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Extent of Seagrass in Tauranga Harbour: a Comparison of Satellite Automated classification versus Manual Delineation of Aerial Imagery

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Seagrass in Tauranga Harbour. Photo: Zhanchao Shao

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EXECUTIVE SUMMARY

This report outlines a comprehensive analysis of seagrass extent in Tauranga Harbour, New Zealand, using advanced remote sensing techniques. It compares two key methods: satellite-based supervised classification using Random Forest, and manual delineation via aerial photography. The research focuses on mapping the spatial and temporal changes in seagrass habitats, particularly the native species *Zostera muelleri*, from 1959 to 2024. Complementary to the satellite-based classification method, an easy-to-use graphical user interface (GUI) was developed, enabling users to efficiently access satellite imagery and generate detailed habitat maps.

The supervised classification using Random Forest proved effective for detecting seagrass habitats, achieving high accuracy (0.92 for Sentinel-2 and 0.88 for Landsat imagery). Despite occasional misclassifications, particularly in areas with high chlorophyll concentrations (i.e., the Tauranga Harbour sub-estuaries), the method provided a more time-efficient and scalable alternative to manual delineation. Aerial photography and manual mapping served as an important comparison but are less efficient in capturing less visible seagrass areas in deeper water or fragmented meadows.

The outcomes of this study derived from satellite data using the aforementioned supervised classification method showed that Tauranga Harbour experienced a substantial 49% reduction in seagrass extent between 1959 and 1990, with particular declines observed in the southern basin. However, a significant recovery trend was observed since 2016, reaching a peak of 4,327 hectares in the summer of 2023. However, since 2018, the seagrass extent started to decrease by more than 1000 ha during the winter months. Overall recovery was mainly observed in open Harbour regions, while sub-estuaries continue to experience degradation, likely driven by sedimentation and land-use changes. Intertidal seagrass remains the dominant contributor to total seagrass extent. However, subtidal seagrass has been steadily expanding since 2011, contributing to the ecological diversity of the Harbour. The subtidal seagrass extent reached 1,138 hectares in 2024, marking a significant recovery from earlier losses. Seagrass density mapping from 2019 to 2024 indicates central Harbour hotspots with nearly 100% density coverage in 2022. Declines in seagrass density have been observed towards the coastline, with fluctuations mainly occurring in peripheral regions.

The decline in seagrass coverage has been closely linked to increased turbidity, sedimentation, and nutrient inflows, reducing light availability for photosynthesis. Improvements in land management, fencing, and environmental plans have contributed to the seagrass recovery observed in recent years. Climate factors, such as rising water temperatures, have further supported the growth and expansion of seagrass beds, particularly in the subtidal zones.

Recommendations:

Continued Monitoring and Mapping: The use of high-resolution satellite imagery (e.g., RapidEye, Worldview) and bathymetric data should be incorporated in future mapping efforts to improve accuracy and distinguish between habitats like mangroves, saltmarshes, and seagrass.

Targeted Land Management: Continued efforts to reduce sedimentation and improve water

clarity in sub-estuaries are essential for supporting seagrass recovery. Focused interventions in high-risk sediment zones can further promote sustainable seagrass expansion.

Adaptation to Climate Change: While moderate warming trends are currently beneficial for seagrass growth, future management should consider potential risks associated with heat stress and ensure interventions are in place to mitigate these effects. The potential useful interventions include creating artificial shading structures in vulnerable areas during extreme heat events and promoting adjacent ecosystems like mangroves that can buffer temperature fluctuations.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

1. INTRODUCTION

1. 1 Purpose

The purpose of this study is to provide accurate mapping of seagrass extent throughout Tauranga Harbour in the Bay of Plenty Region using remote sensing techniques. The mapping should provide:

- Analysis of the seagrass extent by utilising satellite imagery;
- Development of an online GUI using the satellite classification method;
- Comparison between the satellite-derived results and the manually delineated layer based on aerial photos and
- Summary of the advantages and disadvantages of both methods.

1.2 Background

1.2.1 Seagrass

Seagrasses are vital components of coastal and marine ecosystems, characterised by their unique ability in sediment stabilization and carbon sequestration. The dense root and rhizome networks can trap fine particles and bind sediments, which effectively reduce erosion and mitigate the impact of hydrodynamic forces such as waves and currents. (Ondiviela et al., 2014). Apart from accumulating carbon from sediments, seagrass fixes a substantial amount of carbon by photosynthesis, which helps remove carbon dioxide from the water (Silva et al., 2009). The ecological importance of seagrass is now extended further, considering they provide habitat and nourishment for estuarine species, thereby enhancing biodiversity and productivity of coastal ecosystems (Duarte, 2002).

Zostera muelleri, the only native seagrass species in New Zealand, shows several unique characteristics that enable it to thrive in the country's temperate coastal environments. One notable feature is its high tolerance to fluctuating salinity levels, allowing it to colonize a wide range of estuarine and intertidal habitats (Collier et al., 2014). *Z. muelleri* has been shown to produce the highest shoot density at 12 psu (per salinity unit) after ten weeks compared to higher or lower salinities. This adaptability makes *Z. muelleri* particularly resilient to environmental variability, such as changes in tidal flow, sediment movement, and water temperature (Collier et al., 2014). The species is also capable of vegetative asexual and sexual reproduction in lower salinity environment, where even small fragments of the plant can regenerate and form new colonies (Zabarte-Maeztu et al., 2023). This resilience is crucial for the recovery of seagrass meadows following disturbances like storms or anthropogenic impacts.

1.2.2 Tauranga Harbour

Tauranga Harbour, situated in the Bay of Plenty on New Zealand's North Island, is the largest estuarine inlet in the region, encompassing an area of 201 km² (Figure 1). The Harbour is characterised by its shallow waters and extensive intertidal zones, which constitute 66% of the total area. The geomorphology of Tauranga Harbour is defined by its containment within Matakana Island, a barrier island, and two barrier tombolos: Mount Maunganui at the southern entrance and Bowentown to the north (De Lange & Healy, 1990). The Harbour is divided into two primary basins: the northern and southern basins, which are hydrologically separated during low tide, and the smaller Town Reach basin located in the far south. At mean high water, the northern basin has a volumetric capacity of approximately 178 million m³, while the southern basin holds around 278 million m³. Additionally, Tauranga Harbour encompasses over 20 minor sub-estuaries, contributing to its complex hydrodynamic regime.

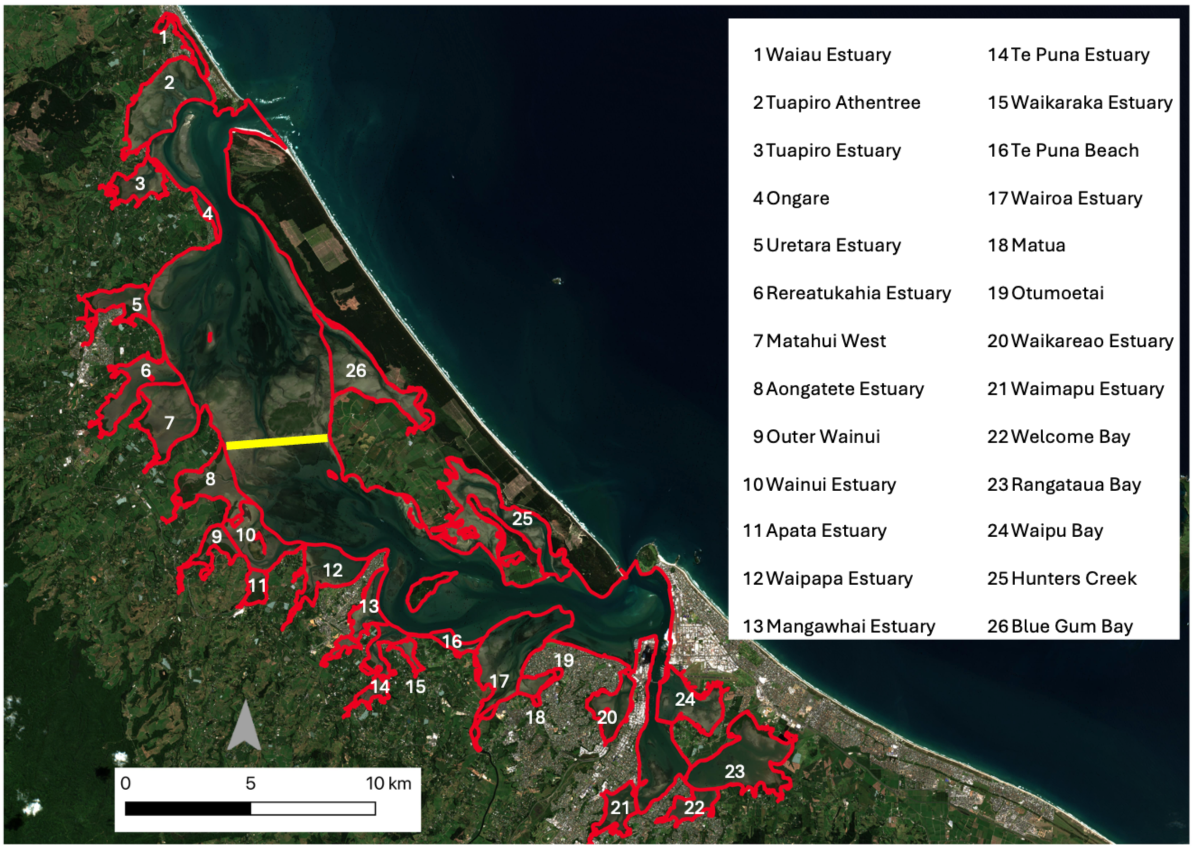

Figure 1 The map of Tauranga Harbour with all sub-estuaries and catchments. The yellow line divides the Harbour into the northern and southern parts.

The catchment area of Tauranga Harbour spans approximately 1,300 km², with substantial development dedicated to horticultural and agricultural activities. The catchment for the northern basin covers 270 km², with a mean freshwater inflow of 4.1 m³/s, equating to 0.1% of the Harbour's volume per tidal cycle. In contrast, the southern basin's catchment extends over 1,030 km², with a mean freshwater inflow of 30.5 m^3/s, constituting 0.48% of the Harbour's volume per tidal cycle (Davies-Colley & Healy, 1978). These freshwater inflows, in conjunction with tidal dynamics, are critical in governing sediment transport processes and the ecological characteristics of Tauranga Harbour. The Harbour supports a diverse range of marine species and holds substantial cultural significance, particularly for the indigenous Māori population. Ongoing efforts are directed towards balancing regional development with the preservation of Tauranga Harbour's natural and cultural resources, ensuring its sustainability for future generations.

2. METHODS

2.1 Data acquisition

To assess the seagrass extent in the long term, a combination of different remote sensing imagery is utilised including aerial photography, Landsat (30m spatial resolution) and Sentinel-2 (10m spatial resolution) (Table 1). The earliest available aerial photography data can be dated back to 1959 (Park, 1999). In order to avoid missing seagrass meadows in the optically deep water, the Landsat and Sentinel-2 imagery was derived at the low tides from United States Geological Survey EarthExplorer (USGS, https://earthexplorer.usgs.gov/) and European Space Agency (ESA, https://dataspace.copernicus.eu/explore-data/data-collections/sentinel-data/sentinel-2), respectively. Due to the difficulty of obtaining sufficient good satellite data with low cloud rates (Landsat revisits New Zealand every 16 days), between 1990 and 2018, one Landsat image collected in late summer (February or March) was selected to estimate the annual seagrass extent in summer. After 2018, Sentinel-2 replaced Landsat for seagrass mapping due to its high spatial and temporal resolution (10 m and 5 days). Therefore, for each season, one image was selected since 2018. In order to ensure seagrass extent derived from the selected images for each season was representative and consistent, the scanning dates of the selected satellite images were collected during adjacent months within each season where possible (see Table 1). We also referenced the published papers and reports to amplify the datasets (e.g. Park 2011 & Ha et al., 2020).

