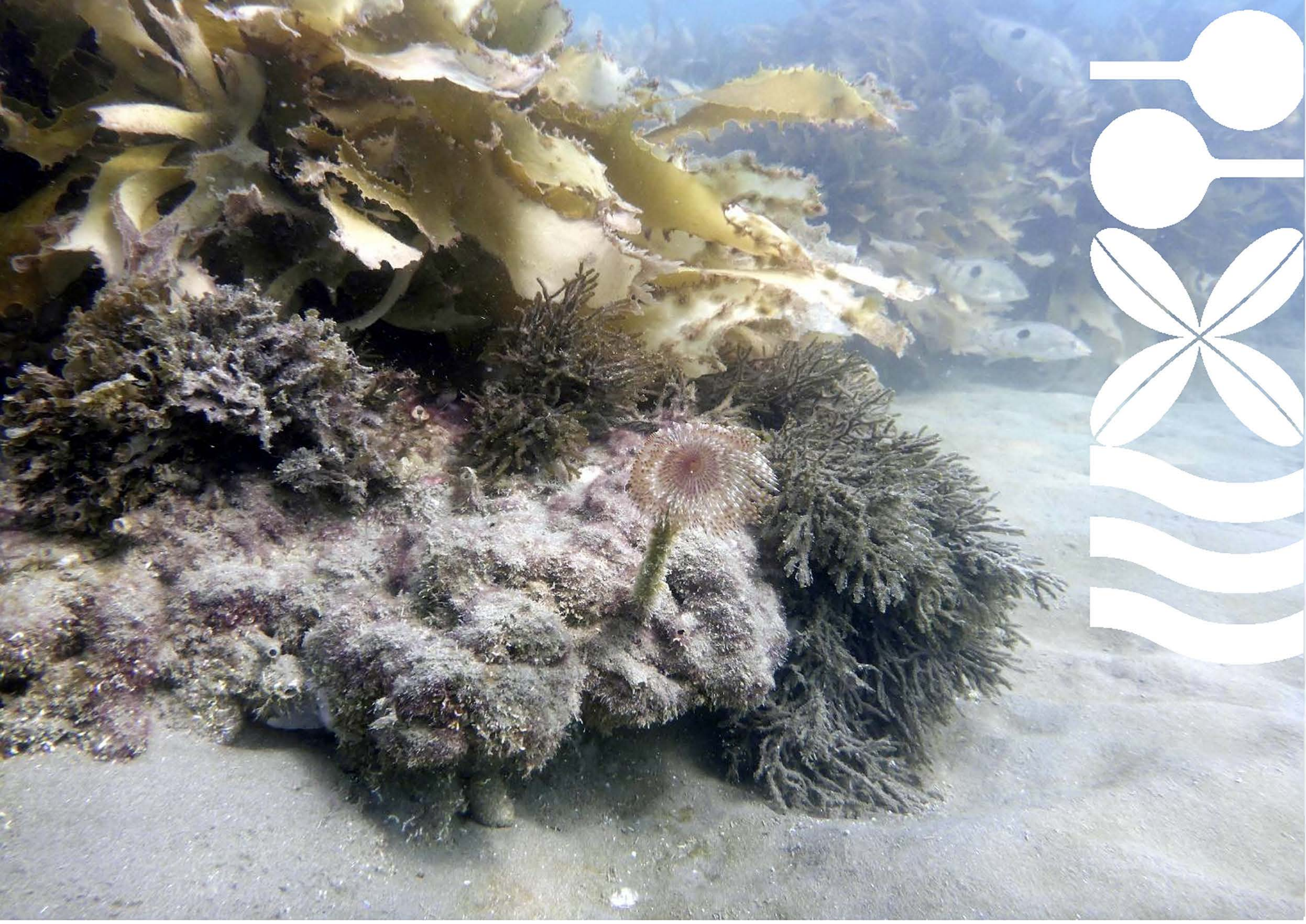

Tāmaki Makaurau / Auckland

East Coast Subtidal Reef Monitoring Report: 2007 to 2025

State of the Environment Reporting

N. Shears

Leigh Marine Laboratory, University of Auckland

September 2025

Technical Report 2025/24

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Common invasive species on shallow reefs at Long Bay; Mediterranean fanworm *Sabella spallanazanii* (centre) and brown seaweed *Cladostephus hirsutus* (centre-right). The native kelp *Ecklonia radiata* in the background. Photograph by N. Shears.

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Executive summary

Subtidal rocky reefs play a critical role in Aotearoa New Zealand's coastal marine ecosystems, supporting high biodiversity and a wide variety of culturally, commercially, and recreationally important species. These reefs, often dominated by macroalgal forests, provide essential three-dimensional habitat, food, and refuge for many species. However, subtidal reef ecosystems face increasing pressure from human impacts, such as overfishing and sedimentation from land-based activities, as well as the escalating effects of climate change. By monitoring the health of subtidal rocky reefs, we can better understand how these threats impact our coastal environments and inform the strategic management of these ecosystems.

Monitoring of shallow subtidal reef sites at Meola Reef-Te Tokaroa (n=3) in the Waitematā Harbour, East Coast Bays-Whangaparāoa (ECB-WGP; n=19), and Leigh (n=4) in the outer Hauraki Gulf, has been carried out for Auckland Council since 2007. The reefs at Meola and ECB-WGP were historically characterised by dense macroalgal forests dominated by large fucoids (predominantly *Carpophyllum* species), with high numbers of gastropods, whereas sites at Leigh are typically dominated by the kelp *Ecklonia radiata*. Sampling across all 26 sites has been carried out on a biennial basis since 2019 and this report examines trends in key species and benthic covers across these locations from 2007-2025.

Across the three general areas, a number of region-wide trends were observed. This included large declines in the endemic fucoids (*Carpophyllum maschalocarpum* and *C. flexuosum*) and in total gastropod numbers (multiple species), as well as general declines in the golf ball sponge *Tethya burtoni* and solitary ascidians. Conversely, there has been a general increase in turfing and filamentous algae, and invasive species (e.g. Mediterranean fanworm *Sabella spallanazanii*), across all areas.

At Meola reef, overall macroalgal canopy cover has remained high but there has been a long-term shift from macroalgal canopies dominated by *C. maschalocarpum* and *C. flexuosum,* to the kelp *E. radiata* and *Sargassum sinclairii.* The dominant substratum covers (sediment, crustose coralline algae (CCA)) have generally remained stable, but there has been an increase in filamentous algae, red turfing algae and oysters in recent years. Increasing numbers of invasive species have been recorded in recent years including the brown seaweed *Cladostephus hirsutus* and the ascidians *Symplegma brakenhielmi* and *Ascidia latesiphonica.*

At ECB-WGP locations, macroalgal forest canopy cover was relatively stable until 2018/2019 after which there have been declines in canopy cover across all but two sites. This has coincided with a decline in CCA and increases in turfing algae, filamentous algae, sediment and the invasive seaweed *C. hirsutus* at many sites. There has also been an increase in *Sargassum* spp. and *Cystophora retroflexa.* At sites where canopies have remained intact, these are largely dominated by *E. radiata* and the changes in substratum covers are less evident. The invasive ascidians *Styela clava, S. brakenhielmi* and *A. latesiphonica* have established across most sites. Multi-species

mortality events including *Carpophyllum* species, *E. radiata,* the small brown seaweed *Zonaria turneriana,* and sponges (*T. burtoni*), have been observed over the last five years at some sites.

At Leigh, the kelp *E. radiata* dominates three of the four sites and has been variable through time, which is generally related to periodic dieback events. However, at Nordic Deep there has been a general decline in kelp in recent years that does not appear to be related to dieback events. The fourth site has transitioned from urchin barrens to turfing algae over the monitoring period. In general, there has been a decline in CCA, increase in sediment, filamentous algae and turfs, and these changes in benthic covers are greatest at sites where kelp canopy is reduced.

The long-term changes over the monitoring period and particularly over the past five years are generally consistent with a transition to an unhealthier state on temperate reefs, with a reduction in canopy cover of forest-forming macroalgal species and an increase in filamentous and turfing algae, sediment cover, and invasive species at many sites. The widespread nature of these changes across the monitoring locations suggests they are regional and likely related to large-scale factors, rather than any specific local-scale or catchment-based impacts. There has been a steep increase in water temperature in the Hauraki Gulf over the monitoring period and there have been a number of unprecedented climatic events in recent years, with record-breaking marine heatwaves in 2022 and extreme storm and rainfall events in 2023. While the exact cause and mechanisms for the observed changes remain undetermined, it is likely that recent changes and observed mortality events are linked to the combined stress of these climatic events. Similar patterns are increasingly being reported on temperate reefs globally and related to increasing anthropogenic stressors and climate change. These trends are expected to continue with climate change, so ongoing monitoring is necessary to document these changes, to understand links with climate variability, and to assess potential recovery dynamics following extreme events. Targeted research is also needed to better understand causal mechanisms which can then help inform what management measures can be implemented to mitigate future impacts, increase resilience of reef communities and potentially restore these ecosystems.

Contents

1 Introduction

Shallow temperate reef ecosystems are typically dominated by forests of kelps and/or large fucoid seaweeds and are of high ecological, economic and cultural importance (Smale et al., 2013; Steneck and Johnson, 2013; Bennett et al., 2016). These underwater forests are generally considered to be characteristic of good ecosystem health (Sala et al., 2012; Peleg et al., 2023a) as they provide habitat and increased structural complexity that supports a plethora of species living on, under, or amidst the plants (Steneck and Johnson, 2013; Teagle et al., 2017). These forests are also important contributors to coastal food web processes by supporting primary and secondary productivity and exporting detrital matter to adjacent ecosystems (Taylor, 1998; Smale et al., 2013; Wernberg et al., 2019).

The loss of kelp and macroalgal forests in many temperate regions has been linked to a range of anthropogenic stressors including the cascading impacts from fishing, land-based impacts such as increased sediment and nutrient run-off, warming oceans and marine heatwaves (Wernberg et al., 2013; Ling et al., 2015; Foster and Schiel, 2010). Destructive overgrazing of kelp by sea urchins is a long-documented cause of macroalgal forest loss, that results in extensive areas of reef lacking kelp and dominated by sea urchins, referred to as “urchin barrens” (Ling et al., 2015). This has largely been attributed to the overexploitation of sea urchin predators but has also been linked to climate-induced range extension of sea urchin species (Steneck and Johnson 2013; Ling et al., 2009). A more recent phenomenon observed in many temperate regions is the loss of kelp forests and shift to filamentous and turfing algal species (Filbee-Dexter and Wernberg, 2018). These shifts are largely linked to changes in environmental and climatic conditions, such as increases in nutrients, sedimentation and water temperature, that favour smaller more ephemeral seaweeds over large kelps and fucoids (Wernberg et al., 2016; Connell et al., 2008). Both urchin barrens and turfs represent less structurally complex, productive and diverse ecosystem states than macroalgal forests and therefore can be considered a shift to a more degraded reef state.

Shallow reefs in Tīkapa Moana / Hauraki Gulf are typically dominated by large brown macroalgal forests that vary in morphology and species composition as a function of wave exposure, depth, and turbidity (Grace, 1983; Shears and Babcock, 2004; Blain et al., 2020). The macroalgal forests that characterise shallow sheltered inner Gulf reefs typically comprise a mix of the kelp *Ecklonia radiata* and the fucoids *Carpophyllum flexuosum*, *C. maschalocarpum*, *C. plumosum*, *Cystophora retroflexa* and *Sargassum* spp. (Shears and Babcock, 2004). On these shallow inner Gulf reefs, the common sea urchin *Evechinus chloroticus* (kina) is present in low abundance and does not have major impacts on benthic community structure through overgrazing (Walker, 2007). In general, these macroalgal forests in the inner Gulf are considered more stable than those found on more exposed reefs in the outer Hauraki Gulf (Walker, 2007; Shears, 2017), which are subject to larger wave disturbance and also greater grazing pressure by sea urchins (Choat and Schiel, 1982). On outer Gulf reefs *Evechinus chloroticus* can occur at high densities and overgraze kelp, forming a distinctive band of urchin barrens at depths of ~2-10m (Choat and Schiel 1982; Shears and Babcock, 2004). Urchin barrens are typically more extensive in fished areas compared to marine reserves and are considered an ecological indicator of the impacts of fishing on important sea urchin predators (e.g., snapper (*Chrysophrys auratus*) and crayfish (*Jasus edwardsii*)) on rocky reefs (Babcock et al., 1999; Shears and Babcock 2002). Below the urchin barrens, monospecific stands of the stipitate kelp *E. radiata* typically dominate. The lower limit of *E. radiata* on reef is determined by water clarity, with *E. radiata* typically reduced to shallower depths in more turbid inner Gulf areas (Blain et al., 2021). *E. radiata* forests are also known to undergo periodic dieback events, where the entire adult canopy dies off (Cole and Babcock, 1996). This is typically followed by high levels of *E. radiata* recruitment, presumably from a "gametophyte bank" beneath the canopy, and the forest fully recovers in 1-2 years. The environmental drivers of *E. radiata* dieback are not well understood, but these events do not appear to be clearly linked with particular climatic events such as marine heatwaves or large storms and runoff events.

In general, there is little evidence of long-term impacts of climate change on reef ecosystems and macroalgal forests in northeastern New Zealand (Cornwall et al., 2025). However, in 2022 the Hauraki Gulf experienced record-breaking ocean temperatures and marine heatwaves (Shears et al., 2024), which impacted on sponge communities (Bell et al., 2023), resulted in expansion of the invasive ascidian *Symplegma brakenhielmi* (Spyksma et al., 2024), and had a number of other anecdotal impacts on reef ecosystems (e.g. Shears et al., 2024). Following these marine heatwave events in 2022, a number of extreme storms and rainfall events occurred in early 2023, including the Auckland Anniversary floods (27 January 2023) and Cyclone Gabrielle (14 February 2023) (Auckland Council, 2023). These events likely exerted additional stress on shallow reef ecosystems in the Hauraki Gulf. While our understanding of the impacts of such events is currently limited, long-term monitoring of reef ecosystems in the Hauraki Gulf provides a unique opportunity to understand how such events may be impacting reef ecosystems.

The Auckland East Coast Subtidal Reef Marine Monitoring Programme has been monitoring shallow reef communities at Meola Reef-Te Tokaroa and a number of locations in the East Coast Bays-Whangaparāoa (ECB-WGP) area since 2007 (Shears, 2017). This programme represents an amalgamation of the Long Bay Marine Monitoring Programme and the Meola Reef State of the Environment monitoring programme (Ford and Pawley 2008; Shears, 2010a and 2010b). These programmes both focus on shallow subtidal reef ecosystems (<3m depth) that have historically been dominated by dense forests of kelp and fucoids. Previous analysis of the data across both programmes from 2007-2013 found that despite variation in benthic communities over time, the dominant macroalgal forest communities persisted, and there were no signs of ecosystem degradation besides an increase in non-indigenous species (Shears, 2017). Further analysis of data from 10 sites in the Long Bay area from 2007-2019 found a large collapse in the gastropod community, a shortening of macroalgal canopies and an increase in the number of non-indigenous species (Peleg et al., 2023a). While the cause of this decline and mechanisms responsible are not fully understood, the decline correlated with an increase in sediment cover on the reefs and warming ocean temperatures over the monitoring period. Peleg et al. (2023a) concluded that overall ecosystem health had declined at five of the ten sites in the Long Bay area. To better understand how these changes at Long Bay related to wider regional changes, historic monitoring sites at Campbells Bay, Manly, Stanmore Bay and Waiwera (originally sampled from 2007-2011/12) were reinstated in 2020 and have been sampled biennially since (see Methods for details). In addition, four sites at Leigh on more wave exposed reefs in the outer Hauraki Gulf have been monitored since 2013 as part of the programme.

This report examines trends in reef communities at Meola Reef-Te Tokaroa in the Waitematā Harbour and at all East Coast Bays-Whangaparāoa locations from 2007-2025, and at Leigh from 2013-2024.

Supporting information

This report is one of a series of technical publications prepared in support of *Te oranga o te taiao o Tāmaki Makaurau – The health of Tāmaki Makaurau Auckland's Natural Environment in 2025: a synthesis of Auckland Council State of the Environment reporting.*

All related reports (past and present) are published on the Knowledge Auckland website.

All data supporting this report can be requested through our Environment Auckland Data Portal. Here you can also view live rainfall data and use several data explorer tools.

2 Methods

2.1 Monitoring programme and site locations

The Long Bay Marine Monitoring Programme was established in 1999 to detect and document the impact of urban development at Long Bay on the subtidal marine environment (Ford and Pawley, 2008; Shears, 2010a). This involved sampling at five sites in each of six locations (Waiwera, Stanmore Bay, Manly, Long Bay, Torbay and Campbells Bay) along the East Coast Bays-Whangaparāoa coast (ECB-WGP). Earthworks for the Long Bay development began in 2010 and extensive housing has now been built at Long Bay, and another large housing development has occurred in the adjacent Ōkura catchment. The Long Bay monitoring programme has been through a number of iterations with sites and methods varying over time (outlined in Shears, 2010a), so data from prior to 2007 is excluded from the current study.

The Meola Reef monitoring programme was established by the Auckland Regional Council in 2001 and included monitoring biological communities at six subtidal sites at Meola Reef (Shears 2010b) using the same methods employed at Long Bay. This programme was a State of the Environment (SOE) monitoring programme and was designed to determine trends in reef communities at Meola Reef. It aimed to compare these trends with changes recorded at other sentinel locations within the Auckland region and interpret any community changes in the context of two major threats to ecosystem health: sedimentation from urban development and toxicity from urban discharges.

In 2011, the Long Bay and Meola Reef monitoring programmes were combined to form the Auckland East Coast Subtidal Reef Marine Monitoring Programme. The number of sites at Meola was reduced to three and in the ECB-WGP region the number of sites were reduced including the cessation of monitoring at Waiwera and Stanmore Bay (Shears, 2017). In addition, four sites at Leigh were added in 2013 to provide an outer Gulf comparison. In 2020, a number of the original sites at Campbells Bay, Waiwera and Stanmore Bay were reinstated to provide a greater regional picture of the reef communities. Since 2019, sampling across all sites is carried out biennially, alternating between Campbells Bay, Waiwera, Stanmore Bay, Meola and Leigh sites in one year and Long Bay, Torbay and Manly sites the next (Table 1). Note that Meola was unable to be sampled in 2021 due to poor weather conditions and extremely low water clarity during site visits. Sites at Meola, Campbells Bay, Stanmore Bay, Waiwera and Leigh were most recently sampled in 2024 and Torbay, Long Bay and Manly were sampled in 2025 (Appendix 1).

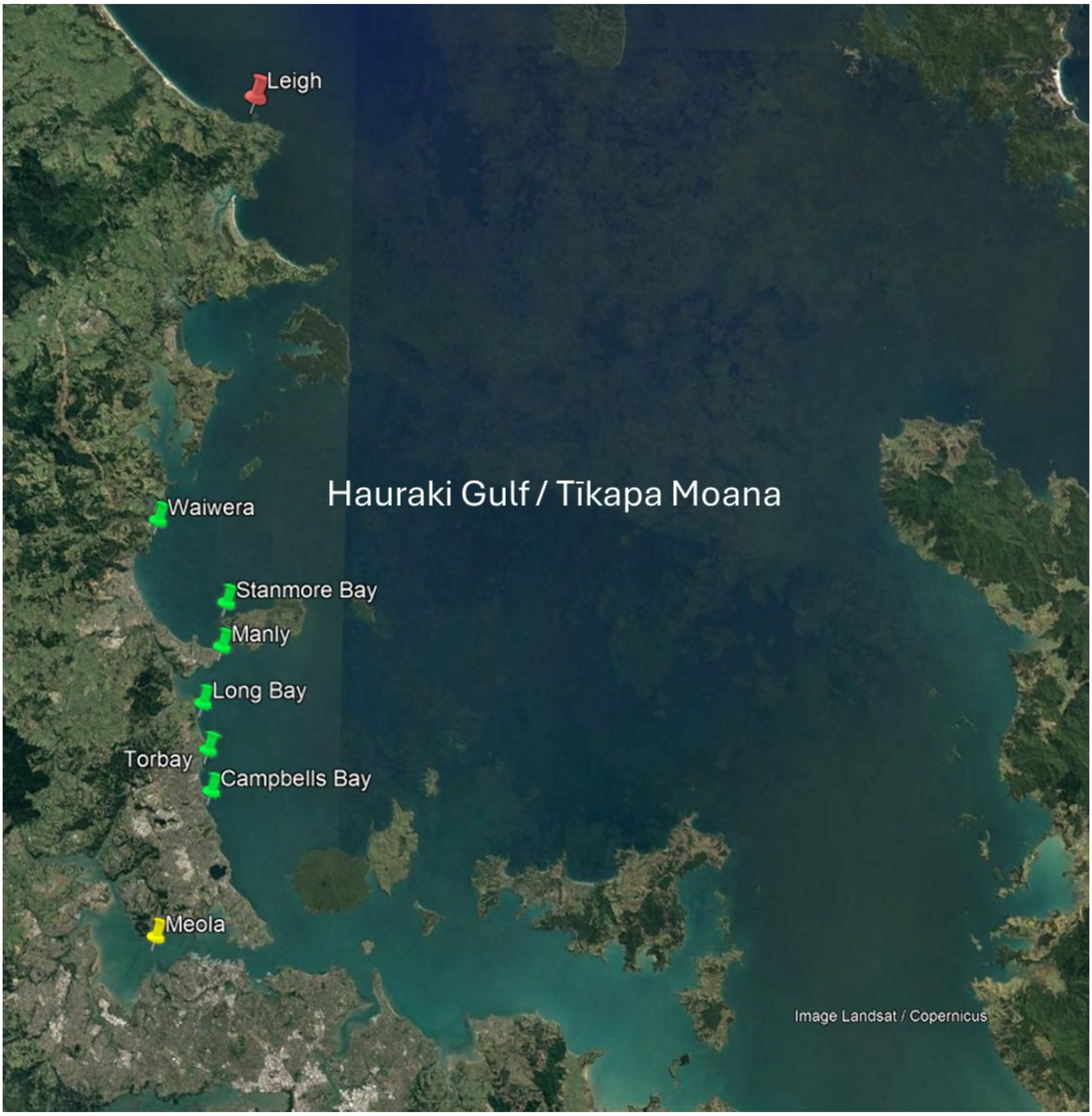
In total, three sites at Meola Reef, 19 sites at ECB-WGP locations (Waiwera, Stanmore Bay, Manly, Long Bay, Torbay and Campbells Bay) and four sites at Leigh are currently monitored biennially (Figure 1, Appendix 1). All sites at ECB-WGP and Meola Reef are located on shallow rocky reef at depths of 1-3m below the mean low water mark. At Leigh, two sites are located on reef at 6-7m depth (ABC and Nordic Shallow) and two are located at 15m depth (North Reef and Nordic Deep). ABC and North Reef are kelp (*E. radiata*) dominated and located in the Cape Rodney to Okakari Marine Reserve. Nordic Shallow and Nordic Deep are outside the marine reserve near Leigh

Harbour. Nordic Shallow was an urchin barren when the sampling began, with patches of kelp nearby, whereas Nordic Deep was a relatively sparse kelp forest comparable to North Reef.

All monitoring sites are marked with a steel base and surveys are carried out within five meters of these markers. At some sites, surveys are carried out on adjacent or inshore reef. Care is needed to ensure this is kept consistent from year to year.

Figure 1. Auckland east coast subtidal reef monitoring locations in the Hauraki Gulf/Tīkapa Moana (A) and monitoring sites within each of the three main areas: Leigh, East Coast Bays-Whangaparāoa (ECB-WGP) and Meola Reef-Te Tokaroa in the Waitematā Harbour (B). Yellow lines indicate marine reserve boundaries at Leigh and Long Bay (B). Site positions given in Appendix 1.

A



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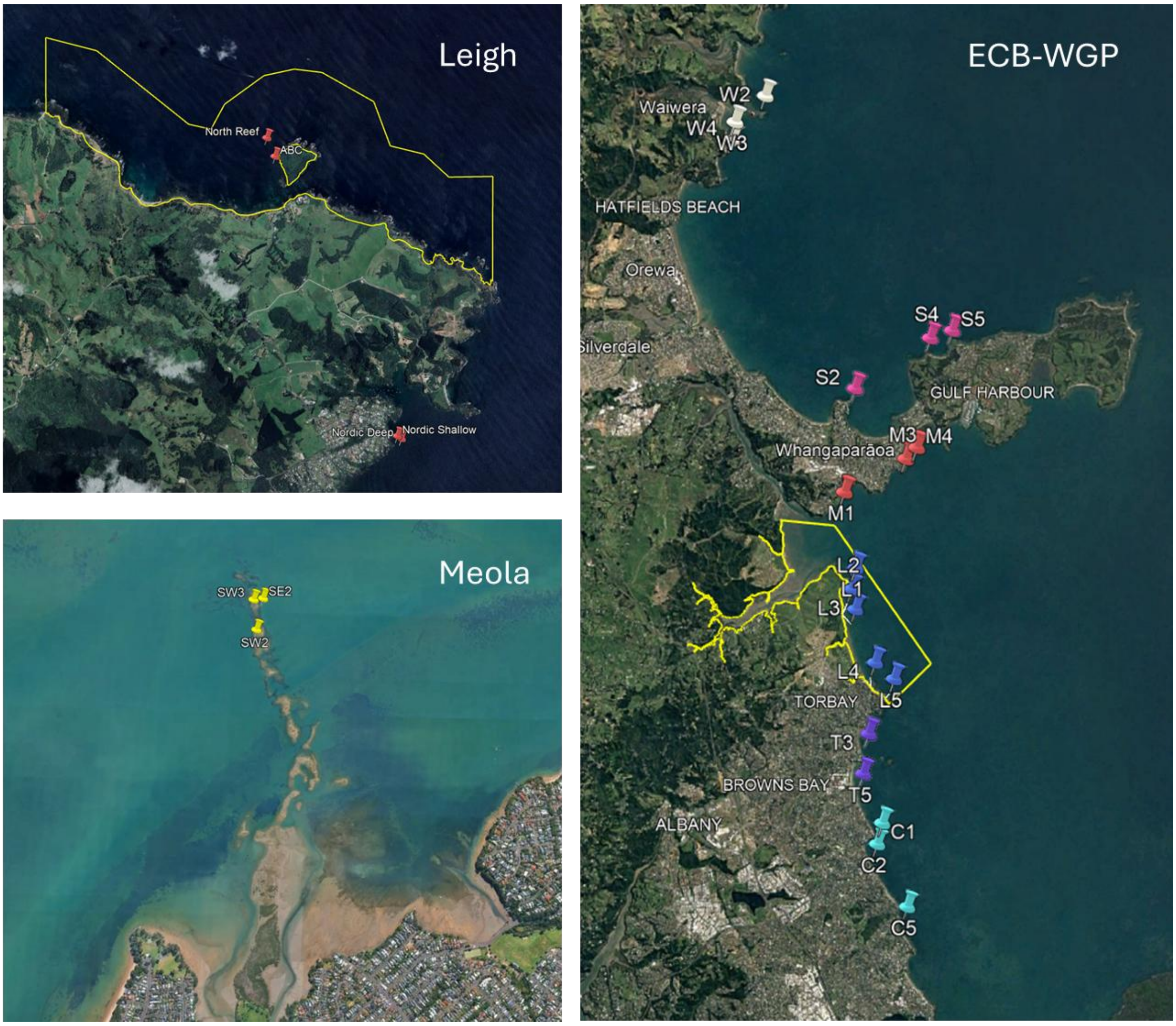


2.2 Biological surveys

Subtidal surveys at each of the monitoring sites are typically carried out in February/March each year, but there has been some variation in timing depending on weather conditions. Sampling in late winter/spring is avoided due to blooms of a number of small seaweed species that occur at this time of year. Large brown seaweeds (excluding the invasive kelp *Undaria pinnatifida*) and invertebrate species generally don't show strong seasonal variation on the shallow reefs sampled. Sampling methods are described in detail in Ford and Pawley (2008) along with details of changes in sampling methods over the monitoring period. In summary, the biological assemblages at each site are surveyed using seven haphazardly placed 1m^2 quadrats. All large brown macroalgae, subcanopy macroalgae, and mobile macroinvertebrate species are counted in each quadrat. Some distinctive and easily defined sessile invertebrates are also counted, including solitary ascidians, golf ball sponges (*Tethya burtoni* and *Tethya bergquistae*), and the invasive clubbed ascidian *Styela clava* and the Mediterranean fanworm *Sabella spallanazanii*. The percentage cover of turfing, filamentous and encrusting algae, as well as sessile invertebrates, is estimated visually within each quadrat. The percentage cover of sand, sediment and bare rock are also estimated. Measurements of large brown algae and mobile macroinvertebrates (e.g. gastropods) are taken for all individuals in five of the quadrats.

