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New Zealand Macroalgae: Distribution and Potential as National Scale Ecological Indicators

New Zealand Aquatic Environment and Biodiversity Report No. 207

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ISSN 1179-6480 (online)
ISBN 978-1-98-859412-5 (online)

March 2019

New Zealand Government

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D’Archino, R.; Neill, K.F.; Nelson, W.A.; Fachon, E.; Peat, C. (2019). New Zealand Macroalgae: Distribution and Potential as National Scale Ecological Indicators.

***New Zealand Aquatic Environment and Biodiversity Report No. 207.* 217 p.**

Canopy-forming large brown macroalgae are internationally recognised as critical components of coastal ecosystems. This project was initiated to evaluate the use of large brown algae for monitoring change in coastal ecosystems, and their potential as national scale indicators of human-induced change in New Zealand’s marine ecosystems.

The first part of the project was a literature review. International studies have reported on the loss of subtidal macroalgal forests in a number of temperate and subtropical marine ecosystems, particularly on urbanised coasts and, in some cases, the local extinction of *Sargassum* and *Cystoseira* species (e.g Queensland and Mediterranean Sea). The review covers monitoring programmes worldwide that are using macroalgae as indicators of ecosystem health, ecological indices developed on macroalgae (e.g Europe) requiring different level of expertise, and the methods being used for mapping coastal macroalgae, including the use of acoustic and optical remote sensing, videography, underwater and aerial observations. In some countries actions to mitigate changes have been initiated as restoration programmes e.g. in Australia, USA, Japan and Korea, or through conservation measures e.g. *Macrocystis pyrifera* beds, considered an endangered ecological community in Tasmania, Victoria and South Wales since 2012.

The review of New Zealand literature provides a comprehensive update by species of the large brown algae, with a focus on ecological studies on their vulnerability to key stressors (e.g. light, temperature, sediment, nutrient loads, ocean acidification), identifying species that are most likely to be susceptible to environmental changes e.g. *Macrocystis pyrifera* (temperature) *Hormosira banksii*, *Durvillaea* spp. (sediments), *Ecklonia radiata*, *Carpophyllum flexuosum* (sediments and low light).

A questionnaire was sent to scientists and to members of the Local Government Coastal Special Interest Group (C-SIG) involved in ecological research and environmental monitoring, to gather data on monitoring programmes using macroalgae within New Zealand and identify areas where monitoring work could be repeated. However, important knowledge gaps on the extent of macroalgal beds and on species distributions were found, indicating an urgent need to initiate monitoring programmes at a national scale to inform decision makers and, where possible, mitigate changes.

The second part of the project tested the responses of five species of large brown algae to stressors (raised temperature, shading and sedimentation), through laboratory-based culture experiments, focusing on early developmental stages (from release of spores to their settlement and growth). Results indicated species specific responses to the factors being tested: *Landsburgia quercifolia, Lessonia variegata* grew better in higher winter temperatures while *Cystophora* growth was better at current rather than future temperatures. Shading had a significant effect on *L. quercifolia, Cystophora scalaris* and *Marginariella boryana* but did not affect *Lessonia variegata. Carpophyllum maschalocarpum* grew better in shaded conditions.

The third part of the project investigated the potential of monitoring macroalgae as a tool for the assessment of ecosystem health. The distribution of large brown macroalgae in New Zealand was documented using data from herbaria, expert surveys, and from a citizen science project,

established to provide new information in areas where data have been unavailable or scarce. Distributional maps were produced for 29 species of large brown algae. In addition, a range of different mapping approaches were tested, including the use of underwater observations, surface ground-truthing via small vessels, aerial photography, satellite imagery and the use of drones. Historical data on the distribution of the giant kelp *Macrocystis pyrifera* in the Wellington harbour were digitised and compared with the current distribution, testing a range of approaches to mapping. Machine learning and computer vision techniques were investigated to test whether these approaches could be used to identify dominant macroalgal species along the Wellington south coast. The trial of machine learning approaches produced very compelling results and indicated great future potential, with *Ecklonia radiata, Lessonia variegata* and *Carpophyllum* spp., correctly identified from videos streams with different levels of confidence.

Human-induced modification of the coastal zone is clearly evident in New Zealand at a range of scales. Macroalgae are appropriate candidates for use in monitoring programme: they are conspicuous, easily identified, and critical for nearshore ecosystems providing a range of services including productivity and three dimensional habitat space. However, the lack of sound baseline information in relation to canopy forming macroalgae in New Zealand, constrains the evaluation of the impacts of environmental change on these algae and the effectiveness of management methods. A standardised approach to the documentation of species distribution and a national repository of verified data in agreed forms would improve accessibility and the utility of such data. Experimental studies have shown that species of large brown algae will respond in species specific ways to the suite of stressors most likely to be encountered in coastal waters of New Zealand, namely rising temperatures, sedimentation and light restrictions. Further work would enable the selection of regional and site appropriate species for on-going monitoring. New technologies have significantly increased the options for more accessible and cost-effective approaches to monitoring coastal systems.

1 INTRODUCTION

Over the past two decades there has been increased international attention on coastal environments and the ways in which human-induced pressures are resulting in changes to coastal ecosystems. This has led to demand for (i) robust methods to separate natural variation in ecosystems from that caused by human activities which could potentially be managed, and (ii) reliable and cost-effective indicators of such change. In New Zealand, the Environmental Reporting Act (2016) and associated requirement for comprehensive State of the Environment reporting has added urgency to these needs. This project was initiated to evaluate the use of large brown algae, both internationally and nationally, for monitoring change in coastal ecosystems, and their potential as national scale indicators of human-induced change in New Zealand's marine ecosystems.

The importance of macroalgae in coastal marine systems has long been recognised, with kelp forests recognised as producing the largest biogenic structures found in marine systems (Dayton 1985) and collectively forming one of the most productive and diverse ecosystems in the world (Mann 1973, Smale et al. 2013). Canopy-forming large brown algae provide three dimensional structures for fish and invertebrate species and create habitat for other algae. Because of their central role in a range of ecological processes on temperate reefs and adjacent habitats, the loss of canopy-forming algae is likely to be associated with a significant loss of associated species and ecological function (Edgar et al. 2004, Teagle et al. 2017). Indeed, the loss of habitat provided by canopy forming large brown algae can drive ecosystem changes comparable to those mediated by environmental stressors such as warming or increased ultraviolet radiation (Hooper et al. 2012).

Bennett et al. (2016) emphasises, not only the ecological values of macroalgal forests of the Great Southern Reef in Australia, but also the social and economic importance of kelp in this region. Raven (2017) reviewed the roles played by algae, including benthic macroalgae in the sequestration of CO_2 as organic carbon, and the potential offsets to increases in atmospheric CO_2 and global temperature, that carbon uptake by algae could provide. Although marine macrophytes were identified as a global carbon sink almost 40 years ago (Smith 1981), the role of macroalgae in carbon sequestration (blue carbon) has received little attention until the past decade (e.g., Chung et al. 2011, Chung et al. 2013, Duarte 2017, Hill et al. 2015, Krause-Jensen & Duarte 2016, Sondak et al. 2017, Trevathan-Tackett et al. 2015). There is increasing evidence that macroalgae may be significant for long-term carbon storage through export of material to the deep sea and as refractory, taxon-specific compounds in sediments.

Recent reviews have focused on various drivers of change and their potential impacts on macroalgae, particularly canopy forming species of brown algae in the orders Laminariales (true kelps) and Fucales. These reviews explore the consequences of numerous drivers of change, including climate change (e.g., Araujo et al. 2016, Boyd et al. 2016, Brodie et al. 2014, Cornwall et al. 2012, Harley et al. 2006, Hepburn et al. 2011, Johnson et al. 2011, Law et al. 2017, Lundquist et al. 2011, MacDiarmid et al. 2012, Merzouk & Johnson 2011, Mineur et al. 2015, Möllmann et al. 2015, Russell et al. 2011, Smale et al. 2013, Tegner & Dayton 1987, Vergés et al. 2014, Wernberg et al. 2016b, Xiao et al. 2015), fishing (e.g., Filbee-Dexter & Scheibling 2014, Ling et al. 2009, Salomon et al. 2008, Steneck et al. 2002), and invasive species (e.g., Jiménez et al. 2015a, 2015b, Mellin et al. 2016).

Loss of subtidal macroalgal forests has been documented in a number of temperate and subtropical marine ecosystems, particularly on urbanised coasts, e.g., the loss of Fucales along the Albères coast, Mediterranean Sea (Thibaut et al. 2015), losses of *Sargassum* species from the Sunshine Coast, Queensland, and *Phyllospora comosa* (Labill.) C.Agardh around Sydney, NSW (Campbell et al. 2014, Coleman et al. 2008), the decline of *Ecklonia radiata* (C.Agardh) J.Agardh around Adelaide, South Australia (Connell et al. 2008) and the giant kelp *Macrocystis pyrifera* (L.) C.Agardh around eastern Tasmania (Johnson et al. 2011), and the retraction in the range of kelps in Western Australia after an extreme heat event (Wernberg et al. 2016a). A number of studies have considered European shores (e.g., Bartsch et al. 2015, Merzouk & Johnson 2011, Yesson et al. 2015b). Understanding the susceptibility of species to extreme events is still far from complete. Some recent studies on *Macrocystis*, for example, suggested that this species may have physiological capacity to cope with the wide range of environmental conditions experienced across its large distributional range (e.g., Buschmann et al. 2014a, Reed et al. 2016).

A number of international initiatives have been implemented using macroalgae, especially fucoids, as biological indicators of system health, particularly in European monitoring programmes, e.g., the state of *Cystoseira* (Fucales) beds has been used to assess the ecological quality of the coastal water bodies in the European Water Framework Directive (2000/60/EC) (Ballesteros et al. 2007b, Pinedo et al. 2007). Various indices have also been applied to the Mediterranean rocky shores where a high ecological status is defined for the communities dominated by the structuring *Cystoseira* species, while a poor ecological status is for shores dominated by less complex species such as Ulvales, Bangiophycidae or cyanobacteria (Ballesteros et al. 2007a, Orfanidis et al. 2001, Sales & Ballesteros 2009).

Research objectives:

This project is comprised of three key objectives:

Objective 1. Provide summaries of

a) the national and international literature on the use of macroalgae in monitoring programmes; and
b) the data available for laminarians and fucoid algae in New Zealand with respect to use in monitoring programmes, distributional data and response to environmental change.

Objective 2. Establish the susceptibility of selected New Zealand laminarians and fucoids to selected environmental stressors.

Objective 3. Establish the utility of selected macroalgae as monitoring tools for the assessment of ecosystem health, and test methods for mapping distribution for baseline monitoring.

a) monitor previously well documented sites repeating similar field protocols
b) evaluate the best (cost-effective) method to map kelp forests to establish reliable baselines
c) map current known distribution of laminarians and fucoids on national scale (presence/absence)

Each objective is addressed as a separate section within this report.

2 LITERATURE REVIEW AND AVAILABLE DATA

2.1 Introduction

This section addresses objective 1 of the project. National and international literature addressing trends in macroalgal abundance and distribution, macroalgal monitoring, and whether the use of macroalgae as indicators of ecosystem health has been found to be useful, is summarised. A descriptive list of New Zealand large brown algae is provided and the results of a survey of the New Zealand research and resource management community is provided to summarise current monitoring practices applied nationally.

2.2 Methods

A detailed literature search was carried out to locate national and international literature on the use of macroalgae as biological indicators, the monitoring and mapping of macroalgal beds, and the effect of climate change and other stressors on macroalgae. The databases searched included standard marine bibliographic sources (Science Direct, Web of Science, Wiley, Google scholar), and also web sites of marine research organisations.

The literature search was made using a combination of key words including: kelp, macroalgae, macroalgae bed, Laminariales, Fucales, *Macrocystis*, canopy-forming algae, large brown, loss, decline, species loss, turfing algae, ecological indicators, indices, monitoring, mapping, modelling, predict modelling, climate change, sedimentation, ocean acidification, productivity, temperature, nutrient, New Zealand, satellite, drone/AUV, single beam, multibeam, side scan sonar, Lidar, videography.

The New Zealand literature was searched by key words and by authors and included references from 1921. Universities databases were searched for theses (Masters and PhD). New Zealand references also included unpublished data such as Department of Conservation reports and personal communications.

The literature obtained was entered into an EndNote database and organised in folders by subject to enable subsequent summaries and analysis. The references, organised by authors, year, title, keywords and reference type were exported in excel, to produce a graph of the references gathered by year. The literature search ended on 28 February 2018.

A summary of New Zealand large brown algae was assembled based on published data, covering both Laminariales (true kelps) and Fucales.

Questionnaires (Appendix 1) were used to gather data on monitoring programmes within New Zealand using macroalgae, targeting:
a) the members of the Local Government Coastal Special Interest Group (C-SIG) involved in environmental monitoring, and,
b) expert scientists and science teams involved in ecological research with macroalgae.

2.3 Results

A total of 1915 scientific papers were located using the search terms described in the methods (Figure 1).

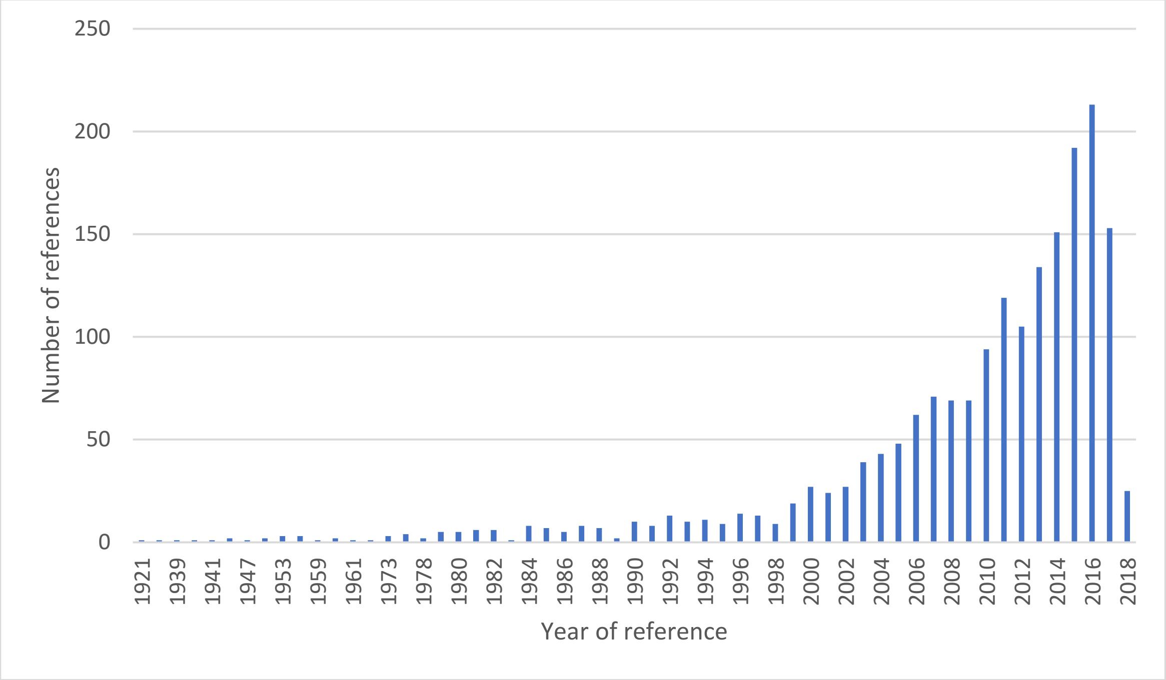

Figure 1: The number of papers published in each year considered during this study.

2.3.1 Monitoring of macroalgae: international trends and monitoring programmes

2.3.1.1 Changes in the abundance and distribution of macroalgae

Global

A global study of trends in changes in kelp forests over the past half-century, detected a high degree of geographic variation with region-specific differences (Krumhansl et al. 2016). The analysis was based on data from 34 of the 99 global ecoregions where kelps exist, and identified declines (38 %), increases (27 %) and no detectable change (35 %) in different ecoregions. Using all available data across the greatest time span for each ecoregion, a high probability of decline (at least 95 %) in kelp abundances was estimated for Central Chile, the Aleutian Islands, the South Australian Gulf, the North Sea, North-Central California and the Bassian ecoregion. Data used from New Zealand in this analysis were restricted to studies from northeastern New Zealand and included only data for *Ecklonia radiata* and *Lessonia variegata* beds, which showed an increasing trend in abundance (Krumhansl et al. 2016). Data from southern New Zealand species (e.g., *Macrocystis pyrifera*) were not included in the analysis. This analysis was based on diver-collected data, and did not include any data from the Mediterranean, and in the northwest Pacific only three sites were included (of the 1454 in the analysis).

Europe

The loss of species, particularly members of the Fucales, has been reported along European coasts. On the Albères coast (Mediterranean Sea) only five species of Fucales remain of the fourteen species reported from the early 19th century, and of the nine species reported in the 1980s (Thibaut et al. 2005). Species of *Sargassum* have become extremely rare or locally extinct along 2970 km of north-western Mediterranean coasts, while the invasive *S. muticum* (Yendo) Fensholt has developed large populations. Stressors that have been linked with the decline of *Sargassum* species include decrease in water clarity, pollution, overgrazing, fishing nets, trawling, loss of habitat and increase in water temperature (Thibaut et al. 2016).

Cystoseira species are also in decline in the Mediterranean Sea and have completely disappeared from some regions. *Cystoseira crinita* Duby is regarded as 'extinct in French Catalonia, near-extinct in Languedoc and western Provence, and functionally extinct on the French Riviera coast', while populations have remained stable in the eastern Provence and Corsica (Blanfune et al. 2016b). A drastic reduction in populations of *Cystoseira abies-marina* (S.G.Gmel.) C.Agardh., a key species for the ecosystems of the Canary Islands, has been documented even in areas with low anthropogenic pressure; in the 1980s a continuous belt extended along 120.5 km of the coastline and occupied 928 ha, and today, this species is found along 37.8 km of the coastline and occupies only 7.4 ha, mainly as scattered patches (Valdazo et al. 2017). *Cystoseira brachycarpa* J.Agardh, *C. sauvageauana* Hamel, *C. spinosa* Sauv., *C. zosteroides* (Turner) C.Agardh, *Sargassum acinarium* Mont. and *S. trichocarpum* J.Agardh, which are characteristic of the Mediterranean Sea photophilic communities on hard substrata, are sensitive to environmental changes and have disappeared from Linosa Island (Sicily, Italy). *Cystoseira* communities were replaced by species producing a lower structural complexity (e.g., species of Dictyotaceae, Sphacelariaceae, Udoteaceae) explaining the decline in biodiversity that has also been observed (Serio et al. 2006). When *Cystoseira* belts decreased in the western Mediterranean Sea, turf forming algae or mussel beds became dominant (Rodríguez-Prieto & Polo 1996, Soltan et al. 2001, Thibaut et al. 2005). On the Istrian Coast (Adriatic Sea) the reduction in *Cystoseira* populations corresponded to increased levels of eutrophication from river outflows, and an associated increase in sea urchin populations (1978–1997), and later a recovery in the *Cystoseira* populations was observed (1999–2013) associated with decreases in nutrient levels in rivers and coastal waters, and associated declines in sea urchins (Iveša et al. 2016).

In the British Isles, an assessment of large brown seaweeds was undertaken using historical survey data, from the 18th century to the present day, to determine whether changes in abundance were linked with climate change. There were both positive and negative trends, with significant declines in the southern regions for kelp species and increases in northern and central areas for some kelp and wracks (Fucales). Temperature alone did not explain these changes and the authors pointed to the need to identify both physical (e.g., increased storminess, turbidity and pollution) and biological factors that are driving these changes, including the need to understand the different phases of the life history of kelp species (Yesson et al. 2015b). Despite recent evidence that "unequivocally demonstrates that the structure of kelp forests in the NE Atlantic is changing in response to climate- and non-climate-related stressors, which will have major implications for the structure and functioning of coastal ecosystems", the kelp-dominated habitats in this region have been "chronically understudied over recent decades in comparison with other regions such as Australasia and North America" (Smale et al. 2013).

The decline in abundance of the subtidal *Laminaria digitata* (Huds.) J.V.Lamour. has been observed along French coasts (Cosson 1999, Davoult et al. 2011). Projections of the future range of *L. digitata* throughout the 21st century, taking into account several environmental and biological variables, showed large shifts in the suitable habitat for this species and suggested the likely disappearance of *L. digitata* in some countries (e.g., France) as early as the 2050s depending on the magnitude of global warming (Raybaud et al. 2013). Change in distribution, from sheltered to moderately exposed open coastlines, and increased abundance have been observed for the warm-water kelp *Laminaria ochroleuca* Bach.Pyl. at its poleward range edge (the southwest coast of the UK) where it now co-exists and competes with the cold-water species *Laminaria hyperborea* (Gunnerus) Foslie (Smale et al. 2015).

The decline of the sugar kelp, *Saccharina latissima* was observed in 2002, along the coast of southern Norway and later surveys showed a large-scale shift from sugar kelp forests to communities dominated by filamentous, ephemeral macroalgae (Moy & Christie 2012). A phase shift, from kelp beds to rocky reefs dominated by opportunistic turf-forming and invasive algae, was also observed in Nova Scotia where the kelp biomass has declined by 85−99 % over the past 40–60 years (Filbee-Dexter et al. 2016). Declines in other habitat-forming species have also been seen in Europe. For example, the bladder wrack, *Fucus vesiculosus* L., formed extensive belts before the 1980s, down to depths of about 10 m along the SW coast of Finland and by 2007, six out of eleven sites had lost their *Fucus* belts, with complete disappearance from three sites. The decline in *F. vesiculosus* abundance was estimated to be over 81 % (Vahteri & Vuorinen 2016).

Australia

Decline, losses or shifts in distribution of habitat-forming species have been recorded in Australia and, as in other regions worldwide, these have been attributed to anthropogenic stressors and climate change. In Tasmania *Macrocystis* kelp beds were once sufficiently large to be commercially harvested but have declined by approximately half along the east coast since 1944 (Edyvane 2003). A further study of the extent of these *Macrocystis* beds, based on a time series of aerial photographs (1946–2007), showed an average canopy extent in the last decade of about 9 % of the average canopy extent in the 1940s. The declines were estimated to be up to 95–98 % in some locations. In general, declines were less pronounced in southern regions of Tasmania than in northern and eastern regions. The decline in *Macrocystis* has been linked with the progressive southward penetration of warm, salty, nutrient-poor EAC water, which has now extended about 350 km further south than 60 years ago (Johnson et al. 2011). Because of this decline and the ecosystem functions that this species provides, *Macrocystis* forests were declared an endangered ecological community under the federal Environmental Protection and Biodiversity Conservation Act 1999 in 2012. *Macrocystis* forests were evaluated against four criteria and judged vulnerable for criterion 1 (decline in geographic distribution) and criterion 2 (small geographic distribution coupled with demonstrable threat), and, endangered for criterion 3 (loss or decline of functionally important species) and criterion 4 (reduction in community integrity) (*Listing Advice on Giant Kelp Marine Forests of South East* from http://www.environment.gov.au/cgi-bin/sprat/public/publicshowcommunity.pl?id=107).

Declines of subtidal habitat-forming species have been observed in major metropolitan cities in Australia. Historical evidence shows that dense canopies of *Phyllospora comosa* (Seirococcaceae, Fucales), a species endemic in temperate regions of Australia, were prominent around Sydney a half century ago, but have not been found recently along 70 km of urbanised rocky coastline in Sydney’s metropolitan region (Coleman et al. 2008). The disappearance of *Phyllospora* was coincident with heavy sewage outfall discharges during the 1970s and 1980s. Despite improved water quality in Sydney harbour, and protection of its habitat, recolonisation has not occurred. It was suggested that competition, particularly with algal turfs or *Sargassum* spp., may inhibit the recruitment of *Phyllospora* (Coleman et al. 2008). Experimental restoration has been carried out with adult thalli of *Phyllosphora* transplanted in two rocky reefs where it was once abundant. Survival of transplanted plants varied between sites suggesting that in some places the conditions were suitable for survival and reproduction (Campbell et al. 2014).

The loss of *Ecklonia radiata* along Adelaide’s metropolitan coast since major urbanisation occurred has been estimated to be up to 70 %, and this decline was considered to have been driven by climate and local anthropogenic stressors such as reduced water quality (Connell et al. 2008). A shift in distribution has also been recorded for *Scytothalia dorycarpa* (Turner) Grev., (Seirococcaceae, Fucales), another important habitat-forming species in temperate Australia. Its northern range limit has contracted about 100 km in less than a year, in response to an extreme heat event (Smale & Wernberg 2013). Reef communities became dominated by persistent seaweed turfs after the loss of kelp forest, and temperate species were replaced by species with subtropical and tropical affinities (Wernberg et al. 2016a). Species loss also occurred on the subtropical Sunshine Coast (Queensland) where five species of *Sargassum*, common before urbanisation began in 1970, were not found during a survey between 2004 and 2006. *Sargassum* beds were replaced by bare substratum, stress-tolerant algal turfs, or smaller foliose algae which resulted in a marked decrease in vegetation heterogeneity and complexity (Phillips & Blackshaw 2011).

South Africa

In South Africa, historical and recent evidence demonstrate that the eastern limit of the major kelp-bed forming seaweed *Ecklonia maxima* (Osbeck) Papenf. has moved about 73 km eastward along the south coast since 2006, after remaining unchanged for almost 70 years (Bolton et al. 2012). The authors hypothesised that the eastward spread is limited by aspects of the inshore water temperature regime, and recent evidence suggests that gradual cooling along this coast may have caused the change in distribution. The expansion northwards of *Ecklonia maxima* was also confirmed by using repeat photography to document long-term changes. (Reimers et al. 2014).

Japan

In Japan the reduction of seaweed beds, called ‘isoyake’, has become a serious problem in most coastal prefectures, although this phenomenon was first recorded in 1830 (Fujita 2010, 2011). *Sargassum* beds have declined drastically along the coast of Japan since 1960s, with more than 100 000 ha of coastal land reclaimed for use as new commercial and industrial areas. The shift in the distribution range of some species has been shown to be related to increased water temperature, for example, temperate *Sargassum* species declined in Southern Japan and were replaced by tropical *Sargassum* species (Haraguchi & Sekida 2008, Tanaka et al. 2012). Restoration techniques have been tested in Hiroshima Bay (Terawaki et al. 2003).

Korea

Forests of *Sargassum*, *Laminaria* and *Ecklonia* were abundant all along the Korean Peninsula (except in the coastal zone of the Yellow Sea where the seabed comprises mud) but have been decreasing from the beginning of 1990s. By the end of 2004, about 13 % of the East Sea and 31.4 % of Jeju Island in the South Sea had become barren ground. This decline has been attributed to various causes, such as global warming, sea urchin grazing, industrialization near the shores, and over releasing of abalone. Different techniques have been applied to restore seaweed beds, including spore-bags which have been used successfully for the re-establishment of *Sargassum* beds, rope-seeding for *Laminaria* beds, and transplantation of adult plants of *Ecklonia* and *Laminaria* (Kang 2010).

A summary of the responses of large brown algae to anthropogenic and environmental stressors is provided in table 1.

Table 1: Responses of large brown algae to anthropogenic and environmental stressors.

Species/habitat	**Changes**	**Country**	**Reference**
Kelp forests	38% decline 27% increase 35% no changes	Worldwide 34 Ecoregions	Krumhansl et al. 2016
Kelp forests	Decline up to 95 % in abundances	Central Chile, Aleutian Islands, South Australian Gulf, North Sea, North-Central California, Bassian ecoregion	Krumhansl et al. 2016
Fucales	9 species extinct	France Mediterranean Sea	Thibaut et al. 2005
Sargassum spp.	Extremely rare but increase of *S. muticum* (invasive)	France 2970 km of north-western Mediterranean	Thibaut et al. 2016
Cystoseira crinita	Extinct in some French regions and stable in Corsica	France	Blanfune et al. 2016b
Cystoseira abies-marina	Drastic reduction from 928 ha in 1980 to 7.4 ha	Canary Islands	Valdazo et al. 2017
Cystoseira spp.	5 species disappeared	Linosa Islands, Italy	Serio et al. 2006

Species/habitat	**Changes**	**Country**	**Reference**
Sargassum spp			
Cystoseira spp.	*Cystoseira* belts decreased	Western Mediterranean Sea	Rodríguez-Prieto & Polo 1996, Soltan et al. 2001, Thibaut et al. 2005
Cystoseira spp.	Reduction in *Cystoseira* 1978-1997 (increased nutrients and sea urchins) Recovery in 1999–2013 (decreases in nutrient levels and declines in sea urchins)	Istrian Coast (Adriatic Sea)	Iveša et al. 2016
Kelp species	Declines in the southern regions and increases in northern and central areas for some kelp and wracks	British Isles	Yesson et al. 2015b
Laminaria digitata	Decline in abundance	France	Cosson 1999, Davoult et al. 2011
Laminaria ochroleuca	Increased abundance at its poleward range edge where it now co-exists and competes with the cold-water species *Laminaria hyperborea*	the southwest coast of the UK	Smale et al. 2015
Saccharina latissima	Loss and shift from sugar kelp forests to communities dominated by filamentous, ephemeral macroalgae	Southern Norway	Moy & Christie 2012
Kelp beds	Kelp biomass declined by 85−99 % over the past 40-60 years. Shift, to opportunistic turf-forming and invasive algae	Nova Scotia	Filbee-Dexter et al. 2016
Fucus vesiculosus	Decline over 81 % of extensive belts from 1980s, to 2007 (6 out of 11 sites had lost their *Fucus* belts with complete disappearance from three sites	SW coast of Finland	Vahteri & Vuorinen 2016
Macrocystis pyrifera	Declines up to 95–98 %	Tasmania	Edyvane 2003 Johnson et al. 2011
Phyllospora comosa	Disappeared during 1970s and 1980s (heavy sewage outfall discharges)	Sydney 70 km of urbanised coastline	Coleman et al. 2008 Campbell et al. 2014
Ecklonia radiata	Decline up to 70 %,	Adelaide's metropolitan coast	Connell et al. 2008
Scytothalia dorycarpa	Northern range limit contracted about 100 km in less than a year, in response to an extreme heat event	Western Australia	Smale & Wernberg 2013 Wernberg et al. 2016a
Sargassum spp.	5 species lost between 1970 and 2004–2006. *Sargassum* beds	Sunshine Coast (Queensland)	Phillips & Blackshaw 2011
Ecklonia maxima	Eastern limit moved about 73 km eastward along the south coast since 2006, after remaining unchanged for almost 70 years	South Africa	Bolton et al. 2012 Reimers et al. 2014
Sargassum beds	Declined drastically since 1960s, with more than	Japan	Fujita 2010, 2011, Haraguchi & Sekida

Species/habitat	Changes	Country	Reference
	100 000 ha of coastal land reclaimed		2008, Tanaka et al. 2012
Sargassum, Laminaria and *Ecklonia* forests	Decreasing from the 1990s. In 2004, about 13 % of the East Sea and 31.4 % of Jeju Island in the South Sea became barren ground	Korea	Kang 2010

2.3.2 Mapping of macroalgae

A common method for observing changes over time in macroalgal beds is through monitoring the distribution and extent of beds. Many approaches have been employed to measure and map the extent of macroalgal beds, ranging from various types of field surveys (intertidal and subtidal) and direct observations including structured sampling programmes involving measurements of presence/absence, cover/abundance, etc. In some cases, photographic methods, including video, have been used. A range of remote methods have been employed, both optical and acoustic. A summary of literature relevant to this project is presented in Appendix 2, covering international and national reports on macroalgal mapping, the purpose of the mapping and the types of approaches employed.

Background information is given in Appendix 3 about mapping techniques including remote sensing approaches, and how satellite imagery has been applied to monitoring and mapping kelp forests and subtidal macroalgal beds. Appendix 4 presents examples of studies where combined approaches have been employed.

The applications of mapping and monitoring methods are seen in the following sections which summarise their use in the development of indices, and also in a range of field programmes, sometimes incorporating citizen science, and through the use of remote sensing technologies.

The relevant literature on the range of methods used to map macroalgal beds was summarised, to evaluate the best methods appropriate for applying in New Zealand (Table A3.1).

2.3.2.1 Macroalgae indices and monitoring programmes developed in other regions

Europe

Monitoring tools have been developed in Europe to meet the requirements of the Water Framework Directive (EC. 2000) and then the Marine Strategy Framework Directive (MSFD) (EC. 2008, 2013), both aimed at maintaining and improving the ecological status of marine coastal waters. Macroalgae were included as biological quality indicators to characterise the ecological status of intertidal and subtidal coastal habitats. Reference criteria for macroalgae outlined in the Directives are 1) the taxonomic composition corresponding totally or nearly totally with undisturbed conditions, 2) no detectable changes in macroalgal abundance due to anthropogenic activities (Wells et al. 2007).

In the last decade a plethora of indices, summarised in Table 2, has been developed to assess the ecological quality of coastal marine ecosystems based solely on macroalgae, or on specific habitat (e.g., coralligenous, seagrass) or considering the whole ecosystem or based on invasive species. Several studies indicate that *Cystoseira* (Fucales) species are indicators of good water quality, in contrast to opportunistic or ephemeral species indicating poor quality e.g., dominance of Ulvales, Bangiophycidae or Cyanobacteria (Arévalo et al. 2007, Benedetti-Cecchi et al. 2001, Falace et al. 2010, Panayotidis et al. 2004, Pinedo et al. 2007, Sales & Ballesteros 2009, Soltan et al. 2001).

The CARLIT (CARtography of LITtoral and upper-sublittoral rocky-shore communities) index has been officially recognised as an institutional monitoring tool in Spain since 1999 (Ballesteros et al. 2007b), in Italy since 2004 (Mangialajo et al. 2007) and in France since 2006 (Asnaghi et al. 2009). This index is based on the degree of development of *Cystoseira* spp. littoral belts on exposed coasts (Ballesteros et al. 2007b) and is widely applied in the Mediterranean Sea. The sampling survey consists of a visual assessment from a small boat, driven as close as possible to the shoreline, to detect the dominant macroalgal community along the upper infralittoral rocky shore. The result is a partition of the rocky shoreline into several sectors, each one characterised by a community category (corresponding to a single community or combination of communities) (Ballesteros et al. 2007b). Several studies have assessed and implemented the CARLIT (Blanfune et al. 2011, Blanfune et al. 2017, Cavallo et al. 2016, De la Fuente 2015, Jona Lasinio et al. 2017) and this method has been applied to various regions in the Mediterranean Sea, for example in Albania (Blanfune et al. 2016a), along the Lebanese coastline (Badreddine et al. 2018), in four Tyrrhenian Islands (Jona Lasinio et al. 2017), in the Adriatic Sea (Nikolić et al. 2011, 2013, Sfriso & Facca 2011), and in the Alboran Sea (European Coast) (Bermejo et al. 2013). The distribution of *Fucus virsoides* in the Gulf of Trieste was assessed with the CARLIT method (Orlando-Bonaca et al. 2013). A simplified CARLIT method has been recently suggested (Blanfune et al. 2017).

Northern Europe

The ALGAMONY project was developed to bring Nordic macroalgae monitoring scientists together to harmonise monitoring methodologies applied in Finland, Norway, Sweden and Denmark and to agree on common metrics that, without extensive cost, could be implemented in the different national monitoring programmes. Two metrics were chosen for further common work: total cover of erect macroalgae species and the lower depth distribution limit of selected macroalgae. A summary of the methodology applied in each country is given in the Nordic intercalibration of hard bottom macroalgae monitoring methodologies (Moy et al. 2010).

NI (Nature Index) developed in Norway for coastal water, includes a total of 65 indicators covering algae, invertebrates, fish, seabirds and sea mammals. The index assessed that the overall status of coastal waters was good, with a slight worsening from 1990 and 2000 to 2010. The benthic status had deteriorated due to reductions in kelp, invertebrates and fish. It is proposed that the NI become a permanent tool for use in Norwegian nature management and political planning, with yearly updates (Oug et al. 2013). The Norwegian Programme for mapping and monitoring of marine biodiversity began in 2003 and integrates data on habitat and species distributions in coastal areas (Bekkby et al. 2013). The Norwegian Climate and Pollution Agency in 2003 funded a mapping and monitoring programme of the sugar kelp, *Saccharina latissima*, suggesting different actions for restoring kelp forests and associated biodiversity (Bekkby et al. 2013).

Table 2: European Indices based only on macroalgae or which include macroalgae

Indices	Acronym	Macroalgae/ecosystems	Country	Reference
CARtography of LITtoral and upper-sublittoral rocky-shore communities	CARLIT	*Cystoseira* spp. belt *Fucus virsoides*	Spain, Italy, France, Albania, Lebanon, Tyrrhenian Islands, Adriatic Sea, Alboran Sea	Ballesteros et al. 2007b; Mangialajo et al. 2007; Asnaghi et al. 2009. Blanfune et al. 2016a; Badreddine et al. 2018; Jona Lasinio et al. 2017; Nikolić et al. 2011, 2013, Sfriso & Facca 2011; Bermejo et al. 2013; Orlando-Bonaca et al. 2013; Blanfune et al. 2017

Indices	**Acronym**	**Macroalgae/ecosystems**	**Country**	**Reference**
BENTHOS		*Cystoseira* spp. vs *Ulva*	Spain	Ballesteros et al. 2007a, Pinedo et al. 2007.
Ecological Evaluation Index	EEI	Macroalgae thick or calcareous thallus, low growth rates, long life cycles vs sheet-like and filamentous seaweed species with high growth rates, short life cycles (opportunistic) and Cyanophyceae	Slovenia; Albania, Istrian coast	Orfanidis et al. 2001, 2003, Orlando-Bonaca et al. 2008; Gogo 2015; Iveša et al. 2009. García-Sánchez et al. 2012
Index of Community Structure	ICS	Macroalgal cover; taxonomic stratification; functional group	Brittany (France) 14 sites	Ar Gall & Le Duff 2014.
Cover, Characteristic species, opportunistic species	CCO	Global cover of macroalgae, number of characteristic species per topographic level/seaweed community, and the cover of opportunistic species	France	Ar Gall et al. 2016
Calidad de Fondos Rocosos 'Quality of Rocky Bottoms'	CFR	General coverage of large characteristic macroalgae and opportunistic species	Spain	Guinda et al. 2014, Juanes et al. 2008
Reduced Species List	RSL	ca 70 algal species, habitat type, community structure, proportions of Rhodophyta, Chlorophyta; opportunistic species	British Isles, Atlantic coast of Andalusia (Spain)	Wells et al. 2007, Bermejo et al. 2012
Macrophyte Quality Index – Expert and rapid assessment	MaQI E-MaQI R-MaQI	Macroalgal species and marine seagrasses including epiphytes e.i. Corallinaceae (expert); or Rhodophyceae/Chlorophyceae ratio (rapid)	Italy	Sfriso & Facca 2010, Sfriso et al. 2009
Marine Macroalgae Assessment Tool	MarMAT	Species richness, proportion of Chlorophyta, number of Rhodophyta, number, proportion and coverage of opportunists, shore description	Portugal	Neto et al. 2012
Rocky Intertidal Community Quality Index	RICQI	Indicator species abundance, cover of morphologically complex algae, species richness, and faunal cover	Spain	Díez et al. 2012
Quality Index of Subtidal Macroalgae	QISubMac	Based on 14 metrics: depth, penetration, composition (sensitive, characteristic and opportunistic) and biodiversity of macroalgal assemblages	French Channel, Atlantic coast	Le Gal & Derrien-Courtel 2015
Ecosystem Based Quality Index	EBQI	Whole ecosystem; different habitat e.g., seagrass meadows, coralligenous; underwater caves (CavEBQI)	France	Boudouresque et al. 2015, Personnic et al. 2014; Rastorgueff et al. 2015; Ruitton et al. 2014
Coralligenous assemblages index	CAI	Bryozoa percent cover, sludge percent cover, builder species percent cover, depth percentage cover of coralline algae and macroalgae	France	Deter et al. 2012
coralligenous monitoring	RECOR	Monitoring network for coralligenous assemblages	France	http://www.observatoire-mer.fr/en/pub_coral.html Holon et al. 2010; Andromède Océanologie 2013.

Indices	Acronym	Macroalgae/ecosystems	Country	Reference
Ecological Status of Coralligenous Assemblages	ESCA	Rocky deep macroalgal assemblages; presence/absence and abundance of sensitive taxa/groups, diversity and heterogeneity of assemblages	Italy	Cecchi et al. 2014, Piazzi et al. 2015b, Piazzi et al. 2017b
Coralligenous Assessment by Reef Scape Estimation	COARSE	Biotic cover and conspicuous species richness, three-dimensional structure of coralligenous reefs; based on the Rapid Visual Assessment	Italy	Gatti et al. 2012, 2015; Piazzi et al. 2017a
Coralligenous index	INDEX-COR	Ratio between sensitive and tolerant species, richness of macrotaxonomic descriptors. Observation (*in situ* or from images) of structural coralligenous complexity.	Italy	Sartoretto et al. 2014, 2017.
Mesophotic Assemblages Ecological Status index	MAES q-MAES	ROV photography and video footage; community structure (number of megabenthic taxa, percent biotic cover in the basal layer, density of erect species), the condition of the dominant erect species (average height, percent of colonies with epibiosis/necrosis), the visible human impact (density of marine litter, including lost fishing gear).	Italy	Cánovas-Molina et al. 2016.
Coralligenous Bioconstruction Quality Index	CBQI	ROV imaging data; coralligenous habitat (rocky cliffs, submerged shoals, and platform banks) shallow and deep bottoms, and calculates their quality status taking into account coralligenous community structuring, bottom abiotic factors and anthropogenic stress evaluation	Italy	Ferrigno et al. 2017
Integrated Sensitivity Level of coralligenous Assemblages	ISLA	Changes in the coralligenous status along a gradient of stress (human-induced pressures), and in time, from a long-term series (1961–2008). Reduction in quality was associated with an increase in stress-tolerant and/ or opportunist species, the disappearance of sensitive macroalgae and macro-invertebrates, and the appearance of invasive alien algal species	Italy	Montefalcone et al. 2017
Ecosystem-Based Quality Index	Reef-EBQI	Shallow, algae-dominated rocky reefs, between 1 and 10 m. fish and habitat-forming species, together with several other compartments (e.g., sea urchins, detritus-feeders). It assesses the ecological quality of water bodies and the health of habitats, and is	Mediterranean	Thibaut et al. 2017

Indices	Acronym	Macroalgae/ecosystems	Country	Reference
		based on a wide set of functional compartments		
PAN for general use, EQ for ecological quality and MAT for Macroalgae Assessment Tool	PAN-EQ-MAT	Intertidal rocky shore seaweed community features	Azores	Wallenstein et al. 2013
ALien Biotic IndEX	ALEX	Invasive macroalgae	Italy	Piazzi et al. 2015a Piazzi et al. 2018
ALGAMONY		Total cover of erect macroalgae species and the lower depth distribution limit of selected macroalgae.	Finland, Norway, Sweden, Denmark	Moy et al. 2010
Nature Index	NI	65 indicators covering algae, invertebrates, fish, seabirds and sea mammals	Norway	Oug et al. 2013, Bekkby et al. 2013

Baltic Sea

Long term monitoring started in 1993/1994 with 11 transects established to reveal temporal changes in the bladder wrack belts, however numerous studies stated that the decline of *Fucus* had already started in the 1970s. Sites were revisited 2000/2001 and 50 sites were added in 2006/2007 to the monitoring plan (Vahteri & Vuorinen 2016). Field work was carried out by scuba on 50 m fixed transects. The effects of temperature, transparency, bottom type, shoreline orientation and location on the bladder wrack distributions and depth penetrations was studied at 61 locations across different archipelago zones (Vahteri & Vuorinen 2016). The benthic primary production in the sea of Kattegat between Sweden and Denmark was estimated from monitoring data, seafloor properties, and model simulations. The abundance and distribution of nine of the most dominant macroalgal species, and factors important for abundance, distribution and growth (e.g., bottom topography and sediment composition, irradiance, nitrogen concentrations and seawater temperature) were considered. A model computed macroalgal productivity based on publicly available macroalgal monitoring data, combined with hydrographic and sediment structure data (Öberg 2006).

Chile

In Chile macroalgae are cultivated or harvested for the extraction of alginates, for human consumption or as a source of food for abalone. Three ‘wild’ *Lessonia* species and *Macrocystis pyrifera* are under a strong and increasing pressure of exploitation (Buschmann et al. 2014b). In northern Chile, the kelp fishery has enormous social, ecological and economic importance, and more than 11 000 people depend directly or indirectly on the collection and harvesting of this resource. Kelp beds have a total value of US $540 million, of which, kelp fishery accounts for 75 % and associated-species fisheries account for 15 % (Vásquez et al. 2013).

The Chilean government has implemented a co-management and conservation plan, ‘Management and exploitation areas for benthic resources (MEABR)’ where several species under commercial exploitation have to be monitored (Almanza & Buschmann 2013). The MEABR co-management system was established by law in 1991 in response to a benthic resources overexploitation crisis with highly negative social and economic consequences (Gelcich et al. 2010). The policy was formally implemented nation-wide in 1997 (Castilla et al. 1998). In Chile more than 620 small-scale fisher organizations exist, but in practice, only 50 % of them apply for MEABRs (Marín et al. 2012).

In Northern Chile, mainly wild populations of *Lessonia nigrescens* Bory are exploited to produce alginates and this represents more than 70 % of the total brown seaweed annual landings (Vásquez et al. 2012). Indiscriminate harvesting has a significant impact on the ecosystem. The use of demographic parameters (e.g., biomass, recruitment, population size structure), as ecological indicators makes it possible to discriminate between coastal areas with different regimes of harvesting practices (Vega et al. 2014, 2016).

The monitoring of populations of *Lessonia nigrescens* was carried out along the northern Chilean coast in four site areas where *L. nigrescens* forms a continuous belt in the exposed rocky intertidal zone. In each site, density and biomass were evaluated for adult plants (more than 20 cm holdfast diameter), juveniles (between 5 and 10 cm holdfast diameter) and recruits (less than 5 cm holdfast diameter). Size structure was established using the largest diameter of the holdfast. Also, the number of stipes and the total length of the plant were evaluated. The population parameters were monitored seasonally for 2 years and compared between *L. nigrescens* populations in Open Access Areas (OAA), in management areas for exploitation benthic resource MAEBR, and in marine protected areas MPA (Vega et al. 2014). The *Scurria-Lessonia* association has been proposed as an ecological indicator to monitor the integrity of kelp beds. *Scurria scurra* is a patelogastropod inhabiting the stipes and holdfasts cavities of *Lessonia nigrescens* complex species forming a highly specialised herbivore-kelp association (Vega et al. 2016).

USA

The giant kelp *Macrocystis pyrifera* is the foundation for one of the most species-rich, productive, and widely distributed ecological communities in the world (Schiel & Foster 2015). There is an extensive literature on *Macrocystis* kelp forests and ecosystems. Aerial surveys of *Macrocystis* on the California coastline were initiated in the 1950s by ISP Alginates. In 1967 the use of infrared photographs started to track the areal extent of giant kelp canopy along the coast of Southern California and in 1989 these surveys were extended along the entire California coastline and they continue today (Bell et al. 2015). Several monitoring programmes received the help of trained volunteers to acquire intertidal or subtidal data. Citizen science has increased awareness in the communities about the importance of kelp forests and the services they provide.

LTER Long Term Ecological Research programme was established by the National Science Foundation (NSF) in 1980 to support research on long-term ecological phenomena. Santa Barbara Coastal Project (SBC) housed at the University of California is part of the LTER Network. The primary research objective of the SBC LTER is to investigate the relative importance of land and ocean processes in structuring giant kelp forest ecosystems. (http://sbc.lternet.edu//index.html). There is an extensive scientific literature developed under the LTER programme accessible at http://sbc.lternet.edu/cgi-bin/publications.cgi.

The Channel Islands National Park (California) “vital signs” programme, began in 1981, has endured because it has proved to be a cost-effective way to reduce uncertainty and increase success of conservation efforts. The term “vital signs” is used to describe a small suite of basic environmental measures, including biological (e.g., giant kelp abundance and distribution), physical (e.g., sea temperature), and chemical (e.g., pH) elements and processes, that represent the entire array of such features in an ecosystem, and that reflect temporal and spatial changes in ecosystem structure and function (Davis 2005).

In Washington State, two species of kelp are dominant: giant kelp (*Macrocystis pyrifera*) and bull kelp (*Nereocystis luetkeana*). While both species occur along Washington's outer coast and coastal Strait of Juan de Fuca, bull kelp is the main species found along shorelines of the inner Salish Sea (Mumford 2007). In 1988, Washington State Department of Natural Resources (WDNR) initiated annual aerial surveys of coastal aquatic vegetation from Port Townsend Bay to the Columbia River. These surveys have continued nearly every year, and in 2010, surveys were extended to include the resources of the Smith and Minor Island Aquatic Reserve (SMIAR) within Island County. In the latest analysis of coastal kelp from 2013 to 2014 (excluding SMIAR), decline in planimeter area of bull kelp around Port Townsend was about 14 %, and range-wide decline in planimeter area of both kelps was 38 % (Van Wagenen 2015). There is also a monitoring programme for *Nereocystis* in Kachemak Bay, Alaska, undertaken by the Kachemak Bay Research Reserve (Mayne Island Conservancy Society 2010). In 2014 the Northwest Straits Commission launched a regional survey of bull kelp beds using a kayak-based survey (Bishop 2016). These on-the-ground surveys were supported by several agencies, including the Department of Natural Resources, to confirm the data based on flight surveys.

Island County Marine Resources Committee (MRC) (Washington State) is an advisory body to county government established in 1999 and comprises community volunteers representing diverse interests and industries, with the common goal to protect and restore marine resources in the Puget Sound area through scientific monitoring, restoration projects, and community education. There are seven county-based Marine Resource Committees in the Puget Sound region funded by US Environmental Protection Agency through the Puget Sound Partnership and Northwest Straits Commission (NWSC) (http://www.islandcountymrc.org/). The MRC's purpose is to investigate local marine resources and habitat issues, recommend remedial actions to Island County agencies and authorities, and build local awareness.

A consortium of organizations called MARINe (Multi-Agency Rocky Intertidal Network) and led by the University of California Santa Cruz regularly carried out intertidal surveys at fixed monitoring sites along the entire Pacific Coast of North America, from Alaska to Mexico, and several East Coast sites in Maine and New Hampshire. The monitoring focus is on key species within fixed plots, annually or semi-annually. This fixed-plot approach allows the dynamics of rocky intertidal species to be monitored with reasonable sampling effort and provides sufficient statistical power to detect changes over space or time. In addition, biodiversity surveys to document species richness and changes in the distribution of species is carried out typically every 3–5 years.

These surveys require a high-level of expertise and consistency in the identification of marine organisms, performed by a group in UC Santa Cruz.
https://www.eeb.ucsc.edu/pacificrockyintertidal/overview/index.html.
A publication list is found here
https://www.eeb.ucsc.edu/pacificrockyintertidal/publications/index.html#publications.

PISCO (Partnership for Interdisciplinary studies of Coastal Oceans) kelp forest monitoring has been running continuously since 1999 in shallow (5–20 m depth) nearshore sites located on rocky bottom habitats. The monitoring includes measurement of density and biomass of macroalgae, invertebrates and fishes within kelp forest communities. Standardised sampling protocols across all PISCO institutions and partners are in use, and these surveys provide valuable data to help assess the causes and consequences of environmental changes. Methods and training material can be found at http://www.piscoweb.org/kelp-forest-sampling-protocols

Reef Check has built a global network of volunteers that monitors reefs worldwide through three programmes: Tropical Reefs Program, the Baja California, Mexico Programme and the California Program. Reef Check California (RCCA), begun in 2005, is a community-based reef monitoring network with a goal of improving marine management in California by collecting data on near-shore rocky reef ecosystems using volunteer scuba divers. All RCCA's volunteers go through a rigorous four-day training and annual recertification program. Since the start of the programme RCCA has trained over 1000 divers in California. The survey methods are based on a visual census developed by the Partnership of Interdisciplinary Studies of Coastal Oceans (PISCO) and have been modified to be taught in a reasonable amount of time to volunteer scuba divers. The data collected are made available to resource managers, universities, researchers and the general public (Freiwald & Wisniewski 2015). Data collected by trained volunteers and professional scientists were compared, and results suggest that volunteers can be taught the appropriate skills to produce similar data to professional scientists, as long as there is sufficient guidance and supervision, a rigorous sampling scheme, and that the taxonomic scope of the work is constrained (Gillett et al. 2012). In 2013 Reef check expanded into the central Baja California (Mexico) BCMP (Baja California Monitoring Program).

A monitoring programme initiated in 2006 under the National Park Service (NPS) Southwest Alaska Network inventory and Monitoring Program, focused on the nearshore benthic food web in the Gulf of Alaska, to understand potential causes and mechanisms of ecosystem changes. Fifteen long-term monitoring sites across more than 1000 km of coastline were established. The programme evaluated six ecological indicators and more than 200 species that range from primary producers to top-level consumers, and examined both bottom- up and top- down dynamics. This monitoring followed the vital signs monitoring programme initiated by the National Park Service (NPS) (Coletti et al. 2016).

Canada

In Nova Scotia, long-term data on kelp biomass and extent, spanning 30 to 65 years, made it possible to monitor kelp beds and estimate their decline. The losses were associated with warming sea temperatures affecting biological processes and interactions (Filbee-Dexter et al. 2016). The Coastal Environmental Baseline Program, part of the national Oceans Protection Plan, was launched in November 2016. The programme extends over 5 years and aims to collect comprehensive baseline data on the state of 6 marine ecosystems in Canada in oreder to detect changes in the environment over time. http://dfo-mpo.gc.ca/science/environmental-environnement/cebp-pdecr/index-eng.html

An estimate of the total standing crop, biomass, and kelp bed area of *Nereocystis luetkeana* (K.Mertens) Postels & Rupr. and *Macrocystis pyrifera* along portions of the central coast of British Columbia was performed for the first time in 2007 using colour infrared photography and digital mapping of kelp polygons directly from georeferenced digital images. Results in 2007 compared with 1993 data estimates for areas common to both surveys showed a decrease in *Nereocystis* stands of 26 % of bed area and 19 % of biomass, while *Macrocystis* biomass increased throughout the area (Sutherland et al. 2008). A collaborative kelp forest ecosystem monitoring programme to establish baselines against which future changes could be evaluated was carried out from 2009–2013 in the shallow rocky reef ecosystems Haida Gwaii, (British Columbia). Monitoring at 11 sites included underwater visual surveys of subtidal reefs associated fish, invertebrate and macroalgal communities (Trebilco et al. 2014).

Australia

Australia has one of the most species rich and endemic temperate algal floras in the world (Bennett et al. 2016). Research on all the aspects of macroalgae (e.g., taxonomy, ecology, biology) has produced an extensive literature that is highly cited worldwide. Four laminarian genera occur in southern Australia (*Macrocystis*, *Lessonia*, *Ecklonia* and the invasive Japanese kelp *Undaria*) and 18 fucoid genera, some of which are particularly speciose e.g., *Sargassum* and *Cystophora* (Womersley 1992). The most widespread species in shallow temperate reefs is *Ecklonia radiata,* with its distribution stretching along more than 8000 km of coastline from the subtropical waters of northern New South Wales down the east coast of mainland Australia, around Tasmania, along Australia's southern coastline and north as far as Kalbarri (Bennett et al. 2016). *Macrocystis pyrifera* occurs on mainland Australia from Cape Jaffa (South Australia) to Walkerville (Victoria) and on the East and South-East coast of Tasmania (Womersley 1992). *Lessonia* only occurs in Tasmania (Womersley 1992).

Long-term monitoring data are available for rocky reef habitats of Marine reserves and Marine Sanctuaries. The first monitoring programme was established in Tasmania in 1991 (Edgar & Barrett 1999) followed by programmes in Victoria and South Wales established in 2002. These monitoring programmes were carried out either by marine consultants (Victoria) or scientists and government agencies.

The Victorian Government has created a system of marine protected areas (MPAs) along the state's coastline. This system provides legislative protection for 13 Marine National Parks (MNP) and 11 smaller Marine Sanctuaries (MS) covering 5.3 % of the Victorian coastline (Keough et al. 2007). The long-term Subtidal Reef Monitoring Programme (SRMP) was established in 14 locations sampled in some cases since 1998 to determine the nature and magnitude of trends in the abundance and diversity of algae, invertebrates and fish on reefs in Victoria's MPAs (Hart et al. 2004). The monitoring involves standardised underwater visual census methods based on an approach developed in Tasmania (Edgar & Barrett, 1997) along a 200 m transect line, surveyed for: abundance and size structure of large fishes; abundance of cryptic fishes and benthic invertebrates; percentage cover of macroalgae; and density of a dominant kelp species (Edmunds et al. 2003, Pritchard et al. 2012).

An Intertidal Reef Monitoring Programme (IRMP) was established in 2003 at six marine sanctuaries and then expanded to three Marine National Parks and Merri Marine Sanctuary. The IRMP began with the aim of using standardised visual census methods to track changes in abundances of invertebrates and macroalgae due to human use of the reef platforms, including trampling and fossicking. At each site, observations were made along five fixed transects from the high to the low shore. Quadrats, randomly placed along the transect, provided data to determine the cover-abundance of macroalgal species and aggregating invertebrates, and the abundance of mobile macroinvertebrates. Annotation about substrate type and morphology and photo quadrats or still video for further analysis were also acquired. Collections were made of specimens as vouchers or for taxonomic, genetic or other studies (Hart & Edmunds 2005). The aims of the programme were expanded to investigate biodiversity, climate change effects and introduced marine species.

Investigations into the health of reefs on the Adelaide metropolitan coast has occurred since the late 1990s with surveys within the Reef Health research program, led by SARDI Aquatic Sciences (Collings et al. 2008, Turner et al. 2006, Turner et al. 2007, Westphalen 2008). As the loss of habitat-forming algae became a public concern, a citizen science programme 'Reef Watch', was officially established in 1997, managed by the Conservation Council of South Australia (CCSA) with the participation of trained volunteers working with marine scientists to gather quality information that contributes to the management of the marine environment. Reef Watch was the first community-based marine monitoring programme in Australia and began by training recreational divers to survey subtidal reefs, and then expanded to include intertidal reef monitoring ('Feral or In Peril' programme) (Westphalen 2008).

In 2000, the Department of Primary Industries, Water and Environment and the University of Tasmania, received funding from the Commonwealth Government's 'Coast and Clean Seas Program' to undertake a detailed conservation assessment of giant *Macrocystis pyrifera* beds in Tasmania. The project included mapping and survey of *Macrocystis* beds, identification of threats including the distribution of *Undaria pinnatifida* and sources of pollution. A conservation assessment was carried out to determine potential for listing under Commonwealth and State /Territory endangered species legislation and evaluation of potential sites for Marine Reserves. 'Kelp Watch' a community-based research and monitoring programme was established to include the local anecdotal knowledge and experience of the marine industries (ie. diving, fishing, shipping) and community groups involved in field work (Tasmania Department of Primary Industries 2004).

In Western Australia the Department of Fisheries (DoF) ecosystem-based fisheries management (EBFM) strategy has complemented the traditional methods of fishery management with a more comprehensive assessment of trophic interactions to devise sustainable and environment-friendly fishery methods. DoF has undertaken a preliminary marine monitoring programme established in marine parks of the Natural Resource Management (NRM) Swan region. This programme gathered baseline data on benthic invertebrate and floral communities, western rock lobster and fish communities. The project intended also to test and develop a robust monitoring regime and identify indicator species for management and early detection of changes due to anthropogenic pressures. Algal functional groups were used as indicators of change to the benthic communities, as they require minimal taxonomic expertise (i.e., surrogates) and could be identifiable from close-up (less than 1 m) digital imagery taken of the substratum, precluding the need to remove vast amounts of material from the marine sites (Bellchambers et al. 2009).

The Department of Environment and Conservation (DEC) has monitoring programmes in place for Marine Reserves, and the Environmental Protection Authority (EPA) has an active monitoring programme in the Perth region. Both agencies evaluate macroalgal assemblages and seagrass in order to assess the quality of benthic habitat or water quality. Measurement of the biomass and diversity of seagrass meadows, macroalgal communities, algal growth and seagrass shoot, density and depth limits are considered. Department of the Environment and Heritage (DEH) also measures extent and composition of macroalgal assemblages on soft and hard substrata (Smale et al. 2011).

A range of ecological indicators derived from benthic primary producers are in use in Western Australia, mostly developed on seagrasses as these are widespread in the region. As macroalgal assemblages are very speciose in Western Australia, and identifing macroalgae to species.is challenging, recent studies have assessed the efficacy of biodiversity surrogates (e.g functional groups, higher taxonomic groups and/or subsets of taxa), as cost-effective monitoring tools (Smale et al. 2011). The value of surrogates for marine macroalgae to predict species diversity and to minimise sampling effort was tested in the Recherche Archipelago. Diversity at higher taxonomic levels, richness at species and genus levels, and richness of dominant taxa were evaluated in in a variety of habitat characteristics such as depth, exposure to wave energy, and substratum type. The study provided support for using species richness as an effective surrogate while higher taxonomic level surrogates and dominant taxa were not consistently able to predict species diversity due to shifts in macroalgal diversity with island location, depth and exposure to ocean swells (Goldberg et al. 2006). Surrogates derived from taxonomic aggregation to genus or family level correlated strongly with species-level patterns, although the family-level surrogate was a less effective predictor of species richness at large spatial scales. Taxonomic aggregation to a genus level resulted in a 50 % reduction of level of expertise and aggregation to a family level led to an 80 % reduction in expertise. The use of surrogates indicated that the marine flora of south-west Australia can be quantified and monitored in a cost-effective manner, provided due consideration is given to the choice of surrogates and the spatial scale of interest (Smale 2010). The use of indicators for ecological monitoring in marine benthic ecosystems “from frond to fish” including case studies from temperate Western Australia were critically assessed and summarised in a comprehensive review (Smale et al. 2011).

Numerous scientific papers have been published about kelp and macroalgal assemblages in Western Australia, assessing the threats to species and coastal ecosystems, and in order to understand or predict ecosystem changes. While this extensive literature is mostly based on ecological work to test different hypotheses about ecosystem responses to different stressors (e.g., sediments, temperature, nutrients, waves, ocean acidification) few studies are based on long-term monitoring data. As noted by Wernberg et al. (2009) there are several international examples of climate change impacts on rocky reef ecosystems, but only very few examples from Australia because of a lack of long-term and broad-scale data sets (Wernberg et al. 2009).

South Africa

In South Africa surveys were carried out on two species of kelp exploited commercially, *Ecklonia maxima* and *Laminaria pallida* Grev., both reaching the surface at low spring tides. Monitoring was carried out on 900 km of coast using several methods, infrared aerial photography, multispectral aerial imagery, Landsat satellite imagery and physical mapping with hand-held GPS. The most effective methods were the infrared aerial imagery and multispectral imaging which accurately quantified areas of kelps (Anderson et al. 2007). Repeat photography of historical images gathered from different sources e.g., book, thesis, public, specialist made it possible to detect visual changes in coastal habitat including progressive easterly spread of the cold-water kelp *Ecklonia maxima* (Reimers et al. 2014). South African kelp forests have been a valuable source of scientific information particularly as a result of the Kelp Bed Ecology Programme that took place in the 1970s and 1980s but there has been little long-term monitoring of kelp forests in the region (Blamey & Bolton 2017).

Japan

Japan has a wide range of climate zones from cold temperate to subtropical, reflecting its wide geographical extent from north to south, as well as the influence of two major warm and cold currents, the Kuroshio and Oyashio, respectively, bring very diverse nutrient conditions, from oligotrophic to highly eutrophic waters, and different degrees of wave exposure (Okuda 2008). Japan has the highest diversity globally of kelp species, with 38 species recorded from the region, and a long history of harvest for utilization as a food material (Fujita 2011). *Laminaria* beds appear predominantly in the northern part of Japan, with about 75 % of *Laminaria* beds located in Hokkaido. Other kelps include species of *Alaria*, *Undaria*, *Agarum*, *Costaria* and more. Beds of *Eisenia/Ecklonia* are distributed in the temperate zone and account for 24 % of the total area of seaweed beds in Japan, while *Sargassum* beds consist of many species and represent 27 % of the total area of seaweed beds in Japan (Okuda 2008). Data of habitat-forming seaweeds (fucoids and temperate kelps) have been collected at 7673 sites on the Japanese coast from warm to cold temperate zones. Data recorded from 1887 to 2014, include 86 species (21 168 presence and 20 845 absence records), compiled from 355 literature sources. This rich data set is important for the study of the biogeography and long-term changes and summarise the diversity of habitat-forming seaweeds of the Japanese coast (Kumagai et al. 2016).

In 2003 Japan's Ministry of the Environment (MOE) established 1000 Monitoring Sites designed to promote monitoring of important ecosystems nationwide for a long-term (5 years) survey (Watanabe et al. 2012). The project aimed to collect high quality data to develop strategic conservation measures considered important for biodiversity conservation, and to detect degradation of ecosystems in its early stages by long-term continuous monitoring of ecosystem components. The monitoring sites included ecosystems such as nearly untouched natural forests, tidal flats and seaweed beds, as well as urban parks. The Ministry of Environment has also regularly carried out national surveys on the country’s vegetation, coastal areas and other natural environments since 1973, to gather basic information on natural terrestrial, inland-water, and marine ecosystems across the country (Watanabe et al. 2012).

To restore seaweed beds, the Fisheries Agency carried out a national project in cooperation with 16 prefectures and 19 specialists from 2004 to 2006 (Kuwahara et al. 2010). The major decreasing bed-forming seaweeds were the cold temperate kelp *Saccharina*, the warm temperate kelps such as *Ecklonia* and *Eiseni*a, and *Sargassum* and *Gelidium* in some local areas. In 2007 the Isoyake Taisaku Guidelines were published for promoting restoration of seaweeds and encouraging collaboration among administrative boards, researchers and citizens and the Large-scale Project for Promotion of Recovery from Isoyake’ started (2007–2009) (Fujita 2010). The restoration of seaweed beds has often been unsuccessful for various reasons e.g., drastic changes in coastal environments, inappropriate methods and substrata. Therefore the Fisheries Agency started a new project for supporting the conservation of environmental ecosystems in 2009 and provided financial support to keep monitoring, maintenance and restoration of seaweed beds. The guideline suggested trial experiments prior to the restoration and assigned experienced specialists. More than 250 groups started to restore healthy and functional ecosystems, among which 150 groups worked on seaweed beds, and 50 groups worked for restoration of kelp beds (Fujita 2011). The use of citizens as volunteer divers in the removal of sea urchins was successful (Watanuki et al. 2010) and was shown to be the most effective of the four methods tested, (removal by diver, collecting from the boat, by basket, and pump) (Kuwahara et al. 2010). Several studies have focused on urchins as they are considered to play a key role in isoyake. High grazing pressure of sea urchins reduced the survival of juvenile sporophytes of *Eisenia* and *Ecklonia* near the lower limit of the bed. Seaweed beds have been progressively receding from the offshore side of the beds every year impeding the recruitment of juvenile sporophytes to replace adult sporophytes that have died due to senescence, storm action and/or grazing (Muraoka 2008). Sea urchin removal was considered to be the most effective method to regenerate seaweed beds at three sites while at two sites the browsing activity of herbivorous fishes inhibited seaweed recolonization (Taino 2010).

2.3.2.2 Macroalgae as bioindicators of nutrients and heavy metal enrichment

Several macroalgae have been tested and used as bioindicators to detect land-based nutrient enrichment in coastal waters e.g., from sewage treatment facilities, shrimp farms, and agricultural runoff. In Moreton Bay, Australia, the red macroalga *Catenella nipae* Zanardini was used to map a sewage plume after incubation in situ, and measurement of the d15N isotopic (Costanzo et al. 2001, 2005). The brown macroalga, *Sargassum flavicans* (K.Mert.) C.Agardh, was tested as a potential bioindicator of nutrient-enriched industrial effluent originating from a nickel refinery in tropical north-eastern Australia (Alquezar et al. 2013). *Ulva lactuca* L. in the Mediterranean was tested as an indicator of anthropogenic sources of nitrogen (N), and was found to have potential as a probe to map pulses of nitrogen inputs of different origins (Orlandi et al. 2014, 2017).

Brown algae have shown to be effective bioindicators for metals due to the high density of carboxylic groups present in alginate (the main component of their cell walls) resulting in high rate of metal accumulation. Calcium-treated *Undaria pinnatifida* was found to be a suitable adsorbent for Zn(II) and Cd(II) pollutants and could be utilised to treat industrial water (Plaza Cazon et al. 2013). The concentrations of heavy metals (Cr, Mn, Ni, Cu, Zn, Cd and Pb) were measured in four brown algae from the Sudanese Red Sea coast. Results indicated *Padina* and *Cystoseira* were better bioindicators than *Turbinaria* and *Sargassum* for their high metal uptake (Ali et al. 2017). The high uptake of metals in green algae (*Ulva lactuca* and *U. intestinalis* L.) and brown algae (*Padina gymnospora* (Kütz.) Sond. and *Dictyota bartayresiana* J.V.Lamour.) indicates that these algae may be used as potential biomonitors for heavy metal pollution (Chakraborty et al. 2014). A review of the potential use of brown algal ecotoxicological assays in monitoring effluent discharge and pollution in southern Australia summarised the species of large brown algae which have been either employed for the development of ecotoxicological assays or have potential for development as indicators (Burridge & Bidwell 2002).

