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Estimating the extent of urchin barrens and kelp forest loss in northeastern Aotearoa, New Zealand

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ABSTRACT

On shallow rocky reefs in northeastern Aotearoa, New Zealand, urchin barrens are indicators of the ecosystem effects of overfishing reef predators. Yet, information on their extent and variability is lacking. We use aerial imagery to map the urchin barrens and kelp forests on reefs (<30 m depth) across seven locations, including two long-established marine reserves and a marine protected area that allows recreational fishing. Urchin barrens were present in all locations and were restricted to reefs <10–16 m deep. Urchin barrens covered 30% (7–49%) of shallow reefs in fished areas, with variation among and within locations likely related to the relative extent and topography of reefs and wave exposure. Within the marine reserves, barrens covered <2% of shallow reefs. Long-term comparison of historical imagery at two fished locations demonstrates that current areas of urchin barrens were historically dominated by kelp forests, but barrens have persisted since at least the early 2000s. We estimate ~30 km² of barrens exist across the region. This demonstrates the widespread nature of barrens, the potential long-term effectiveness of no-take marine protection in restoring urchin barrens, and the need for a multifaceted management approach to restore kelp forests and prevent further expansion of barrens.

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Introduction

Kelp forests have long been recognised for the numerous ecosystem services they provide, including supporting biodiversity and primary production (Mann 1973; Teagle et al. 2017; Pessarrodona et al. 2022; Eger et al. 2023b). More recently, their role in carbon storage and sequestration has increased interest in kelp forest restoration, conservation and management (Ortega et al. 2019; Filbee-Dexter and Wernberg 2020; Bayley et al. 2021). Globally, kelp forests are impacted by a multitude of stressors and the causes of kelp forest loss vary regionally (Krumhansl et al. 2016). Overgrazing by

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sea urchins has had catastrophic impacts on temperate reef ecosystems and is a ubiquitous driver of kelp loss across many regions (Ling et al. 2015; Steneck 2020). In many cases, the proliferation of sea urchins is linked to the loss of predators through fishing, which enables the destructive grazing of kelp forests and the formation of denuded urchin-dominated reefs, commonly known as ‘urchin barrens’ (Filbee-Dexter and Scheibling 2014; Ling et al. 2015). In regions where there is a link between kelp loss, sea urchins and harvest of urchin predators, rebuilding predator populations through marine protection can provide a viable ecosystem-based approach to restoring resilient kelp forest ecosystems and their associated ecosystem services (Babcock et al. 2010; Peleg et al. 2023). However, an important step in developing strategies to manage these ecosystem effects of fishing is understanding the spatial extent and drivers of variation in the distribution of urchin barrens and the effectiveness of existing management strategies such as different types of MPAs in reversing these impacts.

In Aotearoa New Zealand, kelp forest habitats are extensive across shallow subtidal reefs (Choat and Schiel 1982, Shears and Babcock 2007), but are vulnerable to a number of anthropogenic and climatic stressors, including warming temperatures (Cornwall et al. 2023), increased coastal darkening and sedimentation (Blain et al. 2021), and fishing-induced trophic cascades (Shears and Babcock 2002). Where environmental conditions are not limiting, sea urchin grazing is often the primary cause of kelp forest loss, particularly in northeastern New Zealand (Wing et al. 2022). Here, fishing of important sea urchin predators, in particular tāmure (snapper *Chrysophrys auratus*) and kōura (spiny lobster *Jasus edwardsii*) has facilitated the proliferation of the sea urchin *Evechinus chloroticus* (kina) and the formation of extensive urchin barrens (Shears and Babcock 2002). Long-term protection of predators in marine reserves has been shown to result in declines in urchins and a recovery of kelp forests in areas that were previously urchin barrens (Babcock et al. 2010; Leleu et al. 2012; Peleg et al. 2023). This demonstration of the ecosystem-level effects of fishing and restoration potential of fully protected MPA’s have not been incorporated into or addressed by fisheries management. A recent New Zealand High Court ruling confirmed evidence that the widespread loss of kelp forests in the northernmost region of New Zealand is linked to fishing and that government officials have acted illegally in setting fishing quota for spiny lobster by not considering the ecological and cultural impacts of fishing on kelp forest ecosystems as required under the Fisheries Act 1996 (MPI 2022). Following this decision, there is now a directive from the Court that the fisheries management process must address these issues and set fisheries targets that ensure species fulfil their ecological role. Consequently, gaining a broader understanding of the extent of urchin barrens across the region, and how their extent varies among locations and within different types of MPAs, is necessary to start developing and guide a more ecosystems-based approach to managing fisheries of important sea urchin predators.

Urchin barrens in northern New Zealand are predominantly restricted to a shallow depth band, typically from 2–10 m on open rocky coasts (Choat and Schiel 1982; Grace 1983; Shears and Babcock 2004). These are bounded by mixed algal forests in shallow water, dominated by wave energy tolerant species (e.g. *Carpophyllum maschalocarpum*, *Carpophyllum augustifolium* and *Lessonia variegata*), and largely monospecific forests of the stipitate kelp *Ecklonia radiata* at depths below the barrens (Figure 1). The lower depth extent of the kelp forest is often truncated where the reef terminates in sand but can extend to depths beyond 30 m

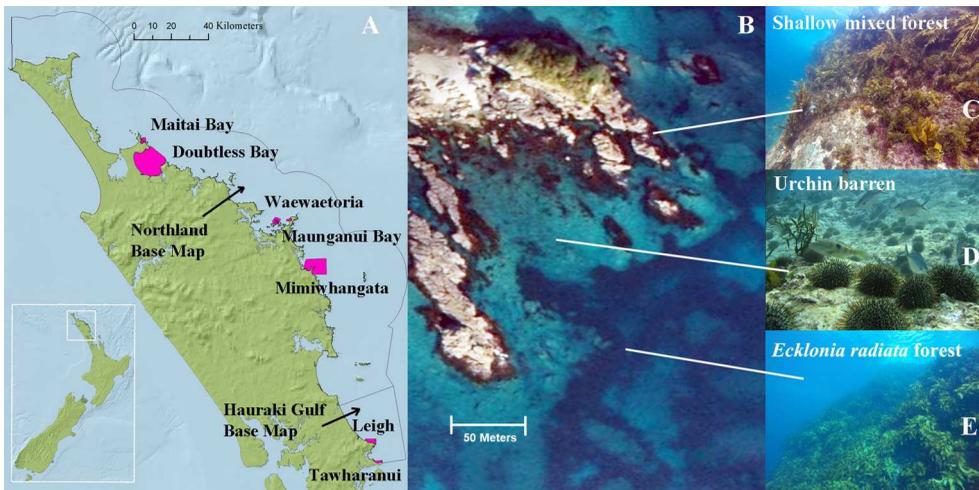


Figure 1. Study locations on the northeastern coast of the North Island, Aotearoa New Zealand (A). The seven mapping locations are shown in pink and labelled. The thin grey outline is the extent of the two regional scale base maps used to estimate the extent of subtidal reef <30 m depth. An aerial photo taken at Maitai Bay showing typical zonation of dominant shallow subtidal reef habitats in the region (B), with shallow mixed forests from 0–2 m depth (C), urchin barrens from 2 to ~10 m (D) and kelp forest from ~10 to 15 m in this location (E), with patches of sand below.