The canopy cover of habitat forming macroalgal species (large brown algae) has also been visually estimated in each quadrat since 2018. Total canopy cover data for large brown algae from 2018-2025 was used to estimate historic canopy cover throughout the time series based on the equation below relating canopy cover to cumulative length of large brown algae (summed lengths of all species) for each quadrat ($R^2 = 0.882$):

Estimated canopy cover = 1.708 x sqrt(cumulative length of large brown algae)

Estimates of canopy cover were only made for quadrats where sizes were measured (n=5 per site). If large brown algae were absent from these quadrats an estimated canopy cover of zero was used. If estimated cover was greater than 100% based on the equation above, this was constrained to 100%.

2.3 Analysis

To highlight the contrasting community composition of reefs between Meola, ECB-WGP and Leigh over time, a principal coordinates (PCO) analysis was carried out based on square-root transformed mean density of the 73 most abundant taxa at each location. Data for each of the three general areas (Meola, ECB-WGP and Leigh) is subsequently analysed and presented separately in the results.

For each of the three general areas, a multivariate analyses of community composition was carried out based on count data of the most dominant species over time. All analyses were carried out in PRIMER 7 with PERMANOVA+. For all areas multivariate analyses were completed using Bray-Curtis similarities, based on location means for each year at ECB-WGP, and site means at Leigh and Meola. Mean values were square root transformed to account for high counts of certain species. Principal Coordinates Analysis (PCO) was used to visualise patterns in community composition among locations or sites and years (the greater distance apart in the ordination means less similarity in community structure). In addition, correlations between PCO axes and dominant species/substratum types were presented as biplots to provide an exploratory analysis of the species contributing to the variation among areas and over time.

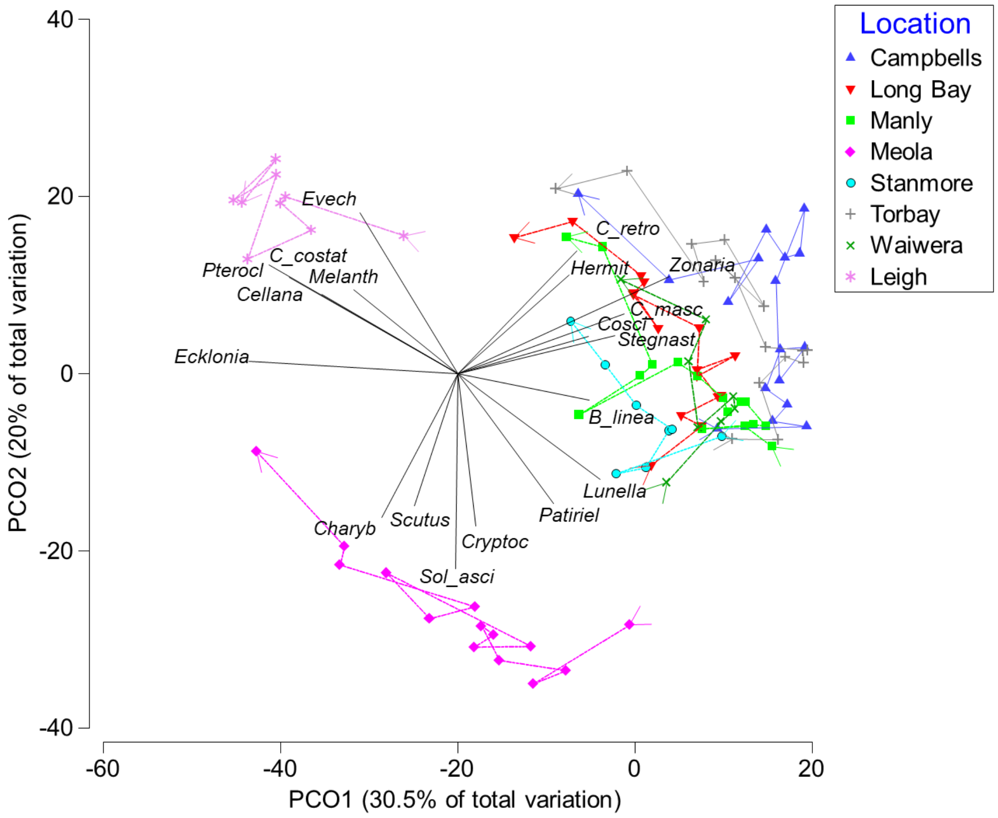
For each area, general site-level trends were examined for canopy cover of large brown algae, density of large brown macroalgal species, subcanopy seaweeds, dominant mobile and sessile invertebrates, and the cover of dominant benthic groups (e.g. sediment cover, crustose coralline algae). For each area, changes in density or cover of any other particular species of note (e.g. invasive species) were also presented. Analysis of trends in key species over time was carried out for each site using linear regression in R v4.5.0.

3 Results

3.1 Region-wide variation in reef assemblages

Principal coordinates (PCO) analysis based on the mean density of the 73 most abundant taxa across locations highlights the contrasting community composition between Meola Reef, ECB-WGP locations and Leigh (Figure 2). Meola and Leigh are clearly separated from the ECB-WGP locations, which overlap in a large cluster on the right of the ordination. Species that distinguished the Leigh sites included the limpet *Cellana stellifera* and seaweeds *Carpomitra costata, Pterocladia lucida* and *Melanthalia abscissa*, whereas the invasive paddle crab *Charybdis japonica* was only recorded at Meola sites. The community composition at Meola exhibited the largest shift in multidimensional space (from right to left), which largely corresponded to the shift from a macroalgal canopy dominated by *Carpophyllum* spp. to *Ecklonia radiata*.

Figure 2. Location-level variation in reef community composition. Principal coordinates ordination based on square-root transformed mean density of the 73 most abundant taxa at each location from 2007 to 2025 (Note: Leigh 2013-2024). Vectors indicate taxa with correlation coefficient >0.45. Taxa codes: Evech: *Evechinus chloroticus*, C_costat: *Carpomitra costata*, Melanth: *Melanthalia abscissa*, Pterocl: *Pterocladia lucida* Cellana: *Cellana stellifera*, Ecklonia: *Ecklonia radiata*, Charyb: *Charybdis japonica*, Scutus: *Scutus breviculus*, Sol_asc: Solitary ascidians, Cryptoc: *Cryptoconchus porosus*, Patiriel: *Patiriella regularis*, Lunella: *Lunella smaragdus*, B_linea: *Buccinulum linea*, Stegnast: *Stegnaster inflatus*, Cosci: *Coscinasterias muricata*, C_masc: *Carpophyllum maschalocarpum*, Hermit: Hermit crab, Zonaria: *Zonaria turneriana*, C_retro: *Cystophora retroflexa*.

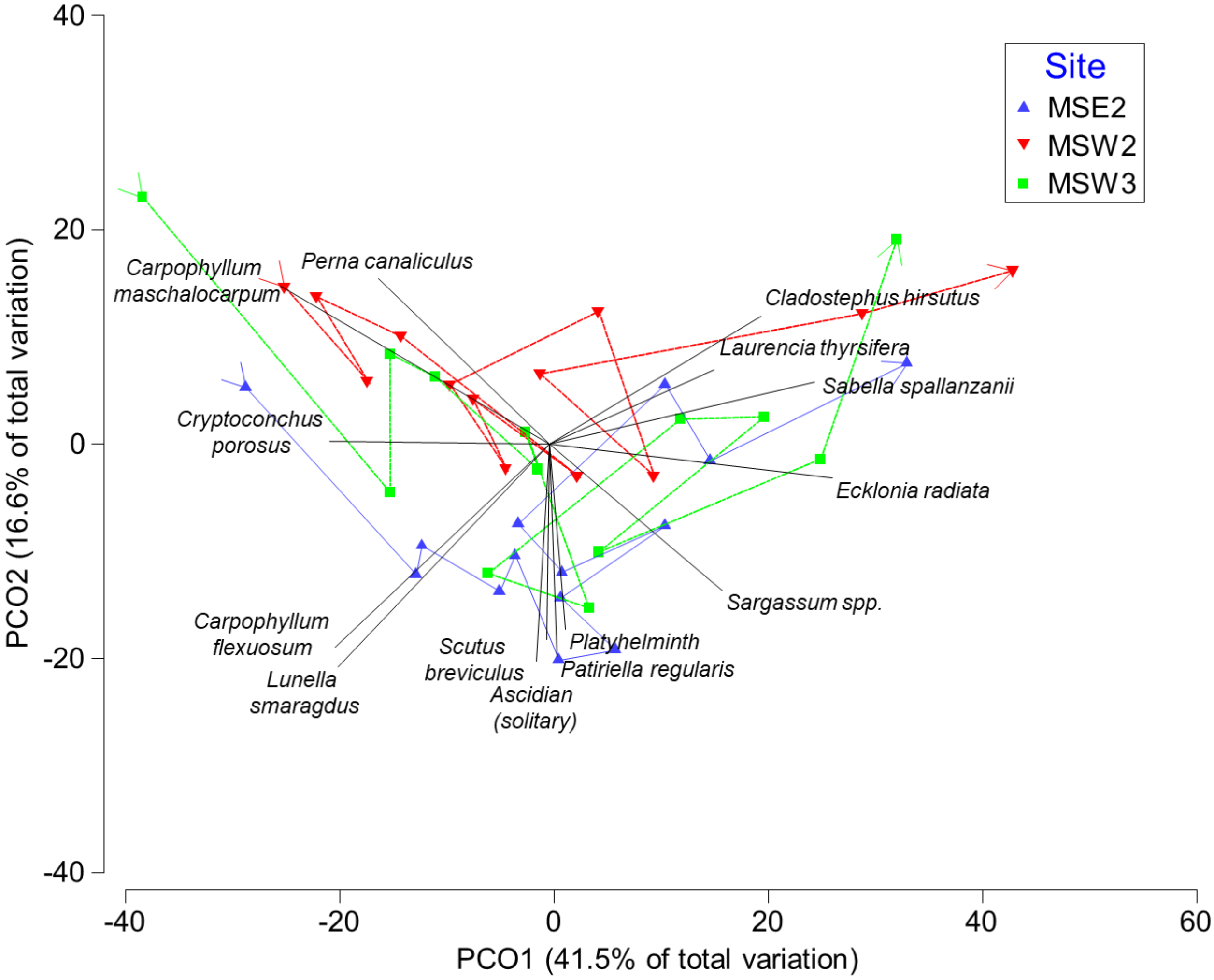


3.2 Meola Reef-Te Tokaroa

Reef community composition (counts)

Overall reef community composition has undergone a steady change over time across all three Meola sites, as indicated by a movement of all sites from left to right in the PCO ordination (Figure 3). Initially all sites were dominated by *Carpophyllum maschalocarpum* and/or *C. flexuosum* and high densities of *Lunella smaragdus*. However, these species have declined at all sites and there has been an increase in *Ecklonia radiata, Sargassum* sp. and the invasive seaweed *Cladostephus hirsutus* and fanworm *S. spallanzanii*.

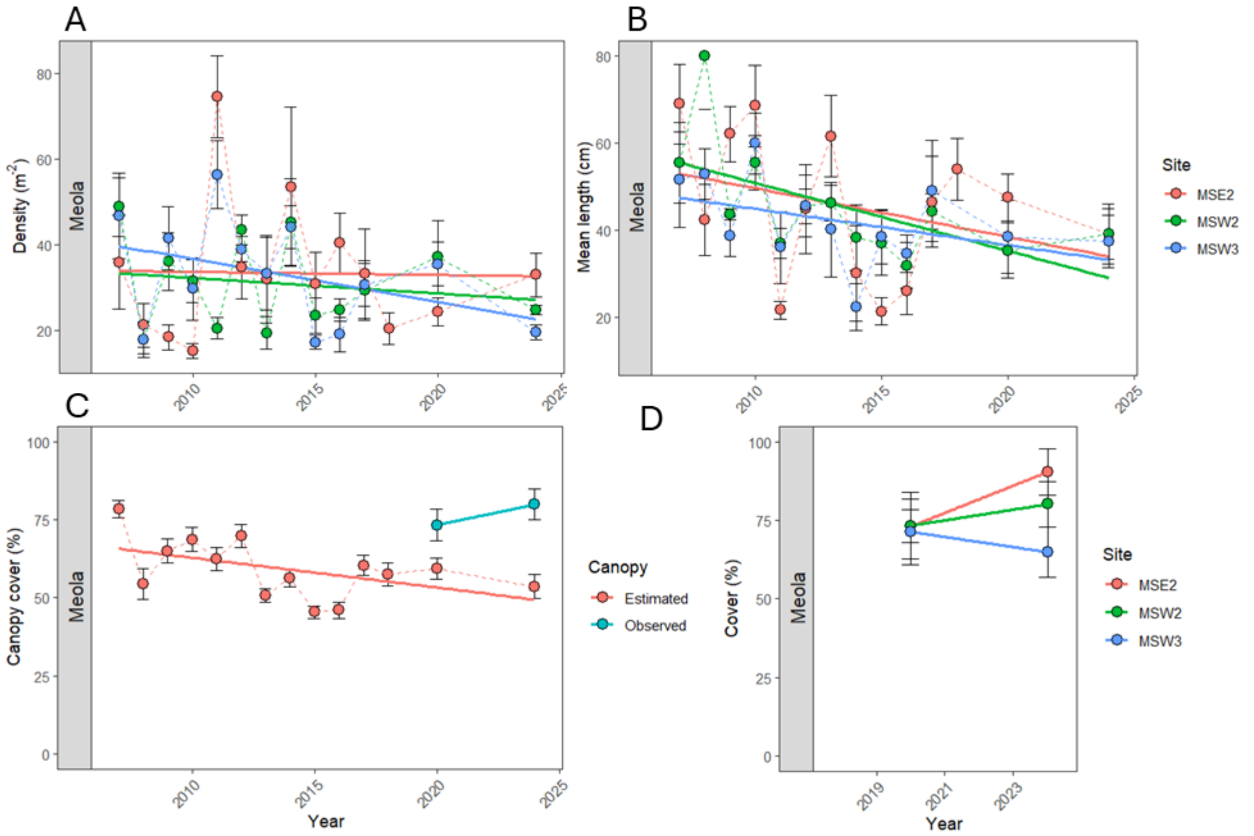
Figure 3. Site-level variation in reef community composition from 2007-2024 at Meola Reef. Principal coordinates ordination based on square-root transformed mean density of the 44 most abundant taxa at each location from 2007 to 2024. Vectors indicate taxa with correlation coefficient >0.4.



Large brown macroalgae

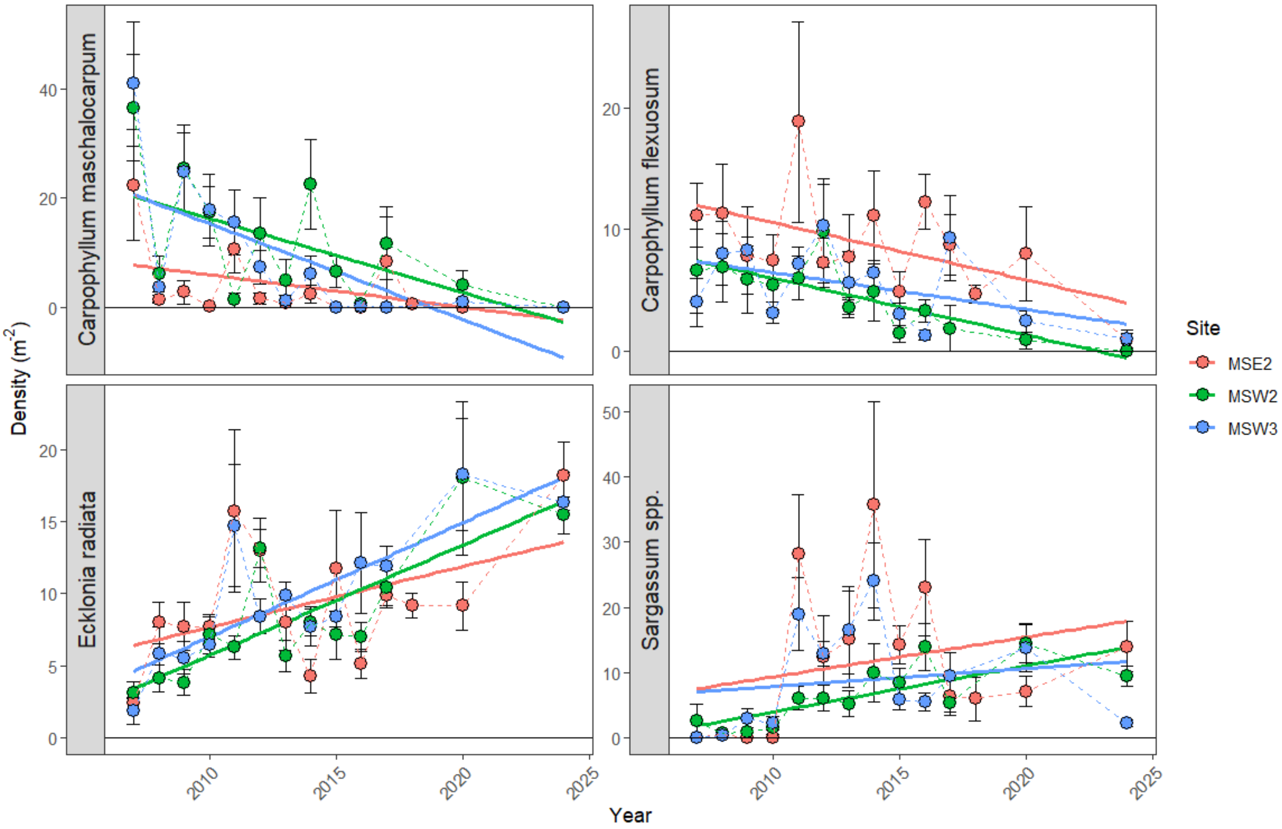
Total density of large brown algae has fluctuated at all Meola sites, but declined significantly at MSW3 (Figure 4A, Appendix 2a). The mean length of large brown algae declined at all three sites (Figure 4B), but overall canopy cover appears to have remained stable and increased at MSE2 (Figure 4C), with current covers at all sites ~70% or greater (Figure 4D). Estimated canopy cover is lower than observed canopy cover in 2020 and 2024 (Figure 4C), which is likely due to the prevalence of *E. radiata* which has higher canopy cover relative to density than *Carpophyllum* species.

Figure 4. Variation in large brown macroalgal density (A), mean length (B), overall estimated and observed canopy cover (C) and site-level observed canopy cover (D) at Meola Reef.



There has been a complete shift in the dominant forest-forming species at Meola over the time series with significant declines in *C. maschalocarpum* and *C. flexuosum* and a significant increase in *E. radiata* at all sites (Figure 5, Appendix 2b). *Sargassum* spp. has also generally increased, but this was only significant at MSW2. While *Sargassum* spp. can be numerically abundant, plants are primarily small (<25cm), and canopy cover is <10%. In contrast, *E. radiata* plants are typically 30-80cm total length and dominate the canopy, covering ~60-80%. In 2024, no *C. maschalocarpum* was recorded in quadrats and only a single large *C. flexuosum* plant (~150cm) was recorded.

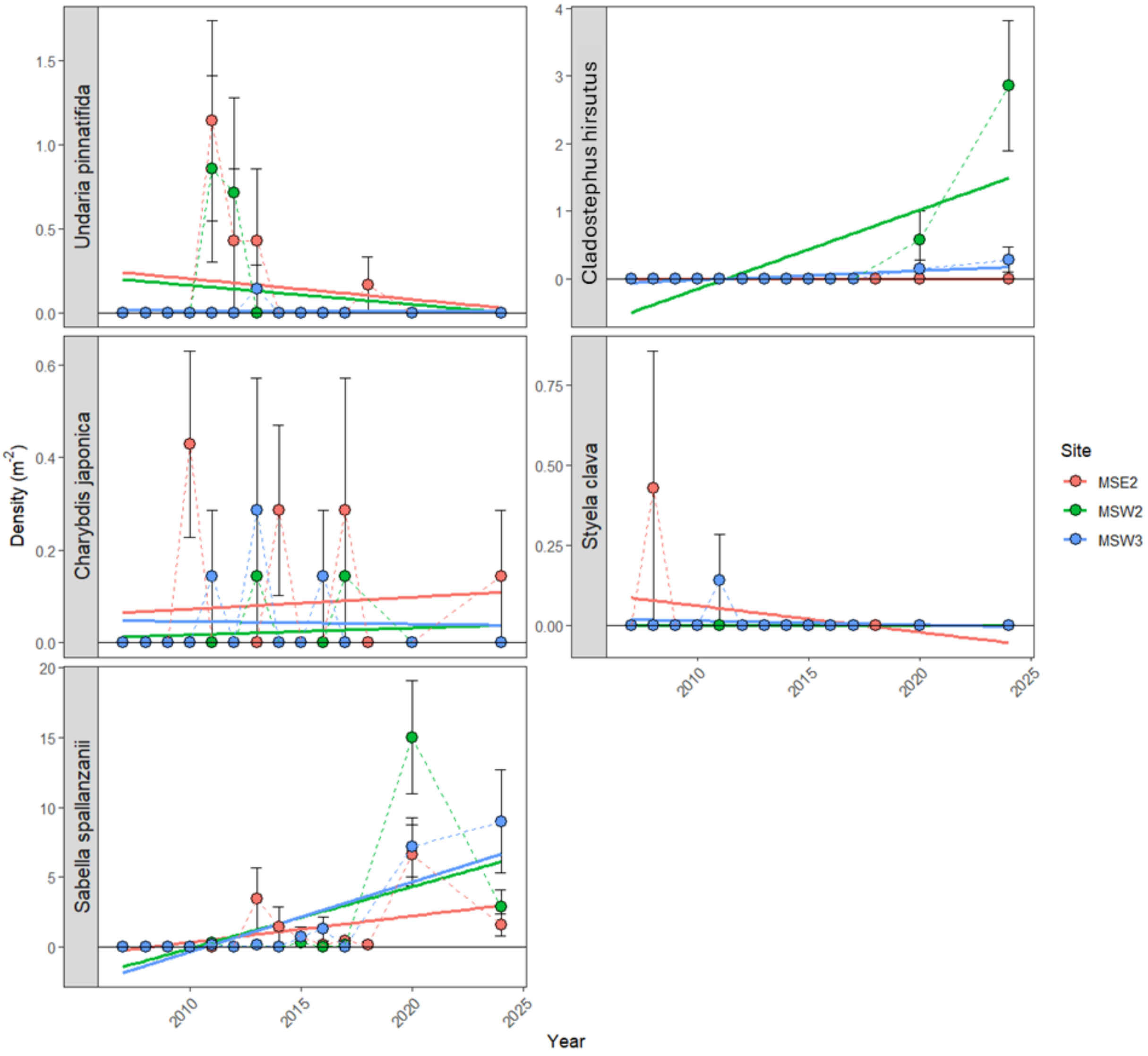
Figure 5. Site-level variation in the density of dominant large brown macroalgal species at Meola Reef from 2007 to 2024.



Density of invasive species

The invasive kelp *Undaria pinnatifida* has been recorded at MSW2 and MSE2 but is highly seasonally variable and generally does not occur during summer when surveys are typically carried out (Figure 6). The small brown algae *C. hirsutus* was first detected at MSW2 and MSW3 in 2020 and has increased in abundance in 2024. The invasive paddle crab *Charybdis japonica* has been recorded in low numbers throughout the time series and clubbed ascidian was recorded in 2008 and 2011 but not since. The large sabellid worm *Sabella spallanzanii* was first recorded at Meola in 2011 and underwent a large increase across all sites (Figure 6, Appendix 2c). It has not increased further between 2020 and 2024.

Figure 6. Site-level variation in the density of common invasive species at Meola Reef from 2007 to 2024.



Density of key mobile and sessile invertebrates

The catseye snail *L. smaragdus* was by far the most numerically dominant mobile invertebrate recorded at all Meola sites but has undergone almost complete declines with only two individuals recorded at MSE2 in 2024 (Figure 7A, Appendix 2d). Other gastropod species have generally been rare at Meola throughout the time series (<10 individuals recorded in total), with the exception of the butterfly chiton *Cryptoconchus porosus* (Figure 7A). However, this species has also declined at all sites and not been recorded in quadrats since 2016. *Patiriella regularis* is common at all sites and has remained stable (Appendix 2e). Both *E. chloroticus* and *C. muricata* were present in low densities but were not recorded in the 2020 and 2024 surveys (Figure 7A). Densities of both solitary ascidians and the orange golf ball sponge *Tethya burtoni* increased in the early part of the time series but have subsequently declined in recent years (Figure 7B). Only two *T. burtoni* were recorded in 2024.

Figure 7. Site-level variation in the density of dominant mobile (A) and sessile (B) invertebrates at Meola Reef from 2007 to 2024.

A

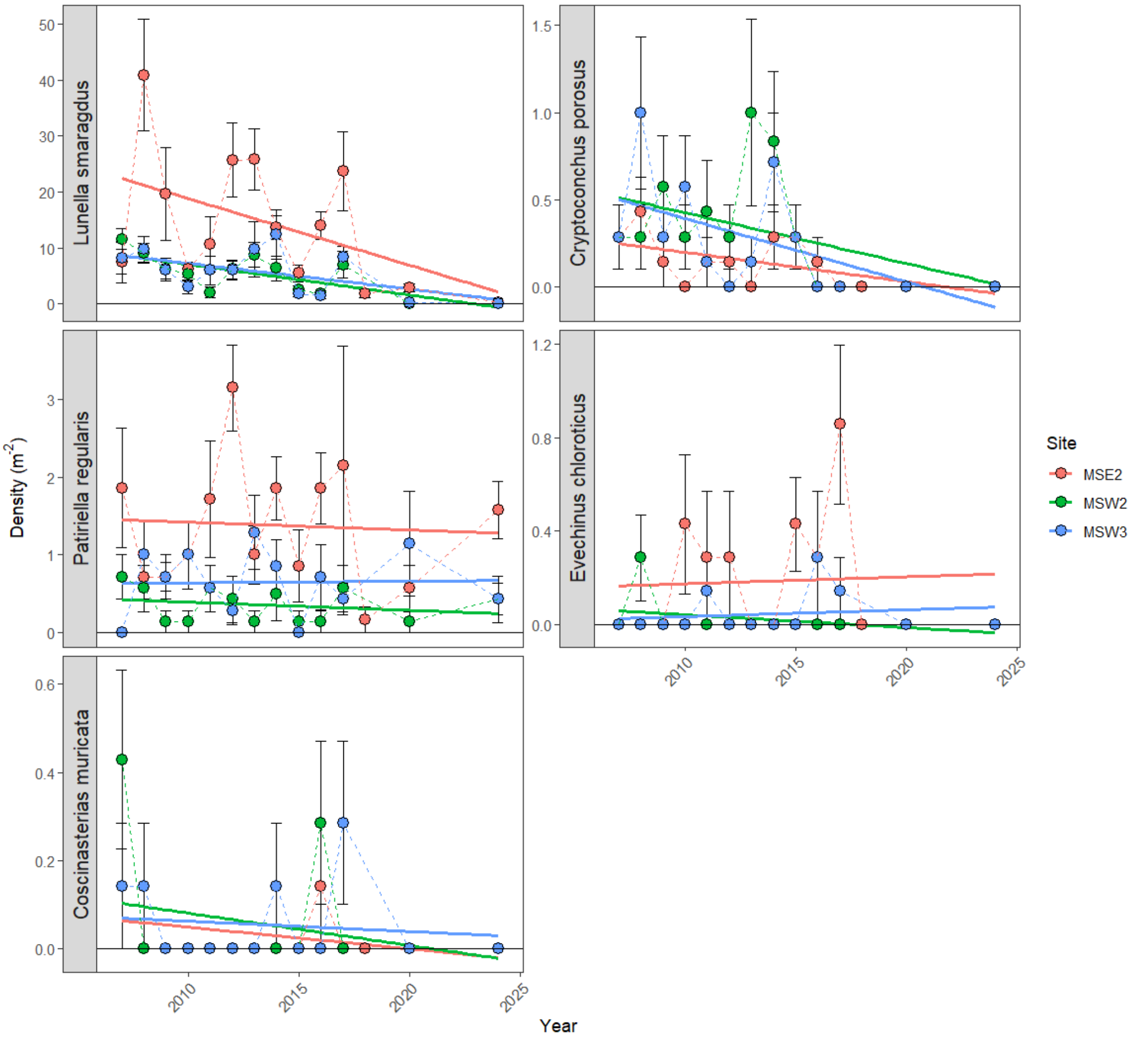
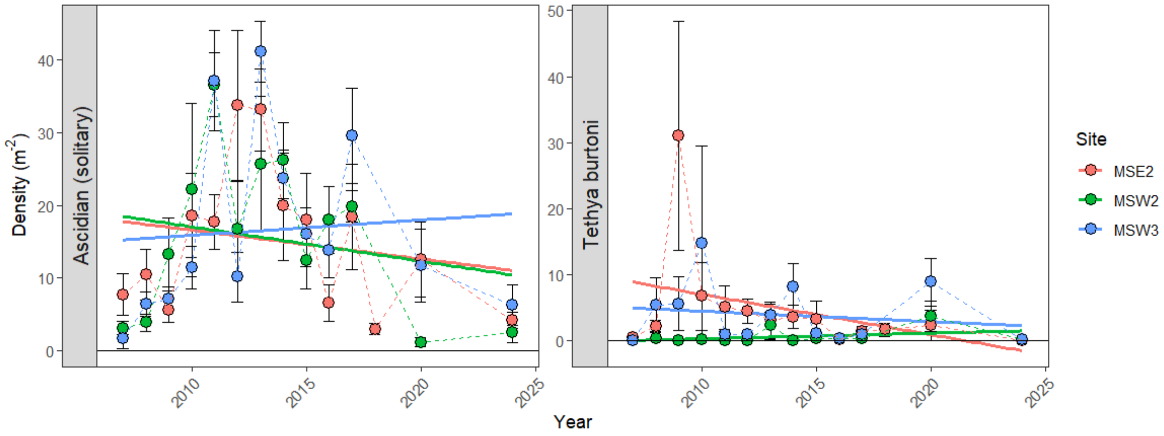


Figure 7. (Continued).