2.3.3 New Zealand macroalgae and the responses of macroalgae to stressors

2.3.3.1 Habitat-forming New Zealand large brown algae - Laminariales, Fucales

The Laminariales is represented in New Zealand by 3 native genera (*Ecklonia, Macrocystis, Lessonia*) and one introduced genus (*Undaria*), with 10 species known, 3 of which are undescribed. Eight species are distributed within the North, South, Chatham and Stewart Islands. Members of this order have a 2-phase life history with microscopic gametophytes alternating with the macroscopic, conspicuous sporophytes. A summary of the taxa covered in this section is provided in Table 3.

Table 3: Summary of habitat-forming large brown algae Laminariales and Fucales.

Genus	Brief description	Species
Ecklonia	Thalli large, with smooth, cylindrical, unbranched stipe, flattened blade with lateral lobes, over 1 m high. Stipe and blade length vary considerably	*E. radiata*
Lessonia	Thalli large, with long, parallel sided strap-like blades growing from perennial solid, dichotomously branched axes, over 1 m high.	*L. variegata sensu lato* *L. tholiformis*

	L. variegata sensu lato includes *L. variegata* and an additional three undescribed species.	*L. adamsiae* *L. brevifolia*
Macrocystis	The thalli consist of multiple cylindrical axes growing to the surface, which form leaf-like blades with pear-shaped air bladders. Massive dome-like holdfasts made up of many branching cylindrical haptera.	*M. pyrifera*
Undaria	Prominent, flattened main axis, extending into divided blade with a well-defined midrib. Thalli lie in one plane. Sporophyll (fertile portion of the plant where sporangia are produced) at the base of the blade, deeply folded, and appears to spiral around the stem and the base of the blade forming a thick, dark ruffle	*U. pinnatifida*
Durvillaea	Thalli massive with the leathery blades divided into long strap-like branches. There is an undescribed species found only on the Antipodes Is.	*D. antarctica* *D. chathamensis* *D. poha* *D. willana*
Hormosira	Thalli are composed of branched chains of bead-like hollow segments joined by thin constrictions, up to 20–30 (occasionally 50) cm long.	*H. banksii*
Carpophyllum	Elongate, flattened main axis (except *C. angustifolium*). Holdfast flattened and creeping extension of the main axis, with peg-like attachments along the edges. Rounded to oval vesicles (rarely found on *C. angustifolium*). Leafy branchlets.	*C. angustifolium* *C. flexuosum* *C. maschalocarpum* *C. plumosum*
Cystophora	Plants are shrub-like and up to1–1.5 m. Branchlets narrow and cylindrical (*C. retroflexa, C. scalaris*), swollen and club-shaped (*C. torulosa*) or flat, leaflike (*C. platylobium*). Holdfast disc or solid and conical.	*C. platylobium* C. *retroflexa* C. *scalaris* *C. torulosa*
Landsburgia	Cylindrical main axes growing up to 1.5 m high. Alternate branching with branchlets naked at bases. Leaves oak-shaped, flattened with a faint mid rib in the lower portion.	*L. ilifolia* *L. myricifolia* *L. quercifolia*
Sargassum	Thalli bushy, cylindrical main and secondary axes, with or without spines, growing up to 1 m high. Basal leaves with a distinct midrib, serrated margins; upper leaves narrow and thinner. Cryptostomata present or absent. Vesicles round to oval and smooth with or without apical leaflet.	*S. scabridum* *S. sinclairii*
Phyllotricha	Thalli have short and stubby axes at the base and basal leaves elongated and lobed, with the main axis becoming slender towards the apex and zigzagging, growing up to 1 m or more. Leaves lobed to dichotomously divided, with a faint midrib, prominent cryptostomata. Vesicles round, scattered among the upper branches, smooth or with a short tip.	*P. verruculosa*
Marginariella	Axes bear long, alternate, flat narrow strap-like lateral fronds that are usually less than 3 cm wide, growing to more than 2 m long. Vesicles oval (*M. boryana*) or rounded (M. *urvilliana*)	*M. boryana* *M. urvilliana* *M. parsonsii*
Xiphophora	Thalli fan shaped, with regular dichotomous branching lying in one plane, to 50 cm or more in height	*X. chondrophylla* *X. gladiata*

The Fucales is represented by 10 genera (*Carpophyllum, Cystophora, Durvillaea, Hormosira, Landsburgia, Marginariella, Notheia* (an obligate epiphyte, primarily found on *Hormosira*), *Phyllotricha, Sargassum, Xiphophora*) and 29 species (1 undescribed), of which 22 are distributed within the North, South, Chatham and Stewart Islands. Members of this order have a direct life history, i.e., with no alternate life history phase.

LAMINARIALES

Ecklonia:

Ecklonia radiata (C.Agardh) J.Agardh is found in the low intertidal to subtidal zones, on rocky reefs, occasionally on cobbles and shells, on moderately sheltered shores well as on exposed coasts, providing habitat and food for a great diversity of reef-dwelling species. In northern New Zealand it is particularly important in coastal assemblages, forming extensive beds along rocky coastlines. The thalli are large, with a smooth, cylindrical, unbranched stipe and a flattened blade with lateral lobes, up to 1 m or more high. Both the length of the stipe and size of blade vary considerably.

A number of theses have been completed investigating various aspects of *Ecklonia* biology, ecology, reproduction and physiology in New Zealand (e.g., Haggitt 2004, Hanns 2014, Miller et al. 2011, Novaczek 1980, Rodgers 2014). Studies in different parts of the country have yielded valuable insights into how this species responds to a range of environmental factors such as light and temperature (e.g., northeastern North Island (Novaczek 1984a, 1984b, Rodgers & Shears 2016); Fiordland (Miller et al. 2011, Wing et al. 2006, Wing & Jack 2007).

Lessonia:

There are at least seven species of *Lessonia* currently recognised in New Zealand: *L. tholiformis* is found only on the Chatham Islands, *L. adamsiae* only on the Snares Islands, *L. brevifolia* is restricted to the subantarctic islands (Adams 1994, Hay 1987, 1989). The name *L. variegata* has been used for *Lessonia* growing round the North, South and Stewart Islands but recent research has revealed that there is greater genetic diversity than previously recognised, with an additional three species to be described (Martin 2011, Zuccarello & Martin 2016). For the purposes of this account the species within this complex are referred to as *L. variegata sensu lato*. *Lessonia* is found in the upper subtidal through to deep water on rock on exposed coasts, in places forming extensive beds. The thalli are large, with long, parallel sided strap-like blades growing from perennial solid, dichotomously branched axes, up to 1 m or more in height.

Nelson (2005) documented the life history of *Lessonia variegata* in culture, based on material collected in Wellington, determining optimal growth conditions for gametophytes. Schwarz et al. (2006) carried out a field study of *Lessonia variegata* growing in depths of 8–10 m on the south coast of Wellington, investigating morphometric and biomass indices following the growth and reproduction over a 2-year period. The highest proportion of fertile blades (up to 100 % of sampled blades) occurred during winter, and the end of the fertile period coincided with the onset of the longer days and warmer temperatures of summer. Two parameters, holdfast diameter and girth of all stipes collectively, both relatively easily measured on live macroalgae in situ, reliably predicted total biomass, suggesting that these could be useful tools for assessment of this resource.

Macrocystis:

Macrocystis pyrifera (L.) C.Agardh, giant kelp, is present in both the northern and southern hemispheres, and is a species of considerable ecological and economic importance. The extensive beds along the Californian coast have been harvested for decades. *Macrocystis* grows in the subtidal zone on open coasts in areas where there is shelter from direct wave action. The thalli consist of multiple cylindrical axes growing to the surface, which form leaf-like blades with pear-shaped air bladders. The thalli are attached by massive dome-like holdfasts made up of many branching cylindrical haptera.

Schiel & Foster (2015) published a detailed treatment of all aspects of the biology and ecology of *Macrocystis*. Moore (1942) and Rapson et al. (1942) investigated aspects of *Macrocystis* biology and distribution in New Zealand at a time when the resource was being considered as an alternative supply for potash and other commercial uses. A series of theses and other studies have been completed on *Macrocystis* in New Zealand investigating a wide range of aspects of its biology including growth, productivity, biomass, harvestable yield, seasonality, distributional range in relation to temperature, responses to epiphytism, aquaculture and nutrient physiology, impacts of ocean acidification (e.g., Amado et al. 2012, Bartsch et al. 2008, Brown et al. 1997, Chew et al. 2013, Cummack 1981, Fernández et al. 2017, Fernández et al. 2014, Fernández et al. 2015, Fernández 2015, Hay 1990a, Hepburn 2003, Hepburn et al. 2006, 2007, 2012, Hepburn & Hurd 2005, Kain 1982, Leal et al. 2014, 2016, 2017b, Macaya & Zuccarello 2010a, Macaya & Zuccarello 2010b, Macaya 2010, Pirker 2002, Stephens 2015, Stephens & Hepburn 2014, 2016, Stewart 2015). *Macrocystis* known as bladder kelp (KBB G), was introduced into the Quota Management System in 2010, within Fishery Management Areas 3 (eastern South Island) and 4 (Chatham Islands). The total allowable commercial catch is 1237 T (eastern South Island) and 274 T (Chatham Islands). Only a 0.1 T catch is allowed in eastern South Island for non-commercial customary and 0.1 T for recreational. The single restriction on KBB G harvest, implemented when the species was introduced to the Quota Management System (1 October 2010), is a maximum cutting depth of 1.2 m. The harvest of bladder kelp by customary Maori is currently unrestricted, however, the customary harvest of attached bladder kelp is likely to be negligible. Currently there is insufficient information on canopy area and density to allow for a stock assessment (Fisheries New Zealand, 2018). Commercial allowance and trend organised by region and year can be found at https://fs.fish.govt.nz/Page.aspx?pk=7&sc=KBB&ey=2019.

***Undaria*:**

Undaria pinnatifida (Harv.) Suringar, wakame, is an introduced kelp native to the northwestern Pacific, classified as an 'unwanted organism' under the Biosecurity Act 1993. It is considered a pest in New Zealand because it can compete for space and light with native species and it has successuly invaded New Zealand from Northland to the Snares Islands, including the Chatham Islands. *Undaria* is capable of living in a wide range of habitats from sheltered harbour areas through to open coasts and from the intertidal zone to more than 20 m depth. It can tolerate a wide range of temperatures, exposures and substrate types. The thalli are very distinctive as each blade has a prominent main axis that is flattened and extends into the divided blade as a well-defined midrib. The blade is divided into numerous lobes that all lie in one plane. The sporophyll (fertile portion of the plant where sporangia are produced) is at the base of the blade, deeply folded, and appears to spiral around the stem and the base of the blade forming a thick, dark ruffle.

First discovered in New Zealand in 1987 in Wellington Harbour (Hay & Luckens 1987), there was some initial work on its presence in Wellington Harbour in 1988 (Hay 1988), and on the dispersal of sporophytes (Hay 1990b). Hay & Villouta (1993) published the results of work on the seasonality of *Undaria* in New Zealand, and Forrest et al. (2000) investigated the role of natural dispersal mechanisms in the spread of *Undaria pinnatifida,* while James (2016) studied the factors influencing the establishment and the phenology of the invasive kelp in northern New Zealand.

Genetic diversity of *Undaria* in New Zealand and the likely origins of the multiple haplotypes that have been introduced were investigated by Uwai et al. (2006). The expanding range of *Undaria* has been documented by a range of authors including James et al. (2014), Russell et al. (2008), Schiel et al. (2018). A number of theses have been produced with a focus on *Undaria* e.g., Balbas 2012, Chen 2012, Hau 2012, James 2016, Jiménez 2015, Leal 2016, Morelissen 2012, Thompson 2004. Since its discovery in New Zealand several publications have reported on the impact of this species in New Zealand (e.g., South et al. 2016, Stuart 2004, Thompson & Schiel 2012). South et al. (2017) reviewed the international literature on this species and assessed its impacts noting that "the impacts of *Undaria* are not well understood due to a deficit of studies, but it seems likely that its advent has modified patterns of biogenic habitat provision and trophic subsidies locally and across ecosystems, despite its seeming inability to displace many native taxa".

FUCALES

Durvillaea:

Species of the bull kelp *Durvillaea* provide considerable biomass and a major habitat in the lower intertidal and upper subtidal zones zone of exposed shores in New Zealand. Five species are known to occur in New Zealand waters, one of which (from the Antipodes Is) is still not formally described.

The morphology, anatomy, reproductive biology and seasonality of *Durvillaea*, particularly *D. antarctica*, as well as the impacts of and responses to harvest have been well documented (e.g., Hay 1977, 1979a, 1979b, 1979c, 1994, Naylor 1949, Naylor 1953b). The systematics of *Durvillaea* has been revised, and in the light of multigene phylogenetic analyses and morphological evidence, has resulted in the description of the species *Durvillaea poha* (Fraser et al. 2010, Fraser et al. 2012). Over the past several decades theses on different aspects of the biology and ecology of *Durvillaea* have been completed, covering such topics as the responses to wave exposure; dispersal and settlement of zygotes; herbivory; the impact of removal on understorey species and recruitment and the contribution of stranded seaweed to sandy shore food webs (e.g., Dufour 2011, Dunmore 2006, Taylor 2002, Taylor & Schiel 2003, 2005, 2010). *Durvillaea* is not included in the quota management system and as taonga species, commercial harvest in the South Islands is restricted (Ngai Tahu Claims Act 1998).

Durvillaea antarctica (Cham.) Har., bull kelp, rimurapa:
Durvillaea antarctica is present in the low intertidal zone on wave-exposed coasts. Thalli are massive with the leathery blades divided into long strap-like branches, 3–5 (up to 10) m long, extending from the cylindrical trunk-like stipe, and dome-shaped holdfasts. A distinctive feature of this species is the honeycomb-like internal structure giving buoyancy to the thalli. The shape and width/length as well as the splitting of the blade into long straps or thongs are influenced by the wave exposure of the sites where this species is found. The blades have a somewhat inflated appearance resulting from a honeycomb-like internal structure, making the blades buoyant.

Durvillaea chathamensis C.H.Hay:
D. chathamensis can be found in the low intertidal and upper subtidal zones, to about 2 m depth on exposed reefs in the Chatham Islands. The thalli are large, with thin, parallel-sided blades with undulating margins, up to 15 m in length. The stipe is unbranched and the blades are usually simple although occasionally divided into narrow straps.

Durvillaea poha C.I.Fraser, H.G.Spencer & J.M.Waters:
This species was described recently and is separated from *D. antarctica* on the basis of both morphological features and genetic data. It had been recognised for some time and regarded to be a variant correlated with living in more wave-sheltered habitats, and referred to the "cape morphotype". *D. poha* overlaps part of the range of *D. antarctica*. *D. poha* is likely to be under-represented in herbarium records as not all herbarium specimens have been verified and updated. *D. poha* is found in the intertidal zone, commonly at mid-tide level, on exposed rocky coasts, usually in a band above *D. antarctica*, and on semi-sheltered coasts throughout the intertidal zone. Thalli are large with flattened blades, inflated, 'honeycombed' buoyant tissue, and grow up to 5–10 (occasionally 12) m in length.

Durvillaea willana Lindauer:
Found in the low intertidal and upper subtidal zones, only on exposed coasts, the thalli of *D. willana* are massive and leathery, up to 5 m long, with thick trunk-like stipes from which stalked blades arise.

Hormosira:

Hormosira banksii (Turner) Dcne., Neptune's Necklace, is a very common and easily recognised seaweed of New Zealand and southern Australian shores found in the intertidal zone, on rock or on stones, on open coasts through to sheltered situations in harbours and inlets. Thalli are composed of branched chains of bead-like hollow segments joined by thin constrictions, up to 20 –30 (occasionally 50) cm long. It is a monotypic genus in a monotypic family.

Bergquist (1959) examined the size of segments, and how these varied with environmental conditions. The pattern of egg release in *H. banksii* from northeastern New Zealand and its relationship with a number of environmental factors (lunar cycle, light / dark cycle, and height on shore) was investigated by Kusnadi (2014).

Hormosira, considered to be the most successful intertidal fucoid in the southern hemisphere (Kain 2015), has been the subject of a number of studies both in Australia and New Zealand, including research on dispersal, the impact of temperature, the use of sporelings to measure impacts of sewage effluent, and restoration of *Hormosira* on effluent affected sites (e.g., Bellgrove et al. 2010, Doblin & Clayton 1995, Kevekordes 2000). A large body of research on *Hormosira* has been conducted on the east coast of the South Island by research workers from the University of Canterbury (e.g., Alestra et al. 2014, Dunmore 2006, Lilley 2004, Lilley & Schiel 2006, Schiel & Lilley 2007).

Carpophyllum:
This genus is endemic to New Zealand commonly found, fringing the low intertidal margins of rocky reefs. There are four species - *Carpophyllum flexuosum* has the widest geographic range as well as extending into deep waters (to depths of up to 20 m). All species have an elongate main axis that is flattened, with the exception of *C. angustifolium*, where the main axis is rounded or oval in cross section. All species, including *C. angustifolium*, have a holdfast that is a flattened and creeping extension of the main axis, with peg-like attachments along the edges. All species have rounded to oval vesicles, although these are rarely found on *C. angustifolium*, and all have leafy branchlets.

Hybrids between different species of *Carpophyllum* have been reported based on morphological features (Dromgoole 1973, Lindauer et al. 1961, Shears & Babcock 2007), and more recently confirmed with molecular data (Hodge 2009, Hodge et al. 2010). An investigation of contemporary and historical dispersal patterns in *C. maschalocarpum* was conducted using mitochondrial DNA variation in populations throughout its distributional range (Buchanan 2011, Buchanan & Zuccarello 2012). A pattern of high northern diversity and low southern diversity was found, suggesting that this reflects a relatively recent southward expansion with warming of the ocean waters following the last glacial maximum sea surface temperature (LGM).

The development of reproductive structures in species of *Carpophyllum* was investigated by Delf (1939). All four species possess unisexual receptacles, and sporelings appear to develop *in situ* until the development of rhizoids. Dawson (1940) examined the development of conceptacles, and embryology, and reported the fertile period to be December-January. In *Carpophyllum* species sporelings are multicellular before their release from the mesochitonous sheath. Naylor (1954b) observed female conceptacle and young sporelings in 4 species of *Carpophyllum*, basing her observations on preserved material and attached sporelings from field collected material. Dromgoole (1973) completed a PhD on *Carpophyllum* physiology, and noted that "attempts to culture *Carpophylllum* were not entirely successful".

Carpophyllum angustifolium J.Agardh:
Found in the low intertidal to subtidal zones on rock on exposed coasts or in surge channels, *C. angustifolium* thalli have a narrow main axis (about 2–4 mm wide) that is oval in cross section with no leaves towards the base, and thalli grow up to 1.5 m in length. Lower leaves are wider and longer than upper leaves. Vesicles are rare, elongated oval in outline often with a terminal leaflet.

Carpophyllum flexuosum (Esper) Grev.:
Carpophyllum flexuosum is found in the upper subtidal or in deep intertidal pools and channels through to deep water in sheltered areas. Thalli have flattened main axes bearing leaves of various shapes and sizes, up to 2 m high. The leaves at the base are large often with a distinct mid rib, whereas upper leaves are narrower and elongate. The vesicles are rounded to oval with a sharp point and sometimes a leaflet at the tip.

Carpophyllum maschalocarpum (Turner) Grev.:
Carpophyllum maschalocarpum is very commonly found in the low intertidal and upper subtidal, in many areas forming a horizontal band at low water, on more or less vertical rocks, occurring on open coasts as well as in sheltered harbours, lagoons and tide pools. Thalli are flat, with a sinuous main axis to 9 mm wide and narrower branches, up to 1.5 m high. The leaves are marginal and alternate, lying in one plane. The ellipsoid vesicles have pointed tips, often with a terminal leaflet.

Carpophyllum plumosum (A.Rich.) J.Agardh:
Carpophyllum plumosum is found in the low intertidal zone to upper subtidal on moderately sheltered to exposed coasts. Thalli have a flat main axis fringed by pinnately lobed leaves lying in one plane, and grow up to 1 m high. The leaves vary in shape and width, from oak-shaped through to thin hair-like leaves. The vesicles are stalked, smooth and round.

***Cystophora*:**
Cystophora is a large genus found only in New Zealand and Australia, with four species recognised in New Zealand (*C. platylobium, C. retroflexa, C. scalaris, C. torulosa). Cystophora scalaris* is endemic to New Zealand. The systematics of *Cystophora* and the number of species present in New Zealand were recently addressed by Buchanan (2011). Delf (1941) provided some data on the early development stages of *Cystophora*. Sporelings *of Cystophora* (particularly *C. torulosa*) have been used in settlement experiments by the University of Canterbury research group led by Schiel (e.g., Dunmore 2006, Lilley 2004, Taylor 2002).

Cystophora platylobium (Mert.) J.Agardh:
Cystophora platylobium is found subtidally on rock, on open exposed coasts, typically in deep water. Thalli are large with a flattened main axis and the whole plant lies in one plane, growing up to 2 m in height. The branching is alternate from the edge of the main axis and each branch has alternate leaf-like broad, flat lobes. Vesicles are round to oval.

Cystophora retroflexa (Labill.) J.Agardh:
Cystophora retroflexa is found in the subtidal zone to depths of about 10 m, on rock on open coasts. Thalli have compressed main axes, and are narrow edged, often twisted between alternate, fairly openly to densely branched side axes, growing up to 1.5 m or more in height. Side branches arise from the flat surface of the main axis and are bent back at their point of origin (retroflex) often with broad basal wings. The degree of retroflex branching is variable between specimens. The branchlets are irregularly radially branched, round in cross section and elongated, and when mature develop into the receptacles. Vesicles are round to oval on a short stalk.

Cystophora scalaris J.Agardh:
Cystophora scalaris grows in the low intertidal to subtidal zones, on rock, on exposed coasts. Thalli have rigid, flattened main axes that are usually strongly zig-zag, with thalli usually under 1m high. The branching is alternate with side branches arising from the face of the main axis. The side branches are often closely zig-zag in the lower part and divided into secondary branchlets that are closely packed, lying in one plane, cylindrical in cross section and develop into receptacles. The vesicles are ovoid to round, and without a pointed tip.

Cystophora torulosa (R.Br) J.Agardh:

Cystophora torulosa is found in the low intertidal and upper subtidal zones, on rock on open coasts. Thalli have thick main axes that are cylindrical in the lower part and flattened above, growing up to 1 m in height. The side branches grow crowded together with tight clusters of club-shaped to cylindrical branchlets that taper at the base. The vesicles are produced on short stalks, and are round and without a tip.

Landsburgia:
Landsburgia is endemic to New Zealand. One species, *L. quercifolia*, is widespread around New Zealand, whereas the other two species are very restricted in distribution – *L. ilicifolia* is found only in deep water at the Three Kings Islands (Nelson 1999) and *L. myricifolia* occurs subtidally at the Chatham Islands.

Landsburgia quercifolia Harv.:
Landsburgia quercifolia grows in subtidal to deep water, on rock on open coasts. Thalli have cylindrical main axes that taper towards the tip, growing up to 1.5 m high. The branching is alternate with branchlets naked at bases with scars where branchlets or leaves have been shed. The leaves are oak-shaped, flattened with a faint mid rib in the lower portion. In an MSc thesis study, Wood (1949) cultured *Landsburga quercifolia* collected in Wellington, and documented the early developmental stages of this species. Wood documented this species to be monecious with male and female gametes maturing simultaneously on the same plant.

Sargassum:
This is a very large genus with hundreds of species described from around the world and present in temperate as well as tropical regions. The two species below can be distinguished by the presence/ absence of spiny main axes and cryptostomata.

Sargassum scabridum Hook.f. & Harv.:
Sargassum scabridum is found in the low intertidal to subtidal zones, on rock in pools on open coasts and in moderate shelter. The thalli are bushy, with spiny, cylindrical main and secondary axes, growing up to 1 m high. The basal leaves grow to about 5 cm long with a distinct midrib, and serrated margins, whereas the upper leaves are narrow and thinner with distinct midrib and serrated margins. There are conspicuous cryptostomata on basal and upper leaves. The vesicles are round to oval and smooth without apical leaflet.

Sargassum sinclairii Hook.f. & Harv.:
Sargassum sinclairii grows in the low intertidal and upper subtidal zones, on rock and in tide pools and channels, on open coasts in moderate exposure. Thalli are large and bushy, with cylindrical to slightly flattened, smooth main axes, and secondary axes that are slender with alternate side branches. Thalli grow up to 1 m or more in height. The basal leaves grow up to 10 cm long with a midrib and a lobed to serrated margin, whereas the upper leaves are smaller and thinner with faint mid rib and toothed margins. Cryptostomata are absent, and the vesicles are round with a leafy tip.

***Phyllotricha*:**
Phyllotricha verruculosa (C.Agardh) R.R.M Dixon & Huisman, formerly placed in *Sargassum*, is an Australian species that is thought to have been introduced to New Zealand waters in the early period of European contact. The first New Zealand record was from Akaroa in 1845, it is also known from Kaikoura, Fiordland, Otago Harbour, Bluff Harbour and Stewart Island. It grows in the subtidal zone, on rock in sheltered bays and harbours. The thalli have short and stubby axes at the base and basal leaves that are elongated and lobed, with the main axis becoming more slender towards the apex and zigzagging, growing up to 1 m or more high. The leaves are lobed to dichotomously divided, with a faint midrib, and usually have prominent cryptostomata. There are knobbly scars on the axis marking where branches were once attached. The vesicles are round and found scattered among the upper branches, either smooth or sometimes with a short tip.

Marginariella:
This genus is endemic to New Zealand, and there are three species, two which occur mainly from Cook Strait southwards, and one that is restricted to the Antipodes and Bounty Islands, *M. parsonsii* (not treated here). The receptacles in *Marginariella* fringe the inner edge (margin) of the fronds, and the apex, or growing tip, is hidden in the long strap-like blades. The species can be distinguished by the shape of their bladders: oval in *M. boryana* and round in *M. urvilliana*. The holdfast of *Marginariella* species is very distinctive.

The anatomy and early stages of development of *Marginariella* were investigated by Delf (1937) and Naylor (1953a). *Marginariella* is dioecious with 'hermaphrodite' conceptacles, and Naylor (1953a) recorded that in *M. urvilliana* the gametes are produced in the third year and then annually. She collected young plants collected in November which she estimated to be 3-6 months in age, with the parent plants fertile from mid- May to early August in Dunedin. Naylor suggested that the extrusion of eggs was timed with tidal rhythms, and noted that germinating sporelings are found in situ on the parent plant, with early stages of development taking place whilst still attached to the receptacles, and that the young plants drop off the parent and attach to substrates at varying intervals of between 24–48 hours and to 1 week.

Marginariella boryana (A.Rich.) Tandy:
Marginariella boryana is found in the low intertidal to subtidal zones on rock on exposed coasts. The axes bear long, alternate, flat narrow strap-like lateral fronds that are usually less than 3 cm wide, growing to more than 2 m long. The oval vesicles, often with an apical point, develop along the inside margin of the frond.

Marginariella urvilliana (A.Rich.) Tandy:
Marginariella urvilliana is found in the low intertidal to subtidal zones on exposed coasts and is also found in deep water in sheltered areas. The axes bearing long, alternate, flat, serrate to almost entire, lateral fronds that are usually 1–3 cm wide but much wider in sheltered waters (up to 10 cm), and grow to more than 2 m long. The oval vesicles are produced along the inner margin of the frond.

***Xiphophora*:**
The genus is endemic to New Zealand and southern Australia with two species, both of which are present in New Zealand, *X. chondrophylla* in the northern North Island and *X. gladiata* found distributed from the southern North Island and Chatham Islands, through to the Sub-Antarctic. Both species are found in the low intertidal to subtidal zones, on open to exposed coasts. Heine (1932) investigated the development of both species illustrating early development stages up to 3–4 months. Naylor (1954a) documented the early development of *X. gladiata* (as var. *maxima*).

Xiphophora chondrophylla (Turner) Mont. ex Harv.:
Xiphophora chondrophylla has fan shaped thalli, lying in one plane with regular dichotomous branching, usually to 30 (occasionally 50) cm in height. The branches are rounded to flattened, becoming flatter and narrower near the holdfast, and forking frequently towards the tips which are often bluntly rounded.

Xiphophora gladiata (Labill.) Mont. ex Kjellm.:
Xiphophora gladiata thalli are also fan shaped, with regular dichotomous branching lying in one plane, to 50 cm or more in height, with elongated branches and pointed branch tips. Gillanders & Brown (1994) published data on standing stock, density, and seasonality in the population structure of *X. gladiata* in the South Island, where this species is first reproductive in May with a peak in reproductive activity in July and August but with reproductive activity for 8 months of the year. They found the largest size classes examined had the highest percentage of reproductive plants.

2.3.3.2 Changes in the abundance and distribution of macroalgae in New Zealand

Changes in the abundance and distribution of several habitat-forming large brown algae in New Zealand have been recorded in different regions and at a range of scales. The most detailed documentation of declining kelp and large brown algae in New Zealand is presented in the work of Booth (2017) in the Bay of Islands, northeastern North Island. Booth compared aerial photographs for shorelines in the Bay of Islands, using historical and recent images (obtained in November 2009 as part of the Ocean Survey 20/20, Land Information New Zealand - NZ Aerial Mapping Ltd). Booth identified 29 discrete locations "for which there was a time-series of aerial images in which the extent of shallow-reef kelp cover could be clearly discerned, from the 1950s/1960s through to 2009", assembling at least four aerial images for each of these locations in order to estimate changes in kelp cover over time.

Booth (2017) found that the cover of kelp and large brown fucoid algae, which were once widespread on shallow reefs of the Bay of Islands, had been reduced up to 90 % or more in places, and considered the losses to be both intensive and extensive. Booth (2017) cites evidence of widespread urchin barrens being confirmed in the Bay of Islands during extensive dive surveys carried out for the Northland Harbour Board (1985–86) and for the Department of Conservation (1991). Booth (2017) attributes the development and spread of the sea urchin barrens to a reduction in the density of keystone predators, such as snapper and rock lobsters, that are capable of preying on kina (*Evechinus chloroticus*) (Ballantine 2014, Schiel 2013, Shears & Babcock 2002) and other sea urchins.

Ling et al. (2015) examined the dynamics of catastrophic sea urchin overgrazing in temperate systems worldwide and concluded that there was a globally coherent pattern and a discontinuous regime shift "with hysteresis effect of approximately one order of magnitude in urchin biomass between critical thresholds of overgrazing and recovery. Different life-history traits appear to create asymmetry in the pace of overgrazing versus recovery. Once shifted, strong feedback mechanisms provide resilience for each alternative state thus defining the catastrophic nature of this regime shift. Importantly, human-derived stressors can act to erode resilience of desirable macroalgal beds while strengthening resilience of urchin barrens, thus exacerbating the risk, spatial extent and irreversibility of an unwanted regime shift for marine ecosystems".

In New Zealand there is an example of a reversal of sea urchin overgrazing as a consequence of no-take marine protection, providing strong evidence of the key role predators perform in controlling subtidal reef communities (Shears & Babcock 2002). The first marine reserve in New Zealand was established in 1975 (Cape Rodney to Okakari Point, or Leigh Marine Reserve). At the time of protection, the temperate reef ecosystem in the reserve was characterised by a shallow algal zone, crustose coralline algal 'barrens' at 7–15 m which were virtually devoid of macroalgae, and below these depths the kelp *Ecklonia radiata.* Grazing of the urchin *Evechinus chloroticus* (present in densities of more than 4 m^{-2}) maintained the barrens (Cole & Keuskamp 1998, Cole et al. 1990). By the mid-1990s there were documented changes in the density of urchins, and transitions to increased cover by large brown algae (Babcock et al. 1999). It is estimated that after about 20 years following the establishment of the reserve, macroalgal primary productivity increased by approximately 58 % (Babcock et al. 1999). In some areas the transition to kelp cover was still underway more than 25 years after full protection of the area as a no-take Marine Reserve (Shears & Babcock 2002, 2003). The recovery of the kelp populations following the establishment of the reserve indicates that fishing activities on New Zealand's northeastern coast have significant ecological impacts.

Cole et al. (2001) described the expansion of the distribution of the fucoid *Carpophyllum flexuosum* in northern New Zealand, from sheltered habitats considered to be typical of this species to more open water and sea urchin dominated sites. Babcock et al. (1999) reported stands of *C. flexuosum* on the wave-exposed Leigh coast adjacent to the marine reserve in greater abundance than in the reserve. Cole et al. (2001) noted that “the earliest descriptions of subtidal macroalgal zonation at Goat Island (Bergquist 1960) indicated different macroalgal assemblages, including the presence of *C. flexuosum*, from those later quantified by Choat & Schiel (1982)” and postulated that the changes in the distribution of this species may represent decadal scale changes related to climate cycles and storm frequency.

Schiel (2013) reviewed how the interactions influencing macroalgal communities in rocky reefs in New Zealand are currently understood, and, in particular, two conceptual models of the dominant processes that structure brown algal assemblages, namely, trophic interactions /cascades (involving predators, sea urchins and large brown algae) and multi-effects models. Schiel concluded: “a trophic effects model is unlikely to apply to much of the coastline of New Zealand, and that a model involving multiple effects, including bottom-up forces, environmental and climatic influences, species’ demographics, and catchment-derived sedimentation is more appropriate for kelp communities over most of the country”.

Variation in long term intertidal datasets collected at sites along the east coast of the South Island over a 17-year period were analysed by Schiel (2011). The abundance of *Hormosira banksii* varied very considerably (from near zero to 100 %) with large declines as a consequence of severe storms which occurred during El Niño periods. Recoveries occurred although sometimes taking a number of years. Schiel et al. (2016) evaluated decadal changes in temperature and wave climate on intertidal community structure, and concluded that air temperature and the Southern Oscillation Index (SOI) were the most influential variables influencing cover of fucoid macroalgae. SST and wave height were also important but less influential. The study showed the complexity of understanding variables that operate at different spatial and temporal scales, and the value of long term datasets. Detecting change as a consequence of global climate is challenging in benthic communities when both species-specific responses and species interactions are occurring.

Although there are anecdotal accounts of changes in the distribution of *Macrocystis* in New Zealand, particularly in the northern portion of its range in the Marlborough Sounds, and also in offshore Otago sites, baseline data are very limited, and the extent of population and distributional declines remain unclear. Hay (1990a) compared the distribution of *Macrocystis* at its northern limits as mapped by Rapson et al. (1942), with its distribution in the same region between 1984 and 1988. Hay postulated that the retraction in its distribution was linked to increases in sea surface temperature (SST), and particularly to extreme events.

Pirker (2002) investigated a number of aspects of the biology of *Macrocystis* in New Zealand and followed seasonal changes in several populations around Banks Peninsula. Beds at Akaroa Harbour were found to exhibit strong seasonal canopy declines over the summer months, which was attributed to a combination of warmer water temperatures, nutrient limitation, and sediment inputs. The greatest biomass reductions were associated with boat ramp construction activities adjacent to the kelp bed that delivered significant quantities of sediment which physically smothered the sea surface canopy, covered the seafloor, and prevented kelp recruitment for over a year (Pirker 2002). In contrast to the summer declines reported by (Pirker 2002), Fyfe et al. (1999) observed winter declines in populations of *Macrocystis* on the North Otago coast as a consequence of winter storms.

Boyle et al. (2001) who compared survey data gathered in the late 1950s in Doubtful Sound, Fiordland with surveys conducted 30 years later, following the 3-fold increase in freshwater being delivered to Doubtful Sound from the Manapouri outfall. *Hormosira banksii* once the dominant low shore alga, had almost completely disappeared from the Sound, and other species, formerly found in the inner Sound, were present only closer to the seaward regions of the fiord.

2.3.3.3 Anthropogenic and environmental stressors to macroalgae in New Zealand

The impacts of stressors on macroalgae vary according to the scale at which they operate, both in space and in time. Global scale stressors, such as climate change and biotic homogenisation, and local scale stressors, such as habitat modification, pollution and marine resource harvesting, operate at the level of individuals through to communities and ecosystems (Mineur et al. 2015). Stressors may also differ in their impacts depending on whether they are episodic, long term or acute, and depending on the nature of their interaction with stressors and impacts.

Sedimentation

In New Zealand some stressors that are understood to have significant impacts on the distribution of macroalgae have been studied in considerable detail. The impact of land-derived sediments on the nearshore environment is recognised as a significant stress for New Zealand coastal marine communities (Schiel & Howard-Williams 2016). Sediment smothers sedentary species, scours and abrades organisms as it is redistributed and resuspended with wave action, interferes with the settlement of sporelings and young thalli, and reduces the light penetration through the water column thereby affecting photosynthesis (Airoldi 2003).

Seers & Shears (2015) examined spatio-temporal patterns in coastal turbidity across an estuarine to open coast gradient in the Hauraki Gulf, analysing 22 years of data. They found strong relationships between turbidity and meteorological and oceanographic variables and suggested a number of potential mechanisms by which climate change may increase turbidity in the future. More sediment is likely to be delivered to the marine environment as a consequence of rising sea level, an increase in the frequency and intensity of storm events with associated rainfall, and a consequent increase in coastal erosion (Lundquist et al. 2011). Resuspension of sediments by waves, swell and tidal flows will all contribute to the turbidity at coastal sites.

In New Zealand the effects of sediments on settlement and post-settlement have been investigated for several key habitat-forming large brown macroalgae. Schiel et al. (2006) found that the attachment of sporelings to the substrate was disrupted by sediment, with “a light dusting of sediment” reducing the percentage settlement of zygotes of *Hormosira* by 34 % and *Durvillaea* by 71 %, and a complete cover of sediment prevented attachment altogether. Sedimentation impacts on *Carpophyllum flexuosum* and *Ecklonia radiata* have been examined in the north-eastern North Island (e.g., Ainley 2013, Hughes 2011). It was found that prolonged exposure to accumulated sediments resulted in thalli decaying, with the greatest impacts in areas under low light conditions.

Light availability

Primary productivity in the marine environment is dependant on light availability, a reduction of which will potentially alter the distribution, community composition, and productivity of key benthic primary producers leading to a reduction in habitat and energy provision to coastal food webs (Desmond 2016, Desmond et al. 2015). In a study comparing the underwater light environment of macroalgal dominated shallow subtidal rocky reef habitats on a coastline modified by human activities with a coastline of forested catchments in southern New Zealand, it was found that light attenuation was more than twice as high in shallow subtidal zones along the modified coast than on the forested coastline. The macroalgal biomass was 2–5 times greater adjacent to forested coasts, even in shallow water (2 m). The differences in observed biomass between modified and forested coasts were best explained by the long-term light dose, with light availability over the study period differing by 60 and 90 mol photons m^{-2} at 2 and 10 m, respectively. Higher biomass on the forested coast was driven by the presence of larger individuals rather than species diversity or density (Desmond et al. 2015, Desmond 2016). Desmond (2016) reported that within the East Otago sites investigated, light attenuation resulted in significantly lowered biomass, size and presence of canopy forming macroalgal species. He concluded that a number of macroalgal species may be living at the extreme end of their photosynthetic abilities, and further light limitation may compress depth limits and alter community structure. Desmond also noted that the changes in macroalgal biomass and structure due to light limitation significantly reduce habitat and food availability to upper level consumers, resulting in reduced secondary biomass.

Multi-stressor studies

Multi-stressor investigations of habitat-forming brown algae have also examined the responses to different types of disturbance e.g., disturbance of the canopy has been investigated by Schiel & Lilley (2007), and Schiel & Lilley (2011) demonstrated that the experimental removal of a dominant intertidal fucoid in southern New Zealand had numerous community effects up to 8 years later. Alestra et al. (2014) investigated the impacts of turfing geniculate coralline algae and sediment on fucoid recovery dynamics and their influence on assemblage net primary productivity (NPP). Alestra & Schiel (2015) used short- (48 h) and long-term (eight weeks) experiments to examine the effects of different combinations of nutrient, sediment and temperature on the early life stages of the habitat-forming fucoids *Hormosira banksii, Cystophora torulosa*, and *Durvillaea antarctica*, and how the persistence of algal beds will be affected by different combinations of stressors. Tait (2010) examined macroalgal productivity at a range of scales, in intact canopies as well as in response to differing levels of disturbance (Tait & Schiel 2011a, 2011b), under different temperature regimes (Tait & Schiel 2013). The interactions between elevated temperature and reduced light (simulating reduced water clarity) on *Ecklonia radiata* were investigated by Rodgers (2014), and Rodgers & Shears (2016) modelled *Ecklonia* forest primary production using biomass, in situ measurements of photosynthesis and light.

Hughes (2011) found that *Carpophyllum flexuosum* forests exhibited a partial tolerance to pollution, as well as to grazing by sea urchins, something also reported by Cole et al. (2001) and Cole & Haggitt (2001). The communities associated with *C. flexuosum* forests were strongly influenced by sedimentation levels, with generally lower abundances and diversity of encrusting sponges, compound and stalked ascidians, bryozoans, crustose coralline algae (CCA) and a number of gastropod species at high sediment sites in comparison to low sedimentation sites. Hughes (2011) reported that the removal of *C. flexuosum* canopies had large effects on the understorey communities, suggesting that this species plays a key role in the facilitation of the understorey assemblages.

Ocean acidification

The responses of macroalgae to ocean acidification have been summarised recently (Law et al. 2017). Research on south-eastern New Zealand kelp forests suggests that macroalgae sensitivity reflects the variable DIC physiology of different species (Hepburn et al. 2011). Spore germination in the kelps *Macrocystis pyrifera* and *Undaria pinnatifida* were little-affected by changing pH (Leal et al. 2017a, 2017b, Roleda et al. 2012). Dense kelp and fucoid beds in New Zealand coastal waters attenuate the flow of seawater (Cornwall et al. 2015b), and when combined with an increase in pH during the daytime, the presence of intact forests may buffer the impact of OA on calcifying species within the kelp bed (Cornwall et al. 2013, Hurd 2015). However, pH fluctuations may also enhance the negative impacts of reduced pH on growth and calcification of calcified algae (e.g., Cornwall et al. 2013). As Law et al. (2017) note “Understanding the dynamics and impacts of changing conditions within an ecologically relevant context is an important research priority, given the high biomass of macroalgae on New Zealand’s rocky reefs”.

Climate change

The possible impacts of climate change on New Zealand coasts have been summarised (e.g., Boyd et al. 2011, Lundquist et al. 2011). In an analysis of long term data from intertidal communities on the east coast of the South Island Schiel et al. (2016), found that air temperature and the Southern Oscillation Index (SOI) were the most influential variables on cover of fucoid macroalgae, with sea surface temperature and wave height important but less influential.

Introduced species

The impacts of introduced species are complex and will also be influenced by global climate change (Floerl et al. 2013). With increasing human population and expanding human transport networks, there is recognition that the issues associated with global homogenisation are likely to increase (Mineur et al. 2015). In the New Zealand context, after monitoring coastal reefs around Auckland, Shears (2017) observed "Perhaps the greatest change in reef assemblages over the last seven years is the appearance of, and increase in, abundance of invasive species." Shears noted that long term monitoring sites provide the opportunity to document the spread of species, the rate of increase in populations and to examine the potential impacts on native biota. While there is some debate about the extent of impacts of non-native species on native assemblages and ecosystem functions, intact ecosystems without non-native components have important conservation values as well as meeting intergenerational responsibilities for wider values associated with diversity. Few of the ecological studies on impacts of non-native species have involved detailed examination of native biota, rather grouping species by morphological characteristics and thus overlooking potential shifts in the species composition and functioning. Jiménez et al. (2017) concluded following a study of the epibiota of *Undaria* and other large brown algae in southern New Zealand "that abundances of epifauna at the ecosystem level will be reduced if *Undaria pinnatifida* displaces more structurally complex native seaweed species that host more diverse and dense epifaunal assemblages. The findings suggest that morphological complexity may be key to predicting the impacts of invasive seaweeds on epifaunal assemblages, and potentially on food webs, in other geographic regions."

Human induced changes

Human induced changes in coastal reef communities include the impacts of recreational activities and, in particular, trampling. This has been documented internationally, and in New Zealand impacts have been studied on coralline turf communities (Brown & Taylor 1999) and on *Hormosira* (Schiel & Taylor 1999). In the case of reef flats, trampling can crush canopy plants of *Hormosira*, leading to significant loss of biomass (as much as 25 % after 10 tramples on a single tide), as well as exposure and loss of understorey species. Brown & Taylor (1999) considered the importance of coralline algal turfs as habitat for small invertebrates that are key prey items for a range of fish species "Given the likely importance of these abundant & productive animals in the rocky reef ecosystem, & their vulnerability to low levels of trampling by humans, we conclude that the effective management of marine protected areas may necessitate total exclusion of humans in some cases.".

2.3.4 Macroalgae monitoring in New Zealand

2.3.4.1 Macroalgae indices and monitoring programmes and tools in New Zealand

In New Zealand macroalgae have been recommended as being suitable for use in monitoring programmes based on a range of different research studies, but the implementation of monitoring programmes has not been consistent at local, regional or national scales. The limited extent of baseline knowledge and data is also problematic, providing a very scant framework for the interpretation of change in the environment or in the status of species and populations. There are some examples of long-term datasets gathered by researchers and through programmes implemented by Regional councils and the Department of Conservation.

Early studies on the structure of macroalgal assemblages in New Zealand were qualitative, frequently focusing on zonation patterns of intertidal organisms (e.g., Dellow 1950, 1955). A general description of the patterns of distribution and abundance of habitat-forming fucalean and laminarian algae was published by Choat & Schiel (1982), who also hypothesised about the processes underlying the patterns. Various authors focused on particular species including carrying out ecological and physiological research (e.g., Dromgoole 1973, Hay 1977, Schiel 1980, 1985). Schiel (1988) summarised the understanding of algal assemblages in the shallow subtidal in northeastern New Zealand, and in 1990 reviewed the available data on the structure, interactions and demography of macroalgal assemblages from Manawatāwhi Three Kings Islands, northeastern North Island, Wellington, Chatham Islands, Fiordland and the Auckland Islands. The data presented (abundance and percent cover) were for large brown algae (3–5 species per region) with some data on large invertebrates (particularly kina). Schiel (1990) noted the need for more information about the "specific life history and phenological characteristics of particular species and the demographic consequences of settling and growing under different conditions". Hurd et al. (2004) summarised the studies that had reported quantitative macroalgal abundance at various locations in New Zealand.

Habitat classification has been the focus of a body of work carried out by Shears et al. (2004) and Shears & Babcock (2007). In a series of reports published by the Department of Conservation, Shears and co-workers documented habitat types and community structure of subtidal reefs around New Zealand, the results of a nationwide survey of mainland New Zealand's subtidal benthic reef communities between 1999 and 2005 (Shears 2007, Shears & Babcock 2004, 2007). These reports were based on surveys of shallow (less than 12 m depth) subtidal communities, and both national and regional patterns were described. Analyses included dominant seaweeds, mobile macroinvertebrates and fishes, and biogeographic patterns as well as non-biological factors (e.g., water clarity and wave action) were discussed in relation to the biotas of different regions. These studies documented the biota at a higher level of taxonomic resolution than had previously been recorded. Macroalgae were assigned to 23 species groups, and algal measurements were converted to biomass to enable comparisons between groups.

As summarised by Shears et al. (2004) "Reliable classification of habitats is essential for describing large-scale patterns in communities …, detecting large-scale changes in habitats over time (Babcock et al. 1999; Shears & Babcock 2003), and understanding the mechanisms responsible (Shears & Babcock 2002)." There is a need to standardise approaches and the descriptors employed to ensure that observations made over time and by different people can be compared and interpreted.

Shears & Babcock (2007) provided detailed discussions of the distribution of large brown habitat-forming species around the New Zealand mainland, and how the communities present at national and regional scales were most strongly related to turbidity and exposure. They concluded that "The structure of algal and benthic communities was most strongly associated with water clarity, suggesting that community structure varies most strongly across a gradient from coastally influenced sites (e.g., shallow areas or embayments) with high turbidity to more oceanically influenced locations (e.g., offshore islands)." Shears et al. (2008) evaluated biogeographic classification as a tool for conservation planning in New Zealand. They used systematically collected community data [metrics derived from abundance and presence–absence data for macroalgae (107 species) and mobile macroinvertebrates (44 species)] and compared these with 6 existing biogeographic classifications. They defined 11 habitat types, assessed the specific abundance of dominant species, and concluded that the habitat classification system was biologically meaningful based on the abundances of habitat-forming algal groups. They also considered that their results demonstrated that the habitat types were able to be reliably categorised visually which "has important applications in classification and mapping of the marine environment".

Shears (2010) repeated sampling he had carried out in November 1999 at six subtidal rocky reef sites on Wellington's south coast as part of this national survey. His analysis indicated that there was little change over the 10-year period, which he suggested was indicative of a high degree of stability in the "dominant components of the reef communities of the Wellington south coast". He also noted the dominance of four large brown algal species and that the depth distributions of these species remained relatively constant across the two surveys: *Carpophyllum maschalocarpum* (typically in water less than 3 m depth), *Lessonia variegata*, *Landsburgia quercifolia* and *Ecklonia radiata* (typically dominant at depth more than 5 m). The mean abundance of dominant large brown algal species at each site was found to be relatively similar between the two surveys, except for *E. radiata* which in 2010 occurred at lower abundance across most sites, and *Marginariella urvilliana* which was more abundant in 2010 at a number of sites. The dominant substratum cover on the reef at all sites was crustose coralline algae in both 1999 and 2010, and data were also presented on changes in the distributions of red turfing species, and *Caulerpa* spp.

Schiel (2011) discussed biogeographic patterns and temporal trends in intertidal communities around New Zealand based on both long term and one-off survey data. The results of intertidal surveys on moderately exposed rocky platforms, conducted between 2000 and 2002 at 73 sites around New Zealand during spring, were compared using percentage cover data for macroalgae, barnacles and mussels. Long-term data were also presented from three regions of the east coast of the South Island (Cape Campbell, Kaikoura, Moeraki). Much of the data presented refers to dominant fucoids (particularly *Hormosira banksii*) but the level of taxonomic resolution was not specified in the paper. Schiel concluded that the intertidal communities could be mapped to "four general regions, which relate to coastal morphology and the characteristics of the dominant water masses around the country". He reported that the "entire west coast was dominated by invertebrates, particularly mussels and barnacles" and the mid-low intertidal zones on the east coast were dominated by fucoid algae.

Hewitt et al. (2014) reviewed marine environmental monitoring in New Zealand to assess whether a comprehensive programme could be developed based on existing sampling programmes. They evaluated which datasets would be best for the detection of long-term trends at a national scale in the state of the New Zealand marine environment, provided recommendations on a robust monitoring design focused around present monitoring, and proposed improvements to data collection, analysis and storage, to provide greater cohesion for marine environmental reporting at the national scale. In Hewitt et al. (2014) macroalgae were treated in the category "Reef macroalgae and macrofaunal communities" which was defined as **"**all seaweeds and invertebrates living on the surface of rocky areas, sized over 0.5 mm, generally identified to genera or species level with either counts made or percent cover estimated. This is a composite variable i.e., the data can be analysed as community composition, as one of a number of biodiversity measures (e.g., species richness, species evenness) or even as individual species or biomass.". Hewitt et al. (2014) noted that these types of communities in both the intertidal and subtidal zones, "affect ecosystem goods and services, fish biomass and nutrient and oxygen fluxes between the sediment and the water column. The community data can be used to calculate biodiversity indices and contribute to the calculation of ecological integrity and ecosystem goods and services. However, there are yet no methodologies for relating community structure to community health." They also noted under the section "Responsiveness" that reef communities have been demonstrated to be affected by trampling, anchoring, sedimentation and temperature, and are likely to be affected by eutrophication and climate change (e.g., sea level rise, changes in weather patterns, seawater temperatures and acidification.

Hewitt et al. (2014) summarised the degree of monitoring in New Zealand, noting that there is relatively little long-term data both intertidally and subtidally, with many surveys conducted only once or twice. A wide variety of methods have been employed (e.g., with respect to quadrat size and frequency). They note "A consistent method of monitoring and classifying subtidal reef communities at a fixed depth and slope was developed by Shears (2007), but sampling to determine natural temporal variability over time in this classification has not yet been undertaken". Hewitt et al. (2014) concluded that although the variable "Reef macroalgae and macrofaunal communities" may be able to be used in reporting on biodiversity, ecosystem goods and services and ecological integrity, there are significant issues such as the absence of standardised methodology, a poor current understanding of natural variability, and inadequate national coverage, and thus it is "presently not suitable for being included in an MEMP". There is no mention of the degree of taxonomic resolution which would need to be evaluated when arriving at standardised national methodologies.

Monitoring in the marine environment by the Department of Conservation in marine reserves and protected areas has focused largely on fish and large invertebrates (e.g., Cole et al. 1990). McCrone (2001) summarised monitoring up to the end of 2000, documenting the number of reports, what was monitored, and the frequency of monitoring in New Zealand marine reserves and protected areas. Most commonly, monitoring was focused on reef fishes, rock lobster, pāua and kina, although "selected algal species" were included in studies in 10 of the reserves. Willis (2013) reported on scientific and biodiversity values of marine reserves, noting that marine reserves in New Zealand "have the primary function of protecting spatially delimited areas from the effects of fishing". The report summarised research work that has been undertaken in marine reserves, as well as monitoring activities (e.g., design, methodology, trends in indicators) with the primary focus on finfish and rock lobster.

Cole (2003) considered the time frame over which monitoring in marine reserves should occur. Based on the experience in northern New Zealand where "effects for exploited species may emerge over periods of decades, and that 10 years of annual sampling should be regarded as a minimum for monitoring", Cole noted that "in the oldest existing reserve, important changes were occurring 20 years after imposition of reserve status. The magnitude of response to marine reserve protection will also depend on the severity of impact that has already occurred."

Monitoring for Taputeranga Marine Reserve (TMR) began with the collection of baseline data in 1998 prior to reserve establishment. Pande & Gardner (2009) described the use of four macroalgal, four macroinvertebrate and eight fish species in surveys conducted at eight sites (three inside, five outside the proposed reserve) over three years before the establishment of the Taputeranga Marine Reserve on Cook Strait. Pande & Gardner (2009) included the four large habitat-forming species (*Ecklonia radiata, Macrocystis pyrifera, Carpophyllum maschalocarpum,* and *Lessonia variegata*) in the baseline surveys because of their common occurrence, large sizes thus contributing a significant area of canopy, their perceived importance as a food source and/or an important three-dimensional habitat for many invertebrate and fish species, and because *E. radiata* has been shown to change in abundance in response to changes in abundance of herbivorous organisms using them as a food source. This baseline dataset was used to estimate temporal and spatial variability in size and abundance of these taxa, and the authors suggested that it would be available to quantify taxon-specific changes in size and abundance once the marine reserve was established.

Zintzen (2014) summarised the history of monitoring within this reserve and the objectives of this monitoring, and presented an audit of the methods employed (field and analytical approaches), quality control, and data management. The audit presented was primarily intended to evaluate whether monitoring methodologies were fit-for-purpose and outputs were meeting monitoring objectives. However, since establishment of the reserve in 2008, no data are being collected for macroalgae (including the frequency and abundance of the large invasive kelp *Undaria pinnatifida*). Annual monitoring conducted by DOC has focused on common fish species, kina (*Evechinus chloroticus*), pāua (*Haliotis australis* and *H. iris*) and rock lobster (*Jasus edwardsii*) at eight sites (three sites within the marine reserve and five control sites outside the reserve).

The Kapiti Marine Reserve (KMR) was established in May 1992 following a single baseline biological survey. Battershill et al. (1993) recognised thirteen different habitats around Kapiti Island – with 10 of the thirteen defined by the presence of particular macroalgae (e.g., *Carpophyllum* spp., *Ecklonia*, *Glossophora* (now *Dictyota*), *rhodoliths*, *Ulva*, and three defined by invertebrates. Pande & Gardner (2012) surveyed macroalgal, macroinvertebrate and fish species for abundance and/or size at two sites inside and two outside KMR and then conducted tests to quantify biological responses after 8 years of protection (on the reservation status (inside vs outside KMR) and temporal change i.e., 1992 baseline vs 1999/2000). Reservation status has had a significant effect on abundance or size for four fish species (banded wrasse, blue cod, butterfish, blue moki), but not for heavily fished macro-invertebrates (two species of pāua, kina, rock lobster) nor for macroalgae.

A baseline report on Ulva Island / Te Wharawhara Marine Reserve in Stewart Island (Wing 2006), included consideration of macroalgae as indicator species, particularly identifying large brown algae and the kelp *Macrocystis*. Wing concluded "the continued monitoring programme should consider abundance and distribution of the major habitat-forming organisms in the reserve "as these habitat-forming organisms are effective indicators of disturbance (dredging and trawling) as well as nutrient pollution and oceanographic variability".

A series of baseline reports and maps have been prepared covering areas within the Northland region for the Department of Conservation (e.g., Doubtless Bay - (Grace & Kerr 2005), Mimiwhangata - (Kerr & Grace 2005), Motukaroro Island – (Kerr & Grace 2006a, 2006b) Mangawhai to Ahipara habitat map (Kerr 2010) and for Friends of the Hauraki Gulf for Waiheke Island (Kerr & Grace 2013). Zintzen (2014) indicated that a National Monitoring Framework was being developed by DOC under the Marine Ecological Integrity Programme, with the intension of integrating previous monitoring undertaken within New Zealand's marine reserves. The key deliverables for the programme included:

1. A marine component for the Department's Inventory and Monitoring Toolbox. Toolboxes are detailed documents developed to standardise sampling methods across New Zealand. Each sampling method implemented in marine reserves will have its own toolbox element.
2. A reporting component which will include report cards for individual marine reserves and inputs into the DOC Annual Report.
3. The inclusion of marine reserve monitoring in business plan prescriptions.
4. The development and implementation of marine reserve data storage and management protocols.

The Marine Reserve Report Cards are now being produced and made available on the DOC website. In 2017, report cards for Cape Rodney-Okakari Point, Tāwharanui and Ulva Island-Te Wharawhara Marine Reserves were published in web and brochure format, and in 2018 for Taputeranga Marine Reserve. Report cards for other marine reserves are planned for the future as well as updated versions (with additional indicators) for the reserves already reviewed. (http://www.doc.govt.nz/nature/habitats/marine/type-1-marine-protected-areas-marine-reserves/marine-reserve-report-cards/report-card-rationale/).

The Report Cards present the "health" of individual marine reserves, summarising the health status and trend, based on a selection of measures. The health status may be rated as "superior, good, fair or poor, and the trend over five years is rated increasing, stable, declining or undetermined, based on monitoring data and research studies. Currently the report cards consider habitat, seafood species, water quality, marine pests, and surrounding land. Currently, no macroalgae are being used as indicators at any of the Marine Reserves for which report cards have been prepared.

There is an increasing trend to use local ecological knowledge (LEK) and citizen science to extend observations and baseline information, both temporally and spatially. Taylor et al. (2011) used local knowledge (expert divers) to examine recovery at the Poor Knights Marine Reserve. Although kelp forest was one of the habitat categories designated by the researchers, there were few observations and no trends recorded in the observations for macroalgae, in contrast to detailed observations for specific fish species, and conspicuous invertebrates.

Jones et al. (2016) gathered local ecological knowledge from fifty trawl fishers, and kelp was one of the five most commonly mentioned biogenic habitats. In some areas temporal and spatial reduction in the habitats/species were noted. On the Hawke's Bay coast, the "*Wairoa Hard*" and "*Clive Hard*" are inshore reefs in shallower depths that were characterised mainly by the presence of sometimes dense kelp. Jones et al. (2016) cite reports of the removal of kelp forests from the Wairoa Hard during the 1960s and 70s, where it was regarded as a nuisance for net fishing.

Within the Ngai Tahu rohe there have been Customary Fishery Protection Areas established (CPA Taiāpure-Mātaitai). Ten years of monitoring in CPAs, through Te Tiaki Mahinga Kai (TMK), are providing some baseline data (e.g., East Otago Taiāpure presented in this study). It is not clear whether local fishery restrictions are providing for effective restoration of macroalgae within stressed coastal ecosystem, within socially acceptable timeframes.

2.3.4.2 Macroalgae as bioindicators of nutrients and heavy metal enrichment in New Zealand

Hurd et al. (2014) provided a detailed summary of the impacts on seaweeds of different types of pollutants, ranging from metals (including bioaccumulation and toxicity), oil, synthetic organic chemicals, eutrophication (both from sewage and other anthropogenic nutrient enrichment), radioactivity and thermal pollution. The bioaccumulation of certain compounds makes some seaweeds suitable for biomonitoring and there has been some work on this topic in New Zealand e.g., *Hormosira* as a biomonitor of Fe and Zn (Cooke et al. 2004). Leal et al. (2016) examined the impact of exposure to chronic and high concentrations of copper in both *Macrocystis* and *Undaria* in New Zealand and found that there were differential impacts on meiospore germination, and arrested development of gametogenesis occurred in both species.

The use of stable isotope signatures (particularly nitrogen) as an indicator of nitrogen loading (particularly in *Ulva* and *Carpophyllum*), has been explored, including testing the utility of this approach to measure before and after changes at sewage outfall sites (Barr 2007, Barr et al. 2013, Dudley & Shima 2010, Fry et al. 2011, Rogers 2003).

2.3.4.3 Survey questionnaires

Eighteen responses were received from either research workers or teams, with all respondents involved in some way with the collection of baseline data on macroalgae in different parts of the country, in most cases with one-off surveys, but in several cases with longer term collection of data and monitoring of specific sites (e.g., Marine Research Group at the University of Canterbury). Macroalgae are being used in the studies for a variety of reasons including because of the importance of macroalgae as dominant components of nearshore reef systems, their potential for providing an “early warning signal” for human induced impacts, to follow the spread of invasive pest species, and in order to document changes in particular habitats over time, both natural and anthropogenic. In terms of future focus, there was widespread agreement from respondents about the value of monitoring macroalgae and the desirability that this occurs within a national framework – to determine and assess possible ecosystem-level changes in intertidal and subtidal ecosystems as a result of disturbance (from local to global scales). In response to questions about data storage, respondents largely referred to databases held personally, and there was little evidence of use of larger data repositories or shared data platforms.

Of the seven regional and unitary Councils that responded to the questionnaire, six are involved in monitoring using macroalgae. The frequency of monitoring, the species/communities monitored, whether intertidal or subtidal, in estuaries or on open coasts, and the methods employed vary. In some cases, monitoring has happened at 5 to 10 years intervals, and in other cases annually or quarterly. Where monitoring is occurring, it is primarily to enable reporting on the state of the environment. In some cases, once a baseline was established on-going monitoring was not deemed necessary. Data are stored in variety of formats largely within spreadsheets or databases, and in some cases as GIS layers within maps. For four of the respondents, changes in coastal macroalgae were relevant to their regions and specific concerns included effects of sedimentation, climate change, and invasion of non-indigenous species. Four of the councils have experienced outbreaks of nuisance quantities of macroalgae. In terms of future monitoring, respondents indicated interest in adopting technologies such as drones, and issues of cost constraining sampling particularly subtidally.

The Hawkes Bay Regional Council current State of the Environment (SoE) monitoring is undertaken across various marine habitats within its boundary (estuarine, intertidal soft sediment, intertidal rocky reef, and coastal water quality), and providing valuable data on the state of these systems, essential for the evaluation of trends (stability, deterioration, and improvements) in relation to land-based stressors (Hashiba et al. 2014). Hashiba et al. (2014) note the lack of information concerning how physical and biological subtidal habitats may have changed through space and time across the Hawke’s Bay region, and that “With the exception of limited subtidal rocky reef assessments done within Te Angiangi Marine Reserve, there are generally no data on how these systems are faring elsewhere, especially in terms of habitat distributions, habitat quality, and overall biodiversity.”. They note that it would be beneficial to target biogenic habitats of significance (mussel beds, kelp and sponge habitats) in rocky reef monitoring, and include species of cultural importance (kina, pāua etc). “It would also be advantageous to ensure sampling methodologies are consistent with the depth-stratified sampling presented in Shears & Babcock (2007) including, where relevant, methodological toolboxes for environmental monitoring currently being developed by DOC. Ensuring consistency with those methodologies has the advantage in so far as results generated can be compared at both regional and national scales. Assuming robust methodologies are developed, monitoring of this nature would not be required on an annual basis e.g., every 3–5 years would be sufficient. The subtidal monitoring would be consistent with the existing SoE monitoring aims and objectives.” There is also interest in understanding the effects of specific stressors (e.g., sedimentation, harvesting, vehicles on beaches), the linkages between intertidal and subtidal environments, and the abundance of kaimoana species and condition of stocks.

Council respondents stored their data in a variety of datasheets, reports, and in GIS layers within maps. There is no shared national platform for regional or local data.

The Auckland Council established a long-term programme monitoring subtidal reefs on the east coast of the region to investigate any potential impacts of urban development. Shears (2017) summarised the results of subtidal reef marine monitoring of the east coast of Auckland between 2007 and 2013. Annual sampling of shallow subtidal reef communities at took place at Meola Reef, Campbells Bay, Torbay, Long Bay, Manly, Stanmore Bay and Waiwera, and in addition, monthly measurements of sedimentation were taken at each site using sediment traps. Shears found that the reef community at all sites was dominated by a canopy of large brown macroalgae, with the canopy composition varying in relation to the differing physical conditions among the seven locations. Since 2007 the overall canopy composition remained relatively constant among the different locations, with the exception of a relatively large increase in the kelp *Ecklonia radiata* at some locations. The introduction of the invasive kelp *Undaria pinnatifida* to Meola reef was the other major change in algal canopies over the monitoring period. Shears reported that the biological communities on the reefs surveyed have been relatively stable since 2007, and overall species assemblages on the reefs remain similar to those recorded in older datasets from 1999, other than the increased prevalence of introduced species on shallow reefs.

Shears (2017) observed that the East Coast Subtidal Reef monitoring programme is the only long-term subtidal reef assemblage monitoring programme in New Zealand, and it is providing “an important benchmark to understand changes associated with human impacts, such as the introduction of invasive species and their effects on native reef ecosystems, as well as documenting and understanding how climate change will impact on temperate reef assemblages in the future. Continuation of this programme is important given Auckland’s rapidly growing population, increasing threats and demands on ecosystems and resources in the Hauraki Gulf, and the interacting effects of climate change on these ecosystems”.

Shears (2017) identified some ways in which additions and changes to the sampling methods would increase the information available to interpret trends, enable data to be compared with other studies, and, also to improve the taxonomic resolution of the monitoring programme, specifically:

1. **Measurement of algal canopies**: changes in canopy cover and whether more open spaces are appearing on the reefs cannot be ascertained from the density data currently being recorded. Shears recommended that this include estimates of percentage cover of algal canopies, as this is an easily measured metric and widely used in temperate reef studies, as well as the use of habitat mapping based on satellite or aerial imagery to enable evaluation of change over time.
2. **Improvements in the taxonomic resolution and accuracy of sampling benthic encrusting assemblages**: there is a relatively large diversity of encrusting sponges and ascidians that are not currently being sampled, and gathering data on these communities is particularly important in determining overall species diversity as well as increasing the chances of documenting any invasive species. Testing the use of photoquadrats within the dense algal forests was recommended.
3. **Increased use of temperature and light loggers**: additional information on temperature and light combined with nutrients from regional water quality monitoring programmes may provide insights into interannual variation in key species.
4. **Extension of the sampling to areas beyond the influence of any new developments**: given the spread of developments within the Auckland region, the inclusion of additional sites for baseline records was recommended.

2.4 Discussion

Changes are occurring in coastal marine ecosystems worldwide, particularly affecting habitat-forming kelp forests and subtidal macroalgal beds. These changes include declines, loss and, in some cases, extinction of habitat-forming species. Fundamental ecosystem shifts from canopy-forming species to turfing algae have been documented, with profound impacts on the associated biota. Increased temperature and anthropogenic stressors (e.g., pollution, sedimentation, coastal developments) have been linked to declines in coastal vegetation.

Long term programmes monitoring macroalgae are in place in a number of countries worldwide, and a range of indices based on macroalgae have been developed to assess ecosystem health. The utility of macroalgae for monitoring is based on their distribution as conspicuous components of coastal systems, their size and relative ease of identification, and their roles in provision of diverse ecosystem services. The areal extent of an ecosystem is a commonly used index of its indicator value.

Macroalgae have potential to be used as

- ecological indicator surrogates to provide information about ecological systems and sensitivity to stressors. Sometimes referred to as sentinel species, biomonitoring species, or ecological-disturbance indicator species, these may be chosen because of their known sensitivity to specific stressors e.g., temperature increases, impacts of trampling, reduction in light as a result of changes in water quality, responses to heavy metals, etc.
- management surrogates, primarily used to facilitate achieving management goals, such as maintaining biodiversity, increasing ecosystem resilience, or capacity to report on the state and trends in the condition of the environment. Such management surrogates may be so-called umbrella species, biogenic habitats or flagship ecosystems (e.g., coral reefs, rainforests, kelp forests) that play a critical role in provision of complex ecosystem services and maintenance of biodiversity (Hunter et al. 2016).

Large brown macroalgae in New Zealand coastal ecosystems meet the criteria for both indicator species/ecological indicator surrogates and for management surrogates. There is interest from a number of agencies and stakeholders (central and regional government, iwi) for approaches that will allow comparison of data at regional and national scales, increase understanding of the effects of specific stressors (e.g., sedimentation, harvesting, vehicles on beaches), the linkages between intertidal and subtidal environments, and the abundance of kaimoana species and condition of stocks.