depending on water clarity (Blain et al. 2021). While the lower limit of barrens typically occurs at ~10 m depth, barrens do extend deeper with increasing wave exposure and may occur to depths of ~20 m in highly exposed locations (Grace 1983; Shears and Babcock 2004). While *E. chloroticus* is the dominant barren-forming sea urchins species in New Zealand (Shears and Babcock 2007), and barrens associated with this species are known as ‘kina barrens’, the subtropical urchin *Centrostephanus rodgersii* and barrens associated with this species has recently been reported as increasing in parts of northern New Zealand (Bailemi and Shears 2023). This species can co-occur with *E. chloroticus* in urchin barrens but can also dominate barren areas in deeper water. The generally shallow depths of urchin barrens combined with relatively clear water along the open northern coast of New Zealand make them clearly distinguishable in aerial imagery (Figure 1). Though mapping submerged marine habitats with aerial and satellite imagery poses a unique set of challenges (St-Pierre and Gagnon 2020), the clear water and visual distinctiveness of the main habitat types in northern New Zealand have meant a number of studies have successfully mapped shallow reef habitats using a range of aerial and satellite-based imagery approaches (Kerr and Grace 2005; Leleu et al. 2012; Kibele 2017; Lawrence 2019; Dartnall 2022).

A critical step in addressing the challenge of restoring and protecting kelp forests is understanding the extent and drivers of variation in the spatial extent of urchin barrens and how existing management measures, such as MPAs, can mitigate these impacts alongside a shift to ecosystem-based management of fisheries and achieving the legislative goals of sustainability and a functioning ecosystem. In the case of fisheries management, stock modelling and management decisions are primarily undertaken at regional scales. In this study, we map subtidal reef habitats to depths of 30 m at seven locations in northeastern New Zealand (Figure 1) to better understand (1) the distribution and extent of urchin barrens across the region, (2) how urchin barrens vary

between no-take marine reserves, partially protected MPAs and areas open to fishing, and (3) how the current distribution of urchin barrens has changed through time (e.g. compared to that in historical aerial imagery) from two key locations. Our assessment includes two previously published habitat maps (Leleu et al. 2012; Kerr and Grace 2005) and five additional maps that have been produced for various management, conservation and research purposes. All maps have been produced using a consistent mapping methodology based on visual classification of shallow reef habitats from aerial imagery and supported by ground-truth data. Using the maps produced, combined with regional maps of the extent of a subtidal reef (Kerr 2010; DOC unpublished GIS resources), we also estimate the overall area of urchin barrens and kelp forest loss across the study area from northwestern tip of the Northland Peninsula to the Tāwharanui Peninsula in the Hauraki Gulf (Figure 1). Collectively, this information provides a wider assessment of the spatial extent of urchin barrens and kelp forest loss due to sea urchin grazing at a regional scale and provides an assessment of the potential management role of different types of MPAs in reversing this trend.

Methods

Study locations

Between 2004 and 2019, shallow subtidal habitats were mapped at seven locations along the north-eastern New Zealand coast at scales of 1:200–1:500 (Figure 1, Table 1).

Table 1. Management status and mapping information for locations mapped in northeastern New Zealand. The maximum depth of barrens in each location was used to delineate ‘shallow’ and ‘deep’ reef areas. Aerial imagery sources: Kerr and Grace collection contact the author, Oceans 2020, National Institute of Water and Atmospheric Research 2009. Oceans 20/20 Bay of Islands aerial photo colour mosaic. NIWA GIS data. <https://www.linz.govt.nz/hydro/projects-programmes/ocean-survey-2020/about-ocean-survey-2020> *A Traditional Management Rahui (full protection) has been established since mapping was completed.

Location	Management status at time of mapping	Ground truthing surveys	Reef area mapped	Max. Depth of barrens (m)	Primary source and date of Aerial Imagery	Map reference
Maitai Bay	Fished*	2017–19	218 ha	16	Grace and Kerr 2003, 2009, Google Earth 2018	Supplementary material figure S1
Doubtless Bay	Fished	2005	2,766 ha	12	Grace and Kerr 2005	Supplementary material figure S2
Waewaetorea, Bay of Islands	Fished	2011–15	182 ha	12	BOI Ocean’s 20/20, 2010	Supplementary material figure S3
Maunganui Bay, Bay of Islands	Fished*	2012–16	37 ha	12	BOI Ocean’s 20/20, 2010	Supplementary material figure S4
Mimiwhangata	MPA allows recreational fishing	2003–4	1,141 ha	12	Grace and Kerr 2003	(Kerr and Grace 2005) & (SM figure S5)
Leigh	No-take MPA and adjacent fished area	2005–6	133 ha	10	Grace and Kerr, 2006	(Leleu et al. 2012) & (SM figure S6)
Tawharanui	No-take MPA and adjacent fished area	2005–6	78 ha	10	Grace and Kerr, 2006	Supplementary material figure S7

The locations are geographically spread along the region's mainland coast and are representative of rocky coastlines with moderate to high wave exposure. At Leigh and Tāwharanui, we mapped reef areas inside and outside two full no-take marine reserves, the Cape Rodney to Okakari Point (or Leigh) Marine Reserve, established in 1975 but not opened until 1977, and the Tāwharanui Marine Reserve, established in 1981 but not opened until 1982. The reserve at Tāwharanui was initially established under The Fisheries Act but changed to a marine reserve under the Marine Reserves Act in 2011. In both marine reserves, snapper *Chrysophrys auratus* and spiny lobster *Jasus edwardsii* are larger and more abundant than in adjacent fished areas (Willis et al. 2003; LaScala-Gruenewald et al. 2021; Hanns et al. 2022). At Mimiwhangata, almost all of the mapped shallow reef area lies within the partially protected Mimiwhangata Marine Park established under the Fisheries Act 1984. Recreational fishing is allowed within the marine park, but all commercial fishing ceased in 1994. Unlike the two marine reserves, predatory species have not recovered in this marine park and both species occur at similar sizes and densities to adjacent fished areas (Denny and Babcock 2004; Shears et al. 2006). All of the other locations were considered as fished, but it is important to note that within two of the mapped locations (Maitai Bay and Maunganui Bay) there are now areas with protection under customary management measures (Rāhui). These protected areas were established around the time aerial imagery and ground-truth data were collected so are treated as 'Fished' for the purpose of this study (Table 1). Given the varying management status of the locations, we have structured the analysis and calculations to allow for the comparisons between no-take marine reserves, the partially protected MPA and areas open to fishing.