B



Dominant benthic covers

The reef at Meola sites is predominantly covered in sediment, crustose coralline algae (CCA), the red encrusting algae *Hildenbrandia* sp. and to a lesser extent *Corallina officinalis* (Figure 8A). None of these groupings show clear unidirectional trends over the monitoring periods, although the cover of CCA at MSW3 has declined (Appendix 2f). However, there has been an increase in filamentous algae, red turfing algae and oysters at all sites in 2024.

Sponges also cover a significant component of the reef but have declined across all sites, particularly between 2020 and 2024 (Figure 8B, Appendix 2g). Overall ascidian cover is largely influenced by solitary ascidians which increased and then subsequently declined. Hydroids and bryozoa only cover a small proportion of the reef and have varied over time with no clear directional trends.

Figure 8. Site-level variation in dominant substratum covers (A) and broad taxonomic groupings for sessile invertebrates (B) at Meola Reef from 2007 to 2024.

A

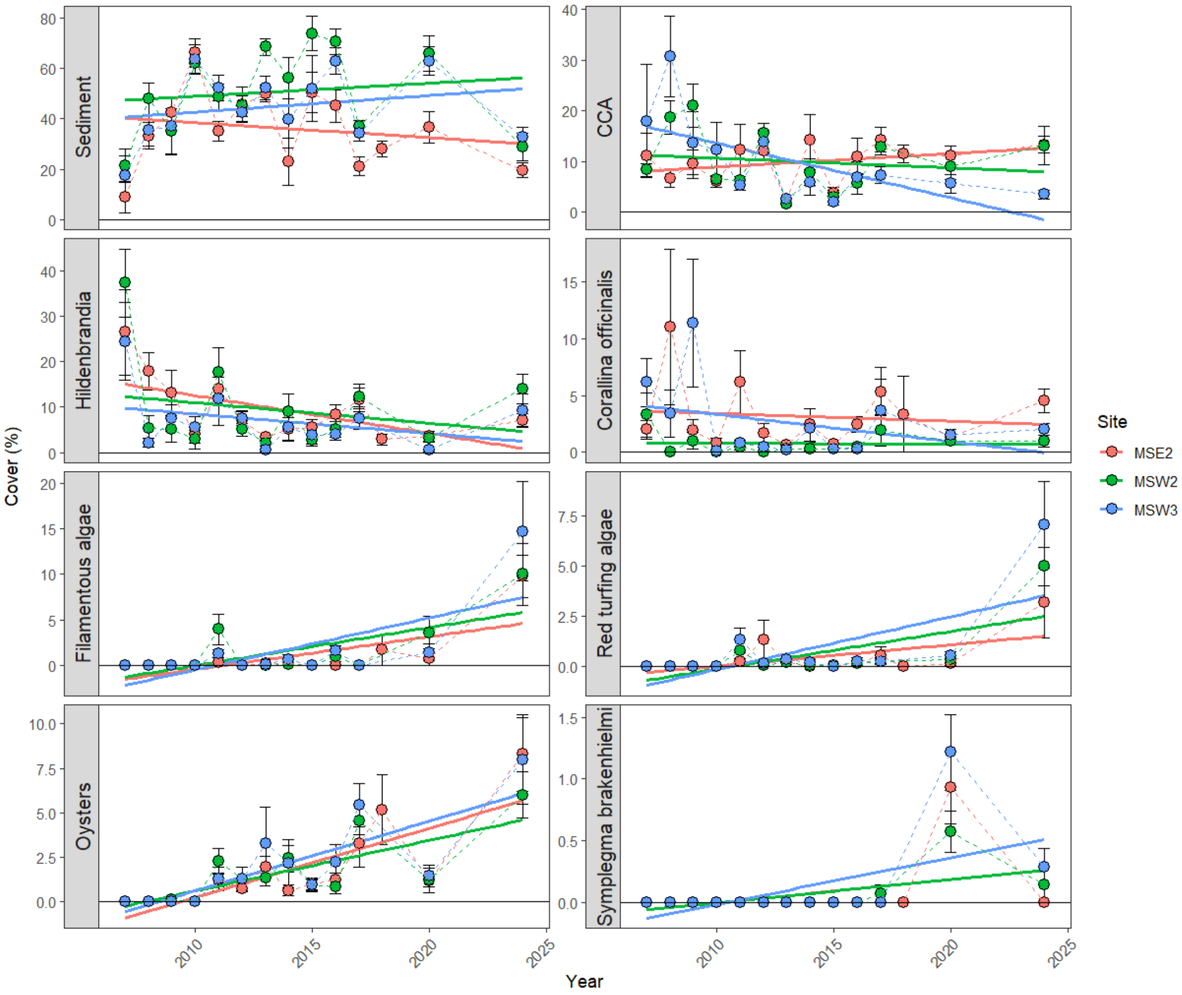
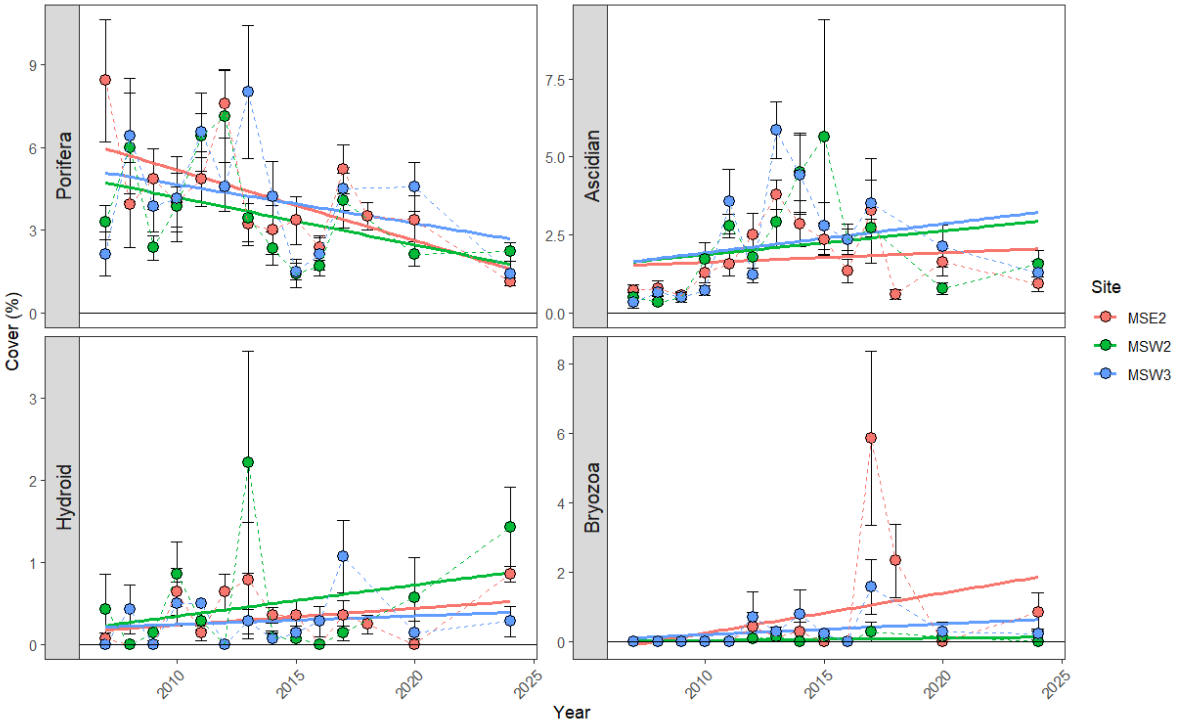


Figure 8. (Continued).

B

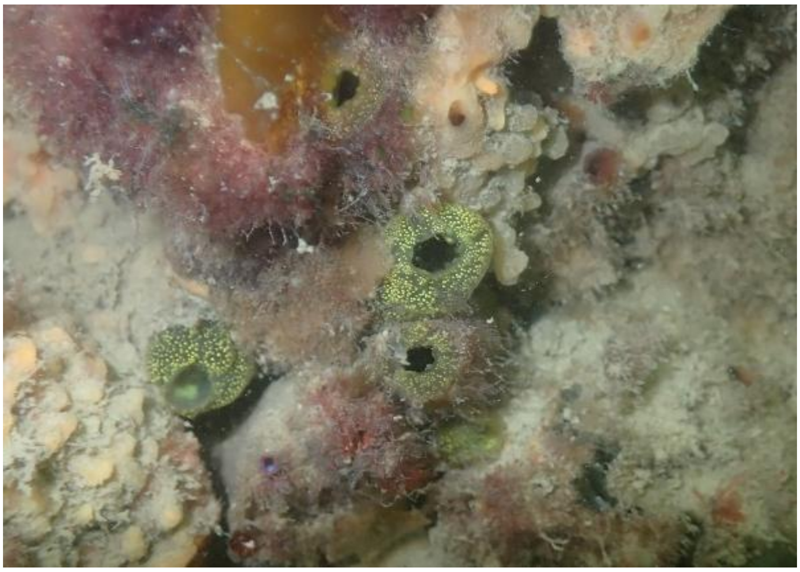


The invasive ascidian *Symplegma brakenhielmi* was first recorded at Meola in 2017 and is now present at low levels (≤~1% cover) at all sites (Figure 8A, Figure 9A). In 2024, the invasive ascidian *Ascidia latesiphonica* was also recorded at Meola (Figure 9B; identified by M. Page, NIWA). The grey encrusting bryozoan *Celleporaria nodulosa* is also present in small quantities at Meola (<1% cover).

Figure 9. Two colour morphs (yellow and orange) of the invasive ascidian *Symplegma brakenhielmi* growing amongst *E. radiata* holdfast and on *Sabella spallanzanii* at Meola Reef in 2020 (A) and the invasive ascidian *Ascidia latesiphonica* first recorded at Meola in 2024 (B; identified by Mike Page, NIWA). Photos: N. Shears)

A

B

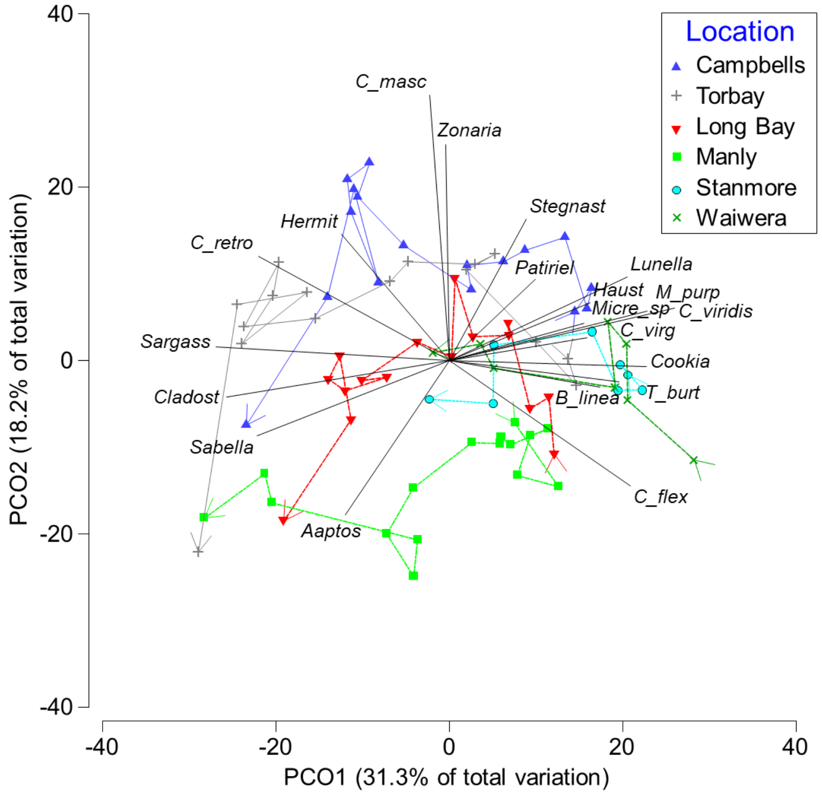


3.3 East Coast Bays-Whangaparāoa

Reef community composition (counts)

Overall reef community composition varied considerably among locations and over time (Figure 10). Locations at the top of the principal coordinates ordination (Campbells Bay and Torbay) tended to have higher abundance of *Carpophyllum maschalocarpum* and *Zonaria turneriana*, whereas Manly, Long Bay and Waiwera were initially characterised by higher abundances of *Carpophyllum flexuosum.* There was a general shift over time across all locations from right to left in the ordination, which generally correlated with a decline in *C. flexuosum* and a number of mobile invertebrate species (e.g. *Lunella smaragdus*, *Micrelenchus purpureus*, *Coelotrochus viridis*, *Cominella virgata* and *Cookia sulcata*), and an increase in *Cystophora retroflexa* and *Sargassum* spp., and the invasive brown seaweed *Cladostephus hirsutus* and Mediterranean fanworm *Sabella spallanazanii.*

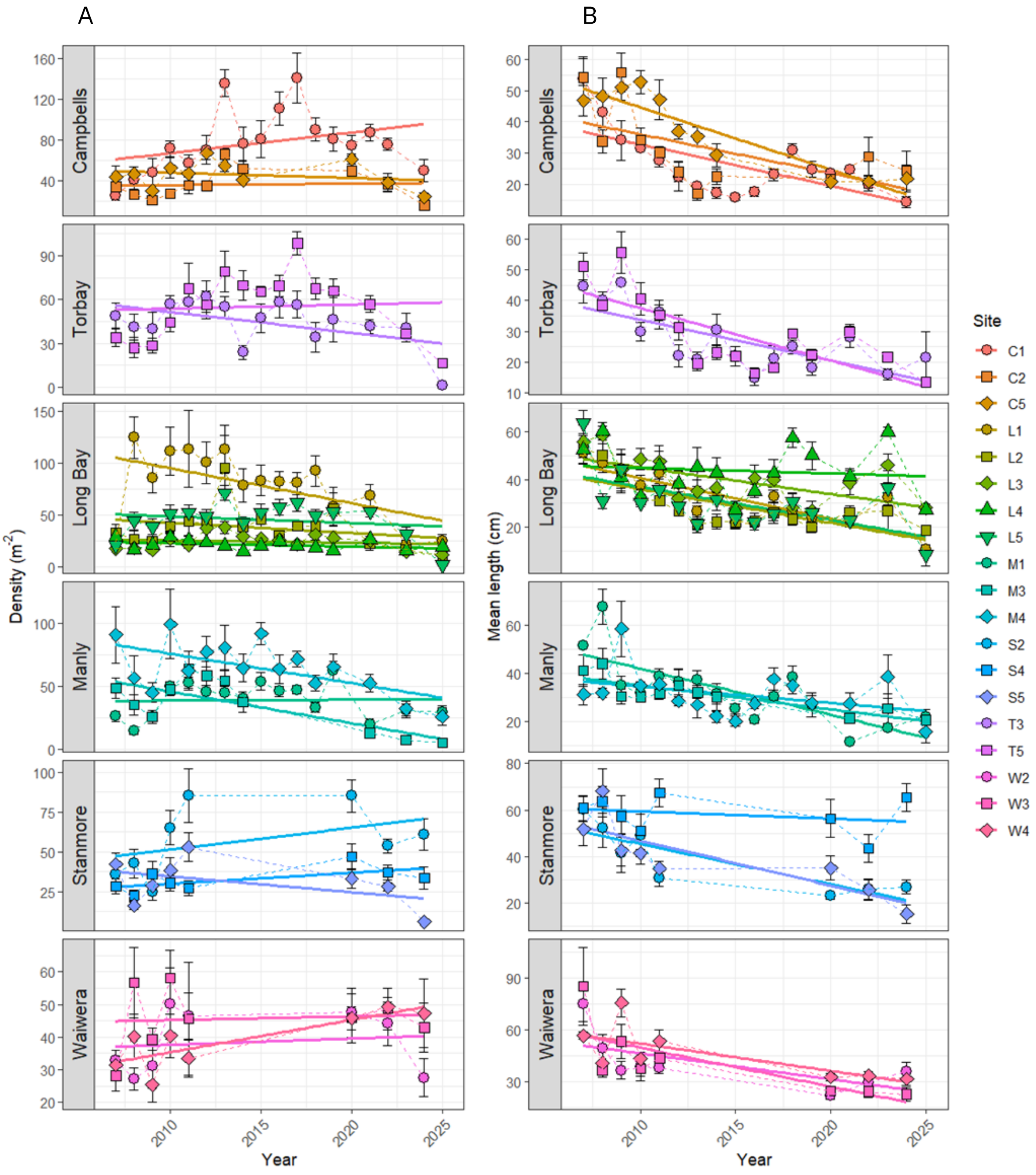
Figure 10. Location-level variation in reef community composition from 2007-2025 at East Coast Bays-Whangaparāoa locations. Principal coordinates ordination based on square-root transformed mean density of the 58 most abundant taxa at each location from 2007 to 2024/2025. Vectors indicate taxa with correlation coefficient >0.4. Taxa codes: Aaptos: *Aaptos aaptos*, Sabella: *Sabella spallanzanii*, Cladost: *Cladostephus hirsutus*, Sargass: *Sargassum* spp., C_retro: *Cystophora retroflexa*, Hermit: Hermit crab, C_masc: *Carpophyllum maschalocarpum*, Zonaria: *Zonaria turneriana*, Stegnast: *Stegnaster inflatus*, Patiriel: *Patiriella regularis*, Lunella: *Lunella smaragdus*, Haust: *Haustrum haustorium*, Micre_sp: *Micrelenchus* sp., M_purp: *Micrelenchus purpureus*, C_viridis: *Coelotrochus viridis*, C_virg: *Cominella virgata*, Cookia: *Cookia sulcata*, T_burt: *Tethya burtoni*, B_linea: *Buccinulum linea*, C_flex: *Carpophyllum flexuosum*.



Large brown macroalgae

Total density of large brown macroalgae has fluctuated with variable trends across sites and locations (Figure 11A, Appendix 2a). However there have been some marked declines over recent years at some sites, e.g. C2, C5, T3, L1, L4, M3, M4, S5, W2. There has been a significant decline in mean length of large brown macroalgae across most sites, with exception of L4 and S4 (Figure 11B, Appendix 2a). At Long Bay, there was a large decline in mean length of large brown macroalgae between 2023 and 2024 across all sites.

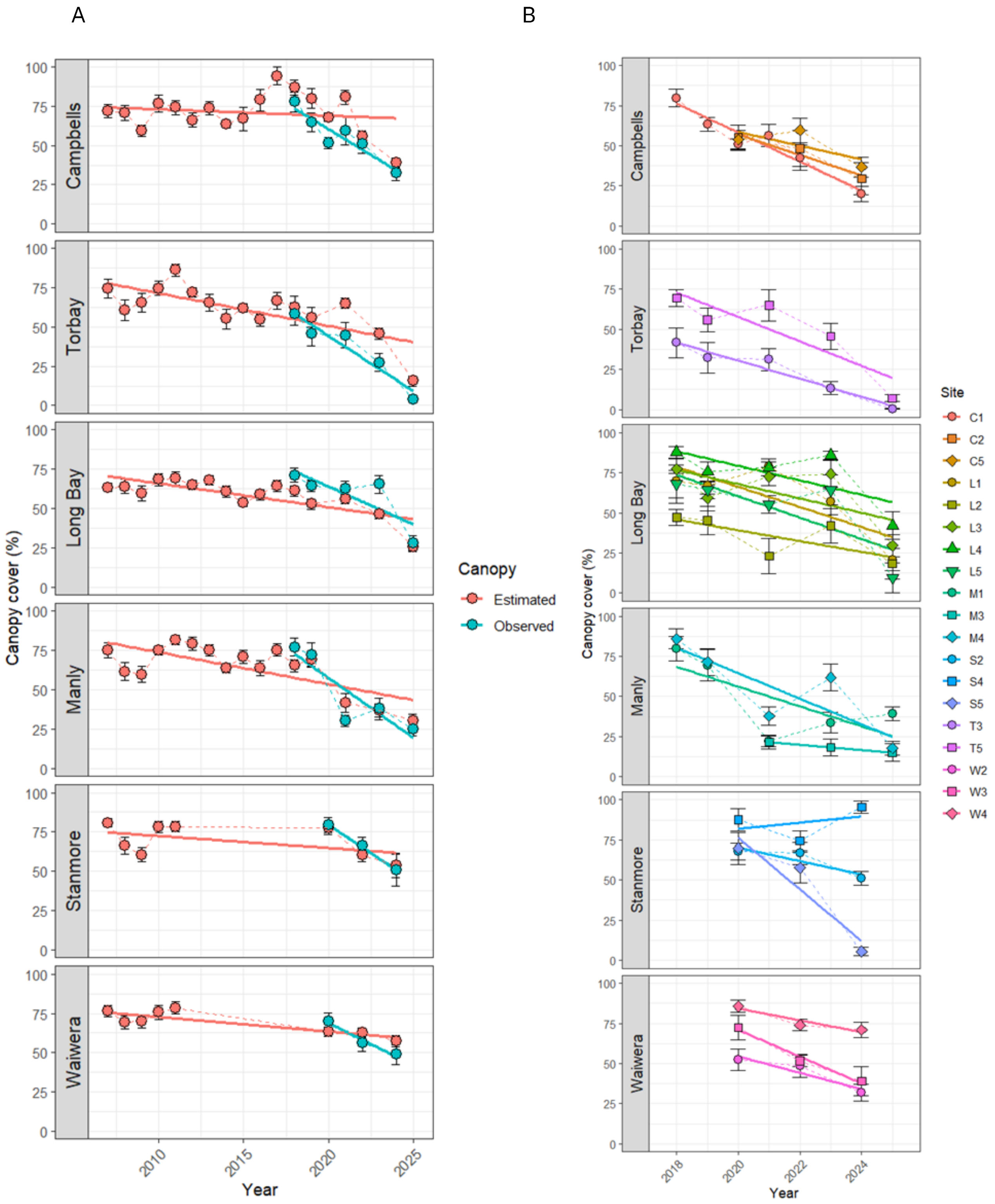
Figure 11. Site-level variation in density (A) and length (B) of large brown algae from 2007-2025 at East Coast Bays-Whangaparāoa locations.



The recent trends in density and size are reflected in changes in overall canopy cover of large brown macroalgae, which has been measured since 2018 (Figure 12A). Site-level estimates of long-term canopy cover (based on density and mean length – see Methods) suggest declines across most sites, except L3, L4, S4 and W4 (Appendix 2a). When averaged across sites within each location, overall canopy cover was relatively constant at most locations until about 2019/2020, with subsequent declines observed across all locations. The area-level trends in observed and estimated canopy cover post 2018 are generally consistent, although the observed decline at Torbay is greater than estimated, whereas observed canopy cover at Long Bay is higher than estimated, likely due to the predominance of *E. radiata*. Mean canopy cover at Long Bay was relatively stable from 2018-2023 but there was a large decline between 2023 and 2025 (Figure 12A), and this was evident across all sites (Figure 12B).

In 2018, observed canopy cover was ≥50% at all sites, but significant declines have occurred at all ECB-WGP sites, except M3 where canopy cover was already low and S4 where the canopy cover has remained high (Figure 12B, Appendix 2a). Across all sites, average canopy cover has declined at 5.8±1.6 % (±95% CI) per annum since 2018. Particularly large declines have occurred at a number of sites between the most recent surveys (2022-2024: Campbells Bay sites; 2023-2025: Long Bay sites, M4 and T5), with complete loss of canopy at S5 and L4. Canopy cover has only remained relatively high (~50% or greater) at L4, S4 and W4.

Figure 12. Location-level trends in estimated and observed large brown macroalgal canopy cover from 2007-2025 at East Coast Bays-Whangaparāoa locations (A) and site-level patterns in observed canopy cover since 2018 (B).



The most numerically dominant large brown macroalgal species was *Carpophyllum maschalocarpum,* which increased at a number of sites over the first half of the monitoring period

(Figure 13). However, this has been followed by large declines in recent years at C1, T5, L2 and L5. A dieoff of *C. maschalocarpum* was observed at T5 in 2020 (see Discussion). At C2, T3, L2, L4, M3 and M4 there has been a significant long-term decline over the time series in *C. maschalocarpum* (Appendix 2b).

Carpophyllum flexuosum was the second most abundant large brown macroalgal species but has also undergone long-term declines across many sites (Appendix 2b), particularly in recent years (e.g. C5, M1, M4, W2 and W3). Dying *C. flexuosum* plants with blackened stipes were observed at W2 in 2020 and heavily fouled remnants of *C. flexuosum* (stipes only) were present at S5 in 2024 (see Discussion). *C. flexuosum* is only common and remains an important component of the macroalgal canopy at S4 and W4, where it is interspersed with *Ecklonia radiata*. *Carpophyllum plumosum* was locally abundant at L1 and M4, but has declined significantly and was at very low densities in the most recent survey in 2024/2025 (Figure 12, Appendix 2b). It also occurred at low levels at other sites but was not recorded in the most recent surveys.

The kelp *Ecklonia radiata* exhibited variable trends across sites and locations (Figure 13). It is generally most abundant at Long Bay, Stanmore Bay and Waiwera sites, but has increased significantly at C5, L1, L4, L5, S2, S4 and W4 (Appendix 2b). There were notable declines in *E. radiata* between 2022 and 2024/2025 at a number of sites. *E. radiata* has dominated S5 since 2007 but there was an almost complete loss of *E. radiata* forest in 2024. Notably, *E. radiata* has remained dominant and stable at S4 which is located ~650m from S5. There was also a complete loss of *E. radiata* at L5 between 2023 and 2025, whereas *E. radiata* persisted at L4 in 2025 (Figure 13).

Both *Cystophora retroflexa* and *Sargassum* spp. have increased over time across a number of sites in all locations (Figure 13, Appendix 2b). *Sargassum* spp. is predominantly comprised of short plants (<30cm) of *Sargassum scabridum* and occasional *S. sinclairii*, which can grow to greater lengths (up to 60cm).

The invasive kelp *Undaria pinnatifida* was also recorded at L5 in 2025, which provides the first record of this species in the monitoring data at ECB-WGP sites. Notably, this occurrence coincided with the loss of native large brown macroalgal canopy at this site between 2023 and 2025 (Figure 12B).

Figure 13. Site-level variation in density of large brown macroalgal species from 2007-2025 at East Coast Bays-Whangaparāoa locations.

