass beds had dark green and luxuriant growth, degraded beds were stunted with a sparse cover of brown fronds (see photo).

Healthy green seagrass (foreground) and dead and decaying brown seagrass (background), March 2021

Changes in seagrass since 1948 are summarised in the adjacent table and show seagrass beds were relatively stable between 1948 and 2013, before undergoing a very rapid decline. Overall, 718ha of high cover (>50%) seagrass has been lost from the estuary since 1948, with most of the losses (531ha, 74%) occurring in the 8 years between 2013 and 2021. The significant loss of seagrass in the last decade likely represents one of largest recent losses of seagrass recorded in New Zealand.

Mapping Year	Ha	%	% loss since 1948
1948	901.6	46.1	-
1990	816.1	41.8	9.5
2013	713.9	36.5	20.8
2016	328.9	16.8	63.5
2021	182.8	9.4	79.7

Due to its relative isolation and low level of catchment development the recent seagrass losses do not appear to be caused by land use activities. Instead, it is postulated that the most likely trigger of losses is climate change, with intense marine heat waves, which are known to cause acute and dramatic die-offs of seagrass meadows, recorded in the summers of 2015/16, 2016/17 and 2018/19. Secondary impacts from the remobilisation of fine sediment following seagrass die-off are also likely. Regardless of the specific drivers of change, the loss of such a large area of high value habitat is of significant concern, particularly as it may signal that seagrass beds in other parts of the region, and New Zealand, are potentially vulnerable to rapid change.

RECOMMENDATIONS

- In response to the recent rapid and extensive seagrass losses it is recommended that mapping of seagrass continues at 5 yearly intervals to monitor ongoing change, with sediment plates measured annually.
- To investigate the likely cause of recent losses, it is recommended that TDC encourage detailed research as part of nationally funded science initiatives e.g., National Science Challenges, university-based research, Envirolink projects.

1. INTRODUCTION

Estuary monitoring is undertaken by most councils in New Zealand as part of their State of the Environment (SOE) programmes. Since 1999, Tasman District Council (TDC) has undertaken SOE monitoring of selected estuaries (Waimea, Moutere, Motueka-Ruiwaka, Motupipi, Ruataniwha and Whanganui/Westhaven) in the region, based on the methods outlined in New Zealand's National Estuary Monitoring Protocol (NEMP; Robertson et al. 2002a-c), or extensions of that approach. NEMP monitoring is primarily designed to detect and understand changes in estuaries over time and determine the effect of catchment influences, especially those contributing to the input of nutrients and muddy sediments. Excessive nutrient and fine sediment inputs are primary drivers of estuary eutrophication symptoms such as prolific macroalgal (seaweed) growth, and poor sediment condition.

The NEMP is intended to provide resource managers nationally with a scientifically defensible, cost-effective and standardised approach for monitoring the ecological status of estuaries in their region. Although it does not provide information on sediment or nutrient sources or other causes of particular condition, the results establish a benchmark of estuarine health in order to better understand human influences, and against which future comparisons can be made. The NEMP approach involves two main types of survey:

- Broad scale mapping of estuarine intertidal habitats. This type of monitoring is typically undertaken every 5 to 10 years.
- Fine scale monitoring of estuarine biota and sediment quality. This type of monitoring is typically conducted at intervals of 5 years after initially establishing a baseline.

Whanganui Inlet (Fig. 1) is of particular interest as it is surrounded by a largely undisturbed native forest catchment (Fig. 2) making it an important reference site for assessing the state of other estuaries in the region and important nationally.

The current report describes the methods and results of broad scale monitoring undertaken on 26 March 2021 to characterise the extent of seagrass (*Zostera muelleri*) in Whanganui Inlet. The survey was initiated following observations of a very large (385ha, 54%) reduction in seagrass extent between 2013 and 2016 (Stevens 2018).

Seagrass grows in soft sediments in most New Zealand estuaries and provides important ecosystem services such as enhanced primary production and nutrient cycling, stabilisation of sediments, increased biodiversity, sequesters carbon and provide nursery and feeding grounds for a range of invertebrates and fish. Although tolerant of a wide range of conditions, seagrass is seldom found above mean sea level (MSL) and is vulnerable to fine sediments in the water column (reducing light), sediment smothering (burial), excessive nutrients (mainly via secondary impacts from macroalgal smothering), and sediment quality (e.g., low oxygen). Any decline in seagrass extent is of concern due to the loss of the important ecosystem services provided. When a decline occurs in the absence of any obvious local catchment land use changes, as is the case in Whanganui Inlet, it suggests external drivers of change (e.g., marine heat waves, offshore sediment or nutrient supply) could be adversely affecting the estuary.

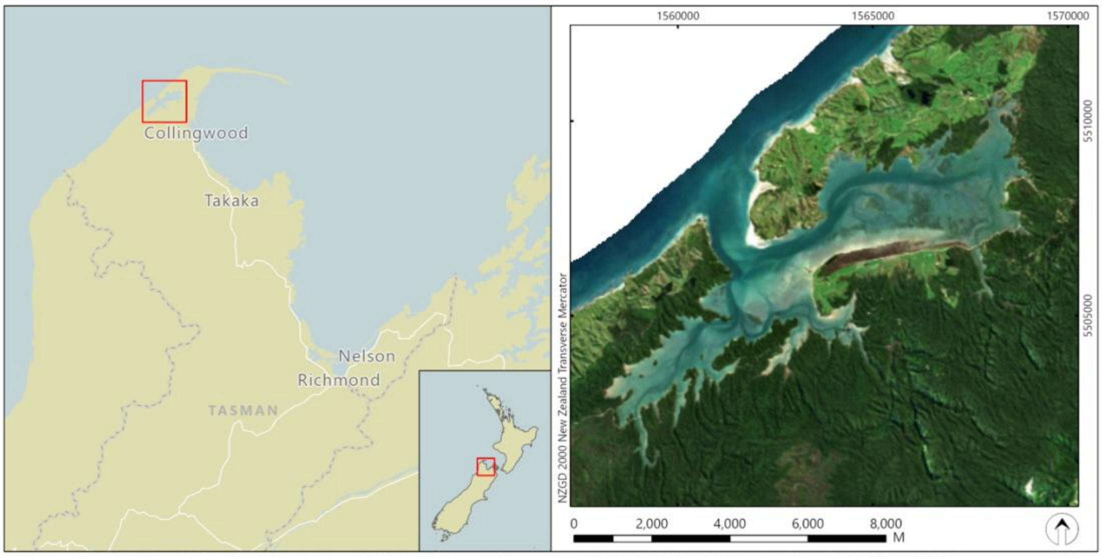

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Fig. 1. Location of Whanganui Inlet.

2. BACKGROUND TO WHANGANUI INLET

The following background information on Whanganui Inlet has been adapted from Stevens and Robertson 2017) and Stevens (2018) and updated as appropriate. Whanganui Inlet is located 19km southwest of Farewell Spit on the top of west coast of New Zealand's South Island (Fig. 1). It is a large (2741ha), relatively unmodified, shallow, well-flushed, seawater-dominated, tidal lagoon (SIDE) type estuary that is open to the sea via a single entrance. It is the third largest estuary of its type in the South Island.

It is fed by four main streams on the south and east sides, (Mangarakau Drain (mean flow $0.66m^3.s^{-1}$), Mangarakau Stream ($0.48m^3.s^{-1}$), Wairoa River ($0.16m^3.s^{-1}$), and Muddy Creek ($0.59m^3.s^{-1}$) - flow data from NIWA Coastal Explorer). Many smaller streams also enter the estuary. Several other water bodies (e.g., the Kaihoka Lakes and Lake Otuhie) are present in the immediate vicinity and increase the value of the estuary/freshwater complex for wildlife. Much of the estuary catchment landcover (see Fig. 2) is native forest (70%), scrub (12%) or herbaceous freshwater wetland or salt marsh (5%). Exotic forest (2%), high producing grassland (2%) and low producing grassland (6%) reflect the main human catchment land uses. The road along the southern and eastern estuary margins has resulted in numerous causeways restricting tidal flushing to many of the upper estuary arms.

Baseline broad scale mapping undertaken by the Department of Conservation (DOC) (Davidson 1990) classified the dominant intertidal estuary features as: seagrass (859ha), sandflats (826ha), mudflats (146ha), salt marsh (96ha), and cobble, gravel and rock fields (27ha). The subtidal zone comprised 769ha (28%) of the estuary area. At the time of the baseline mapping, there had been some historical loss of high value salt marsh habitat due to reclamation and drainage around margin areas (~60ha), with resulting shoreline modification (e.g., seawalls, bunds, roads) restricting the capacity of salt marsh to migrate inland in response to predicted sea level rise.

The estuary is valued for its aesthetic appeal, rich biodiversity, duck shooting, whitebaiting, fishing, boating, walking, and scientific interest. It is a dual protected area with a marine reserve in the southern third and a wildlife reserve over the remaining two-thirds. A Ramsar wetland of international importance application is pending on Whanganui Inlet, Mangarakau Swamp and Lake Otuhie. Ecologically, habitat diversity and condition are high. A significant portion of the intertidal salt marsh vegetation remains intact. The inlet has extensive seagrass beds, as well as dunes, cliffs, islands, rock platforms, underwater reefs, and a well-vegetated terrestrial margin dominated by coastal forest (including kahikatea, pukatea, rata, beech, rimu and nikau). Approximately 30 species of marine fish use the inlet at some stage of their life history. It is an important breeding and nursery area for snapper, flatfish, kahawai and whitebait. It is also important for birdlife (particularly waders) and is connected to large areas of relatively unmodified wetland, freshwater streams and terrestrial vegetation (Davidson 1990).