Mapping of reef habitat types

The methodology used in the seven mapping projects followed a consistent data collection process, field methods and analysis described in the previously published Mimiwhangata and Leigh Marine Reserve mapping studies (Kerr and Grace 2005; Leleu et al. 2012). All seven of these mapping projects were coordinated and conducted by VK and RG. In all cases, the mapping process involved acquiring high-quality aerial imagery and undertaking fieldwork to collect ground-truth information as outlined below. The data were then used to inform a trained visual assessment and manual digitisation of habitat polygons. Additional fieldwork was carried out to fill in gaps in habitat depth zonation profiles and boundaries if necessary. The previously published habitat map produced for the Leigh Marine Reserve, which uses these methods (Leleu et al. 2012) had an accuracy of 87%, which is consistent with other studies that have mapped kelp and urchin barrens from aerial imagery using trained visual interpretation methods (St-Pierre and Gagnon 2020).

The offshore extent and maximum depth limit of the habitats mapped varied among regions and were largely determined by the distribution of reef habitats (See supplementary material). However, for the purposes of this study, the map extent was constrained to a maximum depth limit of 30 m, which generally coincides with the lower limit of the dominant kelp *Ecklonia radiata* in all locations. In all cases, the reef edge was delineated using aerial imagery if visible or combined with ground-truth information when visibility

was limited. The 30 m depth profile was drawn using field bathymetry data, multibeam data where available and nautical charts (as outlined below).

The following reef habitat types were mapped in each location (if present) and are largely consistent with previous mapping studies in the region:

Shallow mixed forest – This combines the Shallow *Carpophyllum* and Mixed algal forests used in Leleu et al. (2012). Reefs are either fucoid-dominated forests (>50% canopy cover) with high abundances (>20 adult plants m⁻²) of *Carpophyllum maschalocarpum*, *C. plumosum*, *C. angustifolium*, or forests are comprised of a mix of fucoids and the kelps *Ecklonia radiata* and *Lessonia variegata*. Sea urchins (*Evechinus chloroticus*) generally occur in low numbers and occupy crevices (<2 exposed urchins m⁻²). This habitat generally occurred from 0 to 2 m depth but did extend to 8 m at the most wave-exposed locations.

Urchin barrens – Low numbers of large brown algae present, substratum typically dominated by crustose coralline algae and urchins present. Usually associated with grazing activity of *E. chloroticus* (>2 exposed urchins m⁻²), which leaves the substratum relatively devoid of macroalgae. *E. chloroticus* was the dominant urchin species within barrens at all locations and *C. rodgersii* was largely found at lower densities when mapping was carried out for this study. *C. flexuosum* and *Sargassum sinclairii* may occur. The shallow boundary was typically the bottom edge of the shallow mixed forest zone (~2 m) and extended to a depth range varying between 10–16 m depending on location (Table 1).

Kelp forest – Almost entirely monospecific stands of mature *Ecklonia radiata*. Canopy cover can vary from sparse *Ecklonia* forest (<50% cover) to dense *Ecklonia* forest. Urchins at low numbers and when present, usually occupy crevices. The underlying substratum is often covered with coralline algae, turf and foliose macroalgae and in places a diverse encrusting invertebrate community. Kelp forest extended from the lower limit of the urchin barrens to the edge of the reef, which was typically between 11 and 17 m. At locations where the reef extended deeper *Ecklonia* generally thinned out below 20–30 m. As outlined below the mapped area of kelp forest was stratified as shallow and deep kelp forest (Section 2.3).

***Carpophyllum flexuosum* forest** – Tall forests of *C. flexuosum* (1–2 m in height) can dominate (>4 adult plants m⁻²) on sheltered reefs, often associated with high levels of sediment. On more exposed reefs plants are short and generally associated with *Evechinus*.

Algal turfs – Reef predominantly covered by turfing algae (e.g. articulated corallines and other red turfing algae <5 cm in height), with a low number of brown algae. Urchins are low in abundance and cryptic or absent.

Sponge garden – This combines the Sponge Garden and Deep Reef habitats used in Leleu et al. (2012) at Leigh. The reef is typically dominated by sponges and has a layer of fine sediment overlying the rock. *Ecklonia radiata* may be sparsely present in the less deep areas. This habitat usually occurs in deep water (>30 m) and often at the reef/sand interface but was recorded on reefs <30 m depth at Leigh.

A relatively large area (37 ha) of shallow patch reefs interspersed with sand was also mapped in Doubtless Bay (Figure S2). This patch reef area consisted mostly of urchin barrens, but there were also patches of kelp and mixed algae. Due to this high spatial complexity, individual habitats could not be mapped effectively at the mapping scale

used. This area was not included in calculations of the total reef area and the area of dominant reef habitats.

Aerial imagery acquisition

Acquiring aerial imagery of sufficient quality during suitable sea conditions poses the most significant challenge to mapping subtidal habitats using aerial image-based methods. Conditions of minimal wind and swell, high water clarity, no cloud cover, high sun angle (time of day) and preferably low tide are needed to accurately distinguish and map reef habitats to sufficient depths over the entire area of interest. For this reason, we utilised local light plane pilots in each region who were on call for the rare ‘perfect conditions day’. The images were taken by the authors (VK and RG) with a handheld high-resolution professional-level 35 mm camera through a downward-looking portal in the rear of the plane. A pre-planned flight path and calculated time interval between photos derived from the flight altitude and camera lens specifications were used to result in full image coverage of the study area. Four photography flights were taken between 2003 and 2009, encompassing the different study locations (Table 1). Additional aerial imagery data for Waewaetoria Island and Maunganui Bay, Bay of Islands, were obtained in 2010 from Oceans 20/20 (<https://www.linz.govt.nz/hydro/projects-programmes/ocean-survey-2020/about-ocean-survey-2020>).

Ground-truth data collection

Ground-truth surveys were carried out for all the mapping projects within a two-year period, with one exception being the Waewaetoria Island study where this work extended to four years. Mapping was completed in the final year of the ground truthing or the following year in all of the studies (Table 1). Broad-scale surveys were conducted to identify the major habitat boundaries between soft sediments and reef areas. Secondary surveys were then used to identify within reef habitat boundaries, with an emphasis on reef areas <30 m depth and dominant reef habitat types. At Mimiwhan-gata, due to the extensive deep reefs offshore, a tethered side-scan unit and an ROV were also used. At all sites other than Leigh, sediment sampling was carried out to assist in interpreting the reef/soft sediment boundaries with a single beam and side-scan sonar. At Waewaetoria Island and Maunganui Bay sites, an additional side-scan sonar layer was available from the Oceans 2020 survey project (<https://www.linz.govt.nz/hydro/projects-programmes/ocean-survey-2020/about-ocean-survey-2020>), which facilitated the delineation of areas of reef and sand. Habitat and algal community descriptive data from fish and rock lobster monitoring was also available and assembled to aid the mapping at all locations except for Doubtless Bay. Lastly, a manta tow video method was used to add to the reef community information (Kerr and Grace 2005; Leleu et al. 2012). The manta tow method was used at all sites except Maunganui Bay and Tawharanui.

