Fisheries New Zealand

Tini a Tangaroa

Summarising and updating knowledge on the distribution of kina barrens in key regions of Aotearoa New Zealand

New Zealand Aquatic Environment and Biodiversity Report No. 365

A. J. P. Spyksma

ISSN 1179-6480 (online)
ISBN 978-1-991380-90-6 (online)

September 2025

Te Kāwanatanga o Aotearoa
New Zealand Government

Disclaimer

This document is published by Fisheries New Zealand, a business unit of the Ministry for Primary Industries (MPI). The information in this publication is not government policy. While every effort has been made to ensure the information is accurate, the Ministry for Primary Industries does not accept any responsibility or liability for error of fact, omission, interpretation, or opinion that may be present, nor for the consequence of any decisions based on this information. Any view or opinion expressed does not necessarily represent the view of Fisheries New Zealand or the Ministry for Primary Industries.

Enquiries should be directed to:

Fisheries Science Editor
Fisheries New Zealand
Ministry for Primary Industries
PO Box 2526
Wellington 6140
NEW ZEALAND

Email: Fisheries-Science.Editor@mpi.govt.nz
Telephone: 0800 00 83 33

This publication is also available on the Ministry for Primary Industries websites at:
http://www.mpi.govt.nz/news-and-resources/publications
http://fs.fish.govt.nz go to Document library/Research reports

© Crown Copyright – Fisheries New Zealand

Please cite this report as:
Spyksma, A.J.P. (2025). Summarising and updating knowledge on the distribution of kina barrens in key regions of Aotearoa New Zealand. *New Zealand Aquatic Environment and Biodiversity Report No. 365.* 45 p.

TABLE OF CONTENTS

PLAIN LANGUAGE SUMMARY

- Kelp forests are highly productive ecosystems but are increasingly threatened by overgrazing from sea urchins, creating persistent "urchin barrens."
- In New Zealand, kina is the main urchin barren forming sea urchin, with fishing pressure on predators like snapper and rock lobster considered a key driver of urchin barren formation, particularly in northeastern New Zealand.
- Despite widespread reporting of urchin barren habitats there has been no previous efforts to collate urchin barren distribution information across the country.
- This study collated 27 data sources to produce a nationwide overview of urchin barren distribution and extent.
- Barrens occur throughout the country, with large areas in northeastern New Zealand, Marlborough Sounds, Tasman Bay, Fiordland, and Stewart Island.
- Additional aerial imagery mapping revealed barrens along ~36% of northeastern NZ's rocky coastline, mostly on shallow, nearshore reefs, though estimates are conservative.
- The resulting datasets provide a national starting point for future work and will help to inform management, track kelp forest recovery, and support mana whenua and community involvement in restoration and management initiatives.

EXECUTIVE SUMMARY

Spyksma, A.J.P.[1] (2025). Summarising and updating knowledge on the distribution of kina barrens in key regions of Aotearoa New Zealand.

***New Zealand Aquatic Environment and Biodiversity Report No. 365*. 45 p.**

Kelp forests are highly biodiverse and productive ecosystems but are in global decline, with sea urchin overgrazing a major driver of loss. In Aotearoa New Zealand, urchin barrens – areas of rocky reef that are largely devoid of macroalgae due to the grazing impact of sea urchins - are widespread in some regions but to date there has been no attempt to collate information documenting their distribution or extent. Additionally in northeastern New Zealand, where there is strong evidence that fishing of key urchin predators (snapper and rock lobster) is a key driver of the patterns in distribution and extent of urchin barrens, developing a region wide understanding of the current distribution and extent of urchin barrens habitat is essential for evaluating the impact of changes to fisheries regulations and other management initiatives aimed at restoring predator populations and facilitating kelp forest recovery in areas of urchin barren.

The aims of this study were to summarise and update knowledge on the distribution of urchin barrens in key regions of New Zealand, with three key objectives:

1. Collate quantitative data on the historical and current distribution of urchin barrens across New Zealand.
2. Map the spatial extent of urchin barrens across northeastern New Zealand.
3. Develop an updatable geodatabase and geospatial layers containing information collected and developed as part of Objectives 1 and 2.

A comprehensive literature review was undertaken to collate information on the historical and current distribution of urchin barrens across New Zealand, drawing upon freely accessible information contained within published scientific articles and technical reports. Additional information was also sourced from unpublished data. Key data source requirements included information on the location and extent of urchin barren habitat. From these data sources a geodatabase was created with rich metadata outlining the data sources, usage rights and key references.

In addition, a habitat mapping exercise was undertaken throughout northeastern New Zealand, from Cape Reinga to East Cape. This was specifically designed to map the current spatial extent of urchin barren habitat. We utilised freely available imagery collected between 2023 – 2024 combined with existing drop camera ground truthing data to conduct this exercise with accuracy assessments and comparisons with previous studies incorporated into the survey design and analysis.

We found 27 suitable data sources containing information on the spatial location and extent of urchin barren habitats throughout New Zealand. This data was mostly in the form of detailed habitat maps and was heavily weighted towards studies conducted in northeastern New Zealand. Data indicated that urchin barrens, primarily associated with the endemic sea urchin ‘kina’ (*Evechinus chloroticus*), are distributed across the country with areas of high urchin barren extent occurring in northeastern New Zealand, the Marlborough Sounds, Tasman Bay, Fiordland and Stewart Island. Time series information showed that in northeastern New Zealand barrens were not a common feature in the 1940s and 1950s but had become common on many reefs by the 1970s and have continued to be a major component of shallow reef ecosystems since. Across other parts of New Zealand time series information is sparse, but the data indicates that urchin barrens have been a feature on many reefs since at least the 1990s. Habitat maps of the Cape Rodney – Okakari Point Marine Reserve support

[1] University of Auckland

the role of predators in controlling sea urchin populations with a decline in urchin barren extent within the reserve following protection. This is reflected in other northeastern New Zealand marine reserves which have tended to have less urchin barren habitat within than in the surrounding fished waters.

Across northeastern New Zealand we evaluated approximately 5875 km of coastline, of which about 1677 km was considered suitable for the potential presence of urchin barren habitat (e.g. coastline with adjacent rocky reef habitat). Urchin barrens habitat was recorded along approximately 36% of suitable coastline and we were able to identify 22 – 28 km² of urchin barrens habitat across the region. Most barrens mapped were in shallow water (<10 m deep) and within 200 m of the coastline. Accuracy assessments, comparing aerial mapping to ground truthing data, indicated high accuracy within those areas mapped as urchin barrens. However, the area of urchin barrens mapped throughout the region is likely to be an underestimate of total area due to limited availability of suitable imagery in some regions (i.e. poor sea conditions and water clarity) and limitations with this aerial-image based mapping approach for accurately mapping deeper areas of urchin barrens (>10 m). Because of this we chose to use the proportion of coastline with adjacent urchin barrens relative to total coastline as the primary way of evaluating urchin barren extent rather than an estimate of total area. Comparisons with previous mapping (2016–2020) showed general consistency with this approach to quantifying urchin barren extent.

The results of this study confirm the widespread presence of urchin barrens throughout New Zealand and support the formation of urchin barrens in northeastern New Zealand occurring after the onset of intensive fishing for urchin predators in the 1950s and 1960s. Current estimates show that urchin barrens are a common and widespread component of shallow rocky reef throughout this region. The geodatabases created as part of this project compiles available information on the known extent and distribution of urchin barren habitat across New Zealand and will serve as a starting point for future work, help guide and evaluate management actions aimed at restoring kelp forest ecosystems and provide an initial source of freely available information about the distribution and extent of urchin barrens for mana whenua and community groups wanting to be actively involved in managing their marine spaces.

Future research to further quantify the distribution and extent of urchin barren habitats could include:

- Repeating the broad-scale northeastern New Zealand mapping exercise on a 10–15 years basis.
- Conducting more detailed mapping to understand urchin barren extent within key areas where management actions are being considered.
- Supporting the collection and integration of high-resolution multibeam bathymetry data into habitat mapping of key areas.
- Conducting a national diver-based survey to provide up-to-date information on the current distribution and extent of urchin barren habitat.
- Funding research to integrate local ecological knowledge (LEK) into barren mapping and to help establishing timelines for when urchin barrens first appeared.

1 INTRODUCTION

Kelp forests are one of the most biodiverse and productive habitat types on earth (Steneck et al. 2002; Eger et al. 2023), yet in many regions significant declines have occurred over the past several decades (Krumhansl et al. 2016). On a global scale one of the primary drivers of kelp forest loss is uncontrolled herbivory by sea urchins (Steneck 2020). Where sea urchin numbers are able to proliferate, excessive grazing pressure on kelps and other canopy forming macroalgae can cause abrupt, catastrophic shifts from a complex, forested reef ecosystem to a comparatively depauperate state known as an 'urchin barren' which is characterised by lower productivity and biodiversity (Ling et al. 2015; Eger et al. 2024). This undesirable urchin barren state can be extremely stable, with widespread evidence of multi-decadal persistence (Filbee-Dexter & Scheibling 2014).

In many systems the ongoing overabundance of sea urchin populations and persistence of urchin barren habitat can be linked, at least in part, to overfishing of key sea urchin predators (Steneck 2020). A wide range of marine organisms are considered as important sea urchin predators and these can play both a direct role in regulating sea urchin populations through consumption (e.g. Tegner & Levin 1983; Sala & Zabala 1996) and indirect role via the modification of grazing behaviour (e.g. Spyksma et al. 2017; Mancuso et al. 2025). Where key predators are significantly reduced, or lost from the system, due to intensive fishing pressure sea urchin populations can proliferate and behavioural transitions occur, prompting a more active grazing approach, which can result in large scale loss of canopy forming kelp and other macroalgae species.

In Aotearoa New Zealand there are two sea urchin species known to be capable of forming urchin barrens, the endemic *Evechinus chloroticus* (kina) and native *Centrostephanus rodgersii*. *Evechnius chloroticus* occurs throughout New Zealand (Shears & Babcock 2007) while *C. rodgersii*, is currently only known to occur in northeastern New Zealand (Thomas et al. 2021) and despite long-term persistence throughout this region has recently undergone a rapid increase in population (Balemi & Shears 2023). While the presence of urchin barrens associated with *E. chloroticus* have been documented throughout several regions (Grace 1983; Villouta et al. 2001; Udy et al. 2019; Lafont & Shears 2025) most attention has focussed on northeastern New Zealand, where extensive observations and experimental research associated with *E. chloroticus* and 'kina' barrens has occurred over the past several decades and there have been strong links between barren formation and loss of key predators. Earliest descriptions of destructive overgrazing by kina in northeastern New Zealand date to the late 1950s (Dromgoole 1964) and coincided with intensification of fishing efforts for red rock lobster (*Jasus edwardsii*) and snapper (*Chrysophrys auratus).* Both species are considered as important predators of *E. chloroticus* (Andrew & Macdiarmid 1991; Shears & Babcock 2002; Marinovich et al. 2025) and have undergone significant declines in abundance due to intense commercial and recreational harvesting (Parsons et al. 2009; LaScala-Gruenewald et al. 2021). On rocky reefs open to fishing, overgrazing by *E. chloroticus* typically presents as the formation of a distinct band of barrens, with high densities of urchins actively grazing the reef, between 2 – 9 m deep (Grace 1983; Shears & Babcock 2004), though these depth limits vary with environmental conditions (Shears et al. 2008). 'Kina' barrens were common on shallow reefs throughout the region by the late 1970s (Ballantine et al. 1973; Ayling et al. 1981; Berben et al. 1988; Dartnall 2022) with examinations of historical imagery from the 1940s and 1950s supporting much of the initial large-scale barren formation occurring between the late 1950s and 1970s (Dartnall 2022; Kerr et al. 2024). In contrast, marine protected areas (MPA) such as the Cape Rodney – Okakari Point Marine Reserve showed the opposite trend, with areas of rocky reef characterised as 'kina' barrens in the 1970s recovering to kelp forest by the late 1990s following predator recovery (Shears & Babcock 2003) and remaining as stable kelp forests since then (Peleg et al. 2023).

In addition to kelp loss caused by *E. chloroticus*, recent proliferations of *C. rodgersii* throughout northeastern New Zealand, particularly at offshore island groups are contributing to the overall extent of urchin barrens present within the region (Balemi & Shears 2023). While *C. rodgersii* is native to New Zealand it has historically persisted at low densities, with limited ecological impact (Schiel 1984; Berben et al. 1988). Warming water temperatures over the past 25 years combined with a lack of natural predators (namely large rock lobster (Ling & Johnson 2012; Smith et al. 2023)) are likely to be driving large population increases and we are beginning to see signs of *C. rodgersii* barren formation as a consequence (Balemi & Shears 2023; Balemi et al. 2025).

The most recent estimates suggest that 30% of shallow rocky reefs open to fishing along the mainland coast of northern New Zealand (from the Tāwharanui Peninsula north) can be characterised as urchin barren formed as a consequence of overgrazing by *E. chloroticus* and/or *C. rodgersii* (Kerr et al. 2024), though in some places these estimates are based on data collected about 20 years ago. Throughout the wider northeastern New Zealand region there is little understanding of the current extent of urchin barrens, with generalisations made by Shears and Babcock (2004) about where they occur in the most recent region-wide assessment.

Similarly, although the drivers of kelp forest loss in other parts of the country are more heavily debated (Schiel 2013) overgrazing by sea urchins clearly plays an important role (Udy et al. 2019; Wing et al. 2022; Lafont & Shears 2025) in some places such as Queen Charlotte Sound/ Tōtaranui where urchin barrens are a dominant rocky reef habitat type (Udy et al. 2019; Wing et al. 2022; Lafont & Shears 2025). Despite the apparent widespread appearance of urchin barrens across New Zealand there has been no concerted effort to bring together information on the current or historical distribution of urchin barren habitat across the country. This information is highly relevant for informing decision making aimed at establishing greater marine protection or implementing active restoration actions such as sea urchin removals (Miller et al. 2022). It is also important for fisheries management, particularly in northeastern New Zealand where a judicial review in October 2023 found that future catch settings for rock lobster within the CRA1 Fishery Management Area (Kaipara Harbour – Te Arai Point) must take into consideration the potential ecological effects of fishing, especially in relation to the regulatory role of lobster on sea urchin populations. Further to this in April 2025 the inner Hauraki Gulf, part of the CRA2 Fishery Management Area (Te Arai Point – East Cape), was closed to all lobster fishing for a minimum of three years to help heavily depleted stocks recover. Here regionwide baseline information on the spatial extent of urchin barren habitat would be beneficial for quantifying the efficacy of management measures such as reduced fisheries allowances or complete fisheries closure on reducing the overall the extent of urchin barren habitat.

1.1 OBJECTIVES

The purpose of this report is therefore to summarise and update knowledge on the distribution of urchin barrens in key regions of New Zealand. To meet this aim three key objectives were set out:

1. Collate quantitative data on the current and historical distribution of urchin barrens in New Zealand
2. Collect comprehensive baseline information on the extent of urchin barrens in northeastern New Zealand
3. Create a series of updatable geospatial layers from the information gathered/created as parts of Objective 1 and Objective 2.

For Objective 1 we conducted a comprehensive literature search for all available information on urchin barren habitats through New Zealand. Using all information sources that contained relevant spatial information on urchin barren location and extent, we then created an updateable urchin barren distribution geodatabase.

For Objective 2 we undertook a northeastern New Zealand wide mapping exercise to provide a current estimate of the extent of urchin barrens across the entire region. This approach leveraged freely available aerial imagery as the primary data source and used drop camera imagery to validate mapping results.

In the sections below we outline the methodological approaches used for Objectives 1 and 2 and discuss key findings of both to help with interpretation of the associated geospatial layers created. Finally, we discuss these key findings along with limitations of the chosen approach and key recommendations for additional or ongoing work associated with quantifying the distribution and extent of urchin barren habitat in northeastern New Zealand.

2 METHODS

2.1 Collate quantitative data on the current and historical extent of urchin barrens in New Zealand

Several studies have been conducted across the country over the past several decades which contain specific information on the extent of urchin barren habitat, however, there has been no formal attempts to compile these into an indexed geodatabase.

For this objective we carried out a comprehensive literature search for New Zealand wide studies referencing urchin barrens. This included a general search on Google/Google Scholar, the University of Auckland library databases and direct communication with researchers across the country who have worked on/are working on sea urchin related projects. Although there are a large number of studies that reference the presence of sea urchin, and/or urchin barrens, many lack the required spatial information for addition to a geodatabase. We therefore limited our results to those studies containing spatial information to quantify areas of reef as urchin barrens, or other relevant habitat types, at the time of the study.

Once compiled these layers, and the relevant spatial information were added to a geodatabase set up in ArcGIS. As each layer was added metadata was created detailing the study, the study authors, any use limitations and the spatial extent of the data. It should be noted that a number of the data layers within the geodatabase are from currently unpublished studies and that the authors, while happy to contribute their data to the geodatabase, would like to be consulted prior to data being used beyond the scope of this project. Usage terms and contact details for each study included in the geodatabase can be found in the metadata.

This section (and subsections) should provide the geographical and physical setting of the research, describe survey design, sampling methods, and so on, depending on the nature of the project. It is important that any statistical techniques and analytical methods be fully explained (if new) or referenced.

2.2 Collect comprehensive baseline information on the extent of urchin barrens in northeastern New Zealand

2.2.1 Methodological rationale

To complement the New Zealand wide geodatabase of existing urchin barren studies, comprehensive baseline information on the current extent of urchin barrens in northeastern New Zealand (Cape Reinga – East Cape, see Figure 1) was required. To quantify urchin barrens over such a large geographic area an approach that utilised aerial/satellite imagery in combination with ground truthing information was considered the most practical/cost effective.

The use of aerial imagery as the basis for mapping subtidal marine ecosystems has become increasingly common globally as access to high quality, georeferenced and orthorectified imagery from unmanned aerial vehicle, conventional aircraft and satellites has increased (Borfecchia et al. 2019; Ventura et al. 2023). While mapping advancements have largely focussed on tropical regions with clear waters that are conducive to the use of aerial imagery there has also been considerable progress made in temperate regions as well (e.g. St-Pierre & Gagnon 2020). This has included several recent projects in northeastern New Zealand that have used imagery for mapping areas of kelp forest and urchin barrens (Kibele 2017; Lawrence 2019; Dartnall 2022; Kerr et al. 2024). These studies have varied in their methodological approach, with manual mapping of habitats (Dartnall 2022; Kerr et al. 2024) and automated mapping processes (Kibele 2017; Lawrence 2019) applied to quantify the spatial extent of subtidal rocky reef habitats from aerial imagery. In most cases ground-truthing information has been used to validate habitat types and ensure that mapping procedures were robust and reliable.