Seagrass beds in the northeastern arm of Whaganui Inlet looking toward the entrance

Whanganui Inlet has largely avoided major human impacts and, with much of the catchment protected within the Kahurangi National Park, it consequently has a low number of potential stressors. Those identified by Robertson and Stevens (2012), include:

- Potential for excessive muddiness if run-off from intensive land use or forest clearance (comprising 10% of the catchment area) is poorly managed. Climate change (increased storms) is expected to exacerbate these issues.
- Loss of high value salt marsh caused by impending sea level rise if inland migration is not facilitated.
- Changes in biological communities as a result of climate changes to seawater pH and temperature (e.g., loss of larger shelled invertebrates).
- Other lesser stressors include a partially modified terrestrial margin, presence of causeways, increased population pressure and margin encroachment (wildlife disturbance, predator introductions, habitat loss), and invasive species (e.g., Pacific oyster).

As part of TDC's coastal SOE monitoring programme, broad scale habitat mapping of the estuary (effectively a repeat of the 1990 baseline survey) was undertaken in 2016 (Stevens & Robertson 2017), with fine scale monitoring of the dominant habitat in the estuary undertaken at three sites in December 2016 (Robertson & Stevens 2016). Fine scale data were also collected in 2017 but have yet to be reported on – raw data are presented in Robertson and Robertson (2017).

The 2016 broad scale habitat mapping identified a rapid reduction in the seagrass extent evident between 2013 and 2016. To better assess temporal changes, TDC commissioned an assessment of baseline seagrass cover (Stevens 2018) based on aerial photography flown between 1945-1948, and digitising of seagrass extent based on the first ground-truthed mapping of the estuary undertaken in 1990 (Davidson 1990). Anecdotal observations of further seagrass losses between 2016 and 2020 led to a further survey of seagrass extent being commissioned by TDC and undertaken in March 2021 and is the focus of the current report.

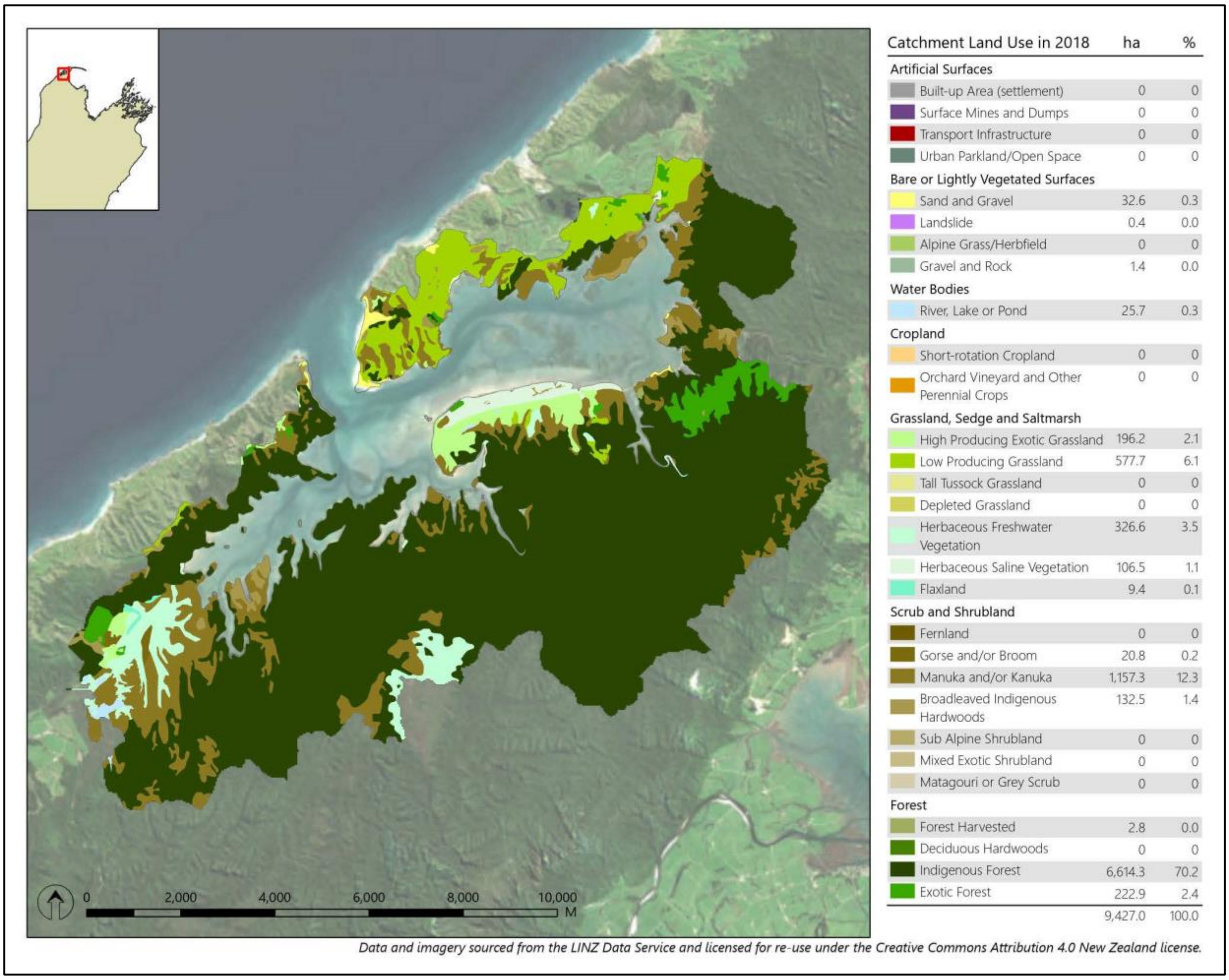


Fig. 2. Whanganui Inlet and surrounding catchment land use classifications from LCDB5 (2017/18) database.

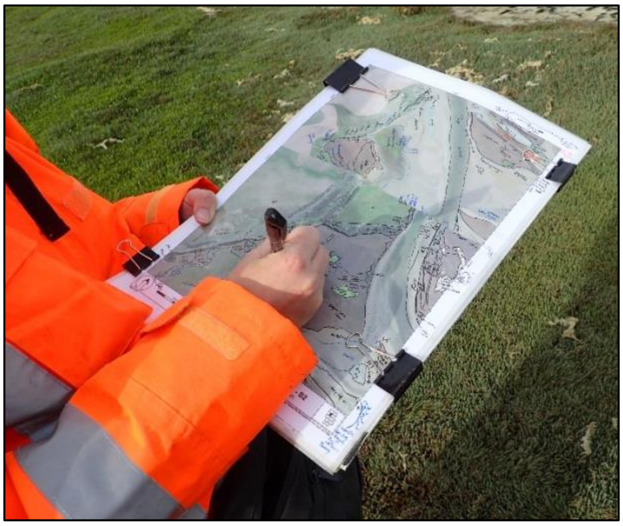
3. METHODS

3.1 BROAD SCALE MAPPING METHODS

Broad scale NEMP surveys involve describing and mapping estuaries according to dominant surface habitat features (substrate and vegetation). The type, presence and extent of substrate, salt marsh, macroalgae or seagrass reflects multiple factors, for example the combined influence of sediment deposition, nutrient availability, salinity, water quality, clarity and hydrology. As such, broad scale mapping provides time-integrated measures of prevailing environmental conditions that are generally less prone to small scale temporal variation associated with instantaneous water quality measures.

In 2021, NEMP methods were used to map and categorise intertidal estuary seagrass and underlying substrate. The mapping procedure combines aerial photography, detailed ground-truthing, and digital mapping using Geographic Information System (GIS) technology. For the present study, rectified ~0.5m/pixel resolution colour satellite imagery captured on 16 Feb 2021 was purchased from Apollo Mapping. Ground-truthing was undertaken on 26 March 2021 by experienced scientists who assessed the estuary on foot to map the spatial extent of seagrass. Because the NEMP provides no guidance on the assessment of seagrass beyond recording its presence when it is a dominant surface feature, seagrass patches were mapped during field ground-truthing using a 6-category rating scale (modified from FGDC 2012) as a guide to describe percentage cover (Fig. 3).

In the field, seagrass features were drawn directly onto 1:5000 scale laminated aerial photographs along with annotated notes on percentage cover and substrate type. The broad scale features were subsequently digitised into ArcMap 10.8 shapefiles using a Huion Kamvas 22 drawing tablet and combined with field notes and georeferenced photographs. From this information, maps were produced for the extent and density of seagrass beds.



Mapping broad scale habitat on laminated aerial imagery in the field

In relation to substrate type, Salt Ecology has extended the NEMP methodology to record the substrate present beneath vegetation, and has revised the NEMP substrate classifications for sand and mud (summarised in Appendix 1) by dividing previously merged categories of 'firmness' and 'muddiness' into independent categories. For 'muddiness', categories were further defined relative to sediment mud content, which can be subjectively assessed in the field and validated using laboratory analyses. These extensions enable a continuous substrate layer for the estuary to be produced, while improved characterisation of sediment muddiness facilitates its assessment as a potential determinant of habitat features and potential drivers of change.

3.2 ASSESSMENT OF ESTUARY CONDITION

The NEMP provides no criteria for assessing seagrass percentage cover or change. However, drawing on approaches from New Zealand and overseas, estuarine health metrics ('condition ratings') have been proposed as part of the Estuary Trophic Index (ETI) (Robertson et al. 2016b) which assign different indicators to one of four colour-coded 'health status' bands (Table 1).