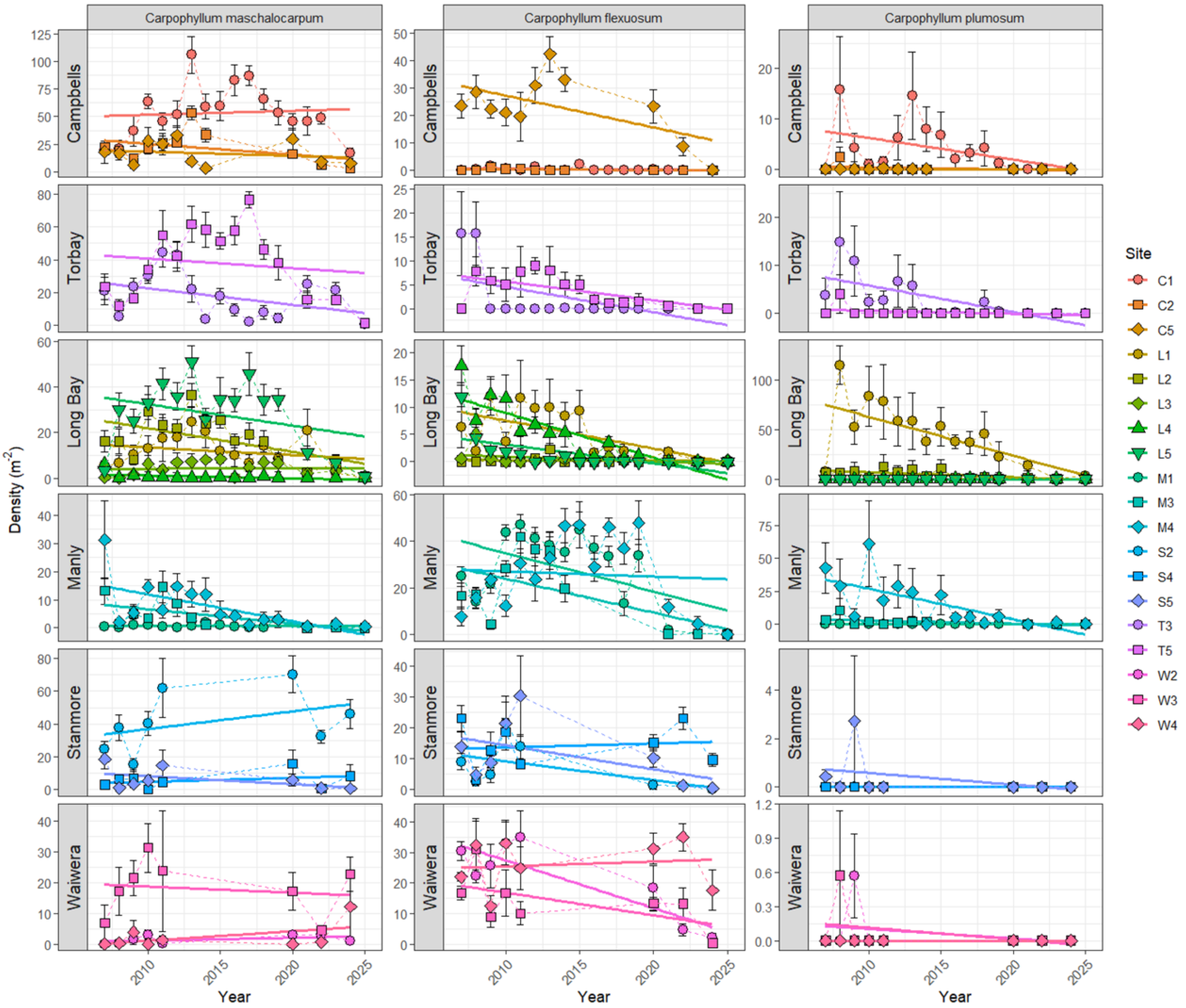
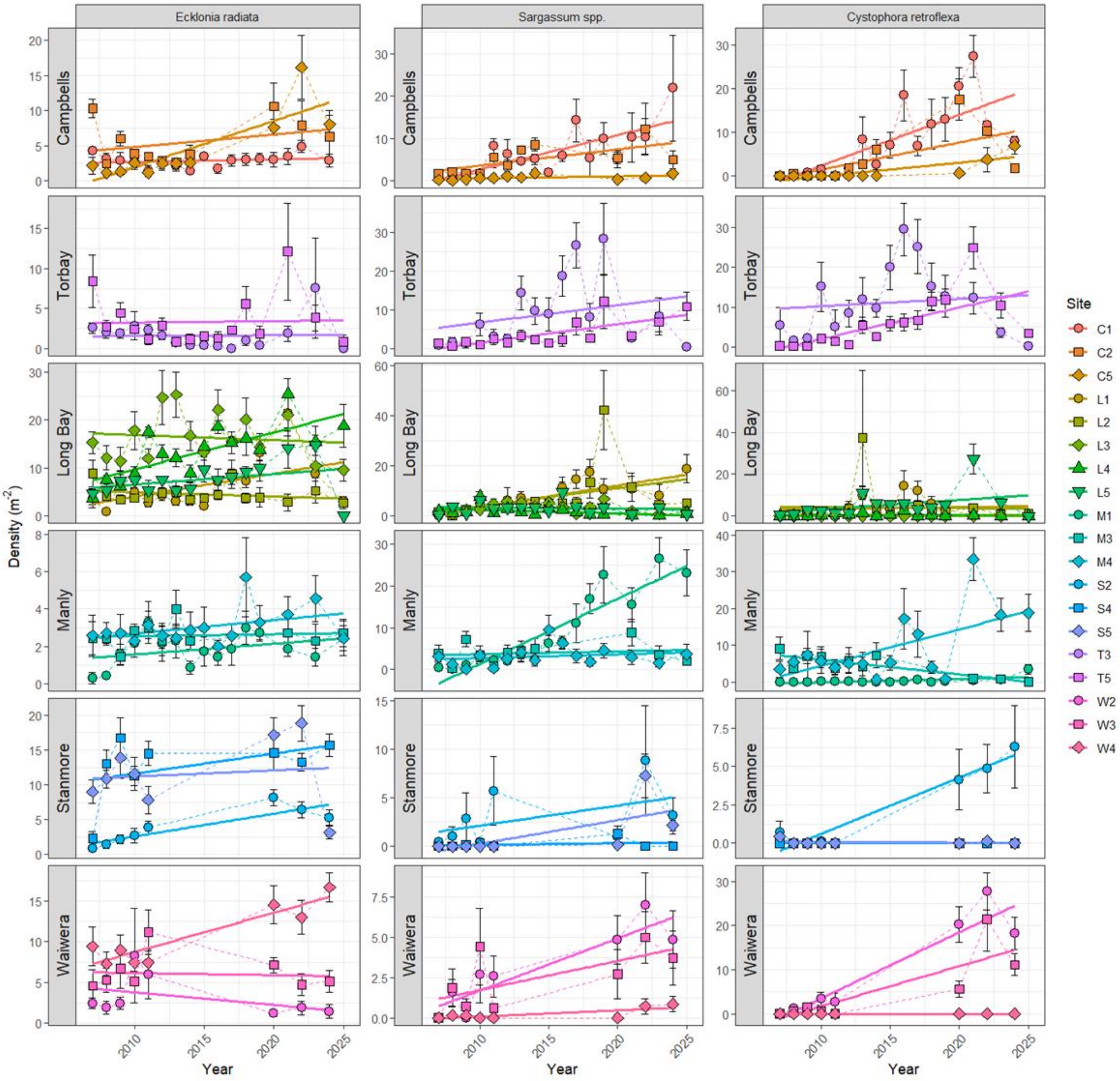


Figure 13. (Continued)

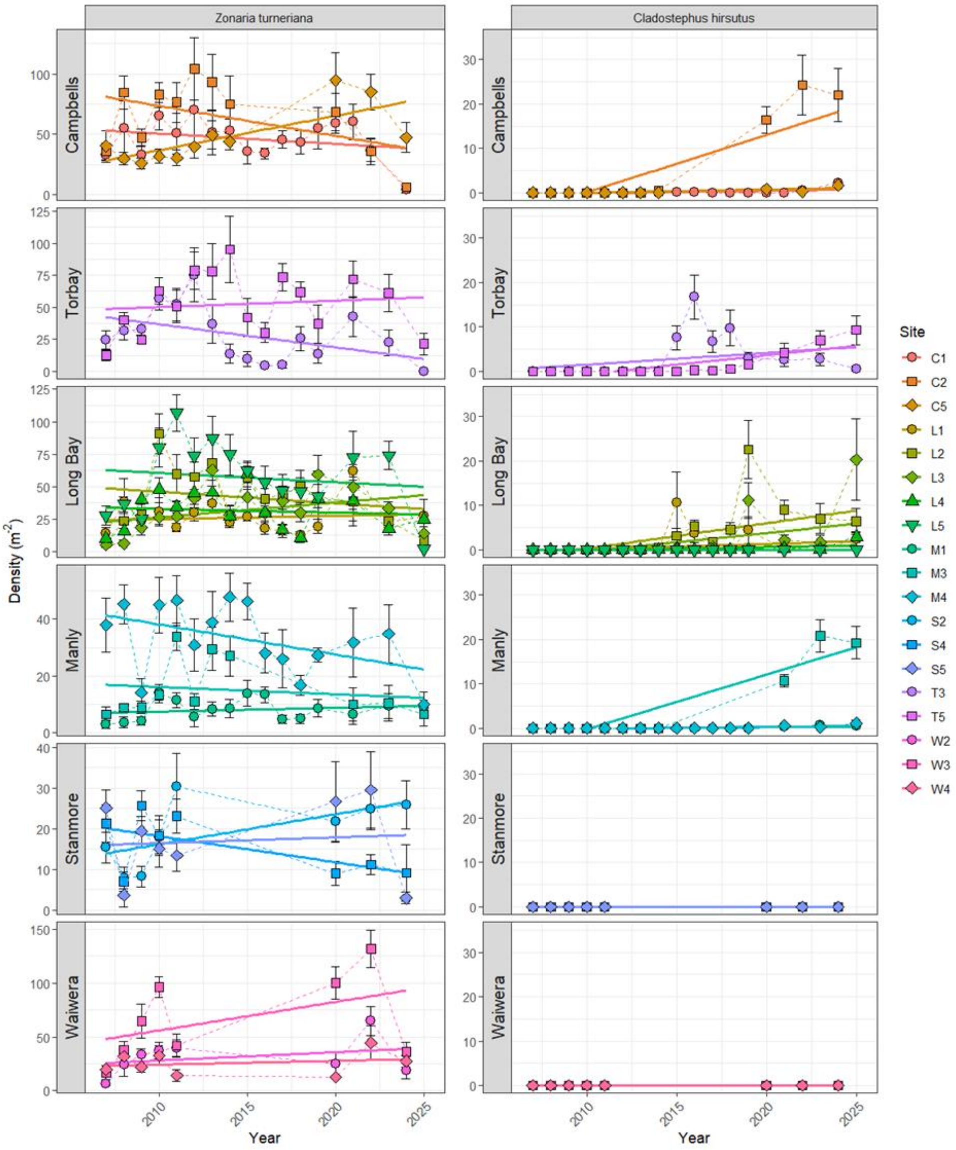


Subcanopy macroalgae

The small brown seaweed *Zonaria turneriana* is a highly abundant subcanopy species across many sites (Figure 14). Densities increased across a number of sites over the first 5-8 years but then underwent rapid declines at a number of sites between 2022 and 2024/2025, e.g. all Campbells Bay, Torbay, Long Bay and Waiwera sites, and S5. Die-off of *Z. turneriana* was observed at W3 and S5 in 2024.

The invasive brown seaweed *Cladostephus hirsutus* was first recorded at Long Bay in 2015 and has now become a dominant and common component of the macroalgal assemblage at some sites, in particular C2, L2, L3 and M3 (Figure 14). Of note, the invasive green seaweed *Codium fragile* also tended to be more common in 2024/2025 at a number of sites (data not shown).

Figure 14. Site-level variation in density of subcanopy macroalgal species *Zonaria turneriana* and *Cladostephus hirsutus* from 2007-2025 at East Coast Bays-Whangaparāoa locations.



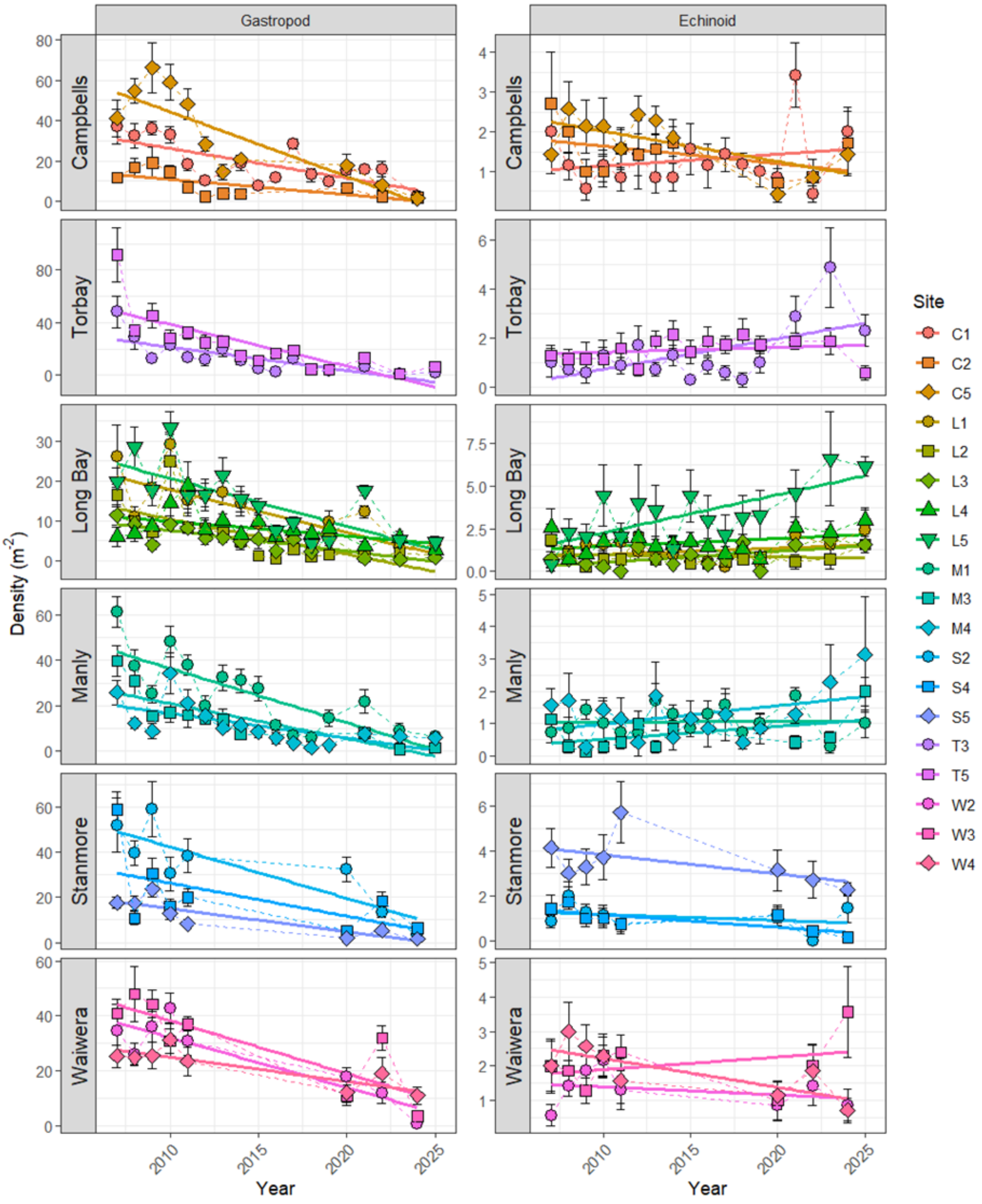
Density of key mobile and sessile invertebrates

Gastropods and echinoderms

Total gastropod numbers have declined dramatically across all sites whereas the total number of echinoderms has generally been stable or increased at some sites (Figure 15, Appendix 2e and f). Among the gastropod species, *Lunella smaragdus* was numerically dominant but has undergone significant declines across all sites (Figure A3a). Other common herbivorous gastropod species *Coelotrochus viridis, Cookia sulcata, Cantharidus purpureus,* and the predatory whelks *Buccinulum linea* and *Cominella virgata,* have also generally declined (Figure A3a). The only gastropod to remain stable or increase at some sites is the suspension feeding turret shell *Maoricolpus roseus* (Figure A3a, Appendix 2e).

The cushion star *Patiriella regularis* is the most abundant echinoderm and has showed variable trends among sites with some stable or increasing and others declining (Figure A3b). Kina *Evechinus chloroticus* mostly occurs at low numbers, but has increased significantly at C1, C5, T3, T5, L2, L5, M3, M4 and Waiwera sites (Appendix 2e). Some of these sites appear to have had recent recruitment with a number of urchins <50 mm test diameter recorded in 2021. The 11-arm starfish *Coscinasterias muricata* has remained relatively stable, except for an increase at L1 and L3. The ambush starfish *Stegnaster inflatus* is relatively rare, but it has declined at C5 and W4 where it was relatively common.

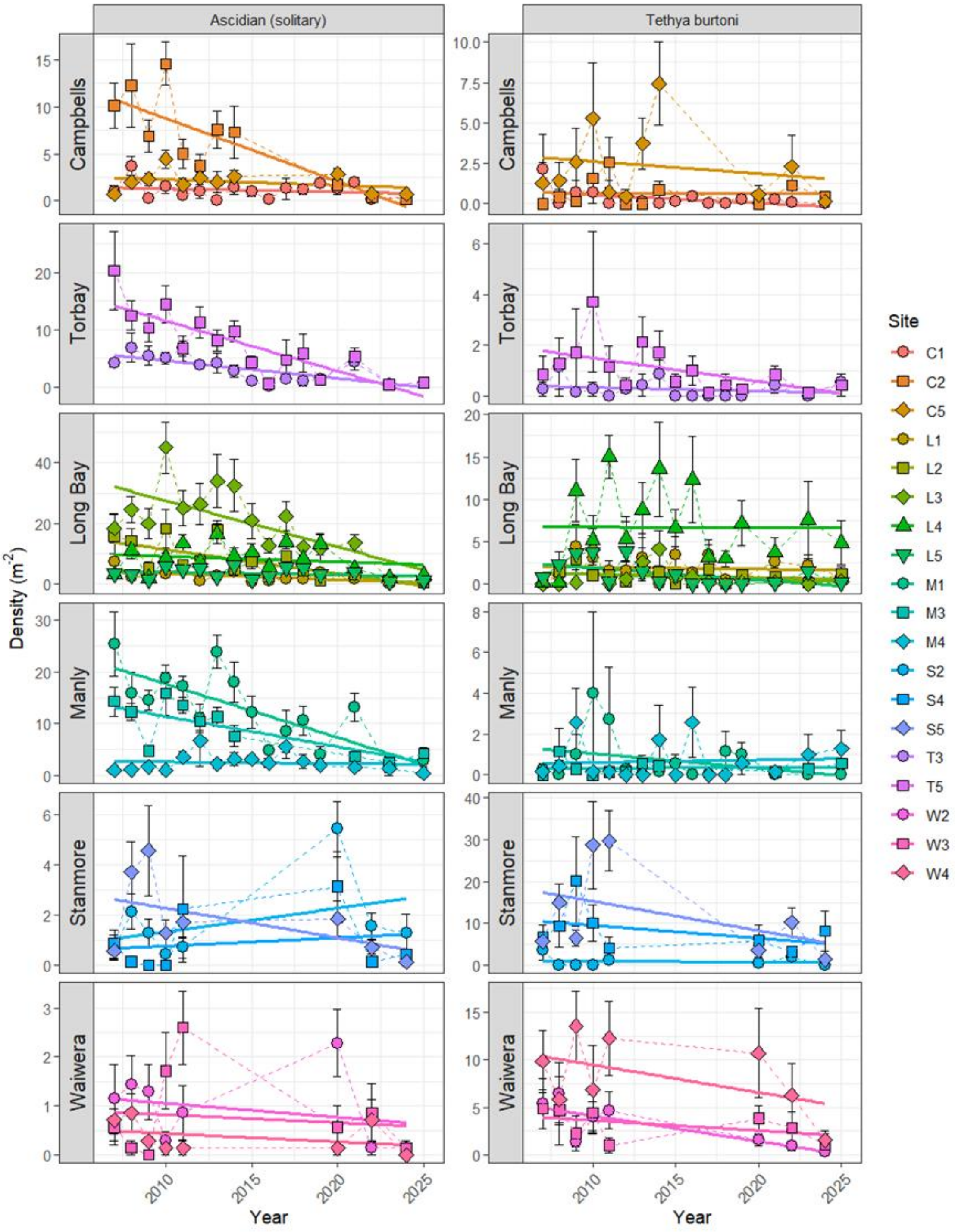
Figure 15. Site-level variation in density of gastropods and echinoderms from 2007-2025 at East Coast Bays-Whangaparāoa locations. See Appendix 3a and 3b for plots of key species.



Key sessile invertebrates

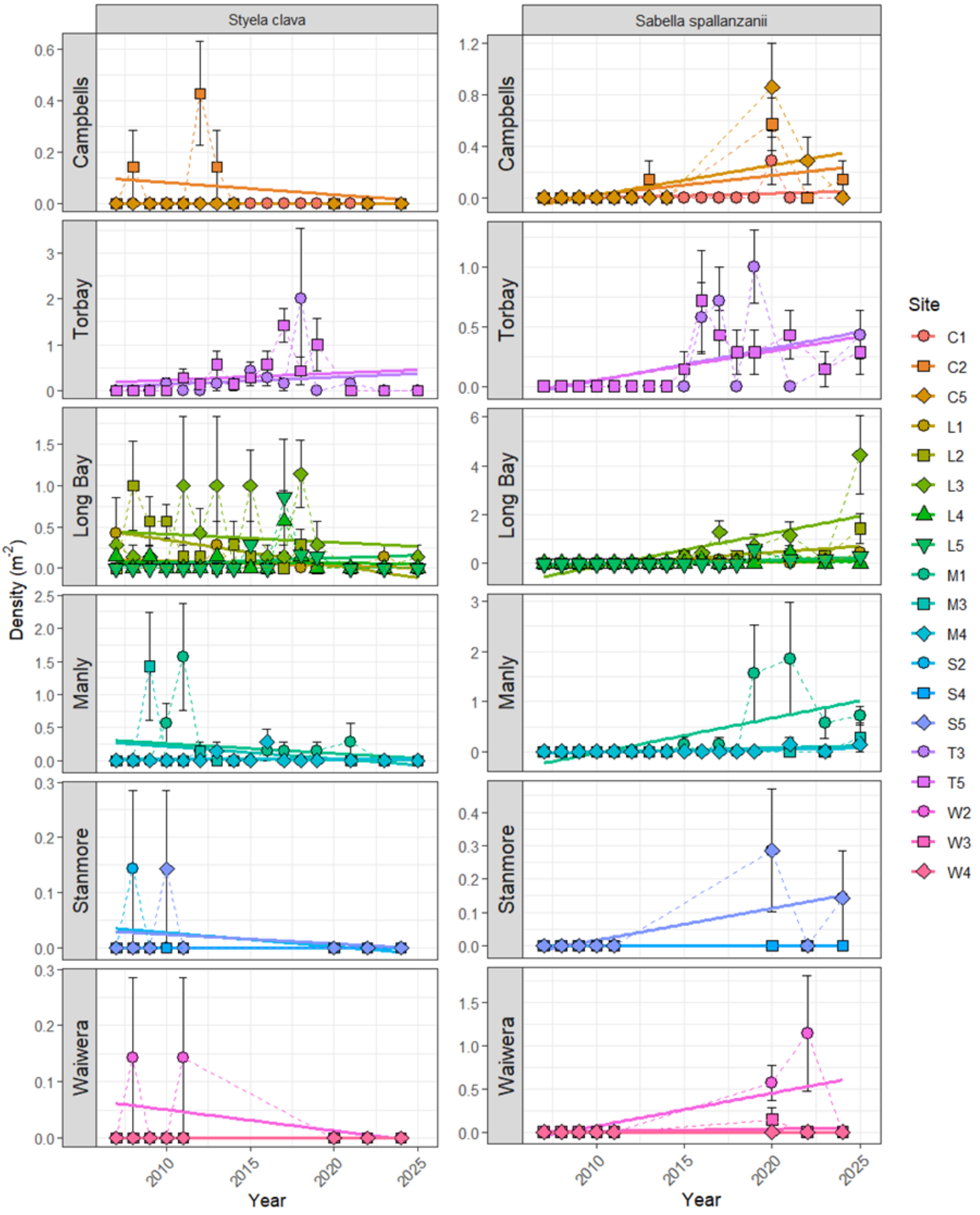
Solitary ascidians and the orange golf ball sponge *Tethya burtoni* are common and counted in quadrats (Figure 16). Solitary ascidians have declined to varying extents across most sites but were at particularly low numbers in the most recent sampling (Appendix 2c). *Tethya burtoni* have also declined at some sites in recent years, e.g. L5, S5 and all Waiwera sites.

Figure 16. Site-level variation in density of gastropods and echinoderms from 2007-2025 at East Coast Bays-Whangaparāoa locations.



The invasive clubbed ascidian *Styela clava* and Mediterranean fanworm *Sabella spallanzanii* have also been counted in quadrats (Figure 17). *Styela clava* established across all locations over the monitoring period, but densities have declined in recent years with only 1 individual recorded at L1 in 2023 and at L3 in 2025. *Sabella spallanzanii* has also established across all locations, but densities have fluctuated across recent surveys (2020-2025).

Figure 17. Site-level variation in density of invasive invertebrates *Styela clava* and *Sabella spallanzanii* from 2007-2025 at East Coast Bays-Whangaparāoa locations.



Dominant benthic covers

Crustose coralline algae (CCA) was the dominant cover at most sites at the start of the monitoring period, but has declined significantly at all sites except L1, L4, and S4, which had a marginally significant decline (Figure 18A, Appendix 2f). This trend is mirrored in a significant increase in sediment cover across all sites except L1, L4, and S4. The cover of coralline turf *Corallina officinalis*, red turfing algae and filamentous algae has also increased at most sites (Figure 18B). There was a particularly large increase in filamentous algae across many sites in the most recent surveys. Other changes of note in cover of macroalgal species include declines in the green turfing algae *Cladophora herpestica* across all sites where it was historically common (Figure A3c).

Sponges are typically the dominant sessile invertebrates across all locations, and overall cover has generally remained stable (Figure 18C, Appendix 2g). Total ascidian cover has generally been variable among sites and over time and there are no clear trends. The invasive ascidian *Symplegma brakenhielmi* has established at sites in all locations, albeit at low covers (<~1%), since 2020 (Figure A3c). The invasive parchment worm *Chaetopterus* sp. also occurs at sites across all locations but has been variable over time (Figure A3c). The invasive ascidian *Ascidia latesiphonica* was first recorded in 2024 at C5, but was commonly found at most Torbay, Long Bay and Manly sites in 2025.

Figure 18. Site-level variation in crustose coralline algae (CCA) and sediment (A), *Corallina officinalis*, red turfing algae and filamentous algae (B) and sponges and ascidians (C) at East Coast Bays-Whangaparāoa locations from 2007 to 2025.

A

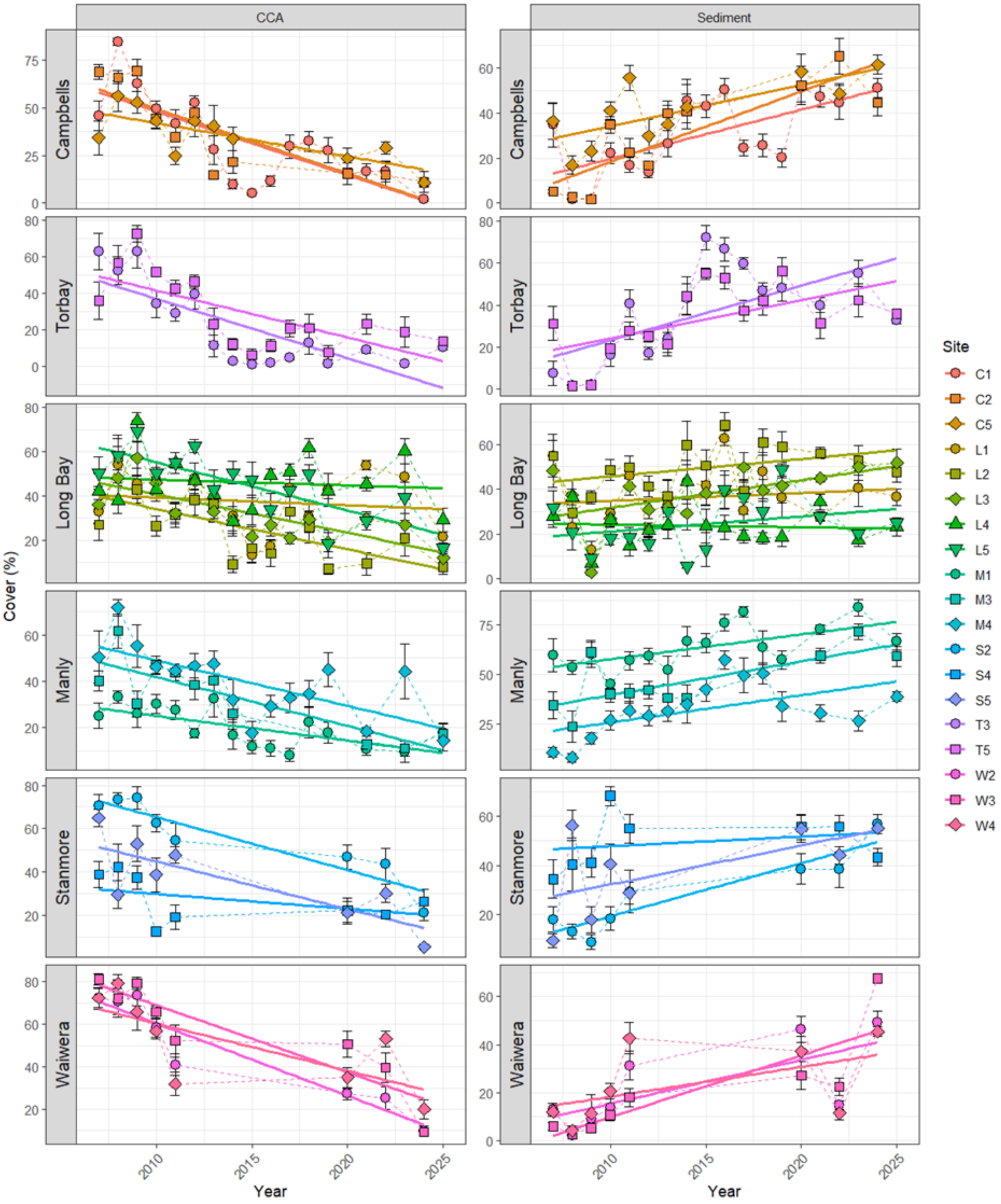


Figure 18. (Continued).