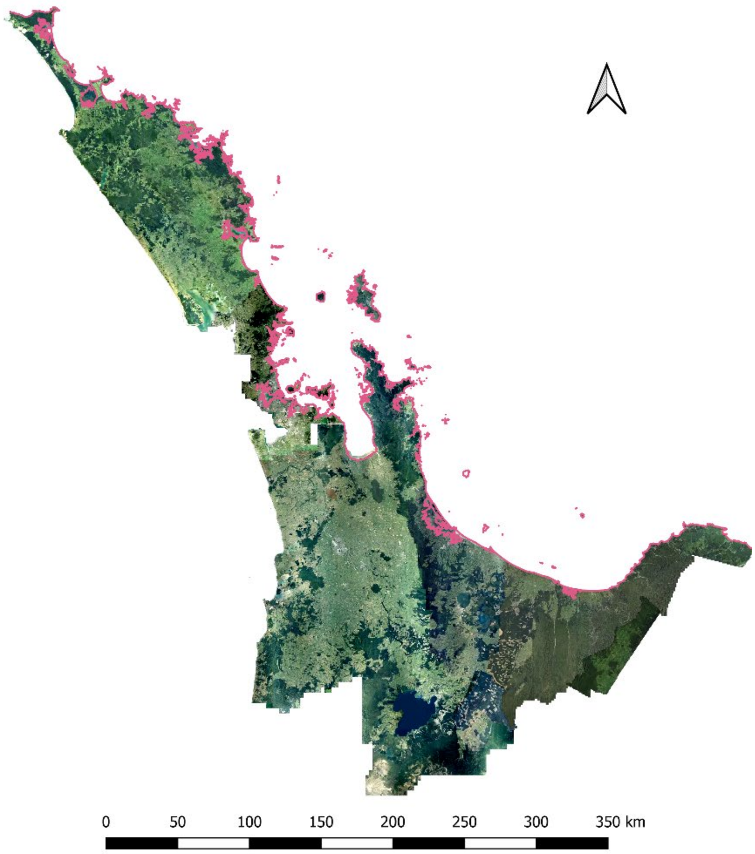

Figure 1: Northeastern New Zealand study area. Pink line denotes extent of coastline from Cape Reinga in the north to East Cape in the south that was evaluated.

Despite growing use for subtidal mapping, this approach in New Zealand has typically focussed on specific locations e.g. Hauturu-o-Toi or the Cape Rodney-Okakari Pt. Marine Reserve. By restricting the size of the study area high resolution satellite imagery can be purchased (or targeted aerial flights undertaken), as well as the collection of extensive ground-truthing information, without becoming cost prohibitive. Replicating this approach with paid satellite imagery or targeted aerial flights and extensive ground-truthing across the entirety of northeastern New Zealand would not be feasible given the geographic scale. Instead, our approach has been to leverage freely available imagery sources (e.g. Land Information New Zealand (LINZ) aerial imagery and Google Earth) in combination with existing ecological data that can be used for ground-truthing.

2.2.2 Image acquisition

All habitat mapping was carried out using QGIS v 3.34.13-Prizren. The project coordinates projection was set to the New Zealand Transverse Mercator 2000 (EPSG 2193), which is the recommended projection for use in mainland New Zealand and its nearshore islands (LINZ 2025a).

Prior to beginning the mapping exercise, a review of freely available imagery was undertaken to establish the most suitable imagery sources for the project. For the purposes of this study, we aimed to utilise only recent acquired imagery, meaning that the search was filtered to imagery captured no earlier than 2020 (i.e. only captured within the past five years). Two primary data sources were identified and considered suitable for use: LINZ aerial photographic surveys and Google Earth Satellite imagery. Other sources such as LANDSAT satellite imagery were considered but ruled out due to low resolution.

LINZ aerial photographic surveys captured between 2020 – 2025 (mostly from 2023 onwards) were available and covered the entire mainland coastline of the survey region, as well as most offshore island groups. To access the LINZ aerial data the LINZ Data Importer plugin was installed in QGIS. This allowed online access to the LINZ Data Services portal. Here available imagery could be selected and imported into the QGIS project as web linked vector layers (Figure 2). Imported imagery was typically clipped by the importing process to about 200 m offshore. Image resolution ranged from 0.075 m/pixel to 0.3 m/pixel.

Google Earth satellite imagery has been used in previous mapping projects (e.g. Hossain et al. 2016; Shahmirian 2020) and imagery is available to download as XYZ tiles within QGIS. This loaded in a global web linked vector layer showing the most recently captured satellite imagery for each specific location (Figure 2). For most coastal areas around northeastern New Zealand the most recent Google Earth imagery was captured between 2023 and 2024 and had a resolution (at scales of 1:300 – 1:1200) of about 1.5 m/pixel.

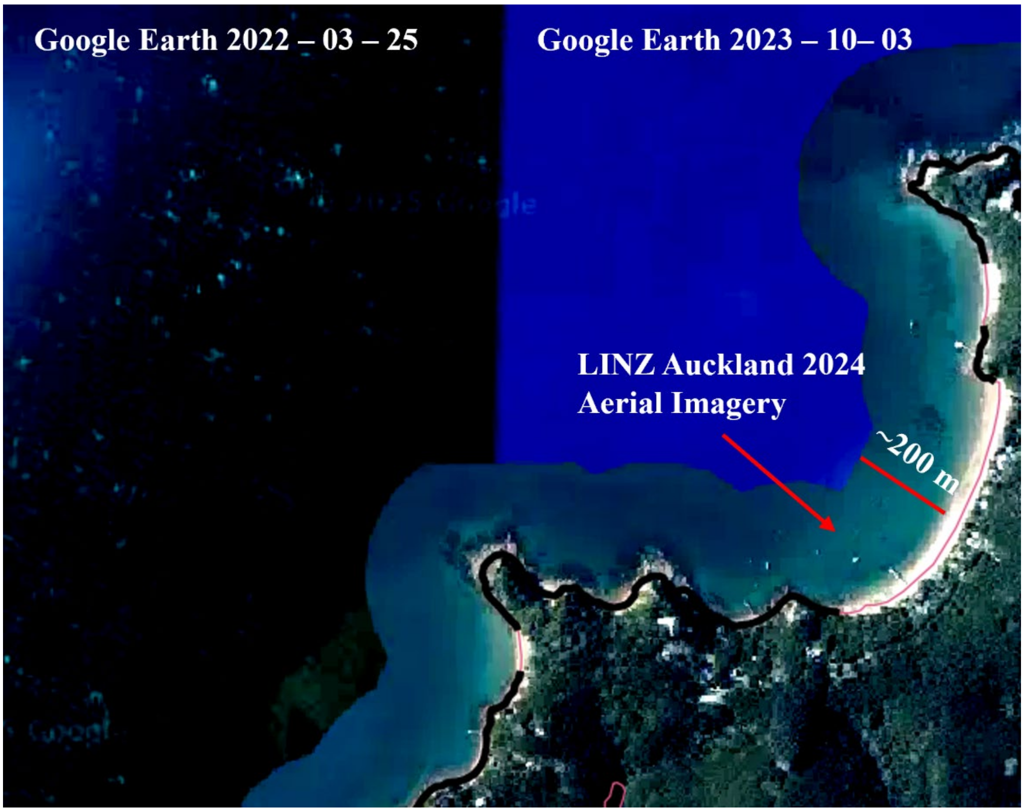

Figure 2: Primary imagery sources used for habitat mapping. Included are an example of the LINZ web vector layer for the Auckland 2024 Urban Aerial Imagery (0.075 m resolution – clear imagery bubble which extends approximately 200 m offshore). Also included are examples of Google Earth Satellite Tiles captured in March 2022 (darker coloured area on left side of image) and October 2023 (lighter coloured area on right side of image). The Google Earth Satellite images sit behind the LINZ imagery in this example, so all visible areas of reef are shown as part of the LINZ 2024 imagery layer.

In general, we endeavoured to use the LINZ aerial imagery as much as possible for habitat mapping due to its superior quality (higher resolution) and capture times corresponding with low tide - limiting confusion with intertidal reefs. Google Earth imagery was used where LINZ data was not available or was considered poor quality (e.g. too much sun glint, swell, land shadow or turbidity).

While the combinations of these two image sources did cover the entire marine space within the study area there were some key limitations to their use which ultimately dictated how urchin barren extent was able to be effectively quantified. Firstly, the imported LINZ dataset only extended offshore to 200 m, effectively missing any areas of urchin barrens that extended farther offshore. This issue was in part overcome by using Google Earth imagery to supplement the LINZ imagery so that habitat tracing could continue farther offshore. However, this was not always possible given the lower quality of Google Earth Imagery and the general diminished ability to identify barrens farther offshore as water became deeper (Lawrence 2019). Further to this we only used LINZ and Google Earth data in tandem where both sources were captured within 12 months of one another. This was done to limit the potential for temporal discrepancies between data sources compromising an accurate estimation of total barren area. The second issue was that offshore reefs, those completely isolated from the coast (e.g. Fair Way Reef, Doubtless Bay which is about 3 km from the nearest land), were not represented at all by LINZ data and were typically poorly represented by Google Earth imagery (tending to be low quality or have less recent capture dates). In effect very few offshore reefs were able to be mapped using this methodology, while urchin barrens on deeper reefs, even those extending directly offshore were likely under-represented. To effectively map these areas would typically require dedicated ground truthing information, including diver surveys to validate effectively.

These key limitations resulted in most mapping occurring within 200 m of the coastline. As urchin barrens throughout northeastern New Zealand are characteristically found in shallow, nearshore waters (Grace 1983; Kerr et al. 2024) and that the goal of the project was to evaluate the entire northeastern region these were considered an acceptable trade-off of the methodological approach. Greater coverage of offshore and isolated areas of reefs would probably only be possible with additional paid imagery or by purchasing specifically tasked imagery (e.g. satellite or aerial surveys) and extensive ground-truthing.

2.2.3 Project setup

Once the imagery sources were determined the project file was set up in QGIS and a series of layers likely to be used for aiding habitat mapping and performing analysis were imported into QGIS. These included:

- NZ Coastlines and Islands Polygons (Land Information New Zealand (Land Information New Zealand - LINZ 2025b), which provided a consistently determined coastal (land/sea interface) boundary (based on mean high water mark) throughout the country.
- Bathymetric model of New Zealand (NIWA 2016), which provided a consistent source of depth information (at a 250 m resolution) across New Zealand. This information was compiled by NIWA in 2016 and is based off published coastal charts, digital soundings archive, navy collector sheets and digital multibeam data.
    - It should be noted that finer scale depth information (e.g. 1 m resolution bathymetry models) were available in some areas, however the NIWA bathymetric model was used, despite coarser resolution, so that a consistent approach could be used across all surveyed areas.
- Regional Council 2023 (generalised) (Statistics New Zealand 2023), which provided official regional council boundaries across New Zealand.
- Subtidal Habitat Polygons (Department of Conservation & Ministry of Fisheries 2011), which provided a consistent seafloor classification for marine habitat types found throughout the country. This layer was clipped to rocky reef areas less than 30 m deep. All polygons were created in 2010 but those associated with the Hauraki Gulf Marine Park (encompassing the Auckland and Waikato Regions) were updated in 2021.

To begin with, using the LINZ and Google Earth imagery, we identified all areas of the coast unsuitable for barren formation. These included areas of sandy shore with no visible offshore reef, as well as harbours, estuaries and sheltered embayment such as the inner Waitemata Harbour and Firth of Thames. These areas tend to have limited and/or very shallow reef and environmental conditions (low wave action and high sediment loads) not generally considered conducive to supporting large sea urchin populations, or the formation of urchin barrens (Shears & Babcock 2004). A polygon was created around each identified length of coastline meeting these criteria and assigned a Status of "Unsuitable location". Coastal areas containing shoreline deemed suitable for the potential presence of urchin barren (e.g. rocky coast outside of those areas classified as "Unsuitable location") but that could not be mapped due to poor image quality or environmental conditions (e.g. too much sun glint, or imagery affected by swell [whitewash] or high turbidity) were also recorded within this polygon layer. These areas were assigned a Status of "Unmapped".

2.2.4 Habitat/Urchin barren mapping

Habitat mapping was then commenced with the focus being the identification of areas of urchin barrens to the greatest extent possible. Given the scale of the project, we did not attempt to map habitat types associated with dense covers of canopy forming macroalgae e.g. kelp forest or shallow mixed algae which have been mapped in previous exercises concentrating on specific locations (e.g. Lawrence 2019; Dartnall 2022; Kerr et al. 2024). Instead, the habitat types mapped focussed primarily on urchin barrens and other habitat types that were characteristically devoid of canopy forming macroalgae within the aerial imagery. Because the focus was specifically on urchin barrens we chose to separate urchin barren habitat from turfing/foliose algae habitats that have previously be treated as one broader habitat type (e.g. Lawrence 2019; Dartnall 2022; Kerr et al. 2024; Shears & Lawerence 2024). This decision was made to try to more accurately account for urchin barren extent, relative to other habitat types. We used ground truthing information, geographical context and descriptions of each habitat and typical reef zonation patterns (e.g. those provided in Grace 1983; Shears et al. 2004) to help guide the delineation of these habitat types. For accuracy we also included a conservative Urchin Barren - Unconfirmed class to account for areas that presented as typical urchin barrens (see Table 1) but either extended into areas of poor image quality, impacting accurate mapping of their true shape, or had greater uncertainty around their absolute classification due to a lack of available ground truthing imagery/information. These Unconfirmed areas were not included in estimations of urchin barren extent or area calculations unless specifically designated. In total we mapped seven habitat types with the description of each listed in Table 1.

A polygon layer was set up allowing for areas of reef to be mapped and classified. Habitat mapping was mostly conducted at scales of 1:300 to 1:1200 depending on the image quality and the area being mapped. Each identified area of urchin barren (or other habitat type) was manually traced using stream digitisation. This effectively allowed the edges of the area to be continually traced without needing to add new vertices intermittently. Once an area was mapped, adjustments to the boundaries were done using tools found in the advanced digitising toolbar e.g. reshape, add ring, split feature. Each polygon contained information on its general location (e.g. Mimiwhangata), the region it occurred within (e.g. Northland), the imagery source used (e.g. LINZ Northland 0.3 m 2023 – 2024) as well as the habitat type and whether a Quality Assurance check was required.

Table 1: Habitat types used for urchin barren mapping and descriptions of each.

Habitat Type	Description
Urchin Barrens	Area of rocky reef largely devoid of canopy forming macroalgae. Reef shows up as a light colour in contrast to macroalgae (which appear as dark colours on the reef, usually green or brown in appearance). Characteristically found as a band between shallow mixed algae and deeper kelp forest. These natural reef features were used as indicators to confirm that the area had suitable environmental conditions to support kelp and other canopy forming macroalgae, and that overgrazing by sea urchins was the likely cause of a lack of canopy forming macroalgae. In sheltered areas urchin barrens may extend to the reef/sand interface (no deeper kelp band) and the shallow mixed algae zone may be very narrow.
Urchin Barrens – Unconfirmed, more information required	As above, but a lack of available ground-truthing imagery reduced certainty or absolute boundaries dictating the alongshore and/or offshore extent were uncertain due to poor image quality or barrens zones extending into water too deep to continue mapping.
Turf and foliose algae	Area of rocky reef largely devoid of canopy forming macroalgae. Show up as lighter coloured reef than canopy forming macroalgae but typically have more texture than urchin barrens or surrounding sand. Largely found towards the reef/sand interface below kelp forest or on low profile reef surrounded by sand, where inundation and scour prevent the growth of canopy forming macroalgae. Urchins may still be present within these areas.
Turf and foliose algae – Unconfirmed, more information require	As above but lack of available ground-truthing imagery reduced certainty or absolute boundaries dictating the alongshore and/or offshore extent were uncertain due to poor image quality or turf and foliose algae zones extending into water too deep to continue mapping.
Mosaic of canopy forming macroalgae and non-canopy forming reef habitat types	Areas of rocky reef containing a highly patchy (mosaiced) distribution of canopy forming macroalgae and non-canopy forming urchin barren and/or turfing and foliose algae habitats. Mapping as distinct habitats not practical.
Unconsolidated	Areas of sand, gravel or cobble. Only mapped where confusion with rocky reef/urchin barrens may occur.
Intertidal	Bare rock in intertidal areas that may be confused with urchin barrens or turfing and foliose algae. Mostly mapped where Google Earth was used as primary image source and intertidal platforms were underwater at the time imagery was captured. Only mapped where confusion with rocky reef/urchin barrens may occur.

2.2.5 Ground truthing

Georeferenced drop camera images from 11 locations were used to ground truth and validate the habitat mapping (Figure 3). Drop camera imagery was collected via boat and divers and much of it was captured as part of unrelated work carried out by the University of Auckland within the two years prior to the commencement of this project. Additional drop camera imagery was captured at 10 locations over the course of the current project. A list of locations is provided in Appendix 1.

Drop camera imagery was captured following the general methods laid out in Kibele (2017). For the boat-based approach a Go-Pro (Hero 11 Black) was attached to a weighted pole about 1.1 m above the seabed. This allowed for an image footprint of about 3 m^2 to be captured. The pole was weighted with a float attached to the top of the shaft to keep it at an upright angle. A line and coaxial cable ran from the top of the line back up to the boat allowing for the pole to be lowered and raised as well as for a Wi-Fi signal to be transmitted between the camera and surface. This cable allowed imagery to be captured remotely via a tablet and the Go Pro Quik app held at the surface. A Garmin GPS device was also used at the surface to record the location at which each image was taken, worn around the neck of the person lowering the camera, to ensure the GPS location most accurately matched where the image was captured. A depth logger was also attached to the pole at the same height as the camera to obtain the depth of each image. As much as practically possible, allowing for wind, swell and current the boat began capturing imagery nearshore and carried out a transect running perpendicular to shore from shallow to deep. The camera was lowered every 3 – 5 m and an image taken to capture the reef profile. The transect continued offshore until either the drop camera reached the edge of the reef or was too deep to continue (>20 m).

Diver drop camera image capture involved a diver swimming with a Go-Pro attached to a lightweight pole, with the camera attached about 1.1 m above the ground. A depth logger was also attached at this height. The pole was connected to a surface float containing a GPS via a bungy cable. As the diver was swimming, constant tension was kept on the bungy cable to keep the float positioned as close as possible to directly overhead to ensure that the GPS position reflected the diver's position. Imagery was generally captured down a fixed transect line at 5 m intervals from shallow to deep. Divers then swam a series of transect lines running alongshore, with the fixed transect line acting as a centre point. Alongshore transects ran approximately 50 m either side of the fixed transect line and got progressively shallower as divers moved up the reef (see Figure 5 for an example).