If macroalgae are to be used in coastal monitoring programmes in New Zealand, there is a need to standardise approaches and the descriptors employed to ensure that observations made over time and by different people can be compared and interpreted. In addition, at present there is insufficient baseline data to enable recognition of trends and status, and a poor current understanding of natural variability. Shears (2017) has presented ways in which the appropriate selection of sampling methods in monitoring programmes would increase the information obtained and thus enable trends to be interpreted, and data to be compared with other studies. Shears also recommends methods that will improve the taxonomic resolution of monitoring programmes.

3 SUSCEPTIBILITY OF SELECTED NEW ZEALAND LAMINARIALES AND FUCOID TO SPECIFIC ENVIRONMENTAL STRESSORS

3.1 Introduction

Two key stressors for coastal communities in New Zealand are elevated temperatures (a consequence of long-term warming of coastal waters - Schiel 2013), and the impacts of increasing sedimentation (and thus decreasing light availability) as a consequence of altered land use practices (Alestra & Schiel 2015). A number of studies have examined the sensitivity of early life stages of fucoid algae to specific stressors (e.g., Kevekordes 2000, Schiel et al. 2006; Irving et al. 2009, Andrews et al. 2014). Studies on the sensitivity of the widely distributed intertidal species *Hormosira banksii* have been carried out in Australia and New Zealand. Alestra & Schiel (2015) tested the combined effects of combinations of stressors in a series of laboratory experiments testing the responses of *H. banksii*, *Cystophora torulosa*, and *Durvillaea antarctica* to combinations of increased nutrient and sediment loads, and temperatures.

This Objective investigates the responses of early life stages of a range of large brown algal species to specific stressors. The species selected are commonly found and have not been investigated in detail until now. In addition to building knowledge about how the selected species respond to specific stressors, we were also examining whether any of them could be candidates to serve as indicator species for tests of ecosystem health and multiple stressors.

3.2 Methods

Small scale culture experiments were used to investigate the responses of selected Fucales, Laminariales to specific stressors (temperature, shading, sediment), using species readily available in Wellington, and informed by previous research. The experiments were conducted on the early life stages of one species of Laminariales (*Lessonia variegata* J.Agardh) and four species of Fucales (*Landsburgia quercifolia* Harv., *Marginariella boryana* (A.Rich.) Tandy, *Carpophyllum maschalocarpum* (Turner) Grev. and *Cystophora scalaris* J.Agardh). The experiments were run in static cultures (no water circulation) in chambers. Water temperatures were set at either 12° C and 18° C, or 18° C and 22° C, according to whether the species was fertile in winter or summer. In each case the lower temperature was close to ambient temperatures in Wellington at the time of fertility. Cool white lights were used with light and dark periods (L:D) of 9 hours light and 15 hours dark (9:15), or 12 hours light and 12 hours dark (12:12). Some cultures were shaded by placing the 6-well culture tray inside a double-layer bag of shade cloth. Shading reduced the average available PAR (photosynthetically available radiation) from an average of 60 µmol photons m^{-2} s^{-1} to 30 µmol photons m^{-2} s^{-1}. All of the experimental treatments were replicated (n=6). A summary of the experiments and conditions is provided in Table 4.

Table 4: A summary of the experiments: species, depth range and known fertile period; culture conditions tested (photoperiod, duration of experiment, factors tested).

Order	Species	Depth range (m)	Fertility period	Light period	# of days	Factors tested	Levels
Laminariales	*Lessonia variegata*	0–20	Winter	9:15 (L:D)	37	Temperature	12° C, 18° C
						Shading	Shade, no shade
Fucales	*Landsburgia quercifolia*	0–40	Winter	9:15 (L:D)	21	Temperature	12° C, 18° C
						Shading	Shade, no shade
	Landsburgia quercifolia	0–40	Winter	9:15 (L:D)	54	Sediment	Sediment, no sediment
						Temperature	12° C, 18° C
						Shading	Shade, no shade
	Marginariella boryana	0–40	Winter	9:15 (L:D)	46	Temperature	12° C, 18° C
						Shading	Shade, no shade
	Carpophyllum maschalocarpum	0–45	Summer	12:12 (L:D)	40	Temperature	18° C, 22° C
						Shading	Shade, no shade
	Cystophora scalaris	0–15	Summer	12:12 (L:D)	36	Temperature	18° C, 22° C
						Shading	Shade, no shade

Filtered seawater (FSW) was used for the establishment of cultures (initially filtered to 5 µm (microns), then re-filtered through 0.45 µm filters). After the first week, all cultures were maintained in f/2 culture medium (AlgaBoost™, AusAqua Pty Ltd). The methods listed below refer to the experiments where cultures were successfully established and maintained for a minimum of three weeks. For most of the species tested, several attempts were made before this was achieved, and details of these unsuccessful cultures are available in Appendix 5.

Temperature and shading experiments with winter fertile plants

Lessonia variegata

A single fertile drift *Lessonia variegata* plant was collected from Waitaha Cove, Lyall Bay on 10 May 2017. Five sori were excised from the blades with a razor blade and each sorus was wiped with an ethanol soaked tissue (80 %) to remove surface contaminants. The sori were left at room temperature for two hours before being added to 300 mL of cool, FSW in a 1 L flask and agitated on the shaker table for 30 minutes before being checked for the presence of gametes. Gametes were present in large numbers. An aliquot of 8 mL of FSW was added to each well of 4 × 6-well culture trays. Two trays were placed in the 12 ˚C cabinet and two in the 18 ˚C cabinet overnight. The following day, the wells were examined under a dissecting microscope before being photographed and a shade bag used to cover one tray per temperature.

Landsburgia quercifolia

Five submerged, but unattached drift plants were collected from Waitaha Cove on May 25th, 2017. Receptacles were removed from each plant, placed in a glass container of FSW (Figure 2), and left in the 18 ˚C cabinet overnight. The next morning the receptacles were removed and a cell scraper and pipette were used to transfer eggs to a fresh container containing 50 mL of FSW. From this slurry, a 1 mL aliquot was added to each well of 4 × 6-well culture trays. Two trays were placed in the 12 ˚C cabinet and two in the 18 ˚C cabinet overnight. The following day, the wells were examined under a dissecting microscope before being photographed and a shade bag used to cover one tray per temperature.

Figure 2: *Landsburgia quercifolia*: receptacles (small, specialised leaves) releasing eggs.

Marginariella boryana

Seven submerged, but unattached drift plants were collected from Waitaha Cove on May 25th, 2017. Receptacles were removed from each plant and dried at room-temperature overnight before being added to FSW. After two hours the receptacles were examined under a dissecting microscope and receptacles (Figure 3) that were releasing were scraped with fine forceps to remove gametes from their surface. Gametes were placed into a small amount of FSW and from this 1 mL aliquots were placed into each well of 4 × 6-well culture trays, and each well was topped up with 9 mL of f/2 culture medium. Two trays were placed in the 12 ˚C culture cabinet and 2 trays into the 18 ˚C cabinet overnight. Photographs were taken the next day and a shade bag was used to cover one tray at each temperature.

After 7 days the cultures were heavily contaminated with paramecia (small unicellular ciliates of the genus *Paramecium*) which appeared to be feeding on eggs and some sporelings. The 18 ˚C cultures were mostly dead at this point and were discarded. In an attempt to lower the numbers of paramecia in the remaining cultures, freshwater was added to each well for 20 seconds before being drained and rinsed with FSW then topped up with f/2 culture medium.

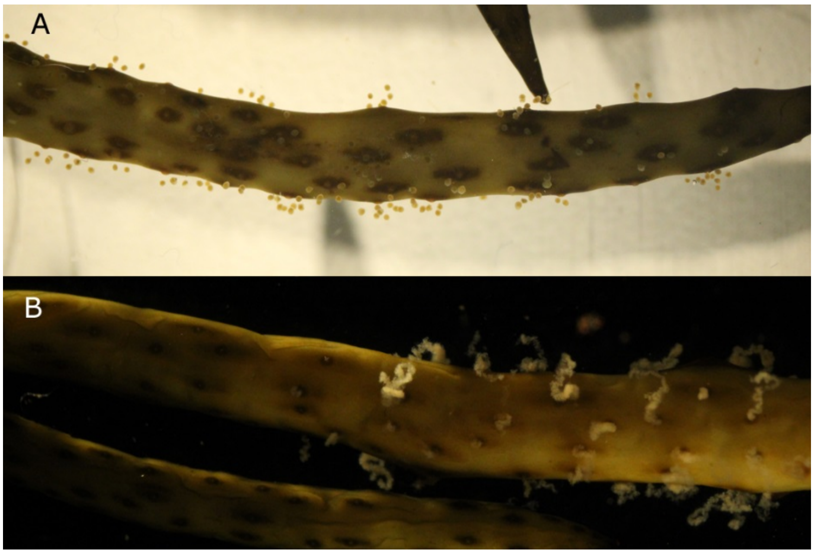

Figure 3: *Marginariella boryana*: receptacles with eggs (A) and sperm (B) being released from conceptacles.

Sediment experiment with *Landsburgia quercifolia*

At the end of the temperature and light experiments, the *Landsburgia* trays were re-photographed under a dissecting microscope and sediment was added to three of the six wells for each prior treatment. The sediment was collected from the Pauatahanui Inlet (Porirua Harbour) adjacent to Horokiri Stream. The sediment was analysed by Gradistat (V 8.0) and was classified as very coarse, silty fine sand. The factors for this sediment experiment then became sediment (sediment or no sediment), shading (shade or no shade) and temperature (12 ˚C or 18 ˚C). The cultures were photographed at 3 subsequent time points (days 39, 56 and 78) and the culture medium was changed on a weekly basis.

Experiments with summer fertile plants

Carpophyllum maschalocarpum

Five attached *C. maschalocarpum* plants were collected subtidally from Moa Point on 7 November 2017 and refrigerated overnight. The next day bunches of receptacles were removed from a minimum of five branches per plant. Receptacles were examined under a dissecting microscope (Figure 4) and individual receptacles scraped with fine forceps, removing as many gametes from inside and outside the conceptacles as possible into a petri dish with a small amount of FSW. From this slurry, a 0.6 mL aliquot was to each well of a 6-well culture tray, and each well was topped up with 9 mL of f/2 culture medium. Two trays were placed in the 18 ˚C culture cabinet and 2 trays into the 22 ˚C cabinet overnight. Wells were photographed the next day and a shade bag was used to cover one culture tray in each cabinet. The culture trays were changed and re-photographed a further 6 times.

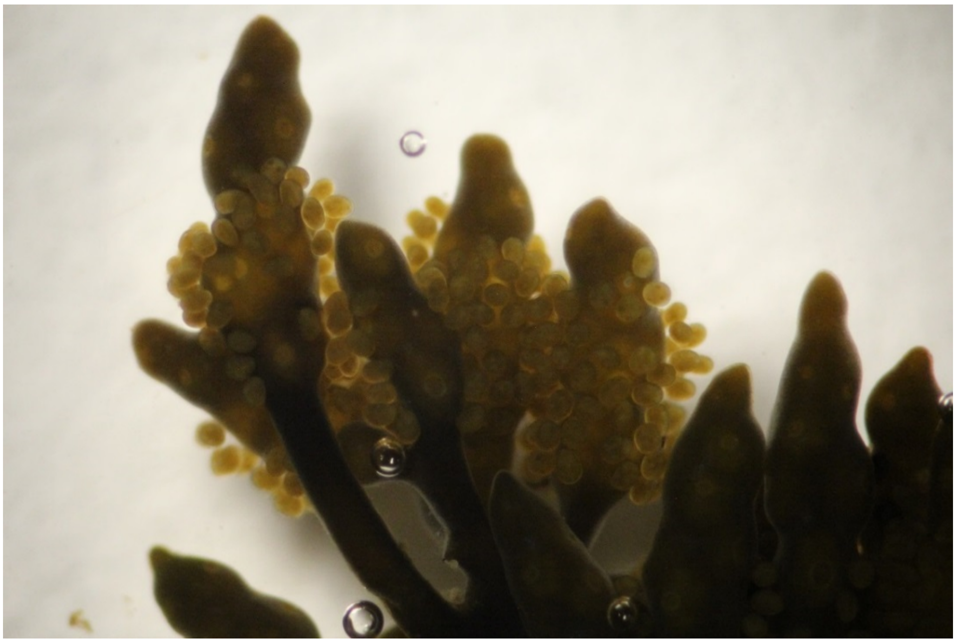

Figure 4: *Carpophyllum maschalocarpum*: a bunch of receptacles releasing eggs.

Cystophora scalaris

Four attached *C. scalaris* plants were collected subtidally from Moa Point on 7 November 2017 and refrigerated overnight. The next day numerous individual receptacles were removed from at least five branches per plant and left to dry for 3 hours before being placed in a glass culture jar and adding refrigerated filtered seawater (FSW). After two hours there had been very little release of gametes, so the jar was left for a further two hours before being re-examined under a dissecting microscope. Only two bunches of receptacles were observed to be releasing, so these were placed into a separate jar of FSW, and both jars were left to release overnight then examined the next morning. The jar containing the majority of the receptacles was heavily contaminated and was discarded. A cell scraper was used to gently dislodge settled eggs from bottom of the remaining jar and create a slurry. Aliquots of this slurry (0.5 mL) were pipetted into 4 × 6-well culture trays and 9 mL of FSW was added to each well. Shade bags were used to cover two of the trays and one shaded and one unshaded tray were placed into each of the 18 and 22 ˚C culture cabinets.

Photography and image analysis

Photographs were taken using an Axiovert 200 microscope, an Axiocam HR digital camera and Axiovision image software v.4.1 (all Zeiss, Jena, Germany). The attached camera records a scale for each image and the complementary measurement software was used to assess the length of fucalean sporelings (fertilised gametes) at each time point. For *Lessonia variegata*, images were taken of 5 random fields of view per well.

Lessonia variegata

The length of the five sporelings closest to the centre of the field of view were recorded. This process was repeated a further 5 times at approximate intervals of 1 week.

All fucalean species

Fucalean eggs and sporelings are much larger than *Lessonia* sporelings. Where only one was present in the camera field of vision it was measured and recorded, and in cases where multiple eggs or sporelings were visible, between three and five were measured and recorded. As the experiments progressed, in some cases fewer sporelings were suitable to photograph due to disintegration or predation, and in later stages, overgrowth by filamentous algae.

We had initially intended to assess growth using relative elongation rates (RER), rather than average length. However, gametes scraped from the outside of the receptacles into the slurry were potentially a mix of unfertilised and fertilised, and further fertilisation is assumed to have occurred in the slurry. This effectively gave two (or more) cohorts of sporelings at the same time point (Figure 5). Calculating the elongation rate from measurements of sporelings of such varying sizes resulted in too many negative elongation rates, and in many cases, there weren't enough surviving sporelings to allow analysis.

Figure 5: Fertile material released from *Landsburgia quercifolia* receptacles at day 0. Two fertilised eggs are shown above a developing sporeling, complete with the rhizoids (top right) that it uses to anchor itself to the substratum.

Statistical analysis

Data at the final time of measurement were analysed using the Sigmastat functions of Sigmaplot v12.5 (Systat Software, Inc.). A series of two-way ANOVAs were conducted, with Factor 1 = temperature (two levels, 12 ˚C and 18 ˚C for winter-fertile species, or 18 ˚C and 22 ˚C for summer-fertile species), and Factor 2 = shading (shaded or non-shaded). Assumptions of normality and equal variance were tested using the Shapiro-Wilk test and subsequent square root or Log10 transformations were made to satisfy these requirements. Where appropriate, post hoc multiple comparison procedures were run using the Holm Sidak Test. In both the May and August experiments with *Marginariella boryana*, not enough sporelings survived in the 18 ˚C to allow analysis of data for that temperature, so for this species a one-way ANOVA was used to assess the effect of shading at 12 ˚C.

Three-way ANOVAs were used to analyse data from the *Landsburgia quercifolia* sediment experiment. In this case, the factors were temperature (two levels, 12 ˚C and 18 ˚C for winter-fertile species, and 18 ˚C and 22 ˚C for summer-fertile species), shading (shaded or non-shaded) and sediment (sediment or no sediment). Equal variance and normality were tested and adjusted using the above procedures.

3.3 Results

For all the experiments outlined below significant results are indicated by *p* values in the text. Full ANOVA tables are presented in Appendix 6.

LAMINARIALES

Lessonia variegata

The *Lessonia variegata* cultures were run for 37 days and remained free of contaminants throughout the culture period (Figure 6). Germination tubes were present by day 1 (Figure 6A) and sporeling lengths ranged between 11.27 and 21.91 µm (microns). By day 37 (Figure 6B) mean lengths of sporelings grown at 18 ˚C were significantly greater than those grown at 12 ˚C (Figure 7) (two-way ANOVA, $p = 0.021$) regardless of whether the cultures were shaded or not ($p = 0.542$).

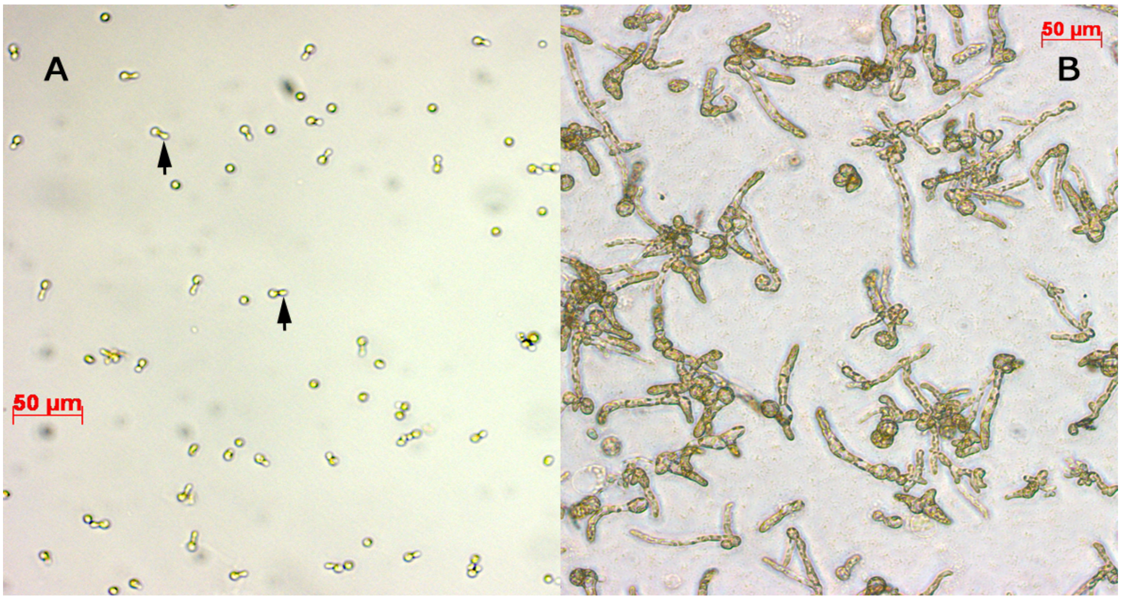

Figure 6: *Lessonia variegata*: (A) sporelings at the time of the establishment of the 12 ˚C non-shaded cultures (germination tubes are marked by arrows), and (B) 37 days later.

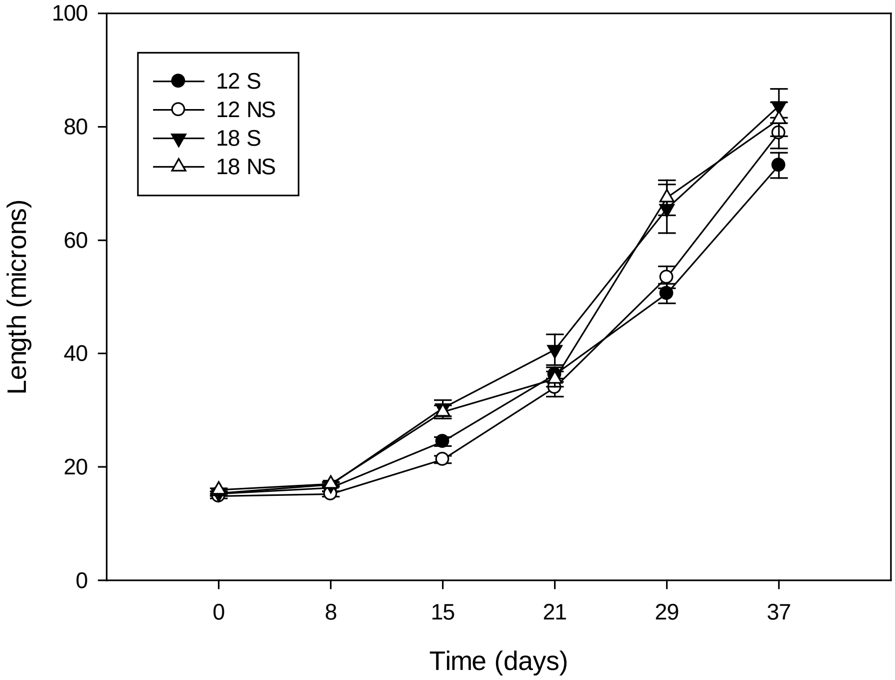

Figure 7: *Lessonia variegata*: sporeling length at two temperatures (12 and 18 ˚C) and two shade levels: shaded (S) and non-shaded (NS). N = 30 for each treatment.

FUCALES

Landsburgia quercifolia

The *Landsburgia* experiment was run over 21 days. The sporelings remained healthy throughout (Figure 8) and little contamination by invertebrates or other seaweeds was evident over the duration of the experiment.

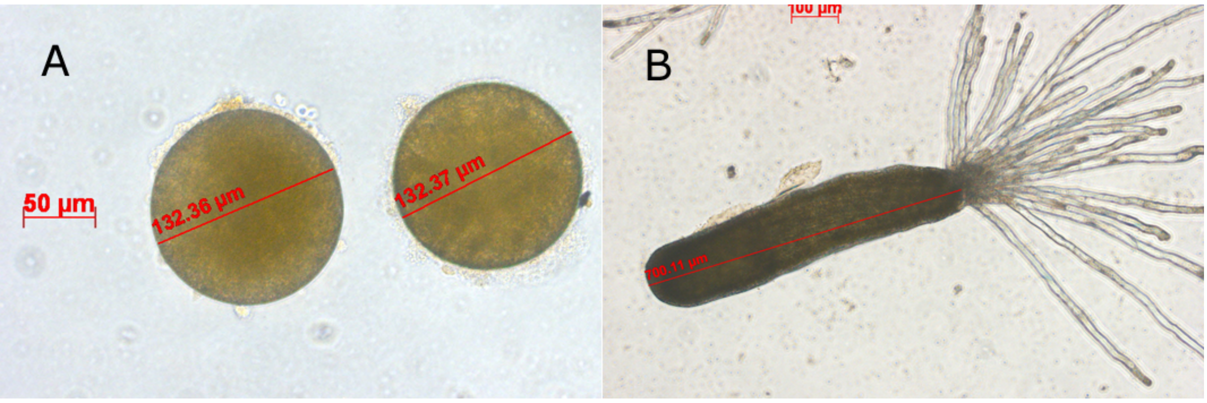

Figure 8: *Landsburgia quercifolia*: (A) fertilised eggs at day 0, and (B) a healthy sporeling at day 21.

A two-way ANOVA assessing the effects of temperature and shading found that both factors had a significant effect on sporeling length at 21 days ($p < 0.001$ and $p = 0.028$ respectively), but also that the effect of temperature was dependent on the level of shading present (interaction term, $p < 0.001$). Shading had no effect on sporeling length within the 12 ˚C cultures (mean 587 µm) ($p = 0.419$), but did within the 18 ˚C cultures (mean 596 µm) ($p < 0.001$). Temperature in the shaded cultures did not have a significant effect on length ($p = 0.903$) but did within the non-shaded cultures, with non-shaded sporelings (mean 765 µm) at 18 ˚C being significantly longer than non-shaded sporelings at 12 ˚C (mean 557 µm) ($p < 0.001$) (Figure 9).

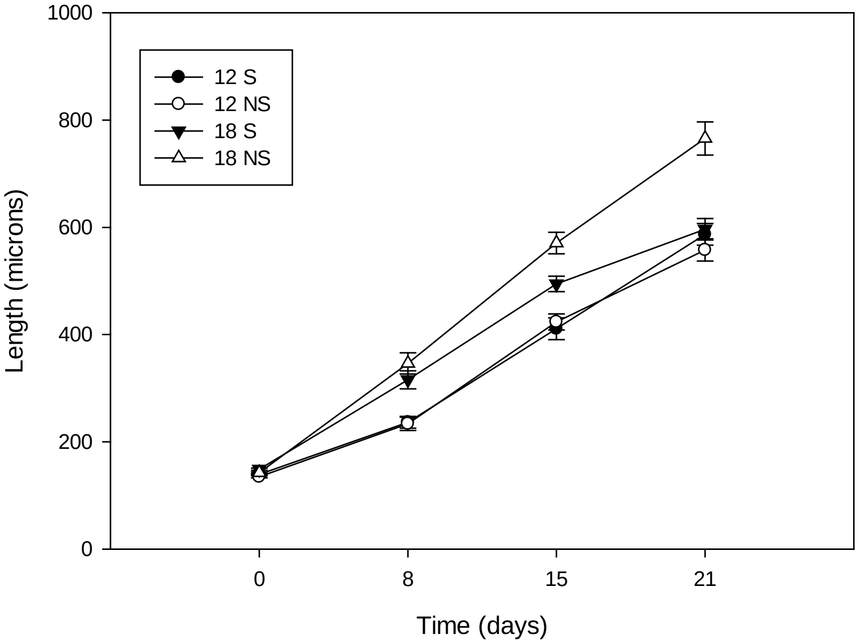

Figure 9: *Landsburgia quercifolia*: sporeling length at two temperatures (12 and 18 ˚C) and two shade levels (shaded (S) and non-shaded (NS). The number of sporelings measured varied between treatments with a minimum n = 37 and a maximum n = 69.

In the experiments testing shading, sediment and temperature, sporelings grown without shade attained greater lengths than those grown with shade, regardless of temperature, and the longest sporelings occurred in cultures that were shaded and had no sediment (mean 2996 µm at 18 ˚C and 2827 µm at 12 ˚C). Shaded cultures with sediment added reached the lowest lengths over all (1369 µm at 12 ˚C and 1154 µm at 18 ˚C, Figure 9). A three-way ANOVA (see Appendix 6) assessing the effects of the three factors at 78 days found that shade and sediment had significant effects on sporeling length (both *p* < 0.001) but temperature did not (*p* = 0.056) (Figure 10). However, the effects of shading depended on whether sediment was present or not (interaction term, *p* < 0.001), and on temperature (interaction term, *p* < 0.001). Within non-shaded cultures, temperature was not significant (*p* = 0.201) whereas within shaded cultures it was (*p* < 0.001).

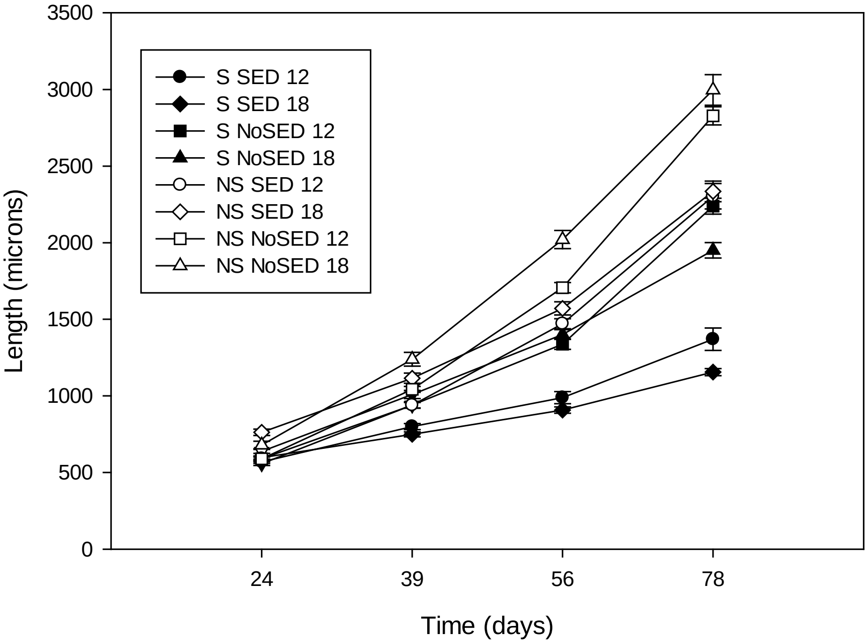

Figure 10: *Landsburgia quercifolia*: sporeling length at two temperatures (12 ˚C and 18 ˚C), two levels of shading (shaded (S) and non-shaded (NS)) and two levels of sediment (sediment (SED) and no sediment (NoSED). n = 10.

Marginariella boryana

Very few sporelings survived in the 18 ˚C treatments (Figure 11) and there were insufficient numbers for analysis. By the end of the experiment, the mean length of non-shaded *M. boryana* sporelings in the 12 ˚C was 880 µm compared to 456 µm in the shaded ones. A one-way ANOVA was run on the 12 ˚C data, assessing the effect of shading at 47 days. Results showed a significant difference between shaded and non-shaded sporelings (*p* < 0.001) with non-shaded sporelings attaining over twice the length of shaded sporelings at that time (Figure 12).

Figure 11: ***Marginariella boryana*****: (A) A sporeling from the 18 ˚C treatment at day 0, and (B) the same sporeling disintegrating at day 7.**

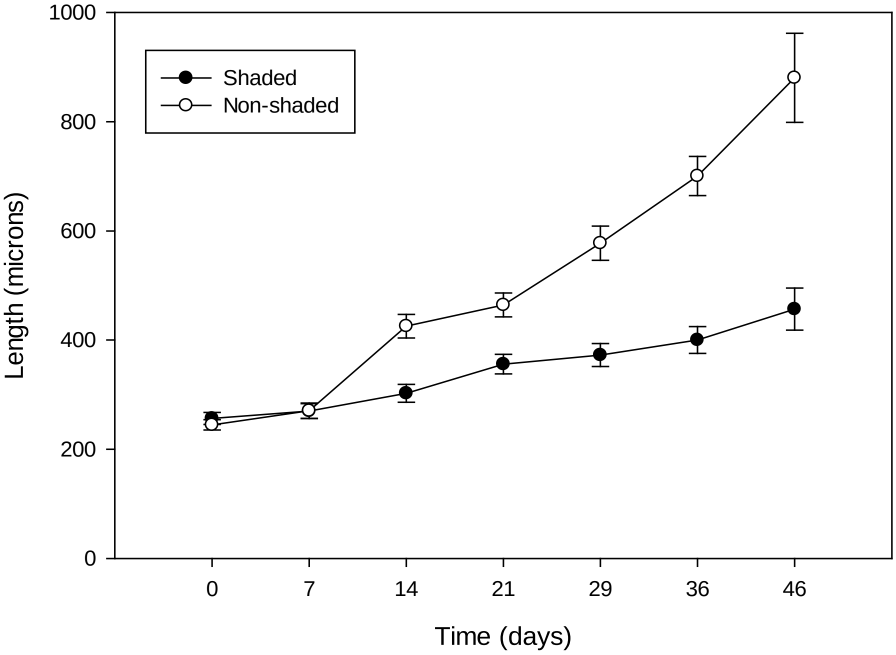

Figure 12: ***Marginariella boryana*****: sporeling length at 12 ˚C and two shade levels (shaded and non-shaded). n = 16 to n = 51.**

Carpophyllum maschalocarpum

Shaded *C. maschalocarpum* sporelings attained greater lengths (1226 µm at 18 ˚C and 913 µm at 12 ˚C) than non-shaded sporelings (831 µm at 18 ˚C and 715 µm at 12 ˚C). A two-way ANOVA assessing the effects of temperature and shading at 40 days found that shade had a significant effect on sporeling length ($p = 0.023$) but temperature did not ($p = 0.08$) (Figure 13).

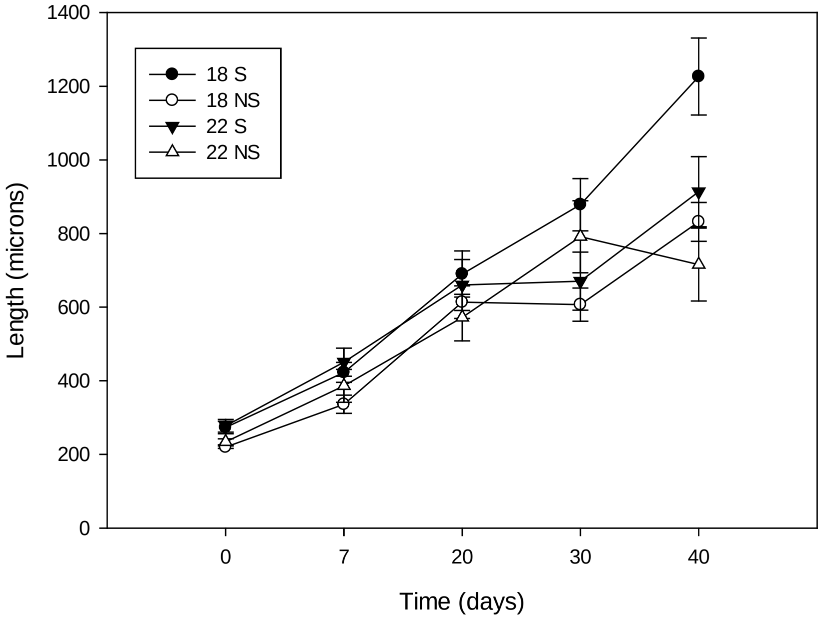

Figure 13: *Carpophyllum maschalocarpum*: sporeling length at two temperatures (12 and 18 ˚C) and two shade levels, shaded (S) and non-shaded (NS). n = 12 to n = 41, with the 22 NS treatment being n = 6 as only six sporelings survived at this time.

Cystophora scalaris

The two-way ANOVA assessing the effects of temperature and shading at 36 days found that both temperature ($p < 0.001$) and shade had a significant effect on sporeling length ($p = 0.03$). The sporelings grown at 18 ˚C were significantly longer (489 µm non-shaded and 449 µm shaded) than those grown at 22 ˚C (344 µm non-shaded and 340 µm shaded), and non-shaded sporelings were longer than shaded ones (Figure 14).

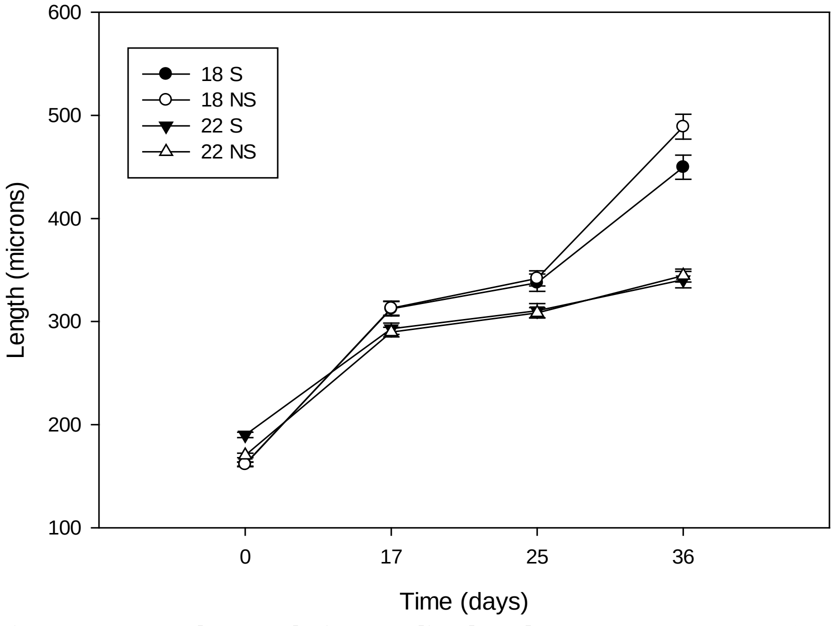

Figure 14: *Cystophora scalaris*: sporeling length at two temperatures (18 and 22 ˚C) and two shade levels, shaded (S) and non-shaded (NS). n varied between 69 and 101.

3.4 Discussion

Species of Fucales have been shown to be useful for experimental studies on the impacts of nearshore stressors. In both New Zealand and Australia, the fucoid *Hormosira banksii* has been employed as an indicator species to evaluate the impacts of stressors (e.g., Alestra 2014, Doblin & Clayton 1995), as it is able to be grown readily in both culture and field experiments.

The short-term experiments conducted in this project provided new data on four species from four genera of Fucales and one laminarian, *Lessonia variegata*. The species investigated (with the exception of *Lessonia variegata*) had not been studied in detail since anatomical and early developmental studies over 50–70 years previously. In some cases (e.g., *Carpophyllum*) the growth in culture has not been successful previously. The culture study was initiated for several reasons: 1) to investigate the ease of culture and manipulation of additional canopy-forming macroalgae that may be more representative of either (a) additional types of coastal habitats, or (b) different geographic ranges within New Zealand, 2) responsiveness of species to stressors known to influence the growth of canopy-forming species.

Light and temperature are critical variables for photosynthetic species, and sediment has been identified as a major influence on New Zealand coastlines (e.g., MacDiarmid et al. 2012, Ministry for the Environment 2016). Sediment has a range of impacts including reducing light, but also causing abrasion and interfering with the settlement of zygotes and spores. The conditions selected were a) current temperatures and raised temperatures 2) light environments of shade and non-shade, 3) with and without a coating of sediment (tested on one species). From the results presented here there is clear evidence of species specific responses to the factors being tested. For example, in two of the winter fertile species (*Landsburgia, Lessonia*) growth was significantly better (expressed as sporeling length) in higher winter temperatures, and one of the summer fertile species (*Cystophora*) grew better at current rather than future (raised) temperatures.

Shading had a significant effect on all species except *Lessonia variegata* and greater lengths were achieved in non-shaded cultures for all affected species except *Carpophyllum maschalocarpum*. The two summer fertile species (*Carpophyllum maschalocarpum* and *Cystophora scalaris*) responded significantly to shading, however, *C. maschalocarpum* attained greater lengths in shaded conditions, and *C. scalaris* in non-shaded conditions. The shade response seen here in *Carpophyllum* warrants further investigation; Dromgoole (1973) examined the physiology of *Carpophyllum* in great detail, revealing the complexity of sun and shade adaptation within a single thallus; Schiel (1985) examined the growth of *Carpophyum maschalocarpum* in low and high density stands and found that high density aggregations may be important for the establishment and maintenance of populations of this species. This may indicate that early stage sporelings are adapted to low light, sub-canopy conditions.

In *Landsburgia quercifolia*, germlings at 18 ˚C grew longer in non-shaded cultures and there was no difference in shaded cultures at this temperature. The effect of sediment deposition was only tested on *Landsburgia* and temperature became less influential in this experiment, with sporelings at both temperatures subjected to both shade and sediment achieving the lowest lengths of all treatments, while those at both temperatures with no sediment and no shade attained the greatest lengths overall. In the experiments conducted here, sediment was introduced after successful settlement, so in this situation it may be producing the negative response through interfering with light availability as well as nutrient and gas exchange and uptake at the thallus surface.

Of the species tested here, *Landsburgia* and *Lessonia* were the most straightforward experimental subjects with respect to establishment and manipulation of cultures. With the other species there were significant issues with contamination, a likely consequence of the retention on parent plants of early germination stages, regardless of whether attached or drift plants were collected and also the time of year. The longer the sporelings are retained on the parent thallus, the greater the opportunity for cross contamination with the epibiota of the parent.

The experiments conducted in this study provide preliminary data for further research, outside the scope of this project.

4 UTILITY OF MACROALGAE AS MONITORING TOOLS AND TESTING DISTRIBUTION MAPPING METHODS

4.1 Introduction

In order to establish the utility of selected macroalgae as monitoring tools for the assessment of ecosystem health, and to test methods for mapping distribution for baseline monitoring, we employed several different approaches, as case studies.

One of the aims was to monitor previously well documented sites repeating similar field protocols. The study chosen for this was in the East Otago Taiāpure with monitoring conducted by Ngai Tahu in 2009 and 2017.

Secondly, four different approaches were used to evaluate the most cost effective method for mapping kelp forests with the aim of establishing reliable baselines: comparing historical data with present day investigations of the distribution of *Macrocystis pyrifera* in Wellington; exploring the use of drones/unmanned aerial vehicles (UAV) for mapping of macroalgal beds; underwater video imagery; the development of machine learning and computer vision techniques for analysis of underwater video.

Finally, we explored the use of citizen science observations to extend the information available about the distribution of large brown algae around New Zealand, and then mapped the current known distributions of laminarians and fucoids at a national scale using herbarium data, expert survey observations and the data from citizen scientist.

4.2 Macroalgal Monitoring. Case study: East Otago Taiāpure

The Otago coastline is characterised by exposed headlands interspersed with protected inlets and bays, providing a diversity of habitats that support rich benthic communities. Flowing northward along the coast, the Southland Current buffers the region from colder offshore subantarctic waters (Sutton 2003) and allows temperate assemblages to develop. Mixed macroalgal beds form the foundation of East Otago's temperate rocky reef communities, with fleshy brown species of laminarians and fucoids providing a main source of habitat and food (Desmond et al. 2015, Hepburn et al. 2011, Richards 2010). Large stands of *Macrocystis pyrifera* are established along the coastline and play a particularly important role in supporting local fisheries species such as pāua (*Haliotis iris* Gmelin, 1791) and blue cod (*Parapercis colias* (Forster, 1801)).

The East Otago Taiāpure was established in 1999 with the purpose of enabling sustainable management of local marine ecosystems and preserving customary fisheries. Mātauranga (local knowledge) and research efforts within the Taiāpure are being used to design regulations that promote the recovery of historically overexploited resources (East Otago Taiāpure Management Plan 2008). Located north of the Otago Peninsula, the East Otago Taiāpure border extends from Cornish Head to Potato Point, and includes 24 square kilometers of marine and estuarine waters that encompass Blueskin Bay as well as the mouths of both the Waitati and Waikouaiti rivers. Land use adjacent to the East Otago Taiāpure consists mainly of productive grassland, with several coastal communities established along the shoreline (Figure 15).

Marine habitats of the East Otago Taiāpure face environmental challenges similar to those of other coastal communities, as well as challenges unique to the area. An invasion of the laminarian kelp *Undaria pinnatifida* has reshaped marine landscapes with implications that have yet to be fully understood (Schiel & Thompson 2012). Initially spread throughout New Zealand by shipping (Hay 1990b), *Undaria* is now ubiquitous in Otago Harbour and has more recently spread to pockets of the open coastline (Richards 2010, Russell et al. 2008).

The East Otago coastline is subjected to a number of human induced changes. Local watershed outlets and coastal erosion provide a potential source of sedimentation, as well as eutrophication derived from agricultural runoff. Additionally, these ecosystems will experience the global effects of climate change, which include warming waters (Lima & Wethey 2012) and ocean acidification (Law et al. 2017). All of these factors influence macroalgal communities, which in turn may affect customary fisheries and overall coastal health.

Survey Purpose

Brown macroalgae, particularly laminarians and fucoids, are critical habitat-forming species that make up the majority of biomass and canopy cover in the East Otago benthic community (Jiménez et al. 2015a, 2015b, Jiménez et al. 2017). These species provide a source of food and shelter, as well as complex three-dimensional habitat within which a variety of organisms can thrive (Win 2010). Furthermore, the presence or absence of these macroalgae can serve as indicators of broader environmental conditions. For example, some authors suggest that *Macrocystis pyrifera* has a low thermal tolerance (Hay 1990a), and some populations appear to be susceptible to sudden disappearance above certain temperature thresholds (Ladah et al. 1999). These environmental sensitivities and habitat-forming roles make brown macroalgal species ideal candidates for long term monitoring programmes.

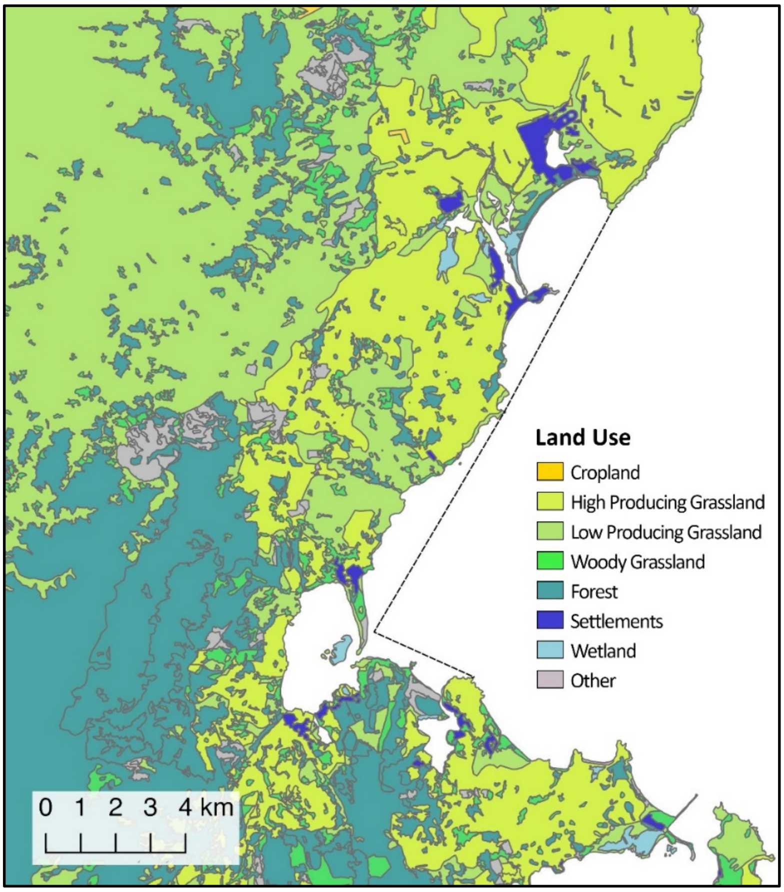

Figure 15: Map displaying land use and prominent features around the East Otago Taiāpure as of 2012 (Ministry for the Environment, Data Service). Land use adjacent to the coastline consists

primarily of productive grassland, interspersed with coastal settlements and small patches of forest. Coastal wetlands and estuaries located at river mouths also make up prominent regional features. The seaward border of the EOT management area is marked by the dashed line. Comparisons of available GIS data between 2009 and 2012 did not indicate any drastic changes in land use designation or relative coverage for this area.

Methods: Macroalgal Monitoring: Case Study - East Otago Taiāpure (2009 – 2017)

Surveys

Macroalgal community surveys were conducted in 2009 and in 2017 by a team consisting of graduate students and staff of the University of Otago Marine Science Department in collaboration with the Ngāi Tahu Te Tiaki Mahinga Kai team. Rocky reef sites were originally selected from across the EOT coastline based on the presence or absence of *Undaria* (Richards 2010). Of the initial sites surveyed in 2009, six were chosen for resampling in 2017 (Figure 16), with the goal of assessing the progression of *Undaria* invasion and determining any other changes in native brown algal assemblages.

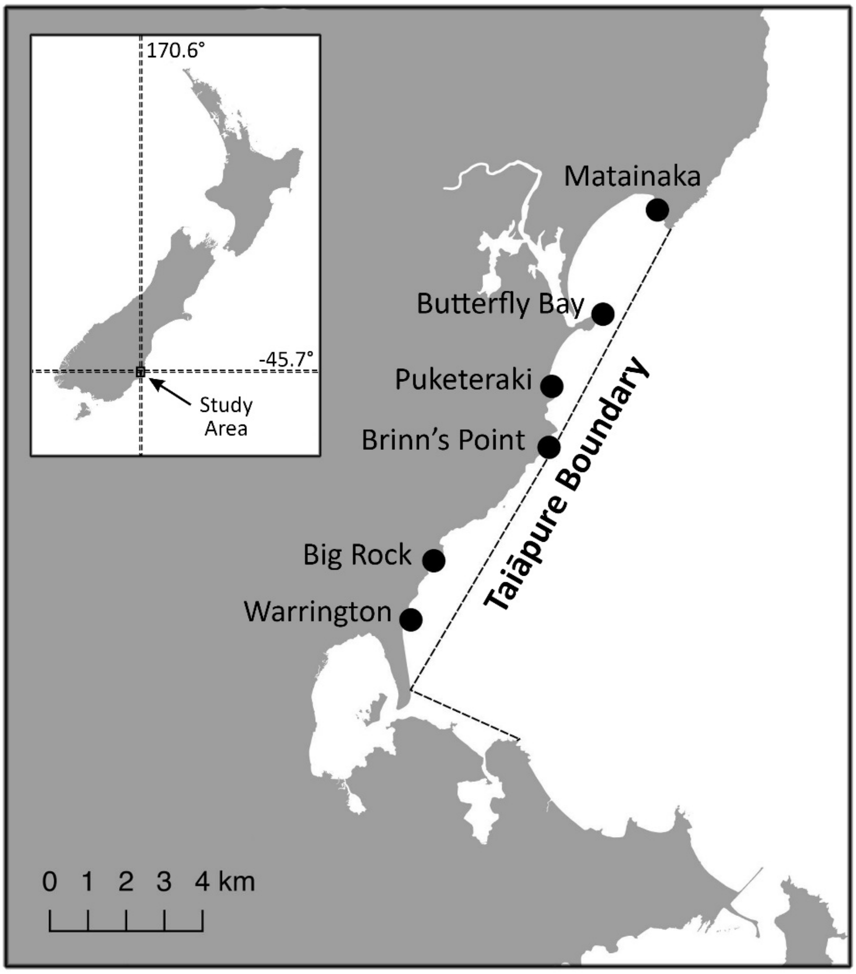

Figure 16: Locations of the six macroalgal monitoring sites designated within the East Otago Taiāpure. The boundary of the Taiāpure is indicated by the dashed line.

At each site, 30 m transects were extended parallel to the shore along four different depth strata, and ten 1 m^2 quadrats were placed randomly along each transect. The shallowest depths (0 and 0.5 m) were sampled as near as possible to low tide by either wading or snorkelling, while the deeper strata (1–3 and 4–6 m) were sampled using SCUBA. To achieve consistency between sites and time points, all depths were adjusted to mean low water. During the 2009 surveys, sites were marked for repeated sampling with GPS positioning, as well as with cattle tags attached to boulders by Marine Expocrete at each transect end. These markers were used to position the 2017 survey.

All large brown macroalgal species were identified and recorded by individual abundance, and substrate was categorised by percent cover for each quadrat. Species identifications were confirmed by guides when necessary (Adams 1994). For the laminarian macroalgae, distinctions were made between juvenile and adult individuals based on the presence or absence of key morphological structures (Table 5). During the 2009 survey all macroalgal members in each quadrat were identified, with some species grouped into broader categories (e.g., crustose coralline algae, filamentous reds, etc.); since counts for these species were not repeated in 2017, they are not included in the time point comparison analysis for this report.

Duration of survey varied widely by site and day, ranging from 20 minutes to over 2 hours. Time to completion was strongly influenced by team size, with a minimum of two and a maximum of six surveyors. All diving work was conducted in pairs. Sampling was postponed whenever large swell or low visibility made conditions unworkable.

Table 5: Species of large brown algae and morphological structures used to distinguish between juvenile and adult individuals (Richards 2010).

Species	**Adult morphology**
Durvillaea antarctica	Internal honeycomb structure developed within blades
Durvillaea willana	Stalked lateral blades developed on stipe
Macrocystis pyrifera	Pneumatocysts (floating structures) present at blade base
Undaria pinnatifida	Folded sporophyll developed at bottom of stipe

Data Analysis

A three-factorial permutational multivariate analysis of variance (PERMANOVA) was used to test for differences among macroalgal communities across sites, depths, and time points. Bray-Curtis similarity measures were calculated with untransformed abundance data, and 9999 permutations were used to obtain p-values. Non-metric multidimensional scaling (NMDS) plots were created to visualise dispersion between points based on the dissimilarity matrix. A post-hoc Tukey test was used to identify significant differences among groups. All analyses were performed using the vegan package (Oksanen 2015) and other tools in R Version 3.1.2. PERMANOVA is a commonly employed ecological tool for analysing multivariate datasets with non-normal distributions (Anderson 2001), and is often used to interpret results of macroalgal abundance surveys (Cornwall et al. 2015a, Hepburn et al. 2011). One limitation of PERMANOVA is that it relies on homogenous dispersion of replicates within groups, an assumption that is often incompatible with real-world observations (Warton et al. 2012). In the case of this dataset, a formal test of the dispersion assumption revealed heterogeneous dispersion among groups. Visual inspection of the NMDS plot was used to separate significant differences in centroid locations from significant differences in dispersion.

Results: Macroalgal Monitoring: Case Study - East Otago Taiāpure (2009 – 2017)

There were significant differences in macroalgal community structure across year (year: pseudo-$F_{1,397} = 18.8$, $p_{perm} < 0.001$, depth strata (depth: pseudo-$F_{3,397} = 40.9$, $p_{perm} < 0.001$) and site (site: pseudo-$F_{5,397} = 25.9$, $p_{perm} < 0.001$). The interaction between all three factors was also significant (Table 6; year × depth × site: pseudo-$F_{15,397} = 4.9$, $p_{perm} < 0.001$); NMDS plots and ordination vectors are displayed in Figure 17.

Xiphophora gladiata, *Undaria pinnatifida*, and *Macrocystis pyrifera* contributed the most to community differences between years. Overall, change in abundance over time varied by depth for each species (Figure 18), with the majority of native fucoids increasing in recorded abundance (Table 7) and some expanding their depth range. Of the largest of the brown algae surveyed, *Undaria* increased significantly in density and geographic spread, while *Macrocystis* and *Durvillaea antarctica* populations decreased.

Table 6: PERMANOVA of species abundance data testing for differences between year (2009 and 2017), depth (0, 0.5, 1–3 and 4–6 m), and site (Big Rock, Brinns Point, Butterfly Bay, Puketeraki, Matainaka, Warrington).

Factor	DF	Sum of squares	Mean square	Pseudo-F	P (PERM)
Depth	3	20.5	6.83	40.9	<0.001
Year	1	3.1	3.14	18.8	<0.001
Site	5	21.6	4.32	25.9	<0.001
Depth × Year	3	4.8	1.59	9.5	<0.001
Depth × Site	15	19.9	1.33	7.9	<0.001
Year × Site	5	12.3	2.46	14.8	<0.001
Year × Depth × Site	15	12.3	0.82	4.9	<0.001
Residuals	397	66.3	0.17		

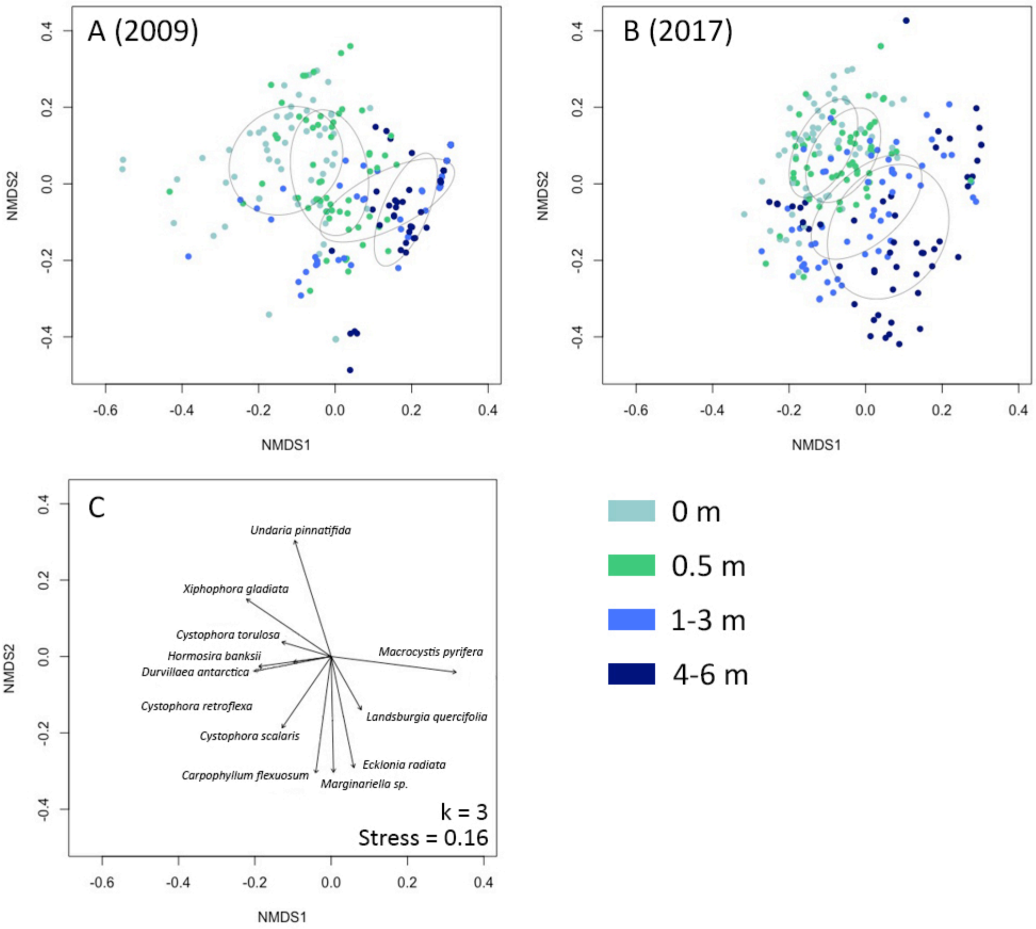

Figure 17: Non-metric multidimensional scaling (NMDS) plots display the separation between macroalgal assemblages based on the dissimilarity matrix. Each point represents a quadrat (N=480), color indicates transect depth. Separate plots were made for A) the 2009 survey and B) the 2017 survey. Ellipses are drawn around the centroid of each depth group in order to visualise relative location and dispersion within the plot ordination. Plot C displays species vectors used to ordinate points. Three dimensions were used (k=3) in order to minimise plot stress (stress = 0.16).

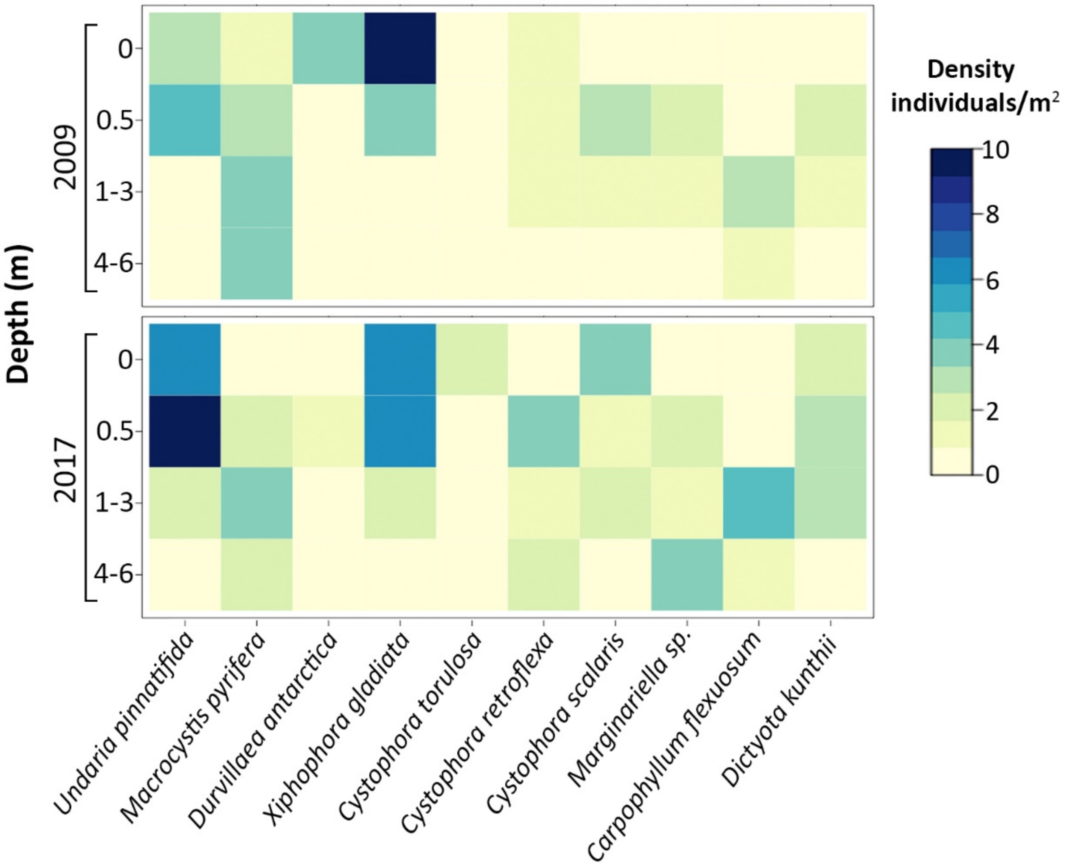

Figure 18: Mean abundance (indicated by colour) of top ten most common species by depth and year. Averages are calculated from depth data pooled across all six survey sites.

It should be noted that while this analysis focuses mainly on interpreting relative species abundances, this dataset can also be used to calculate metrics of diversity and richness (Smale 2010), locate ecotones, and map community associations (an example of a community web is included in Figure 19). Depth, substrate, and exposure information can be integrated with abundance measures in order to track changes in preferred species habitat over time. Developmental data (e.g., juvenile vs. adult kelp counts) from repeated surveys could be used to analyse timing of seasonal recruitment patterns and to understand settlement vs. survival dynam

Table 7: Brief summary of data for each species surveyed. Percent presence for each year was calculated from the ratio of quadrats that were positive for the given species to total quadrats recorded. The depth strata(s) with the densest species abundance also noted for each year. Finally, the mean abundance (from pooled depth and site data) was calculated with standard error.

Species	Presence 2009 (%)	Presence 2017 (%)	Depth with greatest density 2009	Depth with greatest density 2017	Mean abundance 2009	Mean abundance 2017
LAMINARIALES						
Undaria pinnatifida	23	48	0.5 m	0.5m	2.14±0.32	4.40±0.55
Macrocystis pyrifera	64	47	1–3, 4–6 m	1–3m	2.92±0.27	2.01±0.24
Ecklonia radiata	2	12	4–6 m	4–6m	0.06±0.04	0.47±0.9
FUCALES						
Carpophyllum flexuosum	19	32	1–3 m	1–3 m	1.22±0.22	1.63±0.24
Cystophora scalaris	20	27	0.5 m	0 m	1.16±0.21	1.85±0.29
Cystophora retroflexa	19	30	0, 0.5 m	0.5 m	1.02±0.17	1.88±0.26

Cystophora torulosa	6	10	0 m	0 m	0.21±0.09	0.62±0.14
Durvillaea antarctica	11	5	0 m	0.5 m	0.96±0.27	0.32±0.18
Durvillaea willana	<1	2	0.5 m	0 m	<0.01	0.05±0.03
Xiphophora gladiata	37	45	0 m	0, 0.5 m	3.40±0.49	3.85±0.51
***Marginariella* sp.**	18	29	0.5 m	4–6 m	1.02±0.19	1.74±0.24
Sargassum sinclairii	0	2.5	-	1–3 m	-	0.05±0.02
Landsburgia quercifolia	4	13	4–6 m	4–6 m	0.09±0.03	0.30±0.06
Hormosira banksii	1	3	0 m	0 m	0.03±0.02	0.13±0.06
OTHER						
Desmarestia ligulata	4	12	0.5 m	0.5 m	0.11±0.05	0.38±0.10
Dictyota kunthii	22	30	0.5 m	0.5, 1–3 m	1.20±0.19	1.78±0.25

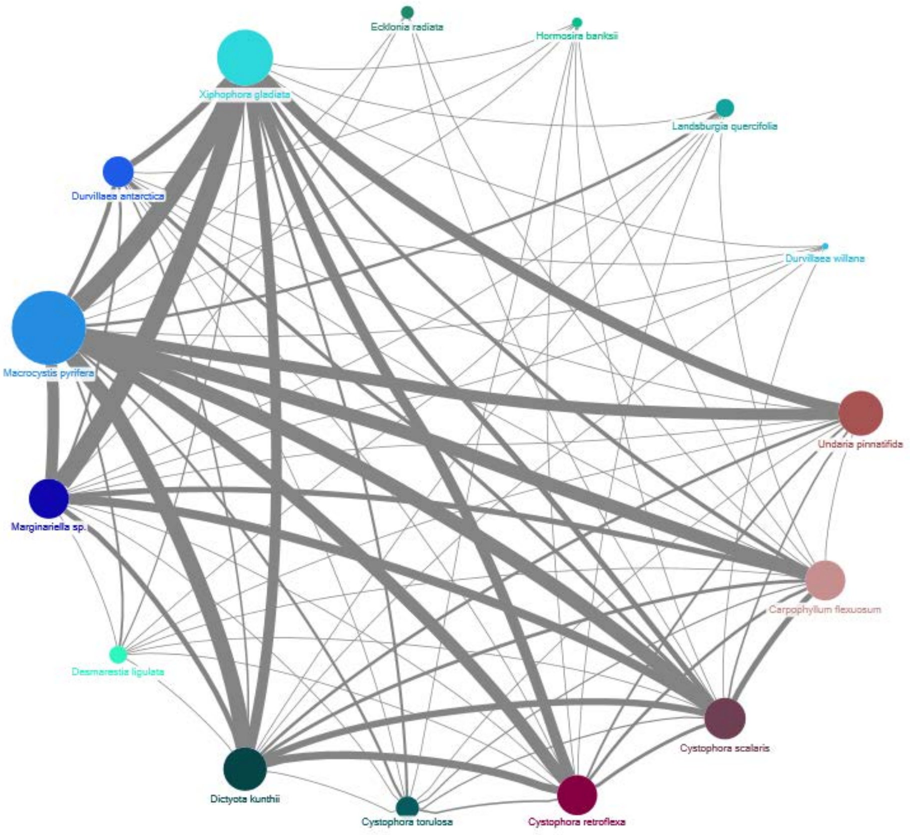

Figure 19: East OtagoTaiāpure community web (created using NodeXL) from 2009 survey data. Size of node for each species is determined by individual occurrence, while thickness of connections is determined by the frequency at which species co-occur. This is a qualitative approach to visualizing this dataset, allowing the viewer to explore relationships and identify patterns of association between species or among groups. For example, the many thick lines spreading from the *Macrocystis* node illustrate its role as a foundational member of the community near which many other species can coexist. By contrast, *Undaria* associates most strongly with *Xiphophora* and *Macrocystis*, and was never recorded near some of the less common species, such as *Hormosira banksii* or *Landsburgia quercifolia.* A community web for the 2017 survey data would show stronger *Undaria* relationships for many of these species, resulting from its spread in geographic range and increase in overall abundance. In this way, graphics created for subsequent surveys could help illustrate community change over time. This is intended as a supplement, not a replacement, for more commonly employed methods of data visualization.

Laminariales

Undaria pinnatifida

The 2017 survey revealed populations of *Undaria pinnatifida* at all sites which were previously uncolonised (Figure 20), as well as an increased presence of *Undaria* in the 0 to the 1–3 m depth strata (Figure 18). *Undaria pinnatifida* doubled in density, from an East Otago Taiāpure wide average of 2.1 ± 0.3 individuals/m^2 in 2009 to 4.4 ± 0.5 individuals/m^2 in 2017 (Table 7). Nearly half of all quadrats surveyed in 2017 contained at least one individual of *Undaria*, while the maximum density recorded was 53 individuals in one square metre. A closer look at the breakdown between adult and juvenile individuals (Figure 21) shows that the increase in *Undaria* is almost entirely due to a greater presence of juveniles. This is likely to be due to the discrepancy in seasonality of surveys, as the 2017 survey occurred at the end of October, following spring recruitment.

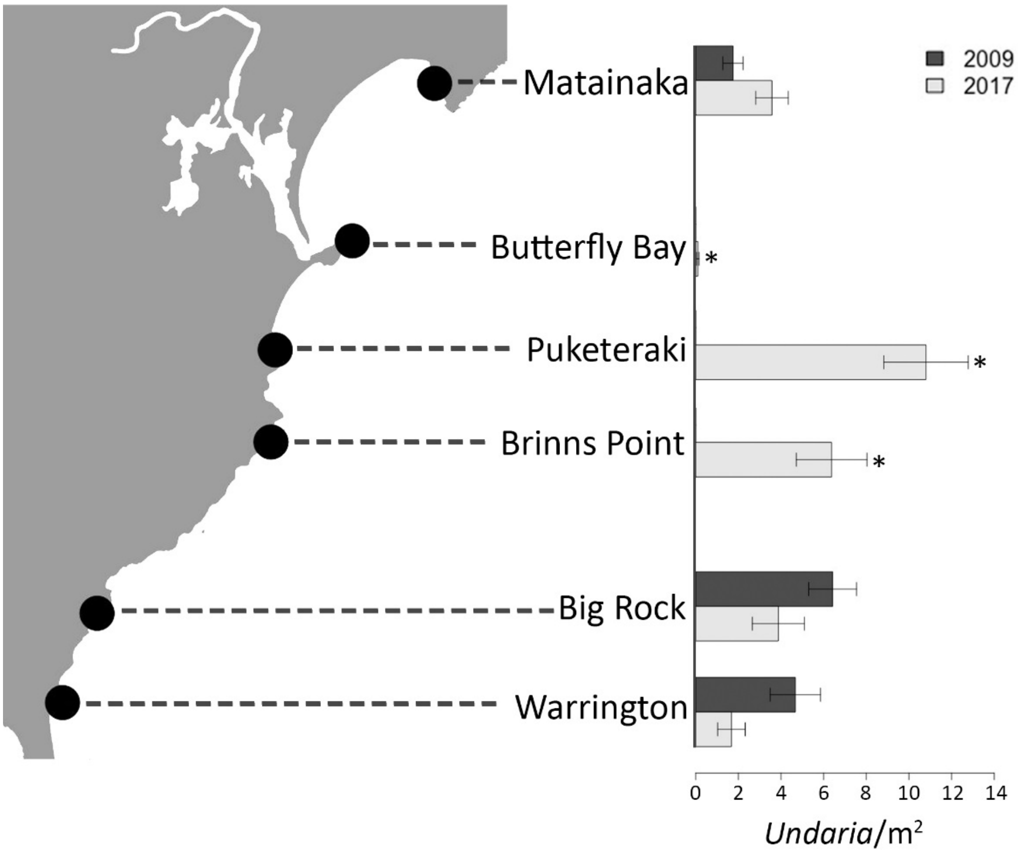

Figure 20: *Undaria pinnatifida* abundance (individuals/m^2) by site and year, bars represent the mean across depths (+/- 1 SE). Asterisks indicate sites where *Undaria* was newly observed in 2017.