All field data had GPS coordinates, notes and depth recorded from our survey boat depth sounder. Bathymetry data from the field was tide corrected and checked against our base bathymetry; the NZ Marine Charts series (<https://www.linz.govt.nz/products-services/charts>). A full set of habitat maps used in this study is provided in the

supplementary material (Fig S1–S8). GIS resources for the study maps are also available online (Kerr 2024).

Aerial imagery mapping technique

Aerial imagery was georeferenced within the GIS project. Ground truth data, presented in line or point form, was processed into a GIS shapefile, and a symbology was applied to facilitate visualisation as an overlay on top of the aerial imagery. For side-scan imagery, a raster layer was generated for the GIS project. The next step involved manually drawing polygons to delineate the different reef habitat types over the aerial imagery to create the map layer. Depending on the site and the quality of the aerial imagery, identifying reef and kelp forest edges was sometimes straightforward, allowing for confidence in the polygon drawing without heavy reliance on field data layers. Ground truthing data assisted the mapping process in cases where the reefs extended beyond the visible depth of the aerial imagery. Side-scan imagery, single beam sonar and drop-down video were employed to verify boundaries between reef and soft sediments. The mapping layer was overlaid onto the side-scan layer, and a ‘swipe’ tool was used to assess the reef boundaries mapping accuracy between the two layers. Upon completing the initial drawing of the reef polygons, mapping checks were conducted against all available ground-truthing layers. The entire process of checking through the data layers over time contributed to ‘training’ and building the experience of the person (VK) doing the mapping with the visual method.

Ideally, for each mapping project, high-quality imagery suitable for mapping at the scale of urchin barrens would be sourced in the same year of ground truthing and the mapping exercise. In two of the study areas, this was not achieved. At Maitai Bay, the best images available were obtained ten and twelve years before the ground truthing and mapping year. However, we sourced a usable satellite image in the mapping year to check for changes since the older imagery. We also had previous diving experience and local knowledge of the hāpu to inform our mapping process. For the two areas in the Bay of Islands, Waewaetoria Island and Maunganui Bay, the best imagery was obtained for four and five years, respectively, prior to the completion of ground-truthing and mapping. All the other mapped areas had aerial imagery dated within two years of ground truthing and mapping (Table 1).

Habitat area analysis

For each location, the total area of each reef habitat type mapped was calculated, allowing assessment of the overall area of a subtidal reef (to a maximum depth of 30 m) and the percentage cover of the different reef habitat types. The overall area of the reef mapped was dependent on the maximum depth and extent of the reef, e.g. in some locations (Leigh and Tāwharanui), the maximum extent of the reef edge was predominantly less than 20 m depth and only a small area of the reef at Leigh extended beyond 20 m. In other locations (Mimiwhangata and Doubtless Bay) the reefs extended beyond the 30 m depth limit used in this study.

To investigate the overall prevalence of urchin barrens across locations and with management regime, the extent of urchin barrens was expressed as a percentage of the overall

area of reef mapped to 30 m depth, but also as a percentage of the ‘shallow reef area’ which was defined separately for each location. Urchin barrens are primarily restricted to shallow reefs (<~15 m) in northeastern New Zealand (Grace 1983; Shears and Babcock 2004), meaning that the relative coverage of urchin barrens will vary depending on the relative extent of shallow and deep reefs present. For example, locations with steep-sloping shallow reefs and extensive deep reefs would inherently have a low overall coverage of urchin barrens, even if barrens completely dominate shallow reefs. Therefore, to provide a more ecologically meaningful and comparable measure of the extent and prevalence of urchin barrens in each location, the area of barrens was also calculated as a percentage of the ‘shallow reef area’ mapped in each location. The depth extent of the shallow reef area was set as the maximum depth limit at which urchin barrens were observed in each location (Table 1). This depth limit was calculated separately for each location as the lower limit of the urchin barrens generally increases with increasing wave exposure and water clarity (Grace 1983; Shears and Babcock 2004). Across the mapped locations, the lower limit of barrens was shallowest in the southernmost locations (10 m at Leigh and Tāwharanui) and deepest in the northernmost (16 m at Maitai; Table 1). Within each mapped location, wave exposure variation exists, so using the maximum depth of barrens provides a conservative estimate of their overall coverage on shallow reefs.

A contour line was added to each map to split the reef between a ‘shallow’ (encompassing all urchin barrens) and a ‘deep’ zone. We used the NZ Series Marine Charts lines and point data, our depth-corrected survey data, and a layer we created from the *Navionics* Marine Chart (navionics.com). Multibeam bathymetry data were also available at Leigh. The depth split contour was then used to split habitat polygons into shallow and deep reef habitats.

Regional assessment of the total area of urchin barrens in Northland

Regional scale maps of subtidal reefs spanning northeastern New Zealand, including Northland to the Hauraki Gulf (Kerr 2010) (DOC GIS resource) were used to estimate the total reef area <30 m depth. Estuaries were excluded from the study. The area covered extends from Cape Reinga on the northern coast of New Zealand down the east coast to the Tāwharanui Peninsula in the Hauraki Gulf (Figure 1). The relative cover of urchin barrens and other reef habitats in the seven fished locations was then used to estimate the total area of each reef habitat type at the regional scale.

Long-term variation in urchin barren extent at key locations

Historic and more recent aerial images were compiled for two sites at Mimiwhangata and Maitai Bay to provide insights into long-term changes in urchin barren extent and the persistence of urchin barrens since the mapping was carried out in the current study. These sites were determined based on the availability of suitable imagery to map subtidal habitats that could be compared over three time periods: (1) the earliest available aerial imagery (1940s–1950s), (2) the year of the imagery used in the mapping conducted in the present study (2000s), and (3) the most recent imagery available from each site (2020s). Historic images were sourced from Retrolens Historical Image Resource,

<https://retrolens.co.nz/>. The most recent images of Mimiwhangata in 2020 and Maitai Bay in 2023 were sourced from Digital Globe and Google Earth respectively. At each site, we identified and created a common spatial area in the GIS project that had suitable and overlapping imagery to allow us to evaluate and map the area of different reef habitats in each of these periods. At Maitai Bay a time series of aerial imagery was created for a 7 ha area at Waikura, with images from 1944, 2003 and 2023, and at Mimiwhangata for a 47 ha area between Black Rocks and Lunch Bay, with images from 1950, 2009 and 2020. Historical imagery is black and white, and ground-truth data was lacking, so it was not possible to distinguish between different types of macroalgal forests. Reef habitats across all images were therefore classified as Urchin barrens or Macroalgal forests, i.e. combining the Shallow mixed algal forest, Kelp forest and *Carpophyllum flexuosum* forest habitat types. At both study sites the mapped reefs extend to a maximum depth of ~15 m before giving way to sand.