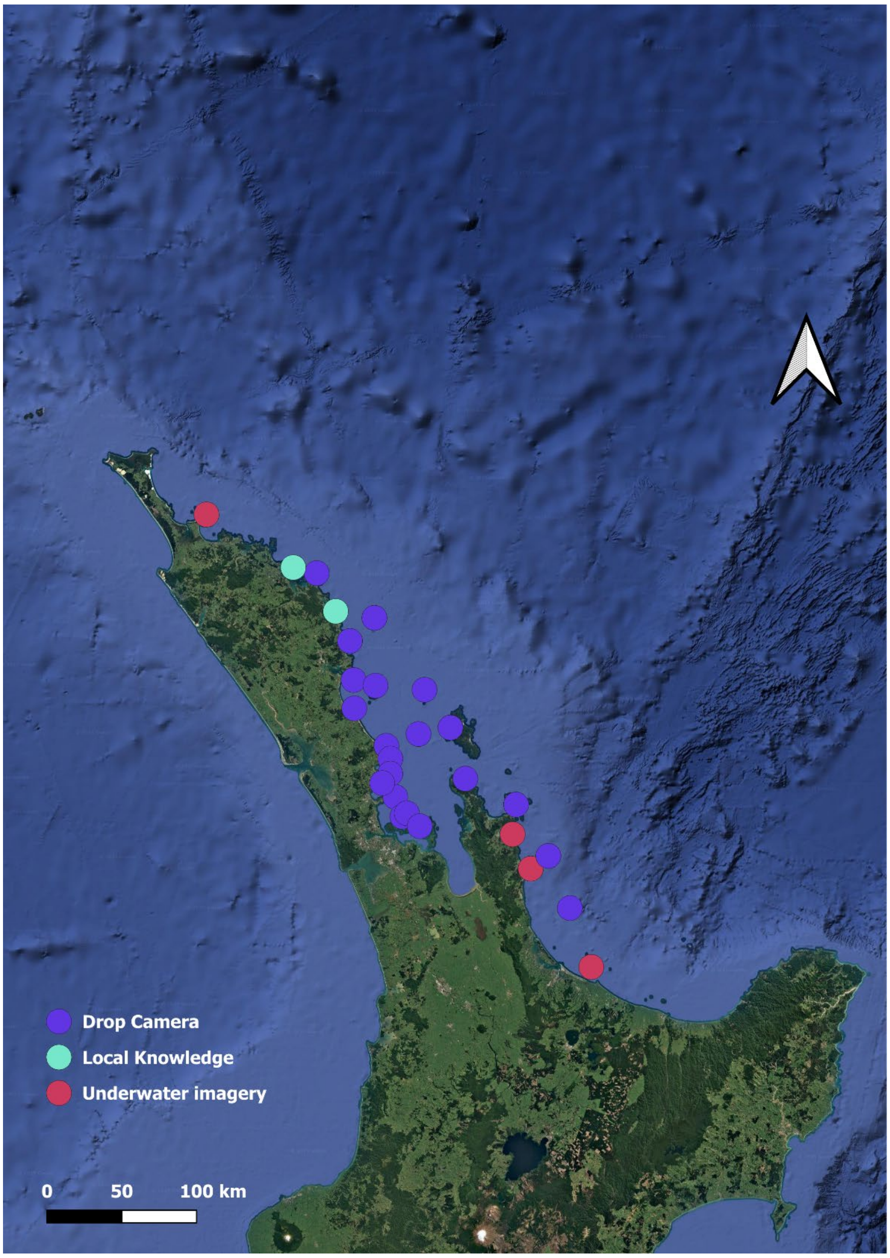

Figure 3: Location of drop camera imagery (purple), local knowledge (teal) and underwater imagery (red) used for ground truthing habitat mapping across northeastern New Zealand. Drop camera imagery was georeferenced and used in validation exercises. Local knowledge came from conversations with mana whenua and community members with knowledge of the current extent of urchin barrens within their local area. Underwater imagery provided general information on reef state, but could not be used for validation due to a lack of georeferencing. See Appendix 1 for a list of location names.

Imagery was geotagged using Benthic Photo Survey (BPS) methods (Kibele 2016). Once geotagged each image was classified into one of several habitat categories designed to aid mapping and provide validation. The description of each can be found below in Table 2 and are based on descriptions provided by Shears et al. (2004). Examples of the different habitat types are provided in Figure 4.

Table 2: Description of habitat classes used for drop camera imagery.

Habitat Type	Description
Urchin barren	Large brown macroalgae rare or absent, substratum typically dominated by crustose coralline algae and sea urchins present. Seasonal filamentous, turfing or foliose algae may be present. Usually associated with grazing activity of *Evechinus chloroticus* (kina) and/or *Centrostephanus rodgersii* (centro). No distinction between barrens associated with either species were made (all collectively called urchin barrens).
Urchin barren mosaic	Clear urchin barren habitat interspersed among patches of canopy forming macroalgae, turf and foliose algae or unconsolidated material.
Turf and foliose algae	Image with high cover (>30% cover) of turfing or foliose algae. Canopy forming macroalgae rare or absent. Sea urchins may be present with minimal or no signs of active grazing. For the purposes of this study this habitat type amalgamated the Red Foliose Algae, Turfing Algae and Caulerpa habitat types described in Shears et al., (2004).
Kelp	Image dominated (>50% cover) by *Ecklonia radiata.* Urchins may be present but no signs of active sea urchin grazing.
Mixed algae	Mixed canopy (no dominant species) of *E. radiata* and other large forest forming macroalgae. Often interspersed with turfing or foliose algae. Little to no signs of active sea urchin grazing.
Other Canopy Forming Macroalgae	Image dominated (>50% cover) by other species of large forest-forming macroalgae (e.g. *Carpophyllum* spp.). Generally captured on very shallow reef or deeper reef where dense stands of *C. flexuosum* can occur. For the purposes of this study this habitat type amalgamated the *Shallow Carpophyllum* and *Carpophyllum flexuosum* forest habitat types described in Shears et al. (2004).
Unconsolidated	Image dominated by sand, gravel or small mobile cobbles

Ground truth imagery was randomly split into training and validation data using an 80:20 split. Training data was added to the project to aid with mapping (Figure 5) while the validation data was used only to validate results.

In addition to drop camera imagery we used local knowledge and underwater imagery sourced from research partners to improve confidence in mapping areas where drop camera data was limited or absent (Figure 3). Local knowledge was gained from conversations with mana whenua and locals with good knowledge of existing locations of urchin barrens within a given area. Underwater imagery included photos and videos that provided information on the reef habitats at specific locations, but did not include specific GPS information and were in formats that were not consistent with drop camera imagery, so could not be used for validation.

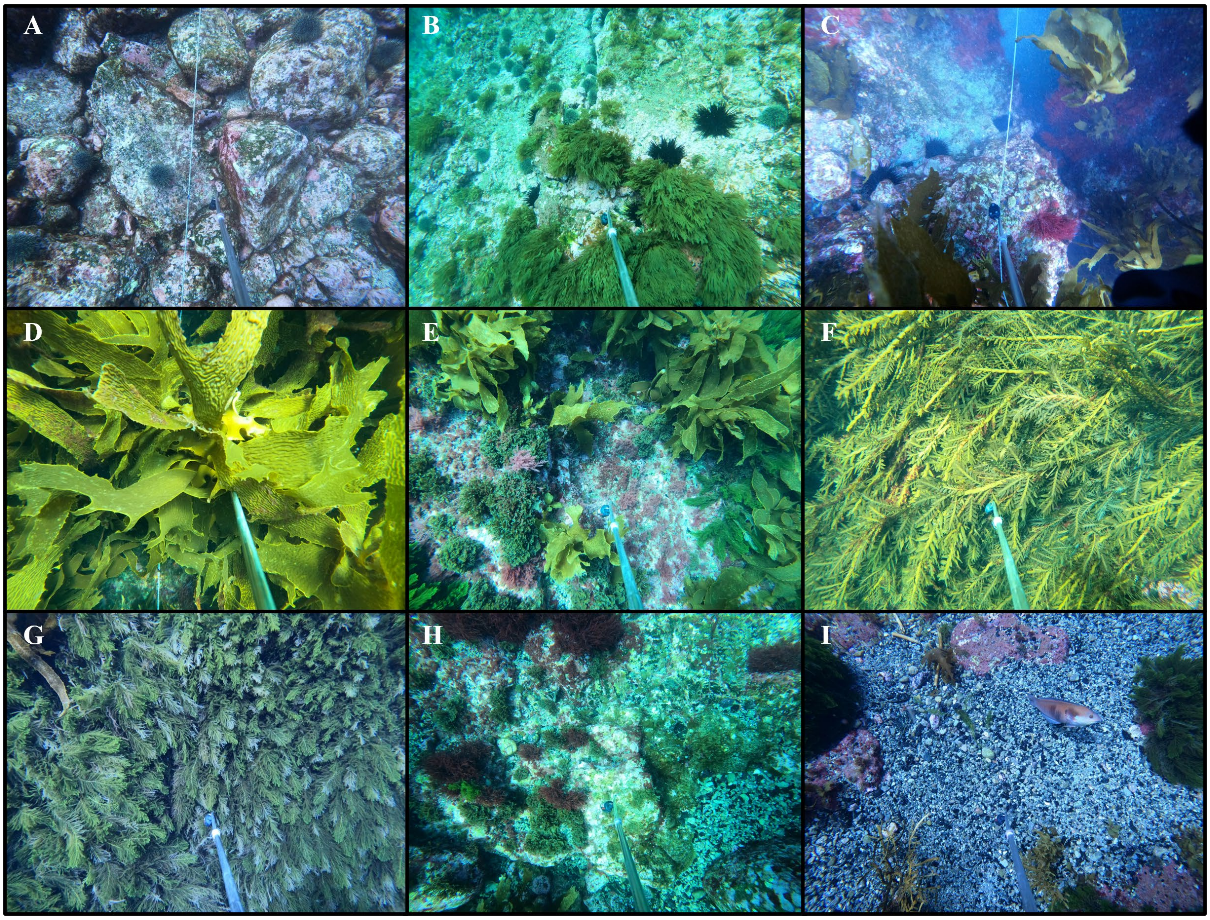

Figure 4: Examples of habitat classifications from diver and boat-based drop camera imagery. A) urchin barren (kina present), B) urchin barren (kina and centro present) with seasonal foliose algae growth, C) Urchin barren mosaic (centro present) with clear areas of urchin barren and intact macroalgae, D) Kelp (*Ecklonia radiata*) forest, E) Mixed algae with kelp cover interspersed with turf and foliose algae, F) Other canopy forming macroalgae, G) Turf and foliose algae – dominated by native Caulerpa (green foliose algae), H) Turf and foliose algae – dominated by turfing algae no urchins present), I) Unconsolidated (gravel).

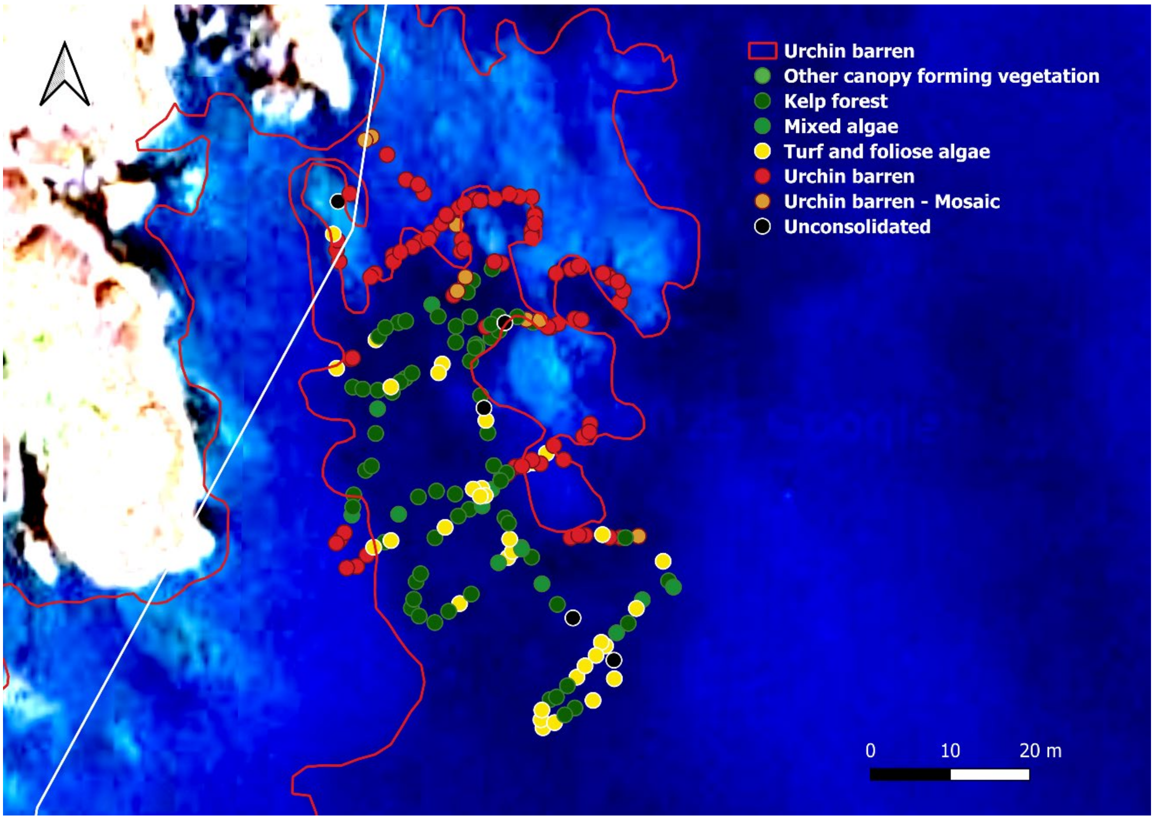

Figure 5: Example of ground truthing imagery used to help guide habitat mapping and provide additional confidence that mapped areas were likely to be urchin barrens. This example is from the Marotere Islands using Google Earth imagery (November 2023). Ground truthing imagery was also captured in November 2023 with divers first swimming down a fixed transect line and then continuing in a series of alongshore transects that got progressively shallower as they worked back up the reef. The red polygon denotes a mapped area of urchin barren. Red and orange points represent drop camera images associated with urchin barren habitats, green points represent drop camera images associated with canopy forming macroalgae and kelp, yellow points represent drop camera images associated with turf and foliose algae and black dots represent unconsolidated material.

2.2.6 Validations

Mapping was validated using the drop camera imagery as well as a second exercise to gauge the difference in area mapped between different annotators.

An accuracy assessment was carried out to evaluate the accuracy in which specific habitat types were being correctly labelled. This was done by matching the total number of drop camera validation points per classified habitat type to the classified habitat polygons. Because the project focussed on mapping urchin barrens, validation points overwhelmingly fell within urchin barren polygons, therefore we only conducted an accuracy assessment on the ‘Urchin barren’ habitat type and “mosaic” habitat type. Validation points that were not captured within a habitat point (i.e. on reef deeper beyond the mapping limits) were not used for the assessment. The accuracy assessment was done using the strict GPS positions.

To understand the discrepancy in mapped area between annotators 20 areas of rocky coastline were selected at random. This represented four locations from each of the primary image sources used (LINZ Northland 0.3 m 2023 – 2024, LINZ Auckland 0.05 m 2024, LINZ Waikato 0.3 m 2023 – 2024, Google Earth 2023 – 2024) so that we could evaluate overall discrepancy and whether particular image sources were more prone to difference. Within each polygon the area of urchin barren (and visibly similar habitat types) was mapped independently by two separate annotators. Any prior mapping within or ground truthing imagery the area was hidden from view as to not influence annotator results. Once complete the total area of mapped urchin barren within each polygon was averaged across the annotators and a relative difference in total area to this averaged value calculated.

2.2.7 Urchin barren extent analysis

Following the habitat mapping exercises the total length of mapped, unmapped and unsuitable coastline was calculated. Calculations were also made for the total mapped area of visible urchin barrens as well as other habitat types. These calculations were made at a regional level as well as across the entire project area (Cape Reinga – East Cape).

Using the length of mapped coastline and total area of visible urchin barrens within each region we then calculated the area of visible urchin barrens per km of suitable coastline. This allowed extrapolation of additional potential urchin barren area along those sections of coastline that were considered suitable for barrens but could not be mapped due to image quality or environmental condition issues.

We investigate the influence of depth and distance from shore on the total area of visible urchin barrens that were mapped. Urchin barren polygons were clipped to a 10 and 20 m depth contour (using the 2016 NZ Bathymetry Model) with the area of mapped barrens expressed as a percentage of total mapped barren area. Similarly, because the LINZ aerial imagery mostly extended to a maximum distance of 200 m offshore, we looked at the percentage of total urchin barren area that was within 100 m and 200 m of the coastline (based on the NZ Coastlines and Islands Polygons).

Based on the data collected throughout the project, and in combination with other available data sources there were two key approaches that could be used for providing an estimate of urchin barren extent across northeastern New Zealand. 1) a calculation of total area of mapped urchin barrens relative to the total area of subtidal rocky reef (available from the Subtidal Habitat Polygons), as was done in Kerr et al. (2024) and/or 2) a calculation of the proportion of suitable coastline (e.g. rocky coastline) with adjacent urchin barrens relative to the total length of suitable coastline.

We chose Option 2 for three key reasons. First, it was clear that not all areas of urchin barrens were able to be mapped using the freely available aerial imagery used in this study. To do so would have required extensive additional ground-truthing of areas poorly represented by the imagery, which would be entirely impractical over such large spatial scales. Thus, any area calculations were likely to underestimate total urchin barren area. Second, the Subtidal Habitat Polygon layer often grossly overestimated reef extent. For example, across some areas of Hauturu-o-Toi, it extended 500 m beyond the clearly visible reef edge seen in high-resolution imagery used for habitat mapping by Dartnall (2022; Figure 6). Using satellite-derived reef extent, urchin barrens at Hauturu-o-Toi were estimated at 30% of reef area in 2019 (Dartnall 2022), dropping to 10% when using the Subtidal Habitat Polygon layer to provide total reef extent. Such discrepancies were widespread, and some reef areas were omitted entirely (Figure 7), potentially skewing urchin barren estimates relative to total reef area. Finally, kina barrens are largely restricted to shallow reef areas, so it is not ecologically meaningful to express the extent of barrens as a percentage of overall reef area.

Calculating the proportion of coastline with adjacent urchin barrens did not rely on area estimates, instead assigning a binary classification to the data (1, urchin barren present, 0 urchin barren absent). In this way the presence of urchin barren polygons, as opposed to the total area represented within each polygon, became the important factor for estimating barren extent. All mapped areas of coastline were split into discrete sections with a maximum length of 100 m. Line section midpoints were then generated and each segment's mean bearing was calculated. A transect line running perpendicular to this bearing was then created and scored as to whether it intersected with any areas of adjacent urchin barren or not (Figure 8). The same urchin barren could be scored on multiple transect lines, accounting for overlapping transect lines where the coastal morphology was complex and reflexive, however a given transect line was not scored multiple times if it crossed through multiple urchin barren polygons (each transect only had one score - 0 or 1). We used two transect lengths; 200 m to reflect the offshore limit of most LINZ imagery and 400 to account for the inconsistent nature of intertidal reef extent (relative to position of the coastline) throughout the region and ensure that urchin barrens farther from shore were captured. Using the 200 m transect lines we also looked at the extent of coastline containing any habitat type that may be associated with overgrazing (Urchin Barren, Urchin Barren – Unconfirmed, Mosaic, Short Vegetation).