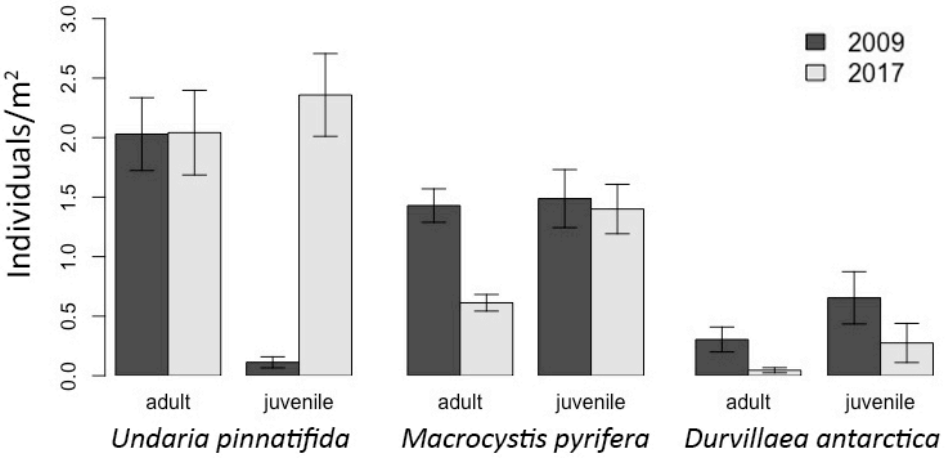

Figure 21: Abundance (individuals/m^2) of adults and juveniles for each of the three largest species by year. Means (+/- 1 SE) were calculated from the entire EOT dataset, pooled across sites and depths.

Macrocystis pyrifera

In 2009, *Macrocystis pyrifera* was the most common large brown macroalgae within the East Otago Taiāpure, exhibiting higher abundance at the 1–3 and 4–6 m strata (Figure 18, Table 7). *Macrocystis* density decreased 31 % in the 2017 survey, from 2.9 ± 0.3 to 2.0 ± 0.2 individuals/m^2 (Table 7). This shift is associated with a reduction in adult abundance, as juvenile counts remained stable (Figure 21). Change in *Macrocystis* was not homogenous across sites, with some locations exhibiting similar or higher abundance even as others decreased (Figure 22). The most notable loss of *Macrocystis* was at Big Rock, a site characterised by large boulders that can roll during storm events and disturb benthic communities. Yet while it is possible to speculate about site-specific factors that may have influenced *Macrocystis* populations (or site variability for other species) the broad time gap between surveys makes it impossible to identify a definitive cause. *Macrocystis* is widespread along the Otago coastline, so the decrease in abundance within the East Otago Taiāpure should not necessarily be concerning at this time. However, future surveys should continue to closely monitor *Macrocystis* abundance in order to identify true population trends.

Ecklonia radiata

Ecklonia radiata was observed at low frequency due to the fact that it generally inhabits deeper strata in this region than the ones surveyed for this study (Win 2010).

Fucales and Dictyotales

Durvillaea antarctica abundance, both adult and juvenile (Figure 21), decreased 70 % between the 2009 and 2017 surveys, while *Durvillaea willana* was observed very rarely during both survey years. Both species were generally found in shallow intertidal waters at exposed sites.

Distribution of fucoid species was not homogenous across the East Otago Taiāpure, with relative densities displaying significant variation between site, year, and depth (Figure 18, Figure 23). Of the fucoid species recorded, *Xiphophora gladiata* was the most abundant, particularly at the 0 and 0.5 m depth strata. *Xiphophora* was the dominant fucoid species at Warrington, while at other sites (e.g., Butterfly Bay) it was found concurrently with high densities of other fucoids. *Cystophora scalaris* and *Cystophora retroflexa* were also prominent members of the shallow intertidal, and both increased in overall abundance, 50 % and 90 % respectively, between surveys (Table 7). As a group, *Cystophora* species were most common at Brinns Point and Butterfly Bay (Figure 23).

Carpophyllum flexuosum was mainly observed at the 1–3 m depth range, but was absent almost entirely from two of the survey sites, Matainaka and Warrington (Figure 23). In 2017, *Marginariella sp.* was the most common genus in the 4–6 m stratum, displaying both greater overall abundance (Table 7) and wider depth range than observed in the 2009 survey (Figure 18). *Dictyota kunthii* was the most common non-fucoid phaeophyte, and was observed mainly in the 0.5 and 1–3 m range during both surveys, with slightly greater densities in 2017.

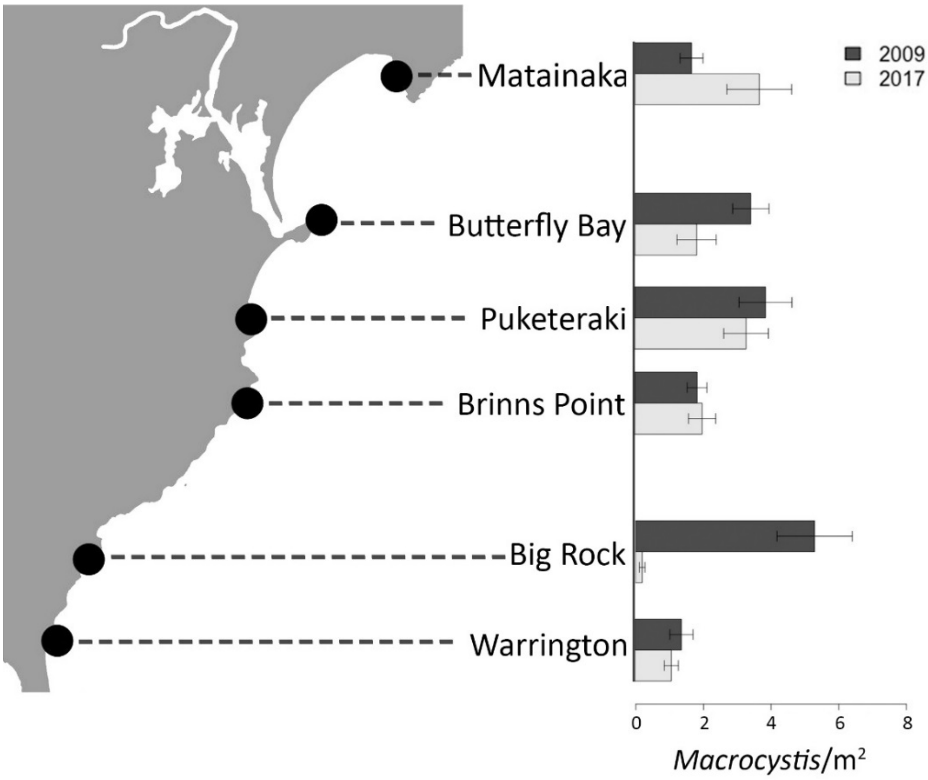

Figure 22: *Macrocystis pyrifera* abundance (individuals/m^2) by site and year, bars represent the mean across depths (+/- 1 SE).

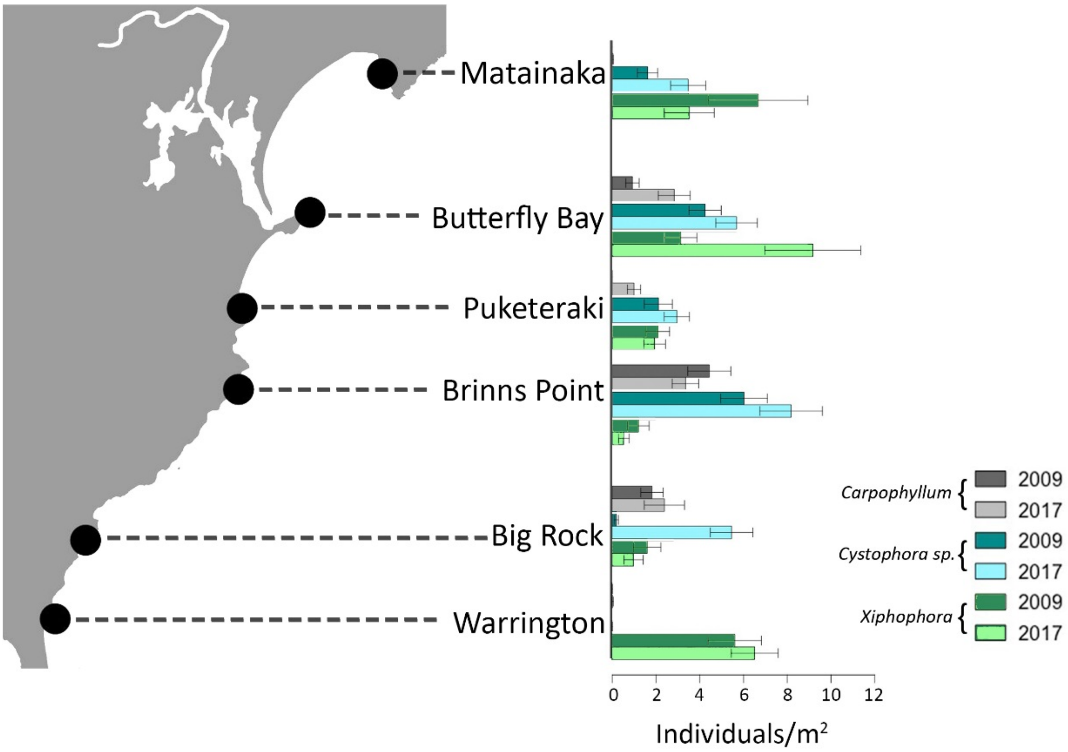

Figure 23: Abundance (individuals/m^2) of most common fucoid species (*Carpophyllum flexuosum, Cystophora spp, Xiphophora* gladiata) by site and year. *Cystophora torulosa, Cystophora retroflexa,* and *Cystophora scalaris* have been combined into one group *(Cystophora* spp*)*. Bars represent the mean across depths (± 1 SE).

Discussion: Macroalgal Monitoring: Case Study - East Otago Taiāpure (2009 – 2017)

The macroalgal monitoring efforts that are being used to assess the state of marine ecosystems within the East Otago Taiāpure were carried out in 2009 and 2017. The replicate surveys were conducted to track community changes over time with the objective of identifying and quantifying the relative abundances of species of large brown macroalgae. While some significant changes were seen in community composition between 2009 and 2017, further surveys will be needed in order to establish meaningful trends and to interpret them within a broader environmental context: it is not possible to draw conclusions regarding long-term trends within the East Otago Taiāpure based on only two survey events.

The eight years that elapsed between surveys make it impossible to relate community shifts to specific disturbance events, anthropogenic impacts, or environmental patterns. For future surveys it is also important to ensure that seasonal timing remains consistent between years. This consistency was not achieved for the 2009/2017 surveys, which makes it difficult to meaningfully compare abundances of annual species (e.g., *Dictyota kunthii, Desmarestia ligulata*) or the breakdown between adult and juvenile laminarians. Furthermore, climate change is expected to influence seasonal onset and duration (Lima & Wethey 2012), which could translate to shifts in macroalgal germination and recruitment success (Harley et al. 2012). In order to track whether such changes are occurring, it is important to conduct surveys at the same time of year.

This highlights the necessity of designing a monitoring programme that has a resolution appropriate to the challenges faced by the East Otago Taiāpure. With the exception of *Undaria*, the spread of which has been well-characterised within the East Otago Taiāpure (Russell et al. 2008, Jiménez et al. 2015a), it is also unrealistic to extrapolate trends for individual species from this dataset. However these survey data represent an important baseline: significant changes identified in this analysis may be used to inform hypotheses and to shape further East Otago

Taiāpure research. Regardless of the timeframe that is selected in the future, this dataset will continue to gain power with repeated surveys as long as base methodology remains consistent.

While this study was originally initiated as a response to the *Undaria* invasion and to understand pāua distribution, the data presented here also offer the opportunity to ask questions about a wide range of species and their relationships. What is the composition of mixed fucoid beds and how are the component species influenced by settlement of *Undaria*? Is *Macrocystis* distribution changing within the East Otago Taiāpure over time? How does this correlate with fisheries populations? It is important to recognise that the potential applications of this monitoring programme will change over time with management goals, and that surveys should evolve accordingly (Magurran et al. 2010). Long-term monitoring within the East Otago Taiāpure should be adaptive rather than rigid (Lindenmayer & Likens 2009), and frequency and methodology of surveys can and should be expanded as new questions arise.

This survey methodology provides the foundation for an adaptive monitoring approach. Using the abundance of canopy forming brown species as a minimum metric, other species (e.g., additional macroalgae, pāua or other invertebrates) and measurements (e.g., biomass, percent cover, morphology) could be incorporated as appropriate to accommodate new research questions or management goals. Sampling frequency could be increased in response to disturbance events or concerns that arise within the East Otago Taiāpure, and as additional resources for monitoring became available. If macroalgal surveys were completed on a yearly basis (e.g., Curiel et al. 2004), the data generated would provide the opportunity to understand the impacts of local disturbances such as storms, sedimentation events or heat waves. New management initiatives within the East Otago Taiāpure, such as the proposed ban on cutting attached kelp species, could be assessed as they progressed and there would be the capacity for adaptive approaches and adjustment as necessary. Higher resolution provided by yearly monitoring may also enable community changes driven by environmental cycles (Gray & Hartvig 1983) to be distinguished from changes caused by anthropogenic factors.

Yearly monitoring is not currently feasible due to the increased effort and resources required. Surveys with wider time spread may be used to track slower processes, such as the community shifts that will be likely to occur due to climate change (Mieszkowska et al. 2006, Hepburn et al. 2011) or coastal development. A sustained long-term dataset will help establish natural patterns of variability within the East Otago Taiāpure, providing context for the interpretation of short-term studies (Wolfe et al. 1987). If surveys are designed iteratively, with methods flexible to new questions, then future research can also be designed to enhance the monitoring programme.

In terms of the methods for the analyses, an alternative approach to this dataset would involve fitting generalised linear models (GLMs) to each species in the community (Wang et al. 2012). This method can provide more statistical power than traditional distance-based analyses, and is more sensitive to variability in rare species because it accounts for the mean-variance relationship. A further benefit of GLMs is that they can be used predictively, making this approach a promising candidate for evaluating long-term community trends and developing hypotheses.

An essential component of a viable long-term dataset is consistent taxonomic identification between years. This can be ensured by having the same team members complete subsequent surveys, and by rigorously training new team members on identification using guidebooks and field practice. One benefit of focusing on large brown macroalgal species is that they are generally easier to identify than many smaller species, requiring less taxonomic expertise while maintaining data consistency. Abundance of canopy forming species has been shown (Smale 2010) to be a reliable biodiversity surrogate above the quadrat/transect level (with improved outcomes with increasing spatial scale), indicating that a reduced taxonomic scope can still be used to monitor diversity and richness within the East Otago Taiāpure.

The depth strata that were chosen for the monitoring are appropriate to EOT-related management questions, as recreational and customary fishing largely occur in shallower waters. However, it could be beneficial to include another deeper stratum (e.g., 10–12 metres) in future surveys in order to include *Ecklonia radiata*. This is an important habitat-forming species for many fisheries in the EOT, such as blue moki (*Latridopsis ciliaris*) and blue cod (*Parapercis colias*) (Win 2010). Additionally, *Ecklonia* has been known to experience mortality events in other areas of New Zealand in the past (Cole & Syms 1999), and is sensitive to changes in environment (Wernberg et al. 2010, 2013). This species is widely distributed throughout New Zealand, providing the opportunity for knowledge of EOT *Ecklonia* populations to be integrated with survey efforts from other regions.

Physiological investigations (Pritchard et al. pers. comm.) and field surveys (Casas & Piriz 1996) have shown that *Undaria* is capable of surviving at deeper strata and in lower light conditions than found in sites it currently occupies within the East Otago Taiāpure. If the depth expansion of *Undaria* observed in this analysis represents a continuing trend, then the addition of a 10-12 m stratum could help track future expansion. Currently, these surveys are being employed to inform discussions regarding the *Undaria* invasion and potential mitigation strategies, an effort that will be ongoing in the East Otago Taiāpure. These data have also been used to support the proposal of a regulation banning the cutting of attached kelp species. Monitoring information can be applied to any future efforts to develop macroalgal harvest, and to evaluate sustainability if harvest takes place (Schiel & Nelson 1990). Macroalgal community information should be integrated with population data for local fisheries species, such as pāua (*Haliotis iris*), that rely on seaweed for sustenance and habitat (Poore 1972). Such an approach would provide a better understanding of fisheries health, particularly in light of the current efforts to support customary fisheries. Beyond this, knowledge of community structure in the present will inform future coastal restoration projects.

4.3 Evaluation of the mapping of kelp forests

The literature review on mapping macroalgal beds (Section 1.3.1.2) pointed out that a combination of techniques is necessary depending on the size of the target area and the nature of the habitats being studied, and the species present i.e., *Macrocystis* with surface floating canopies, intertidal macroalgae, and subtidal beds, all require different approaches.

In this objective, different methods were tested for their effectiveness:

- Mapping *Macrocystis* beds from a small vessel
- Using a drone to map intertidal/shallow subtidal macroalgal beds and *Macrocystis*
- Mapping subtidal forests with underwater videography

As the process of extracting data from videos is very time consuming, the use of Machine Learning to optimise video analysis was investigated, involving the development of software.

Techniques not tested during this study, and their applications, have been evaluated based on recent literature (e.g, acoustic mapping and satellite imagery) and are summarised in Appendix 2.

4.3.1 Distribution of *Macrocystis pyrifera* in Wellington harbour

Methods: **Wellington *Macrocystis* mapping**

The presence of *Macrocystis* beds from surveys in November 1988 and April 1990, illustrated by Hay (1990a), was digitised in ARC GIS (NZGD_2000_New_Zealand_Transverse_Mercator projection). This enabled a snapshot of the number and location of *Macrocystis* beds recorded at that time to be assigned a geospatial reference. This provided a basis for comparison of subsequent *Macrocystis* distribution assessments by ground truthing and with satellite images which can then be overlaid and compared in GIS layers.

An image sourced from the LINZ Data Service: Wellington 3.3 Rural aerial photos (2012-2013) and licensed for re-use under the Creative Commons Attribution 3.0 New Zealand licence was used as a GIS base map background. Probable *Macrocystis* beds were identified and traced at a magnification of 1:1250 (Figure 24).

The criteria for identifying *Macrocystis* beds were assemblages of dark patches some distance from shore that would indicate seaweed present in deep water and growing up close to the surface. *Macrocystis* is distinctive from other species with sometimes dense surface mats of foliage extending from depths of 10 m or more. Each identified bed was given a GIS shape, attribute data for a central GPS latitude and longitude, and an area.

Figure 24: An example of *Macrocystis* beds identified from an aerial photo. Dense, sparse, and long thin beds have been identified.

This technique provided a relatively robust and accurate method of estimating the location and area of *Macrocystis* beds.

Image validation

The harbour was surveyed by boat (6.8 m McLay hard top), cruising near shore. When *Macrocystis* plants were seen, the observations were recorded with a unique number. Latitude and longitude coordinates were taken from the boat GPS chart plotter (Lowrance HDS8 chart plotter) and a photo taken. If a single plant, or isolated group of plants was recorded, a single GPS coordinate was assigned. The GIS shape area for these isolated plant observations was nominated as 9 square meters, being the smallest area that could be still be seen from a large-scale GIS map.

If the *Macrocystis* patch was in a long narrow band, the start and finish point was recorded and a shape of the area within those boundaries was recorded into the GIS data base. An example of this are the observations from along the Hutt Motorway, adjacent to the Ngauranga interchange in Figure 28. For larger and more complex *Macrocystis* beds (e.g., along the eastern side of Miramar Peninsula) the boat was manoeuvred around the fringes of the beds and several GPS coordinates were taken. These were then used to draw a GIS shape representing the bed. Shape files of *Macrocystis* beds were attributed with an area name (e.g., Worser Bay), the date surveyed, the boat used for the survey, the observers, the area of the shape, notes and some effort was made to create a hyperlink to the relevant photo.

Results: Wellington *Macrocystis* mapping

Figure 25 shows the original Hay distribution map (Hay 1990a) and Figure 26 shows the same distribution recreated as a layer in Arc GIS with a satellite background. Although Hay (1990a) exaggerated for clarity the size of the beds in Wellington harbour and a comparison by the areas in 1990 and 2017 is not possible, areas of persistence or change can be noticed.

Hay – Macrocystis *as an indicator of sea temperatures* 331

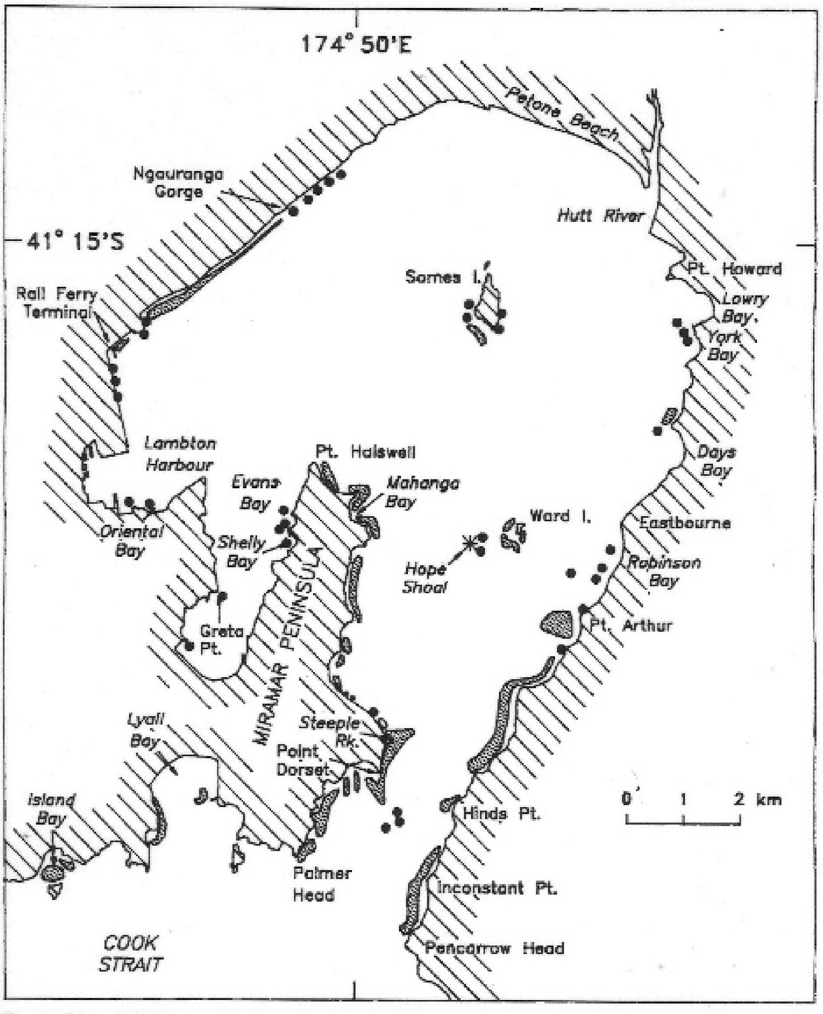

Fig. 4 – Map of Wellington Harbour showing localities mentioned in the text and the distribution of the main beds of *Macrocystis* (stippled). Sporadic distribution and isolated records are marked with solid dots. Data mainly from surveys in November 1988 and April 1990. Note: The size of the beds is exaggerated for clarity.

Figure 25: Original Hay distribution of *Macrocystis pyrifera* in Wellington (Hay 1990a).

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Figure 26: The distribution of *Macrocystis* in Wellington from Hay (1990a) recreated as a layer in Arc GIS with a satellite background.

Figure 27: Comparison of *Macrocystis* beds in the Kaiwharawhara area from Hay (1990a) and aerial photo analysis.

In the Kaiwharawhara area (Figure 27), apart from the three sites identified by Hay from the Northern end of Aotea Quay, *Macrocystis* plants appear to have persisted in similar areas until 2013. The three circles in a row alongside Aotea Quay may indicate that *Macrocystis* was present and growing on wharf piles, but thalli were not obvious in the aerial photograph and may be obscured by the wharf.

Figure 28: A comparison of Kaiwharawhara *Macrocystis* beds recorded by Hay (1990a), aerial photograph analysis 2013 and ground truthing by observations from a boat in 2017.

Ground truthing showed the persistence of *Macrocystis*, although in a narrow stripe, along Hutt Motorway adjacent to the Ngauranga interchange (Figure 28).

Figure 29: A comparison of Miramar Peninsula *Macrocystis* beds recorded by Hay (1990a), aerial photograph analysis 2013 and ground truthing by observations from a boat in 2017.

Macrocystis was not observed in Evans Bay (Figure 29) apart from a few plants at Shelly Bay and was not found at all at Point Halswell, where it had still been present in 2009 (Figure 30). To verify the presence of subtidal plants underwater video were acquired that showed only sporadic plants of *Carpophyllum maschalocarpum*. Intermittent *Macrocystis* beds were observed from Kau Bay to Seatoun (Figure 29).

Figure 30: *Macrocystis* bed at Point Halswell in 2009.

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Figure 31: Mapping of a large continuous bed near Seatoun. The extremities were mapped out by manoeuvring the boat and recording a series of GPS way points.

A large *Macrocystis* bed was observed around Seatoun (Figure 31).

Figure 32: A comparison of Matiu Somes Island *Macrocystis* beds from Hay and ground truth survey by boat 2017.

At Matiu Somes Island *Macrocystis* was not observed in the southern side, in contrast with Hay's observation and few isolated plants were observed on the northern side (Figure 32).

Discussion: *Macrocystis* mapping

Mapping *Macrocystis* beds in Wellington harbour following the contour of the bed with a small boat was is quite inexpensive and could be repeated yearly or could even be proposed as a citizen science project. In Puget Sound (USA) a protocol to map kelp from kayaks has been developed (Bishop 2016). Although from the surface there is the risk of missing subtidal plants, the field work was carried out in very good conditions (e.g., clear water, low wind, sunny) and around low tide. Although we couldn’t compare the area of *Macrocystis* with Hay’s (1990a) data, we were able to identify areas where *Macrocystis* was absent e.g., at Matiu Somes Island and Point Halswell. A recreational photograph (Figure 30) taken in 2009 at Point Halswell showed that *Macrocystis* was still present then. Repeat photography has long been employed as a valuable tool to monitor long-term geological or ecological changes in terrestrial landscapes. Few studies, however, have used marine images to document changes (Reimers et al. 2014).

Mapping macroalgal beds subtidally presents more challenges, and again a multi-approach needs to be considered. An assessment of the best, cost-effective method of benthic habitat mapping techniques for mapping and monitoring seagrass meadows, that tested ultra-high-resolution aerial imagery, surface based underwater videography and single-beam echo sounder, indicated that all methods were suitable for mapping seagrass but all displayed limitations in either detection accuracy or spatial coverage (Puhr et al. 2014).

4.3.2 Mapping by drones/ UAV (Unmanned Aerial Vehicle)

Introduction: **Mapping by drones/ UAV**
The use of drones, or unmanned aerial vehicles (UAVs), is an emerging technique in remote sensing. At the 17th International Symposium GEOHAB Marine Geological and Biological habitat mapping (Nova Scotia – 2017), several presentations explored the use of UAVs e.g., for mapping coastal processes and intertidal marine habitats (Ierodiaconou et al. 2017, Lightfoot et al. 2017; Diesing et al.2017), to assess of seaweed resources (Rossiter et al. 2017), to map ecological phase shifts on coral reefs (Fallati et al. 2017).

Three trials to acquire videos and still images were carried out to test whether this approach is suitable for the New Zealand coastal areas.

Trial 1: Whitireia Park (Porirua) 9 Feb 2017 (Dave Allen, NIWA)
Trial 1 was conducted using a DJI Inspire 1 drone with a Zenmuse X3 camera (https://www.dji.com/). This camera has the 35 mm equivalent focal length of 20 mm. Files produced have the height information included in the file name and metadata. GPS data are also included in the metadata (Figures 33 and 34). Adobe software (Bridge, Camera Raw and Photoshop, https://www.adobe.com/) was used to process the images.

Figure 33: Example of a screen shot from video taken at 19.5 m above the ground.

Figure 34: Example of a screen shot from video taken at 110 m above the ground.

Trial 2: Kaikoura 5–6 October 2017 (Leigh Tait, NIWA)
The second trial was conducted using an ALTUS LRX drone (https://altusintelligence.com/uas-products/) with a Sony A5100 camera with a Voigtlander 15 mm superheliar wide angle rectilinear lens. The flight altitude was 60 m at a flight speed of 8 m/s.

To enable appropriate conversion of images to a georeferenced dataset for spatial analysis, ground targets were laid out evenly across the area to be sampled. These ground targets were then referenced using an RTK (Real Time Kinematic) positioning system (Figure 35).

Figure 35: Example of a ground target used to geo-reference a set of UAV images.

Evidence from mapping the intertidal and nearshore subtidal zone of Kaikoura shows that cloud cover may play a role in observing the subtidal zone (Figures 36–37). The image taken under zero cloud cover (Figure 37) reveals greater detail in shallow water (1–2 m), although identification of species is unlikely to be possible.

Figure 36: AUV imagery covering both intertidal and subtidal zones. Image was taken with cloud cover.

Figure 37: AUV imagery covering both intertidal and subtidal zones. Image was taken during low cloud cover.

It is possible to map the coverage of surface canopies of the giant kelp *Macrocystis pyrifera* (Figure 38, Note that this imagery was from fixed wing aircraft not AUV). Through colour processing protocols of the RGB signal, kelp can be separated from other features, and with appropriate geo-referencing can be used to estimate coverage of the surface canopy.

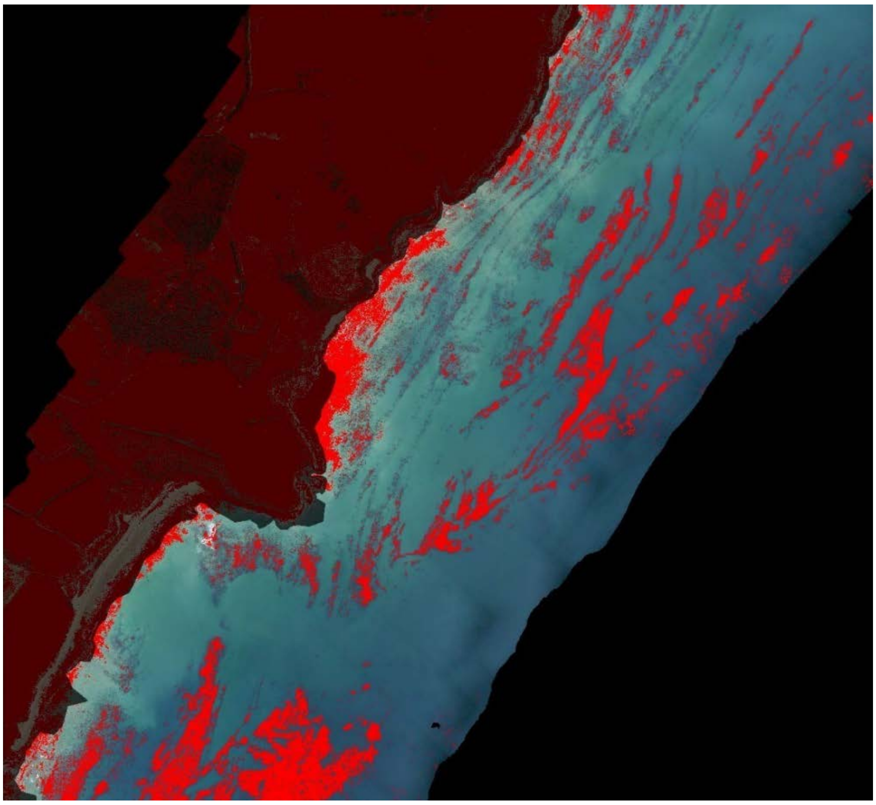


Figure 38: Processed imagery showing surface *Macrocystis pyrifera* (Red) along the Otago Coastline. Images were taken from high altitude on a fixed wing aircraft.

Trial 3: Wellington Harbour Kau Bay March 2018 (Nicole Miller, Wellington Underwater Club)

The third trial was conducted using a DJI Mavic Pro drone with an inbuilt camera (see https://www.dji.com/mavic/info) to fly transects at high tide over Kau Bay, Wellington (Figure 39). Photomosaics were realised with PTGui software (https://www.ptgui.com/) (Figures 40 and 41).

30/03/2018 15:50:31 Maupuia
Flight Distance 1106.7 m
Flight Time 09:12
Height 10.4 m
Distance 55.9 m
Horizontal Speed 1.8 m/s
Vertical Speed 0.0 m/s
-41.288991, 174.834453

Figure 39: Image of the drone trajectories flown over Kau Bay.

Figure 40: Photomosaic of *Macrocystis* bed at Kau Bay, at an elevation of 10.4 m.

Figure 41: Detail from a photomosaic taken at an elevation of 10.4 m.

Results and Discussion: Mapping by drones/ UAV

The drone tests conducted in this project, showed that the best results were achieved on a clear sunny day when the sun was lower in the sky (i.e., not within three hours of midday), as well as preferably a calm sea state for 3 or 4 days prior to the images being acquired. In using drones for mapping there are several issues that need to be considered. There are strict regulations around the use of drones in populated areas and no-fly zones in a radius of 4 km around airports. Another significant limitation of drones is the impact of wind in coastal environments as well as limited flying distance. Drones, however, could be tethered to minimise the risk in populated areas. To create an orthophoto (georeferenced mosaic of drone photos) coastal ground targets need to be surveyed, which are available on the coastline but not in the water. Being a land based activity, the field work involves few people and it is not constrained by boat/skipper availability. However, road access is limited along the New Zealand coastline and drones have a limited distance for flights from the operator. As for other aerial techniques, it is crucial to understand the depth to which macroalgal beds can be identified with confidence. The use of a multispectral camera or radiometric sensors may solve some of these issues but have not been tested during this current work.

Our first drone trial at Whitireia (Porirua) was done in perfect conditions, with blue sky, calm sea, and clear weather for several days prior to the trial. There were some problems with the strong sunlight reflection from the water surface that were solved by changing the camera angle. Videos were taken about 2 hours before low tide and the areas dominated by seaweed were clearly distinguished from the sandy bottom. Identification of species from this test was not possible. Flying at low tide and at a lower elevation should make it possible to identify dominant species. In Victoria, a drone has been used to estimate *Hormosira banksii* on intertidal reefs with less than 1 cm resolution images (Murfitt et al. 2017). The test carried out on a *Macrocystis* bed in Wellington Harbour using an amatorial drone has shown that drones are suitable for mapping giant kelp beds. Moreover, the continued monitoring of bull kelp *Durvillaea*, that has shown a decrease of 70% in East Otago between 2009 and 2017, could be achieved by drone surveys with a protocol being designed to acquire data consistently. Drones are now quite inexpensive and some regional councils (e.g., Hawkes Bay) are already testing their use for a range of applications.

Conversely, the drone testing carried out in Kaikoura has shown more constraints than advantages. Nevertheless, this is a field that shows a great potential with more tests and investigations needed to develop a mapping methodology for seaweed. Drone data could be integrated with satellite imagery to achieve higher resolution or for ground-truthing. In a study to map the green tides in the Yellow Sea, a combination of MODIS satellite images, *in situ* observation data, and drone images were utilised (Xu et al. 2017). MODIS images could detect large (more than 100 m) floating green algae patches but ignore small patches less than 10 m in width that were captured by the drone.

Analysing the drone imagery for coverage of macroalgae requires processing the RGB signal of images to extract colour signatures of kelp or macroalgae. These processing protocols will potentially differ between macroalgal species, and possibly time of year as pigments change. Use of multispectral cameras may improve the processing of this data, especially for comparing between species, although the pixel size of multispectral cameras may also affect the resolution of the images. However, these techniques are less well suited to mapping subtidal beds, especially in deteriorating or variable water clarity. While filters may be applied to the camera (particularly polarizing filters), this will only help in dealing with the surface reflection, but not the penetration depth of the imagery under the typically turbid conditions of New Zealand.

Colour processing protocols will vary with differences in water-quality properties, particularly sediments. There are several requirements for obtaining workable imagery of surface canopies of *M. pyrifera*, similar to those for aerial photography In particular, flights must be done at a low sun angle (i.e., morning/evening), ideally at low tide, during calm periods and with low quantities of suspended particulate matter. The alignment of these conditions is difficult and this is a significant constraint in the use of satellite imagery for this purpose.

Furthermore, geo-referencing images over the water is difficult because there are often no features for referencing, or features are unevenly spaced across the imagery. Few features, or features clustered over only a portion of the images (i.e., features only on the coastal extent of images), greatly affects the ability of the software to factor out lens distortions. For this reason, images of offshore kelp forests must also include images of the coastline in equal or greater proportions.

Currently a Catalyst project (New Zealand-Germany) is developing a small, low-cost camera system for measuring coastal water colour and habitats that can be flown on drones (multicopter and fixed-wing). The measurements of water colour from the aerial sensors will be designed to allow users to estimate water clarity, suspended sediment concentrations, algal biomass, rivers plumes and map habitat types (e.g., rocky reefs, seagrass, macroalgae, mangroves) in conditions ranging from clear oceanic water to shallow, turbid estuaries.

Drones are becoming more affordable, smaller and easier to manoeuvre, with significant potential for environmental studies. It is possible to gather a wide range of environmental data by mounting different sensors. Miniaturised sensors are being developed/adapted for UAV, including hyperspectral imagers, LIDAR, synthetic aperture radar, and thermal infrared sensors (Klemas 2015). UAVs can access remote areas and can provide higher resolution than satellites being closer to the targets and covering smaller areas, providing data for integration with satellite data and in field observations. The use of a low-cost drone allowed coastal fish nursery areas to be identified and accurately mapped in Giglio Island (Italy) (Ventura et al. 2016) who provided a detailed methodology from video acquisition to post-processing image analysis that could be applied to other habitats. In Australia, UAV has been used to capture under 1 cm resolution data from intertidal reefs, providing reliable estimates of the dominant species *Hormosira banksii*. Surveys and analysis of the drone data took approximately half the time of on-ground quadrat observations with comparable coverage estimates (Murfitt et al. 2017). In the northern Gulf of Alaska, drones were used for intertidal monitoring of seagrass and macroalgae communities (Konar & Iken 2017). The combination of the drone with image processing techniques can provide an inexpensive, automated, and accurate system to produce ultrahigh-resolution maps (Koeva et al. 2016). Drone derived data could be integrated with satellite imagery to achieve higher resolution images or for ground-truthing (Xu et al. 2017).

Although internationally satellite imagery has been widely used to map kelp forests and subtidal macroalgal beds (see Appendix 3), during this project we only examined the freely available satellite imagery SPOT maps 2014, with a spatial resolution of 1.5 m, and found that this was not suitable for identifying *Macrocystis* beds. Examples of these images are shown in Appendix 7. Only satellite imagery at very high-resolution should be considered for mapping *Macrocystis* beds in New Zealand and such images are currently available from commercial satellites (e.g., Digital Globe). Google Earth Engine, however, has developed a planetary-scale platform for Earth science data and analysis, available to scientists (https://earthengine.google.com/). With the fast development of satellite technology, high-resolution imagery may become freely available for research in the near future.

The advantage of aerial surveys, in contrast to satellite imagery, is the ability to plan the flight time, as well as the capacity to choose the type of optical sensors to use. However, it is difficult to distinguish which seaweed species are in the water when examining an aerial photo. Dark rocks or rocks covered in short seaweed species are difficult to determine from aerial/satellite imagery. Likewise, seagrass mats can sometimes be indistinguishable from dark coloured sand deposits. Shorter species such as *Carpophyllum maschalocarpum* and *Ecklonia radiata* (about 1–2 m in height) generally appear as a band of dull subsurface brown/grey in contrast to the dark, surface or near surface *Macrocystis* fronds. Tide, current and clarity of water can have a significant impact on how easily and how accurately the *Macrocystis* beds can be recorded. For example, the best opportunity for observing the extent of *Macrocystis* beds would be at low tide, slack current with clear visibility. This would enable plants that were subsurface (and often at the perimeter of the bed) to be included. If the tide was rising and a current began to flow, some *Macrocystis* plants would become submerged and difficult to see. Anderson et al. (2007) noted that aerial surveys of the same area of kelp, surveyed on the same day consecutively with the same method could show differences of up to four times. Thus, as with satellite imagery, tides, currents, waves should be considered and surveys planned accordingly when conducting aerial surveys.

Different sensors can be mounted on aircraft, e.g., hyperspectral sensors which have several applications in agriculture and forestry and have been proven effective in distinguishing green, brown and red seaweeds by their reflectance spectra. Fyfe (2003) showed that it was possible to recognise three species of seagrass by their spectral signatures. As aerial surveys are quite expensive they are being progressively substituted by drones.

Mapping intertidal to shallow subtidal macroalgae and *Macrocystis* beds could be achieved with drones at an affordable cost compared to aerial surveys or commercial satellite imagery. With drone derived imagery, there will also be challenges in identifying macroalgae from the images, dealing with light reflection, attenuation through the water column, and changing water clarity over time.

4.3.3 Underwater Video imagery

Methods & Results: Underwater Video imagery

Image Capture

Field work was carried out with a McLay 5.91 m aluminium vessel (*RV Rahope*) in Wellington harbour and on the Wellington south coast. The underwater videos were acquired with a towed camera system specifically designed for shallow water (Figure 42) and consisted of:

- SplashCam Deep Blue HD
- Atomos Ninja Blade 5 " HDMI On-Camera Monitor & Recorder with 240 GB Solid State Disk Drive
- GoPro Hero4 (GoPro, Inc) in a Dive Housing
- GPS video overlay system to the Splashcam
- Pair of waterproof red 5 mW lasers for determining scale
- Sensus Ultra unit from Reefnet to automatically record depth and temperature profiles
- 2 × 40 m Underwater Diving Video Light 120 LED 1500 lumens with built-in rechargeable battery
- Strong and light framework
- 50 m cable with 75 kg working strength (300 kg breaking strength)

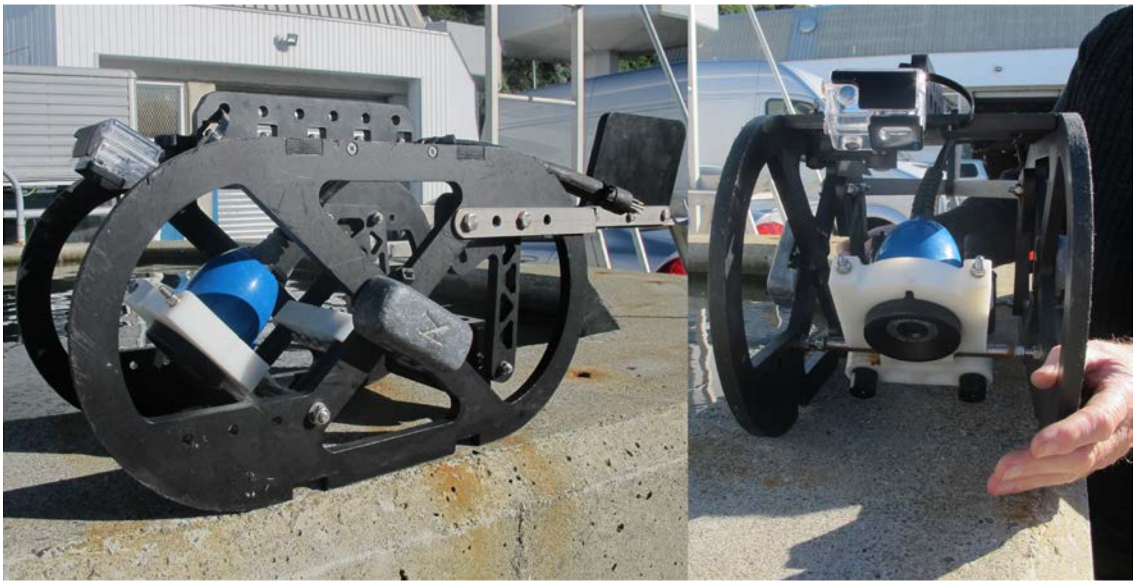


Figure 42: Towed camera system incorporating a splash cam and a GoPro.

The two cameras were set up with one facing down (Splashcam) and the other facing forward (GoPro). The Splashcam was connected to a monitor to enable live viewing on board. The GPS video overlay system was connected to the Splashcam resulting in videos with position (latitude and longitude), date and universal time (UTC) recorded on the video images (Figure 43). The videos can also be edited to display the transect number. The GPS position of the boat was recorded. The vessel was working in very shallow water (usually less than 10 m) and at very low speed, so the issue of cable drift was not considered significant.

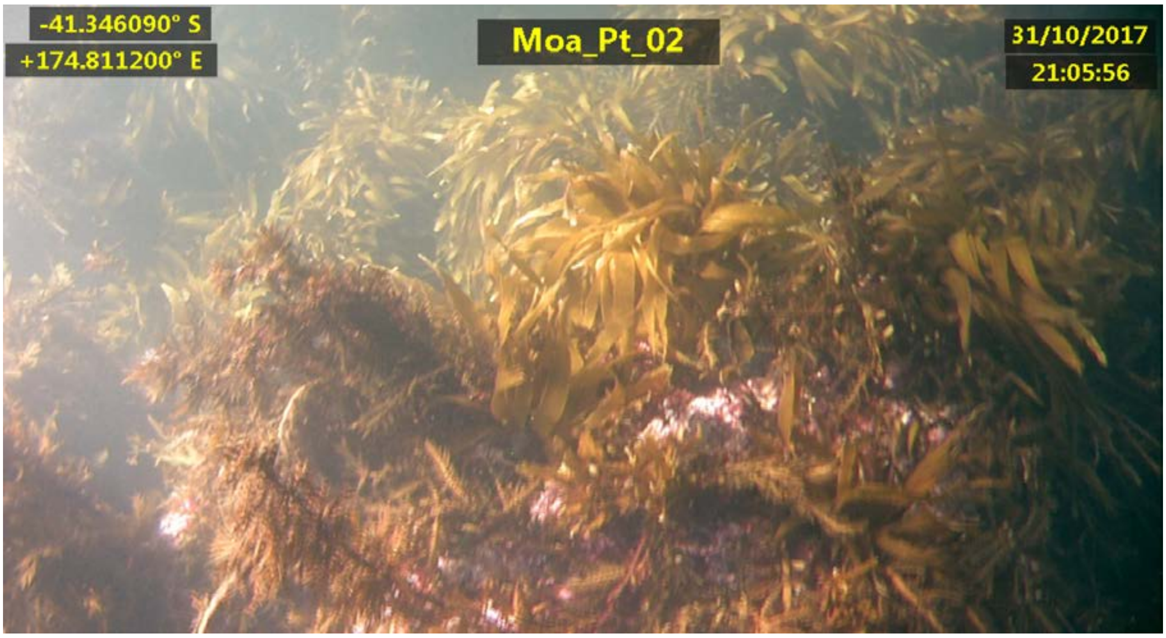
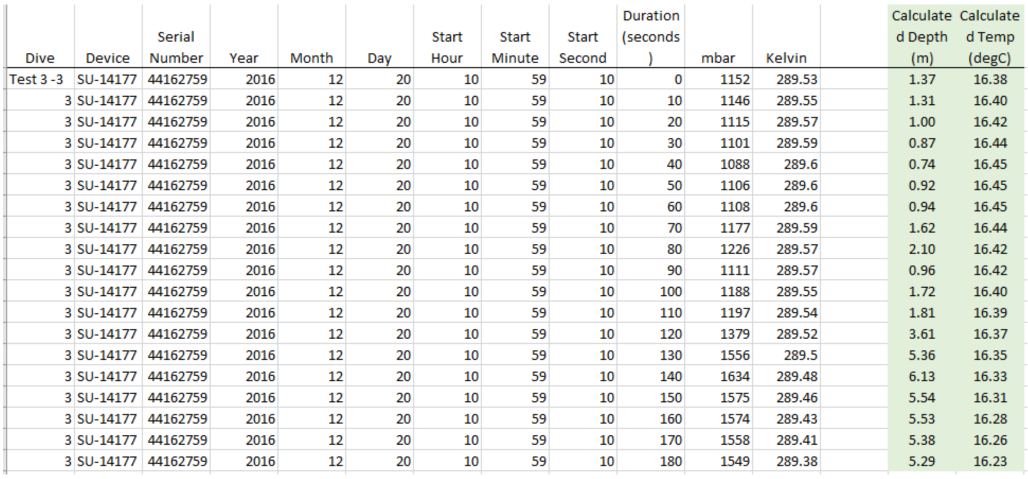


Figure 43: Example of captured image display including GPS coordinates, location, date and time (UTC).

Three people were necessary on board for video acquisition. The system weighs about 10 kg and can be operated by one person (a winch was used). However, a second person was required to watch the monitor and guide the person operating the camera, in addition to the skipper. The boat speed was maintained at about 1 knot, as faster speeds compromised the video quality.

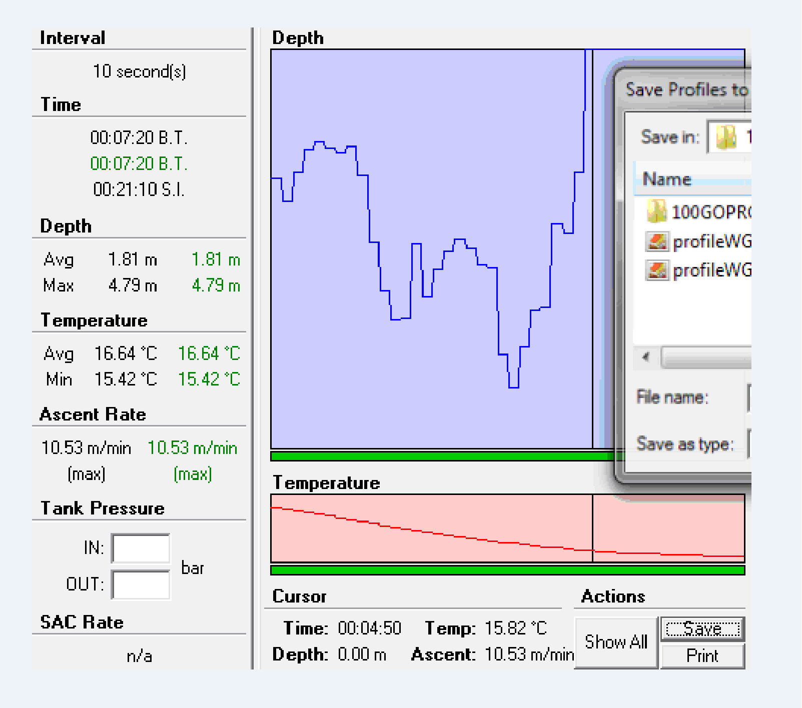
The towed camera system was tested on the 20 December 2016, with three videos taken in Wellington harbour, and six along the south coast and with required modifications made to the camera weight and the lighting system.

The sensor recording water temperature and depth produced a file (Figure 44) and a profile of the dive (Figure 45) associated with each video/dive, providing important data in case of future monitoring.



Dive	Device	Serial Number	Year	Month	Day	Start Hour	Start Minute	Start Second	Duration (seconds)	mbar	Kelvin		Calculated Depth (m)	Calculated Temp (degC)
Test 3 -3	SU-14177	44162759	2016	12	20	10	59	10	0	1152	289.53		1.37	16.38
3	SU-14177	44162759	2016	12	20	10	59	10	10	1146	289.55		1.31	16.40
3	SU-14177	44162759	2016	12	20	10	59	10	20	1115	289.57		1.00	16.42
3	SU-14177	44162759	2016	12	20	10	59	10	30	1101	289.59		0.87	16.44
3	SU-14177	44162759	2016	12	20	10	59	10	40	1088	289.6		0.74	16.45
3	SU-14177	44162759	2016	12	20	10	59	10	50	1106	289.6		0.92	16.45
3	SU-14177	44162759	2016	12	20	10	59	10	60	1108	289.6		0.94	16.45
3	SU-14177	44162759	2016	12	20	10	59	10	70	1177	289.59		1.62	16.44
3	SU-14177	44162759	2016	12	20	10	59	10	80	1226	289.57		2.10	16.42
3	SU-14177	44162759	2016	12	20	10	59	10	90	1111	289.57		0.96	16.42
3	SU-14177	44162759	2016	12	20	10	59	10	100	1188	289.55		1.72	16.40
3	SU-14177	44162759	2016	12	20	10	59	10	110	1197	289.54		1.81	16.39
3	SU-14177	44162759	2016	12	20	10	59	10	120	1379	289.52		3.61	16.37
3	SU-14177	44162759	2016	12	20	10	59	10	130	1556	289.5		5.36	16.35
3	SU-14177	44162759	2016	12	20	10	59	10	140	1634	289.48		6.13	16.33
3	SU-14177	44162759	2016	12	20	10	59	10	150	1575	289.46		5.54	16.31
3	SU-14177	44162759	2016	12	20	10	59	10	160	1574	289.43		5.53	16.28
3	SU-14177	44162759	2016	12	20	10	59	10	170	1558	289.41		5.38	16.26
3	SU-14177	44162759	2016	12	20	10	59	10	180	1549	289.38		5.29	16.23

Figure 44: Water depth and temperature profile data.



Interval
10 second(s)

Time
00:07:20 B.T.
00:07:20 B.T.
00:21:10 S.I.

Depth
Avg 1.81 m 1.81 m
Max 4.79 m 4.79 m

Temperature
Avg 16.64 °C 16.64 °C
Min 15.42 °C 15.42 °C

Ascent Rate
10.53 m/min (max) 10.53 m/min (max)

Tank Pressure
IN: bar
OUT: bar

SAC Rate
n/a

Cursor
Time: 00:04:50 Temp: 15.82 °C
Depth: 0.00 m Ascent: 10.53 m/min

Figure 45: Profile of the towed camera system dive.

On February 10th, 2017, three underwater videos were taken in Wellington Harbour, parallel to the coast around Point Halswell (Figure 46), to verify whether submerged plants of *Macrocystis* were present, as recreational photos taken in 2009 showed a dense bed at the site. Six videos were taken in Breaker Bay (Figure 47), some perpendicular to the coast, some parallel, following the edge of the kelp bed. On November 1st, 2017, videos were taken at Moa Point (Figure 48) and Palmer Head (Figure 49) on the Wellington south coast, along transects perpendicular to the coast.

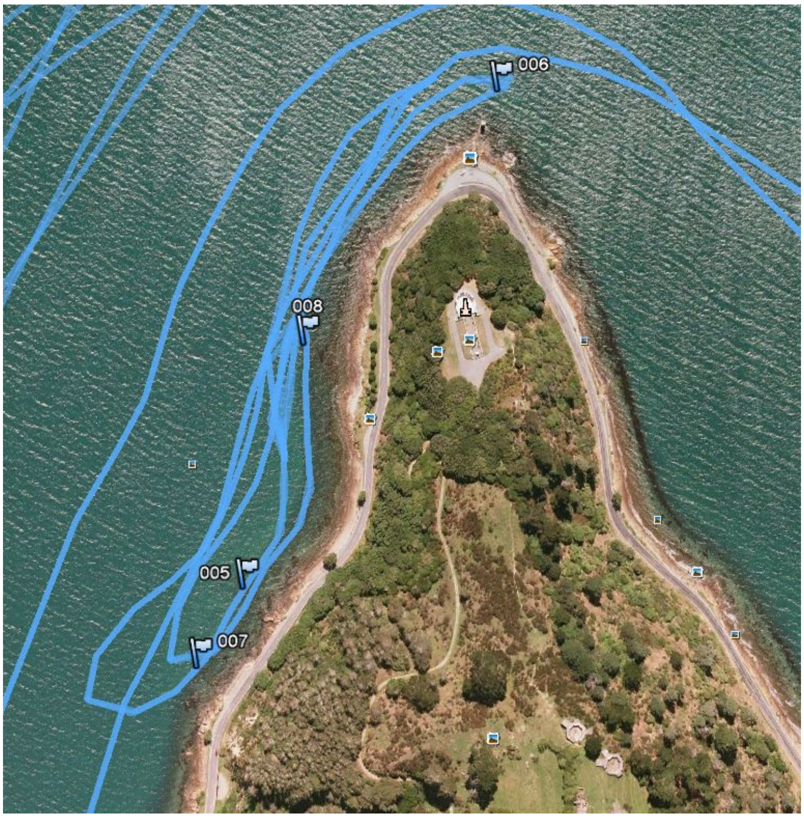


Figure 46: Point Halswell: underwater video transects taken parallel to the coast.

Figure 47: Breaker Bay: underwater video transects taken parallel and perpendicular to the coast.

Figure 48: Moa Point: underwater video transects positioned perpendicular to the coast.

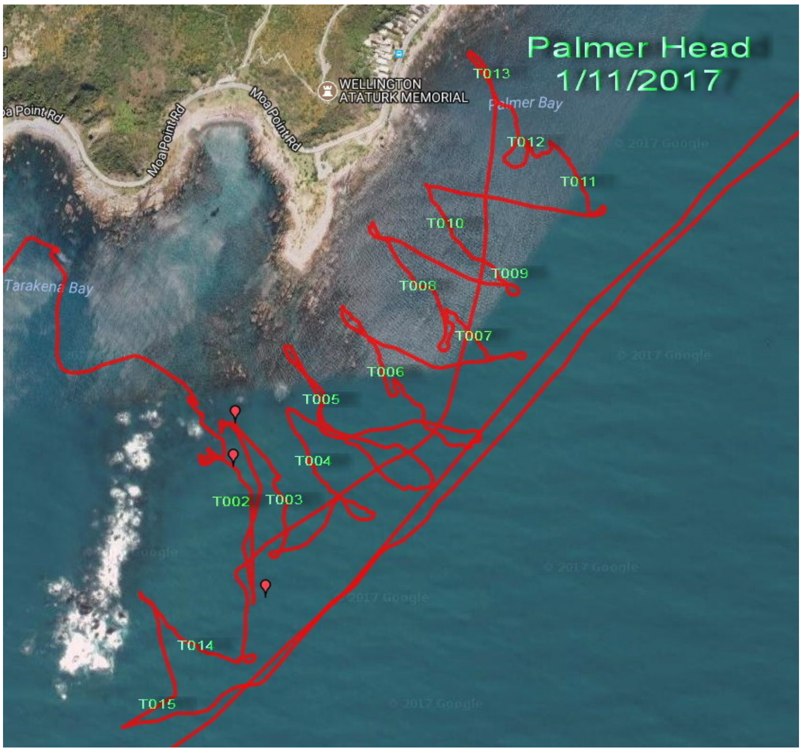


Figure 49: Palmer Head: underwater video transects positioned perpendicular to the coast.

4.3.4 Image Analyses: Use of Machine Learning and computer vision techniques

Machine learning and computer vision techniques were investigated to see whether they would be useful in quickly analysing underwater videos, to provide a cost-effective means of monitoring seaweed presence/absence and mapping macroalgal beds distribution.

This work is divided into four sections. The methods and results are presented in Appendix 8.

A. Metadata Extraction from a video stream: Information about the location of the images was printed on the video stream, and normally would need to be transcribed manually so that the image could be located geographically. We investigated the potential of artificial intelligence (AI) to develop a faster and more reliable method to get information out of this printed text. A tool was designed to extract hard coded metadata from the video stream and return the corresponding string, using machine learning and computer vision techniques.

B. Development of a machine learning classifier: We investigated the use of AI for algal identification, to develop a tool that could recognise the dominant algal species along the video transect, e.g., *Ecklonia*, *Lessonia* and *Carpophyllum* spp.

C. Testing the effectiveness of the approaches. Tests were performed to see how effective different AI approaches were, and to look at the impact of colour vs monochromatic imagery on the effectiveness of the AI classification system.

D. Development of a map from the data acquired: Design choices were examined for processing the raw data (AI ratings and video) into a form that could be easily used with QGIS, a free and open source Geographic Information System.

Discussion: Underwater videography and analysis, machine learning and artificial intelligence

Underwater videography is an effective method for mapping and ground truthing data acquired by remote sensing, usually at a large scale. Towed underwater video systems provide direct observations of species in their natural habitat, are cost-effective, simple to operate, and provide a valuable, non-destructive method to enable habitats to be monitored. As a mapping tool the main limitation is the spatial interpolation between the *in situ* data points or transects (Dekker et al. 2005) and, depending on the size of the boat, access to very shallow water. However, video imagery can be easily shared and re-analysed if necessary (Sheehan et al. 2016). Image-based surveys produce lasting records of the habitat (Chennu et al. 2017) and are effective tools for integrating with other remotely sensed data. The major advantages are that image processing and interpretation are relatively simple, and there is little or no need for subsequent ground-truthing (Grizzle et al. 2008). Camera systems are usually quite light, and can be operated manually from a small boat or with the use of a winch. The operator is able to see in real time the sea floor surface and adjust the length of the cable accordingly. The cost is quite inexpensive compared to ROV and AUV. Some disadvantages are not having the camera at a fixed depth, making it difficult to compare video frames (e.g., calculate algal percentage cover), and that the cable drift in deeper water can affect georeferencing as the position is related to the boat's GPS (although drift can be calculated and taken into account), and the acquisition of videos being limited in turbid water and requiring calm sea conditions.

In addition, video analysis is very time consuming, even simple tasks such as extracting georeferenced data printed on images is not straightforward, and analyses are not always very accurate (e.g., analysis at fixed intervals vs frames that are randomly selected). A map based on visually interpreted video footage was tested to detect fine-scale habitat changes in Goat Island Bay (CROP Marine Reserve). Video transects were georeferenced using the radio acoustic positioning telemetry (RAPT) system which involved the attachment of an acoustic tag to a scuba diver swimming along the transect and recording the substratum on a digital video camera (Parsons et al. 2004). The use of machine-learning software to automatically analyse video footages is an emerging technique recently used to estimate habitat-forming algae in Australia (Griffin et al. 2017), coral reefs (Beijbom et al. 2015), to predict the presence or absence of *Ecklonia radiata* in Australia (Bewley et al. 2012), and to map the distribution of temperate rocky reef habitats in Tasmania (Seiler et al. 2012).

We investigated machine learning and computer vision techniques to identify the dominant macroalgal species along Wellington south coast. The trial of machine learning approaches produced very strong results and indicated great future potential, with *Ecklonia radiata*, *Lessonia variegata* and *Carpophyllum* spp. correctly identified from video streams with different level of confidence. However, more testing is needed to validate the software, and there is scope for a number of future improvements. Collecting more video footage will enable the training of stronger, more robust classifiers: larger datasets tested in this study increased accuracy significantly. Although this will probably plateau at a maximum level of accuracy, currently there are still large improvements in performance to be made. Creating classifiers is not limited to the algal species classified here, and it is expected that this tool will work on any dataset containing natural photography (photos in the visible light spectrum).

In terms of image files and types, storage of video files as they come from the camera is recommended, as the costs of storing the footage in its highest quality form is trivial compared to the costs associated with recording it. However, it is possible to decrease the size of the video files by a factor of 30 with minimal quality loss to generate the video files with an embedded graph. An approach to semi-automation for effective use of the current AI classifier would be as follows:

- Periodically take a frame from each video in a set videos (for example a frame every 30 seconds).
- Manually label these images as either containing the algae of interest or not.
- Use the labelled data to train the AI classifier.
- The AI classifier would then be used to label all other frames in the video footage.

A practical example of this would be, for example, 10 sampling events, resulting in 10 hours of underwater footage, 900 000 total frames, or 36 000 total seconds. If a labelled subset of frames was created, consisting of a frame every 30 seconds, this would result in about 1000 labelled images. This would provide a supportive skeleton of training examples from which the AI can complete the bulk of the labour of accurately classifying/filling out the remaining video frames.

From the GoPro Only and AUV data tests, the consistent trend was that the more representative the training data were of the classification problem, the better the AI performed. To be entirely representative would require all images being labelled for training which defeats the purpose of automation in the first place. The results of the experiments showed that it was possible to get a very good representation with a fraction of the labelled data if the training data were sparsely spread throughout the target dataset, i.e., if only 2% of data are labelled, it is far more useful if every 50th image is labelled as opposed to a 2% section of consecutive images. While this is not a fully autonomous solution it is expected that this should give the most usable results with the AI at present. The gap between semi and fully autonomous is a blurred line: with improvements in dataset diversity/size and algorithms, the expectation is that more sparse labelled training data will become feasible, while still maintaining high levels of accuracy.

Another trait observed from the Breaker Bay tests was the difference in accuracy from two training datasets of similar size. Using the 30 videos, Moa Point and Palmer Head footage was superior to using one Breaker Bay video when classifying other Breaker Bay videos. This shows that the diversity in training data creates a classifier that is also better at classifying a completely new set of data. If this trend continues, it might reach a point where the training data are diverse and large enough to give accurate results in nearly all new situations, essentially becoming fully autonomous, although more testing would have to be done in this area to confirm this.

There are a few algorithmic improvements that can be added to the AI in order to increase performance. The first would be to replace the linear support vector machine with a multi-layer fully connected neural net. This is not limited to learning strictly linear relationships between features and is powerful at learning more abstract relationships. Most state of the art image classifiers use a convolutional neural network followed by a fully connected neural network. With small datasets this extra learning capability is a double-edged sword as the neural network essentially memorizes the training examples which leads to sub-par results on new data. Another method to increase performance would to be to fine tune the convolutional neural network section. This works by applying a small gradient descent (perhaps $1e^{-4}$ with an Adam optimizer) to the last CNN layers, therefore allowing them to specialise in recognizing algae by adjusting to more relevant visual features. Currently the CNN section is not trained beyond Imagenet which contains a very general set of images. This is a step that can be taken after the fully connected neural network layers are trained. Currently this does not improve accuracy, although it is expected that it would once datasets become larger and more diverse.

This study has shown that there are important factors to consider for effective use of the AI classifier:

- At this stage mixing footage from the splash cam and GoPro does not work (a consequence of issues associated with cameras capturing data differently i.e., interlaced images vs global shutter devices)
- Having training data that represents the total dataset leads to high accuracy.
- A representative training dataset can be created by labelling images evenly distributed throughout the total dataset.
- More diverse training datasets have greater capability at classifying new datasets.
- Based on these results a semi-automated approach will likely be the most useful future approach.
- The BENTHOZ-2015 AUV dataset is a useful tool for benchmarking machine learning methods.
- There are algorithmic changes with potential to increase performance.

Government funded/European Union mapping programmes have produced large scale interactive maps that are available online. The marine data and metadata from diverse sources are managed and assembled in a uniform way by a consortium of organisations within Europe (European Marine Observation and Data Network). Data available include broad-scale predicted habitat maps and detailed habitat maps from surveys, with a range of layers including both environmental and biological data that can be selected (http://www.emodnet-seabedhabitats.eu/default.aspx).

4.4 Mapping of the current known distribution of laminarians and fucoids

Methods: **Mapping known distribution of large brown algae in New Zealand**

Maps of the distribution of laminarians and fucoids in New Zealand were produced based on three types of data: specimen-based data, dive survey records and citizen science observations. Data were limited to the New Zealand mainland, the Manawatāwhi/Three Kings Islands, Chatham Islands and the New Zealand subantarctic islands.

Specimen data consisted of all records of Fucales and Laminariales obtained from the databases of the Museum of New Zealand Te Papa Tongarewa (Te Papa) (2546 collections between 1900 and 2016), Auckland Museum (1548 collections between 1887 and 2016) and the NIWA algae team (452 collections between 2001 and 2017). Specimen data were groomed by the removal of duplicate records, records from the Kermadec Islands and records lacking location data.

Specimens collected from ship hulls were also removed, along with anomalous specimens that were recorded as drift. For the introduced species *Undaria pinnatifida*, additional specimen records were sourced from the database of NIWA's Marine Invasives Taxonomy Service (MITS). At the end of the grooming process, a total of 4330 specimen records were available for mapping.

Survey records were obtained from data provided by Nick Shears (University of Auckland) and Clinton Duffy (Department of Conservation). The Duffy data came from dive records around mainland New Zealand and the Three Kings Islands made between 1980 and 2006. The Shears data was collected during a national Department of Conservation project classifying New Zealand shallow reef communities. In both cases we considered these data reliable as the identifications would be consistent. The Duffy data was collected by a single person experienced in seaweed identifications, whereas the majority of the Shears data was collected by two people experienced in identifications and some identifications were confirmed by Wendy Nelson. After grooming, there were 1651 survey records available for mapping.