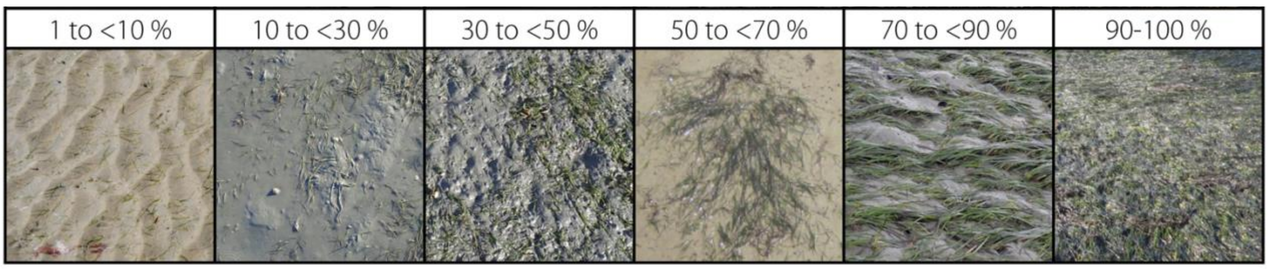


Fig. 3. Visual rating scale for seagrass percentage cover estimates. Modified from FGDC (2012).

A seagrass metric of the percentage change from a measured baseline, developed largely from previous broad scale mapping assessments, is used in the current report to help assess temporal changes in seagrass (Table 1).

Table 1. Percentage decrease from a measured baseline used to assess temporal changes in seagrass in the current report.

Very good	Good	Fair	Poor
< 5	≥ 5 to 10	≥ 10 to 20	≥ 20

3.3 DATA RECORDING AND QA/QC

Broad scale mapping provides a rapid overview of dominant estuary features. The ability to correctly identify and map features is primarily determined by the resolution of available aerial imagery, the extent of ground-truthing undertaken to validate features visible on photographs, and the experience of those undertaking the mapping. In most instances features with readily defined edges can be mapped at a scale of ~1:2000 to within 1-2m of their boundaries. The greatest scope for error occurs where boundaries are not readily visible on photographs, e.g., sparse seagrass or macroalgal beds. Extensive mapping experience has shown that transitional boundaries can be mapped to within ±10m where they have been thoroughly ground-truthed, but when relying on photographs alone, accuracy is unlikely to be better than ±20-50m, and generally limited to vegetation features with a percent cover >50%.

Following digitising of habitat features, in-house scripting tools were used to check for duplicated or overlapping GIS polygons, validate typology (field codes) and calculate areas and percentages used in summary tables.

3.4 PREVIOUS DATA

Data for all previous year's mapping are summarised in Stevens (2018), with underpinning GIS layers supplied to TDC. The table below, and the following text, summarise the broad-scale seagrass assessments undertaken.

Map Year	Imagery date and type	Mapping type
1948[a]	1945-48 B&W orthophoto	Desktop
1990[b]	1990 B&W orthophoto	Field survey
2013[a]	2013 Colour orthophoto	Desktop
2016[a]	2016 Colour orthophoto	Field survey
2021[c]	2021 Colour satellite image	Field survey

Provider: [a] Wriggle, [b] DOC (digitised by Wriggle), [c] Salt Ecology

The same methods and QA/QC processes used in 2021 were applied to previous data to ensure consistency in the mapping. In particular, it is noted that the 1945-1948 black and white photo series, which was digitised and reported on in Stevens (2018), relied on expert judgement to discriminate seagrass from other features within the estuary. Because seagrass cover is difficult to accurately map at low densities, historical mapping only included seagrass where there was a high degree of confidence in the features mapped, in this case seagrass beds with a predicted cover >50%. Therefore, the historical baseline is expected to underestimate total seagrass extent by excluding beds with <50% cover. While the retrospective historical mapping cannot be ground-truthed, it is considered to accurately reflect the >50% seagrass extent based on the author's extensive first-hand knowledge of the estuary and past experience in undertaking broad scale mapping and discriminating habitat features based on aerial photography.

The comprehensive report of Davidson (1990) included hard copy maps of seagrass, salt marsh and substrate that were based on extensive ground-truthing of the estuary. These maps were scanned, imported into ArcMap and overlaid on 1990 imagery. The 1990 seagrass features were then digitised using the NEMP methods described above. Minor changes were made to the originally mapped seagrass beds where improved geo-rectification of imagery or local knowledge of the estuary allowed improvements to be made.

Ground-truthing of the estuary was undertaken in 2016. Due to a delay in the scheduled delivery of 2016 imagery, the field survey was undertaken using 2013 imagery. The 2016 field survey identified substantial differences in the observed seagrass cover compared to that on the older 2013 imagery. Consequently the 2013 imagery was used to map the seagrass extent for 2013, with the 2016 seagrass extent mapped to the 2016 imagery when it became available.

Small bed of seagrass protectd by cobble bed in the northeast of the Inlet

4. RESULTS AND DISCUSSION

A summary of the March 2021 survey in Whanganui Inlet is provided below. Supporting GIS files (supplied to TDC and DOC's SeaSketch national seagrass dataset) as a separate electronic output) provide a more detailed dataset designed for easy interrogation and to address specific monitoring and management questions.

4.1 2021 SEAGRASS

Table 2 and Fig. 4 summarise 2021 seagrass (*Zostera muelleri*) percent cover. Seagrass was extensive (229ha), comprising 12% of the 1954ha intertidal area, with high cover (>50%) extending across 182.8ha (9.4%) of the intertidal flats. The largest seagrass beds were on the southern side of the main drainage channel in the eastern arm of the inlet, often raised slightly higher than the surrounding unvegetated sediment. Most seagrass (78%) was growing in sandy substrate (<10% mud content), 18% was growing in muddy-sand (10-25% mud), and only 4% was growing in sediment with a mud content >25%.

Extensive seagrass beds in the east of the estuary, raised slightly relative to the surrounding substrate

Table 2. Summary of seagrass percent cover categories, Whanganui Inlet, March 2021.

Percent cover category	Ha	%
Complete (>90%)	13.3	5.8
Dense (70 to <90%)	100.2	43.7
High-Moderate (50 to <70%)	69.3	30.3
Low-Moderate (30 to <50%)	9.1	4.0
Sparse (10 to <30%)	30.6	13.3
Very sparse (1 to <10%)	6.6	2.9
Total	229	100

Seagrass beds in the east of the estuary in pockets among rock habitat (top), and adjacent to saltmarsh and native bush (bottom)

Sparse seagrass growing in mobile sands

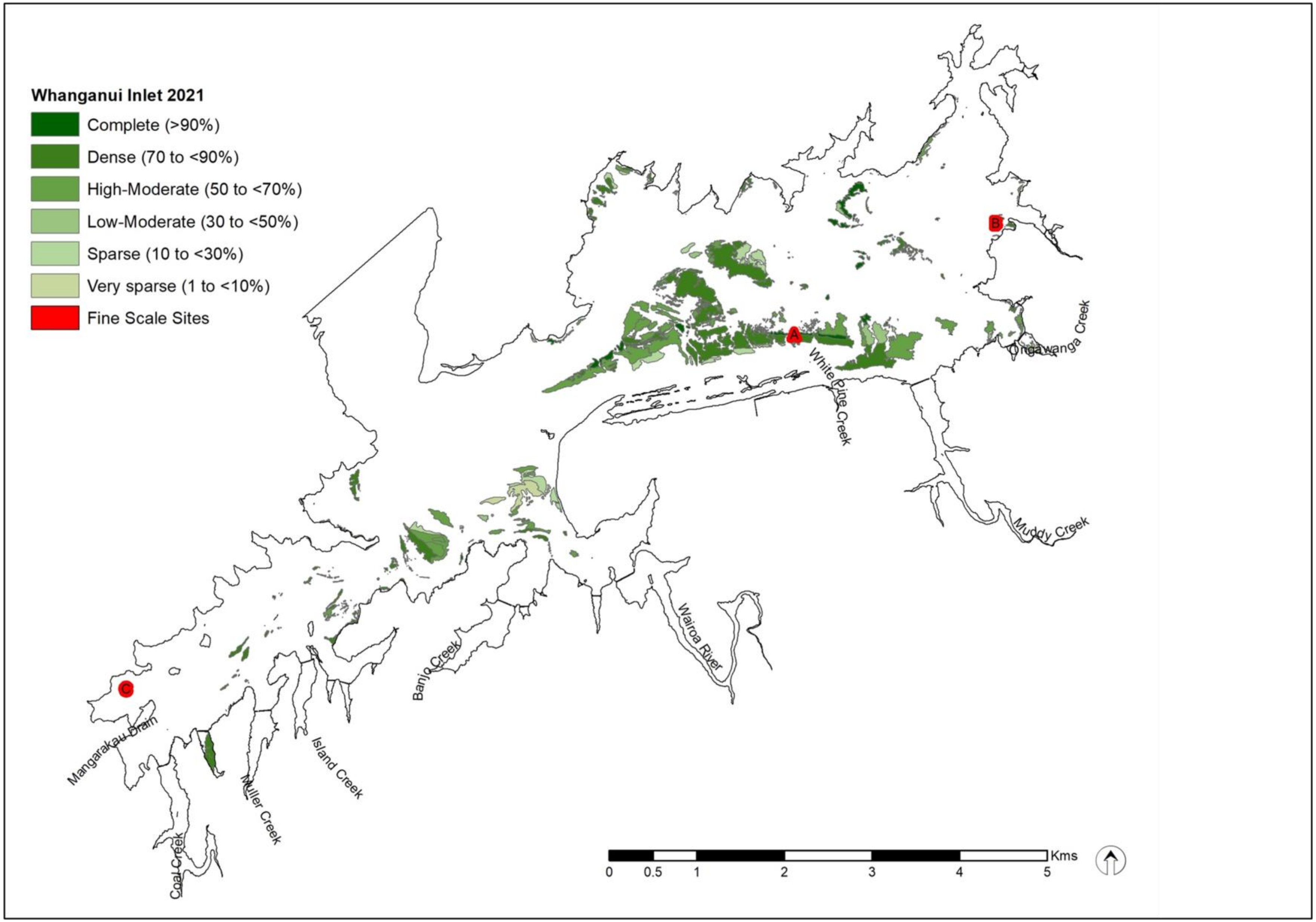
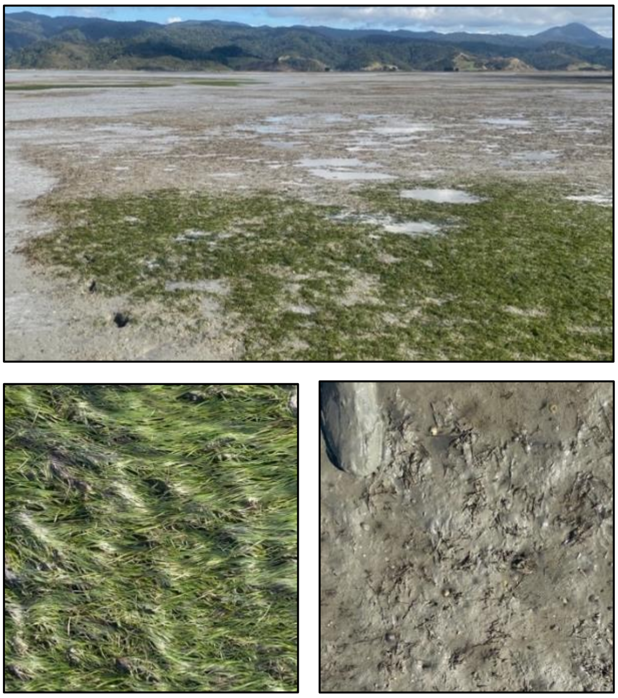