Table 1 The sources of remote sensing data used in seagrass mapping.

Time	Source of data	Reference
1959	Aerial photo	Park, 2011
1990	Landsat-4	Ha et al., 2020
1991	Landsat-4	This report
1996	Aerial photo	Park et al., 2011
2001	Landsat-5	Ha et al., 2020
2002	Landsat-5	This report
2005	Landsat-5	This report

2007	Landsat-5	This report
2010	Landsat-5	This report
2011	Aerial photo	Park, 2011
2014	Landsat-8	Ha et al., 2020
2015	Landsat-8	This report
2016	Sentinel-2	This report
2018	Sentinel-2	This report
2019/03/02	Sentinel-2	Shao et al., 2024
2019/05/01	Sentinel-2	Shao et al., 2024
2019/06/25	Sentinel-2	Shao et al., 2024
2019/10/08	Sentinel-2	Shao et al., 2024
2020/02/05	Sentinel-2	Shao et al., 2024
2020/05/20	Sentinel-2	Shao et al., 2024
2020/09/02	Sentinel-2	Shao et al., 2024
2021/01/10	Sentinel-2	Shao et al., 2024
2021/08/03	Sentinel-2	Shao et al., 2024
2021/09/17	Sentinel-2	Shao et al., 2024
2022/02/24	Sentinel-2 & Aerial photo	This report; Crawshaw & Park, 2024
2022/08/03	Sentinel-2	This report
2022/10/22	Sentinel-2	This report
2023/03/01	Sentinel-2	This report
2023/05/15	Sentinel-2	This report
2023/06/14	Sentinel-2	This report
2023/09/07	Sentinel-2	This report
2024/02/04	Sentinel-2	This report
2024/05/29	Sentinel-2	This report

2.2 Seagrass habitat mapping using supervised classification with random forest

Random Forest is an ensemble learning method that builds multiple decision trees during training and merges their outputs to improve the overall accuracy and robustness of the model. By averaging the results of multiple trees, it reduces the risk of overfitting and produces more reliable predictions (Breiman, 2001). Therefore, in order to extract the spatial extent of seagrass in Tauranga Harbour accurately, we applied supervised classification with random forest on the satellite-derived reflectance in visible and near-infrared bands (Band Blue, Green, Red and NIR) (Shao et al., 2024). Bathymetry (or LiDAR) was also used as an input because it could substantially improve the accuracy of the classification to distinguish seagrass, mangroves and saltmarsh due to their different habitat elevation.

Two classification models were established based on the source of the satellite imagery (either Sentinel-2 or Landsat).

Table 2 The number of samples for different classes used as input (training and validation data) for the two supervised classification models.

Class	Number of samples (Sentinel-2)	Number of samples (Landsat)
Sparse seagrass	1,892,260	137,232
Dense seagrass	455,084	11,096
Unvegetated flats	2,007,884	222,964
Deep channels	1,871,672	100,964
Mangroves	521,652	46,224
Bad points (Sentinel-2 only)	220	-*
Shallow water	1,715,656	106,320
Saltmarsh	24,860	3,480
Clouds	1,512	220
Others	624	21,632
In total	8,491,420	650,132

* Sensor malfunctions or transmission errors can result in bad data points, where parts of the image contain no valid information, leading to 0 values in all bands.

A total of 8,491,420 and 650,132 samples were derived from over 100 polygons throughout all the Sentinel-2 and Landsat images, respectively. Ten different classes for both models were involved: sparse seagrass, dense seagrass, unvegetated flats, deep channels, mangroves, bad points, shallow water, salt marsh, clouds and others (Table 2). These classes were manually labelled based on the expert knowledge of existing habitats in those locations (available on https://legacy.seasketch.org/#projecthomepage/5357cfa467a68a303e1bb87a). Samples were randomly split into training/validation (80%) and testing datasets (20%). The optimum values for the hyperparameters of the models (i.e. number of trees, minimum sample split, minimum sample leaf and maximum tree depth) were derived by applying fivefold cross-validation. After supervised classification, QGIS 3.18 was used to calculate the area covered by seagrass and visualize the results. The classification map was eventually compared to the manually delineated seagrass map based on aerial photography using relative difference (from the Bay of Plenty Council) (Equation 1).

$$Relative\ difference\ (\%) = \frac{|A - B|}{\left(\frac{A+B}{2}\right)} * 100 \quad (1)$$

Where A and B are the detected seagrass extent based on machine learning and manual delineation using aerial photography, respectively. The Harbour was divided into 2,052 tiles with the same area (100, 025 m^2) using Polygon Divider in QGIS to help identify the regions where the greatest difference occurred.

2.3 Seagrass density mapping using machine learning regressions

After the seagrass areas were detected, support vector regression was applied to estimate the percentage cover of each seagrass-dominated pixel. This model was developed based on 500 ground truthing points deployed in summer 2022 using quadrats, which measured the seagrass density (%) along the coastline of Tauranga Harbour. The data has been made available in the recent publication (Shao et al., 2024). Apart from the visible and NIR bands from Sentinel-2, the support vector regression method also required the normalized difference water and vegetation index (NDWI and NDVI). The model was not applicable for Landsat-derived images due to the lack of ground truthing points. Therefore, the seagrass density maps were only provided after 2018.

2.4 Graphical User Interface (GUI) for coastal wetland classification

To enhance the accessibility and efficiency of coastal wetland classification, a user-friendly GUI was developed complementary to the Random Forest classification method applied to Sentinel-2 satellite imagery (Figure 2). The GUI was developed with Streamlit coupled with Python 3.10. The source codes with the manual have been available on GitHub for public use (https://github.com/zhanchaoshao/CoastalHawkeyes). The tool was designed for users who need to classify seagrasses and other wetland types, including salt marsh and mangroves, within estuarine environments like Tauranga Harbour.

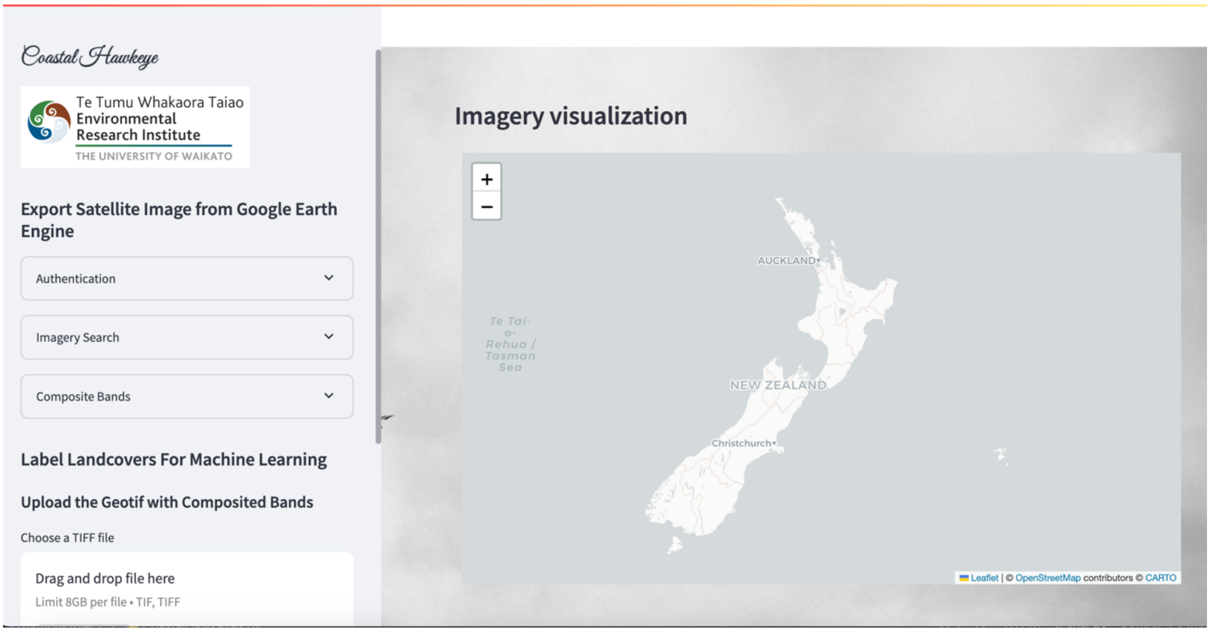

Figure 2 The GUI for coastal wetland classification.

The GUI allows users to easily access and filter freely available Sentinel-2 imagery based on specific criteria, including cloud coverage and date/time range, ensuring that the selected images meet the project's requirements. Users can define their area of interest by entering coordinates or uploading an externally created polygon.

For the creation of training data, users can manually assign polygons to the various habitat classes they wish to identify or upload a shapefile of already existing ground truthing data. This training data serves as the foundation for the classification model, allowing it to learn and accurately classify the different wetland types in the area of interest.

Once the training data is prepared, users can run the classification model. The model processes the data and generates a detailed classification map complementary to a performance score for each of the identified classes. This output can be easily visualised and exported for further analysis in programs like ArcMap, ArcGIS and QGIS, enabling users to compare changes in wetland coverage over multiple years. Further details and guidance can be found in the manual.

3. RESULTS AND DISCUSSION

3.1 Classification results

3.1.1 Performance of supervised classification with random forest

After setting the optimum values for hyperparameters of the random forest model, the supervised classification model reproduced the testing data from both Sentinel-2 and Landsat with high accuracy and precision (Table 3). The optimum values for the number of trees, minimum sample split, minimum sample leaf, and maximum tree depth were 160, 4, 4, and 9 for Sentinel-2 data, and 280, 4, 4, and 8 for Landsat data, respectively. Both models returned high overall accuracy of all classes with values of 0.92 (Sentinel-2) and 0.88 (Landsat). Due to its unique spectral signature and specific thriving elevation (supratidal zones), mangroves had the highest accuracy and precision in both models. Due to the low spatial resolution of Landsat, the classification derived from the Landsat imagery resulted in a decrease in accuracy of 7% ~ 15% compared to the Sentinel-2 images.

Table 3 Performance of the supervised classification model for different habitats in Tauranga Harbour. Other non-vegetation classes are omitted in the table.

Source	Class	Accuracy	Precision	Recall	F1-score	Support
Sentinel -2	Sparse seagrass	0.89	0.89	0.88	0.89	378,452
	Dense seagrass	0.87	0.89	0.84	0.86	91,017
	Mangroves	1.00	0.99	1.00	1.00	104,330
	Saltmarsh	0.96	0.99	0.95	0.97	4,972
	Unvegetated flats	0.91	0.90	0.94	0.92	401,577
Landsat	Sparse seagrass	0.79	0.81	0.75	0.78	27,446
	Dense seagrass	0.74	0.78	0.67	0.72	2,219
	Mangroves	0.93	0.93	0.98	0.96	9,245
	Saltmarsh	0.87	0.94	0.72	0.81	696
	Unvegetated flats	0.89	0.88	0.92	0.90	44,593

After the accuracy performance of the model was confirmed, the model was applied to all pixels of the Harbour. Figure 3 showed the detailed classification of different habitat types using the classification model based on Sentinel-2 and Landsat imagery. The Sentinel-2 image (Figure 3a) offered a higher spatial resolution than the Landsat (Figure 3b), which led to finer detail in detecting habitat boundaries, especially around the mangrove and salt marsh areas. For the pixels mixed with multiple vegetation types (e.g. the of Rangataua Bay), the model returned the classification based on the dominant habitat type within the pixels.