Citizen Science records were collected through a project established on the NatureWatchNZ website and app (http://naturewatch.org.nz/projects/large-brown-seaweed-distributions). NatureWatchNZ is a community nature observation system developed by the charitable trust New Zealand Bio-Recording Network Trust. We set up the "Large brown seaweed distributions project" in April 2016 and collected observations until March 2018. An identification guide to large brown algae (Neill & Nelson 2016, https://www.niwa.co.nz/coasts-and-oceans/marine-identification-guides-and-fact-sheets/beautiful-browns) was made available to the public to assist with identifications. The project limited the observations that people could add to 26 species of fucalean and laminarian seaweeds, and all observations were checked by a taxonomist. Species endemic remote off-shore islands were not included (e.g., *Landsburgia ilicifolia*, Manawatāwhi/Three Kings Islands, or *Lessonia brevifolia*, subantarctic islands). Observations included whether the specimen was attached or drift, images of the whole seaweed, and where possible, images of the stipe and reproductive structures. After grooming, there were 437 citizen science records available for mapping.

Mapping was carried out using QGIS (version 2.18.9) and the "nz-coastlines-and-islands-polygons-topo-150k" shapefiles from LINZ (https://www.linz.govt.nz/). A total of 6418 records were mapped.

Results: **Mapping known distribution of large brown algae in New Zealand**

Figure 50 shows the 6418 distribution records by source. Distributional maps by source and species are available in Appendix 10.

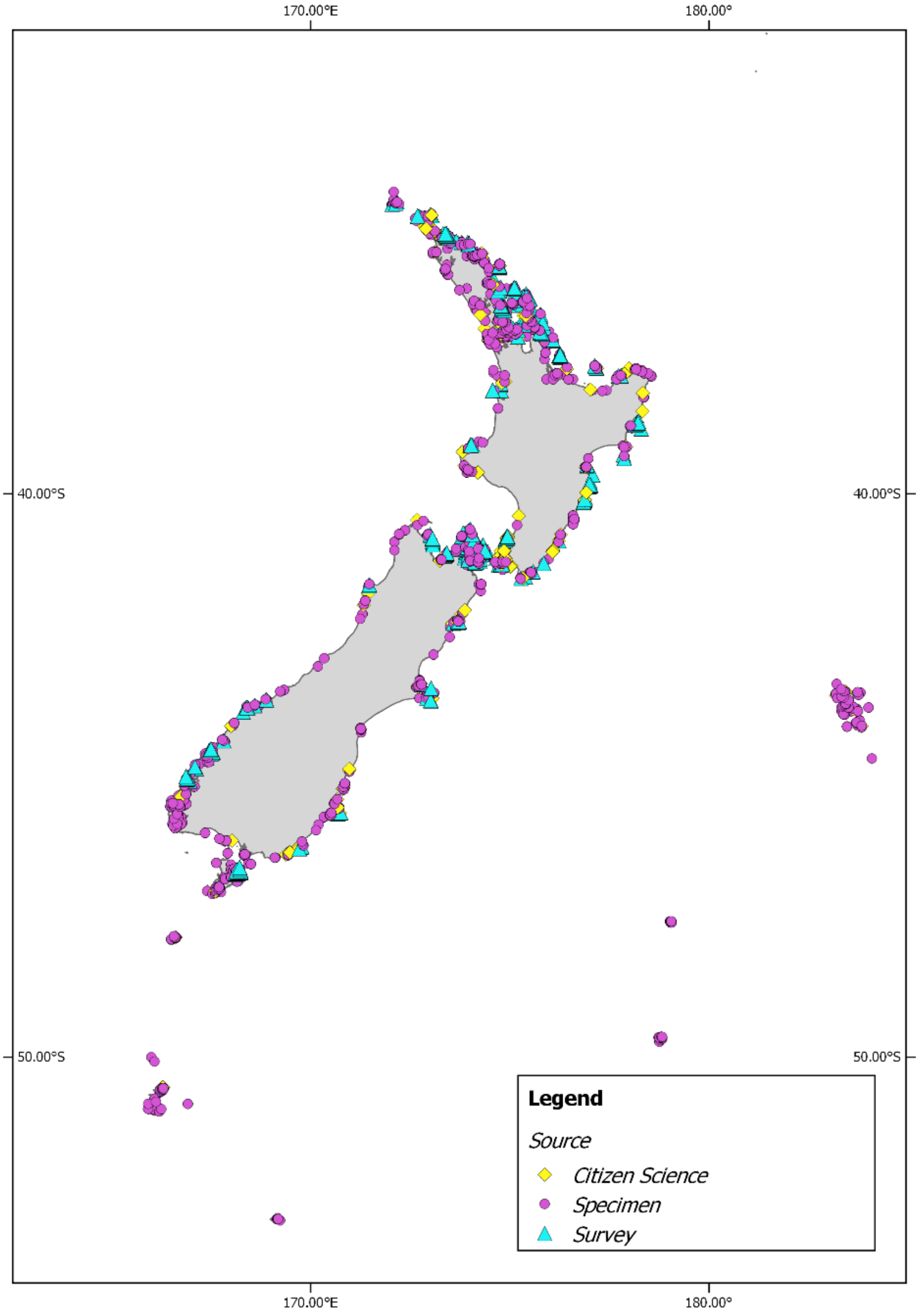


Figure 50. Distribution records of the selected fucoids and laminarians by source: specimen (circles, 4330 records), survey (triangles, 1651 records) and citizen science (diamonds, 437 records).

Discussion: Mapping known distribution of large brown algae in New Zealand

The species maps (see Appendix 10) provide the first national-scale evaluation of the distribution of large brown algae in New Zealand based on location data for verified specimens or specimen observations. Previous treatments have recorded the distribution in terms of regions (e.g., Nelson 1994) but not provided site level data, as presented here. These data enable the interrogation of distributional records in more detail than previously available. When examining the maps there are some interesting issues to consider when attempting to understand the patterns that emerge.

There are some areas of the New Zealand coastline where there are few, if any, records of attached large brown algae e.g., West coast North Island between about southern Taranaki and Kapiti Island, on the South Island east coast between Banks Peninsula and about Timaru and Oamaru. These areas are characterised by long expanses of sand and other habitat that is unsuitable for the establishment of large brown algae. On the West coast of the South Island between about Westport and Jackson Bay there are areas of rocky reef, but apart from *Durvillaea* found in the Buller region (northern west coast to about Rapahoe), and dense beds of brown algae at the Open Bay Islands, the shallow reefs of the west coast are dominated by encrusting fauna including sponges, ascidians and hydroids and have lower diversity and biomass of macroalgae than other regions (Neale & Nelson 1998).

Another challenge when interpreting gaps in distributional records is related to the intensity of collections and representativeness of the records. As summarised by Nelson et al. (2013) there are a range of biases that affect different aspects of natural history/herbarium collections. The position of access roads and settlements, particularly in the case of coastal collecting can lead to spatial biases, as well as access to boats in the large sections of coastline where access is only possible from the sea. Seasonal biases arise for a number of reasons, not least of which is the impact of weather on access to coastlines. The number of collections obtained from particular regions is frequently closely related to the location of active collectors. Inevitably there are biases as far as which species are collected. Although some areas are infrequently visited, their very remoteness means that collections are more assiduously made and preserved and, therefore, the flora of these regions may be more completely represented than other more accessible coastlines. Species that are perceived to be common, frequently are under-collected and so their complete geographic range is not adequately represented in collections e.g., *Hormosira banksii*. Other species remain under-collected because of their size and /or the difficulty of getting access to populations e.g., *Durvillaea* spp. inhabit very wave exposed situations which are sometime difficult to access. Graham et al. (2004) consider that “non-representative sampling in environmental space remains the most difficult source of error to detect and correct”. Importantly, the material in natural history collections/herbaria only provide presence data, establishing that the species was present at that locality when collected, and the interpretation of species absences is complex i.e., the species may not have been at the locality, or was not collected, or not detected.

The maps produced in this study provide a new baseline from which to explore some apparent, or real, distribution gaps in more detail, e.g.,

- *Ecklonia radiata*: although *Ecklonia* is known to occur between Banks Peninsula and Bluff, specimens have not been lodged in herbaria. As pointed out in this report, *Ecklonia* may grow deeper in Otago than other areas and thus has only been seen by divers.
- *Carpophyllum maschalocarpum*: Buchanan (2011) reported that the distribution of *C. maschalocarpum* was restricted to sites from Banks Peninsula northward on the east coast of the South Island, and that the records from Stewart Island were based on drift records. The record from Stewart Island in this report is based on expert diver data. It would be valuable to obtain a confirmatory specimen from this area.
- *Durvillaea antarctica* and *D. poha*: the records of *D. antarctica* are likely to also include records of the newly recognised *D. poha* which was segregated from *D. antarctica.* Although the paper describing *D. poha* included molecular data from a number of populations confirmed to be this species, there were no voucher specimens lodged in New Zealand herbaria, apart from the type material.
- *Durvillaea willana*: the record of this species from the Wairarapa coast (southeastern North Island) was almost certainly drift but this was not recorded in the specimen record. Despite dedicated searches, no attached populations have been found in this region.
- *Landsburgia quercifolia*: there are no records from any source (herbarium data, expert field observations, citizen science) between Kaikoura and SE Otago.

The inclusion of citizen science contributions was valuable, and the observation records submitted to NatureWatch extended the known/recorded distributions or filled in distributional gaps for a number of species (e.g., *Cystophora platylobium* - Kaikoura and Dunedin, *Cystophora retroflexa* - southern Hawkes Bay, *Cystophora scalaris* - East Cape, *Durvillaea poha* - Kaikoura and Stewart Island, *Hormosira banksii* - Wairarapa and north Otago, *Macrocystis pyrifera* - SE Otago).

Internationally, citizen projects are making important contributions assisting scientists to collect data in the field and occasionally participating in data analysis. Benefits of including citizens in research programmes include "enhancing monitoring capabilities, empowering citizens and increasing Ocean Literacy, which can itself lead to the development of environmentally-friendly behaviour" (Garcia-Soto et al. 2017). Garcia-Soto et al. (2017) summarised about 50 citizen programmes in Europe by country, including a few worldwide programmes e.g., Reef Life Survey (fish), and recommended eight Strategic Action Areas for Marine Citizen Science in Europe.

Thiel et al. (2014) carried out an extensive review of marine science studies that have been realised with the involvement of citizens. Of the 227 publications analysed, 84.6% were focused on fauna, 12.8% on flora (20 studies) and 14.1% on contamination. The activities of citizen scientists were concentrated in easily accessible coastal habitats e.g., beaches, estuaries, coral reefs, and seagrass beds, however, in 28.6% of the studies the observations were made subtidally by scuba diving. Some examples include Reef Check California which involves volunteer scuba divers in monitoring *Macrocystis* beds (Freiwald & Wisniewski 2015). The project FindKelp aimed to assess the distribution and the conservation status of six species of kelp along the coast of Portugal (Assis et al. 2009). In Japan, volunteer scuba divers participated in a kelp restoration project removing sea urchins. As GoPros become more popular, a method for habitat analysis has been developed (Raoult et al. 2016). The majority of marine citizen science projects were in North America and Europe, and also in the Indo-West Pacific region (Thiel et al. 2014). In New Zealand, due to strict health and safety regulations it was impossible to involve volunteer divers in our research programme, and it is becoming difficult to train students in scientific diving. Smale et al. (2013) discusses increasing constraints with subtidal sampling - "occupational health and safety culture and ever-growing institutional fear of litigation made scientific diving for ecological research almost unfeasible. Engaging in rational evidence-based discussion relating to actual (rather than perceived) risks associated with subtidal field work, and re-assessing health and safety and legal requirements accordingly, would allow more marine ecologists to get "wet" and facilitate real- world observations of coastal marine ecosystems".

There are opportunities for science contributions from members of underwater and natural history associations. The Wellington Underwater Club, founded in 1951, has members with a broad range of interests, skills and experience, and the club is very well networked in the local marine and conservation community, engaging with other clubs and organisations in Wellington region as well as the National Underwater Association. The Club has Marine Citizen Science projects and outreach activities showcasing the importance of diverse and healthy marine ecosystems and including establishing a monitoring scheme for marine reserve management. Club members are using 3D photogrammetry to document maritime artefacts, and drone and GPS surveys to map seaweed habitats. More information on www.wuc.org.nz. In South Taranaki, a project engaging the community in observing coastal ecosystems was supported by the National Science Challenge Curious Minds programme https://www.curiousminds.nz/projects/south-taranaki-reef-life/ and has since been extended as Project Reef Life with a partnership between local Iwi – Te Kaahui o Rauru and Te Runanga o Ngati Ruanui Trust, as well as the Hawera High School and Patea Area School, and the South Taranaki underwater club underwater clubs in Taranaki https://www.projectreeflife.org/about.

4.5 Discussion: approaches to mapping of macroalgae and recommendations for application to monitoring programmes in New Zealand

Mapping macroalgal beds at a national scale will require a reliable and cost-effective method that deals with the different coastal environments and ocean conditions. Macroalgae predominantly live in coastal shallow water on reefs that are not easily accessible to vessels and boats. Soft sediment habitats also have large algal meadows including rhodoliths and seagrass beds. The New Zealand coastline is about 15 000 km long and includes different habitats and the water clarity typical of specific regions varies greatly. Temperate coastal waters are often turbid for different reasons e.g., tidal currents and waves, sources of sedimentation and pollution, or simply because they are rich in plankton.

The examination of the literature suggests that mapping macroalgal beds can be carried out in a number of ways depending on the scale of interest, by direct observations (e.g., videography, drop camera), or by indirect methods using remote sensing techniques (optical and acoustic). A critical review of 195 studies of the optical and acoustic remote-sensing techniques used to map seagrass beds (Hossain et al. 2015) showed that multiple approaches are required as there is no single technique that can acquire all the required data to map seagrass distribution. This is also true for macroalgae, as evident from our review of mapping techniques (See Table in Appendix 2). The methods applied to date have advantages and limitations, and the mapping methods need to be tailored to specific habitats and environments, considering practicability, efficiency and, not least, cost. The size of the area and the depth range to be mapped also need to be considered. Intertidal macroalgae, subtidal beds, and species with surface floating canopies such as *Macrocystis* all require different approaches. The importance of ground truthing and applying a combination of approaches are recognised in the literatur as key components to monitoring programmes, with the approaches selected dependent on the size of the target area, the nature of the habitats being studied, and the species present.

Optical remote sensing has the advantage of covering large areas at relatively low cost and enables access to remote areas. It has developed hugely in the last decades and different sensors are currently available. However, penetration of the water column is still very shallow, especially in turbid water. Increasing sedimentation with the resultant turbid water has been recognised as a major threat to the coastal environment. With less light penetrating the water column, the maximum depth for growth of macroalgae is expected to decrease (e.g., Desmond 2016), thus monitoring the lower depth of macroalgae in coastal areas will give a measurement of change. At present, relatively few studies have tested mapping subtidal macroalgae forests in turbid water (e.g., Kutser et al. 2006, Vahtmäe et al. 2006, Casal et al. 2011b) and these usually required a multi-approach including ground-truthing with biological samples or images.

Satellite and airborne imagery have been successfully used in temperate water to map kelp beds, particularly species with floating canopies (e.g., *Macrocystis* and *Nereocystis*), and in clear water to map coral reefs, seagrass beds, and intertidal seaweed beds. Satellite imagery has been used extensively for mapping *Macrocystis* beds along the coast of California since 1980s, and with the long-time series of imagery, it has been possible to analyse fluctuations of *Macrocystis* over time. In New Zealand the decline of *Macrocystis* is still mainly anecdotal. An analysis of historical imagery would provide evidence whether a decline has already occurred, and to what extent. However, earlier satellite imagery, e.g., Landsat and SPOT, have a very coarse resolution making it very challenging to recognise *Macrocystis* in New Zealand, as beds are not very large and often close to the coastline. Commercial satellites offer imagery with high spatial resolution (less than 4 m/pixel) which might be sufficient but can be prohibitively costly (e.g., QuickBird, IKONOS) (Ventura et al. 2016). With the development of new satellites and sensors, and also with the constant improvement of computer processing power and data storage capacity, this approach will become more accurate and feasible to apply. Satellite data would need to be combined with field surveys for ground truthing, preferably as close in time to the date of the image acquisition.

Important issues to consider in the coastal zone include wave action, the tide level, and currents that can significantly affect the apparent bed size. In extreme instances, kelp beds virtually disappear from the surface at current speeds exceeding 100 cm/s (Britton-Simmons et al. 2008). Ideally, satellite imagery acquisition should be acquired at low tides, on a fine day (cloud free and with a calm sea surface). In analysing a time series of satellite images, the risk of comparing images at different resolutions and conditions (e.g., tides) should be considered. It is also very difficult to find imagery suitable for analysis covering all the desired aspects.

The Department of Conservation tested the use of satellite imagery (WorldView-2; image resolution pixel size 50 cm in panchromatic and 2 m in multispectral) in the Marlborough Sounds comparing two years 2010 and 2013, showing that it would be possible to follow the evolution through time of the *Macrocystis* canopy (Meng et al. 2015) with high spatial resolution imagery. Although we couldn't test hyperspectral imagery during this study, Bell et al. (2015) observed "the application of hyperspectral imagery to detect changes in the physiological condition of giant kelp is the next logical step in understanding submerged and emergent vegetation population dynamics from satellite observations". Satellite imagery collects valuable data over time that can be used to track changes as in case of catastrophic events. For example, satellite images have been used to estimate the loss of seagrass and seaweeds beds caused by the huge tsunami of the Great East Japan Earthquake in March 2011. Imagery before the tsunami in 2009 showed a coverage 320 ha of seagrass and seaweed beds in Matsushima Bay that was estimated to be only 100 ha after the tsunami. *Zostera marina* beds on sandy and muddy bottoms were severely damaged compared with brown seaweed beds of *Sargassum horneri* on the rocky substrate (Tsujimoto et al. 2016).

Shears (2017) outlined modifications to sampling programmes that will result in more robust data, and enable evaluation of change over time including focusing on measurement of canopies, improvements in taxonomic resolution and accuracy of sampling benthic assemblages, increased use of associated environmental monitoring devices (e.g., light and temperature loggers), and placing monitoring sites along gradients of stressors/human impacts.

5 GENERAL DISCUSSION

The effects of human induced environmental changes on coastal environments at local and global scales are increasing. Pressures of growing populations, urbanisation, land use change, management of waste and pollutants, in addition to the complex suite of impacts of changing climate on temperature, storm intensity, changes in rainfall, and ocean acidification are recognised as contributing to negative outcomes for the coastal environment. In New Zealand, the Ministry for the Environment State of the Environment Report (2016) identified needs for the marine sector, including the need for more data around the ecology and health of coastal habitats and ecosystems, and for "population data across more marine plant and animal groups".

The effects of local change on large brown algae have been recognised in many parts of the world, with research investigating the impacts of particular stressors, individually or in combination, that may be negatively affecting macroalgal growth. Particularly in warm to cool temperate nearshore environments, large canopy forming macroalgae provide a number of critical ecosystem services, with the provision of three-dimensional habitat structure and productivity, the two most commonly identified of these. Because of their size and ecological importance, canopy-forming large brown macroalgae have been identified as indicators of change in a number of programmes worldwide. International attention has been drawn to examples of large scale and climate driven change, such as the retreat of kelp forests in Australia linked to warming of coastal waters and physiological thresholds being exceeded.

Changes to marine species and ecosystems may be the result of pressures that have built gradually or acute events (either in severity or in length), episodic or long-term stress or disturbance, on scales of hours, or days or even intra-annual, inter-annual, or decadal scales. The changes may be highly localised or widespread, caused by a single factor or multifactorial, and there may also be cumulative effects. In many cases, responses will be species or context specific, and also mediated by community attributes which influence the thresholds at which impacts are realised or expressed.

The value of baseline data to reference environmental changes (e.g., latitudinal changes in species ranges, declines in species abundance, alterations in community composition, changes in productivity of coastal ecosystems) is universally recognised in the literature. Baseline data are needed to benchmark the state and condition of the system being managed, for trends to be identified, and distinguished from the inherent variability and dynamics of the system, and provide insights, for example, the correlation of range shifts of species with climate change and other non-localised anthropogenic factors. Unfortunately, no systematic approach to acquisition of baseline data has been developed in New Zealand, and thus the national capacity to employ macroalgae to address a range of questions about environmental change is limited.

Some baseline data on macroalgae are held in herbaria. Unfortunately, the retention of voucher specimens is infrequently part of New Zealand monitoring studies, and thus the data collected cannot be re-examined in the light of new taxonomic understandings (e.g. additional species of the bull kelp *Durvillaea* being identified in the past decade). In addition, the challenges of identifying species – macroalgae and invertebrates – in the field are significant, and it is unrealistic to expect high levels of taxonomic resolution in field data. For this reason, specimens should be collected for confirmation of identification, and vouchers deposited in national herbaria or collections. There needs to be an evidence base for monitoring and establishing long term changes in the biota. The mapping exercise in this report reveals that for even large, conspicuous brown algae the collection of representative specimens around the country does not accurately reflect species distributions.

The distributions of other components of the flora and fauna are even more inadequately and unevenly represented in national natural history collections (e.g., herbaria, museum, NIWA Invertebrate Collection) and this has significant consequences. Following a human-induced or natural event that causes significant changes to coastal ecosystems, understanding the recovery processes and responses of different components within the community is important, in order to learn better approaches to management of coastal resources and to minimise negative impacts in future events where possible. If there are no reference or voucher specimens reflecting the original biota, it is difficult, if not impossible, to fully understand the recovery. In Israel (Krupnik et al. 2018) a comparative study of *Ulva* populations along the coast revealed an almost complete shift in species composition from 2002 to 2014/2016, with the species present in 2016 including a number of species previously not recorded and known to originate in the Indo-Pacific. In New Zealand we have an excellent baseline dataset of species of *Ulva* which has the potential to be used in exploring responses to change (Heesch et al. 2009). Not all species are equal in their contributions to ecosystem services and functions: even species within the same genus that may share similarities in morphology or position of the shore, do not necessarily contribute in the same way within an ecosystem. While it is not possible to retain reference specimens for all aspects of all field studies, having a consistent approach to retention of reference material is important in fully understanding ecosystem processes.

Monitoring macroalgae appears to have greatest utility for 1) identifying changes associated with specific stressors (such as sewage effluent, eutrophication, sedimentation, fishing practices) and 2) measuring the effectiveness of management regimes, expressed in terms of recovery of populations or ecosystem function.

The development of approaches for monitoring macroalgae and the use of macroalgae as indicator species has received considerable attention internationally, and many examples of indices and metrics used in international programmes for monitoring habitat-forming macroalgae have been summarised in this report. In Europe there has been significant work in the development of indices based exclusively on macroalgae, as well as those that consider the whole habitat. The application of these indices e.g., CARLIT, EEI, in the Mediterranean has become part of regular monitoring, and the indices have been used to categorise areas affected by different degrees of pollution or anthropogenic impacts. The disappearance of Fucales/Laminariales and the predominance of turfing algae are frequently used to characterise areas affected by significant environmental impacts.

In order to track trends in New Zealand's coastal marine ecosystems, there is a need for monitoring methods that can be adapted across regions, as the dynamics and drivers of change will vary considerably in different parts of New Zealand (i.e. regional differences in key species and in specific local pressures such as temperature, sedimentation, invasive species). A combination of techniques is necessary for monitoring or mapping, depending on the size of the target area and the nature of the habitats being studied, and the species present i.e., *Macrocystis* with surface floating canopies, intertidal macroalgae, and subtidal beds, all require different approaches. Despite the well-established literature about the productivity, biodiversity and habitat provision that accompanies intact macroalgal canopy structure, indices based on macroalgae which characterise the health of coastal habitats, have not been developed to date in New Zealand. However, such efforts have the potential to contribute to resource management and protection (e.g., the recognition of *Macrocystis* as an endangered species in Tasmania, and subsequent protection, based on data on the range retraction over the past few decades; restoration of kelp in Sydney harbour attempting to return habitats to previous forested state based on baseline information).

This project has provided the opportunity to evaluate a wide range of activities that are currently underway, including examples from New Zealand (e.g., changes in vegetation cover at Leigh – Cape Rodney to Okakari Point Marine Reserve; long-term dataset of major habitat-forming species from the intertidal research programme of MERG at the University of Canterbury; reef community monitoring in the Auckland region supported by the Auckland Council; iwi monitoring of habitats - East Otago Taiāpure). There is a need for a standardised approach to documentation of species distribution data and a national repository of data in agreed forms with associated levels of verification and confidence. Most data that has been gathered in New Zealand in macroalgal monitoring programmes is stored by individual researchers and is not more widely accessible.

In this report a range of approaches have been explored to evaluate how large brown macroalgae can be used in the New Zealand context as either indicator species or as management tools for evaluating ecosystem health. This report has included examining the use of 1) early life stages to evaluate susceptibility of target species, 2) underwater videography with machine learning approaches to data analysis, 3) drones to assess extent of beds and intertidal assemblages, 4) engaging citizen scientists in coastal monitoring. The trial of machine learning approaches produced very strong results and indicated great future potential, with *Ecklonia radiata, Lessonia variegata* and *Carpophyllum* spp., correctly identified from videos streams with different levels of confidence.

There are several requirements for an effective monitoring programme:

- Identification of the spatial and temporal scales that will result in appropriate data in the formats and with the resolution needed for the questions being asked. There is a need to consider differences at geographic scale, intertidal and subtidal, exposed and sheltered shores also in response to stressors.
- Standardised methodologies applied within a nested design to enable the comparison of datasets (Kingsford & Battershill 1998, Van Rein et al. 2009)
- Importance of using multiple approaches (e.g., remote sensing, underwater imaging, and the application of AI analytical methods) and ensuring validation of data collected, not

only with respect to the canopy but also to make sure that the ecosystem properties that are of greatest interest to maintain are in fact being effectively monitored (e.g., diversity, functional relationships). This requires attention not only to the canopy-forming brown algae and most conspicuous fauna, but also developing a structured approach to recording and retaining representative biodiversity samples.

New technologies have very significantly increased the options for more accessible and cost-effective approaches to monitoring macroalgae. The confluence of new methods and concepts, the ready availability of vast new data streams, powerful computing, innovative approaches to data management and integration, as well as machine learning, will substantially enhance capacity from data acquisition through to analysis and interpretation.

6 NON-TECHNICAL SYNTHESIS

Human-induced modification of the coastal zone is clearly evident in New Zealand at a range of scales. There is need for evidence to evaluate the impact and extent of changes in order to develop appropriate interventions and management initiatives.

Large brown canopy-forming algal species are vital for the ecosystem functions in coastal rocky shore environments globally, contributing to nearshore productivity, providing and influencing three-dimensional habitat space and shelter and enhancing biodiversity, elevated secondary production, nutrient cycling, energy capture and flow, contributions to carbon sequestration, coastal defences, as well as ameliorating impacts of environmental stressors. To retain healthy coastal rocky shore ecosystems there is a need to protect and retain intact canopies and the associated communities. Loss of biodiversity has been shown to have impacts on ecosystem services and functions as significant as those from climate change.

Currently there are important knowledge gaps that constrain sound decision making – preventing decision makers from anticipating, and where possible mitigating, changes in the composition of seaweed communities, and accompanying ecosystem-level effects. Baseline data are needed to benchmark the state and condition of the system being managed, for trends to be identified, and distinguished from the inherent variability and dynamics of a system. This lack of sound baseline information also constrains the ability to evaluate the effectiveness of management methods. While New Zealand is not alone in having inadequate baseline data, this lack of critical information impedes the detection of range shifts in response to local or global stressors, as well as the occurrence of local extinctions and the introduction of non-native species.

New technologies have very significantly increased the options for more accessible and cost-effective approaches to monitoring macroalgae. The confluence of new methods and concepts, the ready availability of vast new data streams, powerful computing, innovative approaches to data management and integration, as well as machine learning, will substantially enhance capacity from data acquisition through to analysis and interpretation. This does however, require investment across a number of areas, particularly in interoperable databases for specimen (and species) information, including images, and registration of specimens housed in natural history collections.

7 ACKNOWLEDGMENTS

We acknowledge with thanks the funding from the Ministry for Primary Industries (project ZBD201406) for enabling this research. The research has also built on activities in a number of related programmes and projects within NIWA, as well as programmes conducted by colleagues externally. We thank the Department of Conservation for funding to enable the summer research project on machine learning and artificial intelligence. The case study presented here from research in the East Otago Taiāpure was supported by the Te Tiaki Mahinga Kai partnership (http://www.mahingakai.org.nz), and the dataset was provided by the laboratory group of Assoc. Prof. Chris Hepburn (University of Otago) and was derived specifically from the master's projects of Derek Richards and Eugene Leahy. Advice on the analyses was provided by Dr Daniel Pritchard (Te Rūnanga o Ngāi Tahu). The research was made possible by all other members of the data collection team: Robert Win, Chris Cornwall, Emma Kearney and Niall Pearson. We also thank the East Otago Taiāpure Committee for facilitating ongoing monitoring efforts and application of the results.

This project has involved collaboration and consultation with a wide range of people, and we have greatly appreciated the generosity that has been shown by colleagues sharing data, experience and insights. In particular we thank NIWA colleagues: Pete De Joux (development of camera system and supervision of machine learning project), Peter Notman (field work and *Macrocystis* mapping), Leigh Tait and Dave Allen (drone tests), Neill Barr (assistance with culture work), and also Tara Anderson, Jochen Bind, Caroline Chin, Mark Gall, Warrick Lyon, Peter Marriot, Sadie Mills, David Plew, Graeme Moss, Lisa Northcote, Matt Pinkerton, Christopher Ray, Alexandre Schimel, Tilmann Steinmetz, Rob Stewart, Brenton Twist, Steve Wilcox, and Brent Woods. Alison MacDiarmid is thanked for reviewing this report. We also thank Clinton Duffy (Department of Conservation) and Nick Shears (University of Auckland) for providing their survey data, and Chris Hepburn and Daniel Pritchard for discussions and advice.

Thanks also to international colleagues Luigi Piazzi, Francesco Bignami and Vittorio Brando.
We acknowledge with thanks all the respondents to the questionnaire survey: Auckland Council, Bay of Plenty Regional Council, Hawke's Bay Regional Council, Gisborne District Council, Horizons Regional Council, Tasman District Council, Environment Canterbury; and, Tara Anderson, John Booth, Stephen Brown, Rob Davidson, Matt Desmond, Robyn Dunmore, Sean Handley, Richard Kinsey, the Marine Ecology Research Group, Mark Morrison, Don Morrisey, Don Neale, Kirsten Rodgers, Nick Shears, Te Rūnanga o Ngāi Tahu, Mads Thomsen, Leigh Tait, Mike Wilcox.

Nicole Miller (Wellington Underwater Club President, Friends of Taputeranga Marine Reserve Trustee and Project Baseline Project Manager) is thanked for her assistance, as well as contributions with drone testing. We thank all the participants and contributors to the NatureWatch project (see members list here http://naturewatch.org.nz/projects/large-brown-seaweed-distributions), but particularly those people with the user names 'tangatawhenua', 'rsiddharth97', 'nicolemiller' and 'projectbaselinewellington', who between them added 223 observations. We acknowledge Port Otago Ltd for the aerial photo of *Macrocystis pyrifera* and LUCAS team (Ministry for the Environment) for the satellite imagery SPOTmaps2014.
Karen Tunley and Marianne Vignaux (MPI) are thanked for some very good suggestions and editorial changes.

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9 APPENDIX 1. Survey questionnaires

Questionnaires were sent to staff in local government, specifically members of the Local Government Coastal Special Interest Group (C-SIG) involved in environmental monitoring, and, to expert scientists and science teams involved in ecological research with macroalgae.

Local Government:

1) Do you, or have you, used macroalgae in surveys of coastal or estuarine health, or for assembling local baseline data?

A. If yes, what has been recorded and how frequently?

- Species presence/absence
- Quantity of macroalgae - as a whole or by species
    - Biomass
    - % cover
- Other?

B. If you are monitoring macroalgae, why has this been given a priority for your region?

C. If you no longer record data on macroalgae, what are the primary reasons?

2) Are issues about changes in the coastal macroalgae relevant to your region? – loss of kelp beds, changing position or density of macroalgae on the shoreline?

If so, are there any plans to address this or gather documentation or baseline data?

3) Have there been outbreaks of nuisance quantities of macroalgae in your region?

A. If so, do you monitor these (seasonally, regionally, when reported)

B. Is there a particular season when these are more likely to occur?

Researchers working on Macroalgae

1. Do you, or have you, used macroalgae in surveys of coastal or estuarine health, for assembling baseline data, or as the focus of your research projects?

A. If yes - comments (particularly on frequency):

B. If you are monitoring macroalgae, why have you given this work a priority?

C. If you no longer record data on macroalgae, what are the primary reasons?

D. How are your macroalgal data stored? (e.g., spreadsheet, database, etc)

2. Are issues about changes in coastal macroalgae relevant to your region or a focus of your research? E.g., loss of kelp beds, changing position or density of macroalgae on the shoreline?

If yes, are there any plans to address this, document change or gather baseline data?

3. Have there been any outbreaks of nuisance quantities of macroalgae in your region?

If yes, do you monitor these? (seasonally, regionally, when reported?)

4. Do you use, or have you used, macroalgal data for any reporting purposes?

5. Do you see a need for future monitoring of macroalgae in your area/nationally?

10 APPENDIX 2. Summary of relevant literature on mapping of macroalgae

Authors	Year	Country/ region	Purpose	Optical (satellite, aircraft, UAVs)	Acoustic	Ground truth/ Direct observations	Species/ habitat type
Amorim et al.	2015	Azores	Infralittoral mapping, deep kelp observations	MERIS FR		ROV -Video, drop-down camera	*L. ochroleuca* -four semi-quantitative abundance categories; forest, dense, sparse and absent
Anderson et al.	2007	South Africa	Kelp distribution, biomass	Landsat 5 TM/Landsat 7 ETM; airborne Infrared and multispectral imagery		Boat, kayak, GPS, SCUBA, Quadrats	*E. maxima, L. pallida*
Andréfouët et al.	2004	Tahiti	Mapping, biomass estimation of invasive brown algae	IKONOS		Field data, algal cover	*Turbinaria ornata, Sargassum mangarevense*
Andréfouët et al.	2017	Tuvalu	Algal bloom, biomass estimation	Multispectral Pleiades imagery / series of imagery (2002-2015)		Field surveys	*Sargassum polycystum*
Augenstein et al.	1991	California	Kelp distribution, density	SPOT HRV-XS		Spectral reflectance of kelp acquired in the field	*Macrocystis pyrifera*
Bajjouk et al.	2015	France	Kelp biomass map	MERIS; airborne LiDAR Bathymetry, multispectral imagery	MBES/ Inter-ferometric sonar	Video/ SCUBA quadrats	*L. digitata, L. hyperborea*
Barbera et al.	2012	Spain	seagrass, rhodoliths beds, habitat mapping		SSS	ROV video, underwater drop camera, benthic grabs, beam trawl	*Posidonia oceanica*
Bekkby & Moy	2011	Norway	Predictive sugar kelp distribution maps			Data from monitoring sites	*Saccharina latissima*
Bell et al.	2015	California	Monitoring biomass, physiological condition	Landsat (28 years time series); airborne Hyperspectral (HyspIRI)		Field observations	*Macrocystis pyrifera*

Authors	**Year**	**Country/ region**	**Purpose**	**Optical (satellite, aircraft, UAVs)**	**Acoustic**	**Ground truth/ Direct observations**	**Species/ habitat type**
Belsher & Mouchot	1992	Kerguelen Islands	Biomass, correlation biomass Macrocystis and sea-surface temperature	SPOT-1 /NOAA-11 AVHRR		Field observations	*Macrocystis pyrifera*
Betzabeth & de los Angeles	2017	Yucatan (Mexico)	Habitat mapping	Landsat ETM +		294 sampling sites/ videos	brown algae and seagrass
Bushing	2000	USA	Monitoring the persistence of giant kelp	SPOT NIR, aerial photography, GIS		Field surveys, temporal data	*Macrocystis pyrifera*
Cavanaugh et al.	2010	USA California	Estimating giant kelp canopy cover and biomass from satellite imagery, assess changes	SPOT 5 multispectral (10m res); aerial photos		Diving transects	*Macrocystis pyrifera*
Cavanaugh et al.	2011	USA California	Dynamics of giant kelp at spatial and temporal (up to 25 ys) scale	Landsat 5 Thematic Mapper			*Macrocystis pyrifera*
Cavanaugh et al.	2013	USA California	Environmental controls of giant-kelp biomass	Landsat 5 Thematic Mapper; Monthly mean SST MODIS Terra, Aqua		5 years of data from diver surveys	*Macrocystis pyrifera*
Casal et al.	2011a	Spain	Mapping benthic macroalgae	CHRIS-PROBA		SCUBA, 100m transects/Reflectance spectra of macroalgae	macroalgal beds
Casal et al.	2011b	Spain	Mapping kelp in turbid water	SPOT-4		SCUBA, 100m transects/ recreational divers and commercial fishermen observations	Kelp beds
Casal et al.	2012	Spain	Map macroalgal communities assessing hyperspec	Airborne Hyperspectral Scanner (AHS)		Field data, radiometric measurements, macroalgae abundance subtidal, intertidal	green, red, brown algae

Authors	Year	Country/ region	Purpose	Optical (satellite, aircraft, UAVs)	Acoustic	Ground truth/ Direct observations	Species/ habitat type
			tral scanner				
Casal et al.	2013	Spain	Mapping shallow benthic habitat	Airborne CASI-2 Hyperspectral		Reflectance spectra of 17 species	*Codium tomentosum, Laminaria saccharina, Corallina officinalis*
Casas-Valdez et al.	2016	Mexico	Temporal changes in the biomass and distribution of Sargassum beds	Temperature from satellite data 1988-2014		Field collections, quadrats, observation, historical data of distribution; GPS	*Sargassum* spp.
Che Hasan et al.	2014	Australia	Habitat mapping		Narrow MBES	Towed video observations	Mixed brown, red, green algae
Chennu et al.	2017	Germany	Automated benthic habitats			SCUBA + Hyperspectral sensor	Shallow coral reef
Chust et al.	2008	Spain	Habitat mapping	Airborne Lidar/ multispectral		Biological samples/transects	Intertidal communities
da Silva et al.	2017	Brazil	Seaweed and seagrass spatial distribution and abundance maps	ALOS AVNIR-2		Biological samples/field observations	seaweed, seagrass
Deysher	1993	USA-California	Changes in kelp bed size and location	SPOT; airborne ADAR, multispectral video sensor		Photographic survey, in situ reflectance measurement	*Macrocystis pyrifera*
Funnel et al.	2005	**New Zealand**	Habitat mapping Tuingara Point to Blackhead Point		SSS	Video, drop camera	Mixed algae, *Ecklonia, Carpophyllum* spp. Invertebrates, sponge flat
Fyfe et al.	1999	**New Zealand**	Mapping marine habitat	Aerial photography (colour negative film - UV filter)		Field surveys, Ground control points	*Macrocystis pyrifera, Zostera capricorni*

Authors	**Year**	**Country/ region**	**Purpose**	**Optical (satellite, aircraft, UAVs)**	**Acoustic**	**Ground truth/ Direct observations**	**Species/ habitat type**
Gameiro et al.	2014	Spain	Mapping of algal communities	Airborne LIF LIDAR			red and green/brown algae/ microphyto-benthos
Giardino et al.	2015	Italy	Suspended particulate matter/aquatic vegetation lake	Airborne Hyperspectral		In situ measurements	submersed macrophyte
Grace & Kerr	2005	**New Zealand**	Intertidal and subtidal habitats mapping - Doubtless Bay, Northland	Aerial photography	SSS, MBES	Drop video, scuba and snorkel dives, manta video, sediment sampling	shallow mixed macroalgae, kina barrens, *Ecklonia*, tangle-seaweed forest, deep reefs
Guinda et al.	2012	Spain	Subtidal macroalgae communities, distribution patterns, basic distibution maps			Scuba transects, total coverage dominant algal species	subtidal macroalgae
Hoang et al.	2016	Australia (WA)	Mapping *Sargassum* beds	WorldView-2		Photoquadrat snorkel surveys/ biological samples	*Sargassum* spp.
Holmes et al.	2007	Australia (Tasmania)	Probabilistic species mapping	Aerial photography		Video towed camera	*Amphibolis, Posidonia*
Holmes et al.	2008	Australia (Tasmania)	Modelling distribution of marine benthos		MBES	Video towed camera	macroalgae mixed red, green, brown algae, *Ecklonia*, rhodoliths
Hu et al.	2017	China Yellow Sea	Estimation biomass Ulva green tides	MODIS		In situ and water tank experiments	*Ulva prolifera* -green tides
Ierodiaconou et al.	2007	Australia	Marine benthic habitat mapping		MBES	Video towed camera	Mixed brown, green, red algae, seagrass, invertebrate

Authors	**Year**	**Country/ region**	**Purpose**	**Optical (satellite, aircraft, UAVs)**	**Acoustic**	**Ground truth/ Direct observations**	**Species/ habitat type**
Israel & Fyfe	1996	**New Zealand**	Monitoring intertidal and sublittoral vegetation of Otago Harbour	SPOT XS		Field surveys	*Zostera capricorni*, red algae
James et al.	2015	Global	Mapping global distribution and phenology of *Undaria pinnatifida*	satellite-derived SST		Worldwide distribution literature search	*Undaria pinnatifida*
Jensen et al.	1980	California	Kelp surveys	Landsat /Seasat-A; airborne Color infrared photography, X-band radar		Data compared with monitoring surveys	*Macrocystis pyrifera*
Johnson et al.	2011	Australia	Historical distribution of *Macrocystis pyrifera*	Aerial photography (1946 -2007)		Time series aerial photos compared	*Macrocystis pyrifera*
Jordan et al.	2005	Australia	Habitat mapping		SBES	Video ground-truthed	*Phyllospora comosa, Ecklonia radiata,* seagrass
Kerr	2010	**New Zealand**	Habitat mapping from Mangawhai to Ahipara	Aerial photography	SSS, MBES	Scuba, video	rhodolith beds, seagrass
Kerr	2016	**New Zealand**	Maunganui Marine Reserve	Aerial photography	SSS	Drop video, scuba diving, snorkelling, manta board video, sediment sampling	shallow mixed seaweed, *Ecklonia,* kelp forest
Kerr & Grace	2005	**New Zealand**	Intertidal and subtidal habitats mapping - Mimiwhangata Marine Park and adjacent shelf	Aerial photography	SSS	ROV, Drop video, dredge sampling	shallow mixed seaweed, kina barrens, *Ecklonia,* tangle-seaweed forest, deep reefs

Authors	Year	Country/ region	Purpose	Optical (satellite, aircraft, UAVs)	Acoustic	Ground truth/ Direct observations	Species/ habitat type
Kerr & Grace	2005	**New Zealand**	Intertidal and subtidal habitat Doubtless Bay	Aerial photography	SSS, MBES	Drop video, scuba diving, snorkelling, manta board video, sediment sampling	shallow mixed seaweed, kina barrens, *Ecklonia*
Kerr & Grace	2006	**New Zealand**	Subtidal and intertidal habitat mapping - Motukaroro Island	Aerial photography	SSS	Drop video, scuba diving, snorkelling, knoledge of similar habitats	shallow mixed macroalgae, kina barrens, *Ecklonia*, tangle-seaweed forest, coralline turfs
Kerr & Grace	2013	**New Zealand**	Subtidal and intertidal habitats - Hauraki Gulf, Waiheke Is.	Aerial photography	SSS, SBES	Video and still photos by scuba, by remote suspended camera	shallow mixed macroalgae, kina barrens, tangle-seaweed forest
Kerr & Grace	2015	**New Zealand**	Waewaetorea Marine Reserve		SSS	Drop video, scuba diving, snorkelling, manta board video, sediment sampling	*Ecklonia* forest, mixed macroalgae, algal turf beds, seagrass
Kibele & Shears	2016	**New Zealand**	Bathymetric Mapping in Coastal Waters	WorldView-2/WorldView-3	MBES	Multibeam sonar data used to train and validate the model	*Ecklonia radiata*, kelp beds
Kruss et al.	2008	Arctic Svalbard	Estimation of macrophytes using single-beam		SBES/ MBES	Biological samples	*Laminaria*
Kruss et al.	2012	Arctic Svalbard	Acoustic properties of macrophytes		SBES/ MBES	Biological samples	*Laminaria digitata, Saccharina latissima, Alaria esculenta*
Kruss et al.	2017	Arctic Svalbard	Detecting macroalgae with acoustic		SBES/ MBES/ SSS	Biological samples/ divers	*Laminaria digitata, Saccharina latissima, Alaria esculenta*
Kutser et al.	2006	Estonia	Mapping benthic macroalgal cover in turbid coastal waters	Landsat 7 ETM + IKONOS/ALI/IKONOS		In situ reflectance spectra of benthic macroalgae	*Cladophora glomerata, Fucus vesiculosus, Furcellaria lumbricalis*

Authors	**Year**	**Country/ region**	**Purpose**	**Optical (satellite, aircraft, UAVs)**	**Acoustic**	**Ground truth/ Direct observations**	**Species/ habitat type**
Lee et al.	2015	**New Zealand**	Baseline seabed habitat and biotope mapping			Video camera on ROV/ Drop Down Video	*Ecklonia, Ulva*
Leleu et al.	2012	**New Zealand**	Habitat mapping/ Marine reserve/map comparison 30 years	Aerial photography	SSS	DropDown video/ sampling	mixed algae, *Carpophyllum* spp., *Ecklonia*
Madricardo et al.	2017	Italy	Multibeam and hydrodynamic datasets of tidal channels	Landsat 8 OLI	High res MBES	Dropframe video	macroalgal beds
McGonigle et al.	2011	USA	Detection of benthic macrophytes, deep water		MBES	Video stills	*Laminaria digitata, Laminaria longicruris, Agarum cribrosum*
Mielck et al.	2014	Germany	Predicting spatial kelp abundance		MBES RoxAnn	Video and dive transects	Kelp, mixed algae
Minami et al.	2014	Japan	Quantitative mapping of kelp forests/ harvesting		SBES	Video observations	*Laminaria spp*
Montereale Gavazzi et al.	2016	Italy	Extremely shallow benthic habitats mapping		MBES	Grab sample, underwater imagery	macroalgae and algal turfs
Mumby & Edwards	2002	Turks and Caicos Islands	Mapping marine environments/ clear shallow water/ comparison satellites imagery and airborne sensors	Ikonos 2/Landsat MSS/Landsat TM, SPOT HRV; airborne CASI		Field sites, data acquisition	macroalgae, seagrass
Murfitt et al.	2017	Australia	Monitoring intertidal reefs	UAV		In situ quadrat data	Macroalgae, *Hormosira banksii*

Authors	**Year**	**Country/ region**	**Purpose**	**Optical (satellite, aircraft, UAVs)**	**Acoustic**	**Ground truth/ Direct observations**	**Species/ habitat type**
Noiraksar et al.	2014	Thailand	Mapping Sargassum beds	ALOS AVNIR-2		Serial images taken by manta tow	*Sargassum spp.*
Oppelt et al.	2012	Germany	Intertidal macroalgae habitat mapping	airborne hyperspectral (AISAeagle)		Field spectrometer data	green, red, brown
Parsons et al.	2004	**New Zealand**	Fine scale habita mapping to detect changes in marine reserve			Video transects positioned by a radio acoustic positioning telemetry (RAPT) system	Kelp forest, mixed algae, turfing algae, *Carpophyllum* spp. *Ecklonia radiata*
Rattray et al.	2013	Australia	Detecting patterns of change in benthic habitats		MBES	Towed video sled	Kelp beds
Reimers et al.	2014	South Africa	Detecting and monitoring historical changes in South African coastal habitat			Historic images and repeat photographs were then taken from the same perspectives.	Kelp beds
Schimel & Ierodiaconou	2017	Australia	Hydroacoustic Mapping	airborne LiDAR	MBES		*Macrocystis pyrifera*
Setyawidati et al.	2017	Indonesia	Percentage cover, biomass, distribution of intertidal macroalgae, habitat mapping	GeoEye-1 (Digital Globe satellite), multispectral		Field surveys, quadrats, in situ measurements	*Sargassum, Turbinaria, Padina*
Simms & Dubois	2001	Canada	Map and biomass submerged kelp beds	SPOT HRV, Landsat Thematic Mapper		In situ reflectance spectra measurement	*Laminaria longicruris*
Smith et al.	2015	Antarctica	Habitat mapping		MBES	Video transects	*Himantothallus grandifolius,* macroalgae communities

Authors	Year	Country/ region	Purpose	Optical (satellite, aircraft, UAVs)	Acoustic	Ground truth/ Direct observations	Species/ habitat type
Stekoll et al.	2006	Alaska	Approach to estimating harvestable kelp biomass	airborne digital multispectral imaging system (DMSC); LandSat 7 Enhanced Thematic Mapper (ETM+)		Scuba measurements, transects	*Nereocystis luetkeana, Alaria fistulosa*
Sutherland et al.	2008	Canada	Kelp inventory	aerial colour infrared photography; digital mapping		Field sampling, scuba, mean weight per plant calculated	*Nereocystis luetkeana, Macrocystis pyrifera*
Uhl et al.	2016	Germany	Submerged kelp detection	airborne Hyper-spectral AisaEAGLE		Transect diving mappings	*Laminaria digitata, Laminaria hyperborea, Saccharina latissima, Desmarestia aculeata*
Vahtmäe et al.	2006	Baltic Sea	Benthic macroalgal cover in turbid coastal waters	airborne Hyper-spectral AVIRIS/CASI		In situ measurements of benthic reflectance spectra	*Cladophora glomerata, Fucus vesiculosus, Furcellaria lumbricalis*
Ventura et al.	2016	Italy	Mapping of coastal fish nursery grounds	UAV			macroalgae seagrass
Volent et al.	2007	Norway	Mapping Kelp forests	Hyper- spectral push broom technique		In situ measurements/ SCUBA	Kelp and substatum discrimitation shallow water
Wang & Hu	2016	Central West Atlantic	Mapping and quantifying Sargassum distribution and coverage	MODIS; Hyper-spectral HICO		Local reports	*Sargassum* spp.
Wang & Hu	2017	Caribbean Sea	Predicting *Sargassum* bloom	MODIS		MODIS data collected from 2000 to 2016	*Sargassum* spp. blooms
Xu et al.	2017	Yellow Sea	Monitoring for a green tide	MODIS; AUV		In situ observation data/UAV image	*Ulva* spp.
Young et al.	2015	Australia (Victoria)	Predictive abundance of canopy forming kelp	airborne LiDAR	MBES	Towed video	*Ecklonia radiata*

Authors	**Year**	**Country/ region**	**Purpose**	**Optical (satellite, aircraft, UAVs)**	**Acoustic**	**Ground truth/ Direct observations**	**Species/ habitat type**
Zabloudil et al.	1991	USA	Mapping of giant kelp density, distribution		SSS/down-looking sonar		*Macrocystis pyrifera*
Zavalas et al.	2014	Australia	Habitat classification of temperate marine macroalgal communities	airborne LiDAR		Video data, ROV	kelp and erect fine branching algae

11 APPENDIX 3. Mapping techniques; brief background information

Remote sensing (the science of obtaining information about objects or areas from a distance, typically optical sensing from satellites or aircraft, more recently from drones, or acoustic sensing from the sea surface) has the advantage of covering large areas at relatively low cost and enables access to remote areas.

Acoustic devices

Acoustic devices include Single Beam Echo Sounders (SBES) and Multi Beam Echo Sounders (MBES) as well as Side Scan Sonar (SSS). SBES and MBES provide accurate information about bottom hardness and roughness and have been used for bathymetric and habitat mapping especially in deep waters. SBES were amongst the earliest systems developed and are still the most common form of underwater acoustic system in use today, while the first MBES systems appeared in the late 1970s (Colbo et al. 2014). SBES produces an echogram of the sea floor at a point directly below the transducer, while the MBES emits sound waves in a fan shape beneath the vessel and provides a fuller coverage. As echo sounders are not affected by water clarity, they have been successfully used in turbid and exposed waters and can be mounted on large vessels or small boats, and allow for surveys in shallow waters, although some limitations exist. Acoustic devices, including MBES, have had restricted use in depths of 2–5 metres, mostly due to the sidelobe effect (Hughes Clarke 2006, Urban et al. 2017) and bottom reverberation or multiple reflections (Madricardo et al. 2017). These obstacles often result in an area of no information, termed “the white stripe” between sonar coverage and the coastline (Zavalas et al. 2014). However, recently MBES have been used to map extremely shallow water in the Venice lagoon (Montereale Gavazzi et al. 2016) suggesting improved technology.

Single and Multibeam Echosounders

Single Beam Echo Sounders (SBES) and Multibeam Echo Sounders (MBES) have been used to map macrophytes in the artic fiords of the Svalbard Archipelago (Kruss et al. 2008, 2012, 2017). Presence/absence of algae and heights were calculated from the backscattering strength of SBES and MBES. The MBES was used in water depths as shallow as 0.5 m, and in correlation with SBES measurements it was able to resolve the ‘broad, flat leaves’ typical of the *Laminaria* species, having a nominal vertical resolution of 1 cm. MBES didn’t resolve the thick stalks of *Laminari*a seen in the SBES (Kruss et al. 2008).

Kruss et al. (2017) compared the data from single and multibeams sounders and suggested a method to map macroalgal beds in turbid water that could be also applied to other regions. Surveys were carried out with the single and multibeam sounders mounted on the sides of the boat and working simultaneously, without interference, each one linked to its own GPS. The data received from the echograms were compared with direct observation and biological samples. The analysis of single beam data did not involve a calculation of the beam angle, needed instead for the multibeams, and provided reliable results over flat bottom and slopes less than 20°, as an increase in the slope influenced the echo shapes and acoustic backscatters. As the data analysis from the multibeams needed to take into account the noise coming from the sidelobe, the authors carried out a tank experiment, to calculate the beam angle within which seaweed could be detected with confidence and excluding data from the outer beams that can mask seaweed. They also provided a method to process echograms from the two echo sounders, and a suggestion to improve multibeams data, e.g., the use of a motion reference unit not used in their studies.

Habitat mapping by multibeam and towed video observations were carried out on shallow reefs in Victoria (Australia) which are dominated by kelp and red algal assemblages. A machine-learning algorithm to predict the distribution of benthic biological habitats was developed and the accuracy of the resulting final habitat maps was 88.5% to 93.6% (Che Hasan et al. 2014). MBES data and underwater video were used to model and map marine habitat including mixed red algae, *Ecklonia*, and rhodoliths in Victoria (Holmes et al. 2008).

Image-based classification techniques and multibeam backscatter were used to detect deep water benthic macroalgae in the Gulf of Maine. Two classifications (QTC-Multiview and WCDE), were performed independently and the results were compared showing that both techniques were appropriate for detection of dense canopy of *Laminaria* sp. in shallow water (<30 m). In deeper water the occurrence of solitary individuals or small patches of *Agarum cribrosum* were not detected effectively, but the efficiency of prediction decreased with depth. Both methods were in agreement with the ground-truth data (McGonigle et al. 2011). In the North Sea, single beam (RoxAnn) survey data, consisting of the two seafloor parameters roughness and hardness, ground truthed with underwater videos, detected kelp forests with different densities and other macroalgal communities, between 2 and 18 m depth (Mielck et al. 2014). In Venice's lagoon, areas covered by macroalgae were distinguished from bare bottom using MBES data in tidal channels and inlets (Madricardo et al. 2017). Distribution maps of kelp forest along the coast of Shiretoko Peninsula (Japan) were obtained by MBES data and geostatistical analysis. The acoustic signals were categorised into three groups, kelp forest, sea bottom, and seawater data allowed to estimate densities of kelp (Minami et al. 2010).

High resolution multibeam has been used successfully to map seagrass beds e.g., in the Mediterranean (Di Maida et al. 2011, Pasqualini et al. 2000) and in Japan (Hamana & Komatsu 2016, Komatsu et al. 2003) and also to map vents or seeps (Urban et al. 2017).

In 2015 a multi-beam survey was undertaken in the Marlborough Sounds, Northwest D'Urville, to map the seafloor and identify the diversity of physical habitats (Neil et al. 2015). In 2016 a joint project between Marlborough District Council, Land Information New Zealand and NIWA aimed to map the seabed features three-dimensionally and produce new hydrographic charts for navigation safety. The multi-beam also captured water column features which enabled different habitats to be determined, the identification of seeps and plumes, and detection fish shoals and kelp beds. Two multi-beams were used mounted on two vessels, the Kongsberg EM 2040 MBES on the RV *Ikatere* operating in deeper water and the Kongsberg Geoswath Plus Compact on the RV *Rukuwai* which allowed the acquisition of geo-referenced side scan and bathymetric information in water depths less than 5 m, as well as along coastline areas not surveyed by the EM2040 multibeam. The Geoswath sonar receives information up to an angle of 120° either side of the sonar which allowed it to survey up to the drying line (when the survey was conducted in the upper half of tidal cycle).

The seafloor surrounding Kapiti Island was mapped using the Kongsberg EM2040 Multibeam Echosounder (MBES) by NIWA in 2015 (Pallentin et al. 2016). The map produced is available at https://www.niwa.co.nz/media-gallery/detail/109673/42526). As the vessel could not operate in shallow waters to achieve complete coverage of bathymetry data around the island, multibeam data was augmented with satellite-derived bathymetry for this area (Laferriere 2016). The geophysical data acquisition, using MBES and preliminary data processing are provided in Pallentin et al. (2016).

The Side Scan Sonar is an acoustic imaging device used to provide wide-area, high-resolution pictures of the seabed ideal for object detection, but it does not normally produce bathymetric data. It provides information on sediment texture, topography and bedforms (Kenny et al. 2003). It has also been used in habitat mapping, e.g., to map seagrass beds (Siljestrom et al. 1996), coral reefs (Prada et al. 2004, 2008) and in combination with other techniques, e.g., multibeam (Kruss et al. 2006, Shono et al. 2004), it has been found to be an economical and effective survey method. An automatic method to detect seagrass potholes (boat propeller and anchor scars) in shallow seagrass meadows of Laguna Madre in Texas was proposed, based on side scan sonar data (Rahnemoonfar & Rahman 2016). Density and distribution of giant kelp, *Macrocystic pyrifera*, and substrate type, were estimated using side-scanning and down-looking sonar. Results indicated that these techniques could detect changes in kelp density and distribution (Zabloudil et al. 1991).

Laser line scan technology, developed primarily for military applications, has been used to characterise deep-water benthic habitats (Yoklavich et al. 2003) and for fishery habitat assessment, generating high resolution (1–2 cm across track) imagery (Amend et al. 2007). Currently no studies have tested this technique on macroalgal beds. Yoklavich et al. (2003) were able to detect clearly drift kelp on the seafloor, suggesting that this technique might be useful to detect macroalgal beds.

The main advantage of using acoustic remote sensing is being independent from water clarity thus ideal in turbid water, however, access to very shallow water would be challenging and dependent on the boat size. To interpret acoustic data a complete understanding of the role of sidelobes, sectors and seabed angular response is needed (Hughes Clarke 2006) especially operating in shallow water where the back scatter is usually sub-optimal.

The multibeam operation alone would be hugely expensive, considering that the acquisition of a line 28 km long, 50 m across (1.4 km^2), in 5 m depth of water, at 7 knots, with a 10% overlap and 65 % beam angle, takes 6.5 hours. Processing water column, backscatter and bathymetric data takes 2 days for every day of data acquisition (Steve Wilcox (NIWA) pers. comm.). However, there are regions where multibeam data have been acquired for bathymetry profiles or habitat mapping. When these data become available they could be complemented with data acquired by direct observations (e.g., towed or drop camera) to build accurate maps of macroalgal beds.

Optical sensing

Optical remote sensors collect data by detecting the energy that is reflected from Earth to create pictures of the Earth using electromagnetic radiation of a range of frequencies, from radio waves to gamma rays. Passive sensing uses the sun’s energy as a source of illumination, while in active sensing the sensor emits energy to illuminate the target. As electromagnetic radiation from the sun, or emitted from the satellite itself, hits objects on the Earth, a portion of that radiation is reflected back to the sensor. Sensors measure the intensity of the reflected radiation at selected wavelengths. https://oceanservice.noaa.gov/geodesy/remote-sensing/. Different objects do not reflect radiation in the same way: clear water, for example, will reflect light differently than turbid water (Coggan et al. 2005).

Remote sensing sensors are classified in into several types based on the number of spectral bands used in the imaging process. Panchromatic sensors (e.g., IKONOS-PAN and SPOT HRV-PAN) have a single-channel sensitive to broad wavelength radiation (Thenkabail 2016). The imagery is black and white and targets are identified by their brightness. Multispectral sensors (e.g., LANDSAT MSS; LANDSAT TM; SPOT HRV-XS; IKONOS MS) commonly collect data in three to six spectral bands centred around selected wavelengths, in a single observation, from the visible and near-infrared regions of the electromagnetic radiation spectrum (Govender et al. 2007). Super spectral sensors (e.g., MERIS, MODIS) have several spectral channels (typically more than 10) and the bands have narrow bandwidths, enabling the fine spectral characteristics of the targets to be captured (Thenkabail 2016). Hyperspectral sensors acquire imagery in a high number, typically 200 or more, of contiguous and narrow spectral bands, producing much more detailed spectral data and enabling the construction of an almost continuous reflectance spectrum for every pixel in the scene, thus allowing for detailed applications. Different materials are discriminated by wavelength-dependent absorptions, and these images of reflected solar energy are known as spectral signatures (Govender et al. 2007). Despite the number of bands, sensors are based on the same physical technology, they record radiance in the Visible to Near-InfraRed (VNIR), spanning the 400–1000 nm wavelength range and in the Short-Wave InfraRed (SWIR) 1000–2400 nm of the spectrum (Transon et al. 2018).

Remotely sensed images are classified by four different types of resolution (Klemas 2011b). The spatial resolution is defined by the length and width of the pixels which compose an image.

Spatial resolution is often represented in terms of distance (e.g., 30 m, 1 km, etc.). The spectral resolution is the spacing between the selected wavelengths, for each of which the sensor provides an image (Govender et al. 2007). Radiometric resolution is a measure of a sensor's ability to distinguish between two objects of similar reflectance. Temporal resolution is a measure of how often the same area is visited by the sensor, and does not refer to a single image, but a series of images that are captured by the same sensor over time (Klemas 2011b). To obtain the required spatial, spectral, and temporal resolutions, coastal ecosystems frequently need to be observed from both satellite and aircraft (Klemas 2011b). Remote sensing imagery combined with geographical information system (GIS) offers the advantage of being able to compare images obtained on different dates and thus the ability to detect changes in kelp forests or seagrass beds (Deysher 1993, Fyfe et al. 1999).

Multispectral imagery has been used successfully in a wide range of applications, e.g., to map vegetation of all types including tidal vegetation such as seaweed, salt marshes (Coggan et al. 2005), to assess benthic substrate composition in shallow waters (Werdell & Roesler 2003), to monitor benthic intertidal macroalgae (van der Wal et al. 2014), to map water quality and substrate cover (Phinn et al. 2005), and, through a time series of images, to detect changes in seagrass beds (Dekker et al. 2005).

Since the first hyperspectral sensor in space, Hyperion, launched in 2000, scientists have developed several hyperspectral Earth Observation (EO) remote sensing techniques, and methods of analysis and utilization (Transon et al. 2018). Hyperspectral airborne or satellite sensors have had different applications e.g., mapping shallow water benthic habitat (Kutser et al. 2006), assessing rocky intertidal areas (Henning et al. 2007), detecting benthic macroalgal cover in shallow water (Casal et al. 2013), assessing the trophic status of lakes (Koponen et al. 2002), and water quality in lake, estuarine and coastal waters (Brando & Dekker 2003, Giardino et al. 2007), mapping seagrass species, cover and biomass (Phinn et al. 2008). Some limitations of hyperspectral technologies are a result of the spatial resolution of satellite sensors such as Hyperion (30 m) being too coarse in many circumstances, the production of large data volumes, and the high data-processing effort and cost (Casal et al. 2011a).

Infrared photography, hypespectral imagery

Aerial photographs are available dating back to the 1930s (Klemas 2011b) and were one of the first methods used to map kelp forests, and are still in use, e.g., in California (refer monitoring section). Fyfe et al. (1999) used aerial photography (colour negative film and a UV filter) to map *Macrocystis* beds in Otago. A protocol to estimate the biomass and extent of *Nereocystis* and *Macrocystis* beds has been developed in British Columbia using aerial coloured infrared photography and digital mapping of kelp polygons directly from georeferenced digital images (Sutherland et al. 2008). This method allows replication of measurements and the detection of changes. The use of an airborne hyperspectral sensor was investigated for monitoring and management of kelp forests in the Artic fiords. Synchronously, remotely sensed images as well as in situ measurements of spectral irradiance were acquired at 30 cm depth (Volent et al. 2007). The airborne Hyperspectral Infrared Imager (HyspIRI) has been used to acquire multiple biophysical variables, such as phycocyanin and chlorophyll-a in a freshwater ecosystem (Hestir et al. 2015). High spatial resolution hyperspectral airborne CASI sensor was tested to map benthic habitat in the Baltic Sea. Habitat classification was established depending on spectral reflectance. The main confusion was between brown and red algae as both algae had relatively low reflectance values and differences showed only in narrow spectral regions, however the *Fucus vesiculosus* habitat was relatively well classified (Vahtmäe et al. 2012). An airborne Hyperspectral Scanner sensor was tested to map macroalgal communities. Reflectance spectra of the dominant macroalgae were acquired in the field and macroalgal groups (green, brown and red) were found to be able to be differentiated from sand in up to 8 m depth (Casal et al. 2012).

Airborne hyperspectral AISAEAGLE data were used to determine the distribution of macroalgal communities in the rocky intertidal zone of Helgoland, Germany (Oppelt et al. 2012). A fully automated processor was developed to detect the presence of kelp in submerged habitats, based on airborne hyperspectral AISAEAGLE data, acquired during 13 flight stripes and validated with transect diving mapping. A Water Anomaly Filter was applied to prepare the data for kelp analysis and a spectral kelp Feature Detection was used to separate kelp pixels from non-kelp pixels. Hyperspectral kelp detection around Helgoland (Germany, North Sea) was possible up to approximately 6 m depth. The feature detection processor resulted as a time-effective approach to assess and monitor submerged kelp at the limit of water visibility depth (Uhl et al. 2016).

LIDAR

Airborne laser surveys using LIDAR (Light Detection And Ranging sensor) have become a useful technique for topographic and bathymetric mapping (Allouis et al. 2010, Pe'eri & Long 2011). Lidar uses light in the form of a pulsed laser to measure ranges (variable distances) to the Earth. These light pulses, combined with other data recorded by the airborne system, generate precise, three-dimensional information about the shape of the Earth and its surface characteristics. A LIDAR instrument principally consists of a laser, a scanner, and a GPS receiver (https://oceanservice.noaa.gov/facts/lidar.html). The topographic LIDAR uses a near-infrared wavelength to map the land while the bathymetric LIDAR uses green wavelengths, which can penetrate the water column up to 2–3 Secchi depths, depending on the application (Pe'eri & Long 2011). In clear water LIDAR pulses have penetrated down to 50 m (Klemas 2011a). Using airborne LIDAR one can produce bathymetric maps, even in moderately turbid coastal waters (Klemas 2009). LIDAR has several applications in environmental studies, e.g., mapping coral reefs (Brock et al. 2004, Zawada & Brock 2009), seagrass beds (Ishiguro et al. 2016), river bathymetry (Kinzel et al. 2013), shallow coastal areas (Allouis et al. 2010, Wang & Philpot 2007), beach profiles (Klemas 2011a), coastal research and management (Brock & Purkis 2009, Pe'eri & Long 2011), measuring reef complexity (Wedding et al. 2008), fisheries management (Churnside et al. 2011), and in coastal and estuarine habitat mapping (Chust et al. 2008).

LIDAR can be mounted on Unmanned Aerial Vehicles (UAVs), balloons or blimps (Klemas 2015), thus making data collection more accessible and less expensive for environmental studies. The laser-induced fluorescence–light detection and ranging (LIF LIDAR) sensor was used for mapping microalgal and macroalgal communities inhabiting intertidal sediments in an estuary (Gameiro et al. 2014). LIF sensors could detect and characterise different macroalgae (red, green and brown) based on their specific pigments (Utkin et al. 2014).

Underwater Hyperspectral Imaging

Imaging spectrometers have also been deployed on underwater vehicles to image the seafloor outside the limits of passive remote sensing techniques (Johnsen et al. 2013). Recently underwater hyperspectral imaging (UHI) has been investigated as a way to create biogeochemical maps of seafloor properties. UHI can be deployed on underwater vehicles, such as ROV or AUV, and deployed in proximity to the seafloor to systematically map the benthos over large areas. Gliders are not ideal platforms for UHI as they have low power capacity and are limited in activities near the seafloor. A spatial resolution of approximately 2 mm can be achieved with a UHI when scanning targets at distance of 2 m above the seafloor (Johnsen et al. 2013). A summary of the results from a UHI system deployed on ROV to identify and map different targets on the seafloor including kelp was presented by Johnsen et al. (2016). The potential of UHI as a coralline algal identification and mapping tool was investigated on four species: *Corallina officinalis* L., *Lithothamnion glaciale* Kjellm., *Phymatolithon lenormandii* (Aresch.) Adey, and *Phymatolithon tenue* (Rosenv.) Düwel & Wegeberg. Supervised UHI classification was unable to accurately map different coralline algal species due to the similarity of the optical fingerprints, however, as a group, coralline algae could easily be identified (Mogstad & Johnsen 2017). A diver-operated hyperspectral imaging and topographic surveying system was developed for automated habitat mapping and tested on coral reefs (Chennu et al. 2017)

Satellites

Landsat satellites were the first series of satellites equipped with multispectral scanners to acquire images of the Earth. Landsat 1 was launched in 1972 and since then the technology of sensors and satellites has greatly advanced. Currently, Landsat 8 (launched in 2013) is equipped with two sensors: the Operational Land Imager (OLI) and the Thermal Infrared Sensor (TIRS). These provide seasonal coverage of the global landmass at a spatial resolution of 30 metres (visible, NIR, SWIR), 100 metres (thermal), and 15 metres (panchromatic) (see https://landsat.gsfc.nasa.gov/landsat-data-continuity-mission/). The French SPOT (Satellite Pour l'Observation de la Terre) missions have orbited the Earth for over 30 years, with SPOT-1 launched in 1986 and the current SPOT-7 launched in 2014. These satellites provide images at the intermediate spatial resolution, between 5 and 25 m, and an increased number of spectral bands and higher radiometric sensitivity (Dekker et al. 2006). At the end of the 1990s, very high-resolution sensors such as IKONOS (1999–2015) and Quick-Bird (2001–2014) became available, offering imagery with pixel sizes of 0.6–1.0 m in panchromatic and 2.4 – 4 m in the multispectral bands (Dekker et al. 2006). The commercial satellite, Digital Globe, currently provides high resolution imagery acquired from WorldView-1 to 4 and GeoEye-1.

