Technical options for marine coastal habitat restoration in Te Tauihu

Prepared for Marlborough District Council, Nelson City Council, Tasman District Council

June 2022

Climate, Freshwater & Ocean Science

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NIWA CLIENT REPORT No: 2022170NE
Report date: June 2022
NIWA Project: ELF21401

Revision	Description	Date
Version 1.0	Draft_1	29 June 2022

Quality Assurance Statement		
	Reviewed by:	Dr. Vonda Cummings
	Formatting checked by:	Ms. Jess Moffat
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Executive summary

Te Tauihu (Top of the South Island) Councils: Marlborough District Council (MLDC), Nelson City Council (NLCC) and Tasman District Council (TLDC) are seeking advice on options for activities or actions to reverse the decline in state of coastal and marine habitats, and build resilience in these habitats, which are likely to be impacted in the coming decades by climate change.

Te Tauihu councils engaged NIWA to address the following question: *What are the best restoration options to restore and build resilience of the marine ecology of Te Tauihu?*

To achieve this, we:

1. Review the reasons that restoration may be needed in Te Tauihu.

2. Summarise existing marine restoration techniques that are relevant to Te Tauihu.

3. Identify methods or species to consider for Te Tauihu restoration activities. Projects that are 'shovel-ready' are highlighted.

Coastal-marine restoration and resilience building are sought in Te Tauihu in response to degraded marine habitats and significant loss of species biodiversity. Widespread deforestation of the region after the arrival of Europeans in ca.1850s, and subsequent pressures from marine (shipping, fishing, aquaculture, coastal infrastructure) and land-based activities (land-use change, farming, forestry, effluent discharge, etc.) have driven the changes seen today. Species and habitats now also face risks from climate change, including sea-level rise, ocean acidification and warming, and extreme weather events, that are expected to further reduce marine ecosystem resilience and accelerate biodiversity losses.

To identify activities and techniques that are relevant to restoration in Te Tauihu, we reviewed published literature, reports and articles on marine restoration science, including topic reviews and case studies for local species or related overseas species. Information was compiled from the search engine Google Scholar and complemented by papers and reports held by subject experts from councils, the Cawthron Institute and NIWA. Information was summarised into five topics that had been collectively chosen during discussions with Council staff: (1) coastal wetlands/salt marshes, (2) urban/industrial infrastructure, (3) seagrass and horse's mane weed, (4) shellfish (mussels, oysters, scallops, horse mussels), and (5) artificial reefs/wrecks.

The process of restoration can occur across a 'continuum' from unassisted or spontaneous regeneration to 'active restoration', with many actions or combinations of actions that may be considered as intermediate. Due to the widespread changes in the Te Tauihu marine environment, remedial actions are expected not fully return the system to pre-degraded conditions, therefore, remedial actions may be better termed as 'rehabilitation'. For the purpose of this review however, 'restoration' was used as an aspirational term.

Before carrying out any restoration initiative, there are some key matters to consider, including partnering with tangata-whenua/iwi and involving local communities and stakeholders from the outset. Planning should include thinking through the biology, the ecology, the environment, and mapping out pathways or "what ifs". In doing so, an understanding of the system, and the multiple interacting factors within it that can affect a restoration target/goal, will develop. It is also important to identify and minimise key stressors, and any cumulative effects that may have caused species or

system decline, preventing their natural recovery. This is because restoration may not work if the habitat is no-longer suitable. Be aware of tipping points that involve lags (hysteresis) that may require interventions to reduce ‘establishment thresholds’ or provide ‘windows of opportunity’. Carry out a risk assessment early-on, including considering biosecurity issues, and planning mitigation strategies. Marine restoration is in many respects in its infancy in Aotearoa-New Zealand and therefore there will be an element of “learning-by-doing”.

Drivers of successful restoration projects include emulation of natural systems, acknowledging that successes may be site specific, rather than a one-size-fits-all, and multi-species restoration appears to increase likelihood of success. Marine restoration, habitat creation or habitat enhancement provides demonstrable economic, societal, and ecological benefits, that once initiated can build further support. To scale up restoration, emerging/existing planting technologies could be explored, where possible using existing marine industries to advantage.