Fig. 4. Distribution and percent cover classes of seagrass, Whanganui Inlet, March 2021.

Many parts of the estuary were found to have seagrass beds that were in a very poor condition or had died since 2016. Based on residual plant material present, many of the losses appeared recent (i.e., in the previous 1-2 years). Whereas healthy seagrass beds had dark green and luxuriant growth, degraded beds were stunted with a sparse cover of brown fronds (see below).



Lush healthy seagrass (foreground and lower left) and stunted degraded seagrass (background and lower right)

The degraded and dying beds were widespread and almost exclusively in mud-dominated substrate. In many cases the only indication of previous seagrass extent was the presence of decaying root systems within the sediment (see adjacent photos).

Relative to the large areas of die-back, other impacts on seagrass were minor. There were localised vehicle (quad bike) impacts in the west arm, and evidence of wave scouring along the channel margins in the east arm (see photo below).

Physical erosion of the edge of a seagrass bed near a channel margin

In March 2021, large parts of the estuary had dead or dying seagrass beds

4.2 CHANGES IN SEAGRASS 1948-2021

Table 3 and Fig. 5 summarise changes in the total area of high cover (>50%) seagrass based on the previous mapping of Davidson (1990), Stevens & Robertson (2017) and Stevens (2018), with spatial changes summarised in Fig 6, and presented in more detail in Appendix 2 (Figs 7-11).

Table 3. Summary of high cover (>50%) seagrass (Ha and % of intertidal area) and percent loss compared to the 1948 baseline, Whanganui Inlet, 1948-2021.

Mapping Year	Ha	% Intertidal	% Loss since 1948*
1948	901.6	46.1	-
1990	816.1	41.8	9.5
2013	713.9	36.5	20.8
2016	328.9	16.8	63.5
2021	182.8	9.4	79.7

*condition rating colour bands presented in Table 1

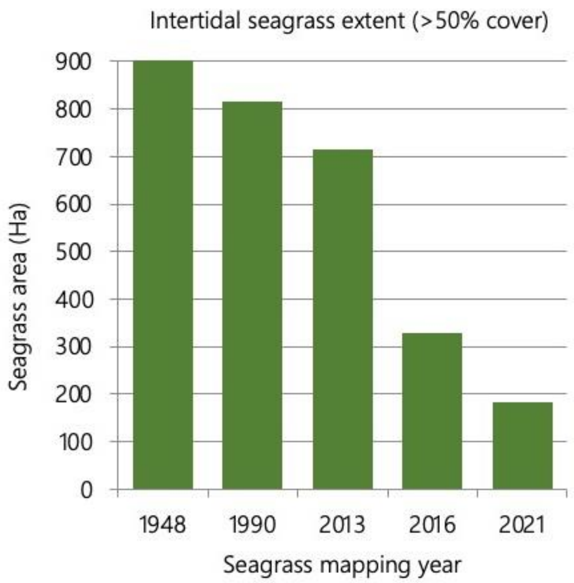


Fig. 5. Summary of changes in high cover (>50%) seagrass (Ha of intertidal seagrass), Whanganui Inlet, 1948-2021.

In 1948, 46% of the intertidal area had seagrass beds with >50% cover, with a continuous cover over most of the upper eastern arm (Fig. 6, Appendix 2 Fig. 7). In the western arm, where sediments appear to be more mobile, seagrass beds were present in smaller beds primarily on the intertidal flats near the south-eastern shoreline.

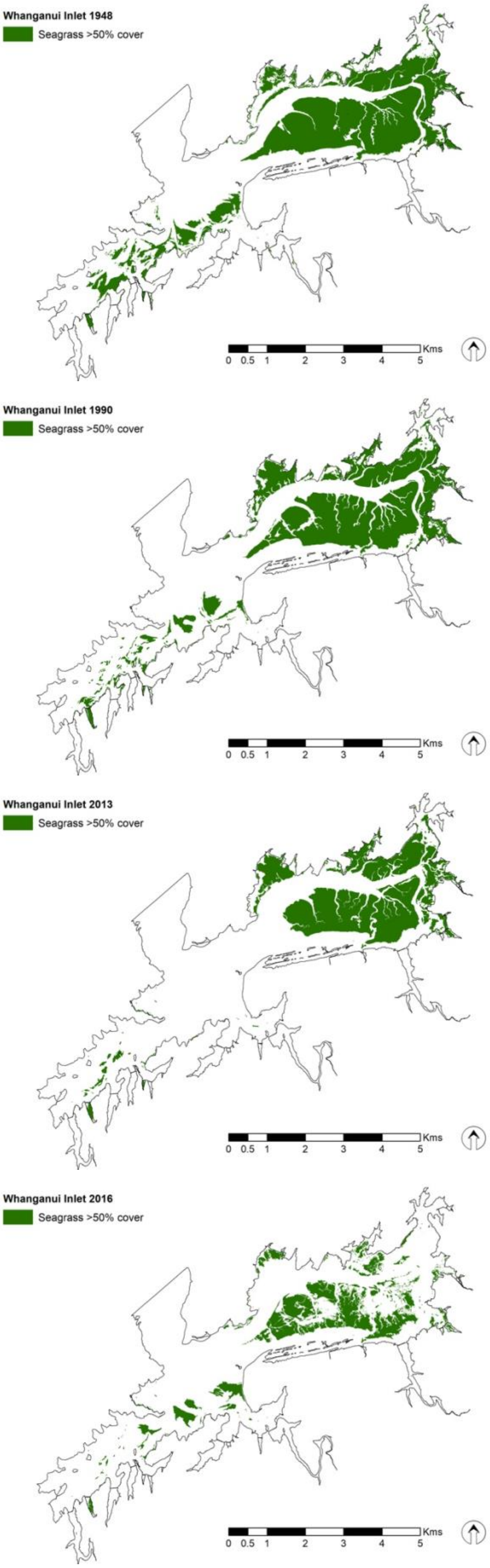
The first comprehensive ground-truthed mapping of the estuary undertaken by Davidson (1990) showed there had been an 86ha (9.5%) reduction in the extent of high cover (>50%) seagrass since 1948. These losses were primarily in the north of the east arm and, to a greater extent, in the west arm along the south-eastern shoreline where previously contiguous beds had begun to break up into smaller beds (Appendix 2 Fig. 8). This reflects a relatively minor change in seagrass extent over the 42 years between 1948 and 1990, with seagrass loss rated as 'good' based on the condition rating criteria presented in Table 1.

Between 1990 and 2013, there was a similar level of change with 102ha of seagrass loss (a 12.5% reduction). The condition rating for the reduction from 1990-2013 was 'fair'. Most losses were in the western arm and lower eastern arm, the latter also experiencing dieback of seagrass within existing beds, with large areas shifting from complete (>90%) cover to dense (70-90%) cover (Appendix 2 Fig. 9).

Between 2013 and 2016 there was a very rapid and extensive loss of seagrass. High cover (>50%) seagrass reduced by 385ha, a 54% reduction in 3 years, and a 64% reduction since the 1948 baseline, a condition rating of 'poor'. The vast majority of the losses were from the eastern arm (Appendix 2 Fig. 10) and occurred in seagrass beds that have been mud-dominated since at least 1990.

From 2016 to 2021, there was a further loss of 146ha of high cover (>50%) seagrass (Appendix 2 Fig. 11), a 44% reduction over the 5-year period and an 80% reduction compared to the 1948 baseline, a condition rating of 'poor'. Most of the losses occurred in the eastern arm where dead or dying seagrass fronds or rotting root masses were the only evidence of previously extensive seagrass beds. There was no evidence of seagrass having been displaced by opportunistic macroalgal growth, nor was there any obvious signs of seagrass being buried by fine sediment. Rather, the mud-dominated sediments previously supporting extensive beds of seagrass appeared to be eroding following the seagrass dieback, as evident in the photos on page 8.