The European Union's Copernicus programme and its CMEMS marine database are based on satellite and in situ observations, as well as numerical model output. The Sentinel-1 mission, launched in April 2014, provides all-weather, day and night radar imagery with cloud-penetrating ability for land and ocean services. The twin satellites Sentinel 2A (launched in 2015) and Sentinel 2B (launched 2017) provide high resolution optical images for land vegetation, soil and water cover, inland waterways and coastal areas. Sentinel-3 (launched in 2016) measures sea-surface topography, sea- and land-surface temperature, ocean colour and land colour with high-end accuracy and reliability (http://copernicus.eu/main/sentinels). Sentinel-2 multispectral satellite remote sensing is now providing free, open, global and systematic high-resolution visible and infrared imagery at a short revisit time (Transon et al. 2018).

In New Zealand, non-commercial satellite imagery is available from SPOT, Landsat series and recently Sentinel-2. High resolution, imagery can be also purchased from Digital Globe.

Mapping floating kelps beds with satellite data

The Landsat Multispectral Scanner (MSS) imagery was tested to estimate the areal extent of kelp beds along the coast of Santa Barbara, California (USA) and data were compared with high-altitude colour infrared aerial photography and aircraft X-band radar imagery. The Landsat MSS estimates of areal extent were highly correlated with estimates derived from aerial photographs; however, the kelp extent was underestimated, mostly due to the limited spatial and radiometric resolution of the Landsat MSS (Jensen et al. 1980, Jensen et al. 1981). Subsequently, a time sequence of Landsat MSS, Thematic Mapper (m) and aircraft Thematic Mapper Simulated (TMS) digital data was tested to document changes in giant kelp distribution for two Santa Barbara beds (Jensen et al. 1987).

The SPOT High Resolution Visible (HRV) satellite system was used to map the distribution and to quantify the abundance of giant kelp (*Macrocystis pyrifera*) along the coast of San Diego, California (Augenstein et al. 1991). Data from SPOT-1 and NOAA-11 AVHRR were used to map and estimate the biomass as wet weight of *M. pyrifera* at the Kerguelen Island (Indian Ocean) and to establish a possible correlation between its occurrence and sea-surface temperature. Field data, collected at the time of image acquisition, validated the remote sensing data (Belsher & Mouchot 1992). However, SPOT imagery was able to map only large *Macrocystis* beds with the 20 m resolution available at the time (Deysher 1993). Multi temporal distribution maps of *Macrocystis* spanning nearly 60 years were generated using satellite data, aerial photography, GIS, and correlations to physical variables (Bushing 2000). In Alaska, to assess *Nereocystis luetkeana* and *Alaria fistulosa* beds for harvesting, an aerial digital multispectral imaging system (DMSC) was calibrated in combination with ground truthing data acquired by scuba divers and LandSat 7 Enhanced Thematic Mapper (ETM+) panchromatic imagery. The canopy extent estimated by the satellite data was approximately 50% greater than the areas derived from the high resolution DMSC that also could detect subsurface kelp plants down to 3 m, providing a very accurate estimate of kelp bed size (Stekoll et al. 2006).

Panchromatic imagery was tested for mapping *Macrocystis* beds and compared with multispectral imagery. Results showed significant challenges with panchromatic images e.g., false indications due to sun glint, clouds, or other bright objects in the images, however these images had much higher spatial resolution and may provide an alternative method to multispectral imagery (Kim et al. 2010). A method using satellite imagery to estimate giant kelp canopy cover and biomass and assess temporal changes, was developed and verified with aerial imagery and with monthly diver observations along transects within the Santa Barbara Coastal Long Term Ecological Research project (Cavanaugh et al. 2010). The dynamics and changes in biomass of *Macrocystis pyrifera* along the coast of California were studied at spatial and temporal scales (up to 25 years) using Landsat 5 Thematic Mapper satellite imagery (Cavanaugh et al. 2011). Biomass changes of *Macrocystis pyrifera* from temporal data (2000–2011) were studied along approximately 550 km of mainland California coast using 30 m resolution multispectral Landsat 5 TM imagery in combination with diver surveys and in relationship other factors e.g., environmental data, sea urchins abundance (Cavanaugh et al. 2013). The use of the hyperspectral infrared imager (HyspIRI) for monitoring the biomass and physiological condition of giant kelp forests was tested in combination with Landsat 8 imagery, Airborne Visible/Infrared Imaging Spectrometer (AVIRIS) and field data. Hyperspectral imagery showed the potential to provide new insights into the ecology and physiology of giant kelp (Bell et al. 2015).

As Landsat imagery covers the entire surface of the earth every 16 days and long-time series of images are available, they have been used to uncover the history of giant kelp forests around the globe. However, software is not able to automatically detect kelp forests. To help scientists with this time-consuming work, a citizen project “Floating Forest” has been launched. Citizen scientists, recruited via internet, are instructed to outline giant kelp patches found on Landsat images. Their findings are crosschecked with those from other citizen scientists and then passed to the science team for verification. The size and location of these forests are catalogued and used to study global kelp trends. The science team includes world experts on kelp e.g., Drs. Kyle C. Cavanaugh, Jarrett E. K. Byrnes, Tom Bell. The project is part of the NASA Citizen Science for Earth Systems Programme managed by Zooniverse. Currently, 2139 volunteers have joined the project, and 51189 classifications have been made (https://www.zooniverse.org/projects/zooniverse/floating-forests).

Moderate Resolution Imaging Spectroradiometer (MODIS) was used to detect the presence and coverage of floating *Sargassum* canopies in the Caribbean Sea (Wang & Hu 2016), and to create probability maps for predicting blooms and non-bloom areas (Wang & Hu 2017). In the Yellow Sea, *Ulva prolifera* biomass was estimated based on its spectral reflectance. An index was developed linking biomass per area to the reflectance of the floating algae which made it possible to analyse a time-series of MODIS observations between 2008 and 2015 and provide quantitative information (Hu et al. 2017).

In New Zealand, the Department of Conservation tested multispectral satellite images (WorldView-II) in two areas in the Marlborough Sounds, Long Island and Tory Channel, to map the extent of *Macrocystis* beds. The images taken in September 2010 and 2013 showed the potential for both mapping large areas and for creating past and future time series. Tides, currents and turbidity may have an effect on the satellites' signal, thus requiring further study (Meng et al. 2015).

Mapping subtidal macroalgae with satellite data

Submerged macroalgal beds have been successfully mapped in clear water by satellite imagery. In Turks and Caicos Islands, satellite (IKONOS, Landsat TM) and airborne (CASI) imagery were compared along an approximately 1 km transect from the shore to a depth of about 18 m for mapping shallow-water marine environments (Mumby & Edwards 2002). In Tahiti, the biomass and distribution of two invasive species *Turbinaria ornata* (Turner) C.Ag. and *Sargassum mangarevense* (Grunow) Setchell, were estimated using 4 m resolution IKONOS imagery and field surveys (Andréfouët et al. 2004). Multispectral Pleiades image and field surveys were used to estimate the biomass of *Sargassum polycystum* along the lagoon shores of Funafuti Atoll (Tuvalu). The analysis of a time series of high resolution imagery (2002–2015) made it possible to establish the initial algal bloom in 2010 and to determine that the biomass remained stable, with algae disappearing in some areas, and developing in others (Andréfouët et al. 2017). In Yucatan (Mexico), habitat mapping was performed by Landsat ETM+ imagery, and 290 field sites were surveyed by underwater videos. Seven habitat types were identified including seagrass, macroalgal forest, and macroalgae on sand or flagstone (Betzabeth & de los Angeles 2017).

High spatial resolution WorldView-2 (WV-2) imagery, validated using a depth invariant index model for water-column correction, and in combination with field observations, resulted in accurate assessment of intertidal and subtidal bed distribution of *Sargassum* spp. around Rottnest Island (Western Australia). The WV-2 imagery, composed of eight bands, six of which are visible and two are near-infrared, had a 2 m spatial resolution. To ground truth the data a minimum of 10 transects with five survey quadrats per transect, was required to assess an area of approximately 20 km^2 (Hoang et al. 2016). Advanced Land Observing Satellite (ALOS) Advanced Visible and Near Infrared Radiometer Type 2 (AVNIR-2) satellite imagery were used to detect and map submerged seaweed and seagrass beds in the northeast of Brazil in combination with field surveys (da Silva et al. 2017).

Sargassum beds were also mapped in Thailand using the non-commercial satellite, ALOS with sensors AVNIR-2, and biomass was estimated in combination with field data (Noiraksar et al. 2014). SPOT-4 was used successfully to map subtidal kelp forests in turbid waters on the Galician coast (NW Spain) up to 10 m depth (Casal et al. 2011b). Submerged kelp beds, dominated by *Laminaria longicruris,* along the Atlantic Canadian coast, were assessed to detect seasonal variation in biomass using satellite imagery SPOT HRV and Landsat Thematic Mapper (TM). Kelp-covered and kelp-free areas were detected up to a water depth 6 m and 7 m with the HRV and TM images respectively (Simms & Dubois 2001).

Several studies have focused on exploring whether macroalgae or seagrass species are discernible from each other based on their optical signatures. Field measurements of reflectance spectra of three seagrass species showed them to be spectrally distinct, suggesting that “species discrimination should be possible in the remote sensing of benthic aquatic vegetation using a hyperspectral sensor that has narrow bands centred on pigment-related spectral features in the visible wavelengths” (Fyfe 2003). In New Zealand, SPOT imagery was tested to monitor intertidal and subtidal vegetation in Otago harbour with seagrass and red macroalgae being clearly distinct. The overall classification accuracy was 87% for intertidal and subtidal vegetation (Israel & Fyfe 1996).

The MERIS configuration of spectral bands allowed the recognition of red, green and brown macroalgae based on their spectral signatures. The reflectance spectra of three indicator species for the Baltic Sea, *Cladophora glomerata* (L.) Kütz. (green macroalga), *Furcellaria lumbricalis* (Huds.) J.V.Lamour.(red macroalga), and *Fucus vesiculosus* (brown macroalga), were measured and a bio-optical model was used to estimate if these algae were distinct from each other and from the sandy bottom. Results indicated that “to some extent” it was possible to map these species with multispectral satellite sensors in turbid waters. However, macroalgae could be detected in shallow water but not down to their maximum depth range (Kutser et al. 2006). MERIS imagery was also used to determine irradiance levels corresponding to the lower infralittoral limit in the Azores, derived from the deepest kelp occurrences that, in combination with underwater video surveys, was estimated at 69 m depth (Amorim et al. 2015). The use of hyperspectral remote sensing for mapping benthic macroalgal cover in turbid coastal waters of the Baltic Sea indicated that the depths where benthic macroalgae could be separated from each other by remote sensing, did not differ significantly in clear or turbid coastal waters (Vahtmäe et al. 2006). An IKONOS satellite image was used to map seaweed beds in Japan. The spectral reflectance of seaweeds and other substrates were measured in the field and results showed effective wavelength bands for distinguishing seaweeds from other substrates (Frouin et al. 2012).

Predictive modelling

A broad-scale approach is becoming essential for marine management and conservation planning. Habitat mapping, defined by biotic and abiotic features of the seafloor, is used to evaluate and predict patterns in benthos distribution (Holmes et al. 2008). Modelling combining surrogates that describe the fine-scale physical characteristics of the seafloor with fine-scale biological data collected from an AUV have been tested in south-eastern Tasmania, to explain and predict the distribution of key sessile biota. These surrogates are environmental and physical data that have greater spatial coverage than biological datasets, the latter being expensive and labour-intensive to collect. Maps of the predicted distribution of biota provide comprehensive and quantitative information for spatial management (Hill et al. 2014). Factors that will influence the distribution of kelp forest are type of substrate, topography, depth, water clarity, temperature, and currents (Méléder et al. 2010, Yesson et al. 2015a, 2015b, Young & Carr 2015a, 2015b).

There are several large international programmes starting to map coastal ecosystems. Some examples include:

European Directives, such as the MSFD and the Horizon 2020 roadmap require a multi-resolution full coverage of all European seas including bathymetry, geology and habitats. ‘Seabed Habitats’ is part of the European Marine Observation and Data Network (EMODnet) initiative, funded by the European Maritime and Fisheries Fund. Seabed Habitats was initiated with Phase I (2009-2013) through the EUSeaMap project and is currently in Phase III (2017–2020). Seabed Habitats has developed, improved and gradually increased the coverage of a broad-scale seabed habitat map for Europe's seabed (Populus et al. 2017) and aims to produce broad scale seabed habitat maps covering all European seas in a consistent way.

A time and cost-efficient way to produce such maps is to use low-resolution datasets and models to predict broad scale seabed habitat types (Manca & Vasquez 2015). EUSeaMap is based on the EUNIS classification (European Nature Information System) which provides a common European reference set of habitat types for terrestrial, freshwater and marine habitats (Populus et al. 2017). Broad-scale seafloor habitat mapping has the benefit of producing maps covering large extents at a reasonable cost and provides immediate resources to guide decision-makers involved in regional or local ocean management (Vasquez et al. 2015). An interactive map is available online (http://www.emodnet-seabedhabitats.eu/default.aspx?page=1934). UKSeaMap 2010 was established to produce an ecologically relevant, full-coverage map of seabed habitats across the entire UK marine area and is based mostly on predictive habitat models (McBreen et al. 2011). MAREANO started in 2006, is mapping depth and topography, sediment composition, contaminants, biotopes and habitats in Norwegian waters**.** Videos and echo sounders on various platforms have been tested and the use of autonomous vehicles to acquire data in very shallow waters (http://mareano.no/en)

12 APPENDIX 4. Mapping macroalgae using combined approaches

The Victorian Marine Habitat Mapping Project was initiated to map over 500 km^2 in 14 marine regions within Victorian state waters using a combination of multibeam sonar bathymetry, backscatter and information collected using towed video systems. Results of a pilot study carried out off the coast of Warrnambool (South West Victoria) allowed for the discrimination of three broad substrate categories and seven distinct biota classes, using an automated decision tree classification system (Ierodiaconou et al. 2007). Subsequently towed video observations and 11 seafloor complexity variables (derived from multibeam) were used to predict the distribution of 8 dominant benthic biological communities off the central coast of Victoria. The utility of two decision trees (QUEST and CRUISE) and a Maximum Likelihood Classifier (MLC) were tested to predict benthic habitat. The video data were classified using the Victorian Towed Video Classification Programme (Ierodiaconou et al. 2011).

Hydro acoustic data acquired with MBES and combined with existing Lidar bathymetric data and field surveys were used to produce map of the seafloor at Wilson Promontory Marine National Park. Areas of seagrass and mixed brown algae were identified (Schimel & Ierodiaconou 2017). Seafloor mapping using the LiDAR and multibeam in combination with models of wave energy provided predictive distribution and abundance maps of *Ecklonia radiata* off the coast of Victoria (Young et al. 2015). At the Kent Island Group in south-eastern Australia, habitat mapping was carried out by using single-beam acoustics and videos for ground truthing to plan MPA. Several habitat types were identified including macroalgae and seagrass beds and sponge communities (Jordan et al. 2005).

A method to detect patterns of change in benthic habitats by acoustic and video data and considering four representative biotic classes enabled quantification of short-term seasonal changes in distribution of kelp-dominated habitats (Rattray et al. 2013). A profiling sonar (Sediment Imager Sonar) and a towed video sledge were used to study a *Zostera marina* bed in southern UK. An algorithm was developed to detect seagrass from the sonar data and tested against video footage. Four parameters were calculated from the sonar data; water depth, a seagrass index, canopy height and patchiness. *Zostera* density, abundance of macroalgae and bottom type were estimated from video data (Lefebvre et al. 2009). IKONOS satellite imagery and side scan sonar measurements were used to map seagrass beds in Japan (Sagawa et al. 2008). Multibeam sonar and underwater video data were used to map benthic habitats and associated communities in East Antarctic revealing a diverse and heterogeneous seabed environment with a complex mosaic of benthic habitats including the brown alga *Himantothallus grandifolius* common on bedrock outcrops and red macroalgae in the embayments (Smith et al. 2015). In Brittany, *Laminaria hyperborea* and *L. digitata* distribution and biomass maps were produced with a multi-approach combining bathymetric and topographic Lidar acquisition, aerial surveys, MBES and spectral imagery (sensor Asia Eagle 1k), underwater videos and scuba diving (Bajjouk et al. 2015). Side Scan sonar, beam trawls, box corers, ROV and an underwater drop camera were used to map benthic habitat, between 50 and 100 m depth, including rhodolith beds and seagrass meadows in Menorca (Spain) (Barbera et al. 2012).

In New Zealand there have been extensive coastal habitat mapping projects carried out in Northland at Mimiwhangata Marine Park (Kerr & Grace 2005), Doubtless Bay (Grace & Kerr 2005), Motukaroro Island (Kerr & Grace 2006a, 2006b), from Mangawhai to Ahipara (Kerr 2010), Maunganui Marine Reserve (Kerr 2016), Waewaetorea Marine Reserve (Kerr & Grace 2015), and in the Hauraki Gulf at Waiheke Island (Kerr & Grace 2013). Mapping was carried out using a combination of techniques e.g., aerial photography, side scan sonar, SBES and MBES, ROV, videography, dredge sampling, scuba diving and snorkelling. *Ecklonia* forest, mixed algae, and rhodolith beds were identified. Habitat mapping was carried out in Hawkes Bay by side scan sonar and drop camera (Funnell et al. 2005) and on the Wellington south coast by drop camera (Byfield 2013). The northeast coast of Great Barrier Island was mapped using underwater video combined with bathymetry and substratum data to acquire baseline seabed habitat and their associated communities including shallow water macroalgae *Ecklonia radiata* and *Ulva* sp. (Lee et al. 2015). Aerial photographs, side-scan sonar, underwater video and a diver towed on a manta board, were used to map sea bed habitats and biotopes in the Cape Rodney to Okakari Point Marine Reserve, and these data were then compared with maps created about 30 years previously (Leleu et al. 2012).

Kibele & Shears (2016) developed a new method for the estimation of high spatial resolution bathymetry over large areas in a wide range of coastal environments. The method can be easily conducted, at low cost and which does not require extensive knowledge of optical remote sensing. This method was developed in the northeastern North Island, and expands "the already considerable value of high-resolution multispectral satellite imagery for marine applications, and could prove especially valuable as a source of depth data for water column correction to aid in the mapping of submerged habitats".

A non-destructive method was applied to the study and mapping of subtidal rocky bottom macroalgal assemblages on the coast of Cantabria (N Spain), with 31 transects spaced at about 2–3 km from each other along 150 km of coastline. Scuba divers recorded the total algal coverage and specific coverage of dominant macroalgae species, substrate and morphology, from the shore to 20–25 m depth. Underwater videos were acquired along the transect to allow further analysis. Interpolated species abundance values were used to representing the spatial distribution of the dominant species along the transect and a depth gradient (Guinda et al. 2012). The biomass and distribution of *Sargassum* beds along the coast of Baja California have been estimated by field collections, diving transects and compared with historical data. Temperature data (1988–2014) from satellite suggested that the biomass is inversely related to water temperature and *Sargassum* beds increased with latitude (Casas-Valdez et al. 2016).

13 APPENDIX 5. Summary of additional culture attempts

Unsuccessful flow-through cultures

Prior to the establishment of the static cultures, attempts were made to culture several species in a flow-through system. The system was designed to test the effects of shading and suspended sediment at two temperatures (i.e., in two different growth cabinets). Sumps containing 4 L of filtered seawater were placed in the bottom of each cabinet. Inside each sump, an aquarium pump was placed and connected to a manifold on the shelf above by silicon tubing. Each manifold had 6 outlet tubes connected to each of 6 jars per treatment. As lighting is delivered from the side of each cabinet, jars were randomised across the shelves to account for light differences. Jars were 400 mL with a drainage hole cut in the centre of the bottom to allow seawater to drain back to the sump via silicon tubing (Figure A).

Figure A. Flow through culture set up testing temperature, shading and suspended sediment.

Figure B

In the first experiment (October 2016) a single species (*Lessonia variegata*) was used. A spore slurry was created using the same method as outlined in Section 2.2.1 and settled onto 1 cm × 1 cm perspex tiles. Four tiles were placed into the bottom of each jar (Figure B) with the intention of removing one tile each week for measurement of sporelings. At each temperature, the jars provided 6 replicates per treatment (Figure C). Treatments were control (no shade or sediment), shade (jars placed inside a double layer of neutral density cloth), sediment (very fine sediment added to the sump) and shade and sediment (sediment added to sump and jar placed in shade bag). The cultures quickly became contaminated and the *Lessonia* sporelings were either smothered by sediment or killed by paramecia. Although the sediment used was intended to remain in suspension, some did settle onto the tiles. The experiment was abandoned after 1 week.

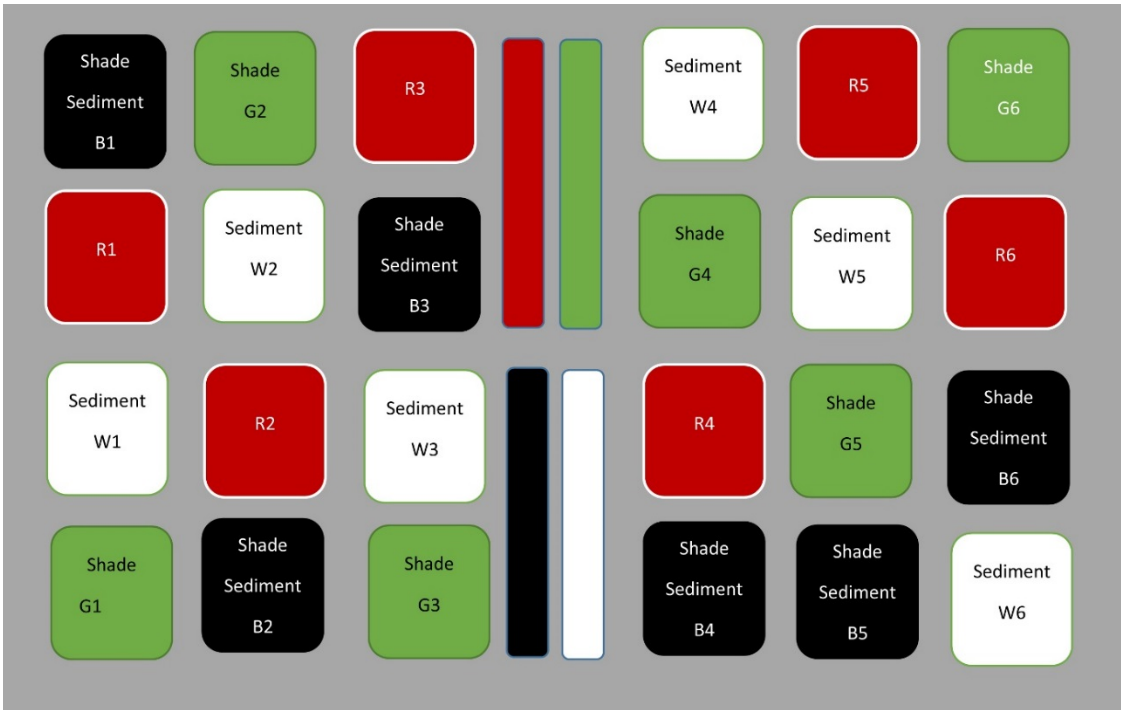


Figure C. *Lessonia variegata*: example of an experimental layout testing the effects of shade, sediment, and shade and sediment against control.

In the second experiment (December 2016) (Figure D), three species were used (*Carpophyllum maschalocarpum*, *Cystophora retroflexa* and *Cystophora torulosa* at two temperatures (15 ˚C and 18 ˚C). Fertilised eggs were settled onto glass tiles (1.5 cm × 1.5 cm) and 4 tiles were placed in the bottom of each jar. After one week some eggs had developed into germlings, but the majority had died and paramecium levels were very high. Cultures were discarded at this time.

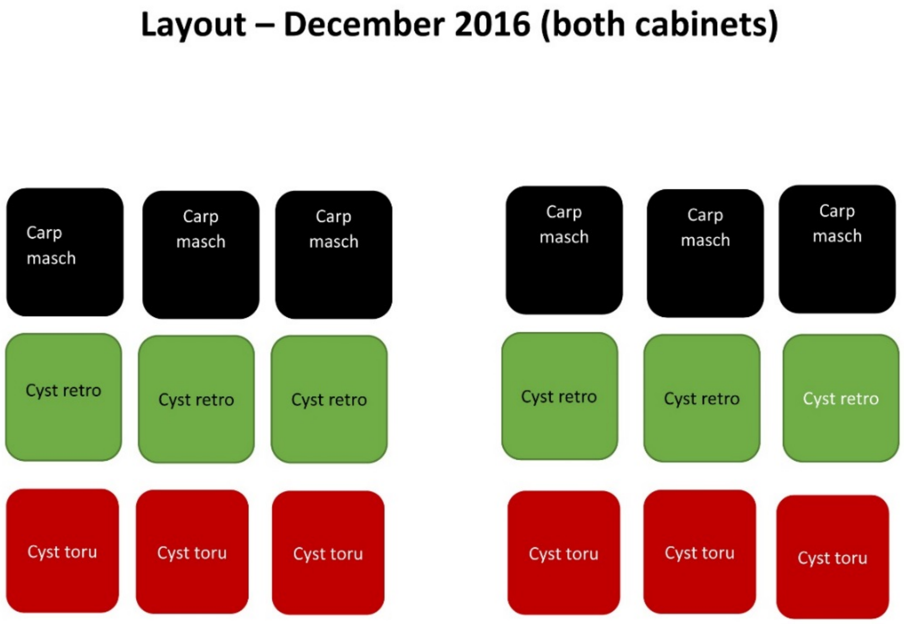


Figure D. *Carpophyllum* and *Cystophora* species: experimental layout in the flow-through system to test the effect of temperature.

Unsuccesful static cultures

In addition to the experiments reported on in Section 2.2, several other attempts were made to establish cultures. These attempts are detailed in Table A5.1.

Table A5.1. Details of unsuccessful static culture experiments. Some were unsuccessful due to the lack of gametes produced. Others were established but could not be maintained for the required period due to contamination.

Order	Species	Fertility period	Collected	Number of plants	Treated	Problem
Fucales	*Marginariella boryana*	Winter	10/05/2017	3	Freshwater rinse, FSW rinse, no immediate release, dried overnight, shaker table, cold FSW	No release
			16/05/2017	3	Rinsed, straight into cold FSW	Only sperm released
			18/05/2017	4	Freshwater rinse, FSW rinse, dried overnight, room temperature FSW	Only sperm released
			26/05/2017	7	Dried and chilled overnight, cold FSW	Good release, but all 18 ˚C germlings

					dead by 2 weeks, experiment discarded
Cystophora scalaris	Summer	9/2/2017	6	Freshwater rinse, dried 1 hour, cold FSW	Good release, majority died after 2 weeks
		7/11/2017	5	Dried for 4 hours, added FSW	Too contaminated
Cystophora torulosa	Summer	7/11/2017	4	Dried for 4 hours, added FSW	Too contaminated
		9/11/2017	3	Chilled overnight, dried for 3 hours, dried for 2 hours	Too contaminated

Ocean Acidification Experiments

Samples of *Lessonia variegata, Landsburgia quercifolia, Marginariella boryana,* and *Carpophyllum maschalocarpum* were incubated at ambient and low pH in the ocean acidification facility and kept at temperature of 12˚ C for about 3 months. Due the low number of replicates results are not included.

14 APPENDIX 6. ANOVA tables and post-hoc testing results for culture experiments

Lessonia variegata

Two-way ANOVA testing the effects of temperature (12 and 18 ˚C) and shading (shade or no shade) at day 37.

Source of Variation	**DF**	**SS**	**MS**	**F**	**P**
Temperature	1	1248.333	1248.333	5.514	0.021
Shading	1	84.739	84.739	0.374	0.542
Temp x Shade	1	485.777	485.777	2.146	0.146
Residual	116	26259.457	226.375		
Total	119	28078.307	235.952		

Landsburgia quercifolia

Two-way ANOVA of transformed data testing the effects of temperature (12 and 18 ˚C) and shading (shade or no shade) at day 21.

Source of Variation	**DF**	**SS**	**MS**	**F**	**P**
Temperature	1	0.227	0.227	12.860	0.001
Shading	1	0.0866	0.0866	4.906	0.028
Temp x Shade	1	0.207	0.207	11.714	0.001
Residual	216	3.811	0.0176		
Total	219	4.321	0.0197		

The effect of different levels of Temperature depends on what level of Shading is present. There is a statistically significant interaction between Temp and Shade. (P = 0.001)

All Pairwise Multiple Comparison Procedures (Holm-Sidak method):
Overall significance level = 0.05

Comparisons for factor: **Shade within 12**

Comparison	**Diff of Means**	**t**	**P**	**P 0.05**
S vs. NS	0.0219	0.809	0.419	No

Comparisons for factor: **Shade within 18**

Comparison	**Diff of Means**	**t**	**P**	**P 0.05**
NS vs. S	0.102	4.234	0.001	Yes

Comparisons for factor: **Temp within NS**

Comparison	**Diff of Means**	**t**	**P**	**P 0.05**
18.000 vs. 12.000	0.127	4.738	0.001	Yes

Comparisons for factor: **Temp within S**

Comparison	**Diff of Means**	**t**	**P**	**P 0.05**
18.000 vs. 12.000	0.00296	0.121	0.903	No

Three-way ANOVA of transformed data testing the effects of temperature (12 and 18 ˚C), shading (shade or no shade) and sediment (sediment or no sediment) on *Landsburgia* germling length at day 78.

Source of Variation	**DF**	**SS**	**MS**	**F**	**P**
Shade	1	6528.774	6528.774	407.068	0.001
Sed	1	3952.551	3952.551	246.441	0.001

Temp	1	59.179	59.179	3.690	0.056
Shade x Sed	1	306.970	306.970	19.140	0.001
Shade x Temp	1	219.325	219.325	13.675	0.001
Sed x Temp	1	1.022	1.022	0.0637	0.801
Shade x Sed x Temp	1	8.710	8.710	0.543	0.462
Residual	232	3720.942	16.039		
Total	239	14797.474	61.914		

The main effects for Shade cannot be properly interpreted since the size of the factor's effect depends upon the level of another factor.

The main effects for Sed cannot be properly interpreted since the size of the factor's effect depends upon the level of another factor.

The main effects for Temp cannot be properly interpreted since the size of the factor's effect depends upon the level of another factor.

The effect of different levels of Shade depends on what level of Sed is present. There is a statistically significant interaction between Shade and Sed. (P = 0.001)

The effect of different levels of Shade depends on what level of Temp is present. There is a statistically significant interaction between Shade and Temp. (P = 0.001)

The effect of different levels of Sed does not depend on what level of Temp is present. There is not a statistically significant interaction between Sed and Temp. (P = 0.801)

All Pairwise Multiple Comparison Procedures (Holm-Sidak method):
Overall significance level = 0.05

Comparisons for factor: **Sed within NS**

Comparison	**Diff of Means**	**t**	**P**	**P 0.05**
NoSed vs. Sed	5.854	8.007	0.001	Yes

Comparisons for factor: **Sed within S**

Comparison	**Diff of Means**	**t**	**P**	**P 0.05**
NoSed vs. Sed	10.378	14.194	0.001	Yes

Comparisons for factor: **Shade within NoSed**

Comparison	**Diff of Means**	**t**	**P**	**P 0.05**
NS vs. S	8.169	11.173	0.001	Yes

Comparisons for factor: **Shade within Sed**

Comparison	**Diff of Means**	**t**	**P**	**P 0.05**
NS vs. S	12.693	17.360	0.001	Yes

Comparisons for factor: **Temp within NS**

Comparison	**Diff of Means**	**t**	**P**	**P 0.05**
18.000 vs. 12.000	0.919	1.257	0.210	No

Comparisons for factor: **Temp within S**

Comparison	**Diff of Means**	**t**	**P**	**P 0.05**
12.000 vs. 18.000	2.905	3.973	0.001	Yes

Comparisons for factor: **Shade within 12**

Comparison	**Diff of Means**	**t**	**P**	**P 0.05**
NS vs. S	8.519	11.652	0.001	Yes

Comparisons for factor: **Shade within 18**

Comparison	**Diff of Means**	**t**	**P**	**P 0.05**
NS vs. S	12.343	16.881	0.001	Yes

Marginariella boryana

One-way ANOVA of transformed data testing the effects of shading (shade or no shade) at 12 ˚C at day 46.

Source of Variation	**DF**	**SS**	**MS**	**F**	**P**
Between Groups	1	0.667	0.667	22.771	0.001
Residual	37	1.084	0.0293		
Total	38	1.751			

Carpophyllum maschalocarpum

Two-way ANOVA of transformed data testing the effects of temperature (18 and 22 ˚C) and shading (shade or no shade) at day 40.

Source of Variation	**DF**	**SS**	**MS**	**F**	**P**
Temp	1	0.109	0.109	3.146	0.080
Shade	1	0.186	0.186	5.352	0.023
Temp x Shade	1	0.0114	0.0114	0.328	0.568
Residual	81	2.814	0.0347		
Total	84	3.250	0.0387		

Cystophora scalaris

Two-way ANOVA of transformed data testing the effects of temperature (18 and 22 ˚C) and shading (shade or no shade) at day 40.

Source of Variation	**DF**	**SS**	**MS**	**F**	**P**
Temp	1	1.407	1.407	173.641	0.001
Shade	1	0.0385	0.0385	4.747	0.030
Temp x Shade	1	0.0158	0.0158	1.956	0.163
Residual	316	2.560	0.00810		
Total	319	4.033	0.0126		

15 APPENDIX 7. Satellite Imagery SPOT maps 2014

Type: Natural Color (RGB)
Resolution: 1.5m
Capture data range: 26 October 2012 to 01 April 2014
Projection: NZTM2000
File format: 450 x geo TIFF (1:50,000 tiles)
Licensing: NZ Organisations – Airbus DS
Access: email Airbus Defence and Space (Australia)
More Information: Metadata Footprints (MfE data service)

Wellington South Coast – from Point Dorset (right) to Arthur Point (left) © Airbus DS (2014).

Detail of Moa Point (© Airbus DS -2014).

Detail of Point Dorset – possible *Macrocystis* bed on the right side (© Airbus DS -2014).

16 APPENDIX 8. Machine Learning and computer vision techniques

Metadata extraction from video stream

Information about the location of the images was printed on the video stream, and normally would need to be transcribed manually so that the image could be located geographically. We investigated the potential of artificial intelligence (AI) to develop a faster and more reliable method to get information out of this printed text. A tool was designed to extract hard coded metadata from the video stream and return the corresponding string, using machine learning and computer vision techniques.

Binary Image

The first step in extracting a string from the image (Figure A8.1) was to discard the colour information and to get a binary image of the characters against the background.

Figure A8.1. Image of GPS data to extract.

To achieve this, a method of sorting the foreground text pixels from the background pixels was needed. A graph showing the RGB (red, green, blue) composition of each pixel in the image (Figure A8.2) was rendered, with two clusters for each category.

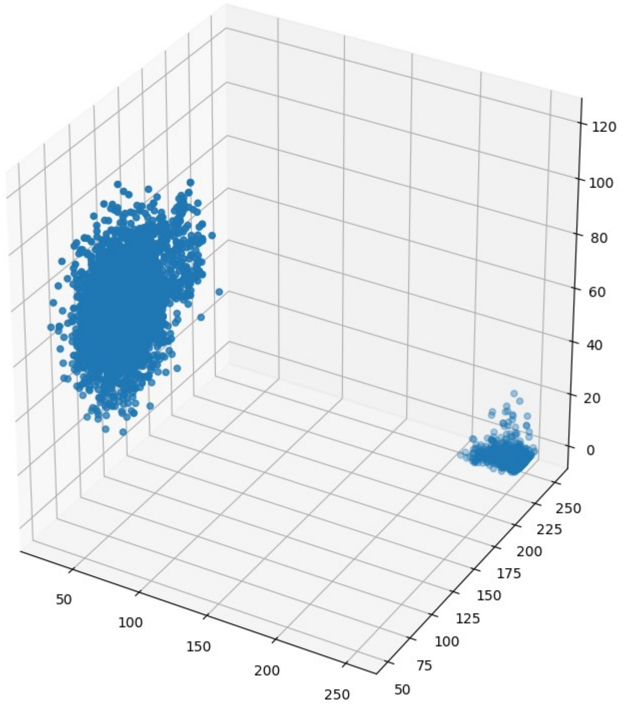


Figure A8.2. 3D graph representing pixel values for Red Green Blue (RGB).

The method used to differentiate these two clusters was a method of unsupervised machine learning known as K-means clustering. This method classifies each data point (pixel) into a predetermined number of classes (2). To do this, an iterative process is used, with the Euclidean distance between a pixel and the current cluster centroid calculated and then added to the closest one. This method requires the calculation to be redone for each image, therefore the difference between frames should not affect optimally separating the pixels.

The final result of this is a binary image as shown in Figure A8.3.

Figure A8.3. Binary image of GPS data.

Character Extraction

The next step was to extract the individual characters into a list. To do this, components that have at least one neighbouring pixel of the same colour were grouped together. This is a form of image segmentation known as connected components analysis.

The result of extracting the individual characters is shown in Figure A8.4.

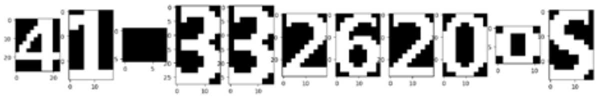


Figure A8.4. Ordered set of individual characters.

Character Identification

The final step was to identify an individual character. This was done by calculating the ratio of the intersection to union of pixels in an image. To do this, the extracted character was placed on a 50x50 matrix along with a template for a possible character. An element wise ’and’ (intersection) and ’or’ (union) is then calculated for the two matrices. This is done for all possible characters and the highest ratio is used to determine the character present.

The example in Figure A8.5 shows this process with the character ’1’ against ’6’ (template). The ratio for this was 0.274 as shown by the white (intersection) to grey(union).

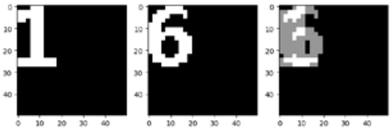


Figure A8.5. Intersection to Union ratio of 1 and 6.

The example in Figure A8.6 shows this process with the character ’1’ against ’1’ (template). The ratio for this was 1.0 as shown by the white (intersection) to grey (union).

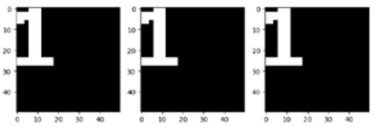


Figure A8.6. Intersection to Union ratio of 1 and 1.

Development of a machine learning classifier:

A machine learning classifier was developed to: test the use of artificial intelligence for algal identification and recognise the dominant algal species along the video transect, *Ecklonia*, *Lessonia* and *Carpophyllum* spp.; automate the analytical workload using current machine learning and computer vision techniques; test the effectiveness of the approaches; perform tests of the impact of colour vs monochromatic imagery on the AI classification system.

Image Classification Method:
The starting point for this problem begins with a 2-dimensional array containing the pixel data for an image. Typically, a convolutional neural network (CNN) would be trained to learn useful visual features (textures, shapes, edges, colours) contained in the pixel data that are useful in identifying the target class. Convolutional neural networks are a type of neural network that work exceptionally well at learning spatial features. Inspired from the animal visual cortex, recent breakthroughs in image processing have been achieved using this method. The drawback with a CNN is that learning these visual features can require upward of 10^4 images. Therefore, a different approach is needed.

This approach is known as transfer learning. Transfer learning in this case will involve using the CNN trained from another dataset as a tool to extract visual features from our images. The CNN used for this is Inception Resnet V2 trained using the ImageNet dataset. ImageNet is an image database containing over 10^7 images with 10^4 labelled classes (species of animals, types of vehicles, miscellaneous objects, etc). Inception Resnet V2 is a neural network published by Google that achieved one of the strongest attempts at classifying this dataset (80.4 % top 1 accuracy and 95.3 % top 5 accuracy). The underlying assumption for using this CNN is that in the vast and diverse number of visual features learned there will be a subsection useful to classifying algae. Therefore, from this step we will receive a 10^5 length array containing numerical confidence levels for every visual feature in Inception Resnet.

Once these visual features have been extracted from the image via Inception Resnet, we can use these to train a custom classifier with a substantially smaller dataset (10^2 to 10^4). The classifier that will be used to classify based on these visual features is known as a linear support vector machine (SVM). This classifier type achieved the highest accuracy when tested at the current data set sizes. This type of SVM works by treating each visual feature value as a dimension therefore making an image a datapoint on a very high dimensional space graph. A hyperplane is then optimally placed to separate these data points based on whether it is a positive or negative label. To demonstrate this (Figure A8.7) shows how a line is used to separate two clusters of labelled data in 2-dimensional space. This is essentially the same in 10^5 space but a little harder to visualize. Once this hyperplane has been calculated each new visual feature array generated from Inception Resnet can be classified simply by determining which side of the hyperplane it lands on.

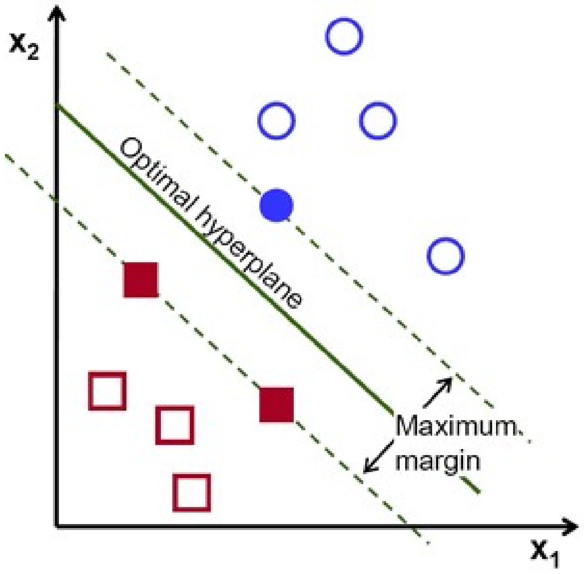


Figure A8.7. Hyperplane generated from a linear support vector machine.

Training an AI Classifier - Requirements to implement machine learning

The machine learning used for this task is as much a branch of artificial intelligence as it is data science. This means that two separate problems had to be solved to create an accurate algae classifier:
1) creating large datasets of images that contain both positive and negative examples to be used for training,
2) creating an AI that learns from said training dataset (from both positive and negative results) then extrapolates to classify new images.
This project focused on designing an AI that will give a strong learning performance for a given training dataset, resulting in accurate predictions of future images. To supply training data, basic datasets were labelled to train the initial classifiers, although much stronger datasets will be created as more video footage is collected and are labeled more accurately.

The first step is to extract the visual features from the training images. To do this, the convolution neural network (CNN) section of Inception Resnet was loaded, and used to extract visual features (textures, shapes, edges, colors). Next a generator was created to load training images from a directory and transform them into an appropriate format for the CNN.
The process of creating the visual feature vectors (known as the bottleneck) was then initiated and saved for the next step. The true class (positive or negative) of each image was also saved as labels for the next step.

Listing 1: Loading Inception Resnet CNN

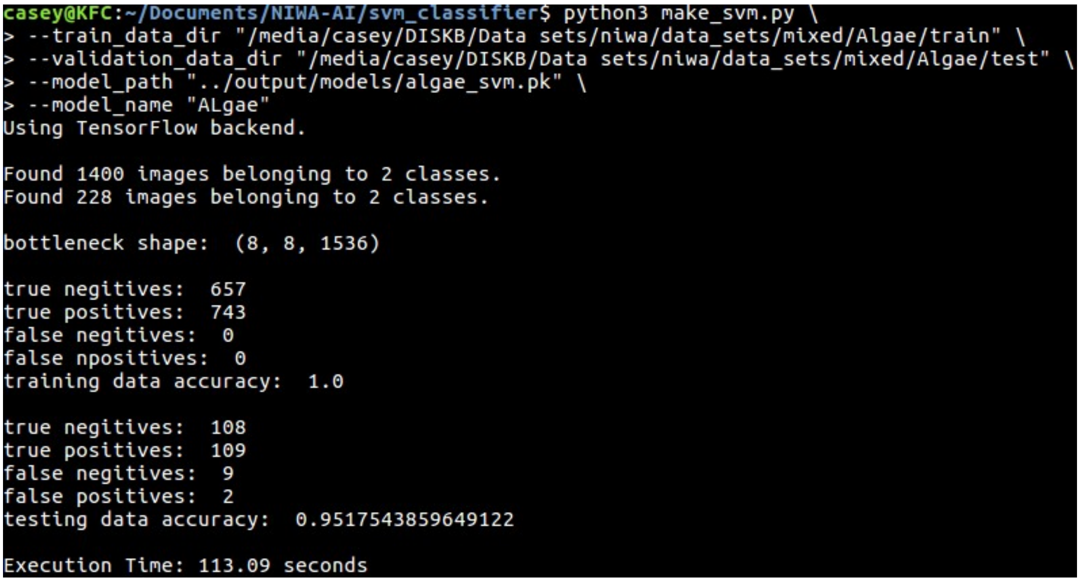
Listing 2: Image Generato

Listing 3: Saving feature vectors

At this point each training image was represented by vectors containing confidence levels for the presence of each visual feature in the image. To train a classifier these vectors and corresponding classes were fed into a support vector machine (SVM) (lst.4), and this learned which of these features contribute to a species of algae being present or not.

Listing 4: Training support vector machine

To ensure that the classifier will be able to classify new images, a set of visual feature vectors not in the training dataset were classified. This is known as the testing/validation set and was created parallel to the training vectors. The results of this were compared to the true classes and accuracy metrics were output to the console in the following format (Figure A8.8).



```
casey@KFC:~/Documents/NIWA-AI/svm_classifier$ python3 make_svm.py \
> --train_data_dir "/media/casey/DISKB/Data sets/niwa/data_sets/mixed/Algae/train" \
> --validation_data_dir "/media/casey/DISKB/Data sets/niwa/data_sets/mixed/Algae/test" \
> --model_path "../output/models/algae_svm.pk" \
> --model_name "ALgae"
Using TensorFlow backend.

Found 1400 images belonging to 2 classes.
Found 228 images belonging to 2 classes.

bottleneck shape:  (8, 8, 1536)

true negitives:  657
true positives:  743
false negitives:  0
false npositives:  0
training data accuracy:  1.0

true negitives:  108
true positives:  109
false negitives:  9
false positives:  2
testing data accuracy:  0.9517543859649122

Execution Time: 113.09 seconds
```

Figure A8.8. Classifier accuracy results.

The new classifier was saved as a file, to be loaded to classify new images in the future.

Classifying Video Footage

This covers the process of using an AI classifier to determine the species of algae present in the video footage.

The first step is to load the support vector machines trained in the previous step (lst.5).

Listing 5: Loading support vector machines

The SVM classifiers require visual feature vectors as input, therefore Inception Resnet is used to extract these features. At this point, the classification process takes place where every frame was processed and given a prediction on whether the class was present or not (lst.6). Every frame will have a 1 or 0 result for each class (Algae, *Ecklonia, Lessonia, Carpophyllum*).

Listing 6: Classifying images

For the first post-processing step, a moving average was applied to create a confidence rating by averaging the results from the last 12 frames (lst.7). This increased accuracy by using information in the time domain. Each result now represented the half a second of footage as opposed to a single frame. The confidence level can take fractional values and was thus more representative than a flat 0 or 1 classification.

Listing 7: Applying the moving average

The final step in classifying the video footage was to multiply each species class by the Algae class (lst.8). The design choice behind this was that the species classifiers are only trained to determine what species of algae were present given an image of algae. Since it is not trained to work with an image of no algae the results will be unpredictable. The Algae classifier on the other hand has the sole purpose of determining whether algae was present in the frame. Therefore, if the Algae classifier returns a 0 all results from the species classifiers get suppressed. This in turn minimized false positive classifications.

Listing 8: Species classifier suppression

Methods and Results of testing colour dependence are presented in Appendix 9.

Accuracy measurements for the classifiers

The following tests were conducted using the image data sets for this project. The images were frames from the video footage labelled as either positive or negative for the species of interest. Each dataset was also split into testing or training. The training dataset was used to create the classifier and the testing dataset was used to determine how well new images were classified.

To ensure that there was no overlap between the training and testing datasets, videos were dedicated to either training or testing. In the case of the Palmer Head sampling, videos 5,6,7,15 were used for the testing datasets. As there were no random processes in these tests, all results can be recreated using the corresponding datasets. The Results for this test are given in Table A8.1. Note that the ratio for random guessing would be 1.

Table A8.1. Algae classifier: determining whether Algae (the class of interest) are present or not.

Dataset used to train classifier	Size of training dataset	Dataset used to test classifier	Size of testing dataset	Accuracy achieved by classifier (%)	True negatives	True positives	False negatives	False positives	Ratio (correct to incorrect)
Moa Point	647	Moa Point	138	88.4	67	55	16	0	7.62
Palmer Head	753	Palmer Head	90	94.4	38	47	0	5	17

From these results we can tell that the classifier is learning how to classify images based on the visual features extracted. With these levels of accuracy there will be some mistakes when comparing directly to the video footage, but it may be enough to create a rough density map of the area and identify trends in said density over time. When the same experiment is repeated but with combining the training datasets together to train the classifier, the following results (Table A8.2) were obtained:

Table A8.2. Algae classifier: determining whether Algae (the class of interest) are present or not using a larger training dataset. MP = Moa Point and PH = Palmer Head.

Dataset used to train classifier	Size of training dataset	Dataset used to test classifier	Size of testing dataset	Accuracy achieved by classifier (%)	True negatives	True positives	False negatives	False positives	Ratio (correct to incorrect)
MP & PH	1400	MP	138	92.7%	66	62	9	1	12.7
MP & PH	1400	PH	90	98.8%	42	47	0	1	89

What these results show is that increasing the number of training images resulted in a far more accurate classifier. This indicates that there is likely to be much more performance to be gained simply by adding images and increasing diversity in the training dataset.

Another observation from this is the 5-fold increase in accuracy when classifying the Palmer Head testing dataset. As shown the Moa Point datasets result in a weaker performance, possibly due to turbidity. Regardless, adding the Moa Point training data greatly improved accuracy when classifying Palmer Head, counter to what was initially expected. This means that the quality of the training images used is less important than the quantity used, especially at the size of the datasets used here.

Species Classifier: determining which species of algae are present

As it is common to have more than one species present in a frame, all testing/training images had from 1 to 3 species present. This ensured that each classifier had to learn to classify the target species while ignoring other species in order to increase accuracy.

Table A8.3. Algae classifier: determining whether algae genera are present or not. MP = Moa Point and PH = Palmer Head.

Genus of interest	Dataset used to train classifier	Size of training dataset	Dataset used to test classifier	Size of testing dataset	Accuracy achieved by classifier	True negatives	True positives	False negatives	False positives	Ratio (correct to incorrect)
Ecklonia	MP & PH	836	MP and PH	100	89 %	45	46	6	5	8.1
Lessonia	MP & PH	797	MP and PH	100	85 %	41	44	6	9	5.7
Carpophyllum	MP & PH	807	MP and PH	103	86.4 %	44	45	6	8	6.4

As shown in the results above (Table A.8.3) the classifiers were learning how to distinguish different species of algae based on visual features. In this current state there are still misclassifications that were apparent when watching the video, but there were also scenes where the classifier has picked up algae that were missed with initial visual classification (such as undergrowth). As mentioned previously, if this information was averaged to create a density map, the results should be accurate enough to determine trends over time. Overall these were promising results that indicate that current AI and computer vision tools were capable of solving this problem.

The resulting video should look like Figure A8.9.

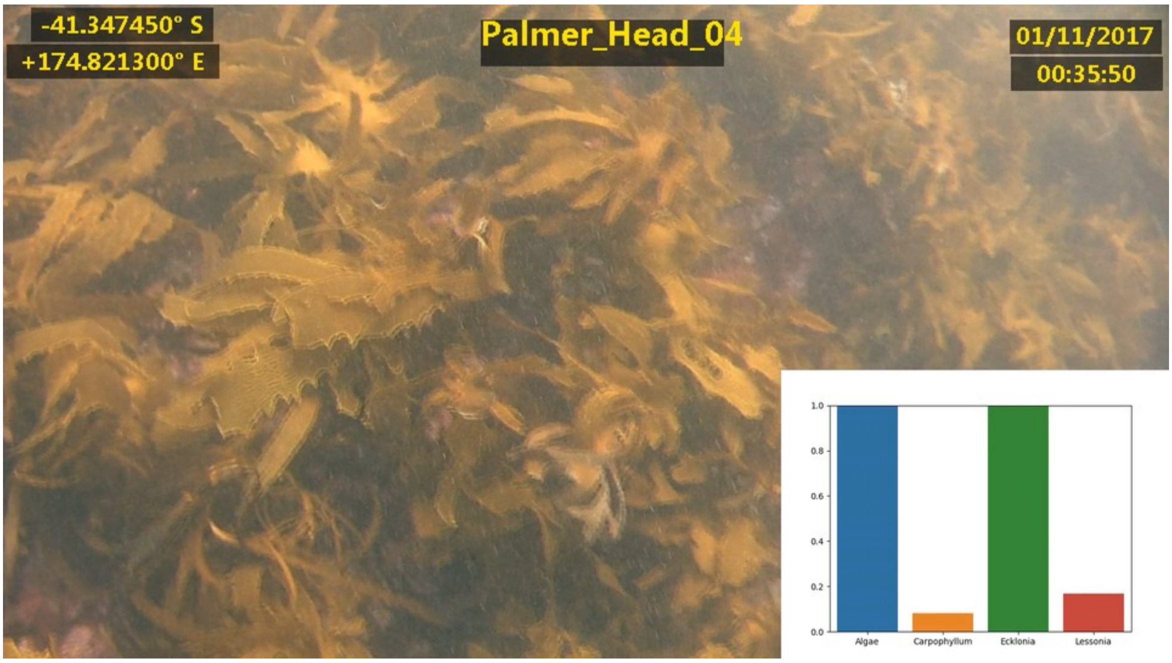


Figure A8.9. Snapshot of new video with graph overlay.

Testing the effectiveness of the approaches

Breaker Bay Testing

A number of tests were conducted using video footage from the Breaker Bay area. The objective was to determine what levels of accuracy can be expected from the current AI system and, if possible, to maximize this, as well as testing what works well or doesn’t work.

For these tests the AI had the task of differentiating *Ecklonia* from other types of algae. All images contained algae, as differentiating algae from rock or sand is an easier task. Rather than testing all species, more detailed testing was performed on a single species, based on the concept that the various insights from these tests should apply to other image classification tasks (such as other species).

Identifying whether algae were present or not showed similar traits but with a higher accuracy (see the AUV dataset test).

Splash cam and GoPro

The first test was to determine whether the AI classifier could accurately classify the GoPro footage, using frames from the splash cam (Moa Point and Palmer Head) as training data.

The GoPro videos used for the testing were ”GOPRO0007”, ”GOPRO0011”, and GOPRO0012”. The results of this for each video were as follows (Table A8.4).

Table A8.4. Splash cam and GoPro Accuracy results.

	GOPRO0007	GOPRO0011	GOPRO0012	Average
Num train	836	836	836	836
Num test	338	845	355	479.3
True negatives	111	277	102	163.3
True positives	123	196	101	140
False negatives	53	318	72	147.6
False positives	51	54	80	61.7
Accuracy	69.2 %	56 %	57.2%	60.8%

The conclusion from this experiment was that the AI perceived significant differences in photography between the splash cam and GoPro footage, which resulted in the severe degradation of performance observed.

A later test "Breaker Bay Splash Cam" shows that the AI was far more successful classifying the splash cam footage from Breaker Bay. This also indicates that the splash cam and GoPro images are not currently compatible.

GoPro Only 1

As shown in the previous test using both splash cam and GoPro footage together did not work. Therefore, the next set of tests were conducted using only the Breaker Bay GoPro footage. This test consisted of two of the GOPRO videos being used for training data and the third for testing accuracy.

For example the test named "GOPRO0011" used "GOPRO0007" and "GO- PRO0012" as training data and "GOPRO0011" was to be tested. The results were as follows (Table A8.5).

Table A8.5. GoPro Only 1 Accuracy results.

	GOPRO0007	GOPRO0011	GOPRO0012	Average
Num train	1200	693	1183	1025.3
Num test	338	845	355	512.7
True negatives	83	245	42	123.3
True positives	124	294	159	192.3
False negatives	52	220	14	95.3
False positives	79	86	140	101.7
Accuracy	61.2 %	63.8 %	56.6 %	60.5 %

The results from this test indicate that the AI classifier was not effective in this test. This may be due to the range of training images from two videos not being diverse enough for the AI to apply what it learned to a new video. If this is true (as opposed to a mislabelled data set) the accuracy should increase as the difference between training and testing dataset decreases.

GoPro Only 2

The next test was to train an AI classifier on two thirds of a video and use this to this to classify the other third of the video. This objective is to determine whether performance increases when the testing data are a part of the same video. This test involved the labelled *Ecklonia* images of "GOPRO0011" with the naming convention of "Test 2" meaning using thirds 1 and 3 for training and 2 for testing. The accuracy results of this were as follows (Table A8.6).

Table A8.6. GoPro Only 2 Accuracy results.

	Test 1	Test 2	Test 3	Average
Num train	564	563	564	563.7
Num test	281	282	281	281.3

True negatives	33	99	67	66.3
True positives	156	119	163	146
False negatives	15	52	8	25
False positives	77	12	43	44
Accuracy	67.3 %	77.3 %	81.8 %	75.5 %

As shown in these results the average classification accuracy increased from 60.5 % to 75.5 %. As explored further in the ”Go Pro Only 3” test this was most likely due to the testing data being closer to the training data.

GoPro Only 3

The final Breaker Bay GoPro test was to generate a random subset of labelled *Ecklonia* frames for training and use this to classify the remaining frames. This was done using the same image set as ”Go Pro Only 2”. Each image was given an 80 % chance of being used for training and 20 % for testing. This was repeated 3 times and gave the following results (Table A8.7).

Table A8.7. GoPro Only 3 Accuracy results.

	Test 1	**Test 2**	**Test 3**	**Average**
Num train	686	677	692	685
Num test	159	178	153	163.3
True negatives	64	62	49	58.3
True positives	80	90	85	85
False negatives	7	11	9	9
False positives	8	15	10	11
Accuracy	90.5 %	85.4 %	87.6 %	87.8 %

As shown in the results, the average accuracy of the testing dataset increased from 75.5 % to 87.8 % (Table A8.7). The conclusion is that the closer the training data set represents the testing data set the higher the accuracy. This makes sense as the AI classifier optimally separates the training data into positive and negative clusters based on their visual features. Therefore, if the testing data is close to the training data in terms of visual features it is more likely to be placed in the correct cluster.

It should also be noted that both the ”Go Pro Only 2” and ”Go Pro Only 3” tests both had a similar amount of training data while the ”Go Pro Only 3” tests resulted in significantly more accurate results. Therefore, as labelling training data is expensive, planning how training data will be labelled is important in efficiently creating an effective AI classifier.

Breaker Bay Splash Cam

The final test measured the accuracy of the Breaker Bay splash cam footage using the Moa Point and Palmer Head splash cam footage as training data. The purpose of this was to determine how well the AI classifier could extrapolate the training data to a new area on the Wellington coast.

The following results (Table A8.8) are a selection of frames from the following Breaker Bay videos, ”NINJABLD S001 S001 T005”, ”NINJABLD S001 S001 T006”, ”NIN- JABLD S001 S001 T007”.

Table A8.8. Breaker Bay splash cam Accuracy results.

	005	006	007	Average
Num train	836	836	836	836
Num test	62	62	47	57
True negatives	28	23	26	25.7
True positives	23	14	9	15.3
False negatives	10	13	11	11.3
False positives	1	12	1	4.7
Accuracy	82.3 %	59.7 %	74.5 %	72.2 %

A few interesting points can be made from these results. The first would be that the Moa Point and Palmer Head data could be used with moderate success to classify another part of the Wellington coast. It is interesting to compare to the ”Go Pro Only” tests where the AI struggled with a new video in the same area. Both these training data sets had a similar number of images, therefore the explanation for the difference is due to the diversity in training data. This data set contained images from about 25 videos over 2 areas compared to all from one video. Therefore, this test indicates that diversity in data is important to creating a robust or general model for classifying. Also, it can be speculated that this trend would continue as the training dataset becomes bigger and more diverse.

DOC GoPro Testing

To see whether it would be possible to classify algae species within video taken from other regions, a video was supplied by the Department of Conservation (DOC) with imagery from Port Pegasus, Stewart Island. From this DOC GoPro footage videos were generated with a graph overlay to view the results. The observation from these is that there are times when it works but with no real consistency (Figures A8.10 and A8.11), and at least not enough to give useful results. It should be possible to create a classifier that gives an accurate analysis with the right training data set, however, the differences between the NIWA and DOC footage are significant, and thus it is not expected that the NIWA datasets can be used to train such a classifier.

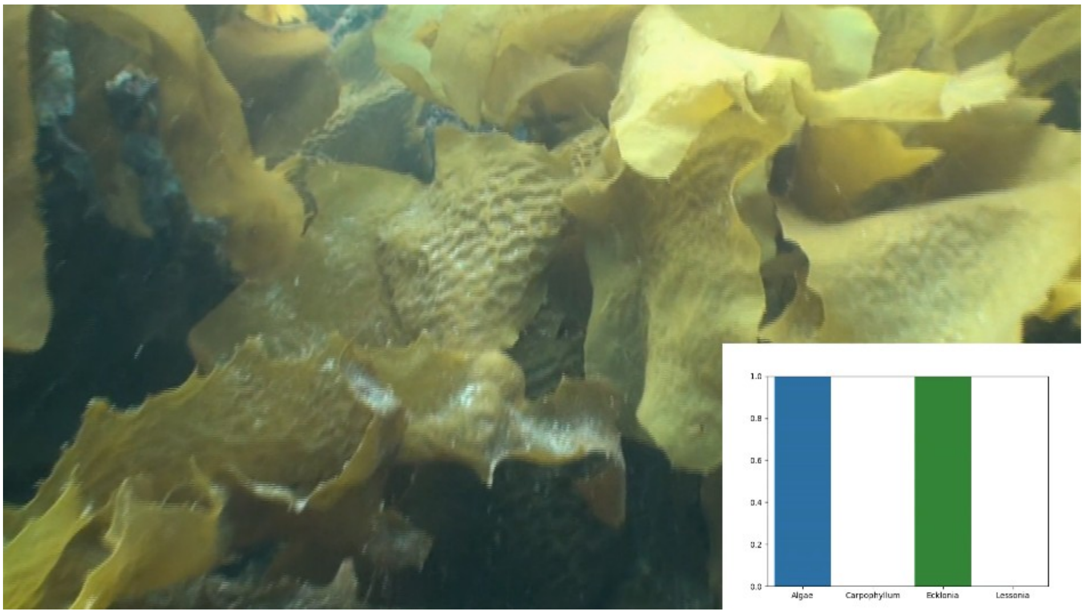


Figure A8.10. Example of working case.

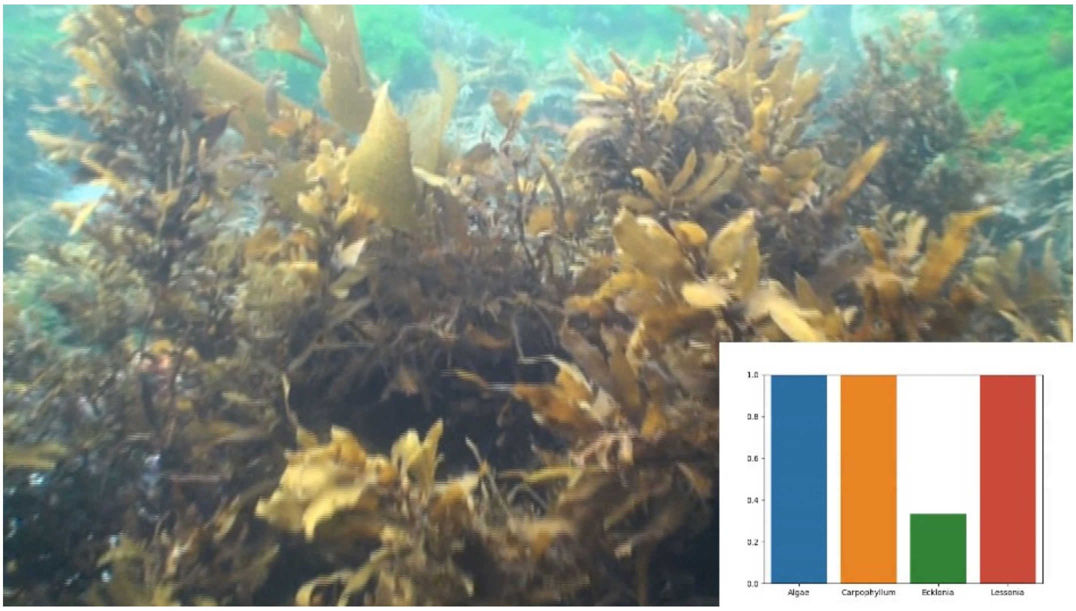


Figure A8.11. Example of not working case.

AUV Dataset

One of the most important parts of creating effective machine learning methods is having a large database of accurately labelled data. Therefore, one approach to increasing performance was to use large amounts of labelled images from the BENTHOZ-2015 dataset (Bewley et al. 2016) to supplement the training process. Unfortunately, this did not end up being useful for supplementing data sets.

The Australian benthic data set (BENTHOZ-2015) consists of about 10 000 images, consisting of 407 968 expert labelled points. Each image is fully labelled with all relevant information over several minutes by a marine scientist. This data set could be extremely useful in further investigation of AI for marine applications. While not being directly applicable to any problem, an extensive database of fully labelled images would mean that future developers would be able to quickly test various ideas and methods beyond what would be possible creating custom data sets due to time and cost restraints. It is also worth noting that each label contains positional information, therefore it could be possible to develop an AI that locates organisms within an image if there was interest in doing so.

Training dataset size test

Creating training datasets containing various percentages of the total images were used to test the accuracy of classifying the rest of the images. The purpose of the test was to determine how using sparse examples throughout an image set for training influences AI classifier accuracy for the entire image set.

To create the dataset each image is given a set percentage of being a part of the training data, therefore the training images should be randomly distributed throughout the BENTHOZ-2015 database. Ideally the testing dataset would be the remaining images, but due to memory constraints 10 % of the remaining images were randomly selected for classification. Table A8.9 shows the results of classifying between 0 and 10+ brown algae tags in an image. Table A.8.10 shows the results of classifying between 0 and 1+ brown algae tags in an image and Figure A8.12 shows a comparison between the two.

Table A8.9. Accuracy of results of classification between 0 and 10+ algae tags.

	2 %	5 %	10 %	20 %
Num train	162	450	843	1723
Num test	899	845	891	839
True negatives	632	625	663	626
True positives	175	168	186	188
False negatives	52	41	23	13
False positives	40	11	19	12
Accuracy	89.7 %	93.8 %	95.3 %	97.0 %

Table A8.10. Accuracy of results of classification between 0 and 1+ algae tags.