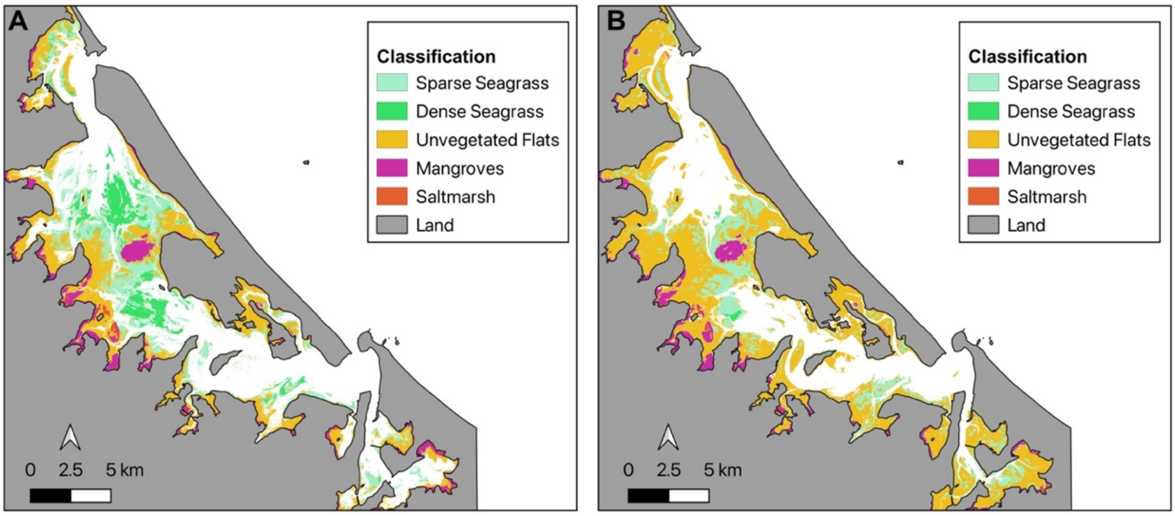

Figure 3 Habitat maps using supervised classification with random forests based on Sentinel-2 (a)and Landsat imagery (b). The Sentinel-2 and Landsat imagery was scanned on 24 February 2024 and 5 January 2001, respectively. The area of unvegetated flats might vary due to the different tidal levels when the satellite revisits the site.

3.1.2 Comparisons between satellite automated classification and manual delineation of aerial imagery

The seagrass extent from the supervised classification and the manually delineated layer based on aerial photography had an overall high-level matchup, especially in the main estuaries. The classification model identified 3,962 ha of seagrass based on the Sentinel-2 image (2022), which was close to the result derived from the manual delineation of aerial photography (3,740 ha, 2022) (Figure 4a). Therefore, the relative difference was mostly below 0.6 (S1). Due to the restriction of spatial resolution of Landsat, supervised classification returned a relatively low value of seagrass extent (2,380 ha, 2011) compared to that derived from the aerial photo (2,740 ha, 2011) (Figure 4b). Manual delineation could detect 12.8% more seagrass extent than automated supervised classification in the regions close to the coastline. However, supervised classification based on satellite imagery was more sensitive to the main Harbour, which contributed 8.4% more of the detected extent.

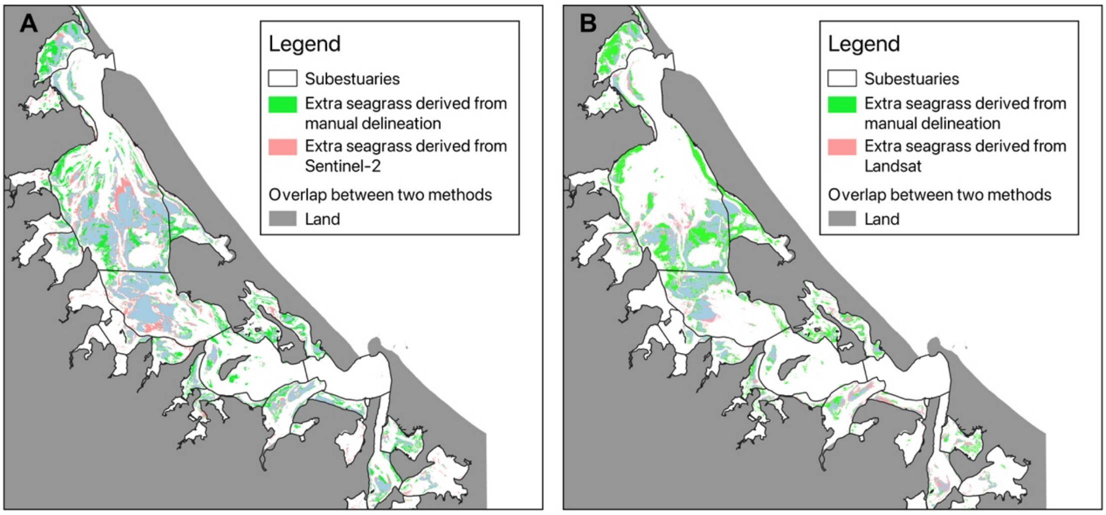

Figure 4 Comparison of seagrass extent derived from supervised classification using satellite imagery (a, Sentinel-2; b, Landsat) with manual delineation from aerial photography. The Sentinel-2 image was derived from 24 February 2024 and the Landsat image was derived from 17 February 2011. Refer to Supplementary Figure S1 for detailed distribution maps from both methods.

Significant discrepancies were observed between manual delineation and supervised classification mostly in sub-estuaries like Blue Gum, Wairoa, and Mangawhai. In 2022, Blue Gum showed a seagrass extent of 141.2 hectares using manual delineation, while supervised classification detected only 79.4 hectares—a difference of 61.8 hectares (Table 4). Similarly, Wairoa had 147.5 hectares identified manually compared to 89.6 hectares through supervised classification. Mangawhai also exhibited a substantial difference, with manual delineation recording 68.2 hectares versus 33.7 hectares from supervised classification (Table 4).

Table 4 Detected seagrass extent in sub-estuaries using supervised classification and manual delineation.

	2022 (ha)		2011 (ha)	
Sub-estuary	Manual delineation	Supervised classification	Manual delineation	Supervised classification
Waiau	18.5	8.9	14.3	7.1
Tuapiro	6.6	2.5	12.2	5.3
Uretara	7.2	1.6	4.5	1.2
Blue Gum	141.2	79.4	215.5	119.7
Rereatukahia	20.3	8.1	8.9	7.1
Aongatete	0.1	0	0.1	0.1
Mangawhai	68.2	33.7	42.5	45.2
Hunters	205.9	210.7	179.2	132.4
Waipapa	78.9	79	50.4	71.6
Te Puna	26	22.5	20.4	19.8

Waikaraka	1.3	1	1.4	1.2
Wairoa	147.5	89.6	105.3	82.2
Waikareao	0.3	0	1.3	0.7
Waimapu	0.01	0	0	0.1
Welcome Bay	0.6	0	2	0.2
Rangataua Bay	36.7	38	25.5	23.5

3.2 Seagrass extent in Tauranga Harbour

Seagrass mapping in Tauranga Harbour has been documented by Park 1999 & 2011, Ha et al., 2020, Crawshaw & Park 2024 and Shao et al., 2024, covering the periods of 1959, 1990, 1996, 2011, and 2014 and from 2018 to 2022. Changes in seagrass extent over time are presented in S2 and illustrated in Figure 5 (slight variations from historical reports may occur due to changes in mapping methodology).

3.2.1 Seagrass extent changes in the northern and southern Harbour

Tauranga Harbour experienced a substantial decline in seagrass extent between 1959 and 1990, with an estimated loss of 49% over the 30 years, particularly in the southern Harbour (Park, 1996). Although there was a short recovery period post-1990, seagrass extent continued to decline, reaching low points between 2001 and 2005. In 2002, the seagrass extent in the northern Harbour fell below that of the southern Harbour for the first time. Following a robust recovery, seagrass extent reached a relatively high level of 3,694 hectares in 2007. However, a substantial decline ensued over the next four years, with the extent reducing to a recorded low of 1,734 hectares by 2011 (Ha et al., 2020), affecting both the northern and southern Harbour areas. Strong seasonal fluctuations were observed from 2019 to 2022 the seagrass extent had increased by over 1000 ha between warm seasons. Despite seasonal fluctuations, a general recovery trend was observed in both the northern and the southern Harbour from 2016 (Figure 5a). Seagrass extent peaked at 4,327 hectares in the summer of 2023, although a noticeable decline occurred afterwards, especially between winter and spring. Trendlines indicated that seagrass areas were expected to continue expanding, predominantly in the northern Harbour.

The spatial mapping results indicated that the most substantial seagrass loss in the Harbour between 1959 and 2024 occurred at the top of the northern harbour and major parts of the southern harbour (Figure 5b). Although the expansion of seagrass extent within the open Harbour areas partially made up for the loss over decades, a renewed decline in the sub-estuaries was observed between 2022 and 2024 (Figure 5c).