Overall, 718ha of high cover (>50%) seagrass has been lost from the estuary since 1948, with most of the losses (531ha, 74%) occurring in the 8 years between 2013 and 2021. The significant loss of seagrass in the last decade likely represents one of the largest recent losses of intertidal seagrass recorded in New Zealand.



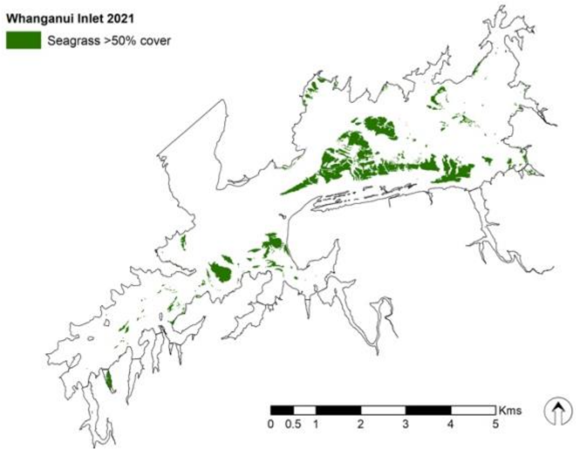


Fig. 6. Summary of changes in high cover (>50%) seagrass (Ha of intertidal seagrass), Whanganui Inlet, 1948-2021. See Appendix 2 for larger images.

5. SUMMARY AND RECOMMENDATIONS

5.1 SUMMARY OF KEY FINDINGS

This report has described the findings of a seagrass monitoring survey conducted in Whanganui Inlet based on the broad scale methods described in New Zealand's NEMP, and method extensions described in Section 3.

Whanganui Inlet is a relatively unmodified estuary set within a catchment dominated by native forest (Fig. 2). Estimates from NIWA's national estuary sediment load estimator (Hicks et al. 2019) predict the current sedimentation rate (CSR) to be only slightly higher than the natural sedimentation rate (NSR) – a CSR:NSR ratio of 1.05, suggesting near natural level of input. The estuary is predicted to be highly efficient at trapping sediment, retaining an estimated 97% of its catchment derived sediment. Based on current sediment loads and retention, the estuary-wide average rate of infilling is expected to be a relatively low ~0.2mm/yr, and under the 2mm/yr national guideline value recommended by Townsend and Lohrer (2015).

Despite this, the estuary has a relatively large area of mud-dominated sediments (1060ha, 54% of the intertidal flats, Stevens & Robertson 2016). The mud-dominated sediments are almost certainly terrestrial in origin, although the timing of inputs, and the specific sources (e.g., catchment erosion, landslides caused by localised high rainfall events or inputs washed into the estuary from West Coast catchments) remain unknown.

It is quite likely that mud-dominated sediments have accumulated over a long period of time. Their widespread presence in 1990 shows that fine sediment has been in the estuary for at least the last 30 years, while the growth of extensive seagrass beds in the fine sediment suggests it has been deposited at a rate within the assimilative capacity of the seagrass and at a level that not been significantly limiting to seagrass growth, notwithstanding the most recent decline.

The results of the 2021 mapping shows that the extensive (385ha) seagrass decline between 2013 and 2016 has continued, with a further 146ha reduction in high cover (>50%) seagrass between 2016 and 2021. These seagrass losses represent a very large reduction in the ecological value of the estuary, particularly through the loss of habitat for birds, fish and shellfish, but also though a reduced capacity to assimilate sediment and nutrient inputs, sequester carbon, and stabilise fine sediments that would otherwise increase turbidity.

To place the 531ha loss since 2013 into a regional context, the extent of high cover (>50%) seagrass in Waimea Inlet is just 21.6ha, Moutere 3.1ha, and Ruataniwha 14.6ha, reflecting a loss of >10 times the combined area of seagrass present in the other large SIDE estuaries in the Tasman region.

Extensive seagrass loss in the eastern arm in 2021 facilitating the mobilisation of mud-dominated sediments

There are several drivers that may potentially be responsible for the recent seagrass losses, although there are no direct data available to conclusively determine the specific causes at this stage.

Seagrasses, because of their high light requirements, are particularly vulnerable to light reductions from smothering or any deterioration in water clarity (e.g., York et al. 2013). In many New Zealand estuaries with intensively developed catchments, excessive fine sediment inputs have resulted in increased turbidity or smothering of seagrass by sediment. However, the prolonged presence of seagrass growing in mud-dominated sediments, and the absence of any obvious changes in catchment land use or land disturbance over the past decade (when the most dramatic decreases in seagrass have occurred), suggests it is unlikely that this is the primary driver of change. It may however be a secondary driver of losses as fine sediments that were previously trapped within the seagrass root masses and fronds are now being eroded from the dead seagrass beds and redistributed in the estuary, likely contributing to localised increases in turbidity and smothering.

This hypothesis is supported by data from fine scale monitoring sites establish in the estuary in December 2015 (see Fig. 4 for locations). At these three sites, sedimentation rates and sediment mud content were measured using methods described in Robertson and Stevens (2016) 13 months after establishment (January 2017), and again in March 2021. Sedimentation rate and sediment mud content results are summarised in Fig. 7 and Fig. 8, respectively, with data presented in Appendix 3. The measurements show localised variance in sedimentation rates and mud content, although the small number of sites, and low frequency of monitoring, limit extrapolation of these results to the wider estuary.

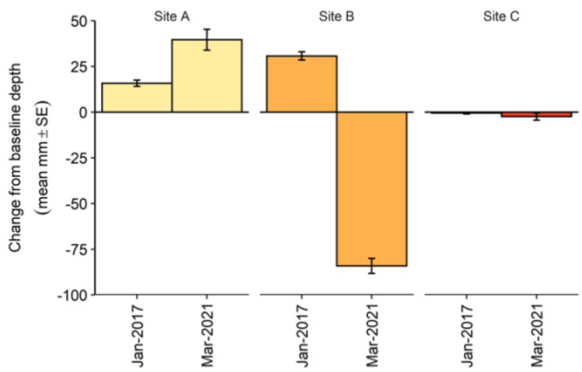


Fig. 7. Change in mean sediment depth over buried plates (±SE) relative to the Dec. 2015 baseline.

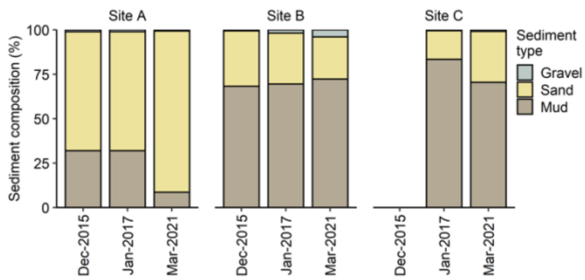


Fig. 8. Sediment particle grain size analysis showing percentage composition of mud (<63μm), sand (<2mm to ≥63μm) and gravel (≥2mm).

Site A in the centrally located seagrass beds had mean sediment accretion of +7.5mm/yr and a reduction in mud content from 32% to 9% suggesting deposition of sands within the seagrass beds. Although the rate of accrual is potentially detrimental to seagrass health, the shift from mud to sand-dominated sediments, and the continued presence of seagrass at the site, suggests impacts are likely minor.

Site C, in the unvegetated southwest arm, showed little change over the monitoring period with slight erosion (-0.5mm/yr), and the sediment mud content remaining high (70-83%).

In contrast, Site B, in the mud-dominated northeast arm, had a very large increase in sediment between 2016 and 2017 (+30.8mm), followed by substantial net erosion (-115mm/yr), and a near complete loss of seagrass, over the following four years (2017-2021). Mud content remained very high (68-73%). Because no monitoring was undertaken between 2017 and 2021, it is not possible to determine the temporal pattern of sediment accrual/erosion or seagrass loss over this period. The result showing substantial sediment erosion following the loss of seagrass at this site highlights that muddy sediments can be remobilised when no longer stabilised by seagrass. The fate of any remobilised sediment is unclear although based on the high predicted rate of sediment retention (97%), most of it it is expected to be retained in other parts of the estuary.

Because of the limited data, it is not possible to conclusively say whether the high rate of mud deposition between 2016-17 triggered the decline of seagrass through smothering, or whether it was due to secondary impacts such as reduced water clarity or from other possible factors as discussed below.

Eutrophic impacts from excessive nutrient inputs have the potential to fuel nuisance macroalgal growths that may result in seagrass smothering (e.g., Stevens et al. 2022), or the establishment of phytoplankton blooms which can cause seagrass losses through reductions in water clarity. However, eutrophic impacts are considered unlikely as the very limited catchment development means the areal nutrient load to the estuary is very low (4mgN/m^2/d), and well below the ~100mgN/m^2/d threshold at which nuisance macroalgae problems are predicted to occur (Robertson et al. 2017), while the high rate of tidal exchange limits the potential of phytoplankton blooms. There were no signs of phytoplankton blooms or nuisance macroalgae smothering of seagrass beds in Whanganui Inlet.

Other common causes of seagrass decline include pollutants (stormwater, herbicides, fuel spills, wastewater discharges etc.), physical disturbance (dredging, reclamation, aquaculture, trampling), introduced species, or climate change (Matheson et al. 2009). Of these, the latter is the most likely driver in Whanganui Inlet as the low level of catchment development and low population pressure minimise the presence of most other stressors.