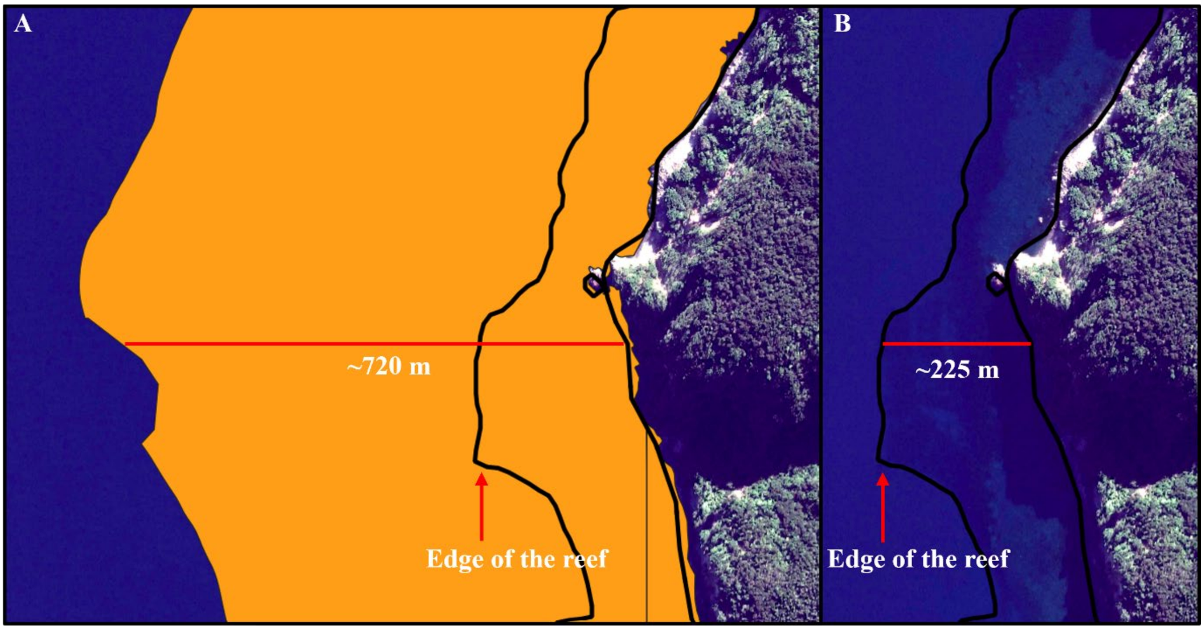

Figure 6: Total shallow reef extent along the western side of Hauturu-o-Toi based on A) the Subtidal Habitat Polygons (orange area) and B) high resolution satellite imagery. The edge of the reef based on satellite imagery is displayed as a black line in A) and B). The reef extends approximately 500 m farther offshore at this location based on the Subtidal Habitats Polygon.

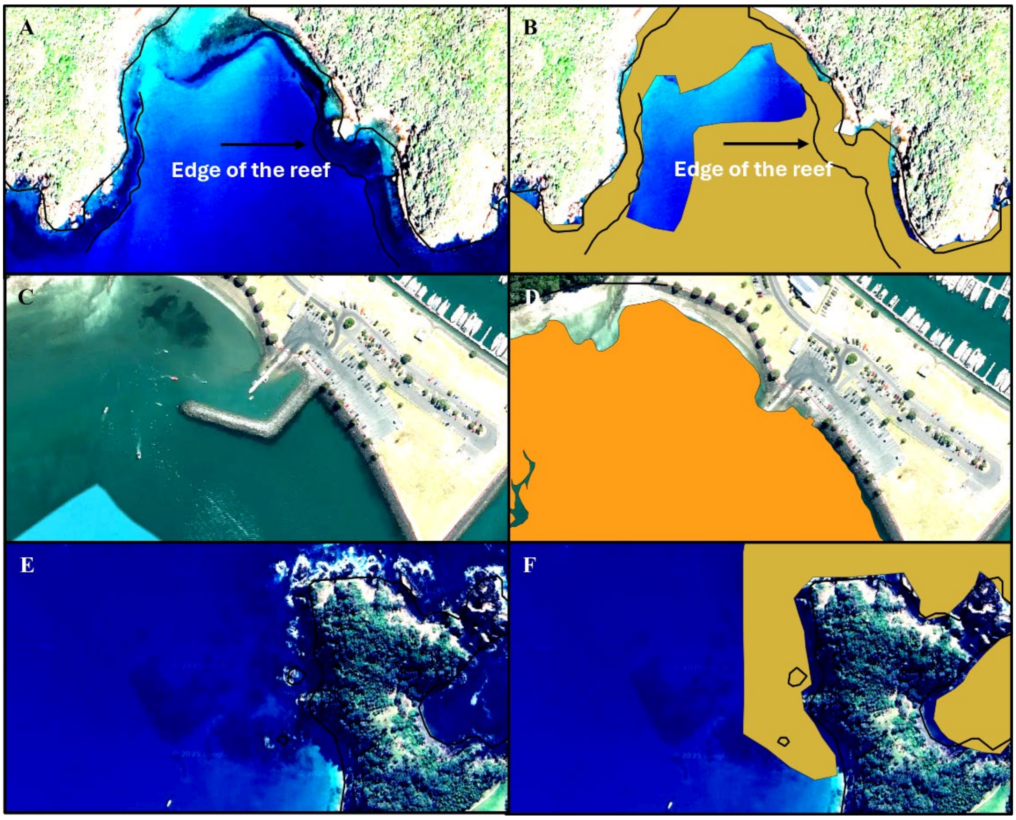

Figure 7: Examples of discrepancies between actual rocky reef extent and the extent present within the Subtidal Habitats Polygons. In A) high quality satellite imagery of a sandy embayment at the Marotere Islands, Northland which in B) is largely classified as rocky reef (yellow area) within the Subtidal Habitats Polygon. In C) the soft sediment areas surrounding the boat ramp at Gulf Harbour, Auckland and D) the same area with the Subtidal Habitats Polygon layer (orange area) turned on. In E and F subtidal rocky reef that is missed by the Subtidal Habitats Polygon layer.

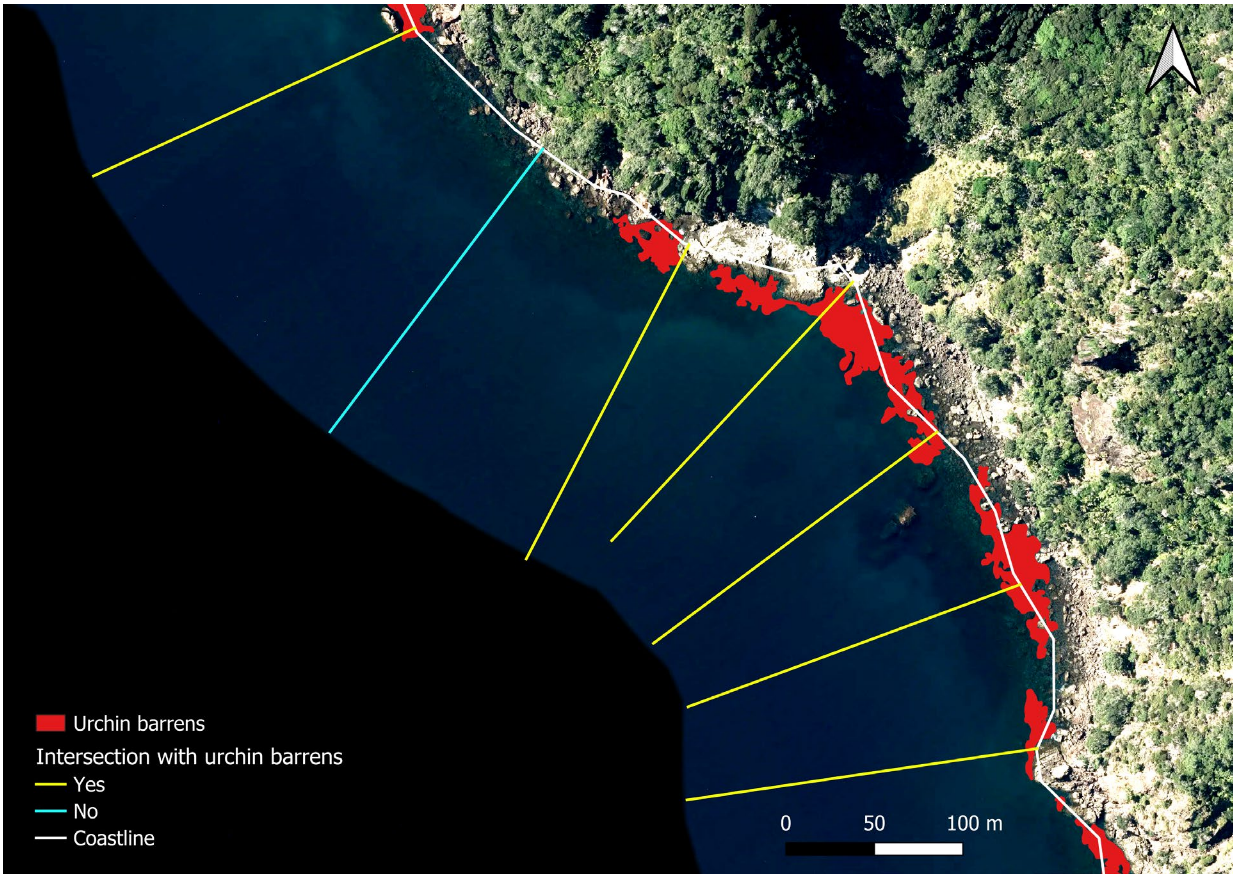

Figure 8: Perpendicular transect lines used to evaluate urchin barren extent. Each transect line runs offshore perpendicular to the bearing of the line segment it was generated from. Transect lines intersecting with an urchin barren polygon (red polygons) are coloured yellow, while those that do not intersect with any urchin barren polygons are coloured teal. The transects in the image extend 200 m offshore from the coastline (white line).

Additionally, we compared the extent of coastline classified as urchin barrens (and associated habitat types) at seven locations mapped as part of this study that were also mapped using satellite imagery sourced between 2016 and 2020 by Lawrence (2019), Dartnall (2022) and Shears and Lawerence (2024). These locations were Mimiwhangata, Mokohinau Islands, Hauturu-o-Toi, Cape Rodney to Okakari Point Marine Reserve, The Noises, Great Mercury Island and Aitu/Green/Korapuki Island. Prior mapping at Hauturu-o-Toi and The Noises followed a very similar methodological approach, with manual mapping of urchin barren habitats supplemented with drop camera ground-truthing information. At the other five locations supervised classification was used for mapping, supplemented with drop camera ground-truthing. We did not compare our results to those of Kerr et al. (2024) because many of the areas were mapped about 15 – 20 years ago and relied heavily on diver observations for ground-truthing and habitat mapping.

Habitat classifications done by Lawrence (2019) and Dartnall (2022) had an ‘Urchin Barren’ habitat classification but no turfing algae or mosaic categories. It is likely that these two categories were included within the ‘Urchin Barren’ classification to some extent. Habitat classification by Shears and Lawerence (2024) combined urchin barrens and turfing algae into a single classification called ‘Urchin Barren/Turfing Algae’. We therefore compared the extent of ‘Urchin Barren’ (Lawrence (2019), Dartnall (2022) and ‘Urchin Barren/Turfing Algae’ (Shears & Lawrence (2024)) habitat to the ‘Urchin Barren’ habitat classified in this study and also to an amalgamated category containing ‘Urchin Barren’ (inc. unconfirmed), ‘Mosaic’ and ‘Short Vegetation’ (turfs).

To compare extent of coastline between time points at each location we selected all 200 m transect lines within the study region (determined by the spatial extent of the earlier study). Following the method described above the number of transects intersecting with the various urchin barren habitat types relative to the total number of transects was calculated.

3 RESULTS

3.1 Collate quantitative data on the current and historical extent of urchin barrens in New Zealand

We identified 27 studies that contained information on urchin barren habitats alongside suitable geospatial information for inclusion within the geodatabase (Figure 9). These included information on reef habitats as far back as 1953 (Hauturu-o-Toi) and as recently as 2020 (Mercury Islands). Included studies spanned the length of the country but there was a heavy bias towards northeastern New Zealand, with 23 of the 27 studies coming specifically from this region. Most studies (25 of 27) were in the form of detailed habitat maps which distinguished between different subtidal habitat types e.g. kelp forest, urchin barren, shallow mixed algae while the only New Zealand wide study, unpublished data sourced as part of the Department of Conservation study conducted by Shears and Babcock (2007) was based on diver survey assessments of urchin barren extent down transect lines to a maximum depth of 10 m. A full list of included studies can be found in Appendix 2.

A broadly consistent definition of urchin barren habitat (see the definition in Section 2.2.5) has been used across the included studies despite some of the older studies, e.g. habitat maps created in the 1970s, using different terms to describe urchin barrens, e.g. Ayling (1978) termed grazed reef with high urchin abundance as ‘rock flats’. These studies indicate the widespread presence of urchin barrens throughout the country with areas of extensive urchin barren occurring throughout northeastern New Zealand, the Marlborough and Tasman Regions, Stewart Island and parts of Fiordland.

Time series information, in the form of habitat maps, existed at six locations; Mimiwhangata (1973, 2005, 2019), the Mokohinau Islands (1978, 2019), Hauturu-o-Toi (1953, 1979, 2019), the Cape Rodney – Okakari Point Marine Reserve (1979, 2006, 2019), the Noises (1978, 2019) and the Te Whanganui a Hei Marine Reserve (2014, 2015). All other studies were based on a single time point. Historical habitat maps compiled from data collected in person at the time, or from historical aerial imagery, indicate the presence of urchin barrens at Mimiwhangata, the Mokohinau Islands, Hauturu-o-Toi, Leigh and the Noise in the 1970s with a notable decline in urchin barren extent only occurring within the Cape – Rodney to Okakari Point Marine Reserve since that time. Other marine reserves such as at Tāwharanui and Hahei showed lower urchin barren extent inside the protected area than outside, however these do not have historical records to understand change through time, with the available data coming from 2006 (Tāwharanui) and 2013/2014 (Hahei).

The New Zealand wide study (Shears and Babcock (2007) was the most comprehensive spatial record of urchin barrens across all other parts of the country, indicating widespread occurrence of urchin barrens at least as far back as 1999. In places such as Stewart Island and Queen Charlotte Sound more recent published and unpublished studies confirmed the continued presence of urchin barren habitat since this time (e.g. Lafont & Shears 2025). Studies such as Udy et al. (2019), which were not included in the study due to a lack of spatial/urchin barren extent information, also confirm continued presence of urchin barrens within Fiordland.

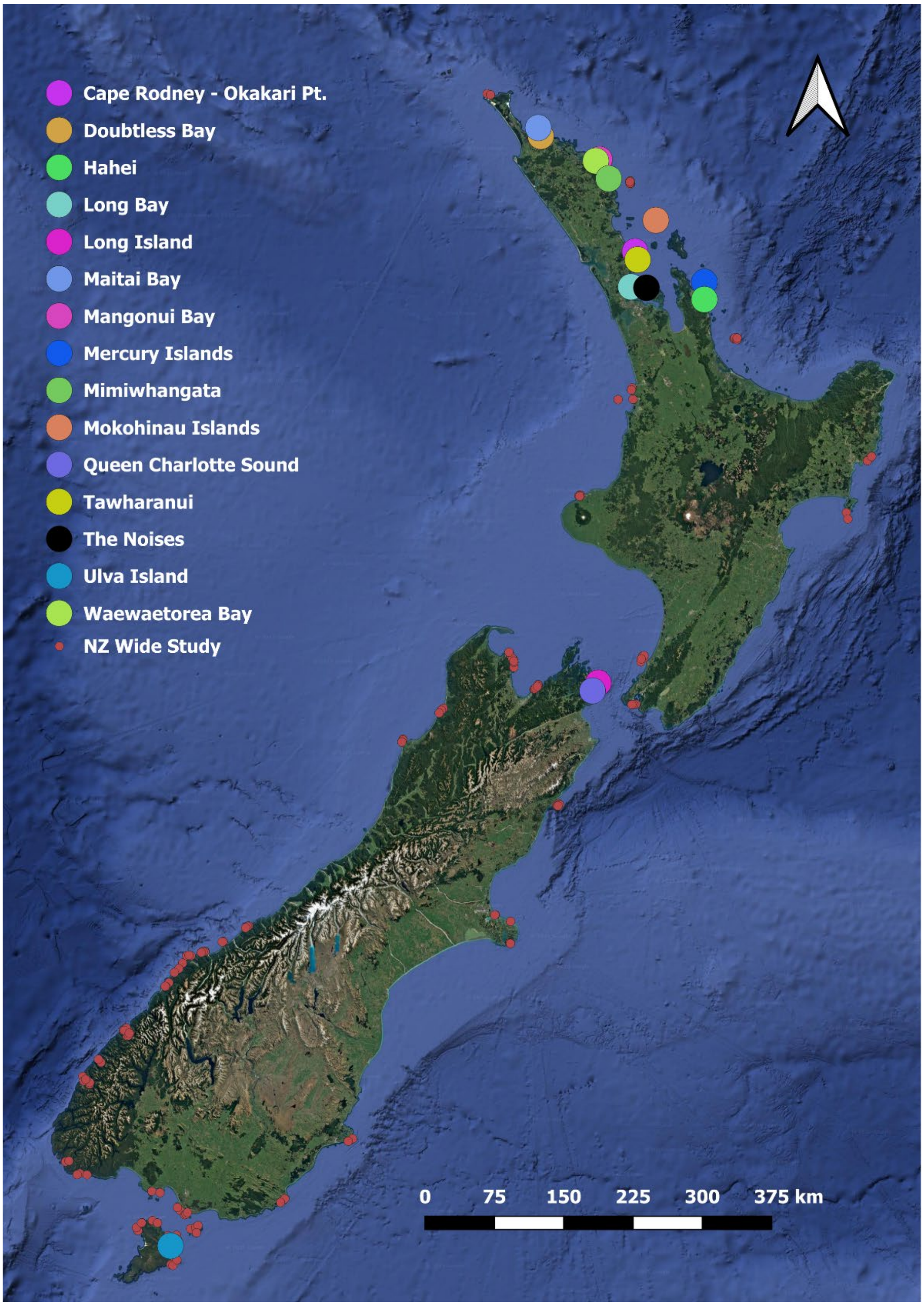

Figure 9: Location of historical studies containing information on the extent of urchin barren habitats as well as suitable geospatial information. Some locations such as the Cape Rodney to Okakari Point Marine Reserve have studies done at multiple time points. A list of studies related to each location can be found in Appendix 2.

3.2 Collect comprehensive baseline information on the extent of urchin barrens in northeastern New Zealand

3.2.1 Northeastern New Zealand coastline

The northeastern New Zealand coastline from Cape Reinga to East Cape was divided among five statutory regions and contained a total of about 5875 km of coastline (Figure 10), of which 29% (about 1677 km) was evaluated for the presence of urchin barrens (Table 3). Overall, Northland had the longest total and evaluated length of coastline while very little coastline fell within the Gisborne region (Table 3). Between Northland and Waikato, the majority (mean ± standard error - 83% ± 2%) of coastline that was considered suitable for containing urchin barrens (i.e. coastline flanked by rocky reef) was able to be evaluated. Throughout these regions offshore islands and headlands were the most common areas where imagery or environmental conditions prohibited effective mapping of urchin barrens and resulted in coastline being labelled as 'Unmapped'. Urchin barren habitat was only able to be effectively evaluated along one third of Bay of Plenty's rocky coastline, owing largely to high turbidity and swell which were problematic along much of the coastline east of Whakatāne. Similarly, the majority (>99%) of suitable coastline within the Gisborne Region (acknowledging that only a small amount of this region's coastline fell within the study area) was unable to be mapped due to issues with turbidity and swell.