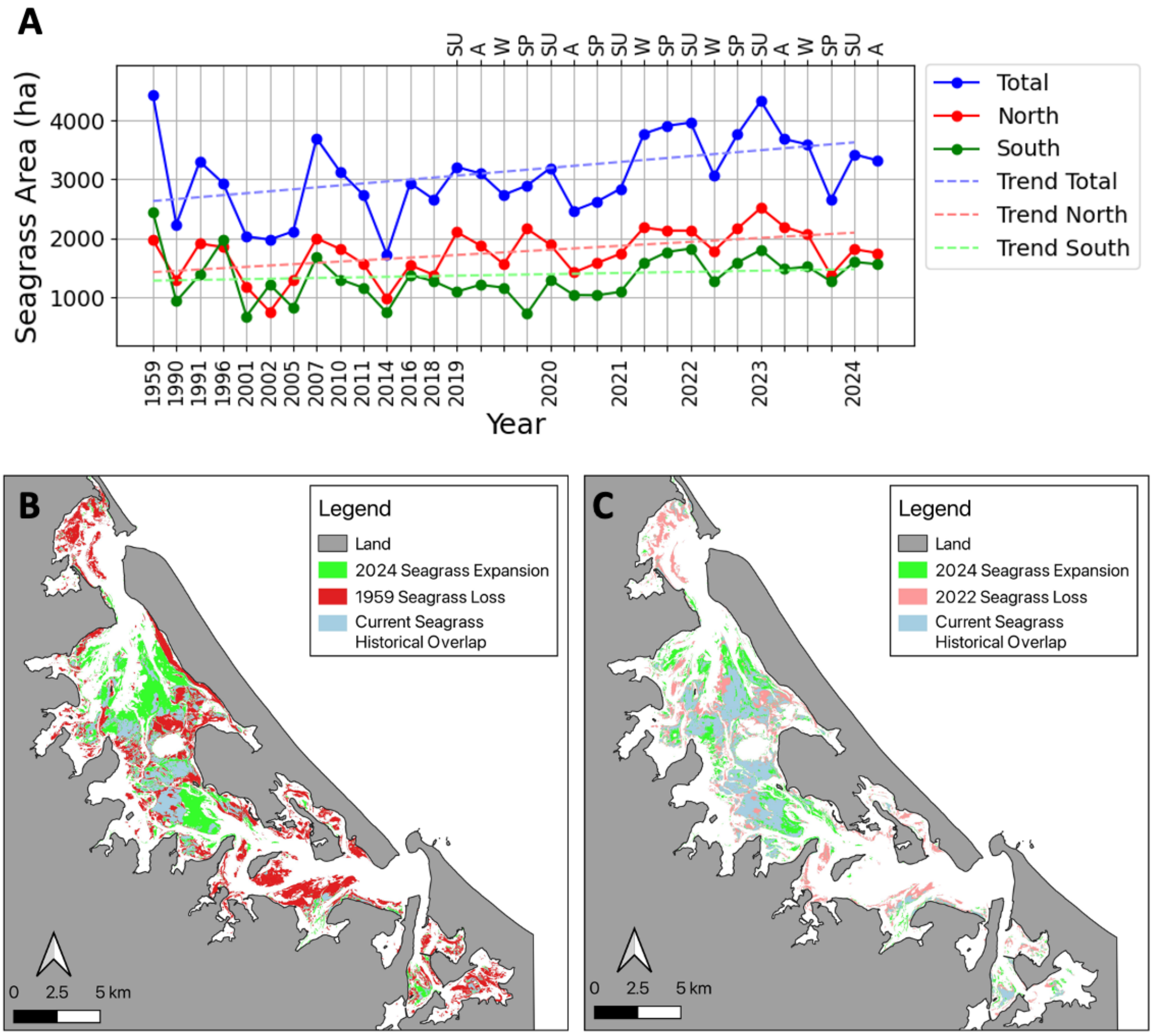

Figure 5 The extent of seagrass from 1959 to 2024 with trendlines (a) and change in seagrass extent between 1959 and 2024 (b) and between 2022 and 2024 (c). The extent of seagrass was calculated seasonally from 2019 (Summer: SU; Autumn: A; Winter: W & Spring: SP). The trendlines were generated based on the seagrass extent during summer each year. Green = most recent expansion, Blue = current seagrass coverage that overlaps the reported historical coverage, Red/Pink = areas of seagrass loss between historical and current extent.

3.2.2 Seagrass extent changes in sub-estuaries and areas

Seagrass extent in the sub-estuaries of Tauranga Harbour experienced significant declines between 1959 and 2024 (Table 5). Aongatete and Waimapu suffered a complete loss of seagrass cover by 2024, with Aongatete showing a decline from 60.8 hectares in 1959 to 0 hectares in the present. Similarly, sub-estuaries like Uretara showed dramatic reductions, with seagrass cover decreasing to just 1.6 hectares in 2024, representing only 11.9% of its 1959 extent. In contrast, other sub-estuaries demonstrated a degree of resilience or recovery in the past two years. Waipapa maintained 100.1% of its 2022 seagrass extent by 2024, while Hunters showed a slight increase, with seagrass cover exceeding its 2022 extent by 102.3%. Rangataua also exhibited a positive trend, with the seagrass extent reaching 103.5% of its 2022 levels by 2024.

Table 5 Seagrass cover (hectares) in Tauranga Harbour sub-estuaries (see Figure 1) from 1959 to 2024 mapping. Percentage shows the 1959 extent compared to 2024. The yellow marked values are after postprocessing for false positive seagrass pixels.

Sub-estuaries	1959 (ha)	2011 (ha)	2022 (ha)	2024 (ha)	% of 1959	% of 2022
Waiau	25.7	14.3	18.5	8.9	34.6	48.1
Tuapiro	9.1	12.2	6.6	2.5	27.5	37.9
Uretara	13.5	4.5	7.2	1.6	11.9	22.2
Blue Gum	215.5	215.5	141.2	79.4	36.8	56.2
Rereatukahia	31.1	8.9	20.3	8.1	26.0	39.9
Aongatete	60.8	0.1	0.1	0.0	0.0	10.0
Mangawhai	53.7	42.5	68.2	33.7	62.8	49.4
Hunters	322.2	179.2	205.9	210.7	65.4	102.3
Waipapa	107.7	50.4	78.9	79.0	73.4	100.1
Te Puna	22.3	20.4	26.0	22.5	100.9	86.5
Waikaraka	3.1	1.4	1.3	1.0	32.3	76.9
Wairoa	269.8	105.3	147.5	89.6	33.2	60.7
Waikareao	6.0	1.3	0.3	0.0	0.2	3.3
Waimapu	0.7	0.0	0.0	0.0	0.0	0.0
Welcome	14.1	2.0	0.6	0.0	0.1	1.7
Rangataua	184.7	25.5	36.7	38.0	20.6	103.5

3.2.3 Intertidal and subtidal seagrass

Although intertidal seagrass remained the dominant contributor to the total seagrass coverage in Tauranga Harbour, the gap between intertidal and subtidal extents was narrowing in recent years due to significant expansion in the middle Harbour (Figure 5 & S2). The intertidal seagrass areas exhibited larger drops and peaks in the area compared to the subtidal seagrass areas, which showed intertidal zones experienced more significant fluctuations in seagrass coverage while the subtidal zones remained relatively stable (S2). The highest intertidal seagrass extent (3540 ha) was observed in 1959 (Park, 1999). Between 1959 and 2014, the extent fluctuated significantly, reaching a low of 1248 hectares in 2014. Since then, intertidal seagrass showed a substantial recovery and reached the peak again at 3482 hectares in the summer of 2023, which nearly matched the extent observed in 1959. Projections based on trend analysis suggested that intertidal seagrass would continue to increase at a slower rate in the coming years. On the other hand, subtidal seagrass experienced a slight increase in extent between 1959 and 1991. However, the subtidal seagrass suffered substantial losses. The extent reduced to 46 hectares in 1996 and further to 30 hectares in 2011. After 2011, subtidal seagrass began to recover and thrive substantially (Figure 5b & c), with the extent reaching a new high of 1138 hectares in the summer of 2024 (Figure 5a). Compared to intertidal seagrass, subtidal seagrass exhibited a more stable increasing trend with fewer fluctuations.

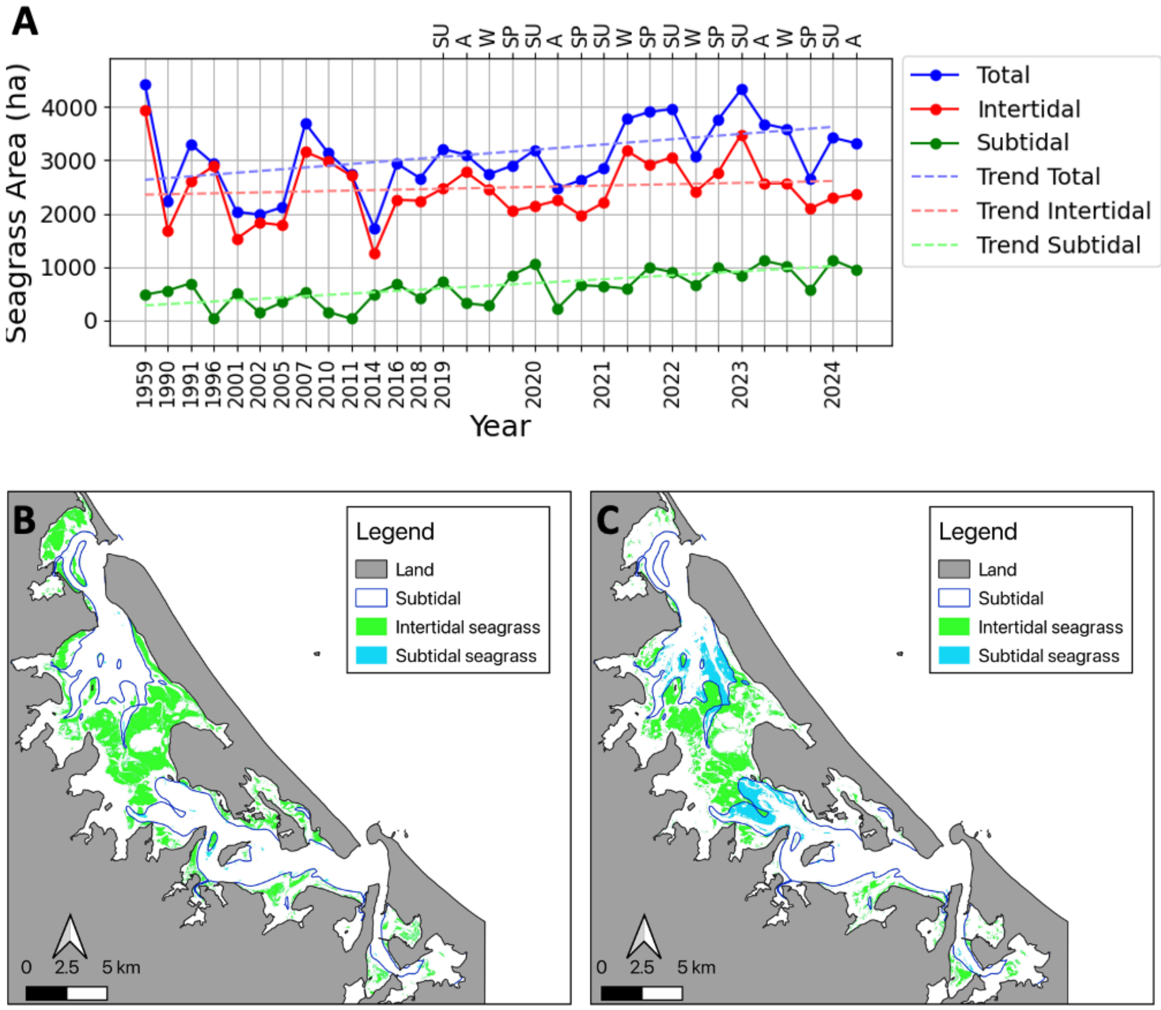

Figure 6 The extent of seagrass in the intertidal and subtidal regions from 1959 to 2024 with trendlines (a) and distribution maps of seagrass in the intertidal and subtidal regions in 2011 (b) and 2024 (c). The extent of seagrass was calculated seasonally from 2019.

3.2.4 Seagrass density

From 2019 to 2024, seagrass density in Tauranga Harbour showed clear hotspots close to the central Harbour (Figure 5). Although seagrass density was relatively uniform in 2021 and 2022, the peak density reached nearly 100% in 2022, particularly within the central Harbour, which corresponded to the year's high seagrass coverage. A general pattern was observed where seagrass density declined from the subtidal/intertidal areas toward the coastline. Spatially, the northern and peripheral regions showed more obvious fluctuations, with some areas experiencing periods of decline followed by potential recovery afterwards. Despite a significant loss in seagrass extent in the southern Harbour, the density remained relatively stable over the six-year period.