B

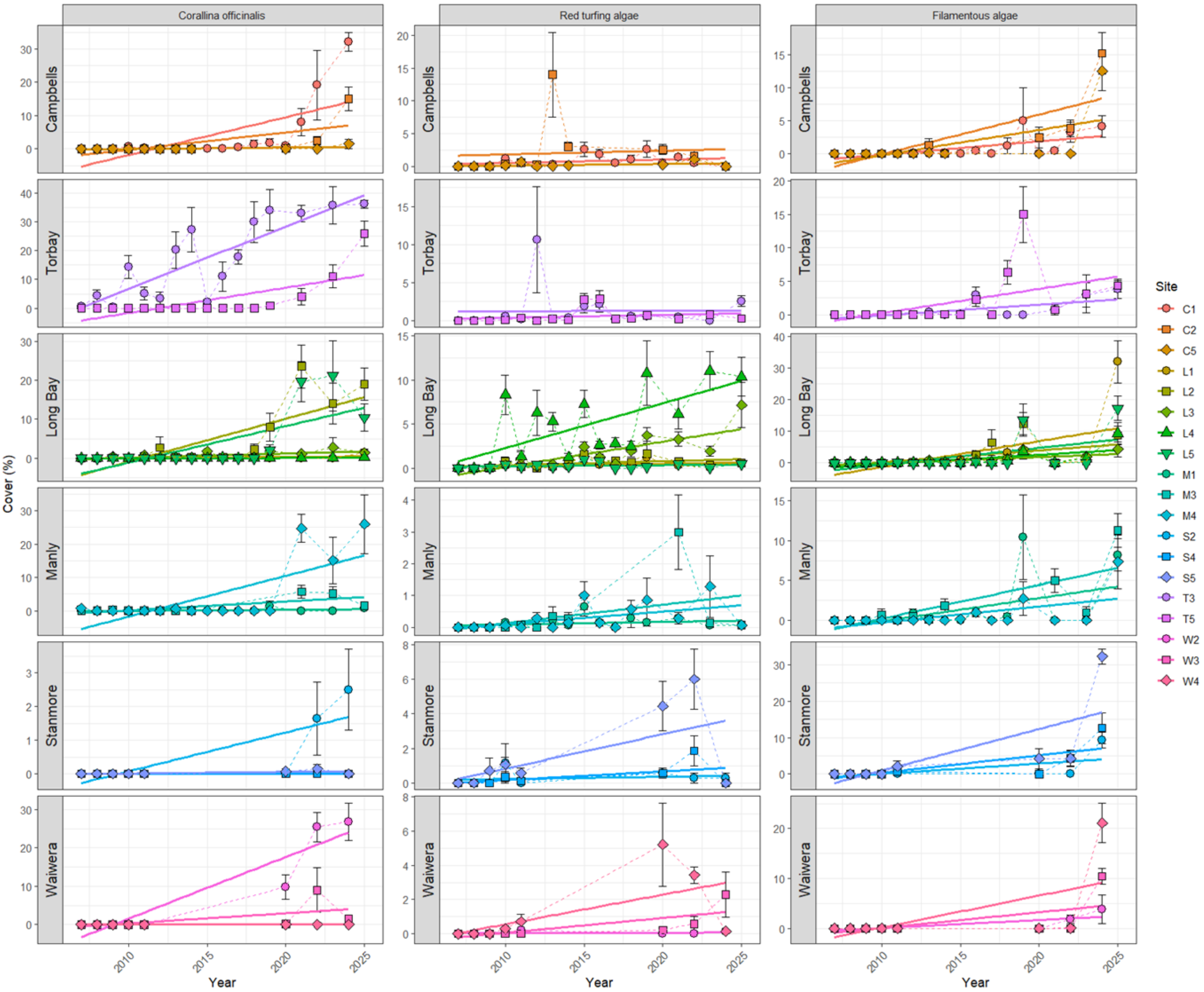
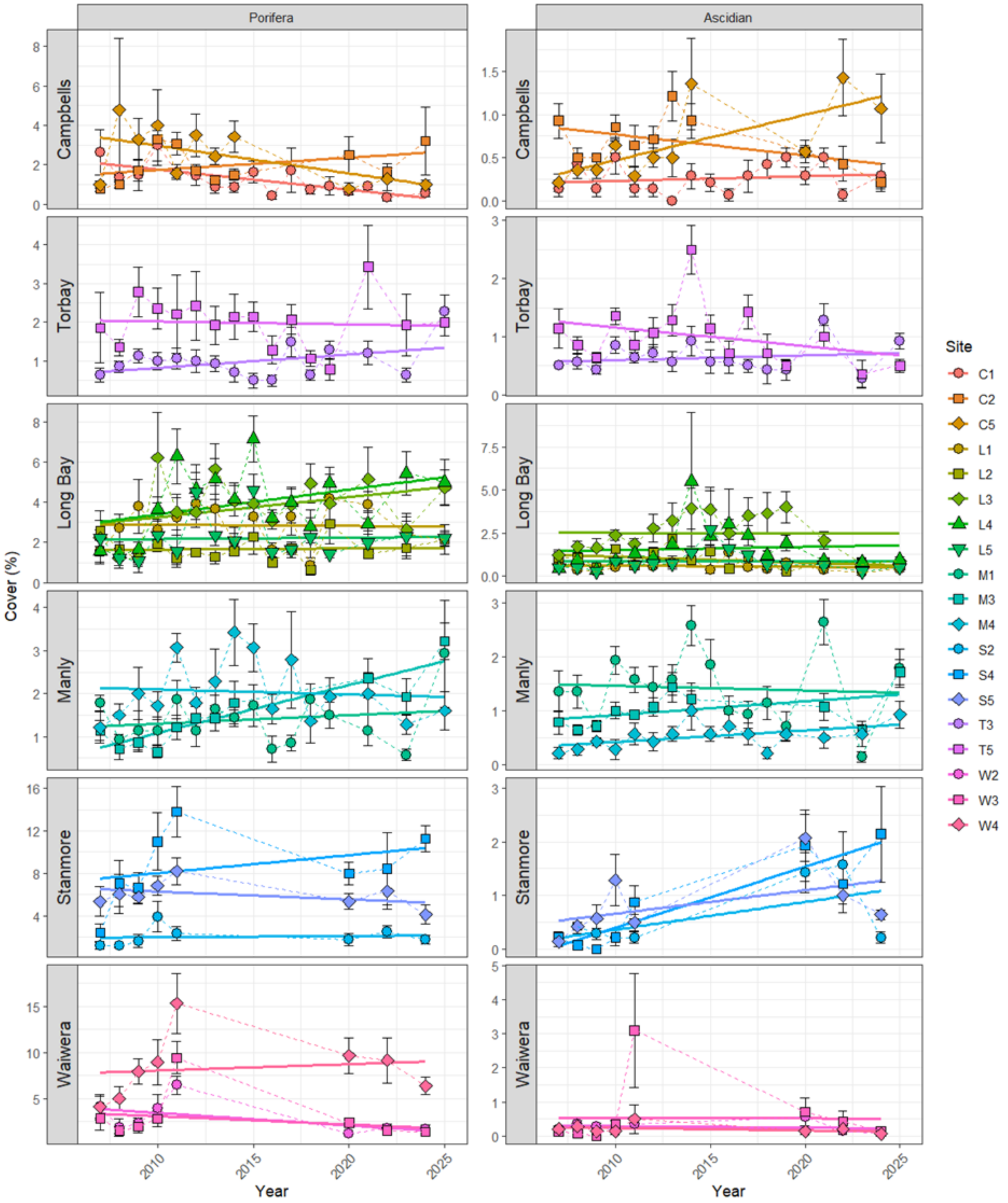


Figure 18. (Continued).

C

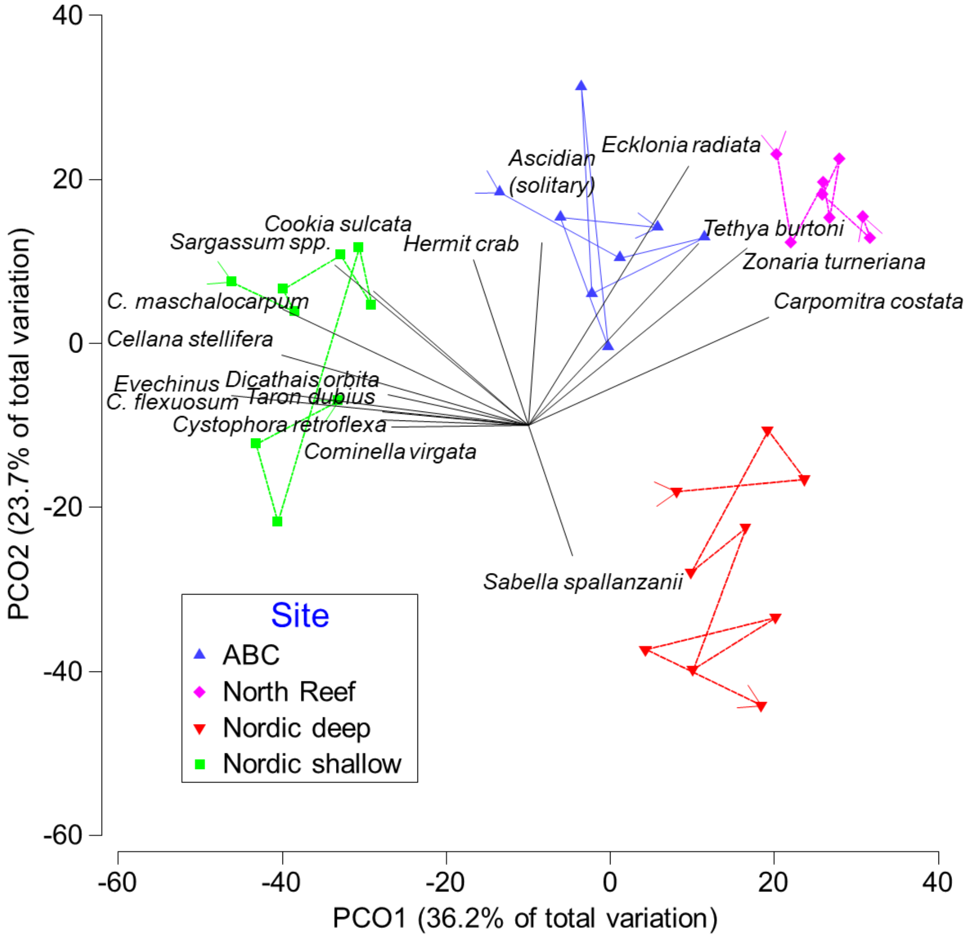


3.4 Leigh

Reef community composition (counts)

Overall reef community composition was highly spatially variable among the four sites at Leigh, and this has persisted over the monitoring period (Figure 19). Nordic Shallow was initially an urchin barren and had a number of species associated with the barrens including seaweeds *Sargassum sinclairii, Carpophyllum maschalocarpum,* and *C. flexuosum,* and mobile invertebrates *Cook sulcata, Cellana stellifera, Dicathais orbita, Cominella virgata,* and hermit crabs. The other sites were typically *Ecklonia*-dominated, with higher abundances of understory species such as solitary ascidians at ABC and the small brown seaweeds *Zonaria turneriana* and *Carpomitra costata* at North Reef. Nordic deep had few characteristic species except for *Sabella spallanzanii,* which was first recorded in 2024. Other invasive species were rare at the Leigh sites, except for the parchment worm *Chaetopterus* sp. which occurs in low abundance all sites (<2% cover). The grey encrusting bryozoan *Celleporaria nodulosa* is also present in small quantities at Nordic Shallow (<1% cover).

Figure 19. Site-level variation in reef community composition from 2013-2024 at Leigh. Principal coordinates ordination based on square-root transformed mean density of the 30 most abundant taxa. Vectors indicate taxa with correlation coefficient >0.4.



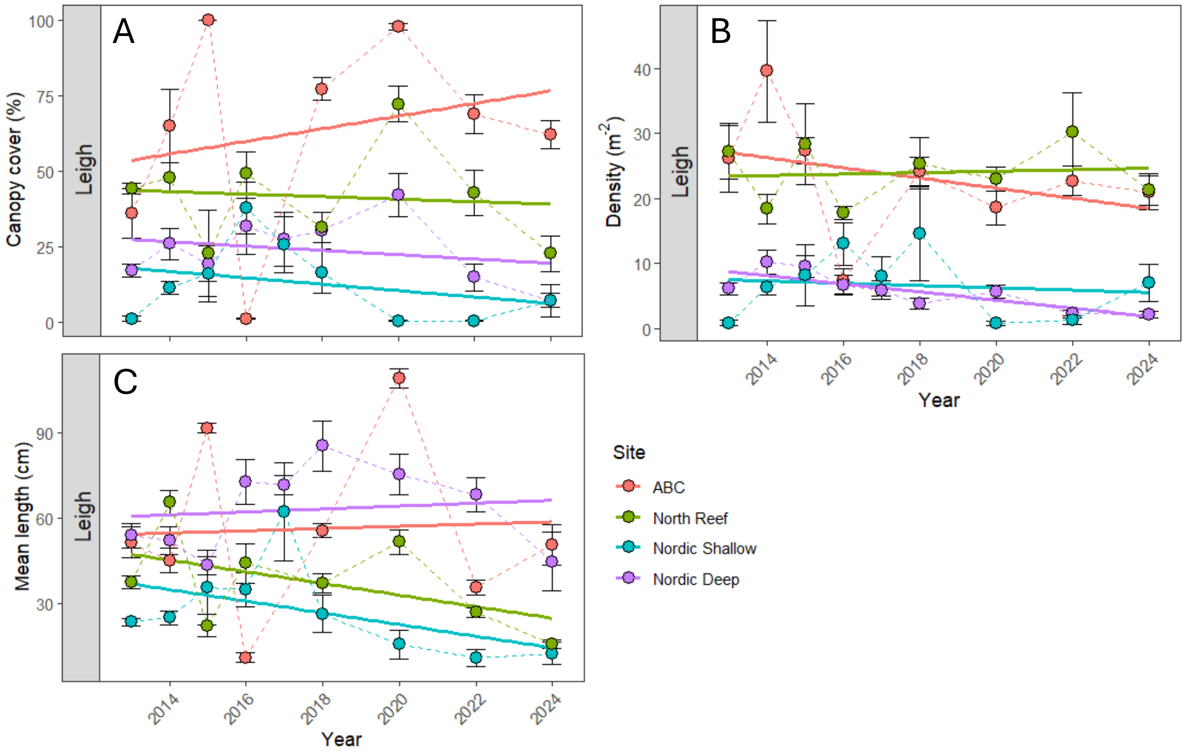
Large brown macroalgae

The kelp *Ecklonia radiata* is by far the dominant large brown macroalgal species at all the Leigh sites. There were no clear long-term trends in *E. radiata* across the Leigh sites, except there was a long-term decline in density at Nordic Deep (Figure 20, Appendix 2a). *E. radiata* typically has the highest canopy cover at ABC, but this site has exhibited large fluctuations in canopy cover associated with dieback events in 2013, 2015, and 2021. These events are typically associated with complete loss of canopy (Figure 20A), low densities (Figure 20B) and reduced mean size of *E. radiata* due to dominance of new recruits (Figure 20C). This is clearly evident in the 2016 sampling. North Reef (15 m depth) typically had lower canopy cover and shorter kelp than ABC, and the fluctuations associated with dieback were less evident. Dieback was evident in the 2015 and 2024 surveys with reduced canopy cover, a smaller mean size, and also high numbers of dead holdfasts (data not shown).

At Nordic Shallow *E. radiata* increased over the first four years (Figure 20A and B), forming patchy areas of kelp forest by 2016 and 2017. However, *E. radiata* subsequently declined to near-zero at this site but there has been a small increase in 2024.

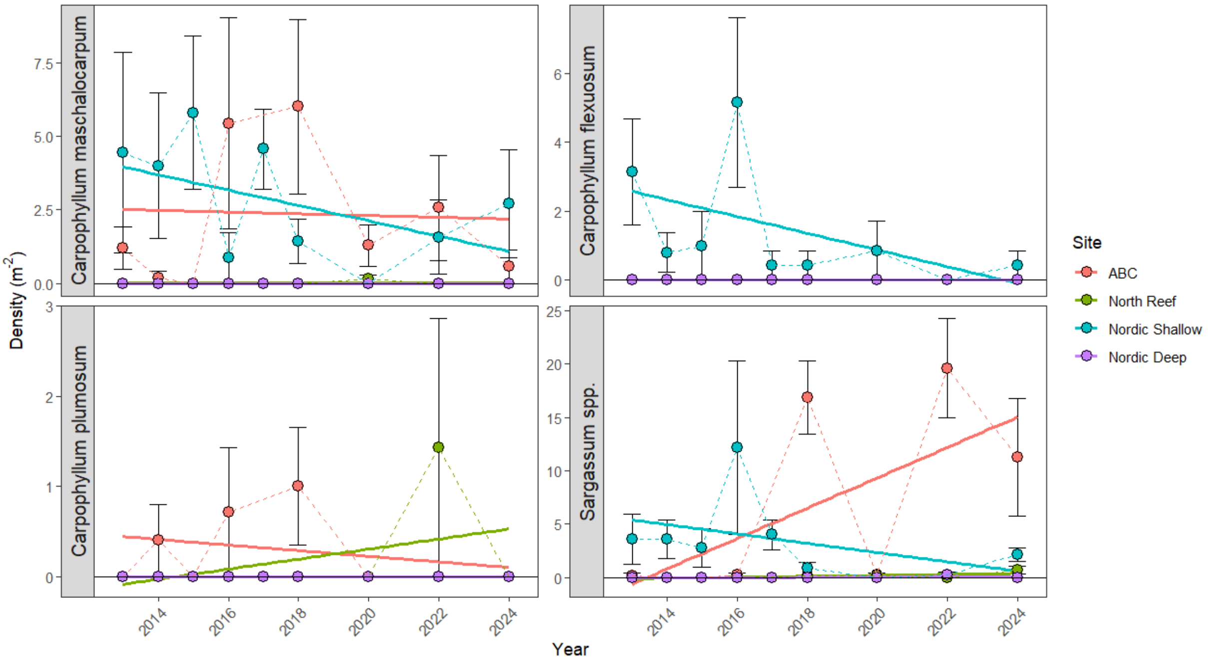
Nordic Deep (15 m) has typically had a sparse canopy (~20-30% cover) of tall *E. radiata* (Figure 20). However, canopy cover, density and mean size of *E. radiata* declined between 2020 and 2024. In 2024, canopy cover was only 8.5% and densities were 2.5 m^{-2}, both the lowest values over the monitoring period.

Figure 20. Site-level variation in the canopy cover (A), density (B) and mean length (C) of *Ecklonia radiata* at Leigh from 2013 to 2024.



Carpophyllum maschalocarpum, C. flexuosum, and *Sargassum sinclairii* were common at Nordic Shallow but have generally declined over time (Figure 21, Appendix 2b). *Carpophyllum maschalocarpum* and *Sargassum sinclairii* were also common but highly variable at ABC underneath the *E. radiata* canopy.

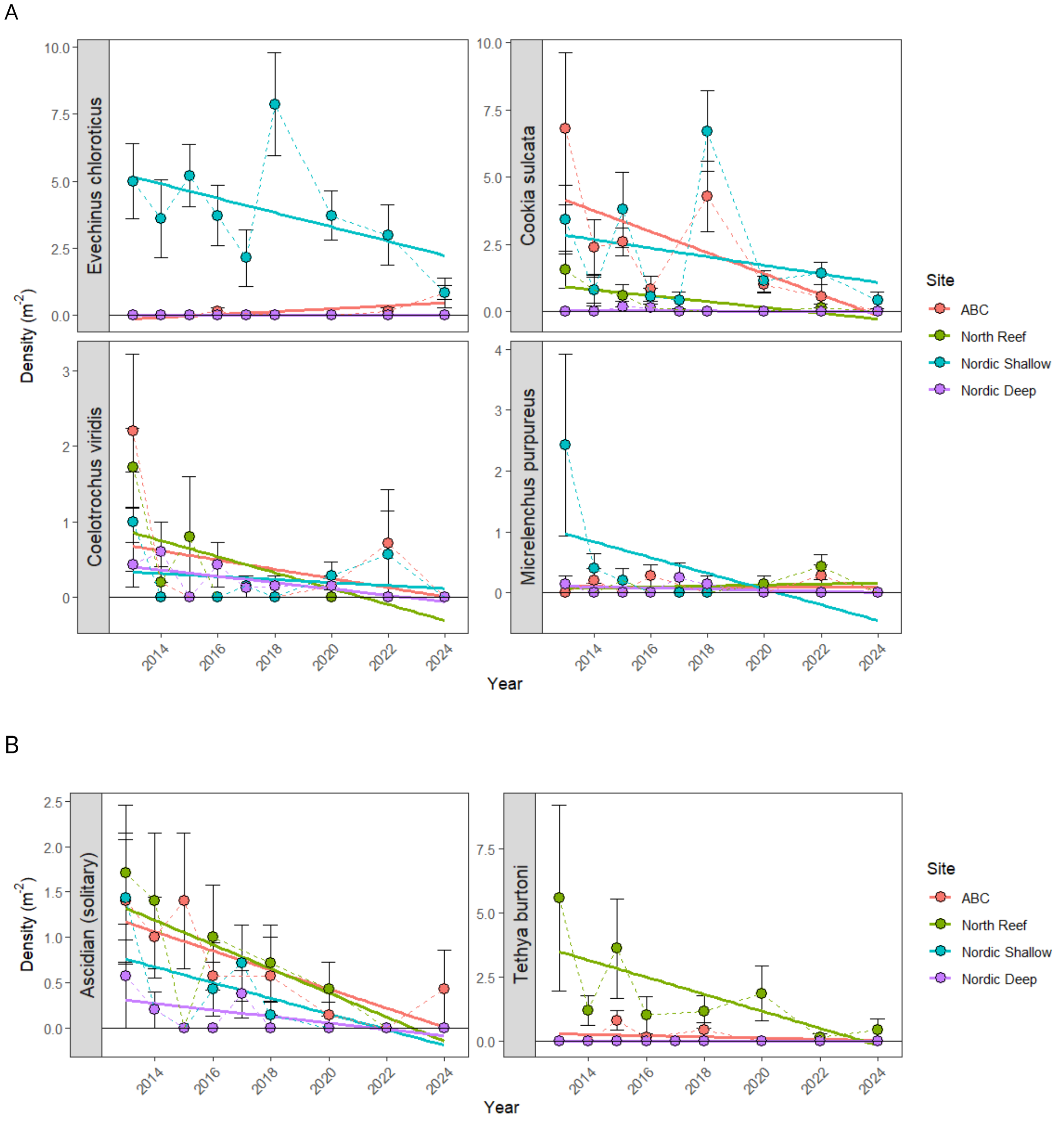
Figure 21. Site-level variation in the density of other large brown macroalgal species at Leigh sites from 2013 to 2024.



Density of mobile and sessile invertebrates

The density of dominant mobile and sessile invertebrates has generally declined across all sites (Figure 22). Total gastropod density declined at all sites except Nordic Deep where they were generally rare throughout the time series (Appendix 2d). Kina *Evechinus chloroticus* was most abundant at Nordic Shallow, but declined over time, particularly between 2022 and 2024 (Figure 22A, Appendix 2e). Dominant gastropod species were also initially variable but were at particularly low levels in 2024. The most common sessile invertebrates that are counted, solitary ascidians and *T. burtoni*, have generally declined, but this was only significant for solitary ascidians at Nordic Shallow (Figure 22B, Appendix 2c).

Figure 22. Site-level variation in the density of dominant (A) mobile and (B) sessile invertebrates at Leigh sites from 2013 to 2024.



Dominant benthic covers and species

The cover of sediment and filamentous algae has increased across all sites except ABC, whereas the cover of crustose coralline algae and other encrusting algae (e.g. *Hildenbrandia* sp. and *Ralfisa* sp.) has declined (Figure 23A, Appendix 2f). *Corallina officinalis* and red turfing algae increased at all sites except Nordic Deep. Overall sponge cover remained relatively stable at North Reef and Nordic Deep but show a weak downward trend at ABC and significant decline at Nordic Shallow (Figure 23B). Overall ascidian cover declined across all sites except Nordic Deep, following general trends seen in solitary ascidian density. Covers of hydroids and bryozoans were generally low but variable. The cup coral *Monomyces rubrum* also occurs at low numbers at all sites (Figure 23C) but has generally declined and was only recorded at North Reef in 2024.

Figure 23. Site-level variation in dominant substratum covers (A), cover of broad taxonomic groups for sessile invertebrates (B) and cover of cup coral *Monomyces rubrum* (C) at Leigh from 2013 to 2024.

A

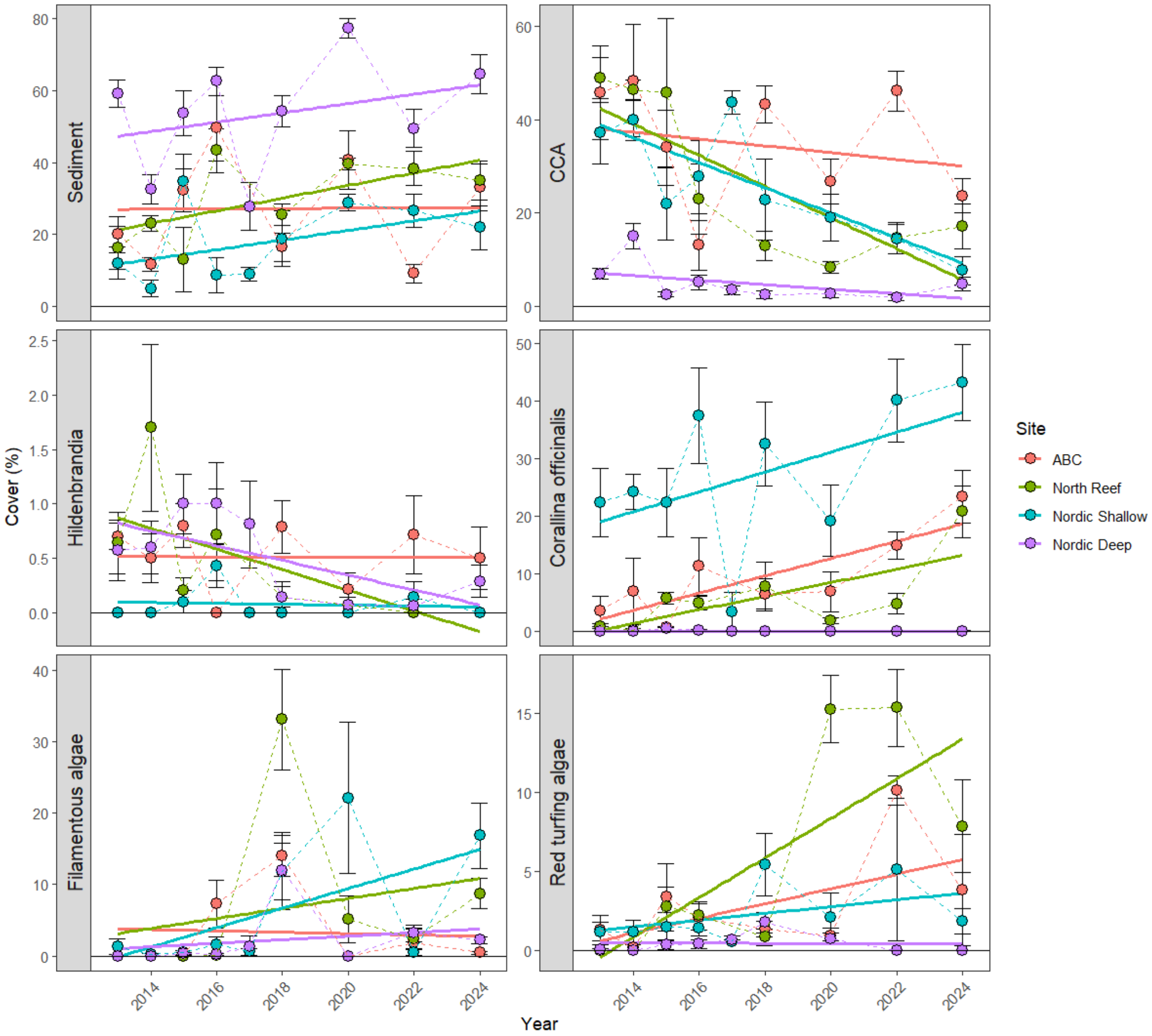
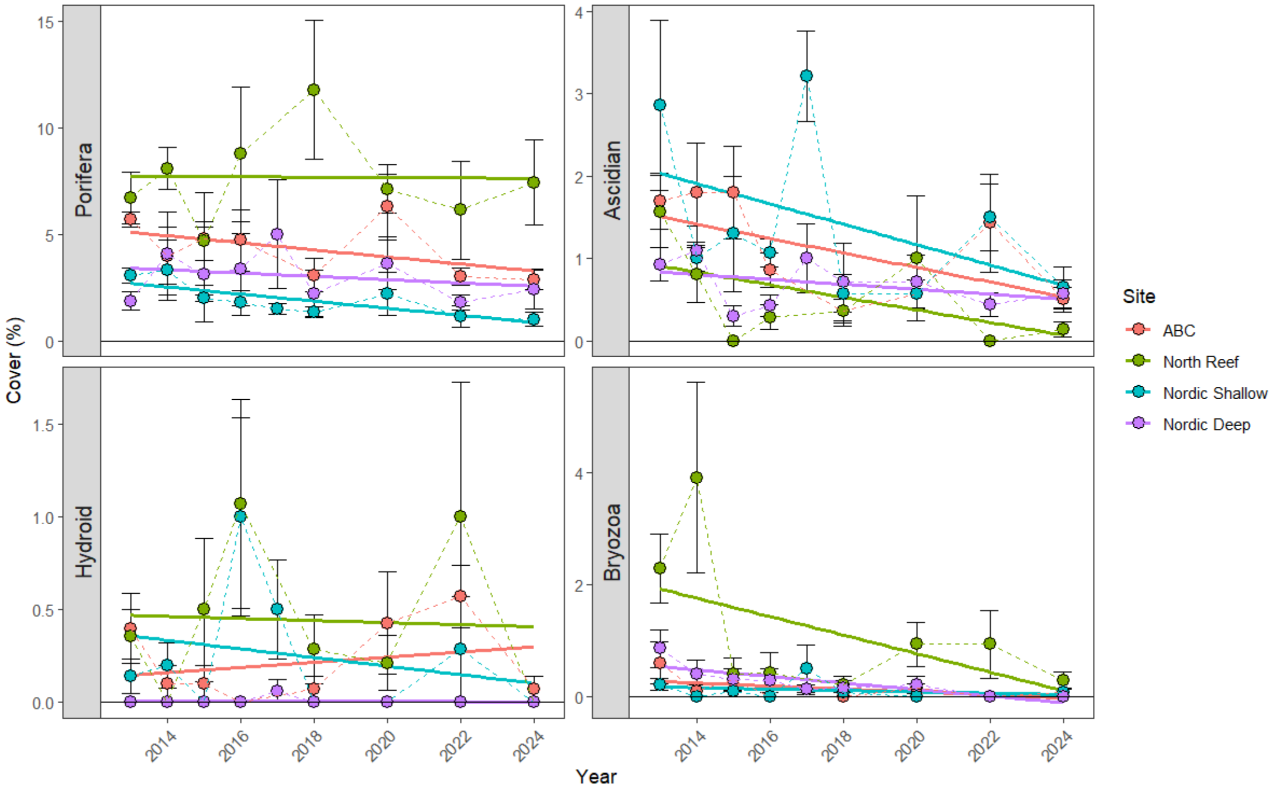
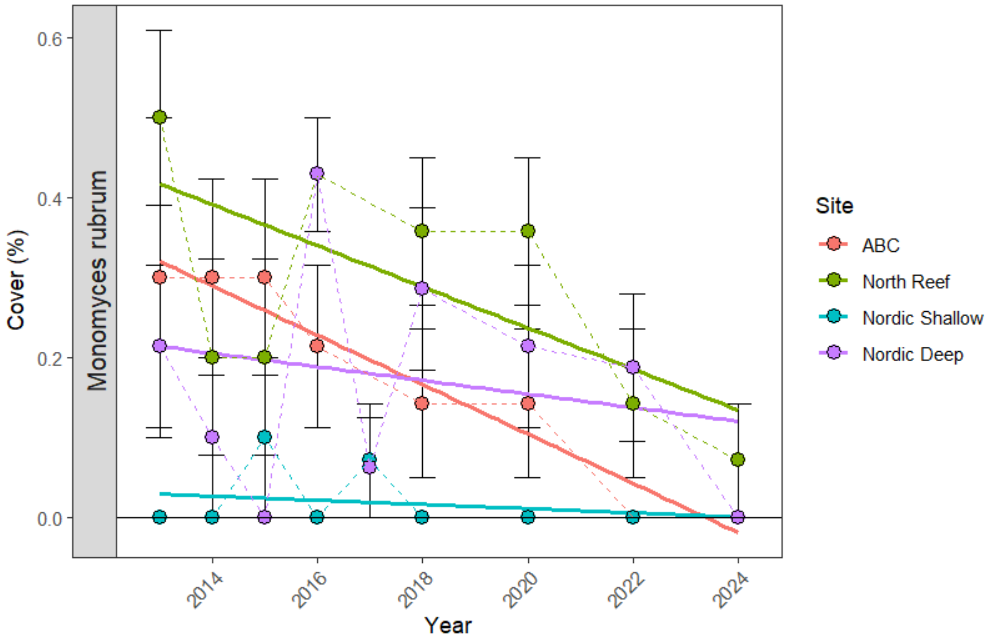