Results

Distribution and extent of subtidal reef among locations

The two largest areas mapped at Doubtless Bay and Mimiwhangata had extensive offshore reef systems (>2 km offshore) that extended beyond the 30 m depth limit of this study (Figure 2A and D). The majority of mapped reefs in these locations was classified as deep, i.e. below the maximum depth that urchin barrens occur (76.5 and 56.5% respectively; Table 2). The other mapped locations had a higher relative cover of shallow reefs (51–92%). Maitai Bay (Figure 2B) and Waewaetoria (Figure 2C) have extensive areas of shallow reef in the bay and sheltered side of the islands respectively, but reefs along the more exposed northeastern coast extend into deeper water (>30 m). Maunganui Bay (Figure 2D) is a more sheltered cove with a narrower margin of subtidal reef that drops relatively steeply to sand at depths between 8 and 20 m. Some deep areas of reefs occur in the outer part of the bay and some of these extend >30 m depth. The two southern most locations, Leigh (Figure 2F) and Tāwharanui (Figure 2G) also had shallower and less extensive reefs that were truncated in sand at depths <25 m.

Spatial variation in major reef habitat types among locations

Kelp forest was the dominant habitat across all locations and dominated all mapped deep reefs (Table 2, Figure 2). Only at Leigh were 'Sponge garden' habitats present at depths <30 m, occurring below the Deep kelp forest. The shallow reef area in all locations was dominated by shallow mixed algal forests, urchin barrens and shallow kelp forest (Table 2, Figure 2). Areas of *C. flexuosum* forest were also present on shallow reef in Doubtless Bay and Mimiwhangata (Figure S2 and S4), and algal turf habitat at Leigh and Tāwharanui (Figure S6 and S7).

The cover of urchin barrens on shallow reefs ranged from 7–49% at the fished locations (Table 2). The highest relative cover of urchin barrens occurred on shallow reefs at Mimiwhangata Marine Park (49%), which allows recreational fishing. Among the fished locations, there was a large variation in the extent of barrens as a proportion of the shallow reef area and overall reef extent. Mimiwhangata, Maitai Bay and the

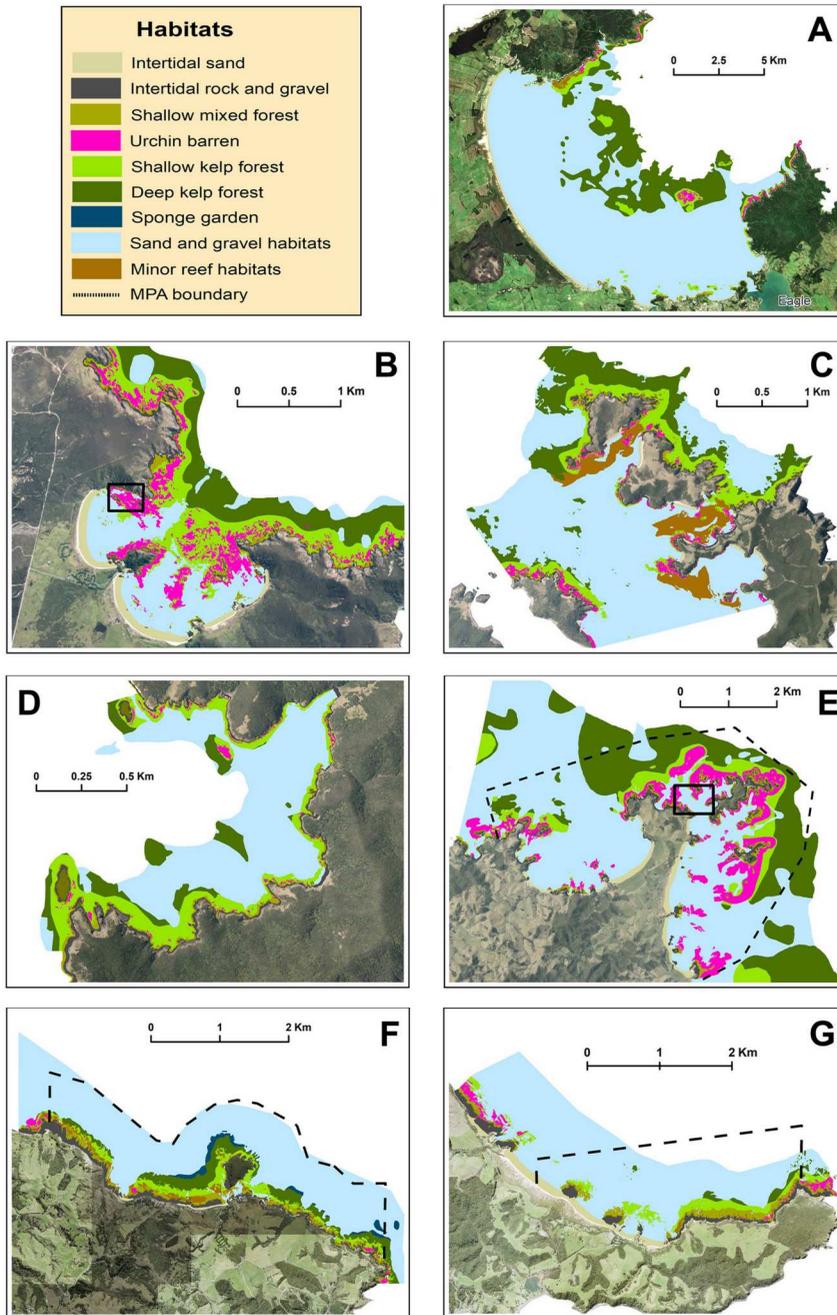


Figure 2. Simplified habitat maps showing dominant reef habitat types (<30 m depth) at each of the study locations: Doubtless Bay (A), Maitai Bay (B), Waewaetoria, Bay of Islands (C), Maunganui Bay, Bay of Islands (D), Mimiwhangata (E), Leigh (F) and Tāwharanui (G). Algal turfs and *C. flexuosum* forest habitats were relatively rare so were combined as ‘Minor reef habitats’ for these simplified maps (see Supplementary material for higher resolution detailed maps of each location). Dashed lines indicate marine protected area (MPA) boundaries. The MPA at Mimiwhangata is partially protected, whereas at Leigh and Tāwharanui the MPA’s are no-take marine reserves. The locations of historical imagery comparisons (Figure 3) are outlined by black squares in (B) and (E).