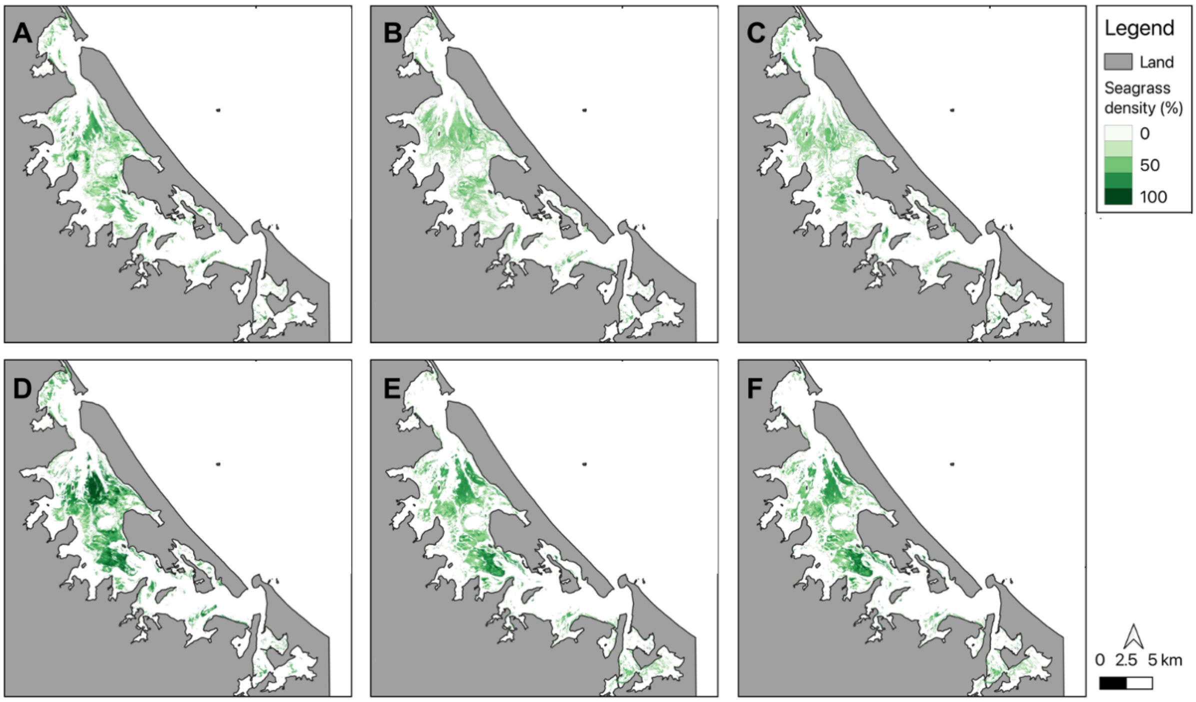

Figure 7 The annual seagrass density distribution maps from 2019 to 2024 (a – f). The maps were derived in the summer of each year.

4. DISCUSSION

This report quantified the seagrass extent and density in Tauranga Harbour from 1959 to 2024 using a machine learning model to classify Landsat and Sentinel-2 imagery as well as aerial photography. Combining existing publications and previous reports, changes in seagrass patterns in the northern/southern and intertidal/subtidal Harbour were derived for the purpose of restoration and management plans.

In this study, supervised classification using the random forest algorithm was used to extract seagrass habitats as well as other land covers. The random forest method is well-suited for handling the complexity and variability inherent in such datasets, as it builds multiple decision trees and merges them to produce a more robust classification. The inclusion of high-resolution satellite imagery (e.g., RapidEyes, Worldview-2 & Worldview-3) and additional relative variables such as DEM/LiDAR in future mapping efforts is expected to further improve classification accuracy, particularly for distinguishing between different coastal habitats like saltmarshes and mangroves (Shao et al., 2024). Despite the general effectiveness of the random forest algorithm, certain challenges remain, such as the potential for misclassification due to spectral similarities between sparse seagrass beds and unvegetated flats and waters with high chlorophyll content. To address this, refining the input features and incorporating advanced techniques, such as bottom reflectance indexes and artificial neural networks, may provide significant improvements in model performance and the precision of habitat mapping (Ha et al., 2021). Compared to manual labelling based on high-resolution aerial photography, supervised classification is more time-efficient and relatively precise. Although extra attention and post-processing are needed for some sub-estuaries, supervised classification with random forest is useful to derive seagrass extent as well as other vegetation covers.

The seagrass in Tauranga Harbour experienced significant changes over time, driven by environmental conditions and human activities. Historically, the decline of seagrass beds was closely linked to increased turbidity, sedimentation, and nutrient loading, all of which reduce the light necessary for photosynthesis. According to previous reports, seagrass coverage is particularly sensitive to sediment grain size in the Harbour. Seagrass becomes rare when mud content exceeds 30% at a site or when muddy sediments cover more than 30% of the estuary area (Crawshaw & Park, 2024). High sediment loads from catchment runoff have exacerbated this decline, highlighting the need for effective land management to reduce sediment inputs (Crawshaw et al., 2023). Analysis of total suspended solids (TSS) data from rivers and streams around the Harbour shows improving trends, signalling progress in recovery efforts (Park, 2016). Conversely, the results presented in this report show a recovery trend for both the northern and the southern Harbour since 2016. It is worth noting that the results of extent recovery can be hindered by climate-related variability, such as years with higher average sediment loading. Recent efforts over the past few years including the implementation of Environmental Plans covering 25,800 hectares of land and the installation of 420 km of new fencing in high-risk sediment zones, have led to improvements in water quality and habitat conditions (Crawshaw & Park, 2024). This has allowed seagrass to recover in areas with lower mud content, aided by the decline of sea lettuce that previously smothered seagrass (Park, 2016).

Since 2019, seagrass in Tauranga Harbour has migrated into subtidal regions, especially in the middle harbour, forming high-density ecological hotspots that are vital for ecosystem functions and carbon sequestration. A key factor in this expansion is the rise in winter water temperatures, which has likely enhanced seagrass growth and resilience. Warmer conditions accelerate photosynthesis and improve nutrient uptake, supporting robust growth (Flowers et al., 2023). However, while moderate warming trends are beneficial, careful management is essential to prevent negative effects, such as heat stress or die-offs (potentially happened between winter and spring in 2022 and 2023), ensuring the continued health and expansion of these critical marine ecosystems.

Challenges remain, particularly in sub-estuaries where high mud loading from the catchment continues to restrict seagrass recovery. To ensure the long-term health and expansion of seagrass habitats, continued focus on improving water clarity by reducing sedimentation through targeted interventions remains essential, supporting both biodiversity and the Harbour's role in coastal carbon sequestration. Implementing artificial shading structures in vulnerable areas during extreme heat events and promoting the restoration of adjacent ecosystems like mangroves can significantly buffer temperature fluctuations and enhance the resilience of seagrass habitats.

5. ACKNOWLEDGEMENTS

We would like to acknowledge Dr Josie Crawshaw and Dr Stephen Park and all other scientists from the Bay of Plenty Regional Council, the University of Waikato and the University of Auckland who were involved in seagrass labelling and sample collection. Their dedicated effort made it possible to have this report.

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7. APPENDIX

S1 The relative difference between automated classification and manual delineation

The greatest difference (>1) mainly existed in the shallow waters of the sub-estuaries with relatively high chlorophyll concentrations (Figure 6). The supervised classification model tended to misclassify these areas because the chlorophyll in the water column has a similar spectral signature to seagrass areas. Therefore, seagrass mapping in these regions needed post-processing after classification to exclude the non-seagrass area based on the original satellite images. However, the supervised classification model is generally sensitive to all potential pixels with the signatures of seagrass. This reduced the possibility of missing individual or less patchy seagrass pixels, especially in the regions where manual labelling usually ignored the species (e.g. deep water), which also resulted in higher relative difference (see Figure 6).

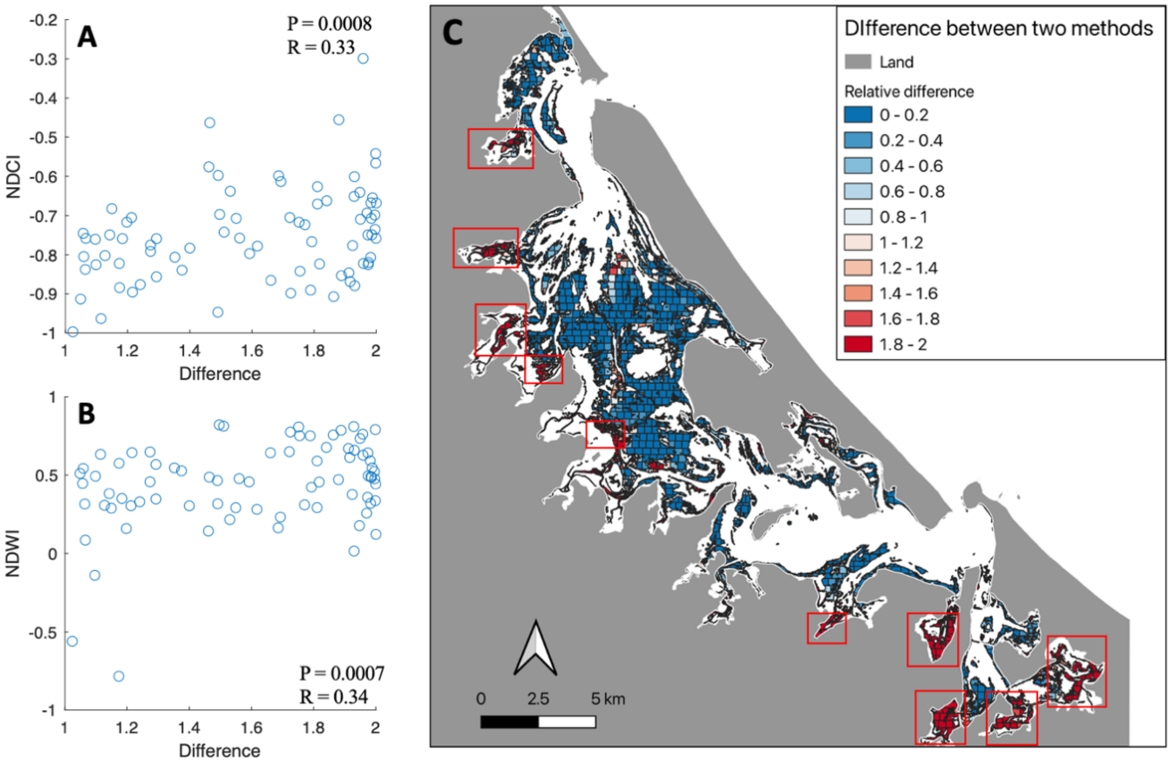

Figure 8 The correlation between NDCI, NDWI and difference of seagrass extent in the sub-estuaries from two methods (a & b) as well as the spatial distribution of relative difference map in the whole harbour.

S2 Seagrass extent changes from 1959 to 2024

Table 6 The seagrass extent in the northern and southern Tauranga Harbour from 1959 to 2024.