Figure 23. (Continued).

B

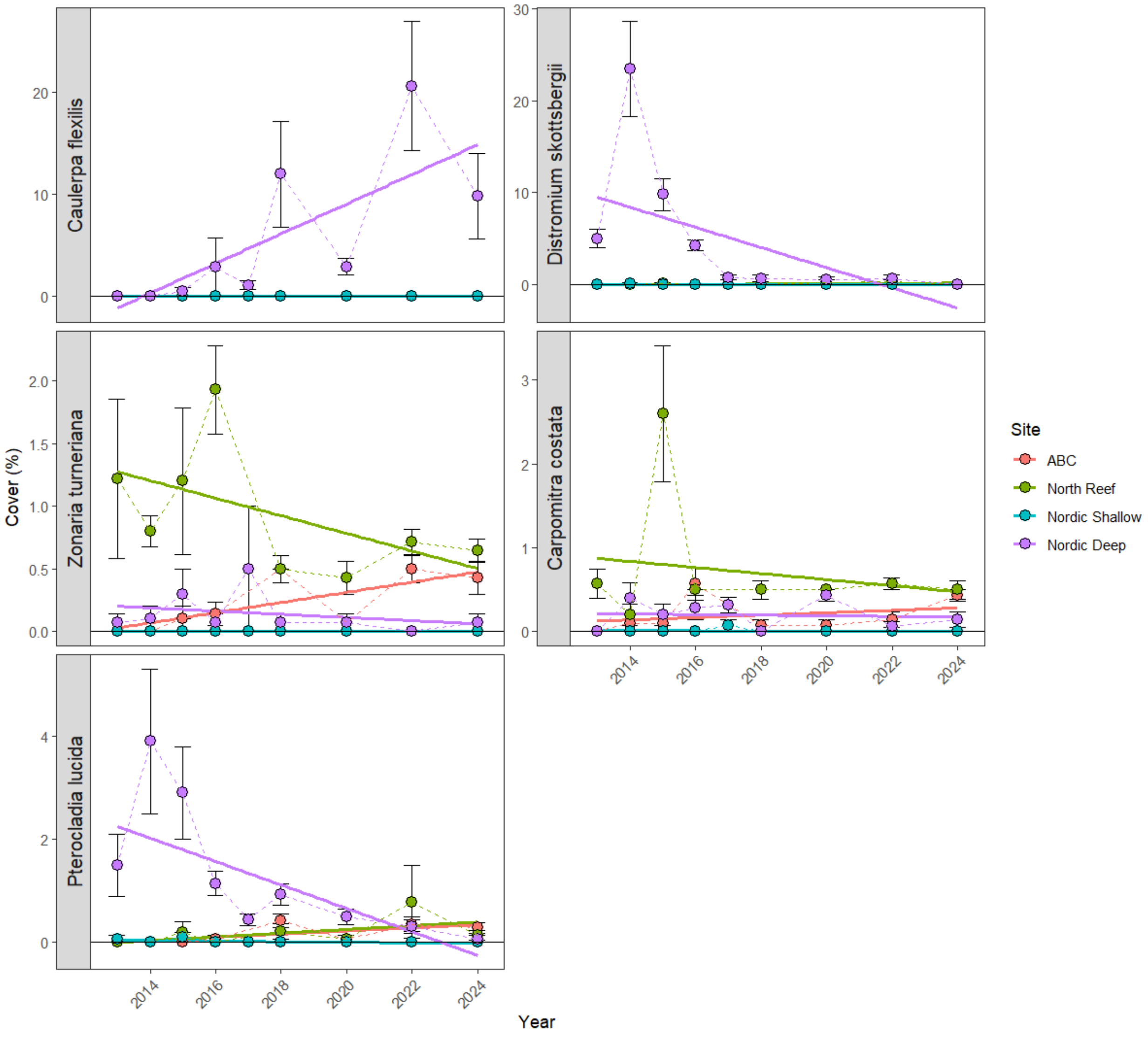


C



A number of seaweed species are common underneath the *E. radiata* canopy at the two deeper sites (Figure 24). At Nordic Deep the cover of *Caulerpa flexilis* has increased whereas the cover of *Distromium skottsberrgii* and *Pterocladia lucida* has declined. *Zonaria turneriana* and *Carpomitra costata* remain relatively common at North Reef.

Figure 24. Site-level variation in substratum cover of dominant subcanopy seaweed species at Leigh from 2013 to 2024.



4 Discussion

The data collected from 2007 to 2025 across Meola Reef, East Coast Bays-Whangaparāoa, and Leigh highlight significant ecological shifts in the Hauraki Gulf's shallow reef ecosystems, with continuation on previously described long-term trends evident, but also some rapid and concerning changes over the last five years. The long-term shifts in macroalgal community composition, declines in gastropods and increases in sediment cover and invasive species previously described at Long Bay (Peleg et al., 2023a) have continued and are generally evident across the wider geographic region from Meola Reef to Leigh. However, several additional changes have been detected in recent years such as emergence of new invasive species, rapid declines in macroalgal canopies and an increase in filamentous and turfing algae. Furthermore, a number of multi-species mortality events and complete canopy losses have been observed at some sites over the last five years. Overall, these changes are pointing to a shift towards a less complex and more degraded reef state at many sites across the sampling programme. Similar patterns are increasingly being reported on temperate reefs globally and related to increasing anthropogenic stressors and climate change (Filbee-Dexter and Wernberg 2018; Farrell et al., 2025).

Widespread shifts and loss of macroalgal canopy species

The predominance of macroalgal forests on temperate reefs is typically considered an indicator of a healthy reef ecosystem (e.g. Sala et al., 2012). All sites monitored in this study were characterised by macroalgal forests when monitoring began, with the exception of Nordic Shallow at Leigh which was an urchin barren. While macroalgal-dominated reefs in the inner Hauraki Gulf have generally been considered more stable than macroalgal forests at exposed locations, large shifts in canopy composition and in some cases loss of canopies have been observed since 2007 (Figure 25A). In particular, the endemic forest-forming fucoids *Carpophyllum maschalocarpum* and *C. flexuosum* have almost disappeared completely on the reefs sampled, being replaced by *Ecklonia radiata* and in other cases by shorter more ephemeral macroalgae such as *Sargassum* spp. or invasive species such as the small brown seaweed *Cladostephus hirsutus* (Figure 25B).

Figure 25. Examples of loss of kelp canopies (A) and increase in invasive species (B) on shallow reefs in the Hauraki Gulf. A. Site S5 at Stanmore Bay during 2024 sampling, demonstrating a multi-species mortality event with complete loss of *E. radiata* (remains of stipes evident) and *Carpophyllum* spp. (foreground) canopies and predominance of filamentous algae and sediment (also note presence of invasive algae green algae *Codium fragile*). B The invasive brown macroalgae *Cladostephus hirsutus* at C2 interspersed with *Cystophora retroflexa, Sargassum* spp. and *Ecklonia radiata. Cladostephus hirsutus* is now a dominant component of the reef assemblage at some sites where the native macroalgal canopy has been reduced. Photos: N. Shears.

A

B

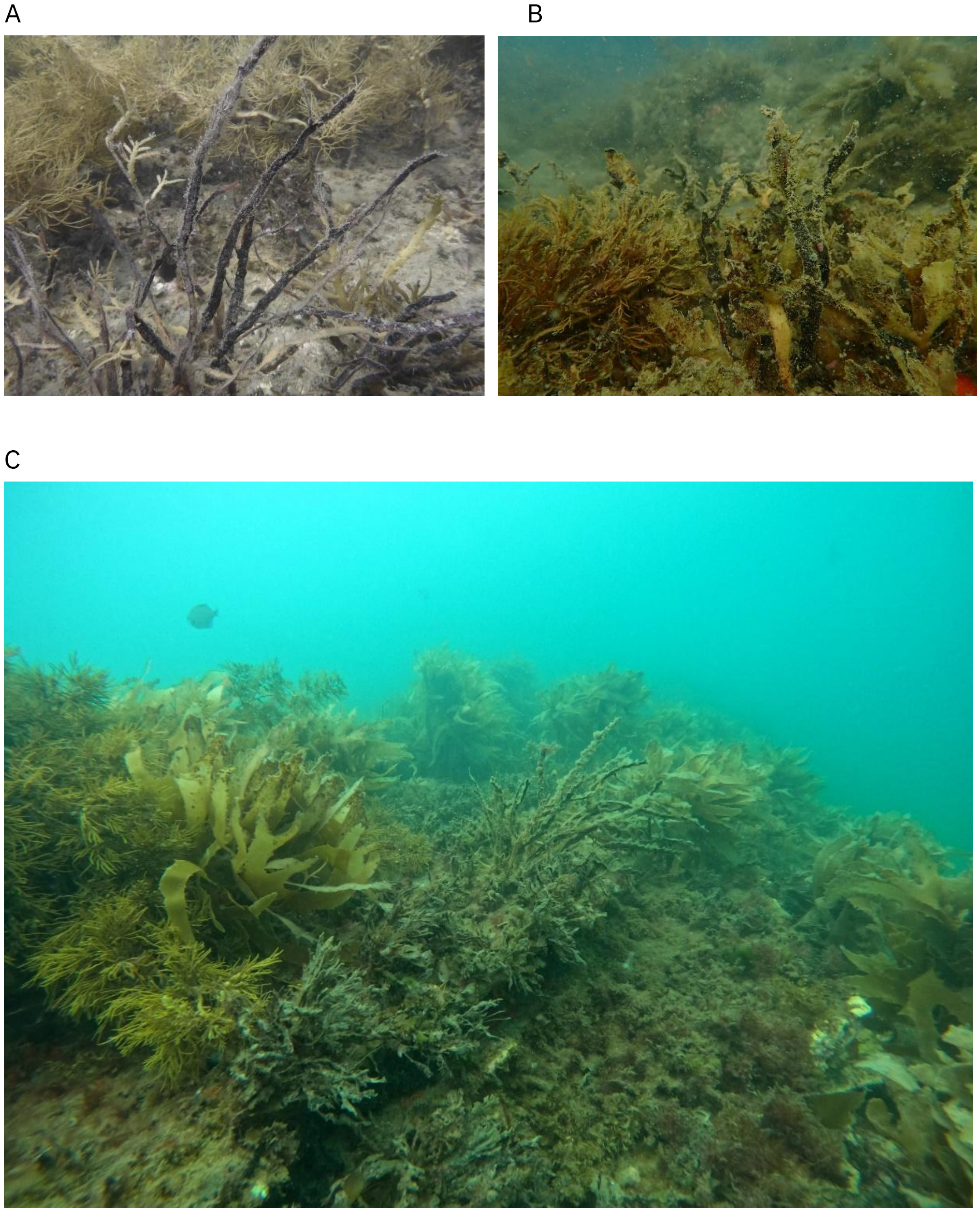
At Meola Reef, overall canopy cover has remained stable, but there has been a shift from *C. maschalocarpum* and *C. flexuosum*, to domination by *Ecklonia radiata*, and to a lesser extent *Sargassum sinclairii*. The increase in kelp and decline in *Carpophyllum* species is contrary to expectations based on kelp and fucoids and the highly urbanised nature of the Waitematā Harbour catchment, with high turbidity and high nutrients. Worldwide, kelp has been lost from urbanised reefs as a result of pollution (Connell et al., 2008) and fucoids are often considered more resilient than kelp to environmental stressors like low turbidity (e.g. Blain and Shears 2020). The reef at Meola is shallow (~1m depth at MLW) with high tidal currents, and therefore presumably subject to sufficient light to support *Ecklonia,* despite the high turbidity.

The decline in *Carpophyllum* species and increase in *E. radiata* at Meola Reef, does however appear to reflect a region-wide trend with similar changes observed at sites at Long Bay, Stanmore Bay and Waiwera. All three *Carpophyllum* species have declined across many of the ECB-WGP sites and are now rare or only comprise a small component of the macroalgal canopy. In some cases, there was a shift to a largely *E. radiata* dominated canopy (e.g. L3-5, S4, W4), whereas other sites have shifted to a mixed canopy also comprising *E. radiata* and the fucoids *Cystophora retroflexa* and *Sargassum* spp. (Figure 25B), which have increased over time. The two species of *Sargassum* (*S. sinclairii* and *S. scabridum*) are often difficult to differentiate, but it appears *S. scabridum* is most abundant at ECB-WGP sites. It is shorter (generally ≤25cm) than *S. sinclairii* and often occurs at high densities on low relief reefs with high sediment cover. Both *C. retroflexa* and *S. scabridum* appear to be tolerant to a high covering of sediment and are most prevalent at sites with the highest sediment cover.

At Leigh, *E. radiata* has remained the dominant canopy forming species, but density and cover have fluctuated over time in relation to dieback events. There has however been a long-term decline in density at Nordic Deep that appears independent of dieback. This site has had high and increasing sediment cover, and has also seen declines in some subcanopy species such as *Distromium skottsberrgii* and *Pterocladia lucida.* Among the Leigh sites, *C. maschalocarpum* and *C. flexuosum* were only common in the barrens at Nordic Shallow and have since declined which is consistent with the wider regional trend in these species.

The cause of the declines in *Carpophyllum* species is generally unknown, but mortality events for both *C. maschalocarpum* and *C. flexuosum* have been observed during surveys since 2019 (Figure 26). Often patches of both species were observed to be heavily fouled by filamentous algae and/or with blackened stipes and with little live tissue or new growth. In most cases, other surrounding species such as *Cystophora retroflexa* and *Ecklonia radiata* appear in comparatively healthy condition. These events have largely been observed in summer when surveys are carried out, but do not appear to have coincided with large marine heatwaves. Further investigation is needed into the cause of such events and in particular any potential links with warming water temperatures.

Figure 26. Examples of mortality of *Carpophyllum maschalocarpum* (A: Site T5 in 2019), *C. flexuosum* (B: Site W2 in 2020) and an example of multi-species mortality (*Carpophyllum* spp. and *Zonaria turneriana*) with high levels of fouling (C: W3 in 2024). Photos: N. Shears.



In addition to the shift in composition, most sites have also had a long-term decline in mean length of canopy-forming species. The overall decline in mean length of canopy forming species at Meola and ECB-WGP is in large part due to the loss of *Carpophyllum* species and shift to *Ecklonia* and other species which are generally shorter than *C. flexuosum*. This has been evident at Meola Reef where macroalgal canopies were initially dominated by *C. flexuosum* up to 2m in length, whereas canopies are now dominated by *Ecklonia* that is typically <1m in length.

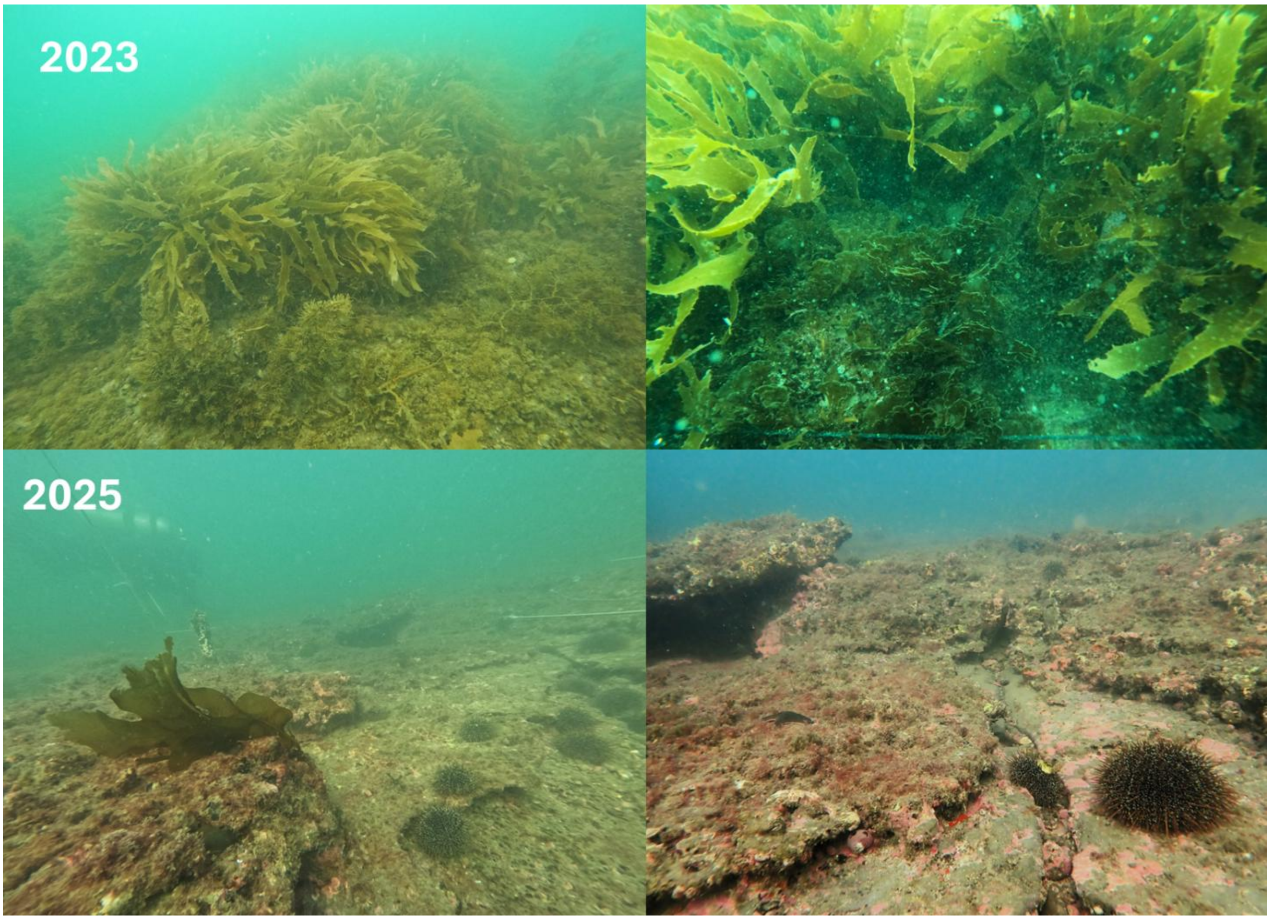
Overall macroalgal canopy cover was relatively stable across most of the time series, but there was a large decline across most ECB-WGP sites over the last five years. In some cases, these declines in canopy included multiple species, with the worst example being S5 where the entire canopy was lost between the 2022 and 2024 surveys (Figure 25A). This included complete loss of *E. radiata* and *Carpophyllum* species, and it is unknown whether the canopy will recover. Similarly, at L5 near the southern boundary of the Long Bay-Okura Marine Reserve there was a complete loss of the canopy between 2023 and 2025 (Figure 27). Only one ECB-WGP site (S4) has maintained a stable macroalgal forest over the last five years.

Marine protection has been shown to increase kelp forest resilience at Leigh, with increased size and abundance of predators in the marine reserve acting to keep sea urchin numbers and grazing impacts down, leading to a more stable kelp forest (Peleg et al., 2023b). However, in the Long Bay-Okura Marine Reserve, sea urchins have historically been at low densities and recent declines in canopy cover appear to be independent of sea urchin densities. While this suggests marine protection at Long Bay may not have provided additional resilience to recent stressors, ongoing monitoring is needed to understand if these changes represent long-term shifts or whether macroalgal canopies recover. In contrast, the dominance of *E. radiata* at the shallow site within the Leigh reserve (ABC) is consistent with low urchin numbers, compared to Nordic Shallow outside the reserve which has fluctuated from urchin barren to kelp forest and now turfing algae. This greater variability in reef state at fished compared to reserve sites at Leigh is consistent with previous studies (Peleg et al., 2023b). The two deep sites at Leigh have generally remained dominated by *E. radiata,* but overall density has decreased at Nordic Deep. The kelp forest at this site does appear to be under greater sediment stress, likely due to its close proximity to Leigh Harbour.

Increases in sediment cover and turfing and filamentous algae

There has been a general increase in sediment cover on the reef across all locations, with the exception of Meola sites where sediment cover has generally been high throughout the monitoring period. While an increase in sediment may be expected to be related to increased coastal development at Long Bay over the study period, increases were evident across most of the ECB-WGP sites excluding two sites at Long Bay (L1 and L4) and one at Stanmore Bay (S4). Furthermore, increases in sediment were also evident at three of the four Leigh sites. These region-wide patterns suggest the increase in sediment cover is related to climatic events and hydrodynamic conditions, rather than tied to specific catchment activities over the monitoring period. Previous reporting highlighted the impacts of cyclone Wilma in 2011 which resulted in extensive cliff landslides and sediment runoff, which was evident in a peak in sediment accumulation and cover at sites in and adjacent to Long Bay (Shears, 2017). Since this time there have been numerous large cyclones and rain events that have likely contributed to increasing regional trends in sediment cover on the reefs monitored (e.g. Cyclone Gabriel and the "Auckland Floods" discussed below).

Figure 27. Loss of macroalgae between 2023 and 2025 at L5, Long Bay. In 2023 the reef was largely dominated by macroalgal canopy, but seaweeds were heavily fouled and had a high covering of sediment (top left). Understory species such as *Zonaria turneriana* were common in 2023 (top right). In 2025, large brown algae and understory species were largely absent, except a few *Undaria pinnatifida* (below left). Sea urchins had also emerged from being cryptic in 2023, to a more exposed behaviour in 2025, and the substratum was predominantly covered by filamentous algae, crustose coralline algae, sediment and bare rock (bottom right).



The sites with the greatest decline in canopy cover were also generally those with the greatest increase in sediment cover. Sites with healthy and stable canopies (e.g. ABC, L4, S4) tended to have less or no increase in sediment cover over time, and in general dense macroalgal canopies can help disperse sediment and prevent deposition on the seafloor. However, once canopies start to decline or are lost, then sediment can settle and increase on the reef, which can inhibit recruitment of large brown algae and therefore impact recovery dynamics following die-off or disturbance events. The increase in sediment has generally been gradual and likely event-driven over the monitoring period, whereas the decline in macroalgal canopy has been more sudden with large declines over the last five years. It is therefore possible that increased sediment cover has reduced resilience and prevented recovery from other stressors rather than having directly driven canopy decline.

Increases in turf and filamentous algae are also increasingly associated with macroalgal canopy loss (Filbee-Dexter and Wernberg 2018, Farrell et al. 2025). In these cases, canopy loss is typically driven by environmental stressors (e.g. marine heatwaves), and this results in a shift to a turf state that inhibits macroalgal recovery (Connell et al. 2008, Reeves et al. 2018). These turf-dominated systems have been shown to support lower biodiversity and functional complexity than macroalgal forests in temperate regions globally (Filbee-Dexter and Wernberg 2018). A general increase in filamentous algae and turfing algal species has occurred across many of the monitoring sites in recent years, including at sites that have not seen any canopy loss. The likely cause is unknown and requires further investigation, but increases in filamentous algae are often related to increased nutrients and/or temperature (Connell et al., 2008; Filbee-Dexter and Wernberg, 2018). In general, there is a paucity of information on the ecology, phenology and potential impacts of these filamentous species, but their increasing abundance in the Hauraki Gulf may be related to recent warming trends (Shears et al., 2024). Furthermore, the taxonomic identity of the filamentous species is largely unknown and it is therefore possible some of these species may be invasive.

Marked decline in mobile invertebrates

Total densities of gastropods have declined dramatically across all monitoring sites, except Nordic Deep where densities have generally been low throughout the monitoring period. At Meola and ECB-WGP locations, the declines have been observed across multiple species, including herbivorous gastropods, chitons and predatory whelks. The only gastropod that has not declined, and has in fact increased at some sites, is the turret shell *Maoricolpus roseus.* Unlike other gastropods this species is a suspension feeder and embeds its spire into sand or holes in the reef, with only their apertures exposed. This behaviour and feeding mode may therefore make this species more resistant to higher sediment loads. The cause of the general declines in gastropods is unknown but may be linked to increases in sediment and/or declines in crustose coralline algae, which occurred over similar time periods. Notably, the large declines in gastropods preceded the more recent declines in macroalgal canopy, which could have impacted on gastropod species that live in kelp canopies.

Declines in gastropods on shallow temperate reefs have previously been linked to smothering by fine sediments or elevated toxicity from urban contaminants (e.g. limpets; Airoldi and Hawkins, 2007; Atalah and Crowe, 2012). While increased sediment may reduce crustose coralline algae (CCA), the primary grazing surface for herbivorous gastropods, the declines in gastropods were also evident at sites where sediment cover hasn’t increased and CCA has remained stable. Warming ocean temperatures have been proposed as a potential contributor to a collapse of multiple gastropod species in the Mediterranean Sea (Rilov, 2016), so it is possible recent temperature trends may have impacted gastropod populations. However, our knowledge of impacts of other factors such as toxins, disease and toxic algal blooms on gastropods is limited and further research is needed to understand these declines.