‘Shovel-ready’ priority candidates for restoration in Te Tauihu, that have been trialled successfully here and elsewhere in Aotearoa-New Zealand, include salt marsh/seagrass, shellfish (cockles/tuangi, green-lipped mussels), and artificial structures to reduce coastal erosion. Examples of recent successes include salt marsh restoration in Waikawa and Maketū, seagrass restoration in Whangarei Harbour, the promising results of green-lipped mussel restoration trials in the inner Pelorus Sound/Te Hoiere, and living terracing installed as intertidal habitats in the lower Maitai River.

Shellfish restoration is the top priority because of the areal extent of historic degradation. Shellfish are essential to ecosystem function and stability (e.g., filtration, nutrient exchange), providing ecosystem services which feedback to soft sediment plants (microalgae, macroalgae and seagrasses); together they can reinforce each other’s coexistence. Restoration of such habitats are very likely to produce additional benefits to fisheries production (shellfisheries, fishes), and contribute to reducing climate change risks (through carbon sequestration and through the greater resilience provided by healthy ecosystems). Successful restoration of shellfish and seaweeds/grasses is more likely if their soft sediment habitats can also be protected from benthic disturbance and if terrestrial sediment discharge into coastal marine areas is reduced.

Initiatives to address coastal erosion is the second priority. This includes restoring salt marshes and seagrass, and softening coastal and estuarine edges (e.g., using terraces, living margins/buffer areas) in locations where coastal squeeze from sea-level rise allows. At locations where intertidal space is limiting, installation of offshore artificial reefs (e.g., temporary/permanent shellfish reefs, or designer-built structures) could be used to buffer coastlines from extreme weather events and inundation. Urban infrastructure and artificial reefs can be designed or modified to enhance habitat, biodiversity and ecological service outcomes – increasing resilience and fisheries production. Artificial structures can also be used either as permanent or temporary living structures (e.g., combined with shellfish or algae) to create ‘windows of opportunity’ to improve high turbidity environments that may be preventing success of onshore or shallow restoration projects (e.g., salt marsh or intertidal/subtidal seagrass restoration).

Recent restoration successes and increasing knowledge of climate change risks provide encouragement and impetus to continue broadening the scope and scale of marine restoration efforts in Te Tauihu.

1 Introduction

The Te Tauihu (Top of the South Island) coastline is the most extensive and ecologically diverse in Aotearoa-New Zealand (A-NZ). It has undergone significant human modification, leading to widespread and ongoing decline of marine habitats, ecosystems and the ecosystem services. Te Tauihu Councils are seeking advice on options for restoration to reverse the decline of coastal and marine habitats, and to build resilience in these habitats, which are also likely to be impacted in the coming decades by climate change.

Three councils have adjoining boundaries extending out to the edge of the territorial sea in the Marlborough Sounds: Tasman/Te Tai-o-Aorere and Golden Bays/me Mohua, Cook Strait/Raukawa Moana and the Pacific Ocean/Moana nui a Kiwa. The Te Tauihu Councils are committed to exploring how coastal and marine restoration activities can improve marine biodiversity across these three regions. This report provides a review of coastal restoration options and their utility to Te Tauihu.

1.1 Background:

Te Tauihu Councils are responsible for Resource Management Act functions in this extensive area, and for safeguarding the life supporting capacity of the water and ecosystems.

The New Zealand Coastal Policy Statement (NZCPS, 2010) includes a focus on marine biodiversity:

- Objective 1 requires protection of the integrity, form, functioning and resilience of the coastal environment, in particular biological systems.
- Policy 11 recognises the need to protect indigenous biological diversity and ‘avoid, remedy or mitigate other adverse effects of activities’.

Each of the three councils are at different stages of regional planning processes which give effect to the objectives and policies in the NZCPS. All three councils are committed to the identification of sites of significant marine biodiversity, and to the protection of marine biodiversity.

The proposed National Policy Statement for Freshwater Management and the proposed National Environmental Standard for Freshwater make provision for increased protections for coastal wetlands from harmful activities (e.g., seabed disturbance, removal of indigenous vegetation; (Tan et al. 2020). The draft of the soon-to-be-published National Policy Statement for Indigenous Biodiversity (NPS-IB) recognises that "The maintenance of indigenous biodiversity may also require the restoration or enhancement of ecosystems and habitats" (Urlich 2021).