	2 %	5 %	10 %	20 %
Num train	211	486	982	1949
Num test	1023	979	975	654
True negatives	636	604	606	647
True positives	220	255	256	267
False negatives	110	60	53	57
False positives	57	60	52	39
Accuracy	83.7 %	87.7 %	89.2 %	90.5 %

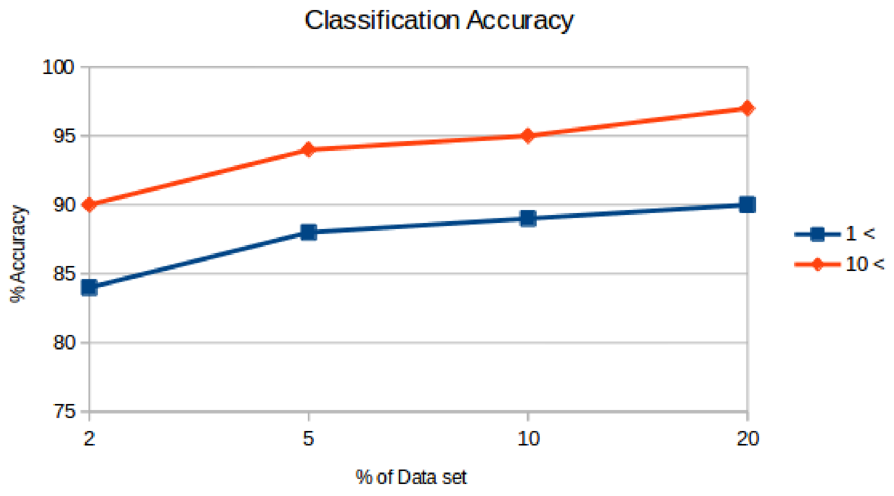


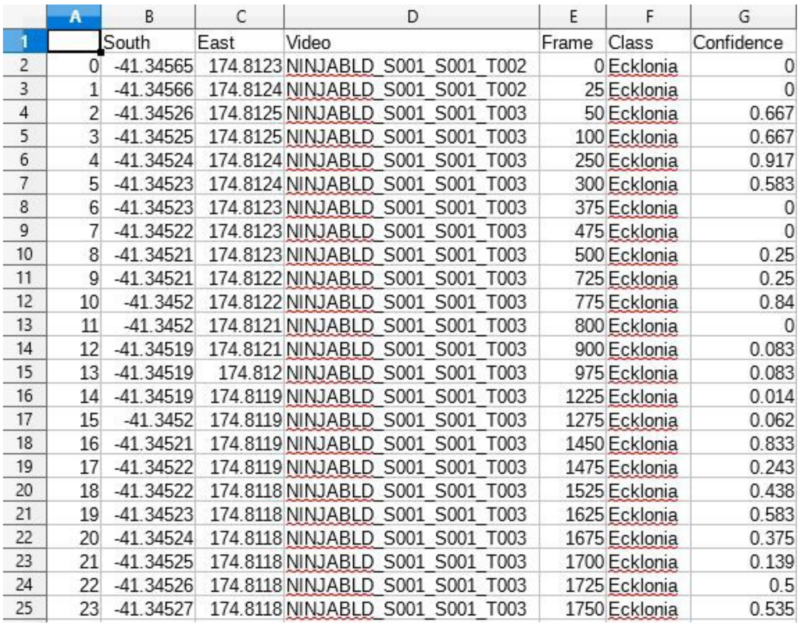
Figure A8.12. Graph of classification accuracy.

The conclusion from these results is that having a sparse training dataset can result in high accuracy if it is representative of the data to be classified, even in training sets with few examples. This indicates that having training data that represents the problem is more important than the absolute quantity of images.

Development of a map from the data acquired: examining the design choices for processing the raw data (AI ratings and video) into a form that can be easily used with QGIS (a free and Open Source Geographic Information System), to map macroalgae beds.

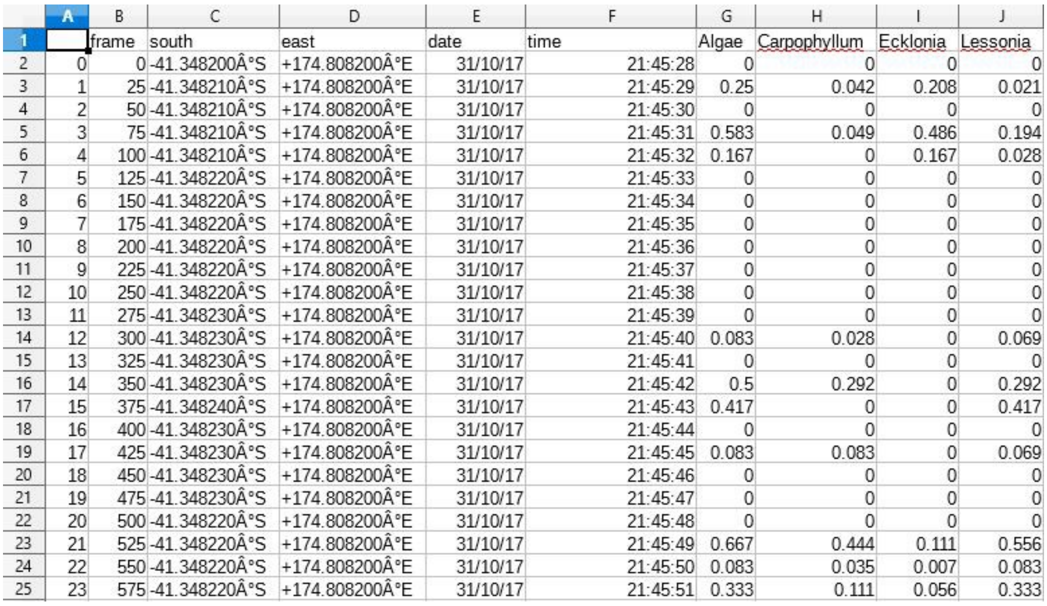
Data processing

The first step of using the mapping system was to transform the algae classification data, in order that it gives meaningful insight when viewed on QGIS. It is important to find the highest classification rating for each coordinate on the map. This was then used as the rating for that coordinate and the rest were ignored. The purpose of this was to reduce the bias towards areas with a high number of ratings per coordinate. The data represent a rating for each coordinate (Figure A8.13), and the raw data represent a rating for every 25th frame (Figure A8.14).



	A	B	C	D	E	F	G
1		South	East	Video	Frame	Class	Confidence
2	0	-41.34565	174.8123	NINJABLD_S001_S001_T002	0	Ecklonia	0
3	1	-41.34566	174.8124	NINJABLD_S001_S001_T002	25	Ecklonia	0
4	2	-41.34526	174.8125	NINJABLD_S001_S001_T003	50	Ecklonia	0.667
5	3	-41.34525	174.8125	NINJABLD_S001_S001_T003	100	Ecklonia	0.667
6	4	-41.34524	174.8124	NINJABLD_S001_S001_T003	250	Ecklonia	0.917
7	5	-41.34523	174.8124	NINJABLD_S001_S001_T003	300	Ecklonia	0.583
8	6	-41.34523	174.8123	NINJABLD_S001_S001_T003	375	Ecklonia	0
9	7	-41.34522	174.8123	NINJABLD_S001_S001_T003	475	Ecklonia	0
10	8	-41.34521	174.8123	NINJABLD_S001_S001_T003	500	Ecklonia	0.25
11	9	-41.34521	174.8122	NINJABLD_S001_S001_T003	725	Ecklonia	0.25
12	10	-41.3452	174.8122	NINJABLD_S001_S001_T003	775	Ecklonia	0.84
13	11	-41.3452	174.8121	NINJABLD_S001_S001_T003	800	Ecklonia	0
14	12	-41.34519	174.8121	NINJABLD_S001_S001_T003	900	Ecklonia	0.083
15	13	-41.34519	174.812	NINJABLD_S001_S001_T003	975	Ecklonia	0.083
16	14	-41.34519	174.8119	NINJABLD_S001_S001_T003	1225	Ecklonia	0.014
17	15	-41.3452	174.8119	NINJABLD_S001_S001_T003	1275	Ecklonia	0.062
18	16	-41.34521	174.8119	NINJABLD_S001_S001_T003	1450	Ecklonia	0.833
19	17	-41.34522	174.8119	NINJABLD_S001_S001_T003	1475	Ecklonia	0.243
20	18	-41.34522	174.8118	NINJABLD_S001_S001_T003	1525	Ecklonia	0.438
21	19	-41.34523	174.8118	NINJABLD_S001_S001_T003	1625	Ecklonia	0.583
22	20	-41.34524	174.8118	NINJABLD_S001_S001_T003	1675	Ecklonia	0.375
23	21	-41.34525	174.8118	NINJABLD_S001_S001_T003	1700	Ecklonia	0.139
24	22	-41.34526	174.8118	NINJABLD_S001_S001_T003	1725	Ecklonia	0.5
25	23	-41.34527	174.8118	NINJABLD_S001_S001_T003	1750	Ecklonia	0.535

Figure A8.13. An example of the processed AI analysis data.



	A	B	C	D	E	F	G	H	I	J
1		frame	south	east	date	time	Algae	Carpophyllum	Ecklonia	Lessonia
2	0	0	-41.348200Â°S	+174.808200Â°E	31/10/17	21:45:28	0	0	0	0
3	1	25	-41.348210Â°S	+174.808200Â°E	31/10/17	21:45:29	0.25	0.042	0.208	0.021
4	2	50	-41.348210Â°S	+174.808200Â°E	31/10/17	21:45:30	0	0	0	0
5	3	75	-41.348210Â°S	+174.808200Â°E	31/10/17	21:45:31	0.583	0.049	0.486	0.194
6	4	100	-41.348210Â°S	+174.808200Â°E	31/10/17	21:45:32	0.167	0	0.167	0.028
7	5	125	-41.348220Â°S	+174.808200Â°E	31/10/17	21:45:33	0	0	0	0
8	6	150	-41.348220Â°S	+174.808200Â°E	31/10/17	21:45:34	0	0	0	0
9	7	175	-41.348220Â°S	+174.808200Â°E	31/10/17	21:45:35	0	0	0	0
10	8	200	-41.348220Â°S	+174.808200Â°E	31/10/17	21:45:36	0	0	0	0
11	9	225	-41.348220Â°S	+174.808200Â°E	31/10/17	21:45:37	0	0	0	0
12	10	250	-41.348220Â°S	+174.808200Â°E	31/10/17	21:45:38	0	0	0	0
13	11	275	-41.348230Â°S	+174.808200Â°E	31/10/17	21:45:39	0	0	0	0
14	12	300	-41.348230Â°S	+174.808200Â°E	31/10/17	21:45:40	0.083	0.028	0	0.069
15	13	325	-41.348230Â°S	+174.808200Â°E	31/10/17	21:45:41	0	0	0	0
16	14	350	-41.348230Â°S	+174.808200Â°E	31/10/17	21:45:42	0.5	0.292	0	0.292
17	15	375	-41.348240Â°S	+174.808200Â°E	31/10/17	21:45:43	0.417	0	0	0.417
18	16	400	-41.348230Â°S	+174.808200Â°E	31/10/17	21:45:44	0	0	0	0
19	17	425	-41.348230Â°S	+174.808200Â°E	31/10/17	21:45:45	0.083	0.083	0	0.069
20	18	450	-41.348230Â°S	+174.808200Â°E	31/10/17	21:45:46	0	0	0	0
21	19	475	-41.348230Â°S	+174.808200Â°E	31/10/17	21:45:47	0	0	0	0
22	20	500	-41.348220Â°S	+174.808200Â°E	31/10/17	21:45:48	0	0	0	0
23	21	525	-41.348220Â°S	+174.808200Â°E	31/10/17	21:45:49	0.667	0.444	0.111	0.556
24	22	550	-41.348220Â°S	+174.808200Â°E	31/10/17	21:45:50	0.083	0.035	0.007	0.083
25	23	575	-41.348220Â°S	+174.808200Â°E	31/10/17	21:45:51	0.333	0.111	0.056	0.333

Figure A8.14. An example of the raw AI analysis data.

The resulting Excel files contained points from all videos in a set, for example the whole Moa Point area. Each excel file only contained one species/class, for example ”*Ecklonia*” or ”Algae”. These changes were for ease of use, as shown in (Figures A8.15 and A8.16) each species was able to be toggled with one click for an entire area.

Figure A8.15. The heat map for *Ecklonia* at Moa Point toggled on.

Figure A8.16. The heat map for *Ecklonia* on Moa Point toggled off.

Thumbnail extraction

The second part of the pre-processing was to create a thumbnail image for each discrete coordinate. This was done by finding each frame that corresponds with each coordinate data point from the process above. The result of this was a folder of all the thumbnail sized images needed, each with the ”(video name) (frame)” naming convention (Figure A8.17).

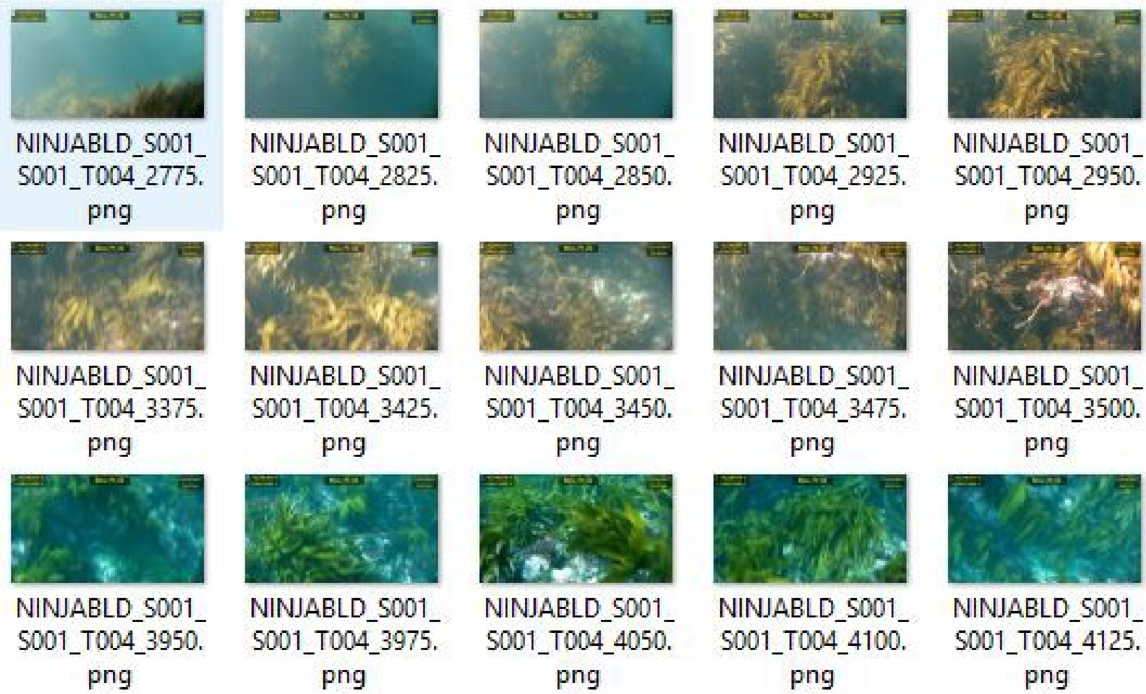


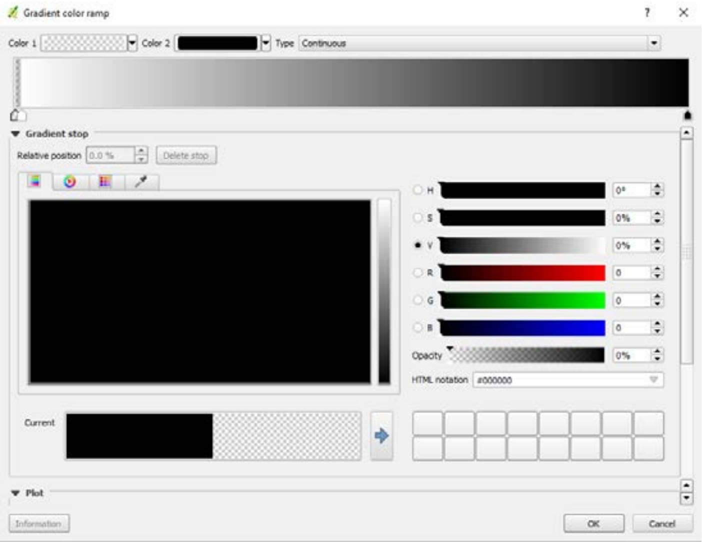
Figure A8.17. The thumbnail images to be used for pop ups in QGIS.

Mapping with QGIS

- design choices made in QGIS to create the desired functionality.

Heat maps

The method of displaying the data was fairly straight forward, using the heat map style and setting the weighting to the ”Confidence” column. The colour scheme was customised (Figure A8.18) to have a colour gradient over non-zero confidence coordinators and complete transparency elsewhere (Figure A8.19).



Type Continuous

Relative position 0.0 %

H 0°

S 0%

V 0%

R 0

G 0

B 0

Opacity 0%

HTML notation #000000

Figure A8.18. The customized colour scheme.

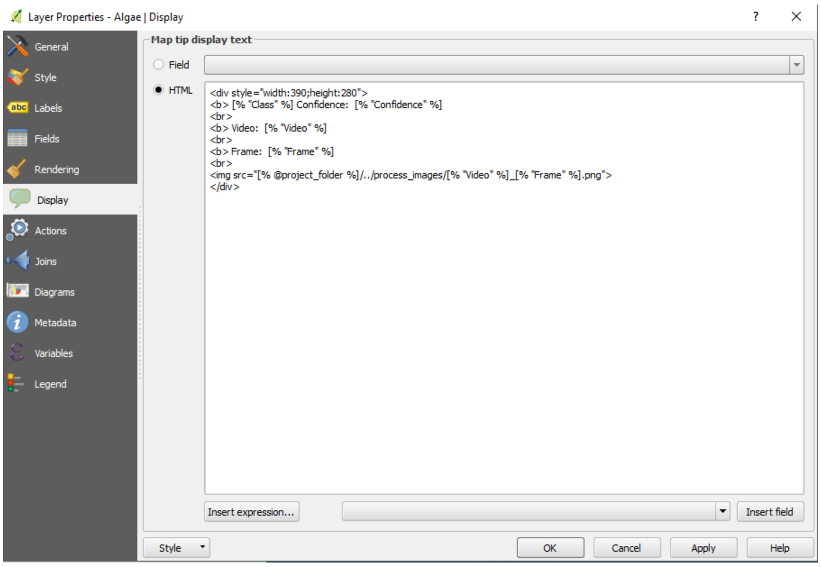
Figure A8.19. An example of the heat map for the Algae class.

Map tips

To have a thumbnail pop up the ”Map tips” functionality was used. Map tips when activated allows information to be displayed about a data point when hovering over it with the mouse (Figure A8.19). This information can be customized from the display section of the properties menu for a data set (Figure A8.20). As shown this was done using HTML markup therefore displaying images can be done using ”img src”.

Ecklonia Confidence: 0.625
Video: NINJABLD_S001_S001_T014
Frame: 2050

Figure A8.20. A thumbnail showing frame used to calculate the confidence.

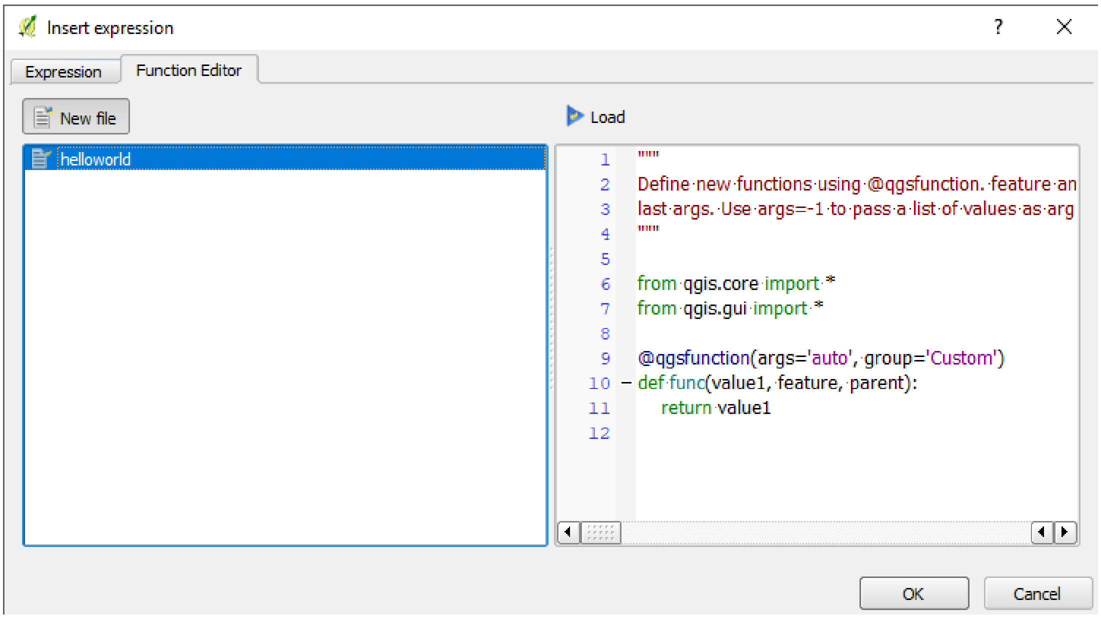


```
<div style="width:390;height:280">
<b> [% "Class" %] Confidence:  [% "Confidence" %]
<br>
<b> Video:  [% "Video" %]
<br>
<b> Frame:  [% "Frame" %]
<br>
<img src="[% @project_folder %]/../process_images/[% "Video" %]_[% "Frame" %].png">
</div>
```

Figure A8.21. The HTML markup used to generate the thumbnail pop ups.

One way that this differs from standard HTML was the use of the [% %] format. These essentially represent some variable that gets substituted in, for example, (Figure A8.20) above "Video" is the name of the video and "Frame" is name frame number, hence the reason for the naming format described in the pre-processing section.

This can be also be used for various functions, as shown in (Figure A8.21) the function [% @project folder %] was used to get the file path of the project. This means that the project can find the relevant project files regardless where the project is stored (for example an on-line network drive). These functions are not limited to the default list and can be created using the Python function editor (Figure A8.22).



```
"""
Define new functions using @qgsfunction. feature an
last args. Use args=-1 to pass a list of values as arg
"""

from qgis.core import *
from qgis.gui import *

@qgsfunction(args='auto', group='Custom')
def func(value1, feature, parent):
    return value1

```

Figure A8.22. The Python function editor for QGIS maps tips.

17 APPENDIX 9. Colour Dependence of AI Classification System

This section provides the methods used to measure how dependent the AI classification system is on colour information.

Benchmark

The first step of the process is to determine the performance using full colour imagery. (Figure A9.1).

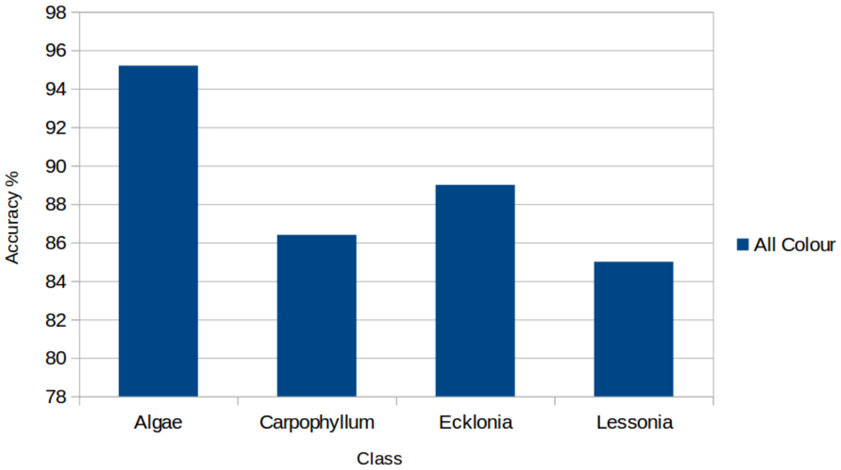


Figure A9.1: Benchmark results using colour training and testing data

Grey scale testing images

The first test to determine colour dependency was by feeding grey scale images to the AI to be classified. The testing dataset for this test is the same as used in the benchmark test. To remove colour information, each RGB pixel is averaged across the three values giving the monochrome equivalent of the image. The results of this test are as shown in Figure A9.2.

From this test it appears as if system performs poorly (close to random guessing in some cases) without colour information to rely on. Another potential explanation for the low accuracy in that, as all the training data has been in RGB colour, the system simply isn't trained to recognize shapes/patterns with a monochrome colour scheme as it would be expecting brown or green patterns. This leads to the second test.

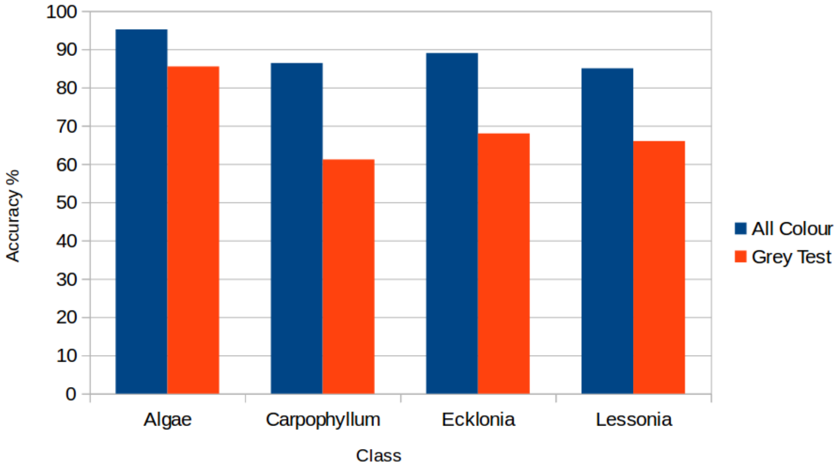


Figure A9.2. Test results using colour training data and monochrome testing data.

Grey scale training and testing images

The second test to determine colour dependency was to do the same conversion from RGB to monochrome as in the first test for both the training images and testing images. This should allow the performance of the AI system to be measured when exclusively using visual features with the absence of colour information. The results of this test are shown in Figure A9.3.

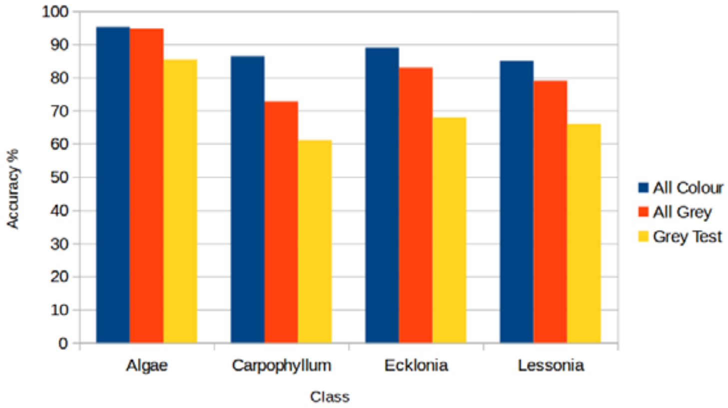


Figure A9.3. Test results using monochrome training data and monochrome testing data.

The results show that this test gives a stronger performance than the first test but is out-performed by the benchmark. The conclusions that can be made from this are.

1. The AI system can make classifications exclusively using visual patterns without colour information.
2. The AI system will use colour information if available to increase performance.
3. If the AI system is trained to utilize colour information, it will perform poorly when classifying monochrome images.

18 APPENDIX 10. Distribution maps by source and species

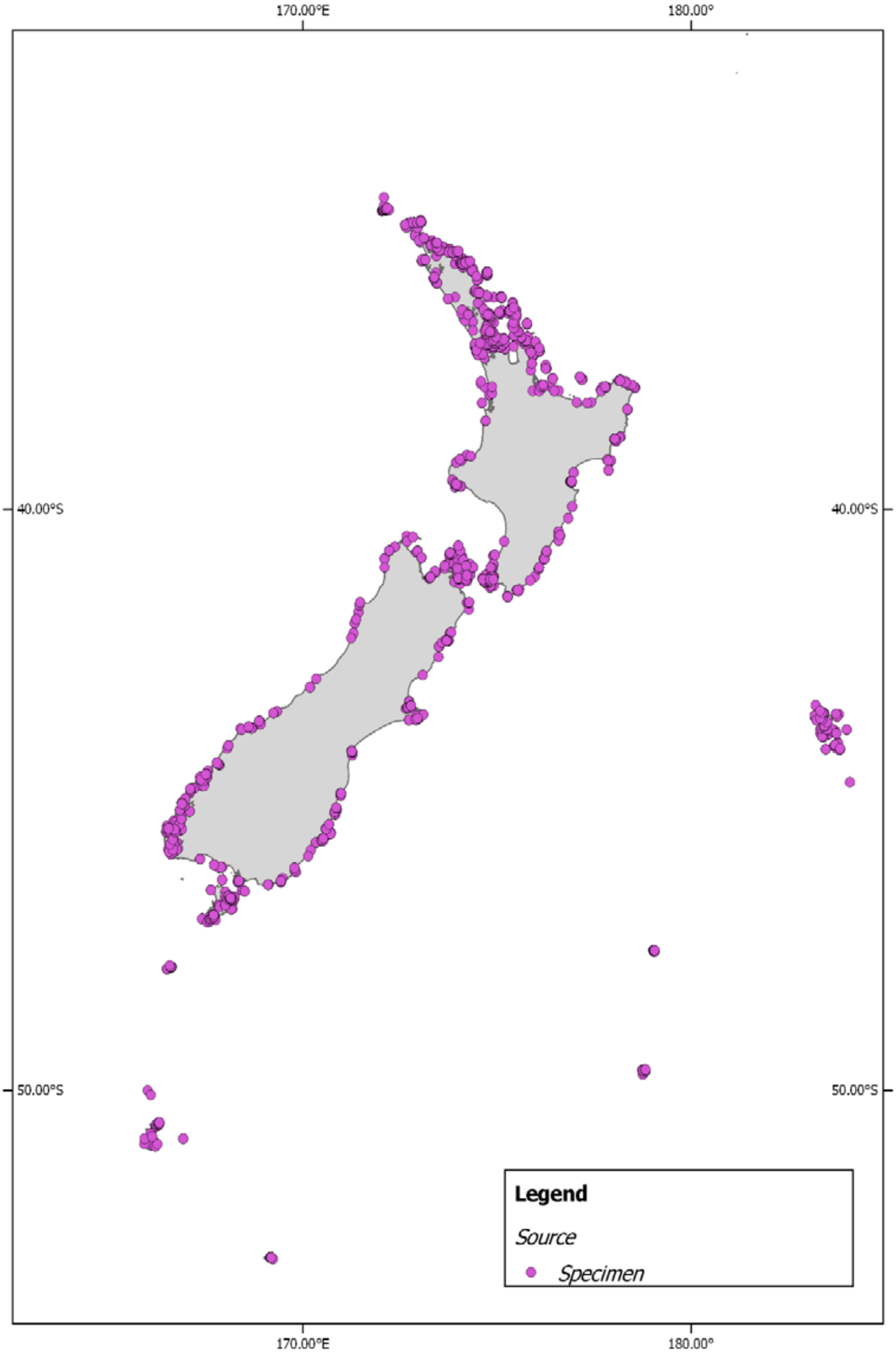


Figure A10.1. Records of large brown macroalgae (Fucales and Laminariales) in New Zealand recorded by specimen records.

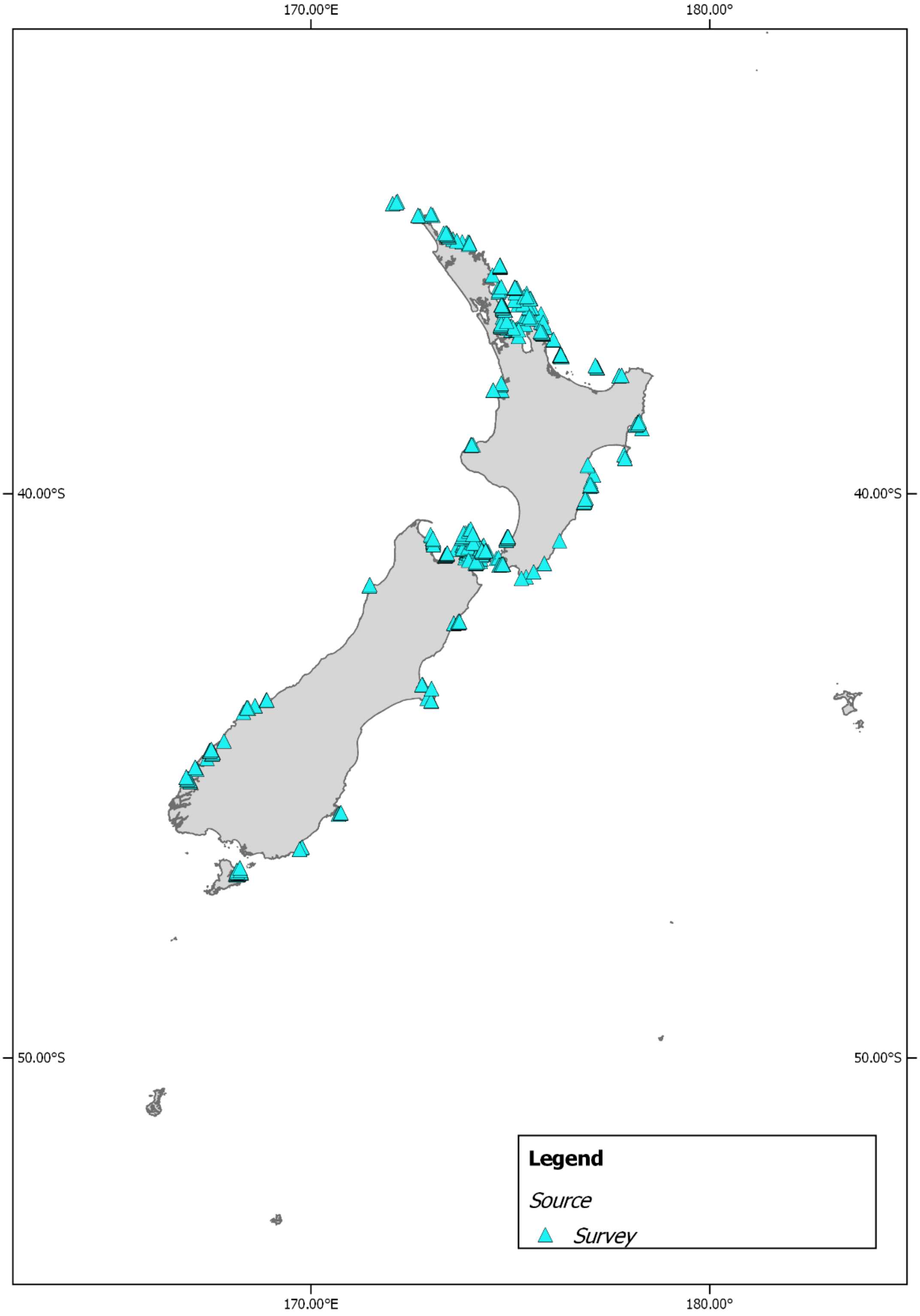


Figure A10.2. **Records of large brown macroalgae (Fucales and Laminariales) in New Zealand recorded by survey records.**

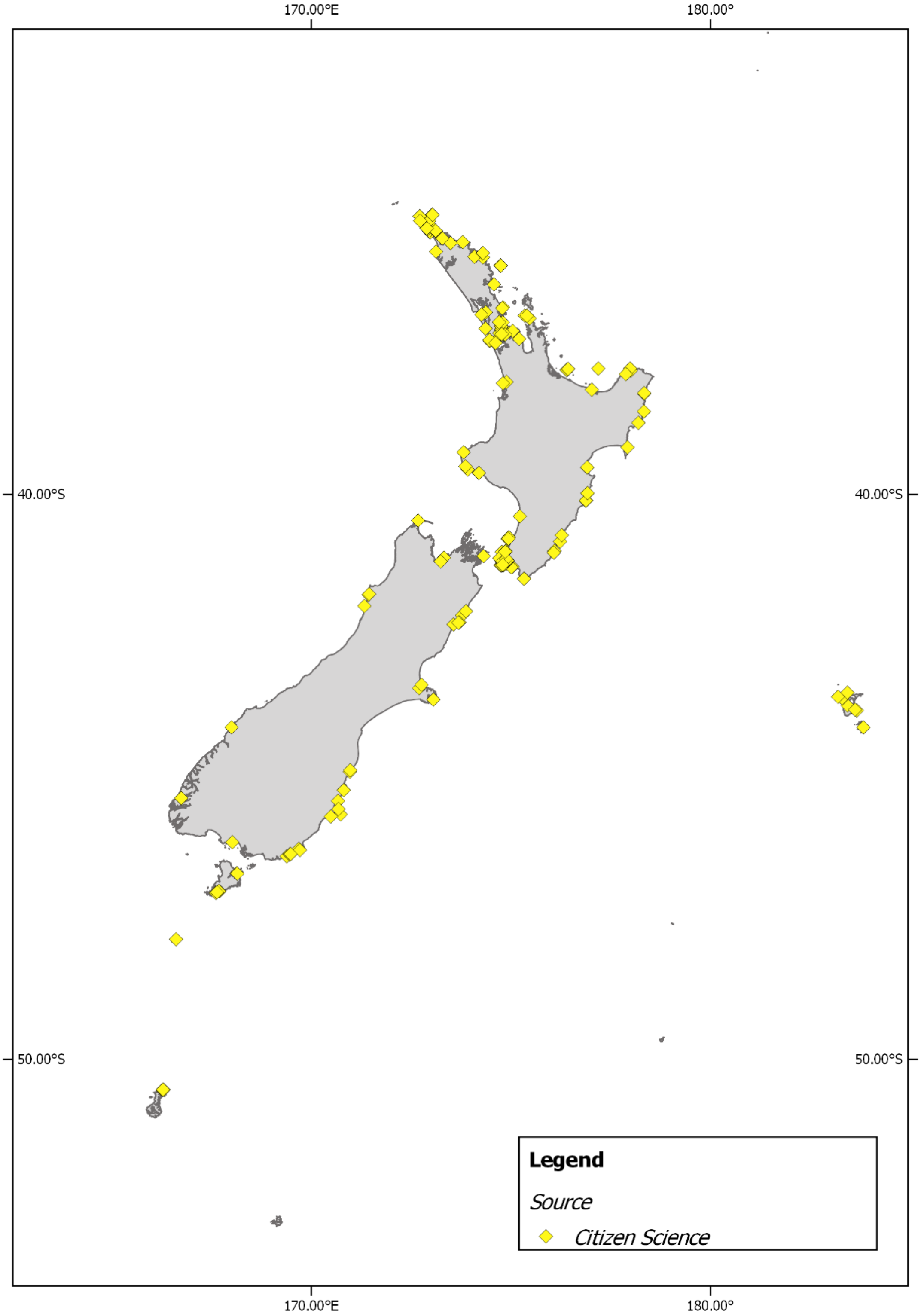


Figure A10.3. Records of large brown macroalgae (Fucales and Laminariales) in New Zealand recorded by citizen science records.

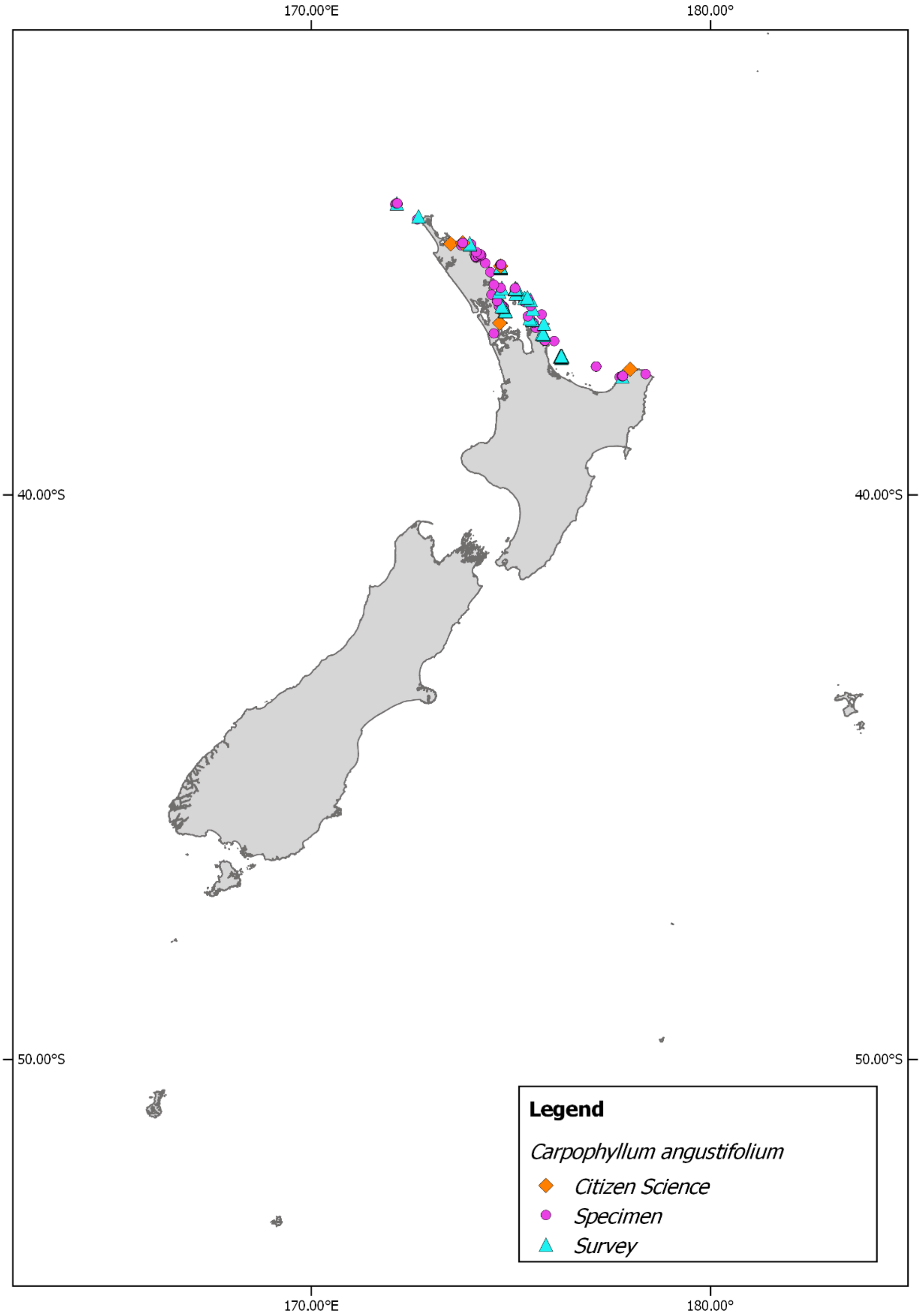


Figure A10.4 The New Zealand distribution of *Carpophyllum angustifolium* (Fucales) as recorded by citizen science, specimen and survey records.

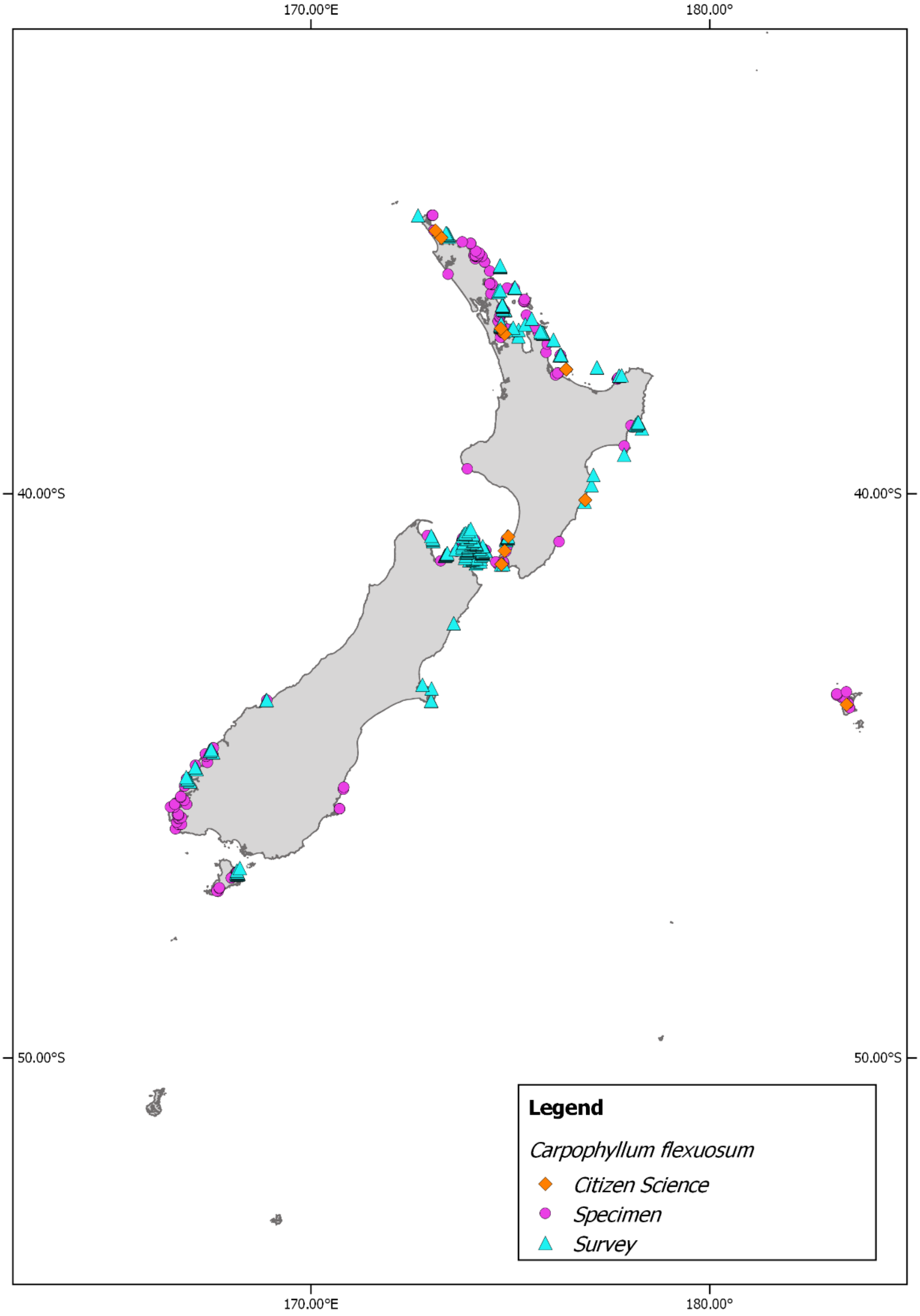


Figure A10.5. The New Zealand distribution of *Carpophyllum flexuosum* (Fucales) as recorded by citizen science, specimen and survey records.

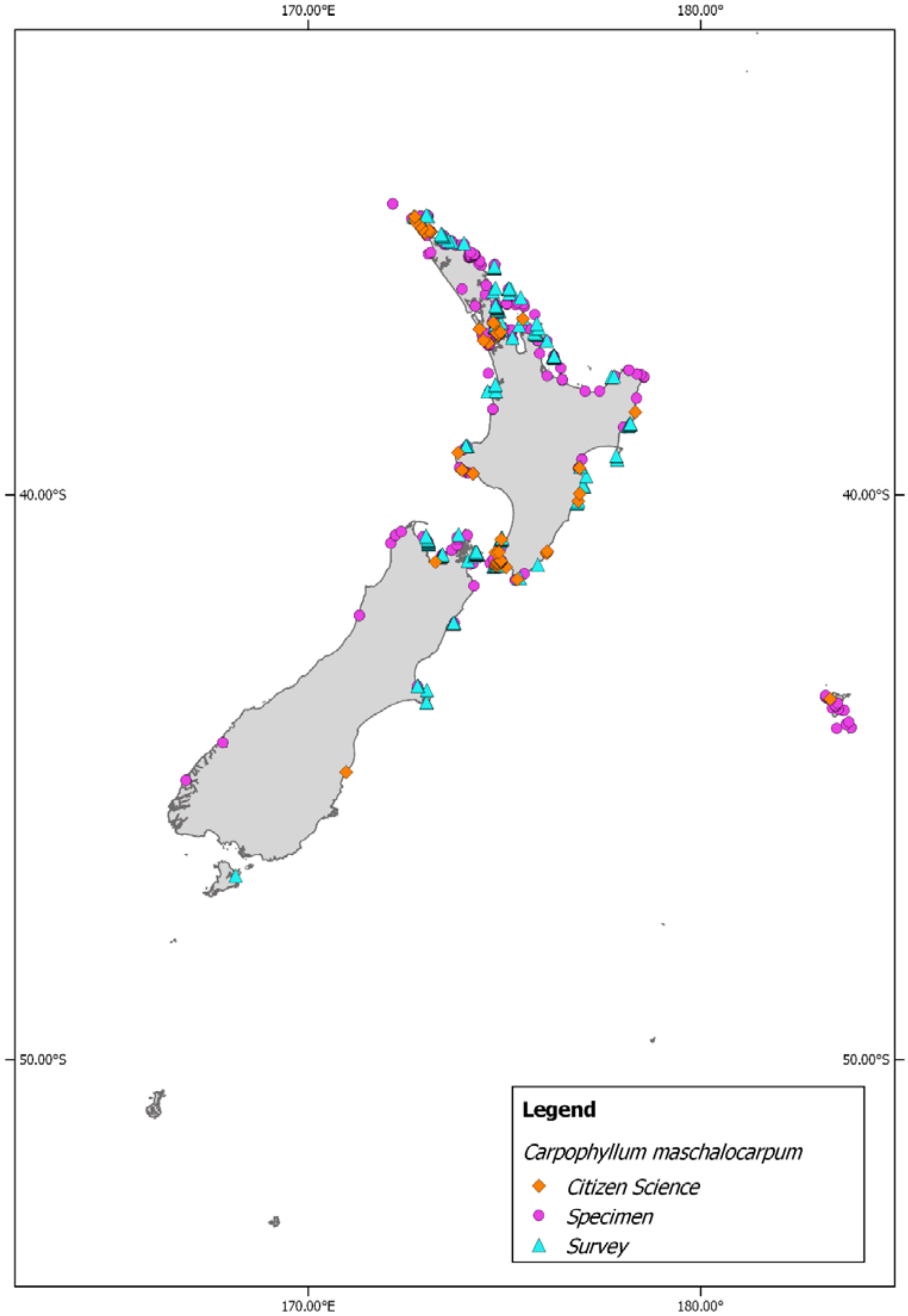


Figure A10.6. The New Zealand distribution of *Carpophyllum maschalocarpum* (Fucales) as recorded by citizen science, specimen and survey records.

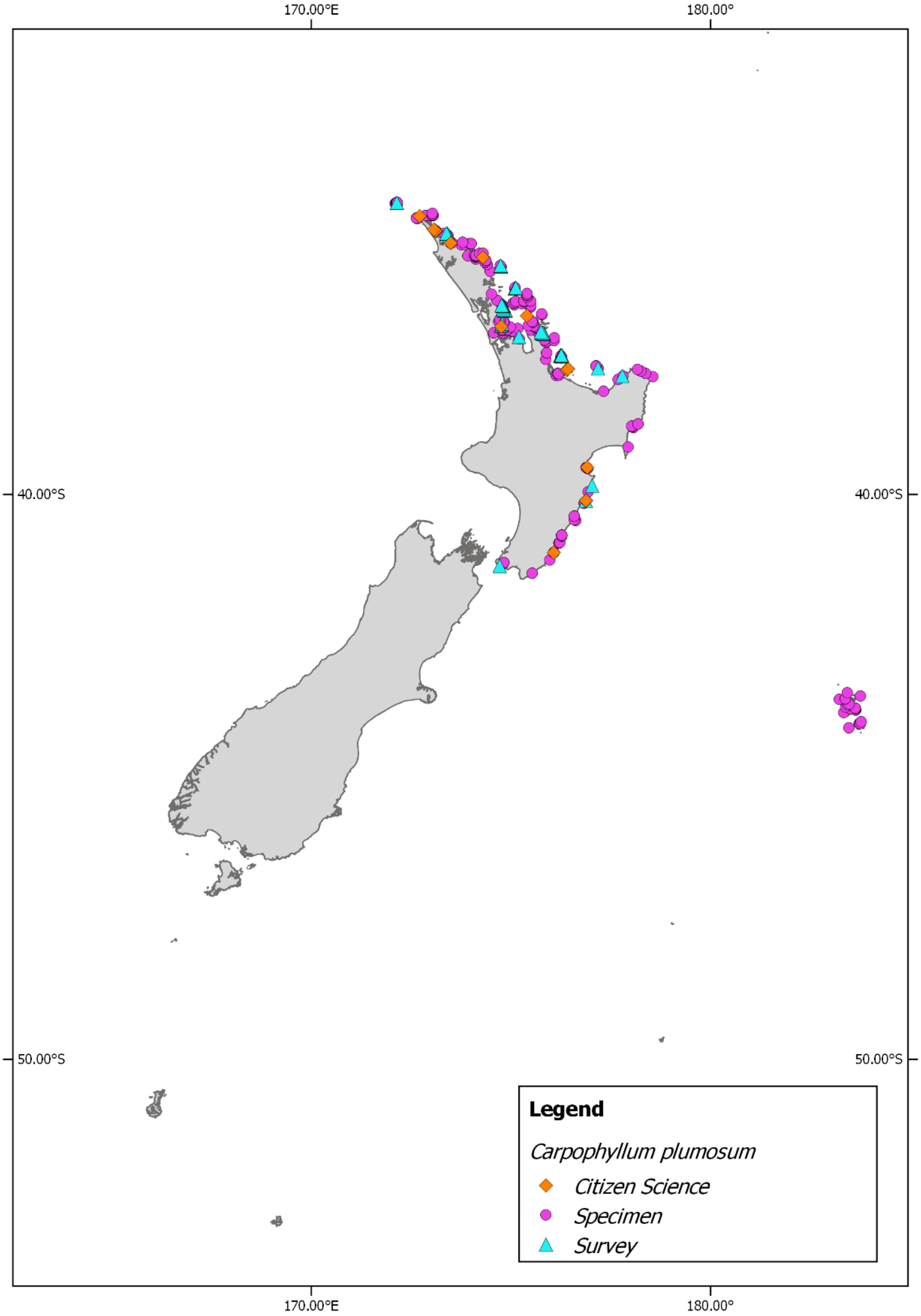


Figure A10.7. The New Zealand distribution of *Carpophyllum plumosum* (Fucales) as recorded by citizen science, specimen and survey records.

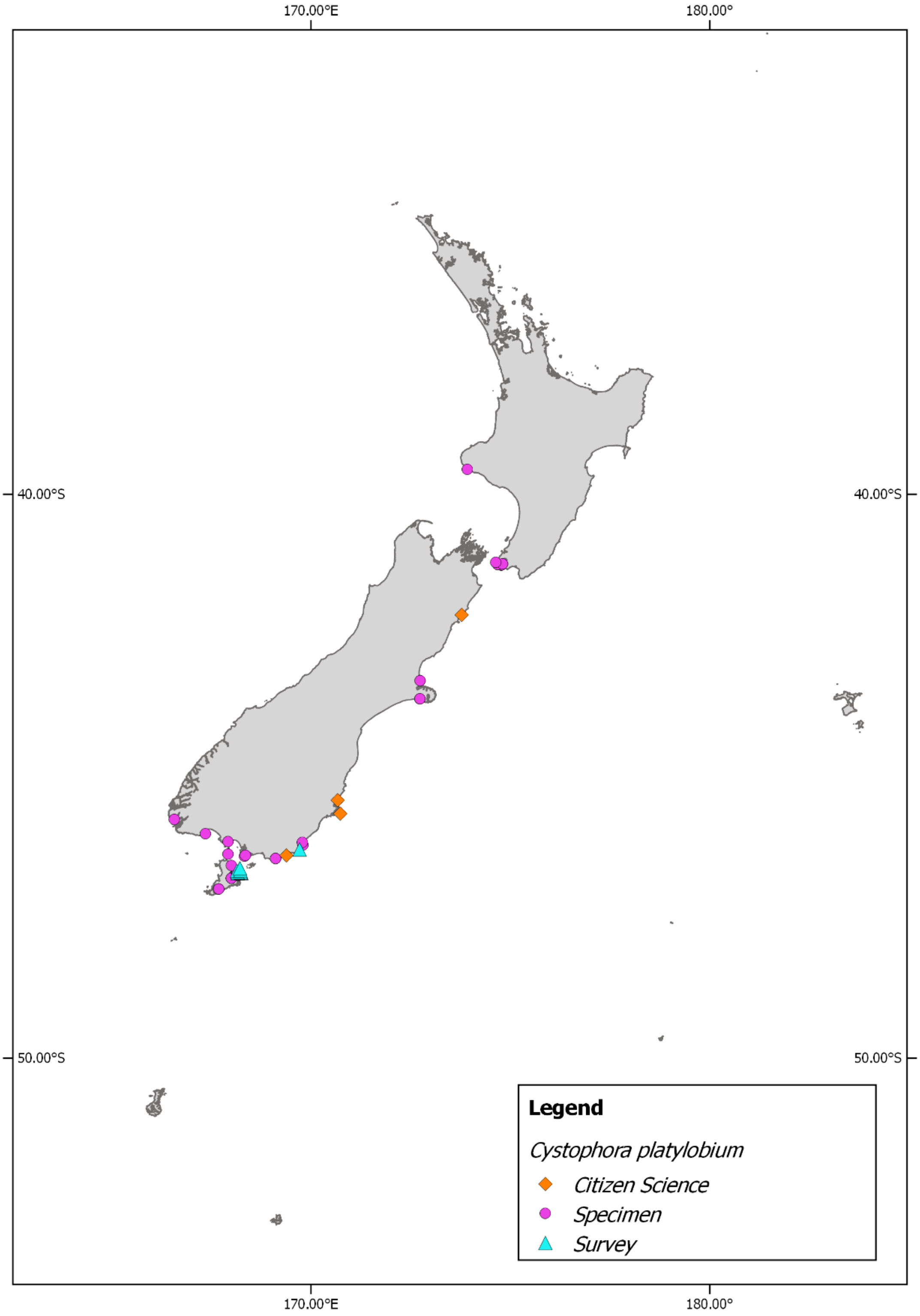


Figure A10.8. The New Zealand distribution of *Cystophora platylobium* (Fucales) as recorded by citizen science, specimen and survey records.

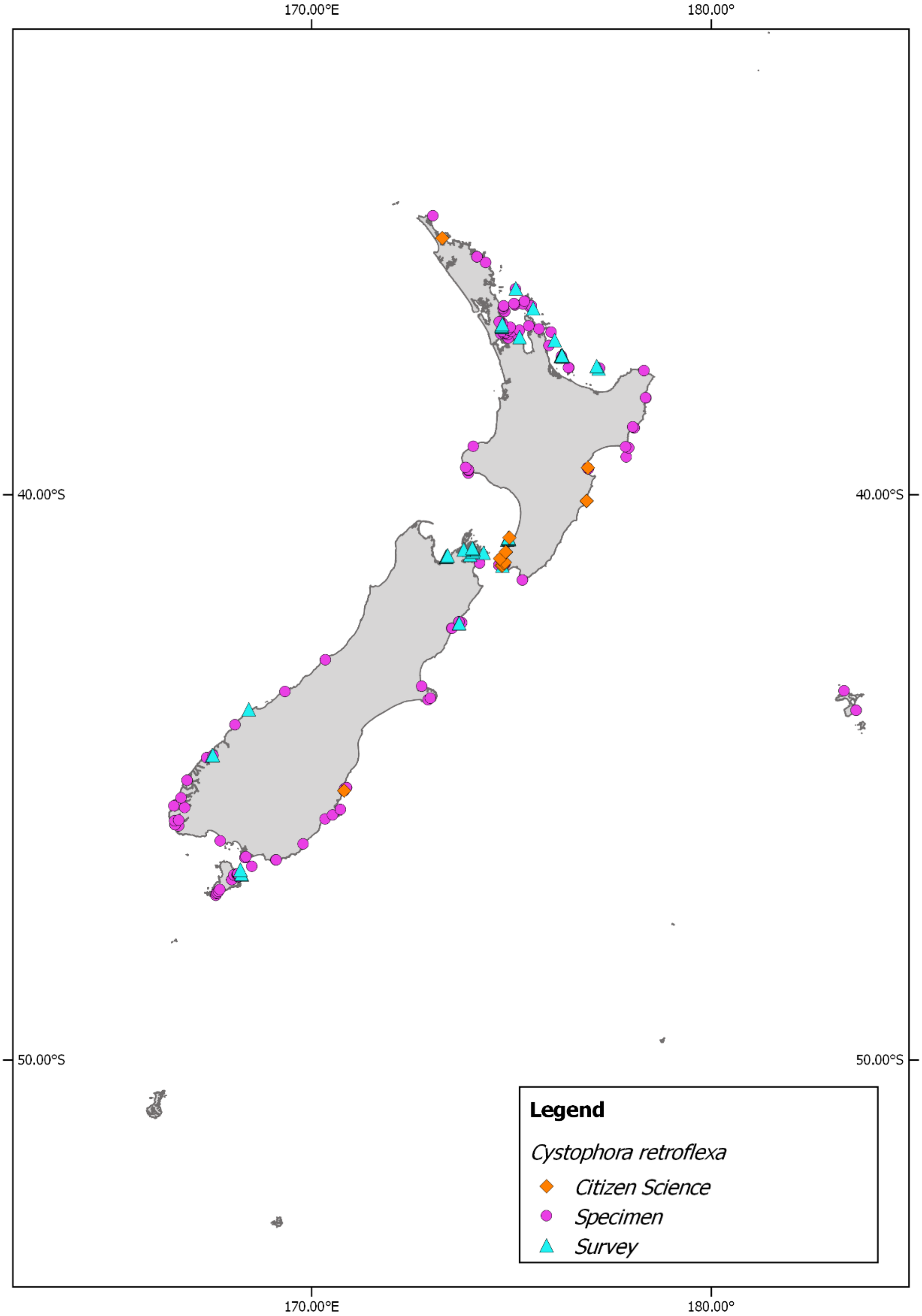


Figure A10.9. The New Zealand distribution of *Cystophora retroflexa* (Fucales) as recorded by citizen science, specimen and survey records.

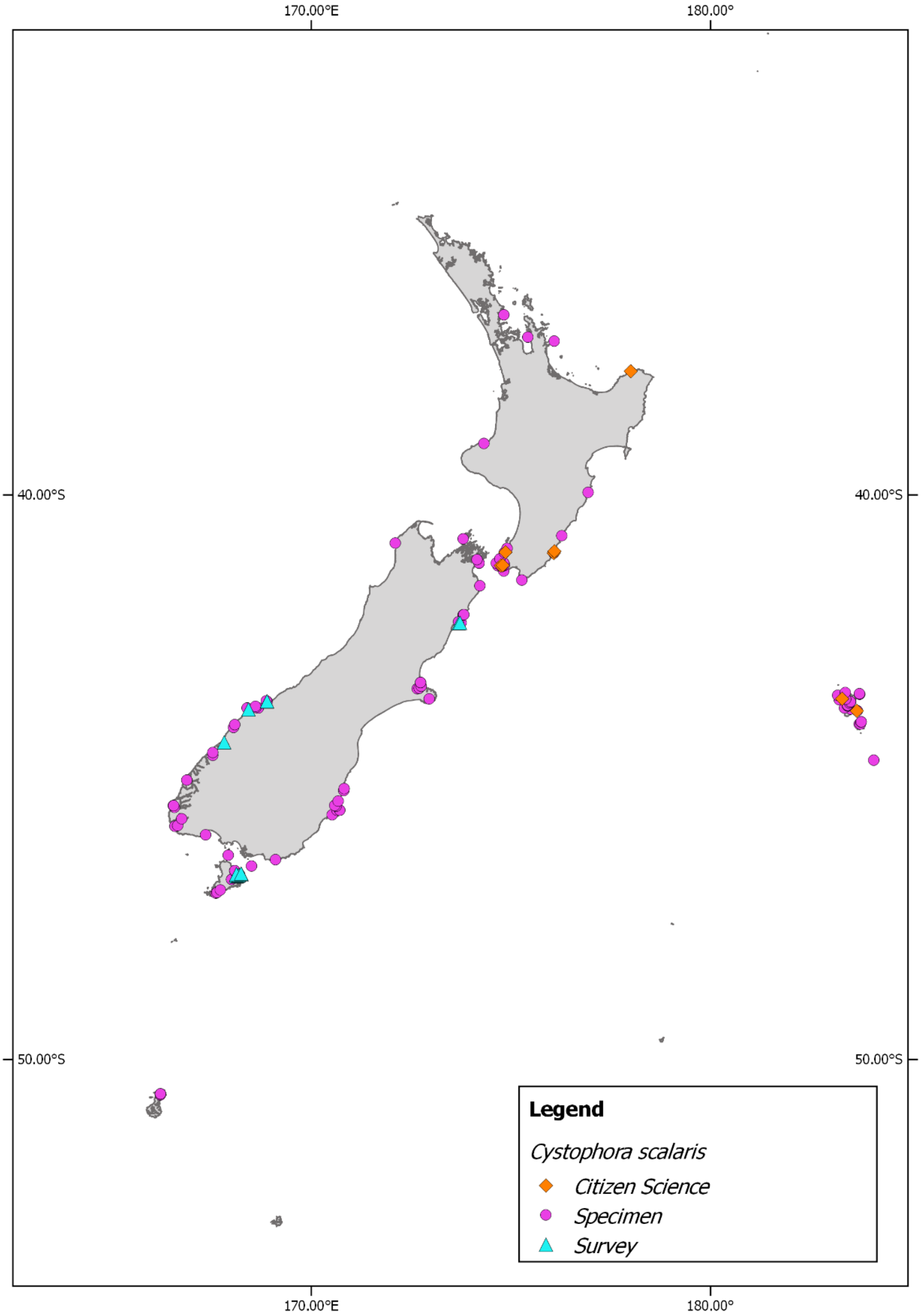


Figure A10.10. The New Zealand distribution of *Cystophora scalaris* (Fucales) as recorded by citizen science, specimen and survey records.

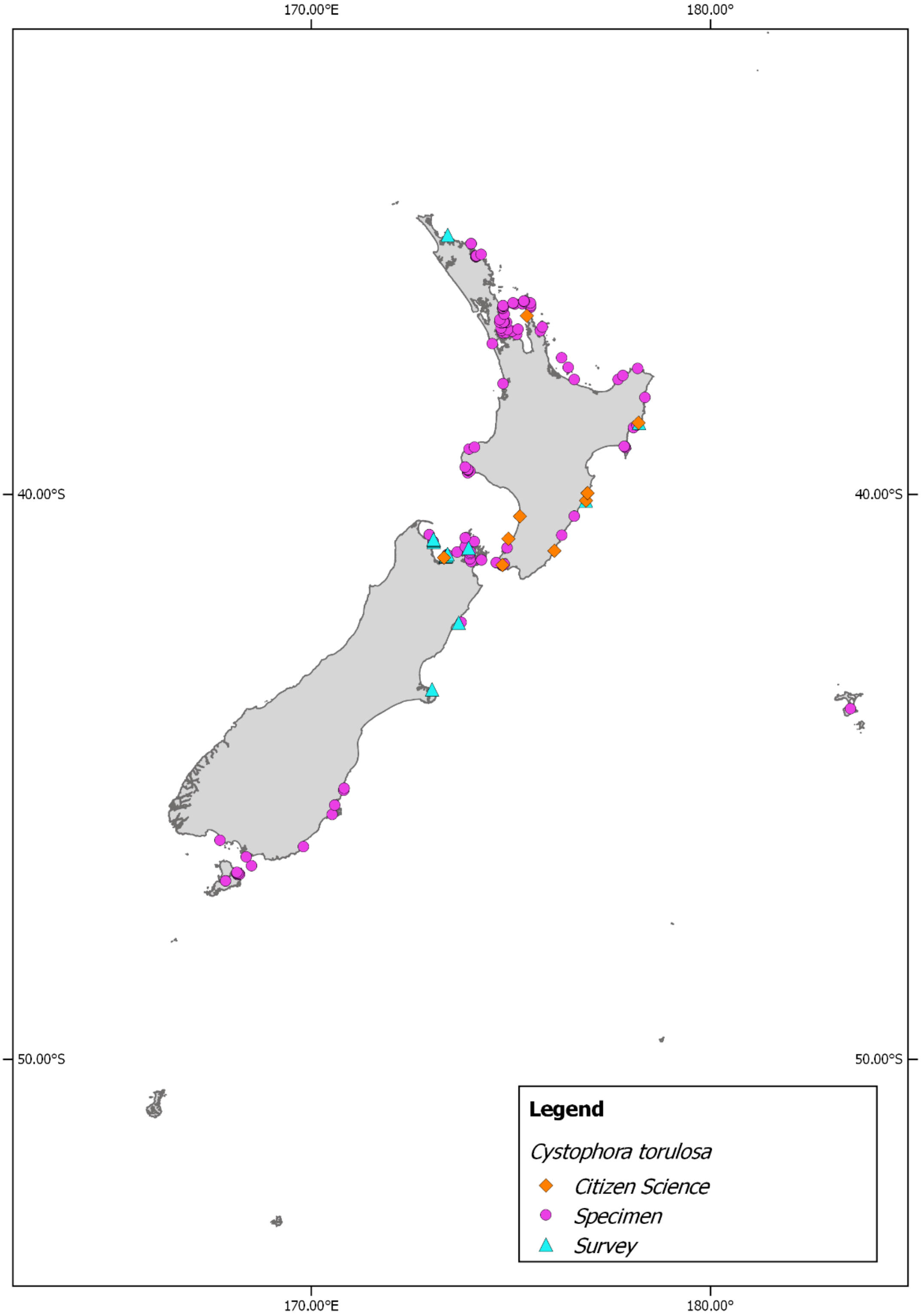


Figure A10.11. The New Zealand distribution of *Cystophora torulosa* (Fucales) as recorded by citizen science, specimen and survey records.