Echinoderm species do not appear to have undergone any unidirectional shifts, with trends in different species varying among sites. Kina have increased at some of the ECB-WGP sites, but still remain at relatively low densities. At two sites (L5 and T3) densities have increased to 4-6m^{-2}, densities at which impacts on macroalgal assemblages may start to occur (Ayling, 1981). In general, kina currently do not appear to have an impact on overall macroalgal assemblages at the ECB-WGP monitoring sites but if recent trends continue this may be expected. The recent declines in macroalgal canopy at L5 has resulted in a shift in kina behaviour from cryptic (in cracks and crevices) to becoming more exposed and actively grazing the reef substratum (Figure 27). Ongoing monitoring at this site will determine whether kina prevent kelp recovery and ultimately lead to the formation of an urchin barren. At Nordic Shallow, kina initially formed barrens, but numbers have fluctuated and there has been a large decline in recent years. This site is prone to blooms of *Ostreopsis siamensis,* which has negative impacts on kina (Shears and Ross, 2009), and was also heavily impacted by large waves during Cyclone Gabrielle in 2023 which led to declines in kina numbers along the Leigh coast (K. Miller and N. Shears, unpubl. data).

Rising incidence of invasive species

Another key finding has been the increase in invasive species over the monitoring period, which was generally greatest at Meola and lowest at Leigh. This general pattern is not surprising given that invasive species are generally spreading throughout the Hauraki Gulf (Aguirre et al., 2016), and the close proximity of Meola Reef to the Auckland Port in the Waitematā Harbour and the ECB-WGP sites to the Weiti River and Gulf Harbour marinas. The spread of invasive species from these locations is expected through natural dispersal pathways but may also be facilitated through high recreational boat traffic in the Hauraki Gulf (Floerl and Inglis, 2003).

The clubbed ascidian *Styela clava* was common at ECB-WGP locations but has generally declined and only one was recorded in 2024. The Mediterranean fanworm *S. spallanzanii* was first recorded at Meola Reef in 2011 and Campbells Bay in 2013 but has now spread from sheltered to more exposed outer Gulf locations, being recorded at Nordic Deep in 2024. It is well established at ECB-WGP locations and while densities are at relatively low levels (<1m^{-2}), they have been highly variable in recent years.

Unlike the well-known invasives, we have also documented increases in abundance and distribution of some lesser-known species. The distribution of the invasive brown algae *Cladostephus hirsutus* is poorly understood (Heesch et al., 2020) but this species now occurs at Meola Reef sites, and is a dominant and common component of the macroalgal assemblage at a number of sites at Campbells Bay, Manly and Long Bay. It appears to be highly opportunistic and colonise reefs with high sediment cover and lacking native canopies (e.g., Figure 25B).

Similarly, the ascidian *Symplegma brakenhielmi* was first recorded at Meola in 2017 and now occurs at low covers across most ECB-WGP locations. While not recorded at Leigh sites, it is well established and increasing in the outer Hauraki Gulf, where it grows on rocks, algae and sessile invertebrates (Spyksma et al., 2024). Another new invasive ascidian species *Ascidia latesiphonica* was recorded at Meola Reef and Campbells Bay (C5) in 2024, and at multiple sites at Torbay, Long Bay and Manly in 2025. Like *S. brakenhielmi*, this species appears to be spreading rapidly.

The potential effects of invasive species on ecosystem functions and services on rocky reefs in the Hauraki Gulf are poorly understood (e.g. Douglas et al., 2020; Tait et al., 2020). In many cases, these species are likely opportunistic and facilitated by degraded ecosystem states or changing environmental conditions, rather than directly impacting or displacing native reef communities and species (e.g. James and Shears, 2016). At this stage, the invasives observed in this study would most likely fall in the former category benefiting from loss of macroalgal canopies and warmer water temperatures. To date, the exotic *Caulerpa* species that are rapidly spreading throughout northeastern New Zealand have not been observed at any of the monitoring sites.

Environmental change

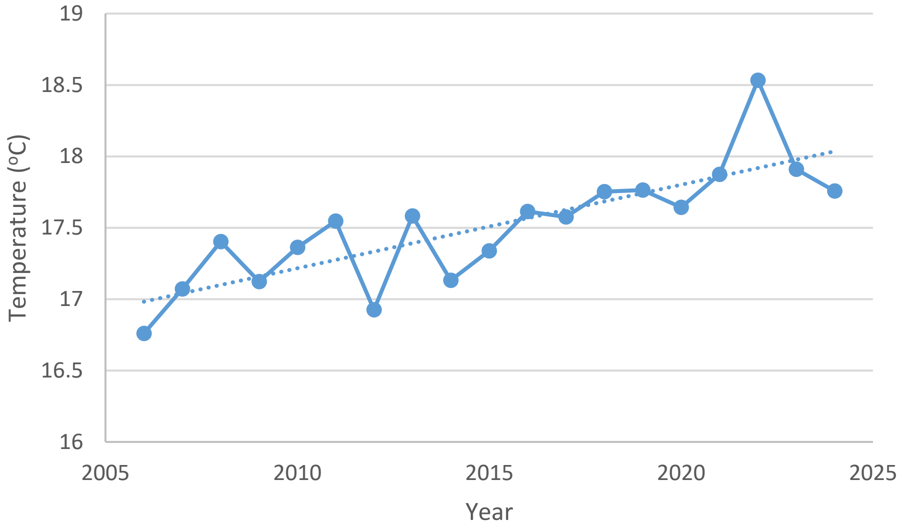
Previous analysis of data from the Long Bay area from 2007-2019 reported that the large collapse in the gastropod community, shortening of macroalgal canopies and an increase in the number of invasive species, coincided with an increase in sediment cover on the reefs and warming ocean temperatures over the monitoring period (Peleg et al., 2023a). Analysis of trends across all monitoring sites, spanning Meola Reef in the Waitematā to Leigh in the outer Hauraki Gulf, demonstrate that similar changes have occurred over a larger geographic area. This suggests these changes may relate to environmental change or climatic events operating over a larger geographic scale, rather than being due to localised impacts. Furthermore, these long-term trends have generally continued, and new patterns are emerging over the last five years, such as die-off of seaweeds and loss of macroalgal canopies. While the exact cause or causes are unknown, these recent trends coincide with extreme environmental conditions in recent years. Ocean temperatures at Leigh have increased steadily over the monitoring period (Figure 28A), with 2022 being the warmest year in the long-term record (1967-2024) with almost continuous marine heatwave conditions (Shears et al., 2024). This was followed by the wettest year on record at the Leigh Marine Laboratory in 2023 (Figure 28B), with a series of large rain events and storms in early 2023 including the Auckland Anniversary Floods and Cyclone Gabrielle (Auckland Council, 2023). These large rainfall events were associated with extensive landslides throughout the Auckland region (Howard and Roberts, 2024) and would have resulted in a high degree of runoff of sediment into the coastal region. In addition, extreme wave action associated with these events would have resulted in greater coastal erosion and resuspension of sediments, which would have contributed to extended periods of high turbidity (Seers and Shears, 2015).

The effects of the marine heatwaves on rocky reefs in 2022 included die-offs of sponges, expansion of tropical invasive species and high coverage of filamentous algae (Shears et al., 2024; Spyksma et al., 2024). Many of these changes were evident during monitoring in 2022 and 2023, such as increases in the invasive ascidian *Symplegma brakenhielmi* and filamentous algae, and mortality of some sponges such as *Tethya burtoni*. The large decline in macroalgal canopy cover that occurred across many sites between 2022 and 2025 is potentially a combined effect of the extreme events that occurred in 2022 and 2023. While declines in canopy cover at a number of Torbay, Long Bay and Manly sites occurred between the 2023 and 2025 surveys, this is likely related to ongoing stress from aforementioned extreme events as well as more recent conditions. Sampling at these sites in 2023 occurred after the Auckland Anniversary Floods and Cyclone Gabrielle and macroalgal canopies were heavily fouled and covered with sediment at this time. Furthermore, recent summer temperatures at Leigh have been consistently higher than average in recent years and available logger data from L4 recorded peak summer temperatures in 2025 at 24.7°C (13th February 2025; N. Shears unpubl. data).

The extent of macroalgal canopy loss is not known based on the spatially restricted monitoring sites, but appears to have occurred across large areas of reef adjacent to many of the sampling sites (e.g. Figure 29). Furthermore, a loss of deep kelp forest (>10m) has also been observed at a number of locations in the Hauraki Gulf since 2023, e.g. Hauturu-o-Toi, Tāwharanui, Tiritiri Matangi and the Noises Islands (N. Shears and C. Blain unpubl. data). The extent of the recent loss of macroalgal forest is poorly understood, but unlike previously documented kelp dieback events (Cole and Babcock, 1996), the recent mortality events appear to have impacted multiple species, across a range of depths and locations, and limited recovery has been observed. It is unclear whether some of these changes may represent long-term shifts or whether recovery may be slower under current environmental conditions. As with the longer-term changes observed across monitoring sites, the mechanisms driving recent declines in macroalgal canopy cover are unknown and further research and monitoring is needed to better understand these changes.

Figure 28. Mean annual sea surface temperature (A) and total annual rainfall (B) at the Leigh Marine Laboratory from 2006 to 2024. The dashed line shows linear regression for temperature (y=0.0585x-100.3; F=31.0, *p*<0.001) and rainfall (y=17.495x-34135; F=2.37, *p*=0.142).

A



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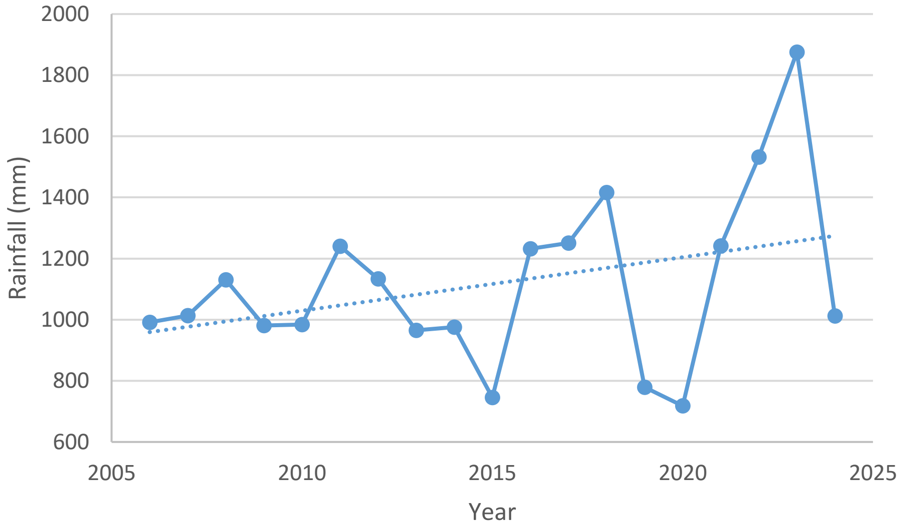
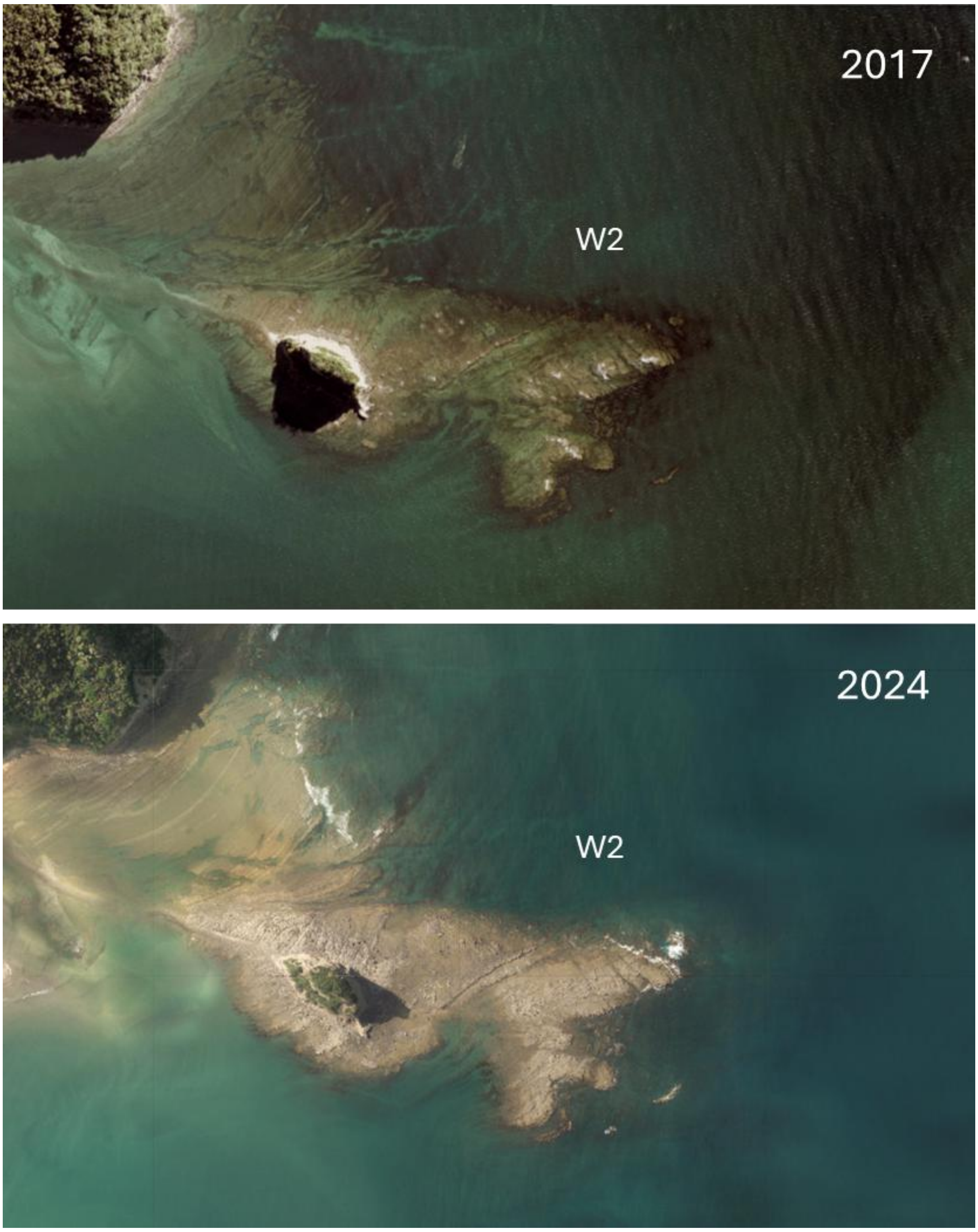


Figure 29. Aerial imagery of site W2 at Waiwera showing loss of macroalgal canopies between 2017 and 2024. Dark coloration shows extensive macroalgal forest on subtidal reef areas in 2017, but much of this has been lost in 2024. https://geomapspublic.aucklandcouncil.govt.nz/viewer/index.html



Implications and application

In many temperate ecosystems worldwide (e.g. in the Mediterranean, South Australia, and California), macroalgal canopies have been lost from urban reefs due to a range of local stressors, including sedimentation and nutrient pollution (Copertino et al., 2005; Mangialajo et al., 2008; Foster and Schiel, 2010), and these impacts are now being exacerbated by climate change (Russell et al., 2009). While such changes had not been observed on the reefs monitored in the East Coast Subtidal Monitoring Programme, previous reporting warned that the general trends were consistent with degradation in reef ecosystem health and may represent an "early warning" signal of an approaching tipping point towards a denuded state (Peleg et al., 2023a). The recent changes indicate that this was likely the case, with subsequent declines in macroalgal canopies at many sites and in some cases the complete loss of macroalgal forest. This is the first documentation of such impacts in northeastern New Zealand where kelp forest loss has historically been attributed to sea urchin grazing (Kerr et al., 2025). The widespread nature of these changes across the monitoring locations suggests they are regional and likely related to large-scale factors, rather than any specific local-scale or catchment-based impacts.

Given the increasing pressures on the Hauraki Gulf and increasing impacts of climate change (Hauraki Gulf Forum, 2023), it is essential that monitoring be continued to document further changes and the extent to which recovery occurs, in particular at sites where macroalgal forests have recently been lost. Understanding the link between the observed changes and environmental stressors will require targeted research and experimentation. This is needed to inform management so that actions can be taken to mitigate these impacts and develop restoration strategies.

Monitoring and research recommendations

It is recommended that monitoring of existing sites continue on a biennial basis to better understand ecological thresholds and track recovery or further degradation. The majority of monitoring is carried out on shallow inner-Gulf reefs (<5m) and therefore some recent changes in the Hauraki Gulf such as mortality of deep kelp forest (N.S. pers. obs.) and increases in subtropical sea urchins have not been detected (Balemi and Shears, 2023) in the current monitoring programme. It is therefore recommended that additional sites be added into the programme to provide greater replication in the Leigh area and also include sites at the offshore Islands (e.g. Mokohinau Islands, Aotea and Te Hauturu-o-toi). These could be incorporated into wider monitoring carried out by other agencies such as the Department of Conservation.

Monitoring of seawater temperature and other environmental conditions should be incorporated into the current monitoring programme to better link ecological changes with potential environmental stressors. Temperature and PAR (photosynthetically active radiation) loggers could be installed to record water temperature and light at one site from each location. Monitoring of sediment inputs into highly dynamic reef environments is challenging but could be explored further including the use of satellite and drone images.

Wider investigation into the extent of macroalgal forest loss is needed to understand the scale and potential drivers. This could be done using available satellite and aerial imagery (as in Figure 29). Targeted research into factors influencing macroalgal forest resilience and conversely the drivers of mortality events is needed. This could involve event driven sampling and additional instrumentation (e.g. dissolved oxygen loggers) at sites, as well as mesocosm experiments investigating the interactive effects of multiple stressors (e.g. temperature and light) on key taxa (e.g. *Carpophyllum* spp.). Additional research could also investigate the effects of invasive species and filamentous algae on the reef communities.

Long-term trends are now clearly suggesting impacts of climate change on reef ecosystems (e.g. warming and increased storm frequency and intensity), highlighting the need for climate action internationally, nationally and regionally. Given the complexity of drivers and the early indicators of regional degradation, there is a need for management strategies focused on both mitigation of local stressors and building long-term resilience. Impacts of sediment observed through the monitoring show the need for continuing to improve the management of catchment-based sediment controls. Invasive species were widespread through the monitoring sites and observations of new species were made. Eradication of invasive species in the marine environment is very resource intensive and in the vast majority of cases is unsuccessful (Simberloff, 2021). Prevention and early detection of marine invasives is essential so management efforts and resources are needed to focus on surveillance and early detection programmes. Fishing is a major stressor that promotes sea urchin populations and therefore undermines macroalgal forest resilience. Evidence is emerging that marine protected areas (MPA) can help to increase macroalgal forest resilience under certain circumstances. Increasing the amount of area protected from fishing will therefore help build greater resilience to other stressors that are more difficult to manage. Opportunities for active restoration in areas where macroalgal forests have been lost (e.g., kelp transplants, reseeding) should also be explored.

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6 Appendix 1 – Site positions and survey schedule

Positions and survey schedule of Auckland east coast subtidal reef monitoring sites. Position format in degrees, minutes and seconds. *ABC was the only Leigh site surveyed in 2012 so data from this survey is not included in this report.

Location/Site	Latitude	Longitude	Year																		
Meola Reef			**2007**	**2008**	**2009**	**2010**	**2011**	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**	**2019**	**2020**	**2021**	**2022**	**2023**	**2024**	**2025**
MSE2	36 50'05.74	174 42'34.43	X	X	X	X	X	X	X	X	X	X	X	X		X				X	
MSW2	36 50'11.38	174 42'33.23	X	X	X	X	X	X	X	X	X	X	X			X				X	
MSW3	36 50'05.94	174 42'32.35	X	X	X	X	X	X	X	X	X	X	X			X				X	
East Coast Bays/Whangaparaoa																					
Campbells																					
C1	36 43'47.23	174 45'29.59	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X	X		X	
C2	36 44'06.71	174 45'24.18	X	X	X	X	X	X	X	X						X		X		X	
C5	36 45'05.34	174 45'58.02	X	X	X	X	X	X	X	X						X		X		X	
Torbay																					
T3	36 42'28.51	174 45'14.99	X	X	X	X	X	X	X	X	X	X	X	X	X		X		X		X
T5	36 43'05.94	174 45'11.40	X	X	X	X	X	X	X	X	X	X	X	X	X		X		X		X
Long Bay																					
L1	36 39'55.38	174 44'56.21	X	X	X	X	X	X	X	X	X	X	X	X	X		X		X		X
L2	36 40'12.77	174 44'55.36	X	X	X	X	X	X	X	X	X	X	X	X	X		X		X		X
L3	36 40'31.22	174 44'59.99	X	X	X	X	X	X	X	X	X	X	X	X	X		X		X		X
L4	36 41'17.37	174 45'24.63	X	X	X	X	X	X	X	X	X	X	X	X	X		X		X		X
L5	36 41'32.85	174 45'36.63	X	X	X	X	X	X	X	X	X	X	X	X	X		X		X		X
Manly																					
M1	36 38'44.20	174 44'44.62	X	X	X	X	X	X	X	X	X	X	X	X	X		X		X		X
M3	36 38'12.90	174 45'57.39	X	X	X	X	X	X	X	X							X		X		X
M4	36 38'2.58	174 46'10.20	X	X	X	X	X	X	X	X	X	X	X	X	X		X		X		X
Stanmore																					
S2	36 37'08.90	174 44'59.60	X	X	X	X	X									X		X		X	
S4	36 36'20.90	174 46'26.00	X	X	X	X	X									X		X		X	
S5	36 36'14.90	174 46'49.91	X	X	X	X	X									X		X		X	
Waiwera																					
W2	36 32'38.72	174 43'16.72	X	X	X	X	X									X		X		X	
W3	36 33'03.60	174 42'40.90	X	X	X	X	X									X		X		X	
W4	36 33'19.50	174 42'38.10	X	X	X	X	X									X		X		X	
Leigh																					
ABC	36 15'56.90	174 47'41.11						X*	X	X	X	X		X		X		X		X	
North Reef	36 15'50.30	174 47'37.51							X	X	X	X		X		X		X		X	
Nordic Shallow	36 17'36.84	174 48'34.98							X	X	X	X	X	X		X		X		X	
Nordic Deep	36 17'37.28	174 48'38.02							X	X	X	X	X	X		X		X		X	

7 Appendix 2 – Site-level trends in key species over the monitoring period

Appendix 2a. Linear trends in total density of large brown algae (m^{-2}), mean length of large brown algae (cm), estimated canopy cover (%) from 2007-2024/2025 (except Leigh sites 2013-2024) and observed canopy cover from ECB-WGP sites from 2018-2024/2025. The slope of the linear regression is given for each site – significant values ($p<0.05$) in bold and marginally significant ($p=0.05-0.10$) in italics. Colour of cell weighted according to the slope, with red indicating a negative slope and blue a positive slope over time.

Area	Site	Total density	Mean length	Est. Canopy cover	Obs. Canopy cover*
Meola	MSE2	-0.05	***-1.12***	**1.03**	NA
Meola	MSW2	-0.37	**-1.57**	0.07	NA
Meola	MSW3	**-1.00**	***-0.83***	-0.37	NA
Campbells	C1	**2.03**	**-1.33**	**-1.58**	**-8.97**
Campbells	C2	0.14	**-1.24**	**-1.05**	***-6.36***
Campbells	C5	-0.52	**-1.98**	**-2.55**	***-4.27***
Torbay	T3	**-1.43**	**-1.45**	**-4.14**	**-5.62**
Torbay	T5	0.27	**-1.69**	**-2.42**	**-7.60**
Long Bay	L1	**-3.38**	**-1.70**	**-2.57**	**-6.33**
Long Bay	L2	-0.97	**-1.42**	**-2.05**	**-3.36**
Long Bay	L3	-0.23	**-1.12**	0.03	**-4.45**
Long Bay	L4	**-0.33**	-0.22	0.56	**-4.74**
Long Bay	L5	-0.65	**-1.35**	**-1.86**	**-6.74**
Manly	M1	0.08	**-1.91**	**-1.67**	**-6.17**
Manly	M3	**-2.53**	**-0.99**	**-3.48**	-1.64
Manly	M4	**-2.35**	**-0.67**	**-2.54**	**-7.95**
Stanmore	S2	**1.37**	**-1.73**	***-0.92***	**-4.14**
Stanmore	S4	**0.69**	-0.31	**1.09**	1.93
Stanmore	S5	**-0.99**	**-1.94**	**-2.25**	**-16.09**
Waiwera	W2	0.19	**-1.50**	**-1.95**	**-5.11**
Waiwera	W3	0.11	**-2.27**	**-1.95**	**-8.34**
Waiwera	W4	**0.99**	**-1.58**	0.22	**-3.68**
Leigh	L-ABC	0.75	0.39	***2.33***	NA
Leigh	L-NR	0.20	**-2.03**	-0.53	NA
Leigh	L-NS	***-1.12***	**-2.18**	-1.11	NA
Leigh	L-ND	**-0.58**	0.33	-0.81	NA

Appendix 2b. Linear trends in density (m^{-2}) of dominant large brown macroalgal species from 2007-2025 (except Leigh sites 2013-2024). The slope of the linear regression is given for each site – significant values ($p<0.05$) in bold and marginally significant ($p=0.05-0.10$) in italics. Colour of cell weighted according to the slope, with red indicating a negative slope and blue a positive slope over time.

Area	Site	*Carpophyllum flexuosum*	*Carpophyllum masch.*	*Carpophyllum plumosum*	*Ecklonia radiata*	*Cystophora retroflexa*	*Sargassum spp.*
Meola	MSE2	**-0.47**	**-0.59**	NA	**0.42**	NA	0.61
Meola	MSW2	**-0.47**	**-1.36**	NA	**0.76**	NA	**0.71**
Meola	MSW3	**-0.30**	**-1.76**	NA	**0.79**	NA	0.27
Campbells	C1	-0.04	0.36	**-0.43**	0.04	**1.17**	**0.81**
Campbells	C2	-0.02	**-0.98**	-0.04	0.18	**0.62**	**0.38**
Campbells	C5	**-1.16**	-0.38	0.00	**0.66**	**0.31**	0.05
Torbay	T3	**-0.61**	**-0.83**	**-0.60**	0.05	***0.49***	**0.76**
Torbay	T5	**-0.39**	-0.59	-0.06	0.02	**0.80**	**0.48**
Long Bay	L1	**-0.51**	-0.33	**-3.96**	**0.48**	0.08	**0.86**
Long Bay	L2	-0.01	**-1.01**	**-0.51**	-0.08	-0.04	**0.73**
Long Bay	L3	-0.09	0.00	-0.01	-0.10	0.00	-0.02
Long Bay	L4	**-0.83**	**-0.13**	0.00	**0.84**	-0.01	**-0.14**
Long Bay	L5	**-0.38**	-0.64	0.00	**0.38**	**0.61**	0.00
Manly	M1	**-1.65**	0.01	0.00	***0.06***	**0.10**	**1.56**
Manly	M3	**-1.45**	**-0.50**	-0.25	0.01	**-0.40**	0.07
Manly	M4	-0.22	**-0.95**	**-2.33**	0.07	**0.99**	0.09
Stanmore	S2	**-0.61**	***1.08***	0.00	**0.33**	**0.37**	0.21
Stanmore	S4	0.13	0.25	0.00	**0.29**	0.00	0.02
Stanmore	S5	**-0.78**	***-0.49***	-0.05	0.09	-0.01	**0.24**
Waiwera	W2	**-1.57**	0.09	-0.01	-0.16	**1.50**	**0.32**
Waiwera	W3	**-0.74**	-0.20	-0.01	-0.03	**0.91**	**0.18**
Waiwera	W4	0.16	0.31	0.00	**0.49**	0.00	**0.04**
Leigh	L-ABC	NA	-0.03	-0.03	-0.61	0.00	**1.42**
Leigh	L-NR	NA	0.00	0.06	0.09	NA	**0.04**
Leigh	L-NS	**-0.28**	-0.29	NA	-0.18	0.00	-0.49
Leigh	L-ND	NA	NA	NA	**-0.54**	NA	0.01

Appendix 2c. Linear trends in density (m^{-2}) of common sessile invertebrates from 2007-2025 (except Leigh sites 2013-2024). The slope of the linear regression is given for each site – significant values (*p*<0.05) in bold and marginally significant (*p*=0.05-0.10) in italics. Colour of cell weighted according to the slope, with red indicating a negative slope and blue a positive slope over time.