Severe marine summer heatwaves are known to cause acute and dramatic die-offs of seagrass meadows (Sawall et al. 2021, Fraser et al. 2014, Thomson et al. 2015), with *Zostera muelleri* sensitive to small chronic temperature increases predicted under future climate change scenarios (York et al. 2013). Over recent years, the Tasman Sea has experienced intense marine heat waves in the summers of 2015/16, 2016/17 and 2018/19 (Behrens et al. 2022), and had a long-term average rate of sea-surface warming of 0.4°C per decade between 1981 and 2018 (Stats NZ 2019). These changes may be sufficient to directly impact seagrass, or other climatic changes could also be important including increased summer desiccation and heat stress, indirect impacts such as changes in salinity (e.g., Nejrup & Pedersen 2008), or changes in rainfall intensity and frequency. In particular, two extreme rainfalls events were recorded in the Tasman region in December 2011 and April 2013 (Macara 2016) which may have potentially contributed to the observed changes in seagrass.

As the recent seagrass losses do not appear to be caused by land use activities, there appears to be little TDC can do to directly prevent such seagrass losses. However, the loss of such a large area of high ecological value habitat is of significant concern, particularly as it may signal that seagrass beds in other parts of the region, and New Zealand, are potentially vulnerable to rapid change, as recently observed in Te Awarua-o-Porirua Harbour (Roberts et al. 2021).

5.2 RECOMMENDATIONS

- In response to the recent rapid and extensive seagrass losses it is recommended that mapping of seagrass continues at 5 yearly intervals to monitor ongoing change, with sediment plates measured annually.
- To investigate the likely cause of recent losses, it is recommended that TDC encourage detailed research as part of nationally funded science initiatives e.g., National Science Challenges, university-based research, Envirolink projects.

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APPENDIX 1: SUBSTRATE CLASSIFICATIONS

Table of modified NEMP substrate classes.

Consolidated substrate			Code
Bedrock		Rock field "solid bedrock"	RF
Coarse Unconsolidated Substrate (>2mm)			
Boulder/ Cobble/ Gravel	>256mm to 4.1m	Boulder field "bigger than your head"	BF
	64 to <256mm	Cobble field "hand to head sized"	CF
	2 to <64mm	Gravel field "smaller than palm of hand"	GF
	2 to <64mm	Shell "smaller than palm of hand"	Shel
Fine Unconsolidated Substrate (<2mm)			
Sand (S)	Low mud (0-10%)	Mobile sand	mS
		Firm shell/sand	fSS
		Firm sand	fS
		Soft sand	sS
Muddy Sand (MS)	Moderate mud (>10-25%)	Mobile muddy sand	mMS10
		Firm muddy shell/sand	fSS10
		Firm muddy sand	fMS10
		Soft muddy sand	sMS10
	High mud (>25-50%)	Mobile muddy sand	mMS25
		Firm muddy shell/sand	fMSS25
		Firm muddy sand	fMS25
		Soft muddy sand	sMS25
Sandy Mud (SM)	Very high mud (>50-90%)	Firm sandy mud	fSM
		Soft sandy mud	sSM
		Very soft sandy mud	vsSM
Mud (M)	Very high mud (>90%)	Firm mud	fM90
		Soft mud	sM90
		Very soft mud	vsM90
Zootic (living)			
		Cocklebed	CKLE
		Mussel reef	MUSS
		Oyster reef	OYST
		Tubeworm reef	TUBE
Artificial Substrate			
		Substrate (brg, bund, ramp, walk, wall, whf)	aS
		Boulder field	aS BF
		Cobble field	aS CF
		Gravel field	aS GF
		Sand field	aS SF

APPENDIX 2: DETAILED MAP OUTPUTS, 1948, 1990, 2013, 2016, 2021

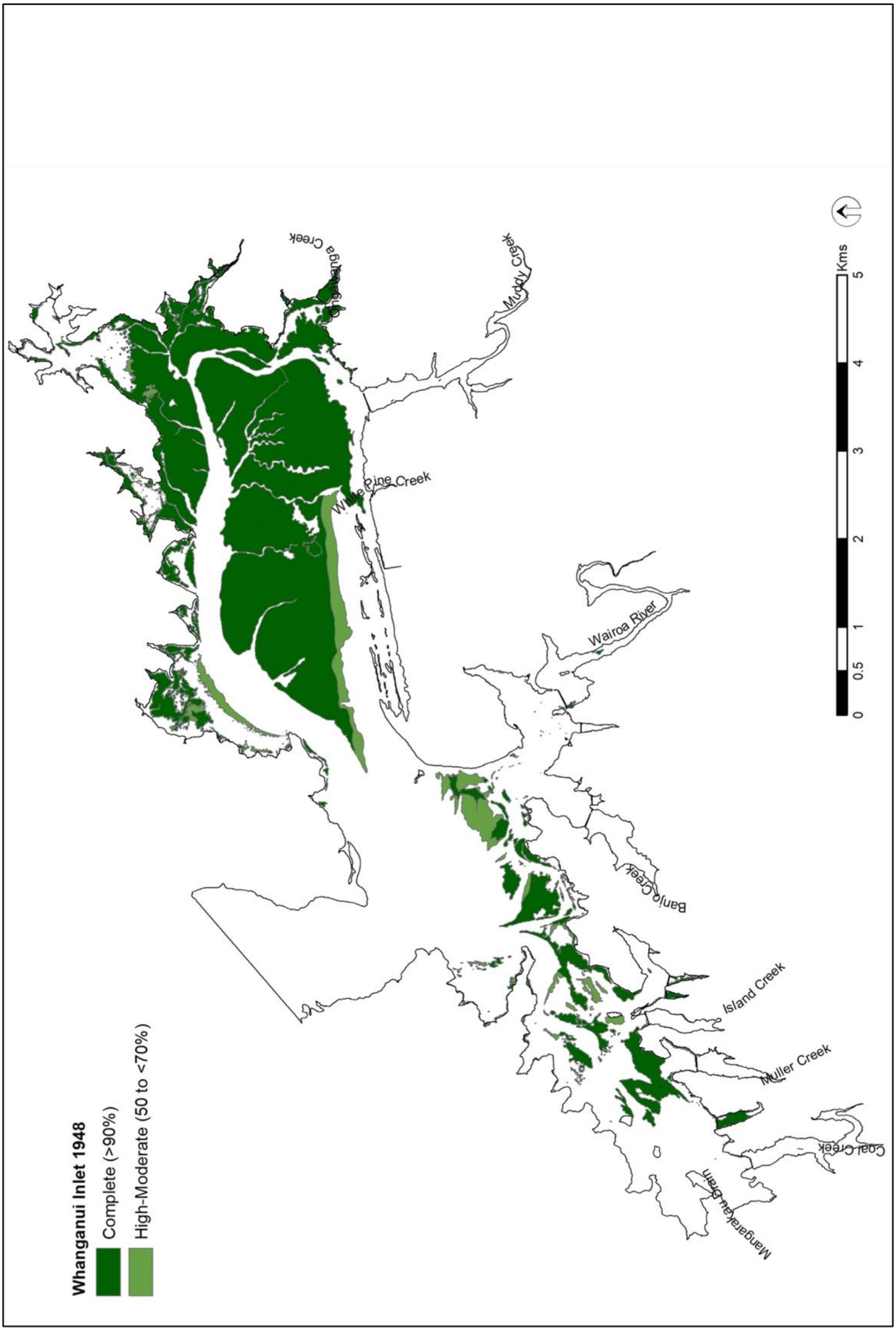

Fig. 9. Distribution and percent cover classes of seagrass, Whanganui Inlet, 1948.