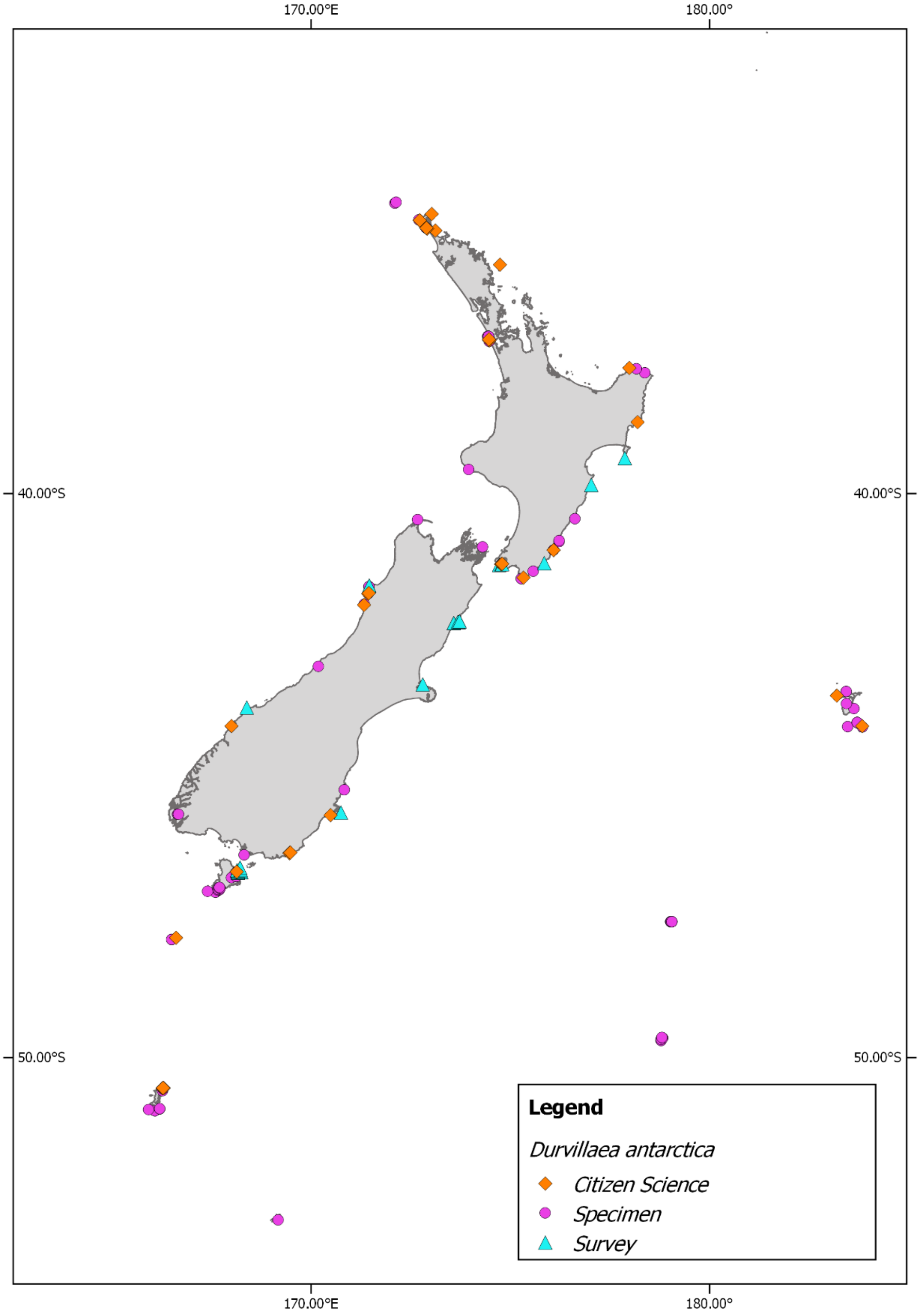


Figure A10.12. The New Zealand distribution of *Durvillaea antarctica* (Fucales) as recorded by citizen science, specimen and survey records.

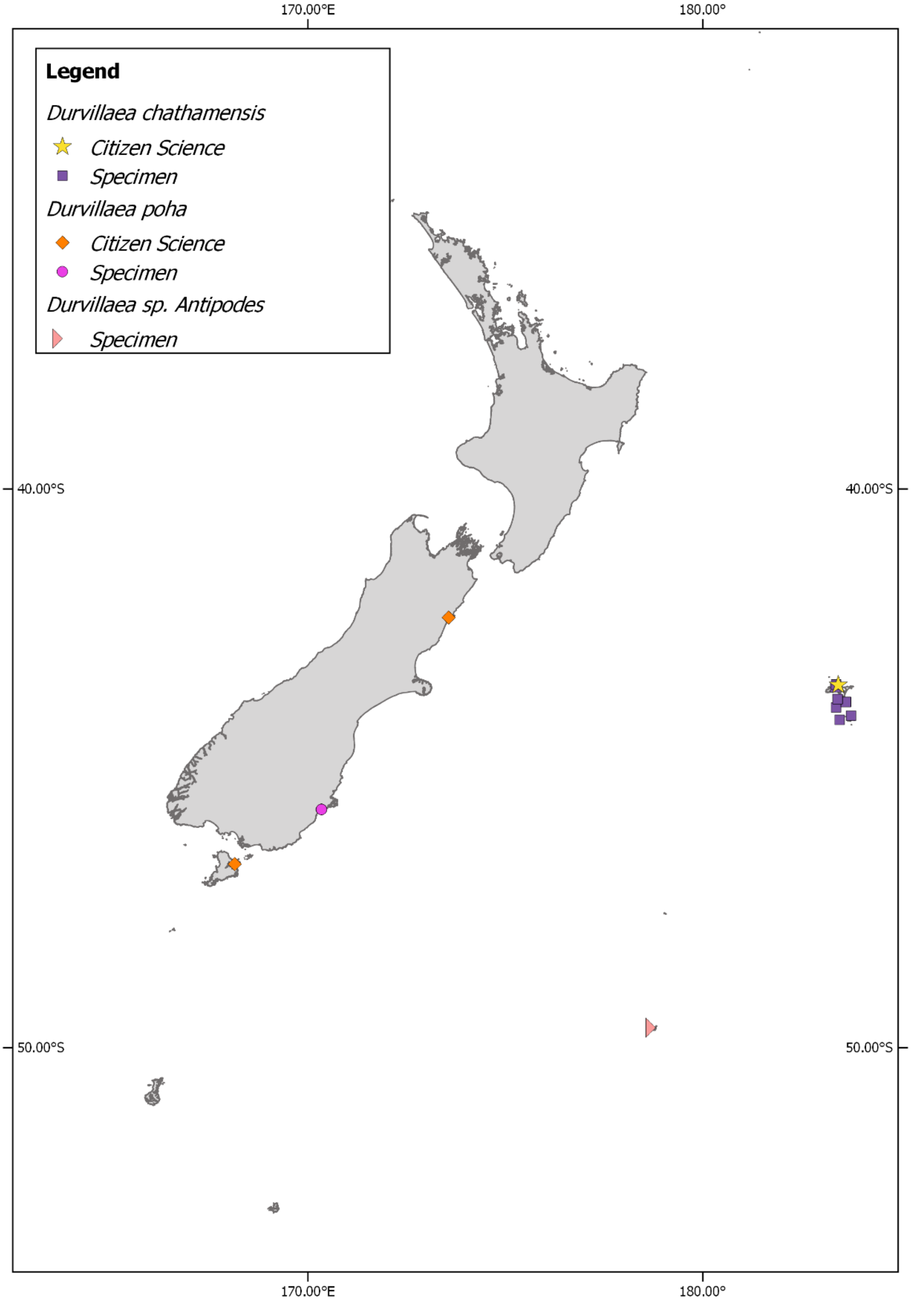


Figure A10.13. The New Zealand distribution of *Durvillaea chathamensis*, *D. poha*, and *Durvillaea* sp. Antipodes (Fucales) as recorded by citizen science and specimen records.

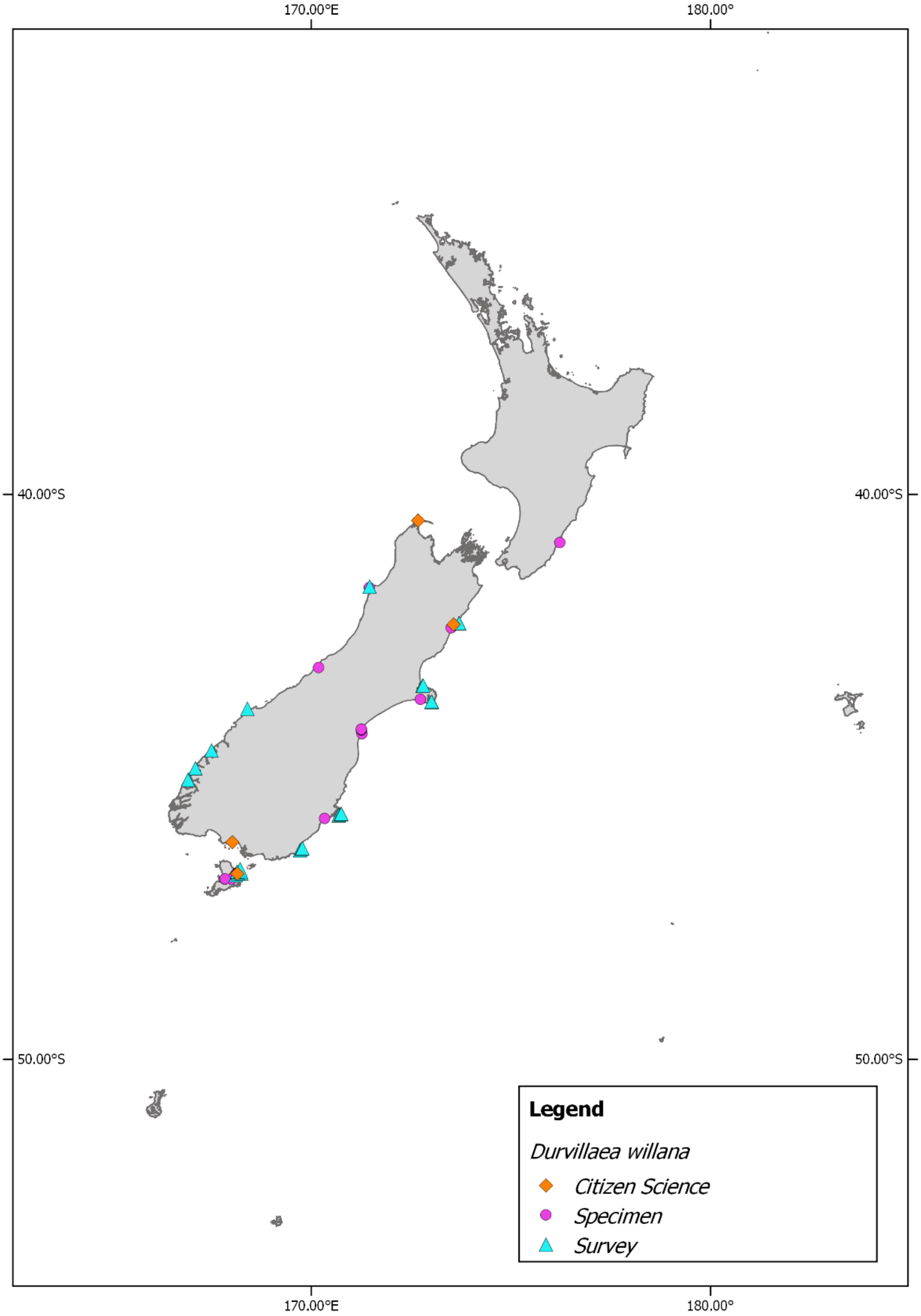


Figure A10.14. The New Zealand distribution of *Durvillaea willana* (Fucales) as recorded by citizen science, specimen and survey records.

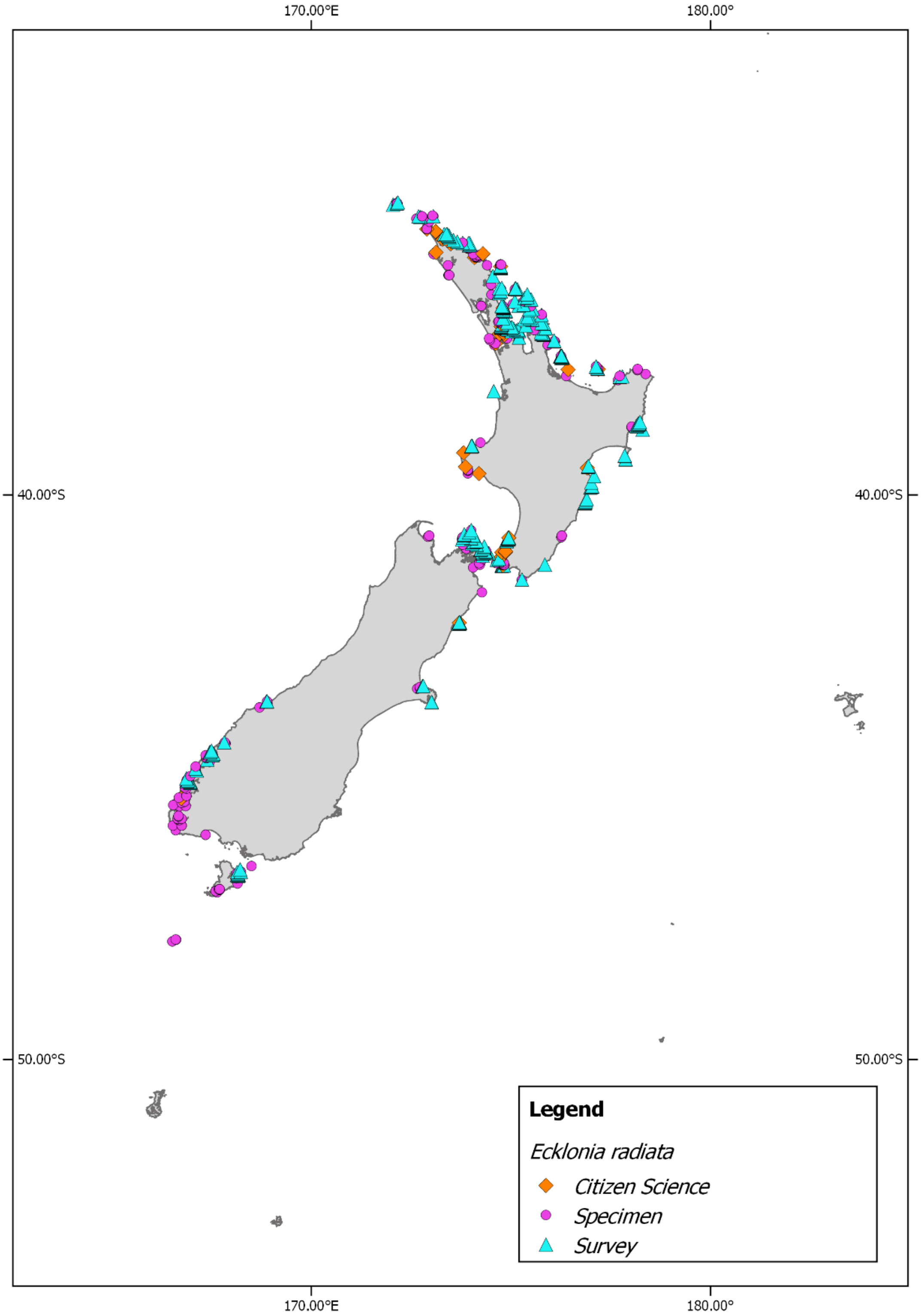


Figure A10.15. The New Zealand distribution of *Ecklonia radiata* (Laminariales) as recorded by citizen science, specimen and survey records.

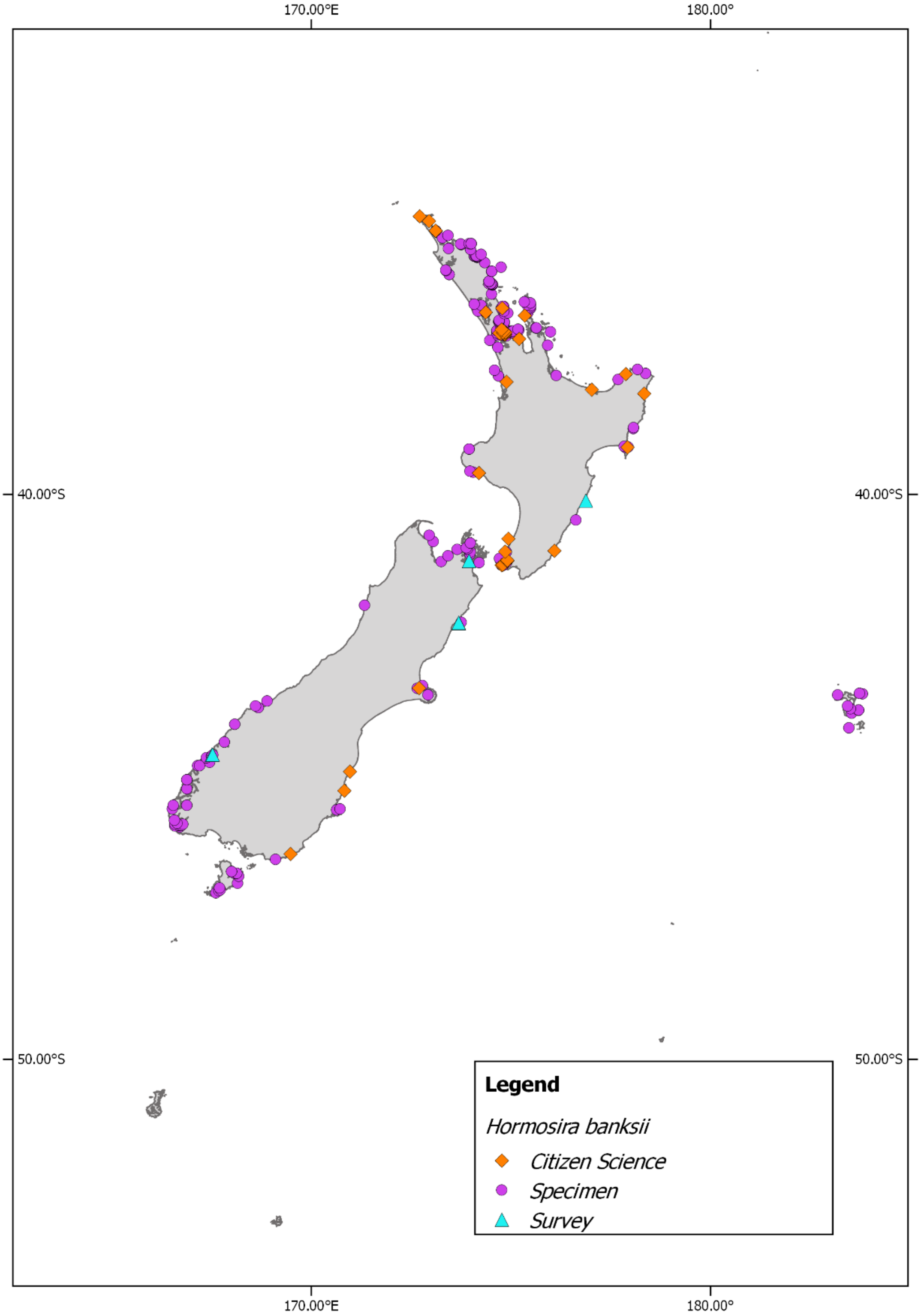


Figure A10.16. The New Zealand distribution of *Hormosira banksii* (Fucales) as recorded by citizen science, specimen and survey records.

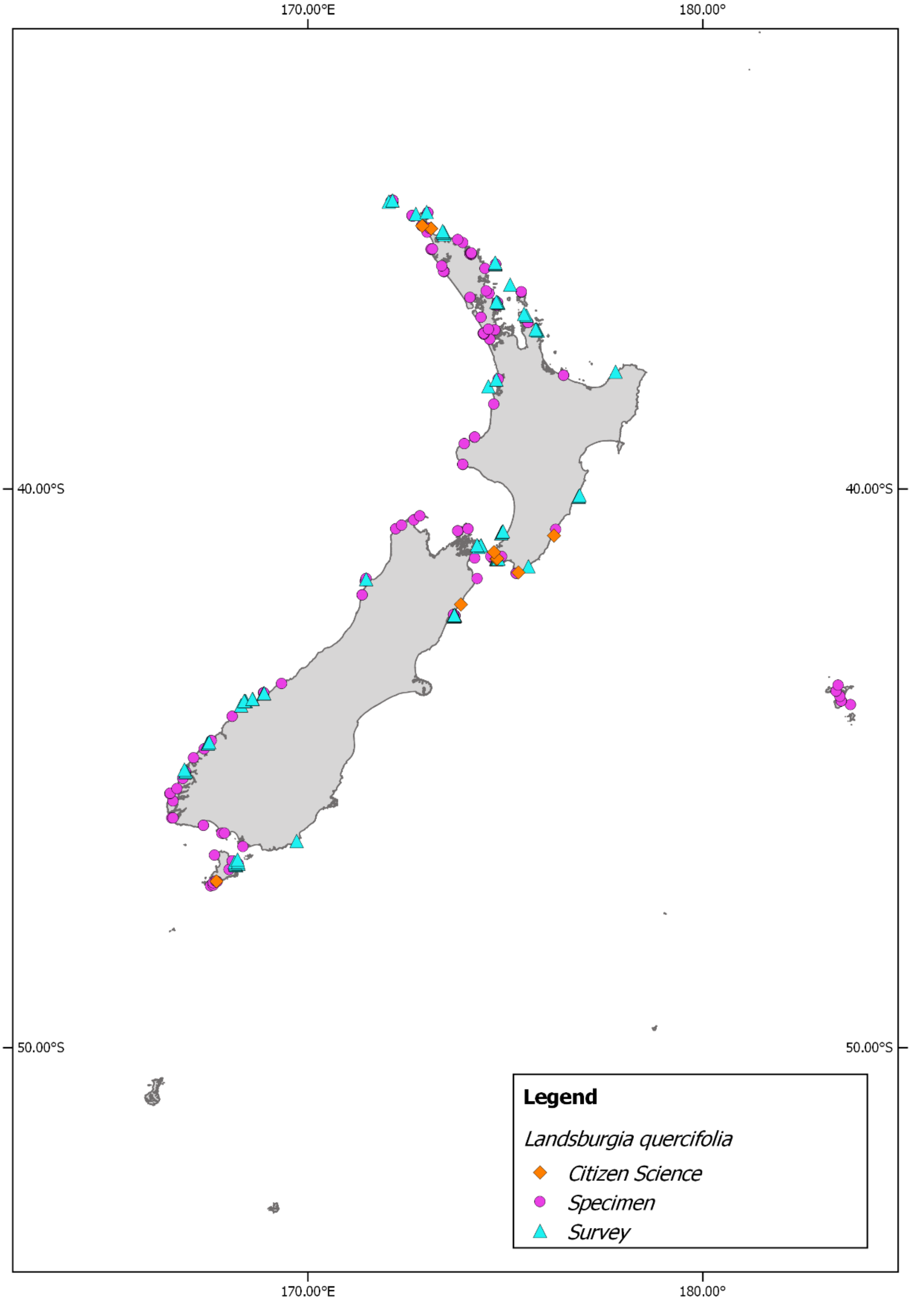


Figure A10.17. The New Zealand distribution of *Landsburgia quercifolia* (Fucales) as recorded by citizen science, specimen and survey records.

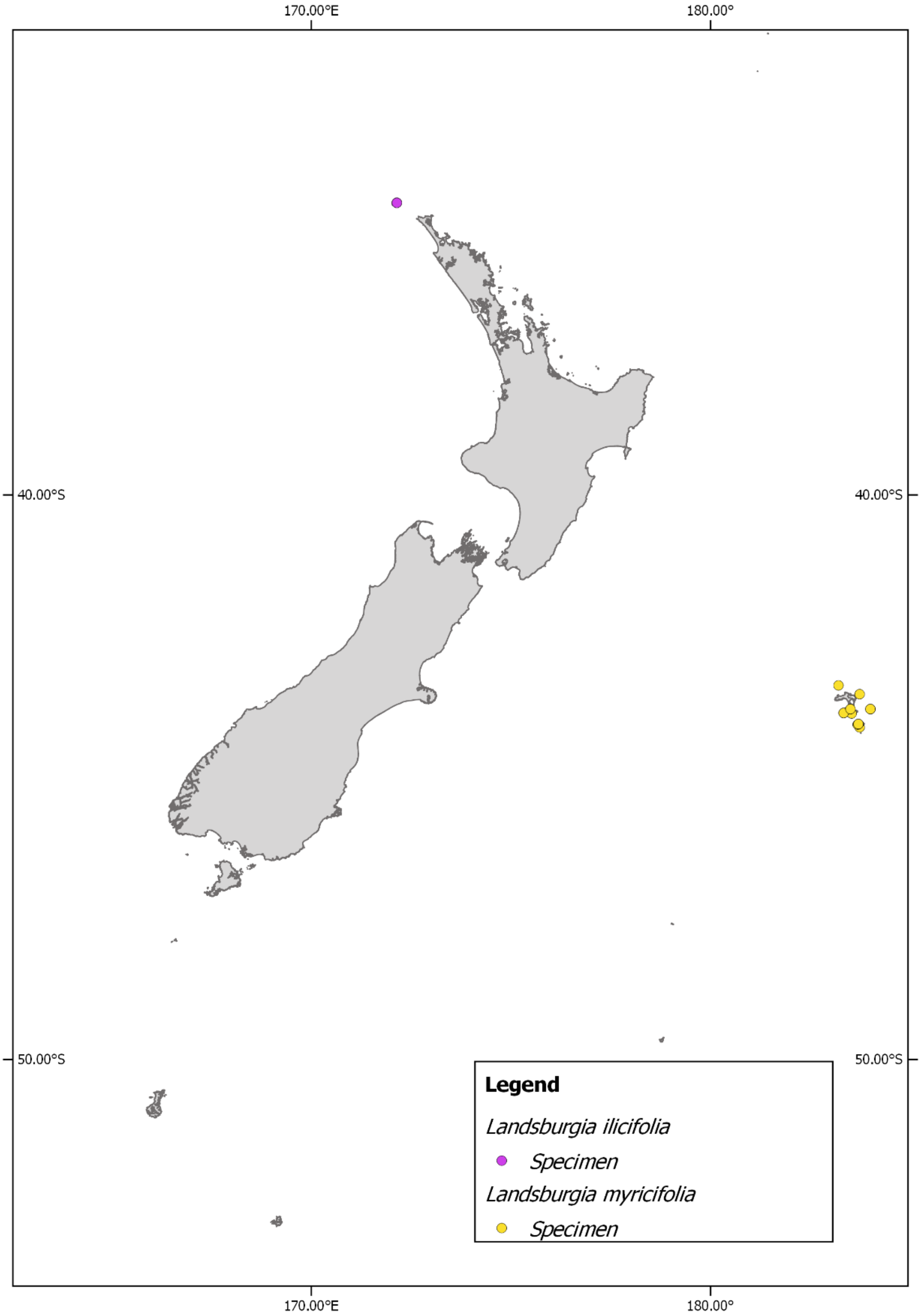


Figure A10.18. The New Zealand distribution of *Landsburgia ilicifolia* and *L. myricifolia* (Fucales) as recorded by specimen records.

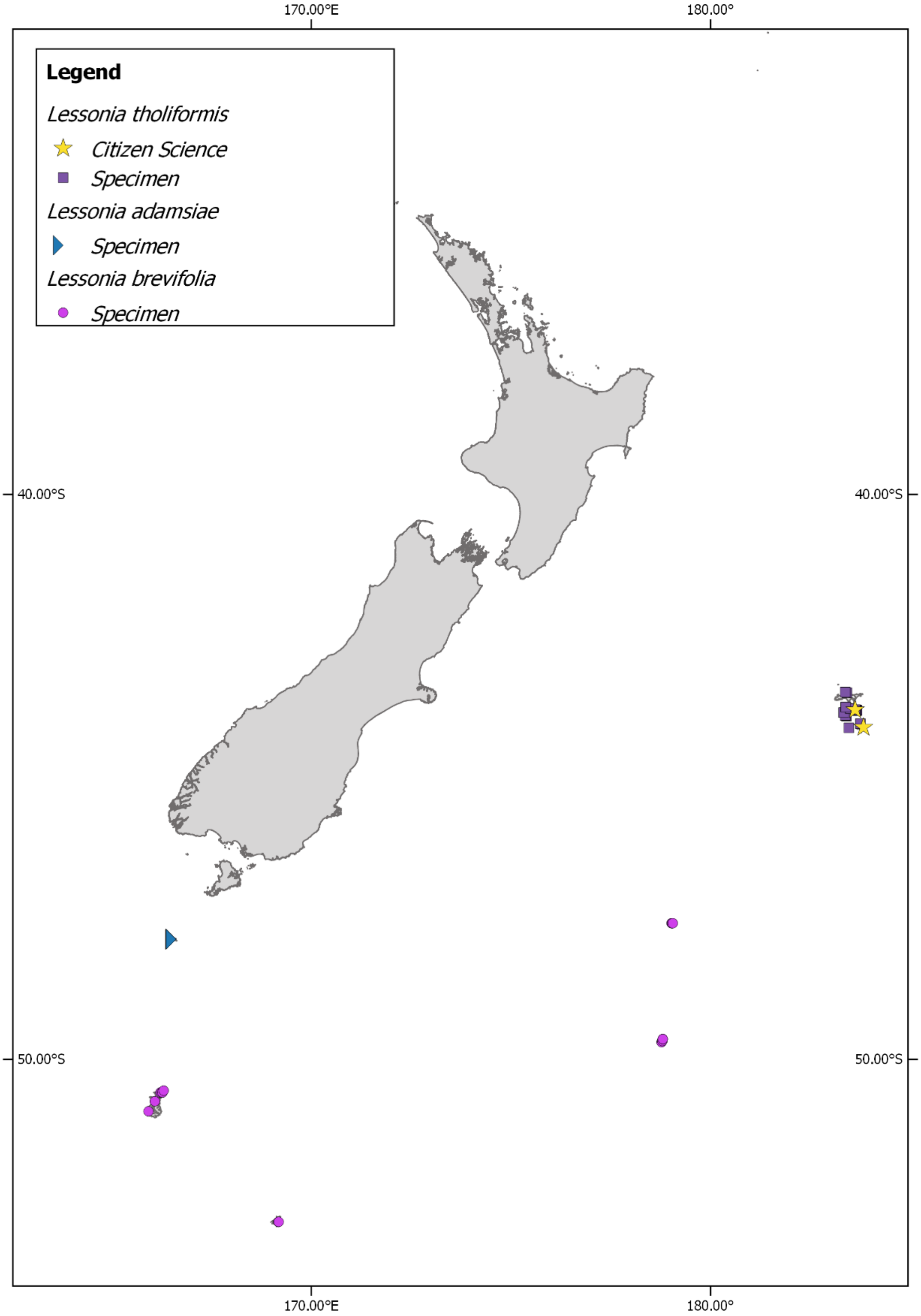


Figure A10.19. The New Zealand distribution of *Lessonia tholiformis*, *L. adamsiae* and L. *brevifolia* (Laminariales) as recorded by citizen science and specimen records.

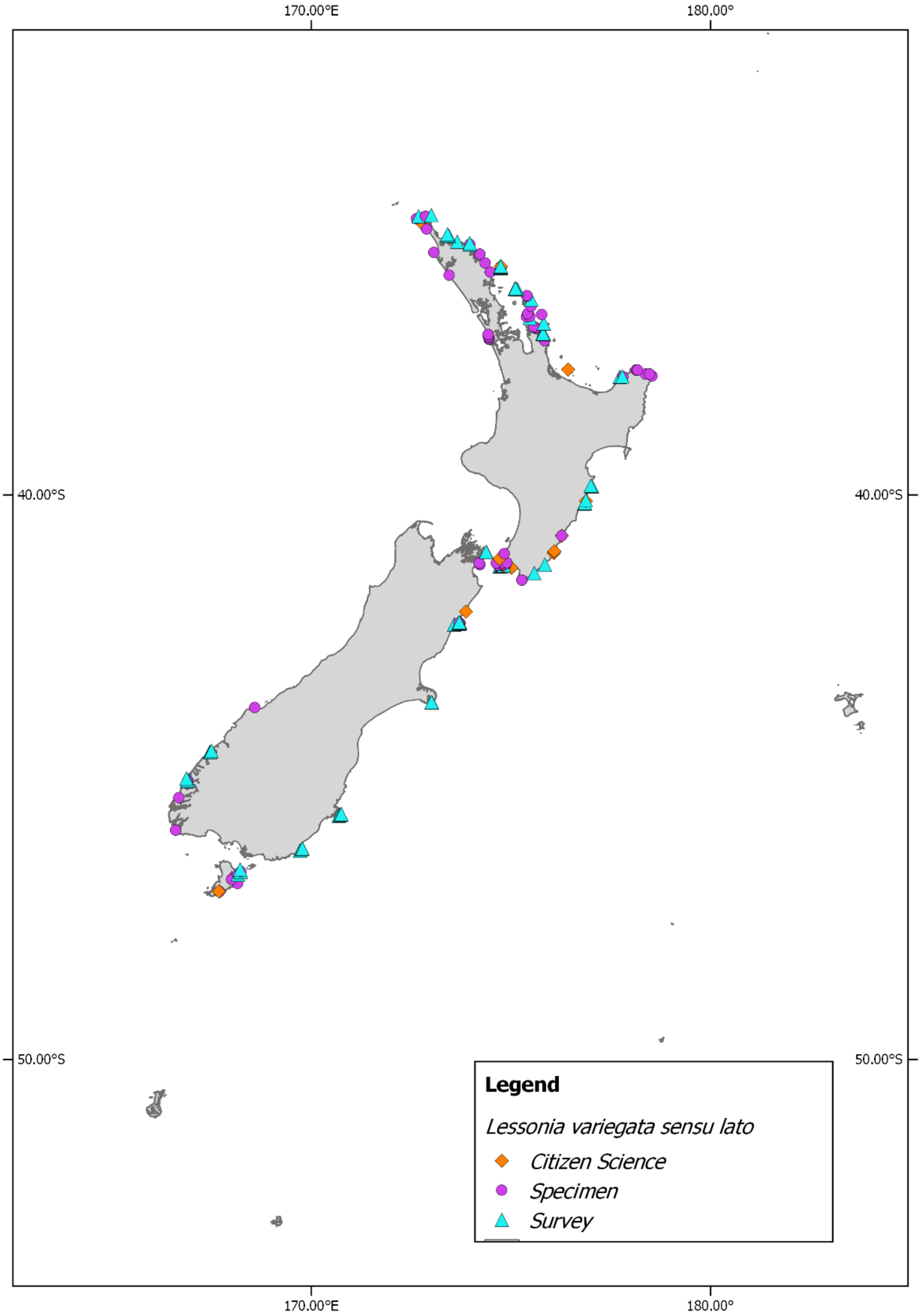


Figure A10.20. The New Zealand distribution of *Lessonia variegata sensu lato* (Laminariales) as recorded by citizen science, specimen and survey records.

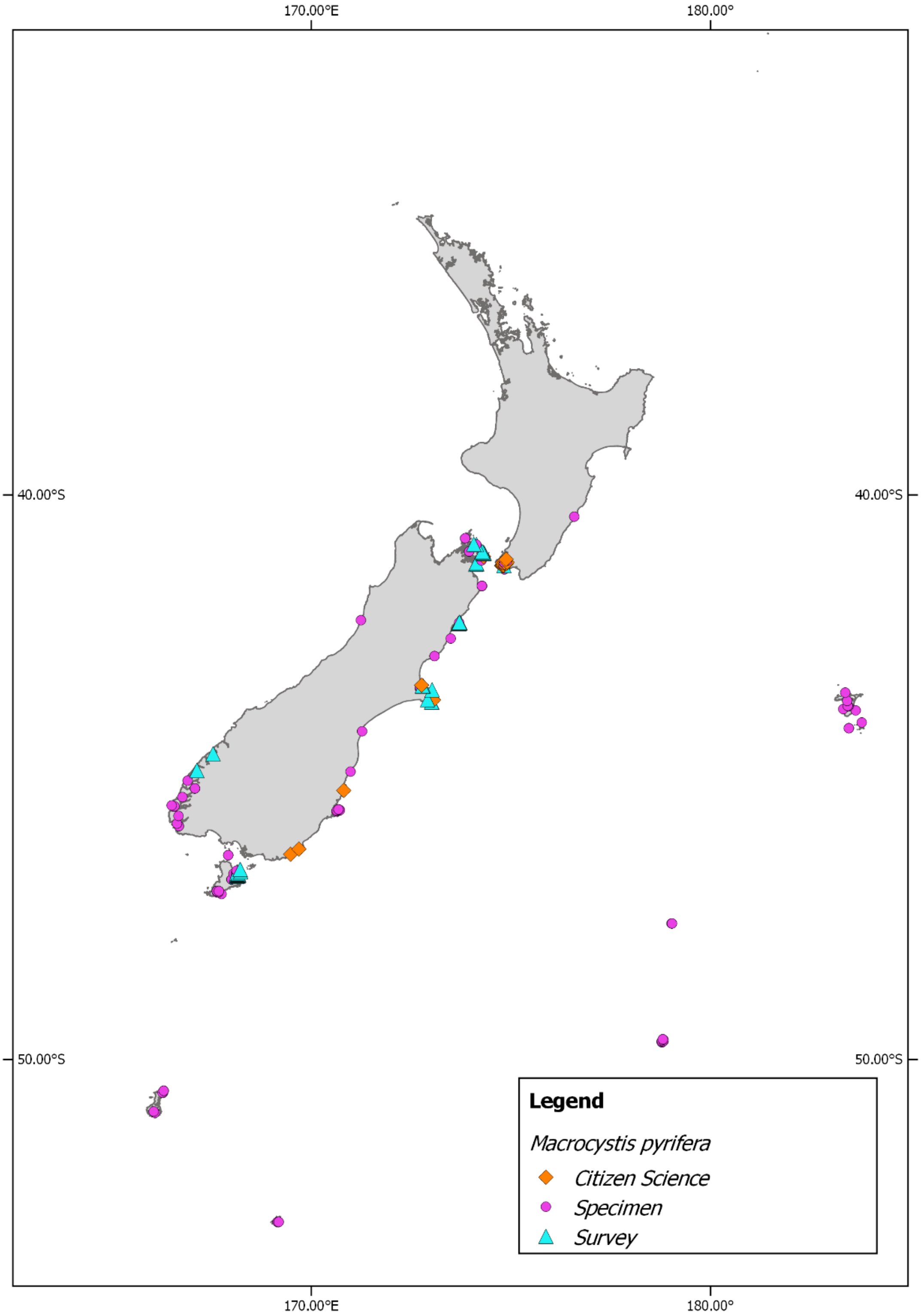


Figure A10.21. The New Zealand distribution of *Macrocystis pyrifera* (Laminariales) as recorded by citizen science, specimen and survey records.

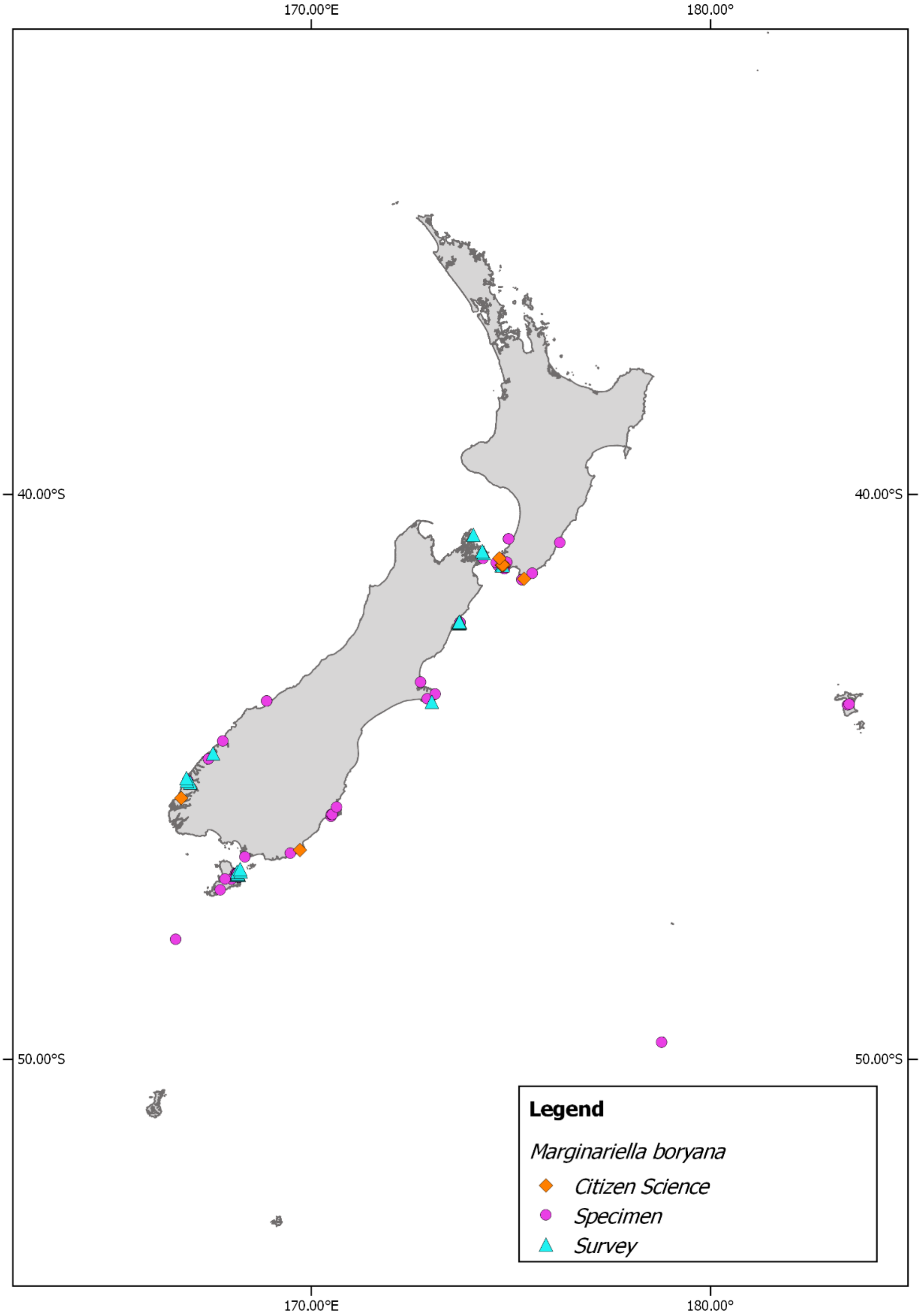


Figure A10.22. The New Zealand distribution of *Marginariella boryana* (Fucales) as recorded by citizen science, specimen and survey records.

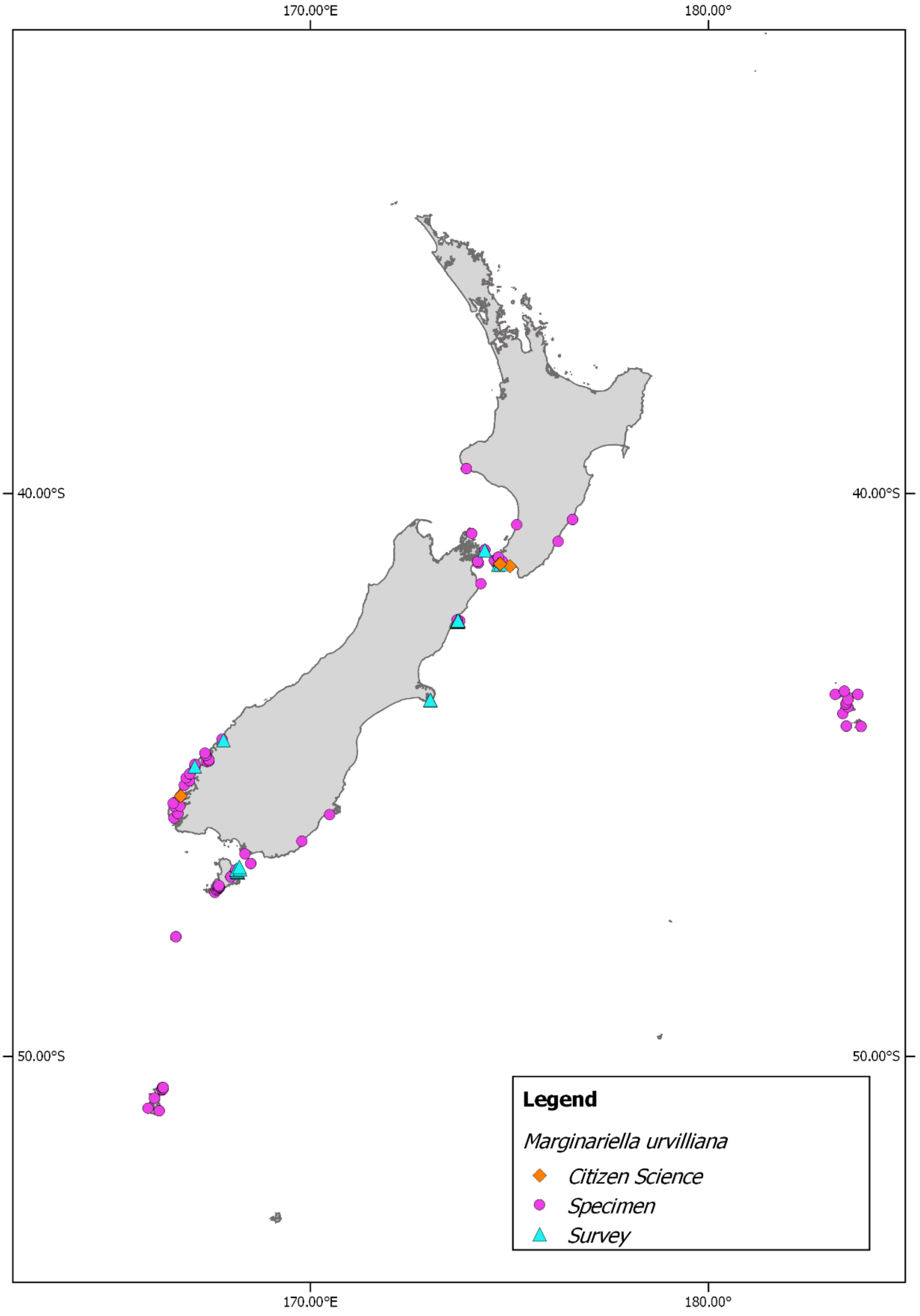


Figure A10.23. The New Zealand distribution of *Marginariella urvilliana* (Fucales) as recorded by citizen science, specimen and survey records.

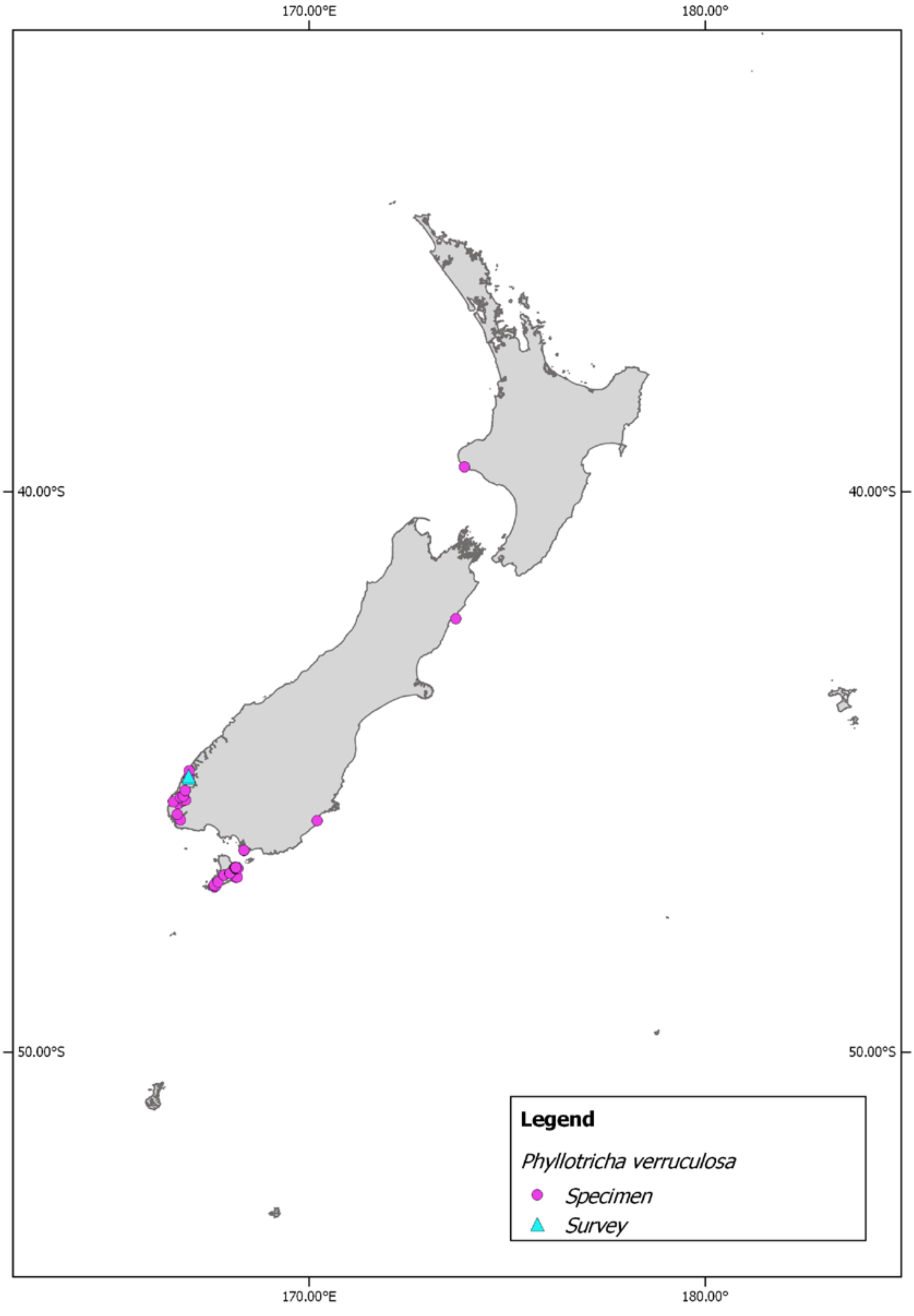


Figure A10.24. The New Zealand distribution of *Phyllotricha verruculosa* (Fucales) as recorded by specimen and survey records.

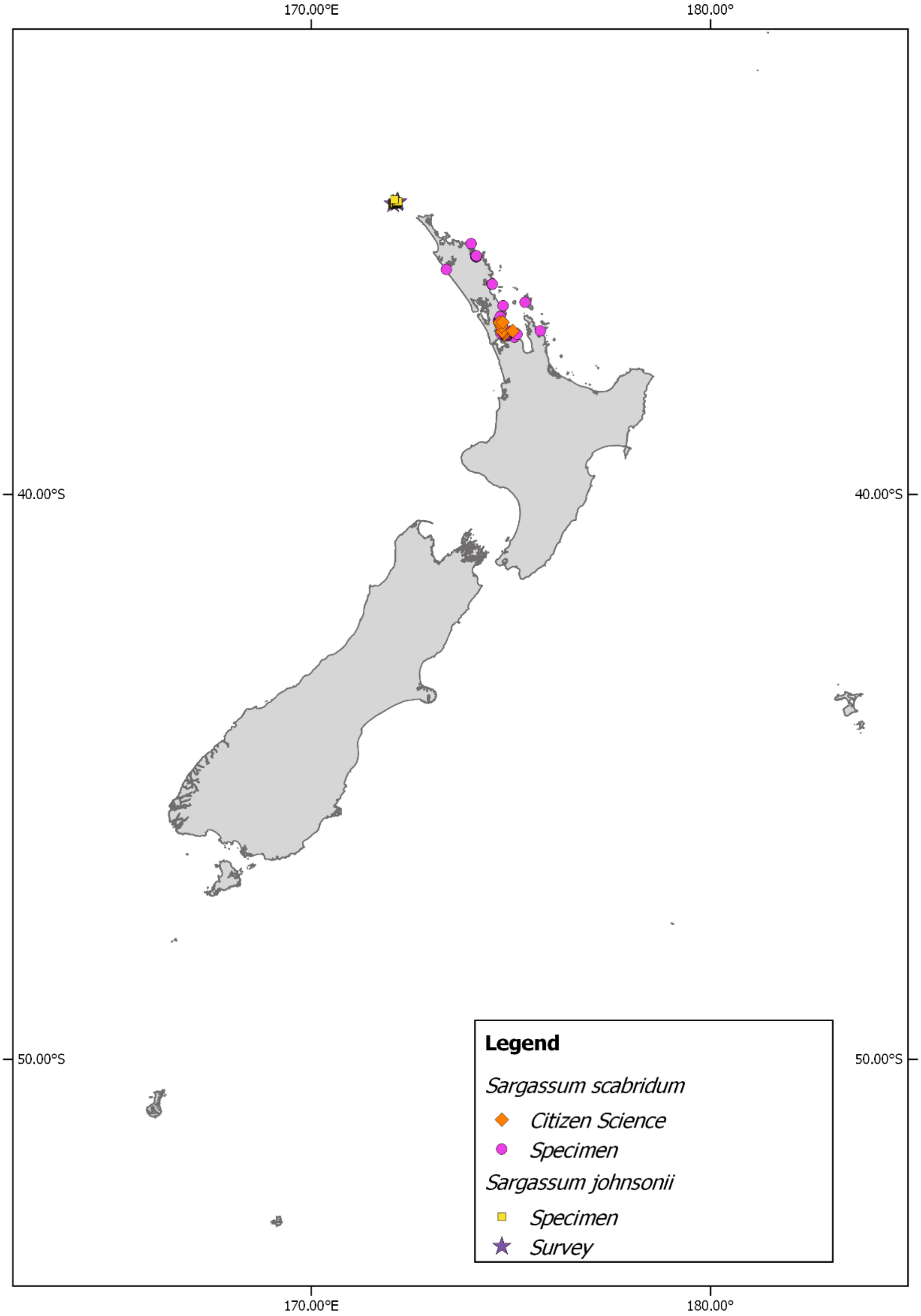


Figure A10.25. The New Zealand distribution of *Sargassum scabribum* and *S. johnsonii* (Fucales) as recorded by citizen science, specimen and survey records.

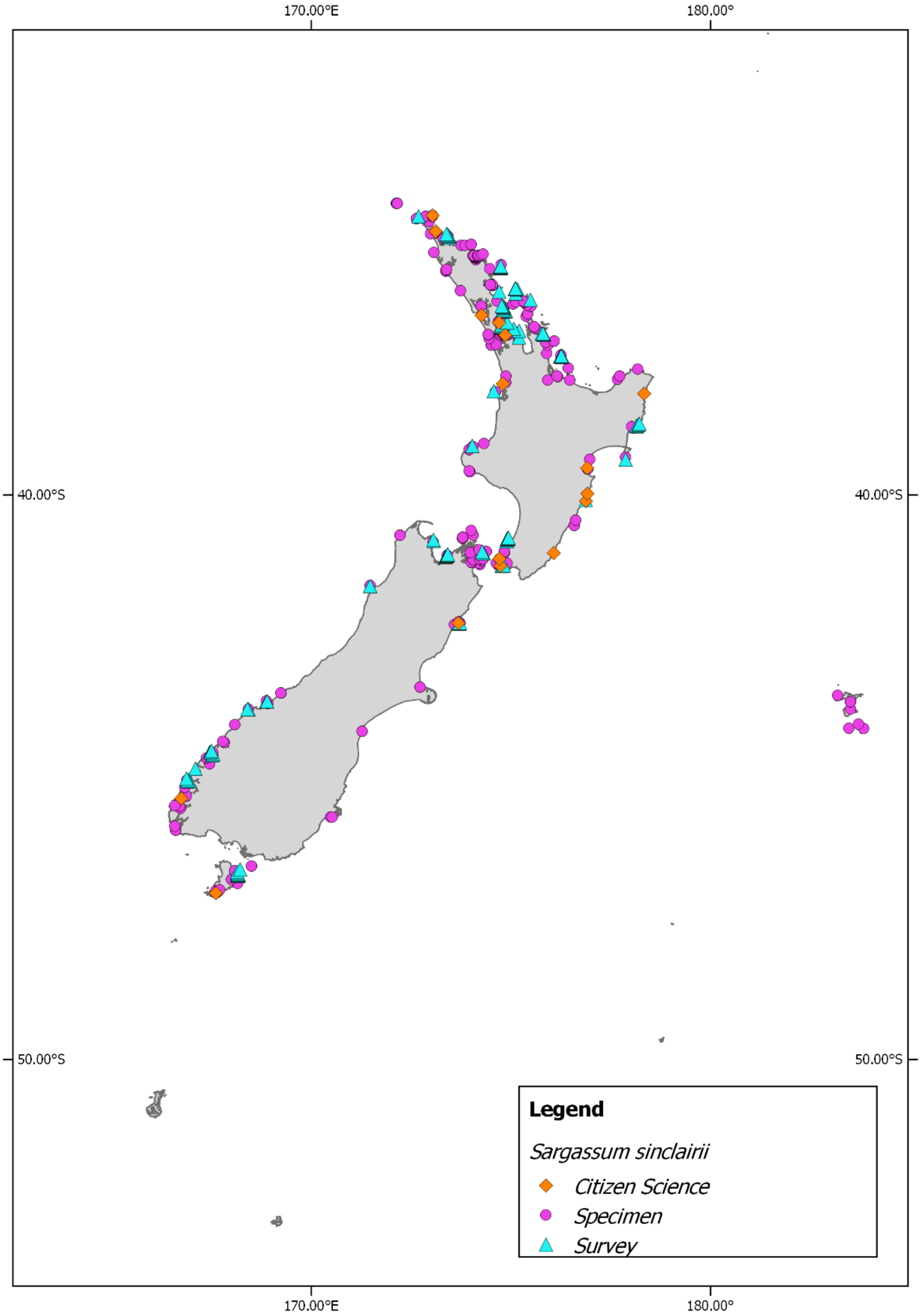


Figure A10.26. The New Zealand distribution of *Sargassum sinclairii* (Fucales) as recorded by citizen science, specimen and survey records.

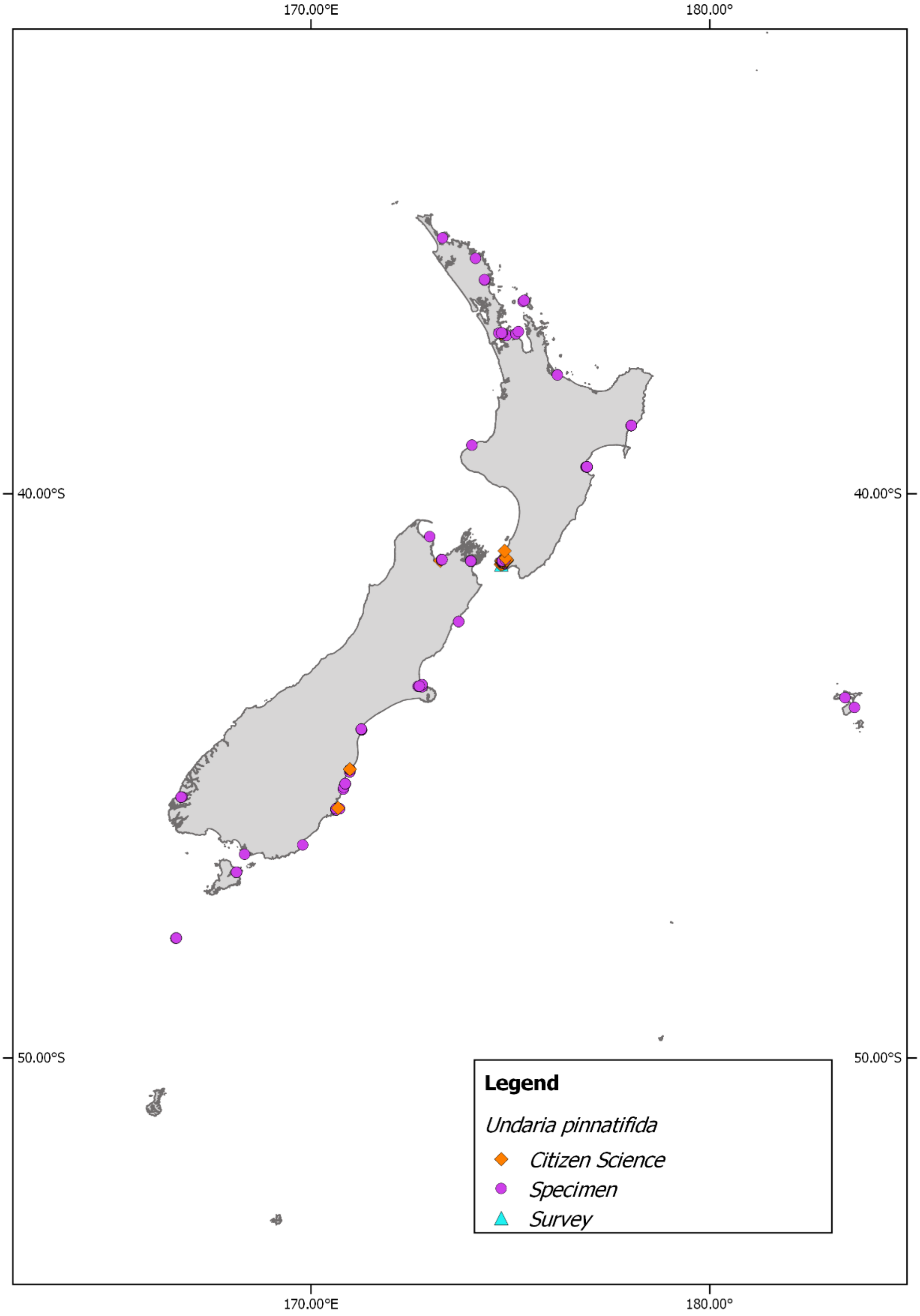


Figure A10.27. The New Zealand distribution of *Undaria pinnatifida* (Laminariales) as recorded by citizen science, specimen and survey records.

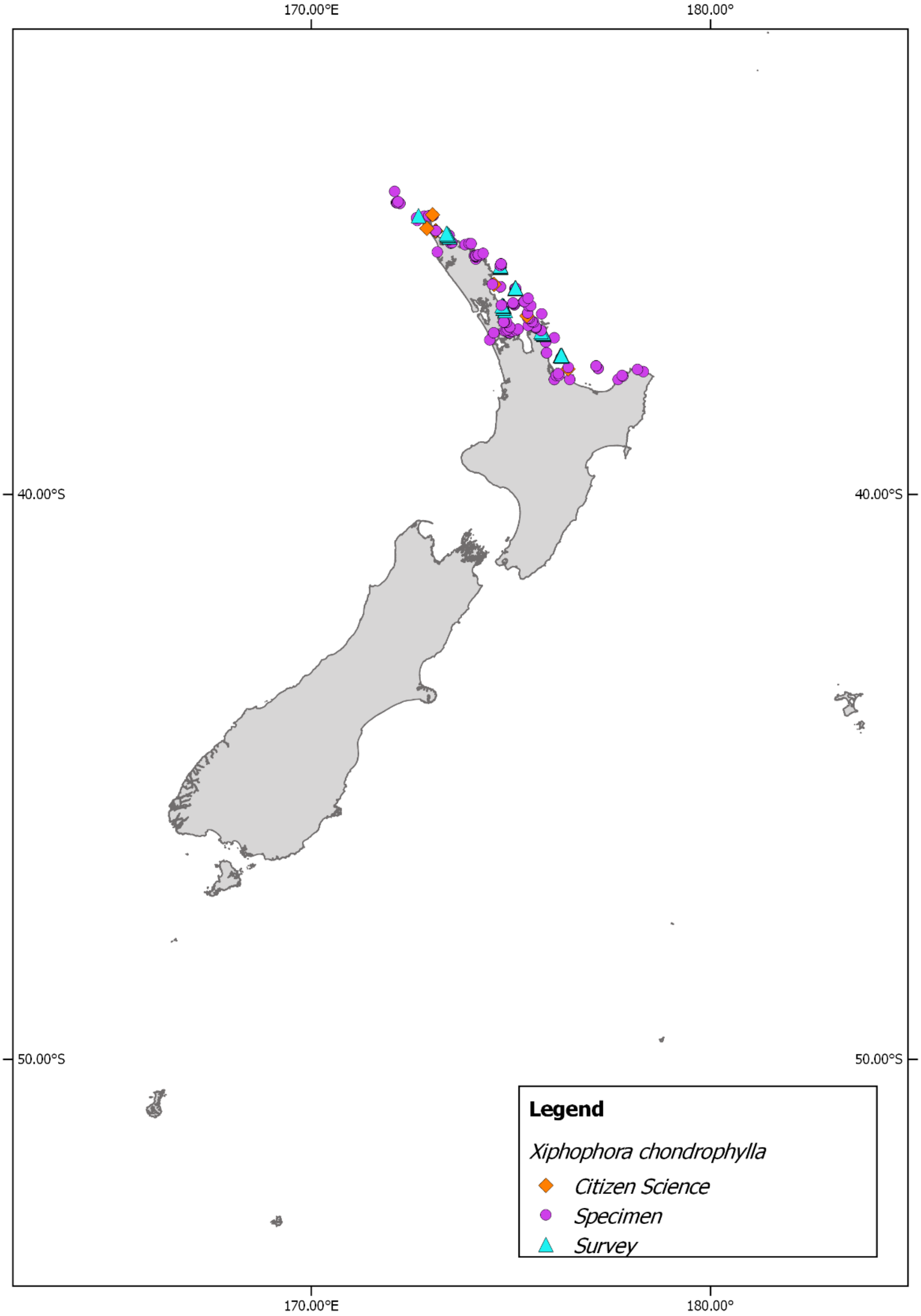


Figure A10.28. The New Zealand distribution of *Xiphophora chondrophylla* (Fucales) as recorded by citizen science, specimen and survey records.

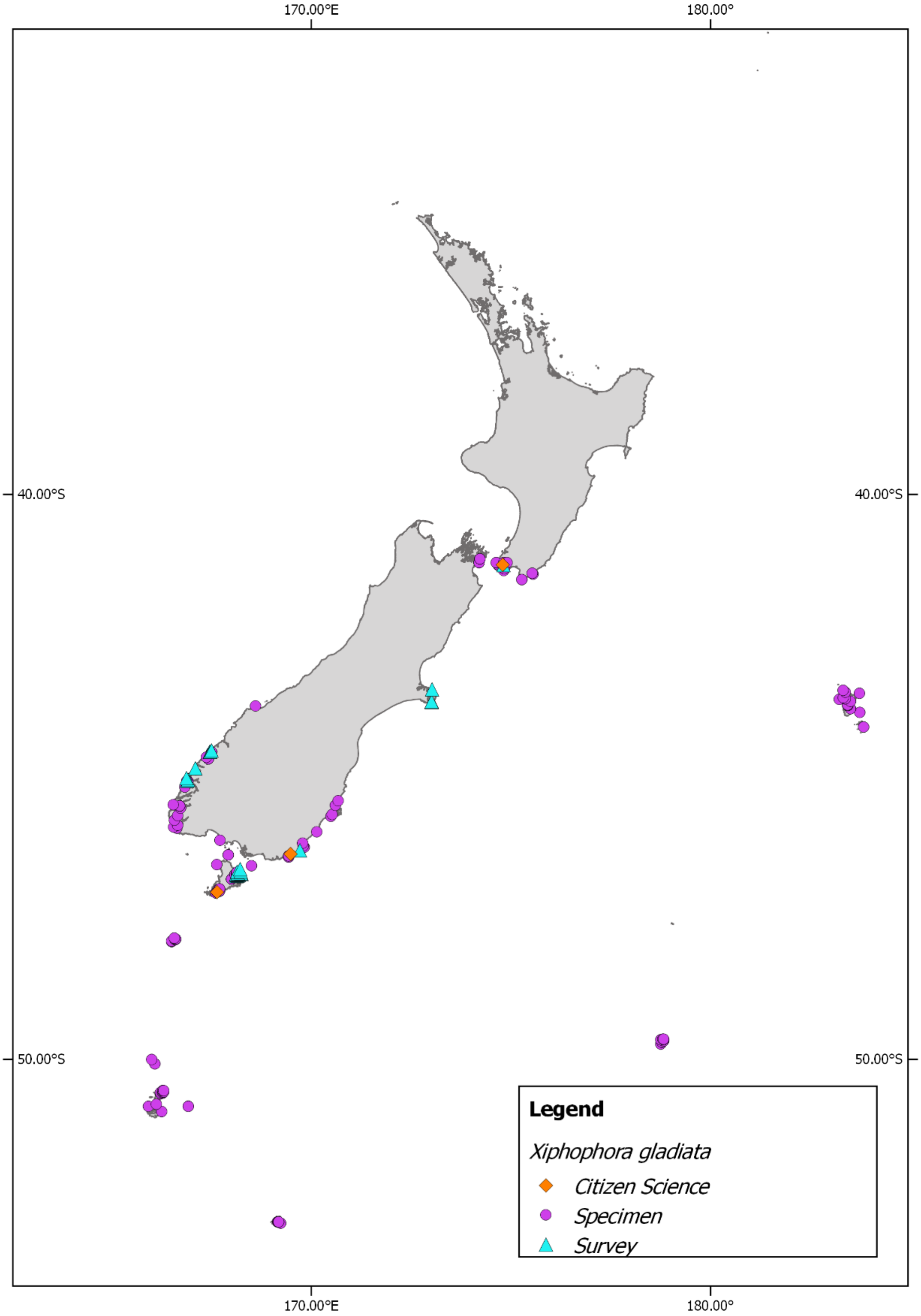


Figure A10.29. The New Zealand distribution of *Xiphophora gladiata* (Fucales) as recorded by citizen science, specimen and survey records.