Area	Site	*Sabella spallanzanii*	*Styela clava*	Ascidian (solitary)	*Tethya burtoni*
Meola	MSE2	**0.19**	-0.01	-0.40	***-0.61***
Meola	MSW2	**0.44**	NA	-0.47	0.08
Meola	MSW3	**0.50**	0.00	0.18	-0.17
Campbells	C1	0.00	NA	-0.03	***-0.05***
Campbells	C2	**0.02**	0.00	**-0.67**	0.00
Campbells	C5	**0.02**	NA	-0.06	-0.08
Torbay	T3	**0.03**	0.02	**-0.30**	-0.01
Torbay	T5	**0.03**	0.02	**-0.88**	***-0.09***
Long Bay	L1	**0.01**	-0.01	**-0.21**	-0.02
Long Bay	L2	**0.05**	**-0.03**	**-0.79**	-0.04
Long Bay	L3	**0.14**	-0.01	**-1.53**	-0.06
Long Bay	L4	0.01	0.00	-0.16	0.02
Long Bay	L5	**0.02**	0.01	-0.10	**-0.14**
Manly	M1	**0.07**	-0.01	**-1.03**	-0.07
Manly	M3	***0.01***	-0.02	**-0.58**	0.00
Manly	M4	**0.01**	0.00	-0.03	0.01
Stanmore	S2	**0.01**	0.00	**0.09**	-0.03
Stanmore	S4	NA	NA	0.03	-0.31
Stanmore	S5	**0.01**	0.00	**-0.12**	**-0.70**
Waiwera	W2	**0.05**	0.00	-0.02	**-0.26**
Waiwera	W3	0.00	NA	-0.02	-0.11
Waiwera	W4	NA	NA	-0.02	-0.29
Leigh	L-ABC	NA	NA	***-0.10***	-0.02
Leigh	L-NR	NA	NA	-0.13	-0.27
Leigh	L-NS	NA	NA	**-0.09**	NA
Leigh	L-ND	**0.14**	NA	-0.08	NA

Appendix 2d. Linear trends in density (m^{-2}) of common gastropod species and total gastropods from 2007-2025 (except Leigh sites 2013-2024). The slope of the linear regression is given for each site – significant values (p<0.05) in bold and marginally significant (p=0.05-0.10) in italics. Colour of cell weighted according to the slope, with red indicating a negative slope and blue a positive slope over time.

Area	Site	*Lunella smaragdus*	*Micrelenchus purpureus*	*Coelotrochus viridis*	*Cookia sulcata*	*Cryptoconchus porosus*	*Buccinulum linea*	*Cominella virgata*	*Maoricolpus roseus*	Total gastropods
Meola	MSE2	**-1.19**	NA	0.00	NA	**-0.02**	NA	NA	NA	**-1.23**
Meola	MSW2	**-0.54**	NA	**-0.01**	NA	**-0.03**	NA	NA	NA	**-0.61**
Meola	MSW3	**-0.47**	NA	0.00	0.00	**-0.04**	NA	NA	NA	**-0.52**
Campbells	C1	**-1.01**	**-0.05**	**-0.25**	0.01	0.00	**-0.06**	**-0.08**	0.01	**-1.47**
Campbells	C2	**-0.08**	**-0.06**	**-0.50**	**-0.02**	0.01	**-0.04**	**-0.18**	**0.11**	**-0.76**
Campbells	C5	**-2.04**	**-0.15**	**-0.91**	0.00	0.00	**-0.03**	**-0.05**	0.01	**-3.20**
Torbay	T3	**-1.04**	**-0.13**	**-0.25**	**-0.08**	0.00	**-0.08**	**-0.04**	-0.11	**-1.78**
Torbay	T5	**-2.31**	**-0.12**	**-0.56**	***-0.01***	***-0.01***	***-0.04***	**-0.06**	-0.02	**-3.16**
Long Bay	L1	**-0.78**	**-0.01**	**-0.12**	-0.01	0.00	**-0.07**	**-0.03**	-0.08	**-1.07**
Long Bay	L2	**-0.51**	0.00	**-0.09**	**-0.01**	**-0.01**	**-0.11**	**-0.15**	0.00	**-0.88**
Long Bay	L3	**-0.12**	***0.00***	**-0.03**	0.00	**-0.07**	**-0.22**	0.00	***-0.08***	**-0.52**
Long Bay	L4	**-0.08**	0.00	**-0.23**	0.01	0.00	**-0.08**	**-0.06**	**0.06**	**-0.37**
Long Bay	L5	**-0.89**	**-0.06**	**-0.11**	-0.01	0.00	-0.02	***-0.03***	0.01	**-1.20**
Manly	M1	**-2.01**	0.00	**-0.03**	0.00	0.00	-0.01	**-0.10**	***-0.21***	**-2.38**
Manly	M3	**-0.40**	0.00	**-0.12**	**-0.07**	-0.01	***-0.05***	**-0.11**	**-0.76**	**-1.54**
Manly	M4	**-0.74**	-0.03	**-0.10**	**-0.06**	-0.01	0.00	**-0.09**	-0.04	**-1.09**
Stanmore	S2	**-1.40**	**-0.09**	**-0.72**	***-0.02***	0.00	**-0.04**	**-0.05**	**0.03**	**-2.28**
Stanmore	S4	**-1.05**	-0.04	-0.33	-0.03	**-0.02**	***-0.02***	0.00	**0.02**	**-1.46**
Stanmore	S5	**-0.18**	**-0.23**	**-0.34**	**-0.18**	**-0.04**	**-0.07**	0.00	**0.03**	**-1.02**
Waiwera	W2	**-1.10**	**-0.24**	**-0.12**	***-0.02***	0.00	**-0.04**	NA	-0.02	**-1.81**
Waiwera	W3	**-0.61**	**-0.79**	0.00	**-0.06**	0.00	**-0.05**	-0.01	**0.05**	**-1.93**
Waiwera	W4	**-0.37**	**-0.23**	-0.25	**-0.06**	NA	-0.01	NA	**0.06**	**-0.88**
Leigh	L-ABC	NA	0.00	-0.06	**-0.39**	NA	-0.01	NA	**0.01**	**-0.52**
Leigh	L-NR	NA	***0.04***	**-0.16**	**-0.15**	NA	NA	NA	***0.06***	**-0.31**
Leigh	L-NS	NA	**-0.13**	-0.02	-0.15	0.00	-0.01	0.00	-0.01	**-0.53**
Leigh	L-ND	NA	-0.01	***-0.08***	-0.01	-0.02	-0.02	NA	0.09	-0.05

Appendix 2e. Linear trends in density (m^{-2}) of common echinoderm species from 2007-2025 (except Leigh sites 2013-2024). The slope of the linear regression is given for each site – significant values (p<0.05) in bold and marginally significant (p=0.05-0.10) in italics. Colour of cell weighted according to the slope, with red indicating a negative slope and blue a positive slope over time.

Area	Site	*Evechinus chloroticus*	*Patiriella regularis*	*Coscinasterias muricata*	*Stegnaster inflatus*	Total echinoderms
Meola	MSE2	0.00	-0.01	0.00	0.00	-0.01
Meola	MSW2	***-0.01***	-0.01	-0.01	NA	-0.02
Meola	MSW3	0.00	0.00	0.00	NA	0.00
Campbells	C1	**0.07**	**-0.04**	0.00	0.00	0.03
Campbells	C2	-0.01	-0.01	**-0.03**	0.00	-0.04
Campbells	C5	**0.02**	***-0.02***	0.00	**-0.07**	**-0.08**
Torbay	T3	**0.12**	**0.03**	-0.01	***-0.01***	**0.12**
Torbay	T5	**0.02**	0.01	0.00	0.00	0.02
Long Bay	L1	0.02	-0.02	**0.04**	0.00	0.04
Long Bay	L2	**0.03**	-0.02	-0.01	0.00	-0.01
Long Bay	L3	0.00	-0.01	**0.08**	0.00	**0.06**
Long Bay	L4	-0.01	**0.07**	-0.01	**0.01**	0.05
Long Bay	L5	**0.21**	0.01	0.00	0.00	**0.22**
Manly	M1	0.00	-0.01	***0.03***	**-0.01**	0.00
Manly	M3	**0.02**	0.00	0.01	***0.01***	**0.04**
Manly	M4	**0.06**	0.01	-0.02	0.01	0.06
Stanmore	S2	0.01	-0.02	-0.02	0.00	-0.03
Stanmore	S4	0.00	**-0.04**	-0.01	-0.01	**-0.06**
Stanmore	S5	-0.02	**-0.06**	-0.01	0.00	***-0.09***
Waiwera	W2	**0.06**	**-0.05**	-0.02	0.00	-0.02
Waiwera	W3	***0.05***	-0.02	0.00	0.01	0.04
Waiwera	W4	**0.03**	**-0.08**	0.00	***-0.03***	**-0.08**
Leigh	L-ABC	**0.09**	NA	NA	NA	***0.04***
Leigh	L-NR	NA	NA	NA	NA	-0.01
Leigh	L-NS	***-0.25***	NA	NA	NA	**-0.32**
Leigh	L-ND	NA	NA	NA	NA	***-0.03***

Appendix 2f. Linear trends in dominant substratum covers (%) from 2007-2025 (except Leigh sites 2013-2024). The slope of the linear regression is given for each site – significant values (p<0.05) in bold and marginally significant (p=0.05-0.10) in italics. Colour of cell weighted according to the slope, with red indicating a negative slope and blue a positive slope over time.

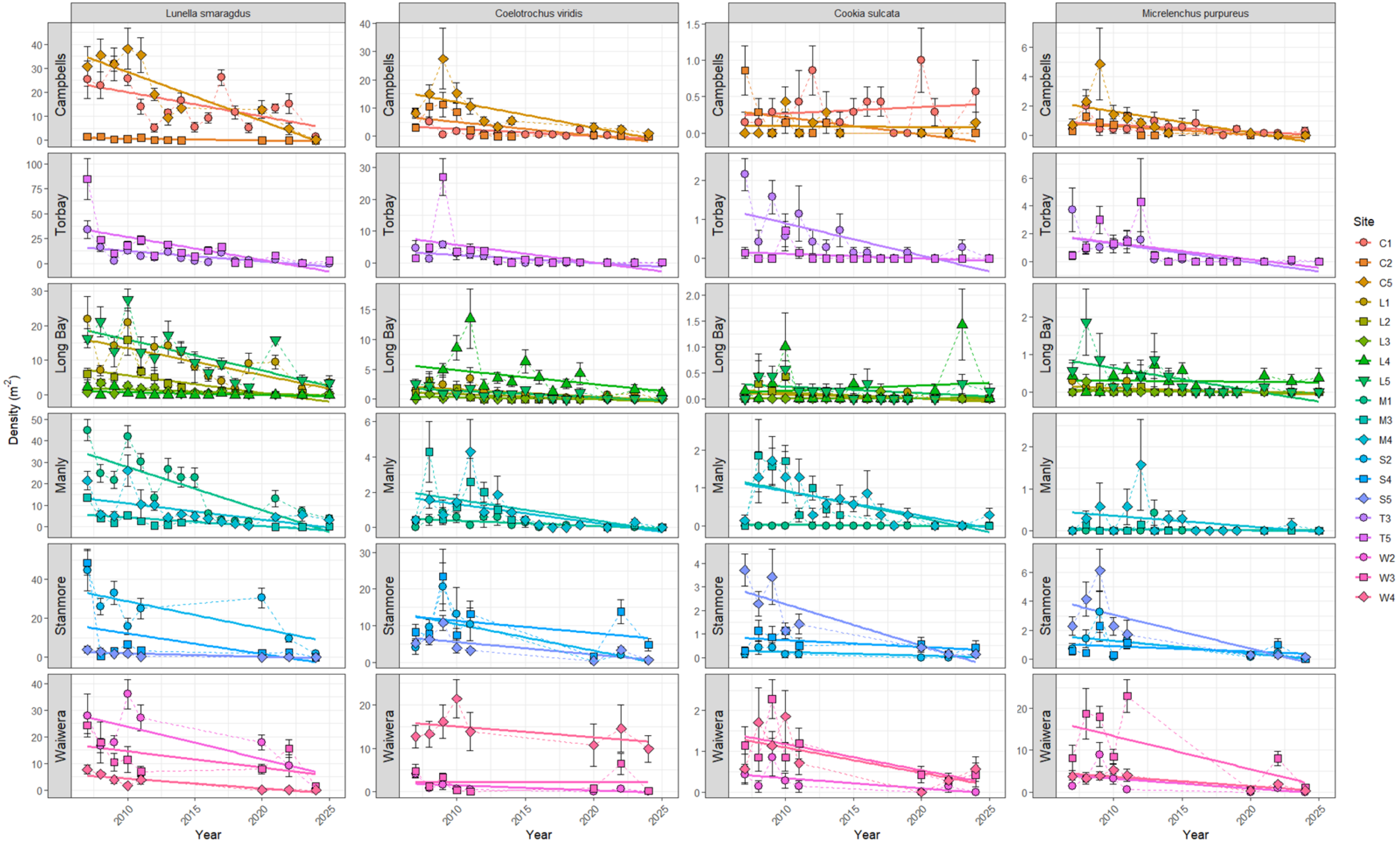
Area	Site	Sediment	CCA	Filamentous algae	Red turf	Coralline turf
Meola	MSE2	-0.59	0.27	**0.36**	**0.11**	-0.07
Meola	MSW2	0.52	-0.19	**0.42**	**0.19**	-0.01
Meola	MSW3	0.66	**-1.08**	**0.57**	**0.26**	***-0.24***
Campbells	C1	**2.19**	**-3.34**	**0.20**	***0.05***	**1.13**
Campbells	C2	**3.14**	**-3.39**	**0.60**	0.06	**0.52**
Campbells	C5	**1.83**	**-1.75**	**0.38**	***0.03***	**0.05**
Torbay	T3	**2.60**	**-3.26**	**0.16**	0.00	**2.17**
Torbay	T5	**1.82**	**-2.57**	**0.36**	**0.04**	**0.88**
Long Bay	L1	0.35	-0.34	**0.82**	***0.03***	0.03
Long Bay	L2	**0.78**	**-1.83**	**0.38**	**0.05**	**1.11**
Long Bay	L3	**1.24**	**-1.74**	**0.19**	**0.29**	**0.11**
Long Bay	L4	-0.12	-0.26	**0.30**	**0.51**	0.01
Long Bay	L5	**0.67**	**-2.18**	**0.54**	0.01	**0.94**
Manly	M1	**1.25**	**-1.06**	***0.29***	***0.01***	0.03
Manly	M3	**1.69**	**-2.11**	**0.42**	**0.06**	**0.27**
Manly	M4	**1.37**	**-1.96**	0.20	**0.04**	**1.22**
Stanmore	S2	**2.16**	**-2.45**	**0.28**	0.01	**0.12**
Stanmore	S4	0.39	***-0.68***	**0.48**	**0.05**	-
Stanmore	S5	**1.58**	**-2.20**	**1.14**	**0.20**	0.00
Waiwera	W2	**1.83**	**-3.41**	**0.16**	0.00	**1.61**
Waiwera	W3	**2.57**	**-3.16**	**0.31**	**0.09**	**0.26**
Waiwera	W4	**1.24**	**-2.23**	**0.64**	**0.17**	-
Leigh	L-ABC	0.05	-0.73	-0.09	**0.47**	**1.50**
Leigh	L-NR	**1.75**	**-3.33**	0.70	**1.25**	**1.17**
Leigh	L-NS	**1.34**	**-2.69**	**1.37**	0.21	**1.72**
Leigh	L-ND	***1.30***	**-0.49**	0.25	-0.01	-0.01

Appendix 2g. Linear trends in cover (%) of dominant sessile invertebrate groups from 2007-2025 (except Leigh sites 2013-2024). The slope of the linear regression is given for each site – significant values (p<0.05) in bold and marginally significant (p=0.05-0.10) in italics. Colour of cell weighted according to the slope, with red indicating a negative slope and blue a positive slope over time.

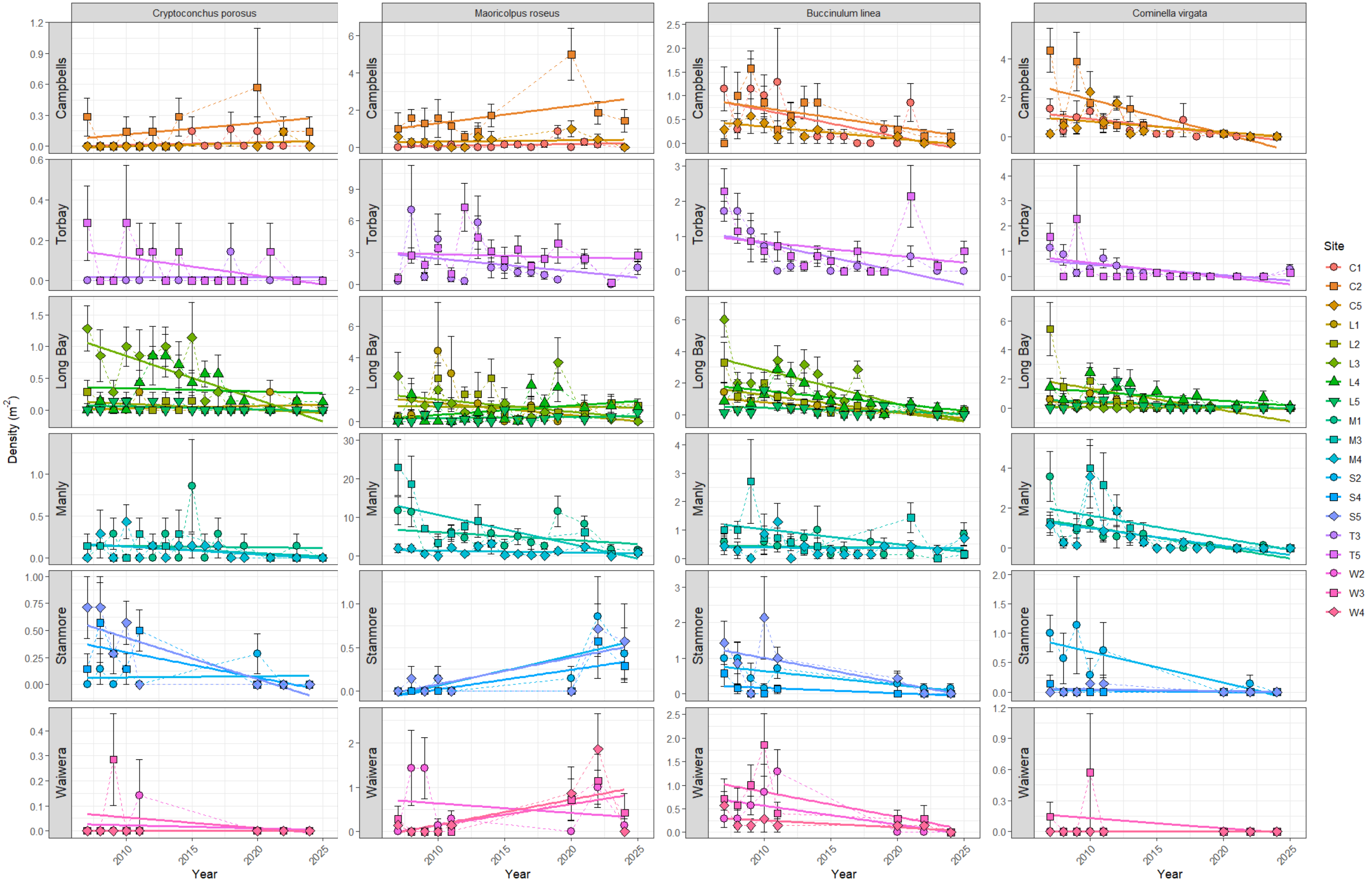
Area	Site	Porifera	Ascidian	Hydroid	Bryozoa
Meola	MSE2	**-0.26**	0.03	0.02	**0.12**
Meola	MSW2	**-0.17**	0.08	0.04	0.01
Meola	MSW3	***-0.14***	***0.09***	0.01	0.03
Campbells	C1	**-0.10**	0.01	0.00	***-0.01***
Campbells	C2	0.06	**-0.02**	0.00	0.00
Campbells	C5	***-0.14***	**0.05**	0.00	0.00
Torbay	T3	0.03	0.01	***0.01***	0.00
Torbay	T5	-0.01	-0.03	0.00	***-0.03***
Long Bay	L1	-0.01	-0.01	0.00	0.01
Long Bay	L2	0.01	**-0.03**	0.01	-0.01
Long Bay	L3	***0.10***	0.00	0.00	0.00
Long Bay	L4	**0.12**	0.02	0.03	-0.01
Long Bay	L5	0.01	0.00	0.00	***-0.01***
Manly	M1	0.02	-0.01	0.00	0.00
Manly	M3	**0.11**	0.03	0.00	0.00
Manly	M4	-0.01	0.02	***0.01***	0.00
Stanmore	S2	0.01	**0.05**	0.00	0.00
Stanmore	S4	0.17	**0.11**	0.00	-0.02
Stanmore	S5	-0.07	**0.04**	0.00	0.00
Waiwera	W2	**-0.14**	0.00	0.00	0.00
Waiwera	W3	-0.09	0.00	0.00	0.00
Waiwera	W4	0.07	-0.01	-0.02	0.00
Leigh	L-ABC	-0.17	**-0.09**	0.01	**-0.03**
Leigh	L-NR	-0.01	**-0.08**	-0.01	**-0.17**
Leigh	L-NS	**-0.17**	**-0.12**	-0.02	-0.01
Leigh	L-ND	-0.08	-0.03	0.00	**-0.06**

8 Appendix 3 – Site-specific trends in key species at East Coast Bays-Whangaparāoa locations

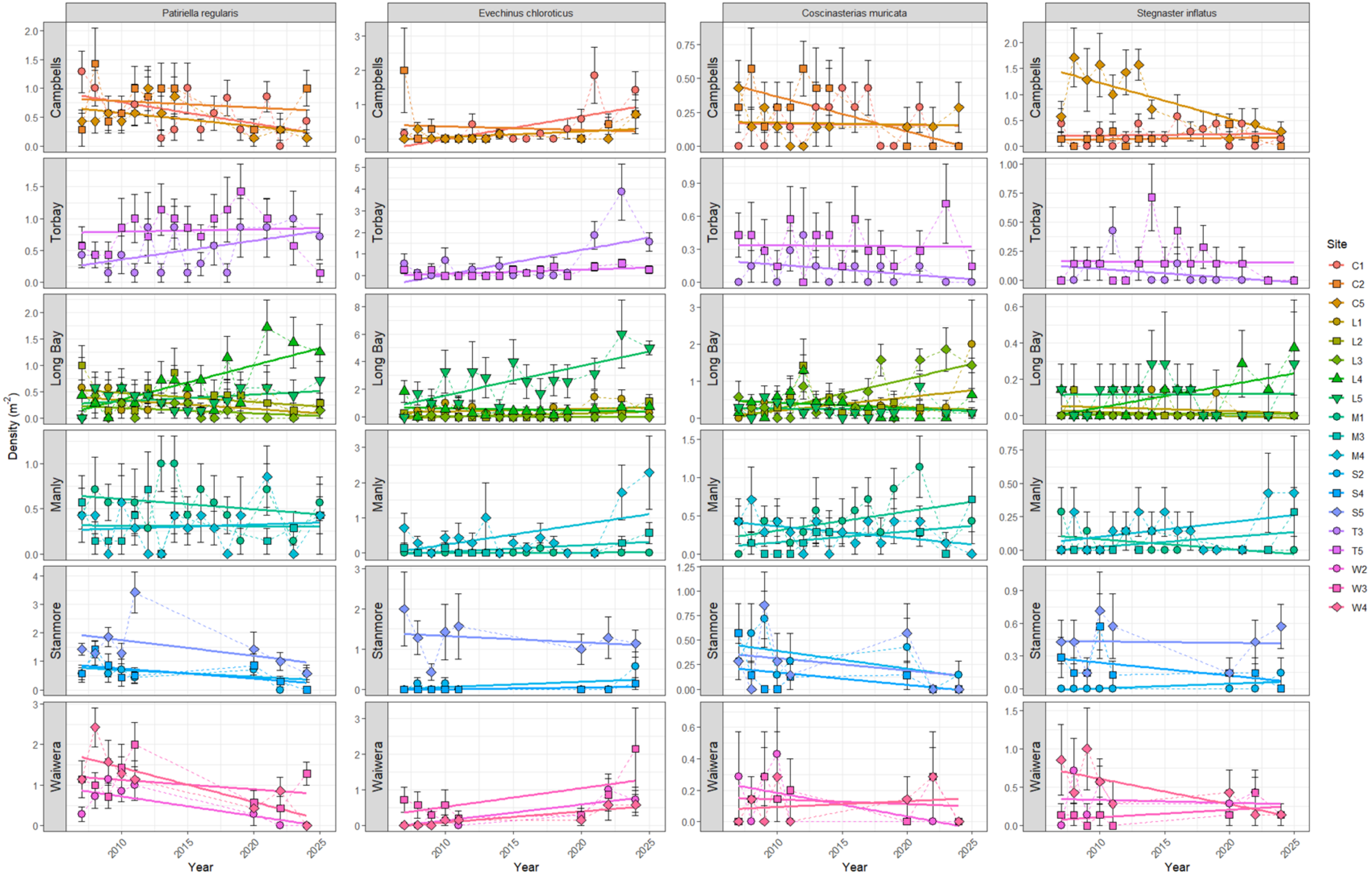
Appendix 3a – Dominant gastropod species at East Coast Bays-Whangaparāoa locations 2007-2025.



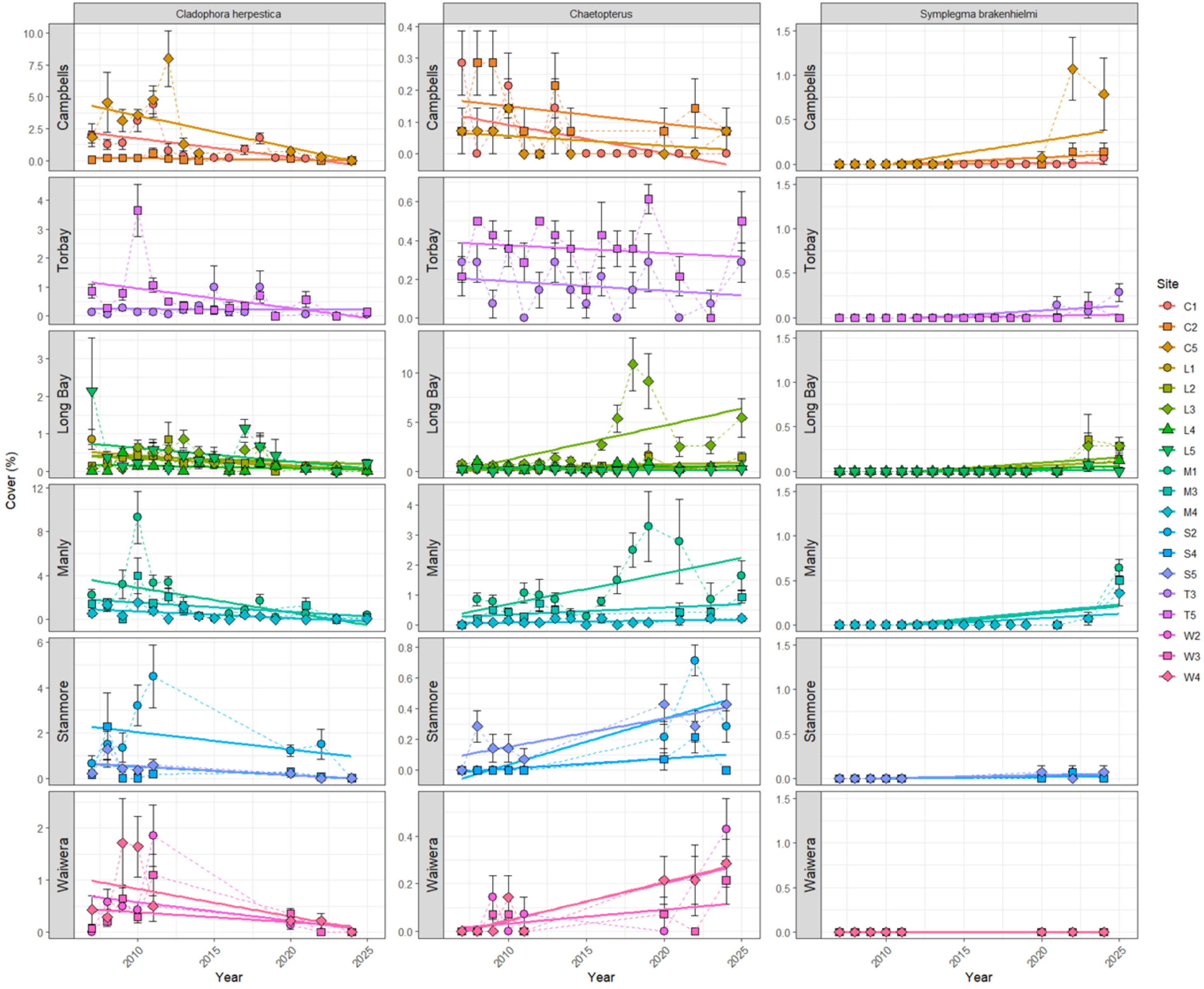
Appendix 3a (Cont.) – Dominant gastropod species at East Coast Bays-Whangaparāoa locations 2007-2025.



Appendix 3b – Dominant echinoderm species at East Coast Bays-Whangaparāoa locations 2007-2025.



Appendix 3c – Covers of key species that have undergone changes at East Coast Bays-Whangaparāoa locations 2007-2025.



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