Year	Season	Data source	Total (ha)	North (ha)	South (ha)
1959	Summer	Aerial photo	4424	1978	2446
1990	Summer	Landsat-4	2237	1297	940
1991	Summer	Landsat-4	3302	1915	1387
1996	Summer	Aerial photo	2937	1855	1982
2001	Summer	Landsat-5	2035	1180	685
2002	Summer	Landsat-5	1982	763	1219
2005	Summer	Landsat-5	2122	1294	828
2007	Summer	Landsat-5	3694	2004	1690
2010	Summer	Landsat-5	3133	1824	1309
2011	Summer	Aerial photo	2740	1572	1168
2014	Summer	Landsat-8	1734	988	746
2016	Summer	Landsat-8	2936	1550	1386
2018	Summer	Sentinel-2	2665	1386	1279
2019	Summer	Sentinel-2	3210	2108	1102
2019	Autumn	Sentinel-2	3102	1884	1218
2019	Winter	Sentinel-2	2739	1569	1170
2019	Spring	Sentinel-2	2897	2166	731
2020	Summer	Sentinel-2	3193	1893	1300
2020	Autumn	Sentinel-2	2472	1427	1045
2020	Spring	Sentinel-2	2627	1586	1041
2021	Summer	Sentinel-2	2843	1744	1099
2021	Winter	Sentinel-2	3774	2186	1588
2021	Spring	Sentinel-2	3907	2133	1774
2022	Summer	Sentinel-2	3962	2133	1830
2022	Summer	Aerial photo	3740	1998	1742
2022	Winter	Sentinel-2	3065	1792	1273
2022	Spring	Sentinel-2	3763	2168	1595
2023	Summer	Sentinel-2	4327	2519	1808
2023	Autumn	Sentinel-2	3683	2199	1484
2023	Winter	Sentinel-2	3591	2067	1524
2023	Spring	Sentinel-2	2657	1382	1275
2024	Summer	Sentinel-2	3423	1818	1605
2024	Autumn	Sentinel-2	3320	1755	1565

Table 7 The seagrass extent in the subtidal and intertidal regions of Tauranga Harbour from 1959 to 2024 with area difference between two continuous seasons.

Year	Season	Total (ha)	Subtidal (ha)	Intertidal (ha)	Difference between seasons (Subtidal) (ha)	Difference between seasons (Intertidal) (ha)
1959	Summer	4424	3945	479	-	-
1990	Summer	2237	1678	559	-	-
1991	Summer	3302	2609	693	-	-
1996	Summer	2937	2891	46	-	-
2001	Summer	2035	1526	509	-	-
2002	Summer	1982	1834	148	-	-
2005	Summer	2122	1782	340	-	-
2007	Summer	3694	3158	536	-	-
2010	Summer	3133	2976	157	-	-
2011	Summer	2740	2710	30	-	-
2014	Summer	1734	1248	486	-	-
2016	Summer	2936	2258	678	-	-
2018	Summer	2665	2242	423	-	-
2019	Summer	2473	737	1736	-	-
2019	Autumn	2782	320	2462	-417	726
2019	Winter	2464	275	2189	-45	-273
2019	Spring	2051	846	1205	571	-984
2020	Summer	2139	1054	1085	208	-120
2020	Autumn	2245	227	2018	-827	933
2020	Spring	1970	657	1313	-	-
2021	Summer	2205	638	1567	-19	254
2021	Winter	3184	590	2594	-	-
2021	Spring	2914	993	1921	403	-673
2022	Summer	3060	903	2157	-90	236
2022	Winter	2404	661	1743	-242	-414
2022	Spring	2774	989	1785	328	42
2023	Summer	3482	845	2637	-144	852
2023	Autumn	2565	1118	1447	273	-1190
2023	Winter	2570	1021	1549	-97	102
2023	Spring	2095	562	1533	-459	-16
2024	Summer	2290	1133	1157	571	-376
2024	Autumn	2366	954	1412	-179	255

S3 Seagrass extent from winter 2022 to summer 2024

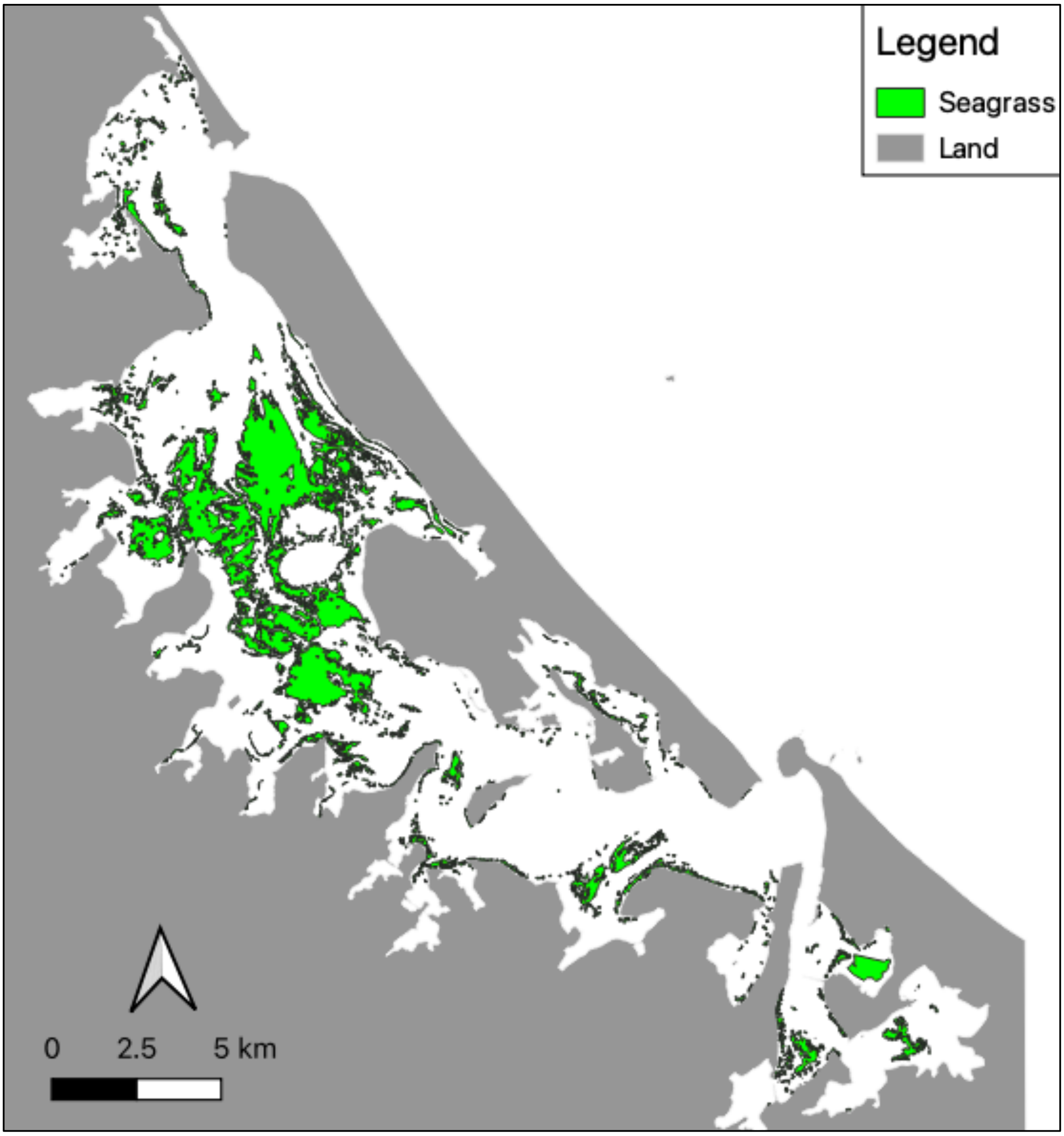

Figure 9 Seagrass extent in Tauranga Harbour in winter 2022.

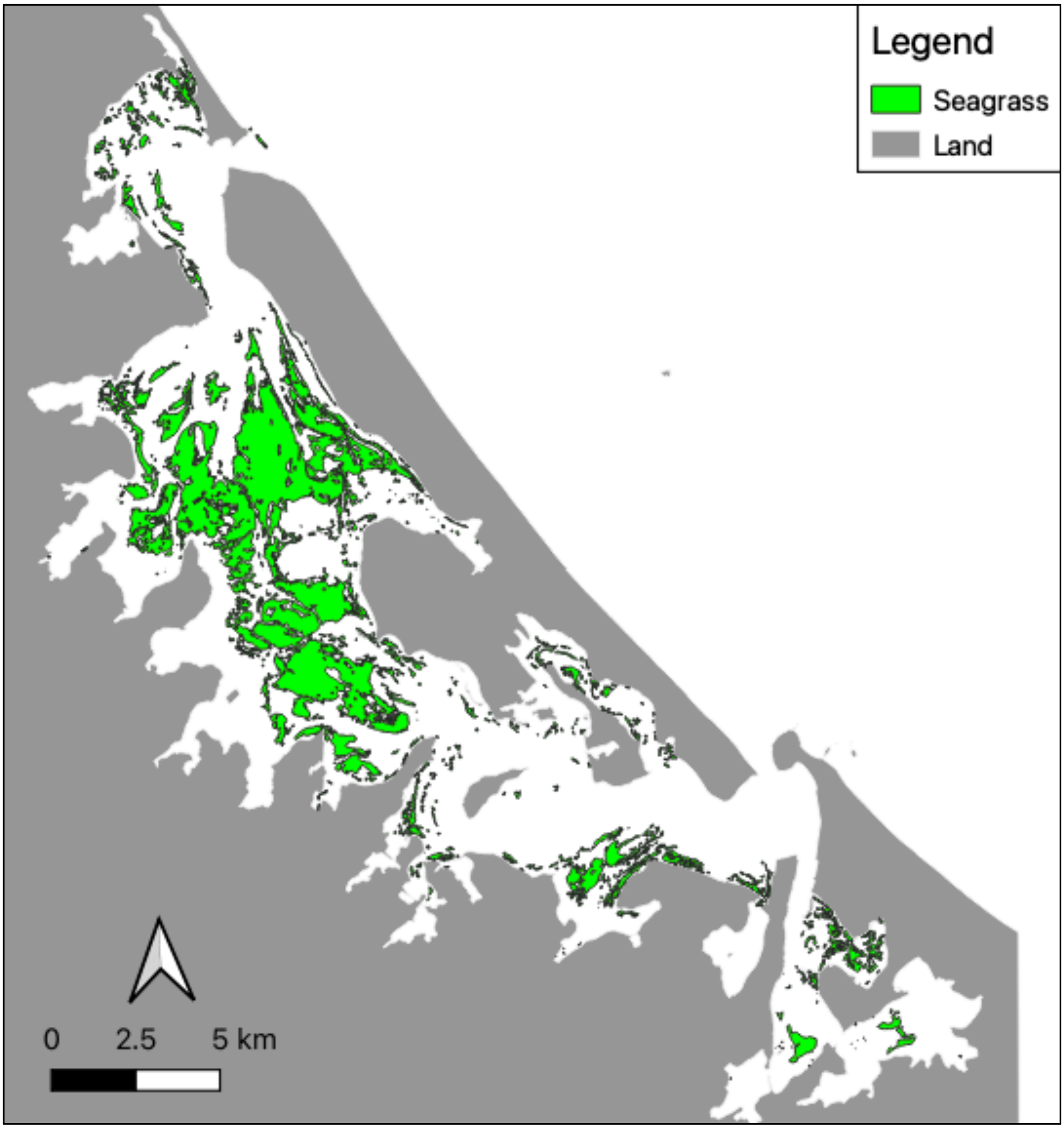

Figure 10 Seagrass extent in Tauranga Harbour in spring 2022.