Table 2. Area of reef habitat types mapped across the seven study locations (A) and regional estimates of the extent of reef habitats (<30 m depth) on the mainland east coast of Northland, New Zealand (B). Shallow reef is defined as the area of reef within the depth limit that urchin barrens occur in each location (Table 1). The cover of urchin barrens is given for each location in (A) relative to the overall (Total) extent of reef (<30 m) and relative to the extent of shallow reef encompassing the maximum depth barrens occur. The regional extent of each habitat is estimated in two ways, based on the overall cover in mapped locations over the total reef area and when constrained to the shallow reef area. *Mimiwhangata is a marine park but allows recreational fishing.

A – Mapped Locations	Reef habitat types (ha)						Reef area (ha)		Urchin barren coverage (%)		
	Shallow mixed forest	Urchin barrens	Algal turfs	<i>C. flexuosum</i> forest	Shallow kelp forest	Deep Kelpforest	Sponge Garden	Shallow	Total	Shallow	Total
Fished locations											
Maitai Bay	15.2	49.4	0.0	0.0	74.0	79.7	0.0	138.7	218.3	35.7	22.6
Doubtless Bay	138.0	107.4	0.0	6.0	390.5	2,086.6	0.0	641.9	2,728.5	16.7	3.9
Waewaetorea	15.3	17.2	0.0	0.0	60.6	89.3	0.0	93.1	182.4	18.5	9.5
Maunganui Bay	4.2	1.7	0.0	0.0	20.2	11.2	0.0	26.1	37.3	6.5	4.5
Mimiwhangata*	71.2	242.6	0.0	6.0	177.1	644.7	0.0	496.9	1,141.6	48.8	21.2
Leigh	2.7	3.3	0.9	0.0	1.7	4.9	13.4	8.6	27.0	38.5	24.5
Tawharanui	8.6	10.2	0.0	0.0	7.6	2.2	0.0	26.4	28.6	38.4	35.5
Total area/Percentage	255.3	431.8	0.9	12.0	731.6	2,918.6	13.4	1,431.7	4,363.7	30.2	9.9
Reserve Locations											
Leigh	27.5	1.2	16.2	0.0	37.2	52.4	1.2	82.0	135.6	1.4	0.9
Tawharanui	23.9	1.0	1.0	0.0	26.0	7.9	0	51.9	59.7	1.9	1.7
Total area/Percentage	51.4	2.2	17.1	0.0	63.2	60.2	1.2	133.9	195.3	1.6	1.1
B – Regional Estimates											
Percentage of shallow reef	17.8%	30.2%	0.1%	0.8%	51.1%	–	–	Estimated reef area			
Percentage of total reef	5.9%	9.9%	0.0%	0.3%	16.8%	66.9%	0.3%	Total			
Reef habitat area (<30 m)	1,805	3,053	7	85	5,172	20,633	95	Shallow	10,122	30,849	

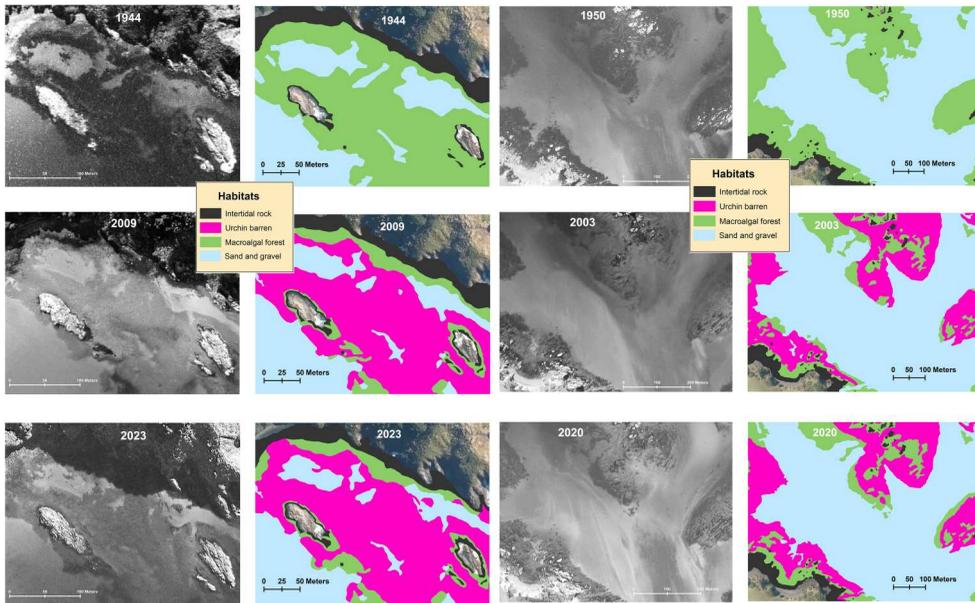


Figure 3. Historical comparison of the extent of macroalgal forest and urchin barrens on shallow reef at a study site at Maitai Bay (A) and Mimiwhangata (B). In both study sites the mapped reefs extend to a maximum depth of ~ 15 m. Note: the boundaries between the different macroalgal dominated habitats could not be distinguished in the historical imagery, so are combined into ‘Macroalgal forest’ for this comparison. Original images are converted to black and white for consistency (left) and comparison with associated habitat map (right).

unprotected coastline surrounding the Leigh and Tāwharanui MPAs were the fished locations with the highest urchin barren coverage. In contrast, Maunganui Bay had the lowest coverage of urchin barren on the shallow reef (7%) and the reefs are relatively steep with limited offshore/horizontal extent. Doubtless Bay and Waewaetoria Island had an intermediate level of urchin barren coverage of 17–19% of the shallow reef. At Doubtless Bay urchin barrens primarily occurred on shallow reefs along the mainland and only a single area (37ha) of urchin barrens was present on the shallow area of reef in the bay at Fairway Reef (Figure 2A). At Waewaetoria Island, there were limited barrens on the exposed northeastern side of the islands, and barrens were prevalent on shallow reefs in the more sheltered areas (Figure 2C).

At Leigh and Tāwharanui, there was a marked difference in the distribution of shallow reef habitats inside and outside of the marine reserves (Figure 2F and G). On the shallow reef in the reserves, urchin barrens covered 1–2% compared to 38.5% outside the reserves (Table 2). The main areas of urchin barrens within both reserves occurred near the reserve boundaries. Shallow reefs inside both reserves were dominated by shallow mixed algal forests and shallow kelp forests, and the relative cover of these habitats was substantially higher inside compared to outside both reserves.