Restoration in the region is aligned with the Te Tauihu Intergenerational Strategy (Wakatū 2020) and the Kotahitanga mō te Taiao Strategy (KMTT 2020). Both have called for restoration of the Te Taiao/natural world with the need for “wide-scale change of behaviours and practices across society to reduce our environmental footprint”, ensuring “ecological connections and resilience are protected, restored and enhanced.”

1.1.1 Environmental degradation

There has been a continued loss of marine habitats and species biodiversity in Te Tauihu, especially following widespread deforestation after the arrival of Europeans in ca.1850s (e.g., Michael et al. 2015; Handley 2016; Davidson et al. 2019). Widespread habitat change has been caused by a multitude of individual pressures from marine activities (shipping, fishing, aquaculture, coastal infrastructure) as well as land-based activities (land-use change, farming, forestry, effluent discharge, etc.) (MacDiarmid et al. 2016a; MacDiarmid et al. 2016b; Handley et al. 2020a; Handley et al. 2020c).

1.1.2 Climate-related change

Projections and measured impacts

The effects of human induced climate change are already being recorded in the marine environment in A-NZ (Lawrence et al., 2022). Of the many marine environmental factors projected to change, warming temperatures, ocean acidification, coastal erosion and sediment loading, salinity and oceanographic conditions (stratification, circulation) were considered most important, along with new threats (e.g., increased disease, invasive species). As an example, for the iconic/taonga species pāua (*Haliotis iris, H. australis*), warming waters and/or ocean acidification are expected to impact various life stages of paua and their various food sources; increased storm frequency and severity will disrupt harvest, and paua habitats and food sources will be affected by increased coastal sedimentation.

In nearby Tasmania, changes in large-scale oceanography are affecting the structure of nearshore zooplankton communities and elements of the pelagic system, with implications for benthic (rocky reef) and pelagic ecosystems. (e.g., Johnson et al. 2011; Tait et al. 2021). These include a regional decline in the extent of dense beds of giant kelp (*Macrocystis pyrifera*), changes in the distribution of nearshore fishes, and range expansions of northern warmer-water species. The latter include commercially important invertebrate species that have colonised Tasmanian coastal waters. Similar large-scale coastal changes have been recorded for kelp in New Zealand. Analysis of satellite imagery showed declines in the surface canopies of the *M. pyrifera* in the New Zealand coastal zone, and especially in the giant kelp's northern range that includes the Marlborough Sounds and Wellington's south coast (Tait et al. 2021). Kelp loss was attributed to marine heat waves in 2017/18, with notable negative effects across the coastal range of this foundation species. These results demonstrated the effects of multiple stressors across latitudinal gradients (Tait et al. 2021), with temperature-induced kelp loss greater when water clarity was poor (Tait et al. 2021). Such pressures on marine ecosystems are expected to increase considerably in the next few decades and are projected to lead to further loss of marine biodiversity and severe degradation of ecosystem functioning (Papadopoulou et al. 2017). Although A-NZ's oceans and marine habitats (e.g., mangroves, sea-grass meadows, kelp forests), may take up and store more CO_2 than our forests (MacDiarmid et al. 2013), rising sea temperatures may reduce the capacity of the oceans to absorb CO_2 (MfE 2020).

Climate change imperative for restoration

A Ministry for the Environment (2020) risk assessment considered climate change driven sea-level rise and extreme weather events to pose the greatest threats to marine environments, and also projected increased coastal inundation and erosion. Those stressors are projected to exacerbate the already greater than 10-fold rates of sediment discharge to coastal environments in Te Tauihu. Unless addressed, they are very likely to further reduce marine ecosystem resilience and accelerate biodiversity losses.

The most recent Intergovernmental Panel on Climate Change (IPCC) report states that coordinated and well-monitored habitat restoration can reduce non-climatic impacts and increase resilience[1]. The Convention on Biological Diversity's (CBD) Strategic Plan for Biodiversity projects that current

[1] https://www.ipcc.ch/report/ar6/wg2/

commitments to reducing CO_2 emissions are insufficient to limit global atmospheric temperature rise below 1.5°C (Northrop et al. 2021). Climate scientists state that warming above 2°C may trigger ‘tripping cascades’ that create planetary conditions incompatible with human existence (Lenton et al. 2008; Steffen et al. 2018; Keen 2021). In response to the United Nations CBD’s global goals and targets, the New Zealand Biodiversity Strategy “Our Chance to Turn the Tide” (MfE 2000) calls to halt biodiversity decline to improve resilience, and to facilitate natural adaptation to climate-change (DBD 2020). In the search for long-term and practical solutions to tackle climate change, the IPCC identified Nature-based Solutions (NbS) as a critical reduction and mitigation approach (Beck et al. 2001; Bindoff et al. 2019). NbS include sustainable management and restoration of ecosystems, that enhance both biodiversity and human well-being (Walters et al. 2016; Seddon et al. 2019).