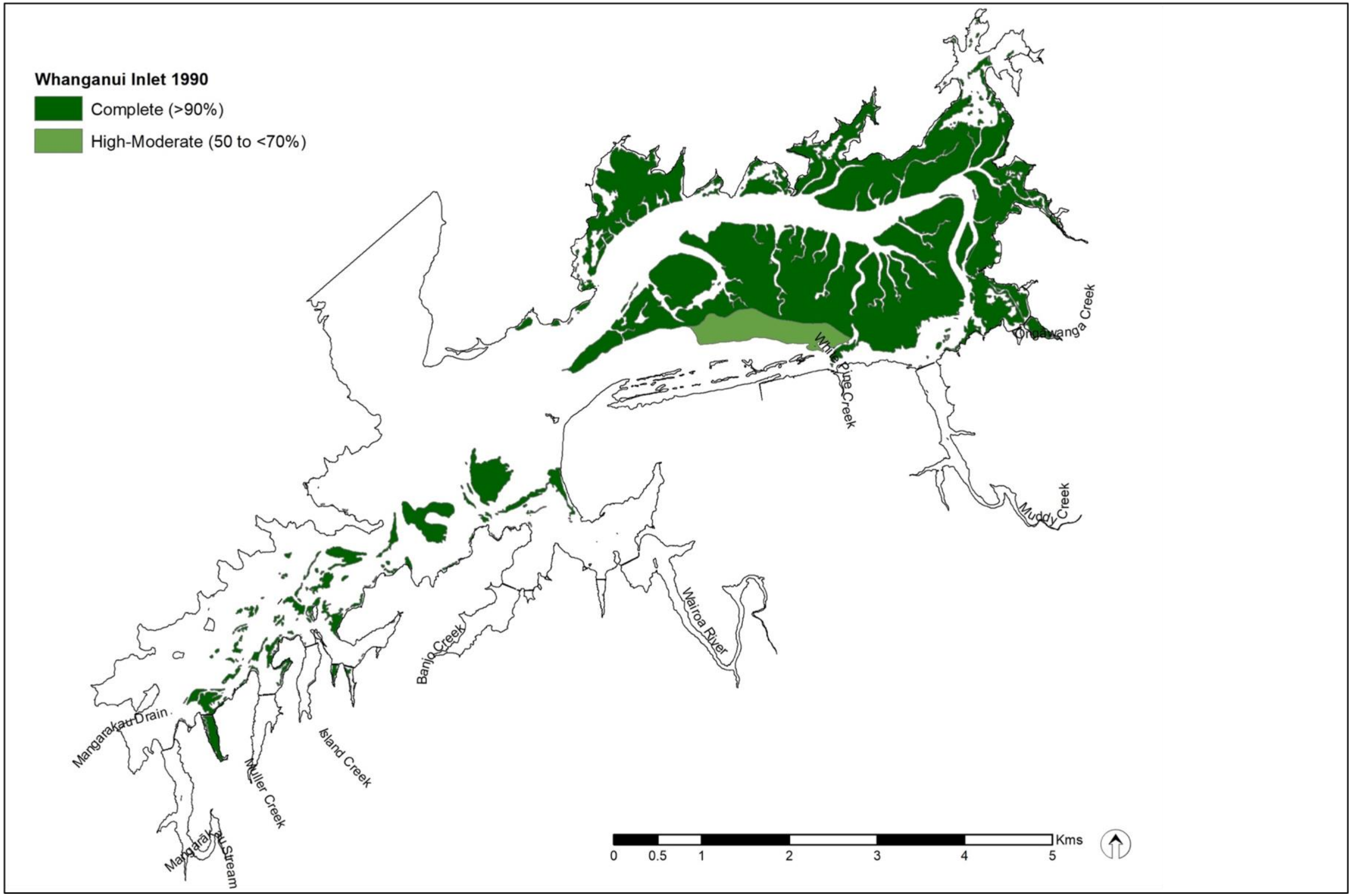

Fig. 10. Distribution and percent cover classes of seagrass, Whanganui Inlet, 1990.

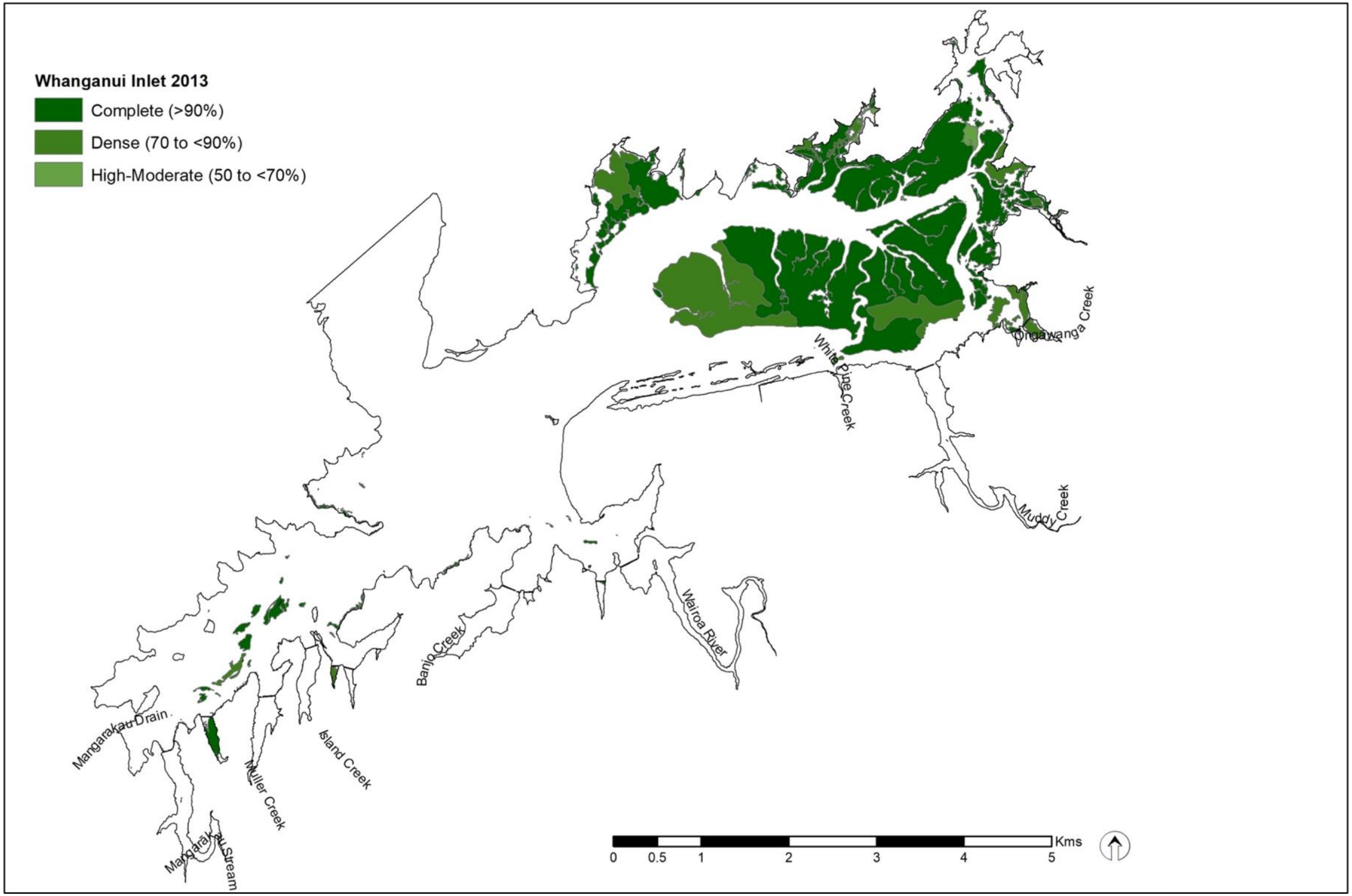

Fig. 11. Distribution and percent cover classes of seagrass, Whanganui Inlet, 2013.

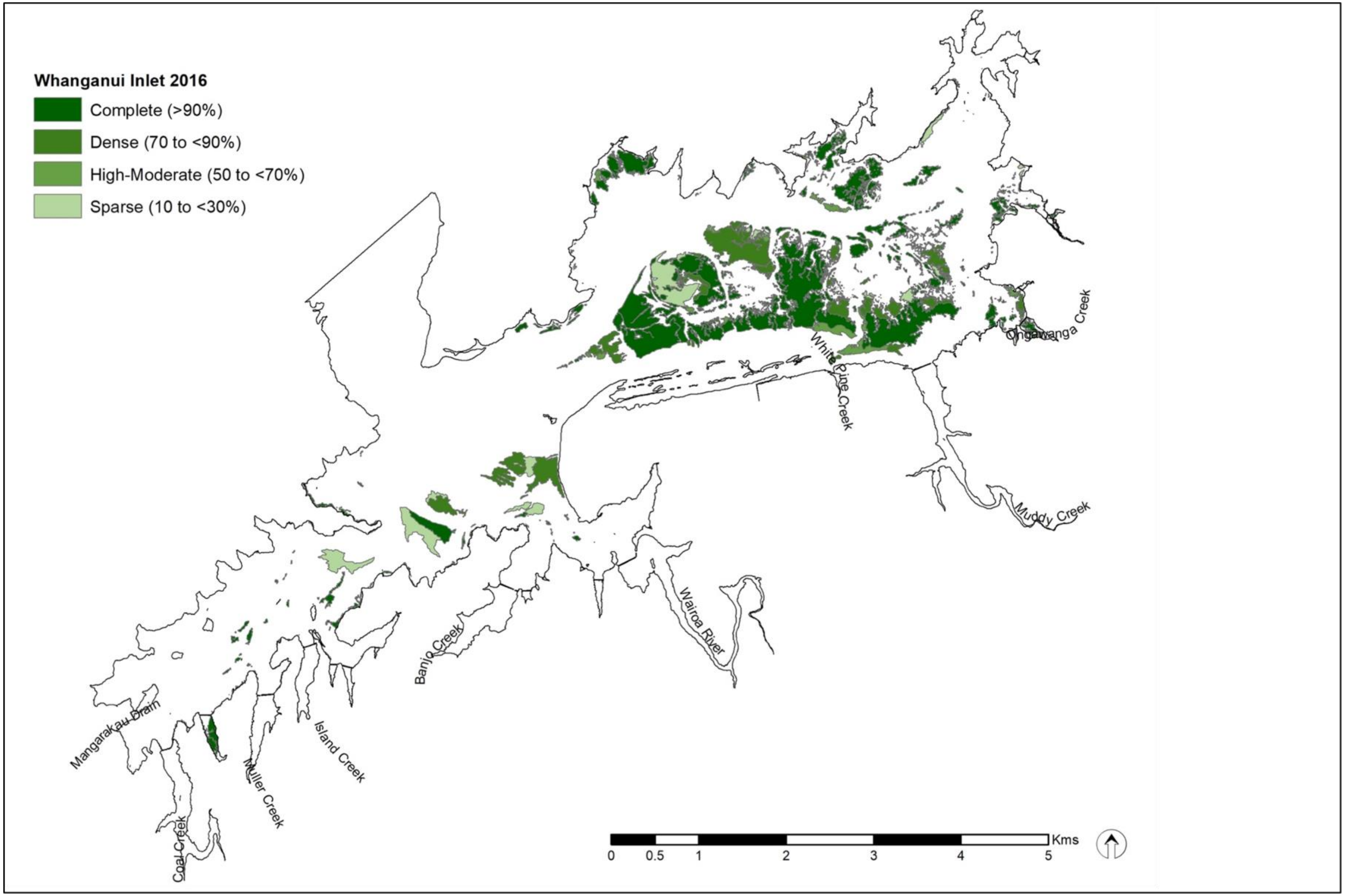

Fig. 12. Distribution and percent cover classes of seagrass, Whanganui Inlet, 2016.