Regional estimate of the area of urchin barrens

A total of 4364 ha of rocky reef was mapped across the 7 locations (Table 2, excluding the two marine reserves), constituting 14% of the estimated area of a subtidal reef

(<30 m depth) in the northeastern New Zealand study region (Figure 1). Using the relative proportion of shallow reefs in the mapped locations (33%), we estimate there are 10,122 ha of shallow reefs in the region within the maximum depth range where urchin barrens occur. By applying the overall percentage of barrens observed in our fished study locations (9.9%) to the estimated total area of reef <30 m in the region (30,849 ha), we estimate there are 3,053 ha of urchin barrens in the regional study area (Table 2). If we take the average cover of barrens on shallow reefs among the seven fished locations ($29.0 \pm 0.11\%$ [95% CI]) and apply this to the estimated area of shallow reef across the region (10,122 ha), this comes up with a similar overall estimate of $2,936 \pm 1,139$ ha (95%CI) of urchin barrens.

Long-term changes in urchin barrens

The time series of aerial imagery compiled for the two sites at Mimiwhangata and Maitai Bay revealed a consistent temporal pattern in the presence and establishment of urchin barrens on shallow reefs (Figure 3A and B). In both locations, there were no urchin barrens evident on the reef in the 1944 and 1950 images and the reefs were dominated by macroalgal forest. In contrast, images and associated habitat maps at Mimiwhangata (2003) and Maitai Bay (2009) showed extensive urchin barrens covering 75 and 79% of the reef area, respectively. These results were similar in the most recent images available, with the Mimiwhangata (2020) site having 61% urchin barrens and the Maitai Bay (2023) site having 81% urchin barren coverage.

Discussion

This study provides the first region-wide assessment of the extent and prevalence of urchin barrens and other important reef habitats in New Zealand. The mapping carried out covered nearly 14% of subtidal reefs (<30 m deep) across the northeastern New Zealand study region. In each of the seven locations examined, urchin barrens were extensive on shallow-fished reefs, in some cases covering nearly half of the shallow reef, but the extent of barrens varied both within and among locations. Overall, approximately one-third of the shallow reefs mapped were dominated by urchin barrens, which when extrapolated across the total area of reef (<30 m depth) across the study region equates to an area of 30.5 km² of urchin barrens. Long-term comparisons at two of the locations demonstrate that macroalgal forests historically dominated these reef areas, but urchin barrens were established by the early 2000s and have persisted for at least the last two decades. Within the two long-established marine reserves, where reef predators are large and abundant, urchin barrens were rare and shallow reefs were instead dominated by kelp forest. The habitat maps presented here provide important insight into the scale and current extent of urchin barrens across the region that can be used to both guide ecosystem-based management and measure kelp forest recovery following management actions.

The extent of urchin barrens at the fished locations mapped in this study ranged from 7–49% of the shallow reef area. This is broadly consistent with previous estimates across wider parts of northeastern New Zealand that found barrens to cover ~10–40% of shallow-fished reefs (<12 m depth) at open coast locations (Shears et al. 2008).

On fished reefs, the abundance and size of key predator species (spiny lobster *Jasus edwardsii* and snapper *Chrysophrys auratus*) are greatly reduced (Eddy et al. 2014; Booth 2017), meaning they are not large or abundant enough to play an ecologically important role in controlling urchin populations. As a result, the distribution of urchins and associated barrens in fished areas is mediated by environmental and other biological factors, which explains the large variability in urchin barrens observed within and among fished locations examined. This variability, and the extent and depth distribution of barrens, was broadly consistent with previous studies in northern New Zealand (Choat and Schiel 1982; Grace 1983; Shears and Babcock 2004). Urchin barrens are most prevalent at open coast locations with relatively clear water in this region, where they typically form a characteristic band on shallow reefs (Grace 1983; Shears and Babcock 2004; Walker 2007). This was evident at Maitai Bay, Doubtless Bay, Mimiwhangata and fished reefs at Leigh and Tāwharanui, with barrens extending deeper in the more exposed areas within these locations (e.g. Maitai Bay and Mimiwhangata).

The clear zonation patterns were less evident at the other locations, which likely relates to differences in reef topography and slope, that influence the horizontal extent of barrens and also the ability to map these habitats using aerial imagery. For example, at Maunganui Bay, the shallow reef area where barrens occur is relatively steep and drops quickly away to sand and therefore the reefs have limited horizontal extent. While barrens are present in this shallow zone in parts of Maunganui Bay (Figure S3), barren areas are hard to distinguish on steeply sloping reefs using 2D aerial imagery and consequently, they are only quantified as occurring in a relatively small and narrow margin. This highlights a limitation of these aerial-based methods for mapping steep reef habitats and likely means that the low percent cover of barrens in this location is a conservative estimate of their relative extent. In contrast, on more extensive and gradually sloping shallow reefs such as at Maitai Bay and Mimiwhangata, there is comparatively more shallow reef habitat for barrens to occur, and habitats on these horizontal areas are more clearly distinguished and easily mapped with aerial imagery-based methods.

Urchin barrens were also relatively rare on shallow reefs on the exposed (northeastern) side of Waewaetoria Island and instead were more extensive on more sheltered reefs in this location. While urchins may be prevented from forming barrens in shallow water at exposed sites by high wave action (Siddon and Witman 2003; Vanderkilt et al. 2009), it is unlikely these reefs are any more exposed than other northeast-facing locations where barrens were prevalent in our study. The reefs on the northeastern side of Waewaetoria Island area are highly complex with highly variable bathymetry including large drop-offs and outcrops of bedrock. High reef complexity has been shown to inhibit sea urchin grazing and promote more resilient kelp forests (Randell et al. 2022), so it is likely highly complex reefs are less prone to barren formation. While we observed some general patterns in the distribution of barrens in relation to wave exposure and depth, there are exceptions, and more investigation is needed into the factors driving spatial variation in the extent of barrens in Northland. Furthermore, the variation in environmental conditions within each mapped location means that our approach of using the maximum depth of urchin barrens as the limit of the shallow reef provides a conservative estimate of the prevalence of urchin barrens in each location.

Within the two no-take marine reserves examined, snapper and crayfish are substantially larger and more abundant than in adjacent fished areas (Willis et al. 2003; LaScala-Gruenewald et al. 2021; Hanns et al. 2022), and the extent of urchin barrens was considerably reduced (<2%) compared to adjacent fished areas (~38%). Previous work has clearly documented long-term declines in urchins and associated barrens in the Leigh reserve (Babcock et al. 1999; Shears and Babcock 2002; Leleu et al. 2012), higher rates of predation on urchins inside vs outside reserves (Shears and Babcock 2002), and ongoing monitoring over the last two decades have shown that the contrasting kelp and barren states between the reserve and fished areas has persisted (Peleg et al. 2023). In contrast, in the Mimiwhangata Marine Park which allows recreational fishing, we found that barrens were extensive, which is not surprising given key predator species remain at low levels (Denny and Babcock 2004; Shears et al. 2006).