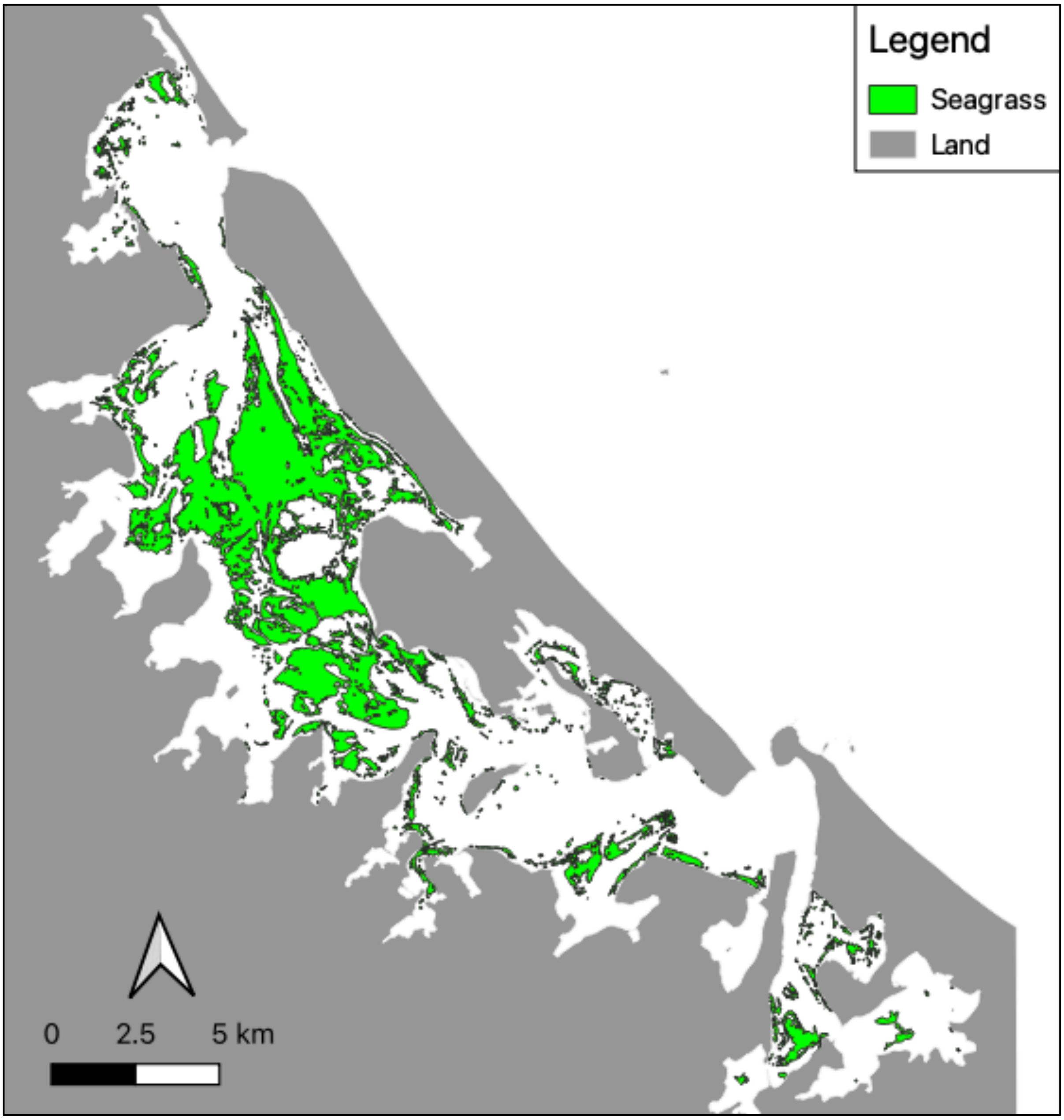

Figure 11 Seagrass extent in Tauranga Harbour in summer 2023.

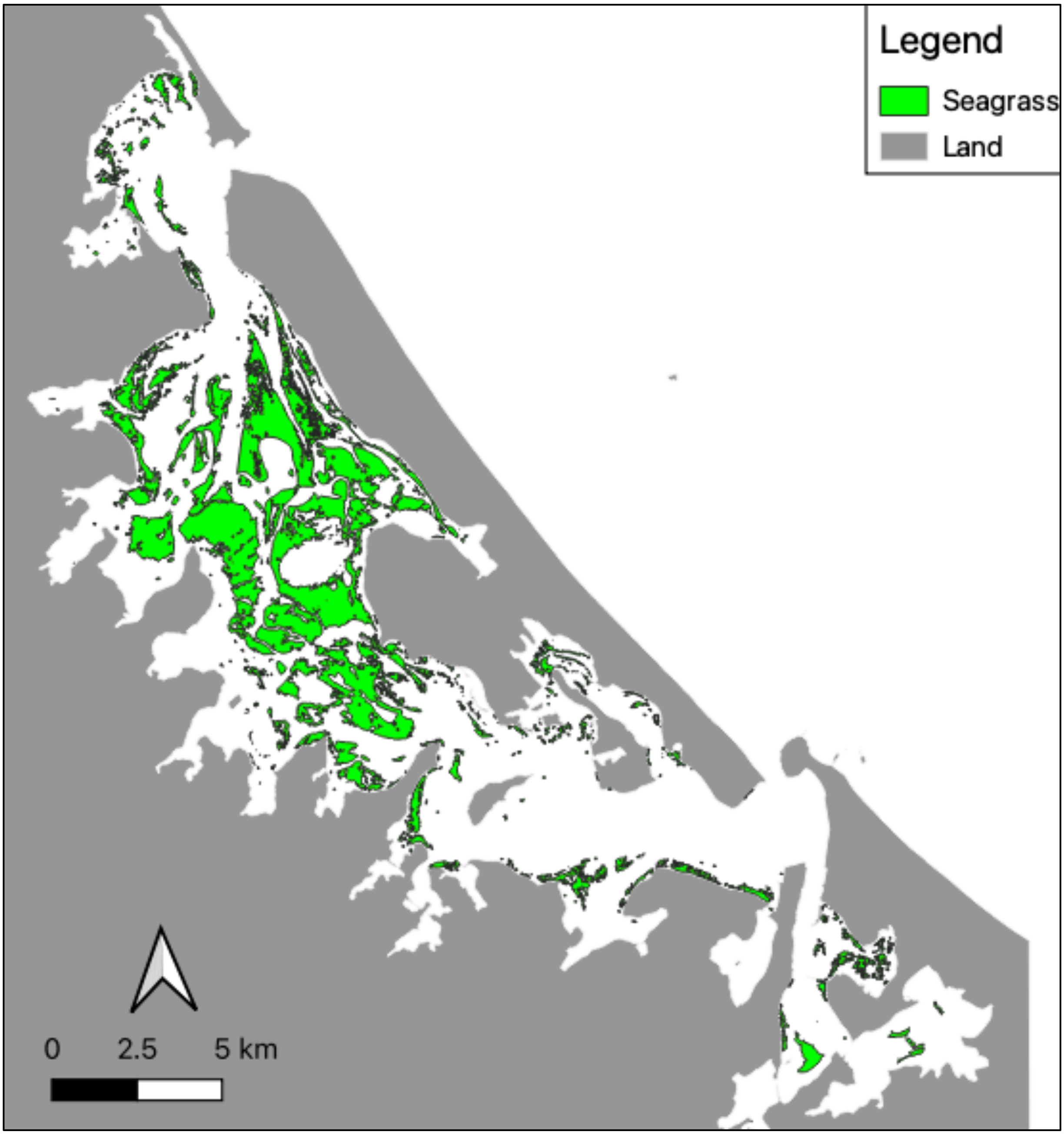

Figure 12 Seagrass extent in Tauranga Harbour in summer 2023.

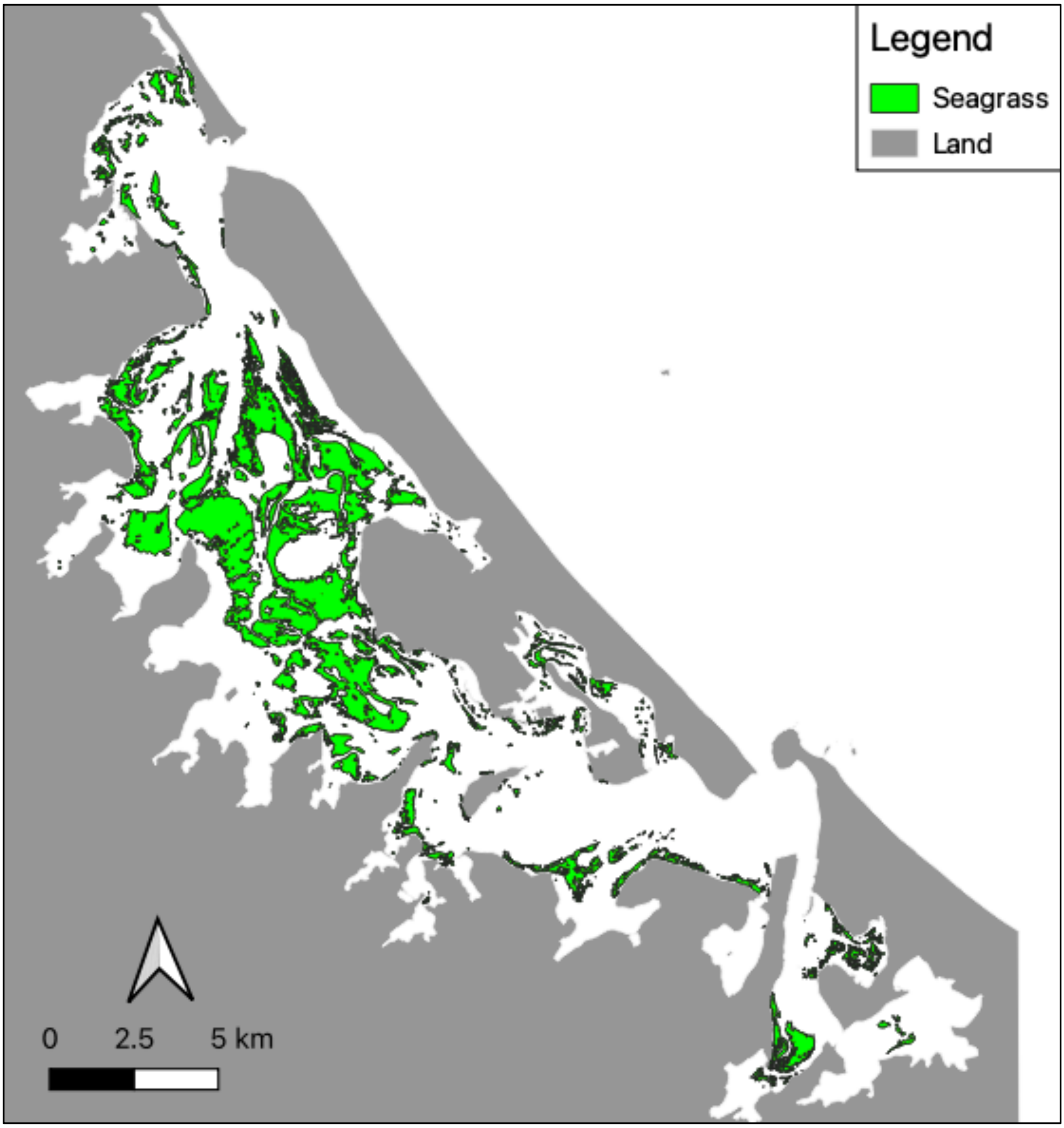

Figure 13 Seagrass extent in Tauranga Harbour in winter 2023.