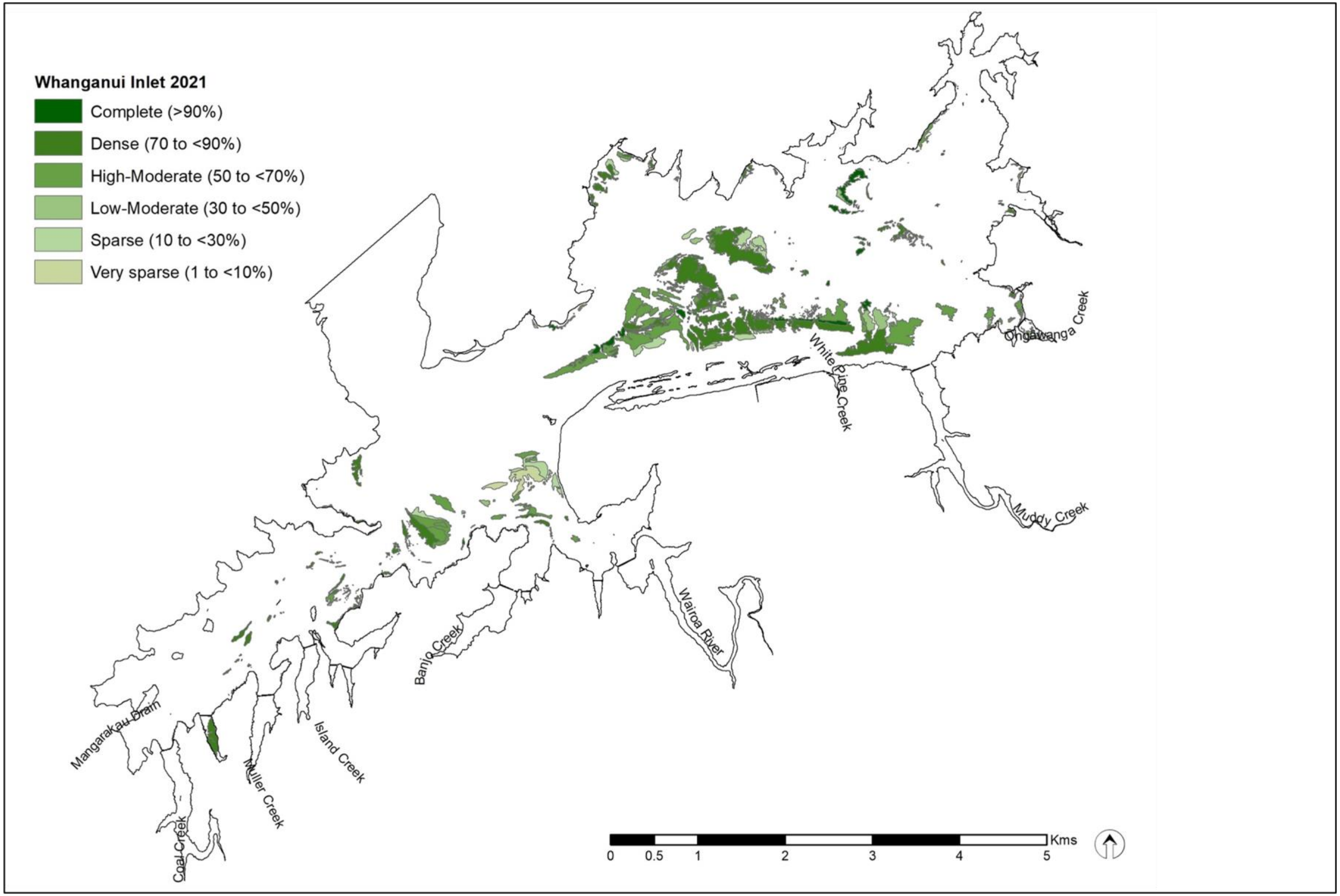

Fig. 13. Distribution and percent cover classes of seagrass, Whanganui Inlet, March 2021.

APPENDIX 3: SEDIMENT PLATE AND GRAIN SIZE DATA

Date	Year	Year-Month	Site	Sediment Texture	Sediment Type	Mud (%)	Sand (%)	Gravel (%)	aRPD (mm)	Plate	Depth (mm)	Baseline (mm)	Interval (days)	Annual adjustment (mm)	Annualised change (mm)	Change from baseline (mm)
2015-12-14	2016	2015-Dec	A	soft	MS25_50	32	67	1	10	p1	96	96	-	-	-	-
2015-12-14	2016	2015-Dec	A	soft	MS25_50	-	-	-	-	p2	96	96	-	-	-	-
2015-12-14	2016	2015-Dec	A	soft	MS25_50	-	-	-	-	p3	112	112	-	-	-	-
2015-12-14	2016	2015-Dec	A	soft	MS25_50	-	-	-	-	p4	104	104	-	-	-	-
2017-01-06	2017	2017-Jan	A	soft	MS25_50	32	67	1.1	10	p1	107	96	389	1.07	10.3	11
2017-01-06	2017	2017-Jan	A	soft	MS25_50	-	-	-	-	p2	113	96	389	1.07	16	17
2017-01-06	2017	2017-Jan	A	soft	MS25_50	-	-	-	-	p3	131	112	389	1.07	17.8	19
2017-01-06	2017	2017-Jan	A	soft	MS25_50	-	-	-	-	p4	120	104	389	1.07	15	16
2021-03-26	2021	2021-Mar	A	firm	S0_10	8.6	90.7	0.7	-	p1	143	96	1540	4.22	8.5	46.7
2021-03-26	2021	2021-Mar	A	firm	S0_10	-	-	-	-	p2	130	96	1540	4.22	4	33.7
2021-03-26	2021	2021-Mar	A	firm	S0_10	-	-	-	-	p3	139	112	1540	4.22	1.8	26.7
2021-03-26	2021	2021-Mar	A	firm	S0_10	-	-	-	-	p4	155	104	1540	4.22	8.4	51.3
2015-12-14	2016	2015-Dec	B	soft	SM50_90	68.3	31.1	0.6	10	p1	116	116	-	-	-	-
2015-12-14	2016	2015-Dec	B	soft	SM50_90	-	-	-	-	p2	95	95	-	-	-	-
2015-12-14	2016	2015-Dec	B	soft	SM50_90	-	-	-	-	p3	88	88	-	-	-	-
2015-12-14	2016	2015-Dec	B	soft	SM50_90	-	-	-	-	p4	86	86	-	-	-	-
2017-01-06	2017	2017-Jan	B	very soft	SM50_90	69.5	28.7	1.8	10	p1	151	116	389	1.07	32.8	35
2017-01-06	2017	2017-Jan	B	very soft	SM50_90	-	-	-	-	p2	123	95	389	1.07	26.3	28
2017-01-06	2017	2017-Jan	B	very soft	SM50_90	-	-	-	-	p3	114	88	389	1.07	24.4	26
2017-01-06	2017	2017-Jan	B	very soft	SM50_90	-	-	-	-	p4	120	86	389	1.07	31.9	34
2021-03-26	2021	2021-Mar	B	very soft	SM50_90	72.3	23.8	3.9	5	p1	43	116	1540	4.22	-25.6	-73
2021-03-26	2021	2021-Mar	B	very soft	SM50_90	-	-	-	-	p2	2	95	1540	4.22	-28.6	-92.7
2021-03-26	2021	2021-Mar	B	very soft	SM50_90	-	-	-	-	p3	3	88	1540	4.22	-26.3	-85
2021-03-26	2021	2021-Mar	B	very soft	SM50_90	-	-	-	-	p4	0	86	1540	4.22	-28.4	-86
2015-12-14	2016	2015-Dec	C	soft	SM50_90	-	-	-	10	p1	75	75	-	-	-	-
2015-12-14	2016	2015-Dec	C	soft	SM50_90	-	-	-	-	p2	103	103	-	-	-	-
2015-12-14	2016	2015-Dec	C	soft	SM50_90	-	-	-	-	p3	77	77	-	-	-	-
2015-12-14	2016	2015-Dec	C	soft	SM50_90	-	-	-	-	p4	77	77	-	-	-	-
2017-01-06	2017	2017-Jan	C	soft	SM50_90	83.4	16.1	0.5	10	p1	75	75	389	1.07	0	0
2017-01-06	2017	2017-Jan	C	soft	SM50_90	-	-	-	-	p2	103	103	389	1.07	0	0
2017-01-06	2017	2017-Jan	C	soft	SM50_90	-	-	-	-	p3	75	77	389	1.07	-1.9	-2
2017-01-06	2017	2017-Jan	C	soft	SM50_90	-	-	-	-	p4	77	77	389	1.07	0	0
2021-03-26	2021	2021-Mar	C	soft	SM50_90	70.5	28.6	0.9	6	p1	75	75	1540	4.22	0	0
2021-03-26	2021	2021-Mar	C	soft	SM50_90	-	-	-	-	p2	98	103	1540	4.22	-1.3	-5.3
2021-03-26	2021	2021-Mar	C	soft	SM50_90	-	-	-	-	p3	71	77	1540	4.22	-1	-6.3
2021-03-26	2021	2021-Mar	C	soft	SM50_90	-	-	-	-	p4	79	77	1540	4.22	0.5	2


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