Our examination of historical aerial imagery demonstrates that in the 1940s and 1950s shallow reefs were dominated by macroalgal forests, but by the early 2000s these areas had extensive urchin barrens, which have persisted since. Similar, long-term changes from before the 1960s have been described at additional sites at Mimiwhangata (Kerr and Grace 2005), the Bay of Islands (Booth 2017), and Little Barrier Island (Dartnall 2022), demonstrating that these changes are reflective of a wider long-term trend across the region. The establishment of urchin barrens in northern New Zealand in the 1950–70's coincides with the onset and peak in commercial fishing for snapper and spiny lobster (Eddy et al. 2014; Booth 2017; Durante et al. 2022). Destructive grazing of kelp by urchins was first observed at Leigh in the late 1950s (Dromgoole 1964) and by the 1970s urchin barren habitats were common at Mimiwhangata, Leigh and other parts of northeastern New Zealand (Ayling et al. 1981; Choat and Schiel 1982; Grace 1983). Historical accounts from local hāpu at Maitai Bay (pers. com. Erik Raharuhi) and Mimiwhangata (Kerr and Grace 2005) are also consistent with the timing of long-term changes. In particular, lifelong divers, respected for their mātauranga (knowledge) and kaitiakitanga (guardianship) of their rohe moana (coastal area), had not seen urchin barrens in their early years of diving and there were no such historical accounts in their knowledge passed down through the generations. The recent aerial imagery (Figure 3) demonstrates that urchin barrens have remained relatively stable on fished reefs for at least the last two decades and this is further supported by long-term monitoring of shallow reefs at Leigh and the Mokohinau Islands (Balemi and Shears 2023; Peleg et al. 2023).

Globally, kelp forests are under threat from a multitude of global and local-scale stressors (Krumhansl et al. 2016). Similarly, in parts of Aotearoa New Zealand kelp forests have been impacted by a range of factors including warming ocean temperatures, marine heatwaves and high turbidity from sediment runoff (Thomsen et al. 2019; Tait et al. 2021; Cornwall et al. 2023). The dominant kelp forest-forming species *Ecklonia radiata* is not near its thermal range limits in northern NZ (Wernberg et al. 2019) and there is no evidence that long-term warming in this region has led to declines in *E. radiata* forests as seen for other kelp species in other parts of the country (Cornwall et al. 2023). Instead, the continued expansion of *C. rodgersii* populations in northern New Zealand (Balemi and Shears 2023) is likely to be a major source of kelp loss and expansion of barrens, particularly into deeper water. The bulk of the mapping in this study was carried out prior to recent increases in *C. rodgersii* (Balemi and Shears

2023), and *E. chloroticus* was the dominant urchin species in barrens at the time of mapping. However, the abundance of *C. rodgersii* has increased at some of the northern-most locations (e.g. Maitai Bay; VK pers. Obs.), highlighting a risk that urchin barrens could expand in the future with further increases in *C. rodgersii*.

While other stressors may impact kelp forests, our results clearly demonstrate that sea urchin grazing is a major driver of kelp loss across this region. Low light can impact the productivity and lower depth extent of *E. radiata* forests at highly turbid locations in northern New Zealand (Blain et al. 2021), but as seen in this study kelp forms extensive forests in deep water on open coasts demonstrating that light is not a limiting factor on shallow reefs. While other factors such as storms or kelp die-off may lead to the loss of kelp in northeastern New Zealand, these generally only represent short-term and small-scale disturbances on the northeast coast (Schiel 1988; Cole and Babcock 1996; Haggitt 2004). In contrast, once sea urchins graze down kelp forests, comparatively low numbers can maintain these barren areas devoid of kelp (Shears and Babcock 2003; Ling et al. 2015) and these areas can persist for decades once established (Figure 3; Peleg et al. 2023). Environmental variation and other stressors can lead to fluctuations in urchin abundance (Shears and Ross 2009; Hernández et al. 2010), which can lead to associated expansion or contraction of barrens over time. However, as seen in this study, where healthy predator populations persist they can be effective in controlling urchin populations and increasing kelp forest stability (Peleg et al. 2023). Precautionary management that aims to rebuild predator populations over large scales may help prevent and dampen further expansion of urchins and associated loss of kelp forests that may be facilitated by future warming (Balemi and Shears 2023).

Our results support that long-term marine protection can remedy these historic and long-term impacts of overfishing, but questions remain whether similar results can be achieved through improved fisheries management of key predators. Here where predators were fully protected from fishing, the recovery of kelp forests took decades (Babcock et al. 2010; Peleg et al. 2023), whereas in the partially protected MPA, these ecological effects have not occurred. Sea urchin harvest (Cresswell et al. 2023) or active removal (Miller et al. 2024) can promote kelp forest recovery in urchin barrens, but these approaches do not provide a long-term solution or restore full ecosystem function and resilience. To effectively control urchins, predators need to be sufficiently abundant and large enough over regional and local scales. It is therefore essential to develop a combination of approaches to maximise the benefits of a network of fully protected reserves and fisheries management reform aiming to rebuild predator populations to ecologically relevant levels. This requires MPA planning and fisheries management to be coordinated in order to achieve and maximise conservation and fisheries goals (Gaines et al. 2010). Active restoration of kelp forests through sea urchin removal provides an additional management tool, but as for MPAs and fisheries management reforms, this needs to be part of a wider and coordinated ecosystem-based approach, rather than be seen as a solution on its own (Miller et al. 2022).

Looking to the future, ocean warming will likely further threaten kelp forests in northern New Zealand (Cornwall et al. 2023) and recent record-breaking marine heatwaves provide insights into what this will look like, with unexpected impacts on rocky reef ecosystems (Shears et al. 2024). Our climate emergency calls for action to manage and reverse the current impacts, and in doing so increase the resilience of kelp forests. It is

time to recognise the ecological and social significance of kelp forests and that full restoration is possible (Bennett et al. 2016, Eger et al. 2023a). This is reflected in the recent global call to restore 1 million and protect 3 million hectares of kelp forests by 2040 (Eger et al. 2023a). The foundation ecological studies are in place, what is urgently needed now is to act proactively for our kelp forests.

A proactive future is now called for. Overfishing appears to be the greatest manageable threat to kelp forest ecosystems. Management should focus on minimising fishing impacts and restoring populations of functionally important species in these systems. (Steneck et al. 2002)

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Data availability statement

An archive of the GIS resources involved in this research have been loaded into a DRYAD archive, <https://doi.org/10.5061/dryad.8gtht76w3>.

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