Of the coastline that was not evaluated, most was classified as 'Unsuitable' for urchin barrens, and this included numerous beaches with no visible offshore reef as well as estuaries, harbours and sheltered embayments with limited or very shallow reef and/or conditions not conducive to supporting large sea urchin populations (e.g. high silt content or variable salinity). The largest areas that were excluded from evaluation were the inner Waitemata Harbour and Firth of Thames within the Auckland and Waikato regions.

Table 3: Extent of coastline across northeastern New Zealand. Coastline that was not evaluated for the presence of urchin barrens was split into 'unsuitable' coastline (beaches, harbours, estuaries etc) and 'unmapped' coastline (rocky coastline that could not be mapped due to image quality issues or poor environmental conditions).

Region	Total coastline (km)	Not evaluated (km)		Mapped (km)
		Unsuitable	Unmapped	
Northland	2 360.21	1 461.42	143.70	755.09
Auckland	1 681.79	1 091.08	84.11	506.60
Waikato	951.61	508.70	96.35	346.56
Bay of Plenty	810.98	638.91	112.77	59.30
Gisborne	71.14	16.52	54.56	0.05
Total	**5 875.73**	**3 707.15**	**491.49**	**1 677.06**

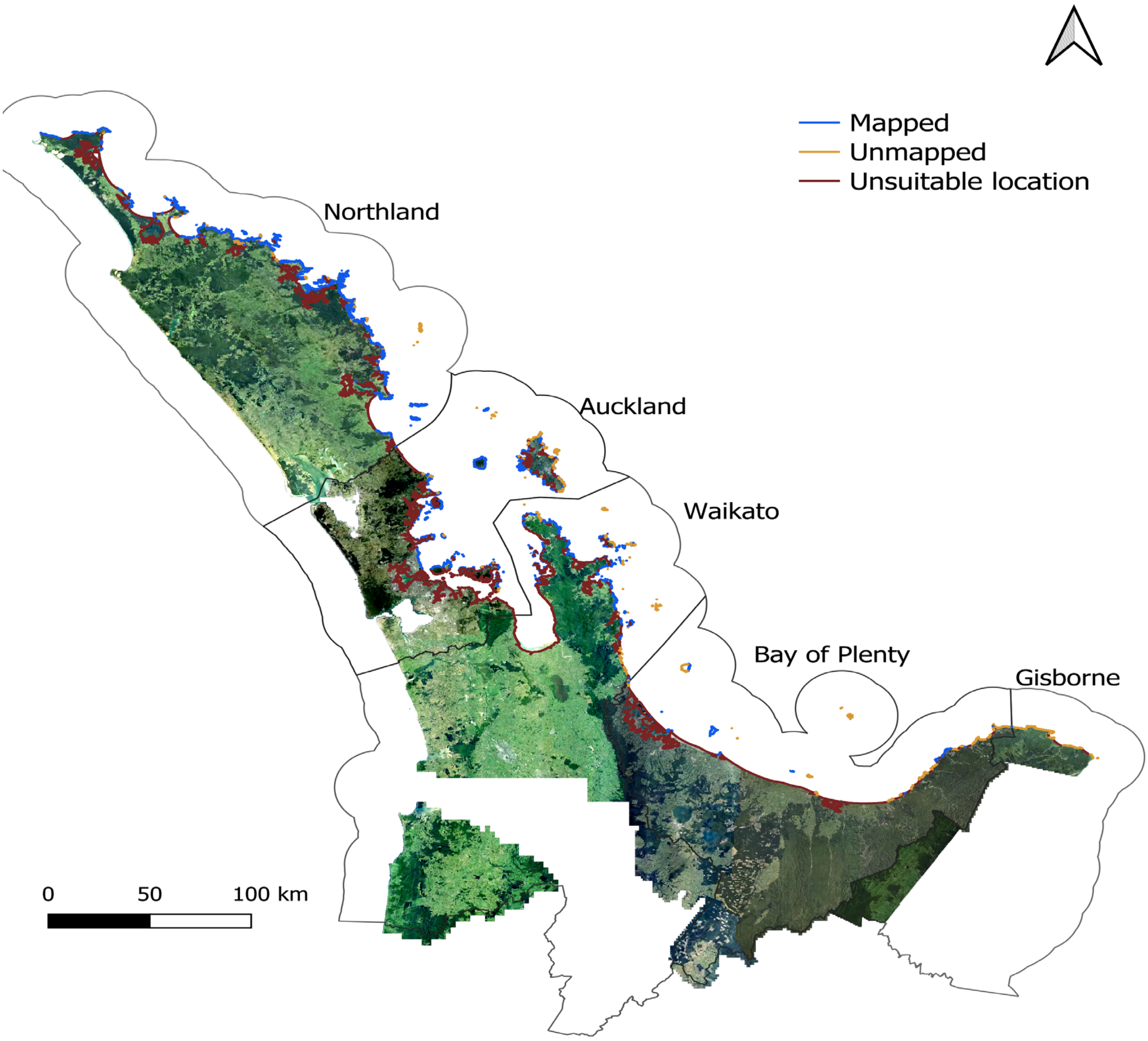

Figure 10: Coastline evaluated across northeastern New Zealand, including sections of coastline that were able to be mapped for urchin barrens (blue lines), that could not be mapped due to poor image quality or environmental conditions but were likely to be suitable for urchin barrens (e.g. had offshore rocky reef – orange lines) and that were unsuitable for urchin barrens so were excluded from mapping (e.g. estuaries and harbours – maroon lines)

3.2.2 Mapped areas of urchin barren

Overall, approximately 86% of all mapped habitats (see Table 1 for details of habitat types) were mapped using high resolution LINZ aerial imagery with the remaining areas mapped from Google Earth satellite imagery. All imagery was captured between 2020 and 2025 with approximately 96% captured from 2023 onwards.

In total approximately 22 km^2 of urchin barren was identified and mapped across northeastern New Zealand (Table 4, Figure 11 and Figure 12). Additionally, about 1.3 km^2 of unconfirmed urchin barren and about 2.8 km^2 of mosaiced habitat were mapped. These additional habitat types represented areas where we had low confidence with the overall barren extent or definitive classification, due to image quality issues or lack of ground truthing information, as well as patchy barren and/or turfing habitat with a high occurrence of macroalgae throughout (i.e. not continuous areas of barren, or macroalgae).

A further approximately 1.6 km^2 of turf and foliose algae habitat was mapped. These areas were free of canopy forming macroalgae, but due to location/context (e.g. low-profile reef surrounded by sand) and/or ground-truthing information could not be reasonably classified as urchin barren habitat.

Overall, 93.93 ± 1.45% of all mapped habitats were in waters less than 10 m deep with 88.93% of urchin barren polygons falling entirely within this depth band and 99.46% falling within the 20 m depth band (Table 4). Similarly, 88.23% of all mapped urchin barren area was within 200 m of the shoreline (Table 4)

At a regional level the greatest overall area of urchin barrens that was able to be mapped occurred in Northland (about 9 km^2), while the Gisborne Region, which was only partially represented within the study area, had only 0.01 km^2 of mapped barrens (Table 5). Despite a greater overall area of mapped urchin barrens within Northland the area of mapped urchin barrens relative to the length of evaluated coastline was similar to that mapped in Auckland, Waikato and Bay of Plenty, ranging from 0.1 km^2/km of evaluated coastline to 0.3 km^2/km of evaluated coastline (Table 5). Extrapolating these values out across areas of coastline that could not be mapped due to image or environmental issues we estimate that an additional 1 – 3 km^2 of urchin barren would likely have been mappable (if conditions allowed) per region between Northland and Bay of Plenty (Table 5). This would have taken the area of mapped urchin barrens from about 22 km^2 to about 28 km^2 across the entire survey region

It should be noted that the approximately 22 km^2 of mapped urchin barrens and 28 km^2 of mapped and potentially mappable barrens do not represent the total of urchin barren area that exists across the entire region. These values are likely to be an underestimation of total area, due to discussed limitations of the broad-scale mapping approach and represent only the area of urchin barren that could be effectively mapped via this methodology.

The influence of depth on mapped urchin barren area was reasonably consistent among the regions (excluding Gisborne, which had limited mapped areas of barren). Most barrens (84.04 ± 4.56%) were in shallow waters, less than 10 m deep regardless of region with >98% occurring in waters less than 20 m deep (Table 6). Distance from shore however was more variable. While the majority of identified urchin barren area in Northland, Auckland and Waikato occurred within 100 m of the shore, only one third of barren area recorded within the Bay of Plenty was within 100 m of the shore (Table 6). Similarly, 85% – 97% of all mapped barren area was found within 200 m of shore between Northland and Waikato but only 64% of total barren area was within 200 m within the Bay of Plenty. A lesser overall area of barrens was mapped in Bay of Plenty, with the largest single area derived from the waters surrounding Motiti Island. This island has extensive areas of shallow rocky reef, expanding well away from the coastline, that were classified as urchin barren. This area, relative to other mapped coastline within the region, and the limited area of barren that could be mapped, is likely to have influenced the lower percentage of barrens occurring within 200 m of the coastline.

Table 4: Total extent of mapped urchin barren and other mapped habitat types across northeastern New Zealand. Depth based on Bathymetric Model of New Zealand (2016).

Habitat types	Total mapped polygons (n)	Total mapped area (km^2)	Polygons ≤10 m depth (%)	Polygons ≤20 m depth (%)	Area ≤ 200 m from shore (%)
Urchin Barrens	6848	21.91	88.93	99.46	88.23
Urchin Barrens – Unconfirmed	247	1.31	91.50	100.00	79.37
Mosaic	1027	2.80	90.75	100.00	94.29
Turf and Foliose Algae	1092	1.55	97.07	99.63	83.85
Turf and Foliose Algae-Unconfirmed	34	0.03	97.06	100.00	100.00
Intertidal	42	0.26	100.00	100.00	89.43
Unconsolidated	347	0.88	92.22	100.00	100.00
		Average (± SE)	**93.93 (1.45)**	**99.87 (0.08)**	**90.74 (2.77)**

Table 5: Extent of mapped urchin barren across the surveyed regions.

Region	Total mapped area (km^2)	Area per km of coastline (km^2/km)	Additional potential area (km^2)	Mapped area + potential area (km^2)
Northland	8.97	0.01	1.69	10.66
Auckland	5.64	0.01	0.94	6.58
Waikato	5.77	0.02	1.60	7.38
Bay of Plenty	1.46	0.03	2.78	4.25
Gisborne	0.01	NA	NA	NA

Table 6: Extent of mapped urchin barrens relative to depth and distance from shore. Depth was determined using the Bathymetric Model of New Zealand (2016).

Region	Percent polygons ≤10 m depth (%)	Percent polygons ≤20 m depth (%)	Percent area ≤ 100 m from shore (%)	Percent area ≤ 200 m from shore (%)
Northland	93.08	99.80	67.62	87.98
Auckland	84.68	99.77	58.46	85.43
Waikato	88.92	99.50	73.37	97.32
Bay of Plenty	86.89	100.00	32.72	64.20
Gisborne	66.67	100.00	100.00	100.00
Average (± SE)	**84.01 (4.57)**	**99.81 (0.09)**	**66.43 (10.90)**	**86.99 (6.32)**

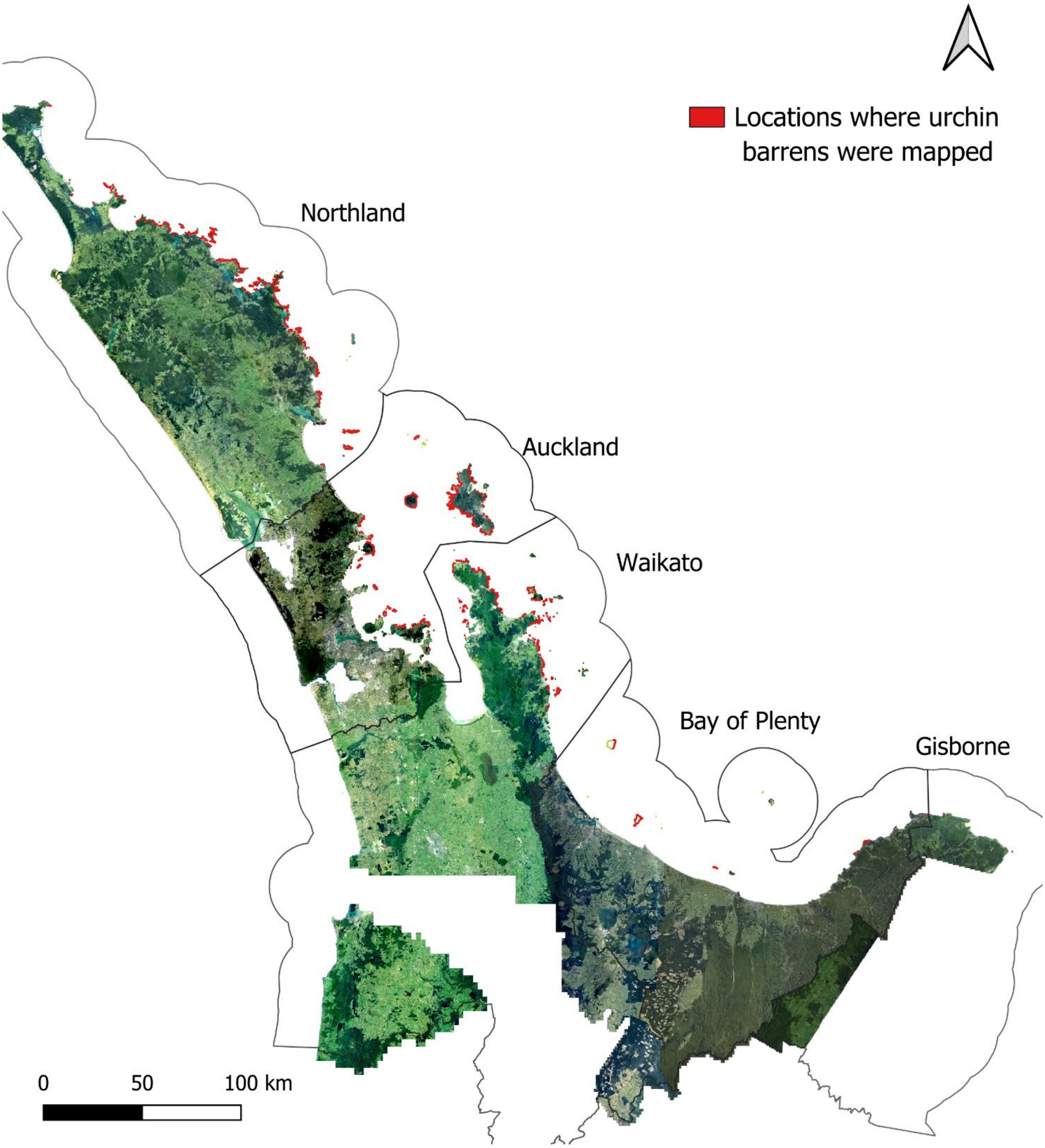

Figure 11: Locations across northeastern New Zealand where urchin barren habitat (red lines) was recorded. Note that visualisation at a regional scale over-represents urchin barren extent as the width of polylines have been exaggerated for clarity. Across the entire region urchin barrens were recorded along 36% of suitable coastline (e.g. with adjacent rocky reef) that was able to be mapped. Locations such as Whakaari/White Island and the Alderman Islands do not have any recorded urchin barrens as mapping was not able to be carried out due to poor image quality and/or unsuitable environmental conditions. Other habitat types mapped (including Urchin Barren – Unconfirmed) are not shown. See Figure 12 for examples of close ups of specific locations.

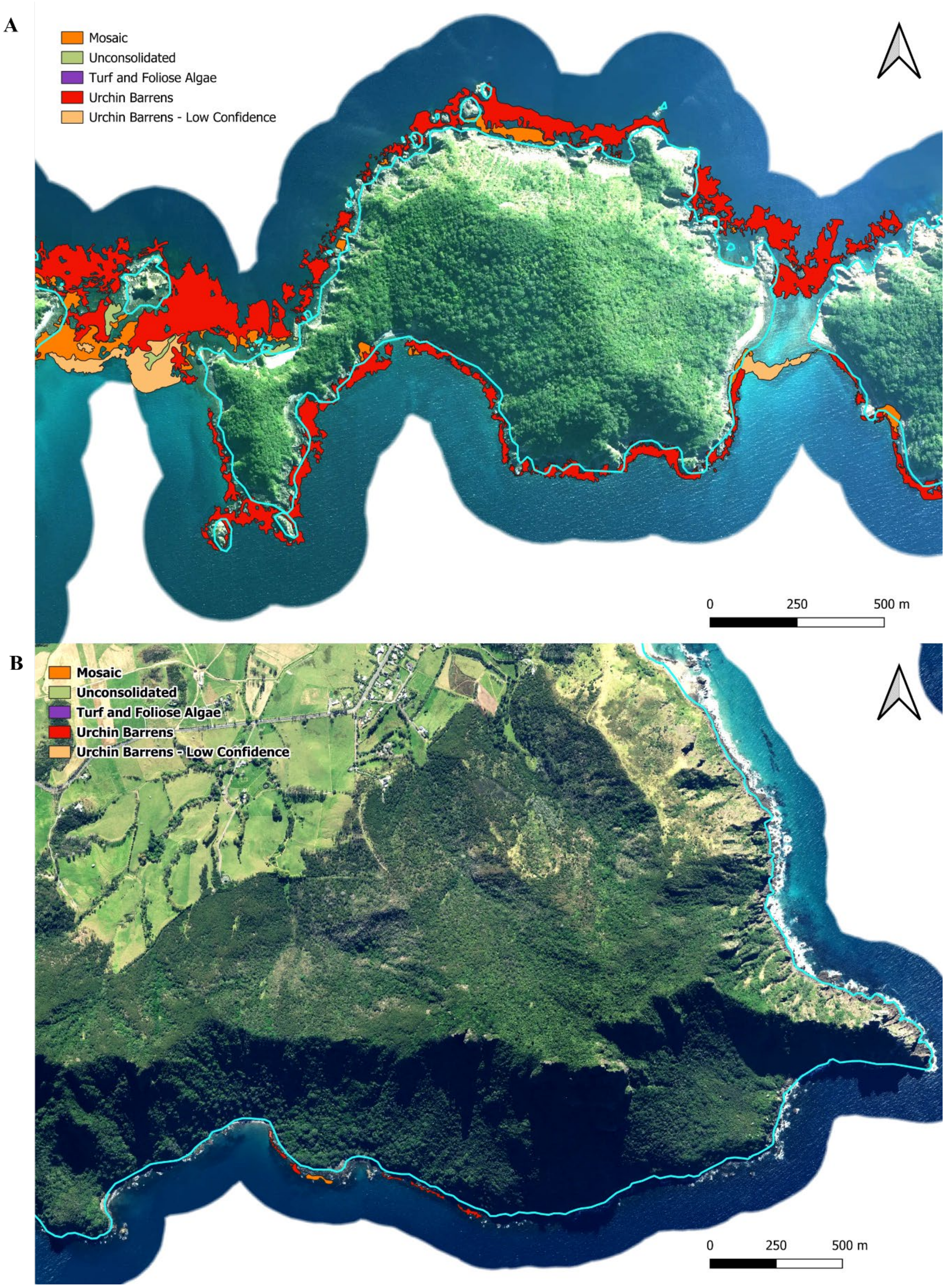

Figure 12: Close up examples of mapped urchin barrens (red polygons) around A) Whatupuke Is. (part of the Marotere or 'Chicken' Islands) and nearby B) Whangarei Heads. Extensive areas of urchin barren were recorded around Whatupuke Is. along with Mosaic, and Unconfirmed Urchin Barren (offshore extent uncertain) habitats while urchin barren habitat was limited along the Whangarei Heads coastline.