1.2 Restoration

For the purpose of this review, **ecological restoration** is defined as “the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed” (Society of Ecological Restoration; (Clewell et al. 2004). The process of restoration can be presented along a ‘restorative continuum’ (Gann et al. 2019), from reducing the causes of decline to full ecosystem restoration. Restoration can range from “passive/unassisted/spontaneous restoration” to “active restoration”, with many actions or combinations of actions that may be considered as intermediate between the two. Most importantly, to ensure restoration success, simultaneous or sequential steps must first be taken to rehabilitate the ecosystem, including by removal of threats, to allow for restoration options to work (Papadopoulou et al. 2017).

Other terms associated with restoration include:

> **Ecosystem Recovery**: “the ability of a habitat, community or individual … species to redress damage sustained as a result of an external factor” (Elliott et al. 2007).
>
> Habitat **Enhancement** (Elliott et al. 2007) is a type of remedial action ‘to rectify or make good’ where complete restoration may not occur (Bradshaw 2002).
>
> **Habitat Creation** describes an anthropogenic intervention which produces a habitat that was not previously there. For example, artificial reefs placed on an otherwise sandy sea bottom create new habitat aiming to increase the biodiversity of an area (Elliott et al. 2007).
>
> **Rehabilitation** aims to reinstate a level of ecosystem functionality for renewed and on-going provision of ecosystem services and goods, rather than full ecological restoration (Gann et al. 2019).

It is important to note that due to the widespread changes in the marine environment in Te Tauihu, restoration actions are unlikely to fully return pre-degraded conditions, therefore, remedial actions may be better termed as ‘rehabilitation’. For the purpose of this review however, ‘restoration’ was used as an aspirational term.

The most significant causes of species decline and extinction in marine ecosystems have been attributed to habitat loss and degradation, and it has been widely recognised that a range of different restoration actions are essential to halt further decline and reverse the current trends (Papadopoulou et al. 2017). A review of marine habitat restoration in Europe was undertaken to identify what prevented restoration projects from being successful. Of 498 publications (projects), 50-70% were successful. Restoration failures were linked to methodological aspects, overlooking important site characteristics, and local threats.

The science and advice of marine restoration is most developed for intertidal species and least developed for deep water species due to subtidal marine habitats being “out of sight, out of mind”, and the larger costs associated with restoring species or habitats in deeper waters (Papadopoulou et al. 2017).

In response to continuing loss of biodiversity linked to habitat degradation, the Convention on Biological Diversity and EU Biodiversity Strategy called for a 15% restoration target by 2020 (Papadopoulou et al. 2017). The United Nations has subsequently declared a “Decade on Ecosystem Restoration” aiming to prevent, halt, and reverse the degradation of ecosystems on every continent and in every ocean. (https://www.decadeonrestoration.org/).

1.3 This project

Te Tauihu councils engaged NIWA to address the following question: *What are the best restoration options to restore and build resilience of the marine ecology of Te Tauihu?*

To achieve this, we:

1. Briefly review the reasons that restoration may be needed in Te Tauihu (see above).
2. Summarise existing marine restoration techniques that are relevant to Te Tauihu.
3. Recommend methods or species to consider implementing for Te Tauihu restoration activities. Projects that are ‘shovel-ready’ are highlighted.

The review focusses on restoration of terrestrial margins (coastal wetlands/salt marshes), intertidal habitats (seagrass, horse’s mane reed), biogenic habitats (mussels, oysters, scallops, horse mussels) and artificial reefs (subtidal reef habitats).

This advice will help guide future restoration efforts in the marine domain of Te Tauihu.