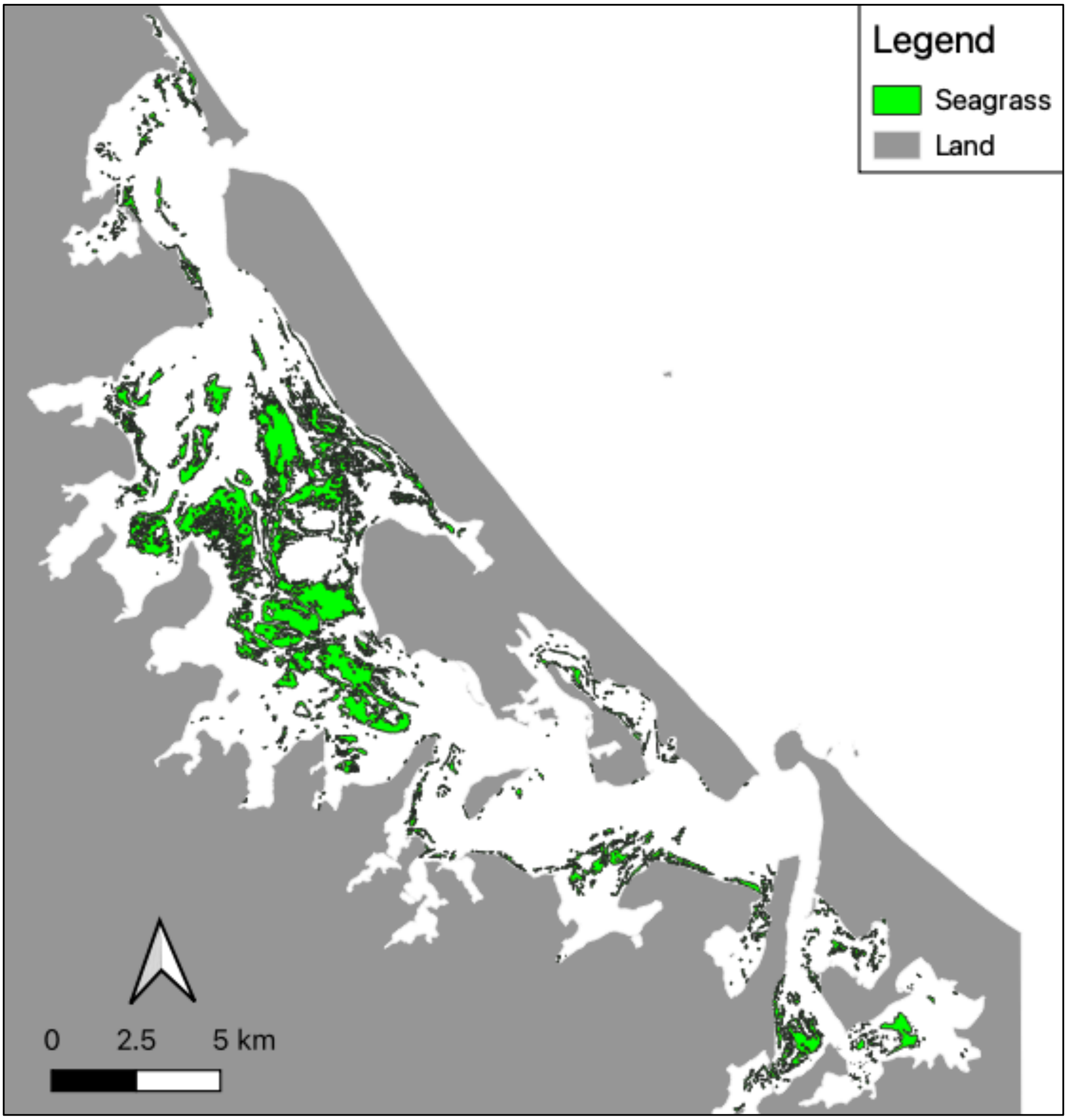

Figure 14 Seagrass extent in Tauranga Harbour in spring 2023.

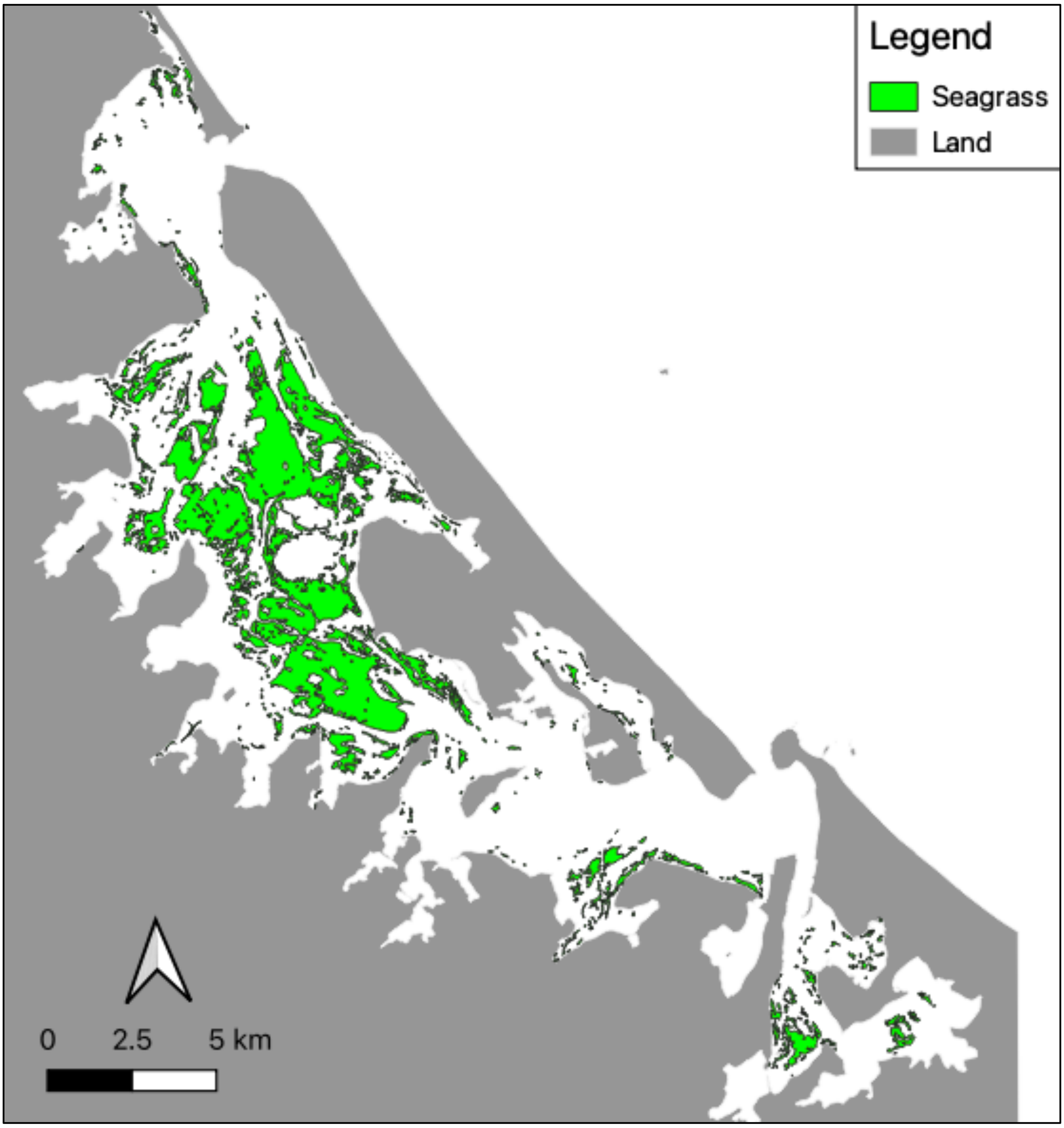

Figure 15 Seagrass extent in Tauranga Harbour in summer 2024.

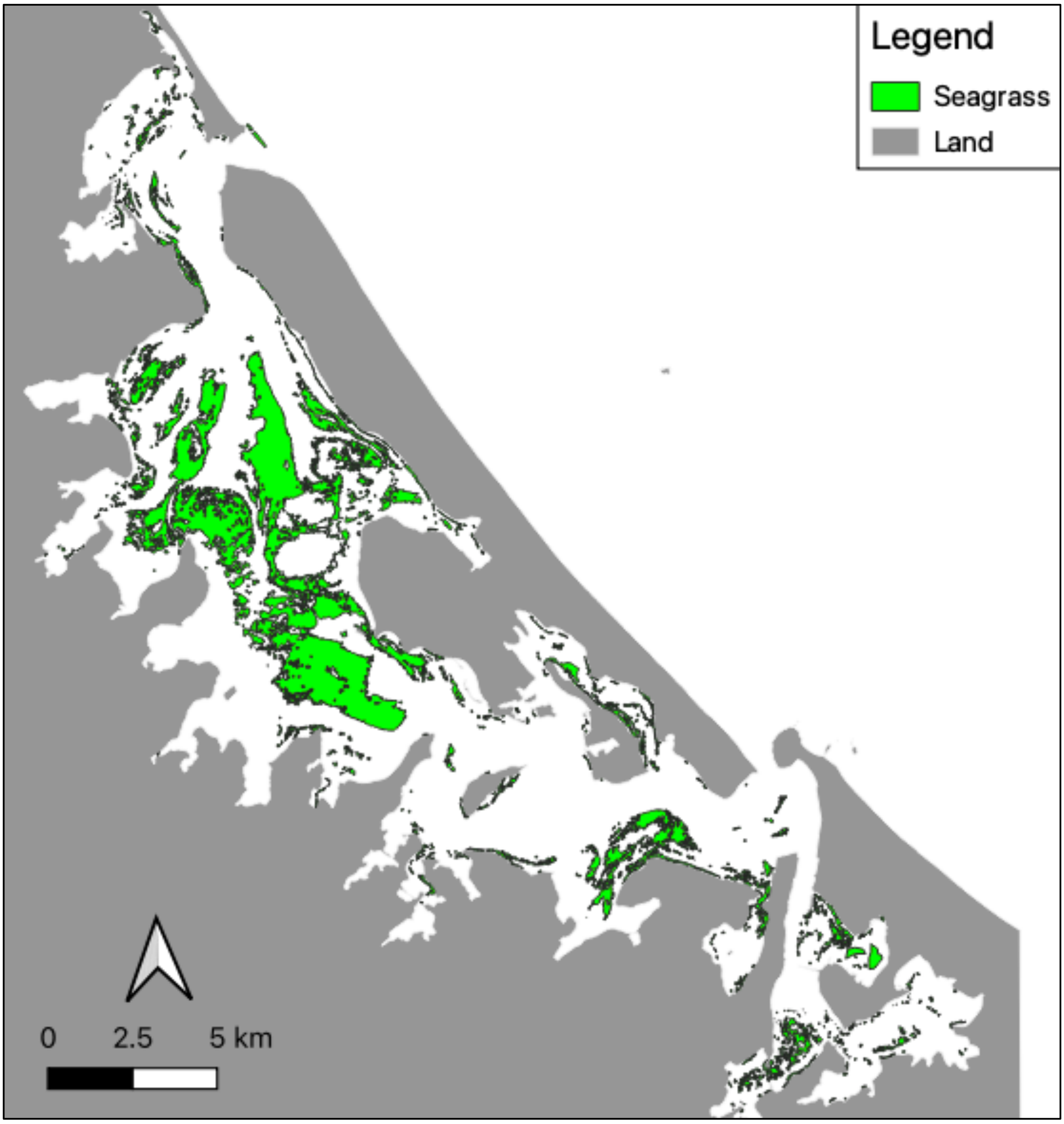

Figure 16 Seagrass extent in Tauranga Harbour in autumn 2024.