3.2.3 Validation

In total 91 polygons contained validation points, of which 75 were classified as Urchin Barren. Urchin barren mapping was considered to have a high degree of accuracy with 79.01% of ground truth points within Urchin Barren polygons classified as Urchin Barren/Urchin Barren Mosaic (Table 7). Short Vegetation (9%) and Kelp (8%) were the next most frequent non-urchin barren ground truth points found within Urchin Barren polygons, with very few Other Canopy Forming Macroalgae or Unconsolidated points identified.

As expected, Mosaic polygons, which represented heterogeneous habitat, had a more even spread of ground truthing points (Table 8), however ground truthing points labelled as Urchin Barren/Urchin Barren Mosaic still represented the most frequent label type (50%). This indicated that these habitats, although not strictly in a barren state still had a high occurrence of active grazing within them. Overall, approximately 60% of Urchin Barren/Urchin Barren Mosaic ground truthing points were captured within urchin barren polygons. Urchin Barren ground truthing points that were not captured within mapped areas of urchin barren typically occurred offshore, beyond the maximum mapped extent for that location.

Table 7: Producer accuracy assessment for urchin barren polygons

						Ground truth labels
Urchin barren polygons	**Urchin barrens/UB mosaic**	**Turf and Foliose Algae**	**Kelp/Mixed Algae**	**Other canopy forming macroalgae**	**Unconsolidated**	**Total**
	350	40	35	7	11	443
Producer accuracy (%)	79.01	9.02	7.90	1.58	2.48	100%

Table 8: Producer accuracy assessment for mosaic polygons

						Ground truth labels
Mosaic polygons	**Urchin barrens/UB mosaic**	**Turf and Foliose Algae**	**Kelp/Mixed Algae**	**Other canopy forming macroalgae**	**Unconsolidated**	**Total**
	18	7	9	1	1	36
Producer accuracy (%)	50.00	19.44	25.00	2.78	2.78	100%

In general expert annotators were consistent with the extent of urchin barrens they effectively mapped within each of the 20 comparison areas utilised (see Figure 13 for an example). Overall mean difference in mapped area between annotators was 9.5% ± 2.3%, however this varied with imagery source, with annotators using the LINZ Waikato 0.3 m 2023 – 2024 aerial imagery having the lowest relative difference (4.8% ± 1.7%) and annotators using the Auckland LINZ 0.05 m 2024 aerial imagery having the highest (13.2% ± 2.8%).

Figure 13: Example of validation exercise showing difference in urchin barren (red and green polygons) size between expert annotators. This example is of a comparison area (yellow polygon) on the northern side of Kawau Island and urchin barrens were mapped using Google Earth Satellite imagery.

3.2.4 Urchin Barren Extent

Across northeastern New Zealand (excluding Gisborne) urchin barrens were present within 200 m of the shore along 36.11 ± 2.45% of evaluated coastline (excluding sandy shores, harbours etc; Table 9; Figure 11). Extending the offshore limit to 400 m or including other habitat types which may also be associated with sea urchin grazing activities (Mosaic and Turf and Foliose Algae) resulted in a small increase in total extent of coastline with adjacent urchin barren habitats to 38.65 ± 2.18 and 43.30 ± 2.16 respectively (Table 9).

Within 200 m of the shore urchin barren extent varied from 33.36% in Northland to 43.41% in the Waikato. The most notable difference in extent of urchin barrens between the 200 m and 400 m transects was in Bay of Plenty where overall extent went from 34.51% of evaluated coastline to 39.36% of evaluated coastline (Table 9). When including other habitat types that may be associated with sea urchin grazing there was a consistent 6 – 8% increase in total extent across the regions (Gisborne excluded; Table 9).

Table 9: Extent of evaluated coastline with adjacent urchin barren and associated habitat types.

Region	**Percent of suitable coastline with adjacent Urchin Barren habitat (%)**		**Percent of suitable coastline with adjacent Urchin Barrens/Mosaic/Turf and Foliose Algae habitat (%)**
	200 m offshore transect	400 m offshore transect	200 m offshore transect
Northland	33.36	35.82	40.35
Auckland	33.16	34.90	42.08
Waikato	43.41	44.52	49.70
Bay of Plenty	34.51	39.36	41.08
Gisborne	NA	NA	NA
Average ± SE (exc. Gisborne)	**36.11 (2.45)**	**38.65 (2.18)**	**43.30 (2.16)**

We found that in general the extent of coastline with adjacent urchin barren habitats mapped in studies between 2016 and 2020 was similar to the extent calculated in this current study (Table 10). These comparisons were made with those recent studies that had the most similar mapping approaches (e.g. use of satellite imagery and drop camera data for validation), however there were inconsistencies between the studies in terms of the habitats that were mapped and we found that areas that had previously been mapped as Urchin Barren habitat (which also included turfing algae) were mapped as Mosaic or Turfing and Foliose Algae in our study (e.g. Figure 14). Comparisons of Urchin Barren habitat extent were therefore closest when comparing previously mapped Urchin Barren habitats to an amalgamated habitat type that included Urchin Barrens, Mosaic and Turf and Foliose Algae from the current study (Table 10).

Table 10: Comparison of urchin barren extent between habitat mapping projects conducted using aerial/satellite imagery sourced between 2016–2020 and the current study (imagery sourced between 2023–2024). Comparisons based on 200 m offshore transects. [1]Previous studies included Turfing Algae and some Mosaic habitats within the Urchin Barren habitat so for comparison we have compared these values to our Urchin Barren habitat and an amalgamated habitat including Urchin Barrens, Mosaic and Turf and Foliose Habitats. [1]Locations mapped by Lawrence (2019). [2]Locations mapped by Dartnall (2022). [3]Locations mapped by Shears and Lawerence (2024).

	Extent of Urchin Barren habitat (%)		
	Previous studies	Current study	
Location	**2016 – 2020 – Urchin Barren or Urchin barren/Turfing Algae**	**2023 – 2024 – Urchin Barren Habitat only**	**2023 – 2024 – Urchin barren/Mosaic/Turf and Foliose Algae**
Mimiwhangata[1]	56%	57%	60%
Mokohinau Islands[1]	58%	33%	40%
Hauturu-o-Toi[2]	83%	77%	87%
Cape Rodney – Okakari Point Marine Reserve[1]	25%	3%	42%
Noises[2]	88%	72%	83%
Great Mercury[3]	54%	40%	50%
Aitu/Green/Korapuki[3]	71%	41%	54%
Average ± (SE)	**62% (8%)**	**46% (10%)**	**59% (7%)**

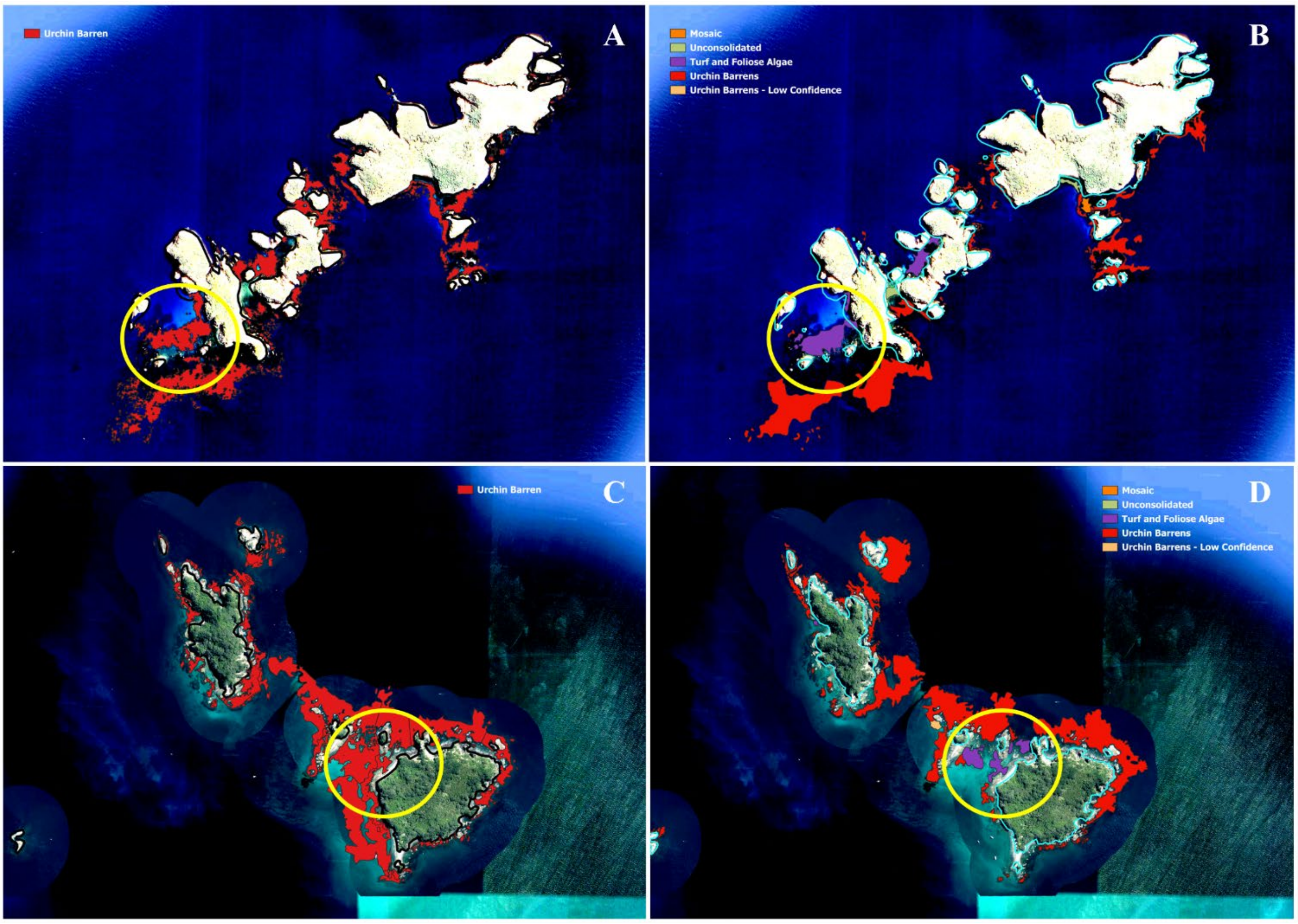

Figure 14: Comparisons of mapped area of urchin barrens at the Mokohinau Islands based on imagery sourced in A) 2018 and B) 2023 and The Noises based on imagery sourced in C) 2018 and D) 2024. Contrast in habitat types mapped are evident with mapping in 2018/2019 only including an Urchin Barren habitat classification, whereas mapping in 2023/2024 also included Mosaic and Turf and Foliose Algae habitats. At both locations areas classified as Urchin Barren in 2018/2019 have been classified as Mosaic or Short Vegetation in 2023/2024 (yellow circled areas) leading to discrepancies in total urchin barren extent.

4 DISCUSSION

4.1 Collate quantitative data on the current and historical extent of urchin barrens in New Zealand

Confirmation via the geodatabase of urchin barren existence throughout large parts of New Zealand was not surprising given the large body of literature, and anecdotal evidence, highlighting their widespread occurrence. The majority of information able to be included in the geodatabase was sources from northeastern New Zealand and was in the form of detailed habitat maps. Habitat mapping created at the time (Ballantine et al. 1973; Ayling et al. 1981; Berben et al. 1988), or based on analysis of historical imagery (Dartnall 2022; Kerr et al. 2024) indicate that urchin barrens were a common component of many shallow reef ecosystems in the 1970s but not the 1940s and 1950s. Urchin barren extent increased over the next three to four decades at most locations resurveyed between 2003 – 2019 and were a dominant habitat type throughout other standalone mapping projects carried out over the same time period (Lawrence 2019; Dartnall 2022; Kerr et al. 2024; Shears & Lawerence 2024). In contrast urchin barren habitat declined within the Cape Rodney – Okakari Point Marine Reserve between 1977 (first habitat map) and 2006 (Leleu et al. 2012) and remained less extensive through to surveys done in 2019 (Lawrence 2019) and as part of the current study. These findings are supported by long-term diver surveys which show that following marine protection kelp forests recovered within the marine reserve by the mid to late 1990s (Shears 2002) and have remained stable over the past 25 years (Peleg et al. 2023). Habitat maps of the Tāwharanui and Te Whanganui-A-Hei (Cathedral Cove) Marine Reserves in 2006 and 2014/2015 respectively also indicated a greater extent of urchin barren habitat outside of the reserves relative to inside (Haggitt 2017; Kibele & Shears 2017; Kerr et al. 2024).

Spatial and temporal information on urchin barren extent/distribution was limited outside of northeastern New Zealand but the New Zealand wide survey of rocky reef community characteristics conducted by Shears and Babcock (2007) in 1999/2000 did indicate that in many locations extensive barrens had already formed by the late 1990s. These included Queen Charlotte Sound (QCS), Tasman Bay, parts of Fiordland and Patterson Inlet, and Stewart Island. Historical aerial imagery from QCS show that in places where urchin barrens are now present canopy forming macroalgae was the dominant habitat type in 1958 (Lafont & Shears 2025), while within Tasman Bay dramatic declines in canopy forming macroalgae occurred between 1966 and 1988 (Davidson 1992). While the exact causes of and time frame over which the loss of these canopy forming macroalgae occurred, overgrazing by kina has been suggested as a likely contributor in both cases (Davidson 1992; Lafont & Shears 2025). What is clear however is that kina barrens have remained as a stable part of the reef ecosystem at many South Island locations since the 1999 surveys. This is confirmed by the presence of urchin barrens within habitat maps compiled for the reefs within and surrounding the Long Island (QCS) and Ulva Island (Patterson Inlet) Marine Reserves in 2015 (Department of Conservation, unpublished data), the dominance of urchin barrens throughout QCS as surveyed in 2018 by Lafont and Shears (2025) and multiple studies that have conducted experimental research within urchin barren habitats throughout Fiordland (e.g. Wing & Wing 2015; Udy et al. 2019).

While information from multiple time points in places such as QCS and Patterson Inlet do indicate that urchin barrens have persisted over the past 20 + years the lack of consistency between data sources makes it hard to quantify whether actual urchin barren extent has changed throughout this period. Even in northeastern New Zealand where habitat maps exist at several locations from different points in time the methodological approach for many have been significantly different meaning that comparisons of urchin barren extent based on calculations of mapped urchin barren area should be approached cautiously. At a national scale, a repeat of the diver surveys conducted by Shears and Babcock (2007) is likely to provide the most effective means of quantifying the current extent of urchin barren habitats and allowing a direct comparison with values calculated in 1999/2000.

To establish timelines for when urchin barrens first occurred across different regions of New Zealand consideration should be given to use of local ecological knowledge (LEK). Local ecological knowledge offers a potentially valuable source of information on species characteristics and ecological state gained through lived experience or as collective knowledge passed down through generations (Teixeira et al. 2013). The use of LEK has been successfully integrated into a number of marine and coastal studies to document changes in ecological health or species abundances and guide management practices (Jung et al. 2011; Taylor et al. 2011; Chewying et al. 2025; Hansford et al. 2025). It has also been used as a way of mapping habitat types (Lauer & Aswani 2008; Teixeira et al. 2013) thus it has potential for use in establishing timelines for the onset of urchin barrens. Mana whenua are likely to hold the longest collective understanding of changes that have occurred along the coastlines of their rohe. Yarning circles with Walbunja Traditional Owners in New South Wales, Australia have previously been used in conjunction with western science to understand kelp forest – urchin barren dynamics (Chewying et al. 2025). Additionally, in locations such as the Poor Knights Islands, where species such as *C. rodgersii* are increasing and have begun to form extensive areas of barrens, frequent divers and diver operators are likely to hold valuable insights into these changes. In some locations local ecological knowledge could potentially be paired with historical imagery which has itself been used to successfully document the onset of urchin barrens (e.g. Dartnall 2022; Kerr et al. 2024; Lafont & Shears 2025), to help establish timelines for kelp loss and/or recovery.

4.2 Collect comprehensive baseline information on the extent of urchin barrens in northeastern New Zealand

Habitat mapping of the entire northeastern New Zealand region, based on recent (mostly 2023 – 2024) and freely available imagery, indicated that urchin barren habitat is currently found along about 36% of suitable coastline (coastline with adjacent rocky reef) and are typically associated with shallow reefs less than 10 m deep on open to exposed coasts. This is consistent with previous descriptions of where the majority of urchin barren occur (Grace 1983; Shears & Babcock 2004). Although we did not distinguish between urchin barrens formed by *E. chloroticus* and *C. rodgersii,* recent region wide surveys indicate that *E. chloroticus* remains the dominant barren forming sea urchin through most of northeastern New Zealand. However, the influence of *C. rodgersii* is growing, particularly along the coastline north of Cape Brett and at many of the offshore island groups that are more heavily influenced by the warm East Auckland Current (A. Spyksma, unpublished data).

Despite no imagery being captured specifically for this study we found that recent LINZ aerial imagery was well suited for broad-scale mapping of shallow water urchin barren habitats due to its high resolution (0.075 – 0.4 m). This data source was however limited to a maximum extent of 200 m offshore (via the LINZ plugin) and while in some situations we could use Google Earth imagery to supplement the LINZ data the lower resolution of this imagery was problematic and often did not provide any additional information for reefs that extended farther offshore, or those completely isolated from the coastline. This is a common problem with freely available satellite imagery (Lewis et al. 2023) which can potentially be overcome by purchasing high resolution imagery. This would become expensive over the entire northeastern New Zealand region and was considered too costly for the aims of this study but should be considered for more detailed surveys of key areas where management aimed at addressing sea urchin overgrazing and kelp recovery are planned. While the producer accuracy assessments of 79% and strong expert agreement indicated that we could have high confidence in the classification and extent of area we were able to effectively map as urchin barrens, our limited ability to map rocky reefs beyond 200 m from the shore or in deeper areas, coupled with a high number of urchin barren ground truthing validation points that fell outside of mapped polygons suggests that our results of about 22 to 28 km^2 of mapped urchin barrens are likely to be an underestimation of total area. Approximately 13% of urchin barren validation points occurred farther offshore or in locations that urchin barrens were not mapped at all and these typically occurred in areas of steep reef, where depth quickly exceeded 10 m (e.g. Cape Brett or the Mokohinau Islands).