2 Methods

Published literature, reports and articles on marine restoration science, topic reviews, and case studies for local species or related overseas species were compiled from the search engine Google Scholar and complemented by papers and reports held by subject experts from councils, Cawthron Institute and NIWA. Information was summarised for each of the following topics: (1) coastal wetlands/salt marshes, (2) Urban/industrial infrastructure, (3) seagrass and horse's mane weed, (4) shellfish (mussels, oysters, scallops, horse mussels), and (5) artificial reefs/wrecks. These topics were collectively chosen during discussions with Council's staff.

Each topic is presented in a separate section. At the beginning of the section, the status of knowledge and advice relevant to that habitat or species is first outlined in a Table (Tables 3-1 – 3-5). The table summarises knowledge in the following categories:

1. Potential use or value, suitable locations to try in Te Tauihu;
2. Status and why the habitat is important;
3. The main, most recent, and/or successful techniques and methods for restoration;
4. New techniques, approaches, or technological innovations that could make a difference in up-scaling restoration actions;
5. Major barriers or deal breakers that prevent scaling-up restoration;
6. Timescales for restoration;
7. Other points/key messages relevant to the habitat;
8. Examples of where this has been tried before and the likelihood of success in Te Tauihu.

Different definitions and concepts are discussed throughout the sections within this report to illustrate the breadth of available restoration options for consideration by practitioners and managers, and their utility. The pros and cons of the different approaches are discussed, along with their potential "value" to society.

Where available, restoration **case studies** are presented. The case study examples are given in text boxes, distinguished by colour - blue for A-NZ studies and green for overseas studies.

At the end of each topic table, the potential likelihood of success of the restoration approach to enhance or improve Te Tauihu ecosystem health is indicated by a coloured tick. Green (✓) = high, Amber (✓) = moderate, Black (✗) = unlikely to be of use.

3 Review

3.1 Coastal wetlands/salt marshes

Table 3-1: Summary of the salient points from the review of coastal wetlands/salt marshes, with relevance to restoration activities in Te Tauihu. The remainder of Section 3.1 details the information behind this table.

1.	**Potential use or value, suitable locations to try in Te Tauihu**
▪	Habitat creation, enhancement, or restoration
▪	Where to try: Degraded/silted locations (e.g., Waimeha Inlet, Moutere Inlet), locations where salt marsh has been removed, drained, or affected by coastal squeeze.
2.	**Status and why the habitat is important**
▪	Status: Impacted/declining (drainage, reclamations, coastal developments, sea-level rise)
▪	Nature based solution (NbS)
▪	Provides habitat and food for invertebrates, fish (e.g., whitebait), and birds
▪	Natural buffer between land and sea, protecting against large storms and tides. Reduces sediment erosion and runoff.
▪	Sequesters carbon and sediment, reduces eutrophication (‘Blue carbon’)
▪	Lower cost cf. hard infrastructure (walls), self-sustaining, low maintenance
3.	**Main, most recent, or successful restoration techniques and methods used**
▪	Weed control, ground preparation, planting designs
▪	Softening edges, terraces, living setbacks
▪	Provision of sills or offshore artificial reef elements to reduce erosion during establishment
▪	Root-trainer plantings
4.	**New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions**
▪	Biodegradable elements to enhance establishment success (‘windows of opportunity’)
▪	Shellfish or artificial reef restoration elements installed offshore to attenuate waves/ reduce erosion
▪	Floating islands
5.	**Major barriers or deal breakers that prevent scaling-up restoration**
▪	Availability of experienced practitioners/subject experts
6.	**Timescales for restoration**
▪	Seasonal, several months to years
7.	**Other points/key messages relevant to the habitat**
▪	Very important ‘blue carbon’ option to reduce effects of climate change

8. Examples of where this has been tried before, likelihood of success
▪ Successful salt marsh and streambank planting at Waikawa Estuary (Salt Ecology) ✓
▪ Success at Ōngātoro/Maketū Estuary. Re-channelling of the Kaituna River, with wetland restorations at Te Pā Ika and restoration of neighbouring Papahikahawai Island (Bay of Plenty Regional Council/Toi Moana)
– Increased invertebrate biodiversity + improved sediment qualities + reduced nuisance algae ✓
Likelihood of success: High if site prepared properly, erosion reduced, weed control undertaken, grazing controlled, community engaged. ✓

3.1.1 Why restore wetland/salt marshes?

The protection and restoration of coastal wetlands, including salt marshes and seagrass meadows, are important to protect coasts from flooding and erosion, provide habitat for commercially important and endangered species (invertebrates, fish and bird-life), and improve water quality (Beck et al. 2001; Mcleod et al. 2011a; Duarte 2017; Papadopoulou et al. 2017) (Table 3-1).