We expect that as *C. rodgersii* become more abundant, estimations of total barren habitat made primarily from aerial imagery will become more inaccurate due to the greater depth range of this species (relative to *E. chloroticus*; A. Spyksma unpublished data) and its known capacity for forming extensive barrens below 15 m (Ling & Keane 2024). Previous habitat mapping projects at locations such as the Mokohinau Islands, which have extensive areas of reef deeper than 15 m, have highlighted the challenges of using aerial imagery to accurately quantify habitats below 15 m deep and on steep sloping reefs (Lawrence 2019). At a site level these issues can potentially be overcome, and more accurate area estimates generated, through the collection of extensive ground truthing data e.g. the approach of Kerr et al. (2024).

However environmental conditions and water clarity will always be an unavoidable factor when working with aerial imagery, as was the case for much of the coastline through eastern Bay of Plenty and Gisborne, and the collection of extensive ground truthing information over an areas as large as northeastern New Zealand would quickly become prohibitively expensive. Thus, our approach to mapping was considered an acceptable trade-off between overall cost and achievable detail over the entire northeastern New Zealand region and provides an indicator of the extent of urchin barrens on shallow reefs. Furthermore, by using the proportion of coastline with adjacent urchin barren, rather than an area calculation, as the primary means for estimating urchin barren extent we have provided an estimate that is less reliant on accurate area calculations of urchin barren polygons that extend into deeper water and avoids the need to relate urchin barren area to the problematic reef area spatial layers that exist throughout the region.

When the proportion of coastline with adjacent urchin barrens relative to total suitable coastline (i.e. with adjacent rocky reefs) calculations were applied to other recent mapping exercises, we found that estimations of urchin barren extent were consistent between studies. The largest discrepancies came from the habitat classifications that were used to calculate overall urchin barren extent. Where we included a Mosaic and Turf and Foliose Algae habitat classification to account for highly heterogenous habitats (urchin barren interspersed with kelp) and those that were devoid of macroalgae, but also not likely to be urchin barren, previous studies only included an Urchin Barren (Lawrence 2019; Dartnall 2022) or Urchin Barren/Turfing Algae (Shears & Lawerence 2024) category. It is highly likely that in previous studies areas we classified as Mosaic or Turf and Foliose Algae would have been incorporated into these broader urchin barren classification and visual comparisons of results between study sites/mapping approaches support this. Where extent of Urchin Barren habitat in our study was compared to these previously used Urchin Barren habitat classifications our estimates were similar but consistently lower. Including Mosaic and Turf and Foliose Algae categories increased our estimate extent of urchin barren habitat and values became more consistent with previous studies. Including Mosaic and Short vegetation into our overall estimations of urchin barren extent across northeastern New Zealand caused urchin barren extent to increase from about 36% to 42%. This increase highlights the caution that is needed when comparing between habitat maps produced using different methodological approaches and indicates that future mapping exercises aimed at specifically comparing extent of urchin barren habitat should follow the same habitat classification schema as this one.

5 FULFILMENT OF BROADER OUTCOMES

This study represents the first attempt to collate information on the distribution and extent of urchin barrens across New Zealand and is also the first to comprehensively map urchin barren habitats across an entire region (northeastern New Zealand). Understanding the distribution and extent of urchin barren habitats is an important part of identifying the patterns and causes of kelp forest loss and is critical for effectively evaluating the success of management actions aimed at reducing the extent of urchin barren habitat and initiating kelp forest recovery (Eger et al. 2022). Importantly in northeastern New Zealand current estimations of urchin barren extent based primarily on imagery sourced between 2023 – 2024 will provide a starting point for evaluating potential ecological effects of recent changes to rock lobster fisheries regulations that have occurred in CRA 1 and CRA 2. Additionally, for some of the High Protection Areas that are proposed for the Hauraki Gulf Marine Park this data may serve as an important baseline for urchin barren extent prior to the establishment of marine protection. This data is likely to be particularly useful at locations such as Hauturu-o-Toi, Kawau, the Noises and Slipper Island where urchin barrens were extensive, primarily occurred close to shore or the reef profile were conducive to mapping full extent farther offshore. At these locations we have high confidence that the majority of urchin barren habitat was captured. Follow up mapping across these areas could then be used to document any change to urchin barren extent that occur following protection.

The geodatabases created could also support tangata whenua or community led initiatives, such as the recent Rehuotane Ki Tai 186a temporary closure, trial sea urchin removals and upcoming mātaitai application for the Tutukaka coastline (Hansford et al. 2025). Even relatively small-scale projects such as this can involve significant time and resource investment to conduct initial and ongoing habitat mapping, making access to free, up to date information on urchin barren extent a valuable resource that should help to lower entry barriers for groups looking to actively manage their local marine environment.

6 CONCLUSIONS

Overall, compiling information on the distribution and extent of urchin barren habitat has confirmed their widespread occurrence throughout New Zealand. While information sources were generally limited outside of northeastern New Zealand, and in some cases the most recent data was about 25 years old, it is clear that expansive urchin barrens currently exist, or have been documented to occur, throughout parts of northeastern New Zealand, the Marlborough Sounds, Tasman Bay, Fiordland and Stewart Island.

In northeastern New Zealand urchin barren habitat is currently extensive and urchin barrens were recorded from the Far North through to East Cape. The mapping approach used, based on freely available imagery and existing drop camera ground truthing information, provided a reliable and cost-effective means of establishing the distribution and extent of shallow water urchin barren habitat through much of the region. Environmental conditions limited the usefulness of aerial mapping along the eastern Bay of Plenty and Gisborne coastlines and deeper reefs or those with steep profiles were also less reliably mapped. Despite these limitations this approach was probably the most effective way of working across such a large spatial scale. Nevertheless, obtaining additional imagery under better conditions could expand the area mapped and improve mapping accuracy. This could be obtained by purchasing satellite imagery or using targeted aerial or drone surveys when conditions are favourable.

The geodatabases created as part of this project will provide freely available information on the extent and distribution of urchin barren habitat across New Zealand and will serve as an important baseline for future work, evaluating recent and upcoming management actions aimed at restoring kelp forest ecosystems and will lower the barrier for mana whenua and local communities wanting to be actively involved in managing their marine spaces.

7 POTENTIAL RESEARCH

Following the conclusion of this project there are a number of options for further work associated with tracking the distribution of and extent of urchin barrens throughout New Zealand:

1. Large scale mapping exercises like those done in this study for northeastern New Zealand may be best repeated over longer times scales (e.g. a 10–15-year basis). Comparisons of sites mapped from imagery gathered between 2016 and 2020 and then between 2023 and 2024 (current study) indicated little overall change in the extent of urchin barren habitat that was mapped between these time points. Established urchin barrens are often highly stable features (Filbee-Dexter & Scheibling 2014) and outside of areas subjected to active sea urchin management, which can promote rapid kelp recovery (Miller et al. 2024), we would not expect to see large scale changes in the general size of urchin barrens that were able to mapped in this study on a short-term basis. Repeating this mapping on a 10–15-year basis would allow a consistent, region wide time series to be created of urchin barren extent and would most likely be at a regular enough interval to capture any significant changes occurring on shallow, nearshore reefs throughout the region, including changes as a result of changes to fisheries regulations for urchin predators or marine protection.
    a. While we expect that significant barren expansion will most likely occur over the coming decades throughout northeastern New Zealand as a consequence of increasing *C. rodgersii* abundances this is more likely to occur in deeper waters that are less reliably mapped by aerial imagery alone. Changes occurring on shallow reefs will be less obvious considering the widespread existence of shallow water urchin barrens already, but would still be detected over a 10–15-year time period (particularly if barrens formed in locations that they are not currently present). Documenting changes in urchin barren extent caused by *C. rodgersii* would be better suited for specific areas and incorporate diver or extensive drop camera surveys that can capture deeper reef environments. Fisheries New Zealand could look to support existing work by DOC and Regional councils if greater information on *C. rodgersii* is required.
    b. While the current project leveraged freely available imagery it would have undoubtedly benefitted from data captured specifically for the purposes of mapping urchin barren habitats. Commissioning a series of aerial surveys that specifically focussed on the coastline and nearshore environment would help to ensure that imagery is fit for purpose. Across northeastern New Zealand for example this could

be captured over a 1 – 2-year period if needed and target days that present the best opportunity to gather high quality imagery of the coastal area e.g. at low tide following prolonged periods of settled weather.

2. In-depth ecological surveys at priority sites could be done to capture the full extent of urchin barren habitats. This could include diver surveys, extensive drop camera and high-resolution satellite imagery. In-depth surveys would allow for greater estimation of barren habitat area within specific locations and provide a means to capture changes in barrens that are farther offshore or in deeper water that are not easily resolvable through aerial imagery alone.

3. High resolution bathymetry could be incorporated into future mapping projects. High resolution bathymetry data derived from multibeam echo sounding (MBES) is seen as an increasingly relevant tool for the production of accurate benthic habitat maps (Che Hasan et al. 2014). Across the country LINZ provides access to a repository of available bathymetric data, including data captured by MBES however there remains significant gaps that should be filled to make this data useful at a regional level. Due to these inconsistencies, we opted not to use available MBES in this study however inclusion in future studies at a site or region level would be beneficial for several reasons:
    a. MBES backscatter and backscatter angular response can be used to distinguish between seafloor characteristics allowing areas of hard rocky reef to be effectively distinguished from unconsolidated material such as sand and mud (Che Hasan et al. 2014). It was clear that currently available data on the extent of rocky reef habitats throughout the country poorly represents the actual distribution and extent of rocky reef habitat. This is in part because of the current reliance on depth soundings alone to quantify reef extent.
    b. MBES data can provide information on rugosity, slope and reef orientation. These characteristics can be used in conjunction with ground truthing information to build predictive models as to where urchin barren habitat is likely to occur (Sward et al. 2022). This would be beneficial, particularly for areas of deeper reef which are problematic for aerial imagery alone.
    c. MBES data also provides a 3D representation of reef extent. Current mapping approaches such as those used in this study provide a 2D representation of reef habitats when in reality they are representing 3D space. This inevitably leads to an inherent underestimation of area, particularly on steeper sections of reef where representations of urchin barrens in 2D space may only show up as a very narrow band. These issues could be addressed using MBES data.

   Fisheries New Zealand could liaise with LINZ and other entities (e.g. the New Zealand Navy) to conduct MBES surveys across priority locations (e.g. northeastern New Zealand) to fill gaps in the current MBES dataset that is available. Particular attention should be given to including high resolution data in shallow waters (<5 m). This area is crucial for effective mapping of urchin barren habitats but is often missed due to vessel navigation/ safety issues.

4. To understand urchin barren extent at a national scale a nationwide survey of urchin barren habitat extent could be done following a similar method to Shears and Babcock (2007). Considering that the most recent information on the extent of urchin barrens in many areas outside of northeastern New Zealand is based on data that is now 25 years old and an updated understanding of the current distribution and extent of urchin barren habitats is merited and would be useful for further understanding of the drivers and patterns of kelp forest loss across the country. As it is unlikely that aerial imagery would be useful across the entire range of locations urchin barrens have been reported, a diver-based survey would likely be the most effective approach. Using a methodological approach like that used in 1999/2000 and across the same range of locations would allow an up-to-date evaluation of urchin barren distribution and extent. Additional priority locations could also be included as needed. Conducting a survey programme at this scale would however require significant funding to achieve.

5. To incorporate local ecological knowledge and mātauranga into our understanding of change in reefs over time a small project could develop a methodology for incorporating LEK into habitat mapping practices and establishing timelines for the onset of urchin barren habitat. This project would be well suited as a project for a postgraduate student and could incorporate marine science, social science and mātauranga Māori.

8 ACKNOWLEDGEMENTS

This work was completed under Objective 1-4 of Fisheries New Zealand project ZBD2023-03. Thanks to Jean Davis for guiding the project and reviewing the document alongside Ian Tuck. Additional thanks go to Bruce Hartill, Phil Ross, Charlotte Mortimer and the Fisheries New Zealand Biodiversity Research Working Group for providing useful discussion and suggestions. Thanks to staff and students at the Leigh Marine Laboratory for assistance with field work and to Nick Shears for providing useful feedback throughout the project. Lastly, a huge thank you to all those who provided information on local areas, additional ground truthing information or access to data sources for collating information on the distribution of urchin barrens across New Zealand.

9 REFERENCES

Andrew, N.L.; Macdiarmid, A.B. (1991). Interrelations between sea-urchins and spiny lobsters in northeastern New Zealand. *Marine Ecology Progress Series*, *70*, 211-222. https://doi.org/10.3354/meps070211

Ayling, A.; Cumming, A.; Ballantine, W. (1981). Map of shore and subtidal habitats of the Cape Rodney to Okakari Point Marine Reserve, North Island, New Zealand in 3 sheets, scale 1: 2000. *Department of Lands and Survey, Wellington*.

Ayling, A.M. (1978). *Cape Rodney to Okakari Point Marine Reserve Survey*.

Balemi, C.A.; Pan, S.; Shears, N.T. (2025). Broad dietary niche of irrupting subtropical sea urchin exacerbates threats to multiple temperate reef habitats. *Marine Ecology Progress Series*, *757*, 131-144.

Balemi, C.A.; Shears, N.T. (2023). Emergence of the subtropical sea urchin Centrostephanus rodgersii as a threat to kelp forest ecosystems in northern New Zealand. *Frontiers in Marine Science*, *10*. https://doi.org/10.3389/fmars.2023.1224067

Ballantine, W.; Grace, R.; Doak, W. (1973). Mimiwhangata marine report. *Turbott & Halstead and New Zealand Breweries Limited, Auckland*.

Berben, P.H.; McCrone, A.; Creese, R.G.; Ballantine, W.J. (1988). *The Mokohinau Islands: a marine survey: with additional notes on the history, climate and terrestrial environments of the group* Leigh Lab Bulliten, 1-9.

Borfecchia, F.; Consalvi, N.; Micheli, C.; Carli, F.M.; Cognetti De Martiis, S.; Gnisci, V.; Piermattei, V.; Belmonte, A.; De Cecco, L.; Bonamano, S. (2019). Landsat 8 OLI satellite data for mapping of the Posidonia oceanica and benthic habitats of coastal ecosystems. *International Journal of Remote Sensing*, *40*, 1548-1575.

Che Hasan, R.; Ierodiaconou, D.; Laurenson, L.; Schimel, A. (2014). Integrating Multibeam Backscatter Angular Response, Mosaic and Bathymetry Data for Benthic Habitat Mapping. *PLoS ONE*, *9*, e97339. https://doi.org/10.1371/journal.pone.0097339

Chewying, K.; Gibbs, M.; Przeslawksi, R.; Rogers, K. (2025). Braiding Indigenous oral histories and habitat mapping to understand urchin barrens in southern New South Wales. *Marine and Freshwater Research*, *76*, NULL-NULL.

Dartnall, L. (2022). *The extent of kina barrens over time at Hauturu-o-Toi and the Noises Islands*. University of Auckland Thesis.

Davidson, R.J. (1992). *A report on the intertidal and subtidal ecology of the Able Tasman National Park*.

Department of Conservation; Ministry of Fisheries. (2011). Coastal Marine Habitats and Marine Protected Areas in the New Zealand Territorial Sea: A Broad Scale Gap Analysis. In (pp. 50): Department of Conservation and Ministry of Fisheries Wellington, New Zealand.

Dromgoole, F. (1964). The depredation of Ecklonia radiata beds by the sea urchin Evechinus chloroticus. *Tane*, *10*, 120-122.

Eger, A.M.; Blain, C.O.; Brown, A.L.; Chan, S.S.W.; Miller, K.I.; Vergés, A. (2024). Kelp forests versus urchin barrens: a comparison of ecosystem functions and services provided by two alternative stable marine habitats. *Proceedings of the Royal Society B: Biological Sciences*, *291*, 20241539. https://doi.org/doi:10.1098/rspb.2024.1539

Eger, A.M.; Layton, C.; McHugh, T.A.; Gleason, M.; Eddy, N. (2022). *Kelp Restoration Guidebook: Lessons Learned from Kelp Projects Around the World*. The Nature Conservancy.

Eger, A.M.; Marzinelli, E.M.; Beas-Luna, R.; Blain, C.O.; Blamey, L.K.; Byrnes, J.E.K.; Carnell, P.E.; Choi, C.G.; Hessing-Lewis, M.; Kim, K.Y.; Kumagai, N.H.; Lorda, J.; Moore, P.; Nakamura, Y.; Pérez-Matus, A.; Pontier, O.; Smale, D.; Steinberg, P.D.; Vergés, A. (2023). The value of ecosystem services in global marine kelp forests. *Nature Communications*, *14*, 1894. https://doi.org/10.1038/s41467-023-37385-0

Filbee-Dexter, K.; Scheibling, R.E. (2014). Sea urchin barrens as alternative stable states of collapsed kelp ecosystems. *Marine Ecology Progress Series*, *495*, 1-25. https://doi.org/10.3354/meps10573

Grace, R.V. (1983). Zonation of sublittoral rocky bottom marine life and its changes from the outer to the inner Hauraki Gulf, northeastern New Zealand. *Tane*, *29*, 97-108.

Haggitt, T. (2016). *Long Island - Kokomohua Marine Reserve Habitat Mapping*. Report prepared by eCoast for Department of Conservation.

Haggitt, T. (2017). *Te Whanganui a Hei Marine Reserve Habitat Mapping*. Report prepared by ECoast for Department of Conservation.

Hansford, J.; Edney, G.; Wellington, P.; Solomon, R.; Amos, C.; Clueard, H.; Holdsworth, J. (2025). *A shallow water benthic habitat survey and trial kina removals in the Whāngai Mokopuna Rohe Moana* (New Zealand Aquatic Environment and Biodiversity Report No. 358).

Hossain, M.S.; Bujang, J.S.; Zakaria, M.H.; Hashim, M. (2016). Marine and human habitat mapping for the Coral Triangle Initiative region of Sabah using Landsat and Google Earth imagery. *Marine Policy*, *72*, 176-191. https://doi.org/https://doi.org/10.1016/j.marpol.2016.07.003

Jung, C.A.; Dwyer, P.D.; Minnegal, M.; Swearer, S.E. (2011). Perceptions of environmental change over more than six decades in two groups of people interacting with the environment of Port Phillip Bay, Australia. *Ocean & Coastal Management*, *54*, 93-99. https://doi.org/https://doi.org/10.1016/j.ocecoaman.2010.10.035

Kerr, V.C.; Grace, R.V.; Shears, N.T. (2024). Estimating the extent of urchin barrens and kelp forest loss in northeastern Aotearoa, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, 1-22. https://doi.org/10.1080/00288330.2024.2336081

Kibele, J. (2016). Benthic Photo Survey: Software for Geotagging, Depth-tagging, and Classifying Photos from Survey Data and Producing Shapefiles for Habitat Mapping in GIS. *Journal of Open Research Software*, *4*. https://doi.org/10.5334/jors.104

Kibele, J. (2017). *Submerged Habitats from Space: Increasing Map Production Capacity with New Methods and Software* University of Auckland Thesis.

Kibele, J.; Shears, N. (2017). *Mapping rocky reef habitats on the eastern Coromandel Peninsula with multispectral satellite imagery*. Waikato Regional Council.

Krumhansl, K.A.; Okamoto, D.K.; Rassweiler, A.; Novak, M.; Bolton, J.J.; Cavanaugh, K.C.; Connell, S.D.; Johnson, C.R.; Konar, B.; Ling, S.D.; Micheli, F.; Norderhaug, K.M.; Perez-Matus, A.; Sousa-Pinto, I.; Reed, D.C.; Salomon, A.K.; Shears, N.T.; Wernberg, T.; Anderson, R.J.; Barrett, N.S.; Buschmann, A.H.; Carr, M.H.; Caselle, J.E.; Derrien-Courtel, S.; Edgar, G.J.; Edwards, M.; Estes, J.A.; Goodwin, C.; Kenner, M.C.; Kushner, D.J.; Moy, F.E.; Nunn, J.; Steneck, R.S.; Vasquez, J.; Watson, J.; Witman, J.D.; Byrnes, J.E. (2016). Global patterns of kelp forest change over the past half-century. *Proceedings of the National Academy of Sciences*, *113*, 13785-13790. https://doi.org/10.1073/pnas.1606102113

Kulins, S. (2021). *Investigating the Ecological Effects of Long Bay-Okura Marine Reserve*. University of Auckland Thesis.

Lafont, D.J.; Shears, N.T. (2025). Assessing drivers of variation in the distribution of macroalgal forests across a complex marine environment in Central New Zealand. *Marine Environmental Research*, *209*, 107171. https://doi.org/https://doi.org/10.1016/j.marenvres.2025.107171

LaScala-Gruenewald, D.E.; Grace, R.V.; Haggitt, T.R.; Hanns, B.J.; Kelly, S.; MacDiarmid, A.; Shears, N.T. (2021). Small marine reserves do not provide a safeguard against overfishing. *Conservation Science and Practice*, *3*, e362. https://doi.org/https://doi.org/10.1111/csp2.362

Lauer, M.; Aswani, S. (2008). Integrating indigenous ecological knowledge and multi-spectral image classification for marine habitat mapping in Oceania. *Ocean & Coastal Management*, *51*, 495-504. https://doi.org/https://doi.org/10.1016/j.ocecoaman.2008.04.006

Lawrence, K.R. (2019). *Mapping long-term changes in reef ecosystems using satellite imagery* University of Auckland Thesis. Auckland, New Zealand.

Leleu, K.; Remy-Zephir, B.; Grace, R.; Costello, M.J. (2012). Mapping habitats in a marine reserve showed how a 30-year trophic cascade altered ecosystem structure. *Biological Conservation*, *155*, 193-201. https://doi.org/10.1016/j.biocon.2012.05.009

Lewis, P.H.; Roberts, B.P.; Moore, P.J.; Pike, S.; Scarth, A.; Medcalf, K.; Cameron, I. (2023). Combining unmanned aerial vehicles and satellite imagery to quantify areal extent of intertidal brown canopy-forming macroalgae. *Remote Sensing in Ecology and Conservation*, *9*, 540-552. https://doi.org/https://doi.org/10.1002/rse2.327

Ling, S.D.; Johnson, C.R. (2012). Marine reserves reduce risk of climate-driven phase shift by reinstating size- and habitat-specific trophic interactions. *Ecological Applications*, *22*, 1232-1245. https://doi.org/10.1890/11-1587.1

Ling, S.D.; Keane, J.P. (2024). Climate-driven invasion and incipient warnings of kelp ecosystem collapse. *Nature Communications*, *15*, 400.

Ling, S.D.; Scheibling, R.E.; Rassweiler, A.; Johnson, C.R.; Shears, N.; Connell, S.D.; Salomon, A.K.; Norderhaug, K.M.; Pérez-Matus, A.; Hernández, J.C.; Clemente, S.; Blamey, L.K.; Hereu, B.; Ballesteros, E.; Sala, E.; Garrabou, J.; Cebrian, E.; Zabala, M.; Fujita, D.; Johnson, L.E. (2015). Global regime shift dynamics of catastrophic sea urchin overgrazing. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *370*, 10, Article 20130269. https://doi.org/10.1098/rstb.2013.0269

LINZ. (2025a). *New Zealand Transverse Mercator 2000 (NZTM2000)*. https://www.linz.govt.nz/guidance/geodetic-system/coordinate-systems-used-new-zealand/projections/new-zealand-transverse-mercator-2000-nztm2000

LINZ. (2025b). *NZ Coastlines and Islands Polygons*

Mancuso, R.T.; Gravem, S.A.; Campbell, R.S.; Hunter, N.; Raimondi, P.; Galloway, A.W.E.; Kroeker, K.J. (2025). Sunflower sea star chemical cues locally reduce kelp consumption by eliciting a flee response in red sea urchins. *Proceedings of the Royal Society B: Biological Sciences*, *292*, 20250949. https://doi.org/doi:10.1098/rspb.2025.0949

Marinovich, J.; Shears, N.; Parsons, D. (2025). Big fish, big mouths, big impacts: gape size of a fish predator determines size of urchins eaten. *Marine Ecology Progress Series*, *764*, 161-173. https://www.int-res.com/abstracts/meps/v764/meps14881

Miller, K.I.; Balemi, C.A.; Bell, D.R.; Blain, C.O.; Caiger, P.E.; Hanns, B.J.; Kulins, S.E.; Peleg, O.; Spyksma, A.J.P.; Shears, N.T. (2024). Large-scale one-off sea urchin removal promotes rapid kelp recovery in urchin barrens. *Restoration Ecology*, *32*, e14060. https://doi.org/https://doi.org/10.1111/rec.14060

Miller, K.I.; Blain, C.O.; Shears, N.T. (2022). Sea urchin removal as a tool for macroalgal restoration: a review on removing “the spiny enemies”. *Frontiers in Marine Science*, *9*. https://doi.org/10.3389/fmars.2022.831001

NIWA. (2016). *NZ Bathymetry 250m Imagery/Raster* (

Parsons, D.M.; Morrison, M.A.; MacDiarmid, A.B.; Stirling, B.; Cleaver, P.; Smith, I.W.G.; Butcher, M. (2009). Risks of shifting baselines highlighted by anecdotal accounts of New Zealand's snapper (Pagrus auratus) fishery. *New Zealand Journal of Marine and Freshwater Research*, *43*, 965-983. https://doi.org/10.1080/00288330909510054

Peleg, O.; Blain, C.O.; Shears, N.T. (2023). Long-term marine protection enhances kelp forest ecosystem stability. *Ecological Applications*, *33*, e2895. https://doi.org/https://doi.org/10.1002/eap.2895

Sala, E.; Zabala, M. (1996). Fish predation and the structure of the sea urchin Paracentrotus lividus populations in the NW Mediterranean. *Marine Ecology Progress Series*, *140*, 71-81. https://doi.org/10.3354/meps140071

Schiel, D.R. (1984). *Poor Knights Islands Marine Reserve" A Biological Survey of Subtidal Reefs*.

Schiel, D.R. (2013). The other 93%: trophic cascades, stressors and managing coastlines in non-marine protected areas. *New Zealand Journal of Marine and Freshwater Research*, *47*, 374-391. https://doi.org/10.1080/00288330.2013.810161

Shahmirian, E. (2020). Utilizing Copernicus and Google Earth Satellite Imagery to Define Habitat Suitability for Flying Squirrels (Glaucomys spp.) in the lower Rouge River basin, Toronto, Ontario.

Shears, N.; Lawerence, K. (2024). *Mapping broad-scale habitat types at the Mercury Islands, northeastern Aotearoa New Zealand, using supervised classification of satellite imagery* W. R. Council.

Shears, N.T. (2002). *Ecological response of shallow subtidal reef communities to marine reserve protection in northeastern New Zealand* University of Auckland.

Shears, N.T.; Babcock, R.C. (2002). Marine reserves demonstrate top-down control of community structure on temperate reefs. *Oecologia*, *132*, 131-142. https://doi.org/10.1007/s00442-002-0920-x

Shears, N.T.; Babcock, R.C. (2003). Continuing trophic cascade effects after 25 years of no-take marine reserve protection. *Marine Ecology Progress Series*, *246*, 1-16. https://doi.org/10.3354/meps246001

Shears, N.T.; Babcock, R.C. (2004). *Community composition and structure of shallow subtidal reefs in northeastern New Zealand* (Vol. 245). Department of Conservation.

Shears, N.T.; Babcock, R.C. (2007). *Quantitative description of mainland New Zealand's shallow subtidal reef communities* (Vol. 280). Department of Conservation.

Shears, N.T.; Babcock, R.C.; Duffy, C.A.J.; Walker, J.W. (2004). Validation of qualitative habitat descriptors commonly used to classify subtidal reef assemblages in north-eastern New Zealand. *New Zealand Journal of Marine and Freshwater Research*, *38*, 743-752. https://doi.org/10.1080/00288330.2004.9517273

Shears, N.T.; Babcock, R.C.; Salomon, A.K. (2008). Context-dependent effects of fishing: Variation in trophic cascades across environmental gradients. *Ecological Applications*, *18*, 1860-1873. https://doi.org/10.1890/07-1776.1

Smith, J.E.; Keane, J.; Oellermann, M.; Mundy, C.; Gardner, C. (2023). Lobster predation on barren-forming sea urchins is more prevalent in habitats where small urchins are common: a multi-method diet analysis. *Marine and Freshwater Research*, *74*, 1493-1505. https://doi.org/https://doi.org/10.1071/MF23140

Spyksma, A.J.P.; Taylor, R.B.; Shears, N.T. (2017). Predation cues rather than resource availability promote cryptic behaviour in a habitat-forming sea urchin. *Oecologia*, *183*, 821-829. https://doi.org/10.1007/s00442-017-3809-4

St-Pierre, A.P.; Gagnon, P. (2020). Kelp-bed dynamics across scales: Enhancing mapping capability with remote sensing and GIS. *Journal of Experimental Marine Biology and Ecology*, *522*, 13, Article 151246. https://doi.org/10.1016/j.jembe.2019.151246

Statistics New Zealand. (2023). *Regional Council 2023 (generalised)* (

Steneck, R.S. (2020). Regular sea urchins as drivers of shallow benthic marine community structure. In *Developments in Aquaculture and Fisheries Science* (Vol. 43, pp. 255-279). Elsevier. https://doi.org/10.1016/b978-0-12-819570-3.00015-9

Steneck, R.S.; Graham, M.H.; Bourque, B.J.; Corbett, D.; Erlandson, J.M.; Estes, J.A.; Tegner, M.J. (2002). Kelp forest ecosystems: biodiversity, stability, resilience and future. *Environmental Conservation*, *29*, 436-459. https://doi.org/10.1017/s0376892902000322

Sward, D.; Monk, J.; Barrett, N.S. (2022). Regional estimates of a range-extending ecosystem engineer using stereo-imagery from ROV transects collected with an efficient, spatially balanced design. *Remote Sensing in Ecology and Conservation*, *8*, 105-118. https://doi.org/https://doi.org/10.1002/rse2.230

Taylor, R.B.; Morrison, M.A.; Shears, N.T. (2011). Establishing baselines for recovery in a marine reserve (Poor Knights Islands, New Zealand) using local ecological knowledge. *Biological Conservation*, *144*, 3038-3046. https://doi.org/https://doi.org/10.1016/j.biocon.2011.09.009

Tegner, M.J.; Levin, L.A. (1983). Spiny lobsters and sea urchins: Analysis of a predator-prey interaction. *Journal of Experimental Marine Biology and Ecology*, *73*, 125-150. https://doi.org/https://doi.org/10.1016/0022-0981(83)90079-5

Teixeira, J.B.; Martins, A.S.; Pinheiro, H.T.; Secchin, N.A.; Leão de Moura, R.; Bastos, A.C. (2013). Traditional Ecological Knowledge and the mapping of benthic marine habitats. *Journal of Environmental Management*, *115*, 241-250. https://doi.org/https://doi.org/10.1016/j.jenvman.2012.11.020

Thomas, L.J.; Liggins, L.; Banks, S.C.; Beheregaray, L.B.; Liddy, M.; McCulloch, G.A.; Waters, J.M.; Carter, L.; Byrne, M.; Cumming, R.A.; Lamare, M.D. (2021). The population genetic structure of the urchin Centrostephanus rodgersii in New Zealand with links to Australia. *Marine Biology*, *168*, 138. https://doi.org/10.1007/s00227-021-03946-4

Udy, J.A.; Wing, S.R.; Jowett, T.; O'Connell-Milne, S.A.; Durante, L.M.; McMullin, R.M.; Kolodzey, S. (2019). Regional differences in kelp forest interaction chains are influenced by both diffuse and localized stressors. *Ecosphere*, *10*, e02894. https://doi.org/https://doi.org/10.1002/ecs2.2894

Ventura, D.; Grosso, L.; Pensa, D.; Casoli, E.; Mancini, G.; Valente, T.; Scardi, M.; Rakaj, A. (2023). Coastal benthic habitat mapping and monitoring by integrating aerial and water surface low-cost drones. *Frontiers in Marine Science*, *Volume 9 - 2022*. https://doi.org/10.3389/fmars.2022.1096594

Villouta, E.; Chadderton, W.L.; Pugsley, C.W.; Hay, C.H. (2001). Effects of sea urchin (*Evechinus chloroticus*) grazing in Dusky Sound, Fiordland, New Zealand. *New Zealand Journal of Marine and Freshwater Research*, *35*, 1007-1024. https://doi.org/10.1080/00288330.2001.9517060

Wing, S.R.; Shears, N.T.; Tait, L.W.; Schiel, D.R. (2022). The legacies of land clearance and trophic downgrading accumulate to affect structure and function of kelp forests. *Ecosphere*, *13*, e4303. https://doi.org/https://doi.org/10.1002/ecs2.4303

Wing, S.R.; Wing, L. (2015). Ontogenetic shifts in resource use by the sea urchin Evechinus chloroticus across an ecotone. *Marine Ecology Progress Series*, *535*, 177-184. https://doi.org/10.3354/meps11410

10 APPENDIX 1 – Locations for ground truthing

Ground truthing data sources used to help inform and validate habitat mapping exercises. Bold data types represent the primary data sources used for each location.

Region	Location	Data Type
Northland	Cape Karikari	**Georeferenced underwater photogrammetry** (underwater imagery), diver surveys, local knowledge
Northland	Purerua Peninsula and inner Bay of Islands	**Local knowledge**
Northland	Cape Brett	**Drop camera,** diver surveys
Northland	Mimiwhangata	**Local knowledge**
Northland	Poor Knights Islands	**Drop camera,** diver surveys
Northland	Tutukaka	**Drop camera,** local knowledge
Northland	Whangarei Heads	**Drop camera**
Northland	Marotere Islands	**Drop camera**, diver surveys
Northland	Bream Tail	**Drop camera**
Auckland	Mokohinau Islands	**Drop camera**, diver surveys
Auckland	Hauturu-o-Toi	**Drop camera**, diver surveys
Auckland	Aotea	**Drop camera**
Auckland	Leigh	**Drop camera**, diver surveys
Auckland	Tawharanui	**Drop camera**, diver surveys
Auckland	Kawau	**Drop camera**
Auckland	Moturoa Island	**Drop camera**
Auckland	Tiritiri Matangi Island	**Drop camera**, diver surveys
Auckland	The Noises	**Drop camera**, diver surveys
Auckland	Rakino Island	**Drop camera**
Auckland	Waiheke Island	**Drop camera**
Waikato	Coromandel Peninsula	**Drop camera,** diver surveys
Waikato	Mercury Islands	**Drop camera,** diver surveys
Waikato	Hahei	**Underwater imagery,** diver surveys
Waikato	Alderman Islands	**Drop camera,** diver surveys
Waikato	Slipper Island	**Underwater imagery,** diver surveys
Bay of Plenty	Tuhua	**Drop camera,** diver surveys
Bay of Plenty	Motiti Island	**Underwater imagery,** local knowledge

11 APPENDIX 2 – Studies included in this project

Data sources compiled within the New Zealand wide geodatabase on the distribution and extent of urchin barren habitat.

Region	Location	Date	Data Type	Source
Northland	Maitai Bay	2018	Habitat Map	Kerr et al. (2024)
Northland	Doubtless Bay	2005	Habitat Map	Kerr et al. (2024)
Northland	Waewaetoria	2010	Habitat Map	Kerr et al. (2024)
Northland	Mangonui Bay	2010	Habitat Map	Kerr et al. (2024)
Northland	Mimiwhangata	1973	Habitat Map	**Original Map:** Ballantine et al. (1973). **Digitisation:** Lawrence (2019).
Northland	Mimiwhangata	2003	Habitat Map	Kerr et al. (2024)
Northland	Mimiwhangata	2019	Habitat Map	Lawrence (2019)
Auckland	Mokohinau Islands	1978	Habitat Map	**Original Map:** Berben et al. (1988) **Digitisation:** Lawrence (2019)
Auckland	Mokohinau Islands	2018	Habitat Map	Lawrence (2019)
Auckland	Hauturu-o-Toi	1953	Habitat Map	Dartnall (2022)
Auckland	Hauturu-o-Toi	1979	Habitat Map	Dartnall (2022)
Auckland	Hauturu-o-Toi	2019	Habitat Map	Dartnall (2022)
Auckland	Cape Rodney - Okakari Pt. Marine Reserve	1977	Habitat Map	**Original Map:** Ayling et al. (1981). **Digitisation:** Leleu et al. (2012).
Auckland	Cape Rodney - Okakari Pt. Marine Reserve	2006	Habitat Map	Leleu et al. (2012).
Auckland	Cape Rodney - Okakari Pt. Marine Reserve	2019	Habitat Map	Lawrence (2019).
Auckland	Tawharanui Marine Reserve	2006	Habitat Map	Kerr et al. (2024).
Auckland	Long Bay Marine Reserve	2020	Habitat Map	Kulins (2021).
Auckland	The Noises	1978	Habitat Map	Dartnall (2022).
Auckland	The Noises	2019	Habitat Map	Dartnall (2022).
Waikato	Great Mercury Island	2020	Habitat Map	Shears & Lawrence (2024).
Waikato	Aitu/Green/Korapuki Islands	2016	Habitat Map	Shears & Lawrence (2024).

Region	Location	Date	Data Type	Source
Waikato	Moturehu/Whakau Islands	2016	Habitat Map	Shears & Lawrence (2024).
Waikato	Te Whanganui-a-Hei Marine Reserve	2013	Habitat Map	Haggitt (2017)
Waikato	Te Whanganui-a-Hei Marine Reserve	2013	Habitat Map	Kibele & Shears (2017).
Marlborough	Long Island Marine Reserve	2014	Habitat Map	Haggitt (2016)
Marlborough	Queen Charlotte Sound	2018	Drop Camera	Lafont & Shears (2025).
Stewart Island	Ulva Island Marine Reserve	2015	Habitat Map	Unpublished data - commissioned for Department of Conservation
New Zealand wide	New Zealand wide	1999-2000	Diver Transects	**Unpublished data** - commissioned for Department of Conservation as part of: Shears & Babcock (2007).