Coastal wetlands are also one of the most efficient carbon sinks on earth, sequestering up to 40 times faster than terrestrial ecosystems, despite occupying ~0.2% of the ocean surface (Nellemann and Corcoran 2009; Mcleod et al. 2011a). Protecting or restoring wetlands could enhance coastal protection and food security through provision of habitat for invertebrates and fish (Macreadie et al. 2019; Guthrie et al. 2022). Coastal wetlands, salt marshes and seagrass meadows have also been proposed as a cost-effective and scalable NbS or Blue Carbon Ecosystem (BCE) to reduce climate change (Costa et al. 2021a; Macreadie et al. 2021). A comparison of the costs of NbS to hard engineering structures has shown that salt marshes and mangroves could be 2 to 5 times cheaper than a submerged breakwater in certain conditions (Papadopoulou et al. 2017).

In the UK, rising sea-levels are projected to completely flood thousands of hectares of salt marsh and mudflats over the next 50 years (Papadopoulou et al. 2017). Those habitats provide a natural buffer against the sea, protecting against large storms and tides. It is estimated that 30% of remaining wetlands could be lost by 2100 due to threats from sea-level rise causing coastal squeeze if landward migrations are not accommodated (Schuerch et al. 2018).

Aotearoa New Zealand

In A-NZ, coastal wetlands are considered taonga by Māori as they hold historical, cultural, economic and spiritual significance. They provide food, plants for weaving, medicines, dyes, and canoe landing sites (Cromarty and Scott 1996). Despite their significance and ecological benefits, salt marshes now account for the scarcest wetland type in A-NZ, having been affected by drainage/channelisation and reclamations in the coastal zone (Thomsen 1999).

Salt marshes are an important habitat for galaxiid fishes including inanga *Galaxias maculatus*, whose juveniles contribute some 90% of "whitebait" catch (McDowall 1965). Inanga egg survival is greatest in riparian vegetation with dense stems and a thick aerial root-mat (*Juncus edgariae, Schedonorus phoenix,* and *Holcus lanatus*) that provides a cooler and more humid micro-environment, as well as shade from lethal ultra-violet light (Hickford et al. 2010).

About 30% of the salt marsh in the Tasman and Golden Bay estuaries (excluding Abel Tasman) has been lost since 1900 (Robertson & Stevens 2012). Moutere and Ruataniwha estuaries have suffered the largest loss at 50% and 40% respectively. Reclamation of high value habitat has severely lowered the natural assimilative capacity of these estuaries which has led to increased sedimentation rates in tidal flat areas and low habitat quality. In the Hauraki Gulf (HG), salt marsh extents have declined more than 90% due to land practices and reclamations (Morrison 2021). Degradation of salt marsh habitat has also been reported in Maketū (see below), in Waikawa Estuary (Table 3-1), and degraded locations like the Havelock Estuary (Stevens and Robertson 2014).

3.1.2 Successful techniques

Interests in restoring or establishing marshes have been to improve water quality and restore lost or damaged habitat (Thomsen 1999; Thomsen et al. 2005). A wetland information fact sheet and a restoration guide have been published by Auckland Regional Council (Bergin 1994; ARC. 2000a; ARC. 2000b).

Restoration can be achieved through natural colonisation, or by planting seeds, seedlings, or plants that have been divided. Seeds of the most common salt marsh species are available from commercial nurseries (e.g., sea rush *Juncus maritimus* and oioi/jointed rush reed *Leptocarpus* (*Apodasmia*) *similis*), local voluntary groups/trusts (e.g., Guardians of Pāuatahanui Inlet), and council nurseries (Thomsen et al. 2005; Morrison 2021). Species choice is important, and stock may be obtained from either natural populations or from nurseries (Cronk and Fennessy 2000). Despite considerable knowledge on the ecology and structure of salt marsh communities (Cockayne 1967; Partridge and Wilson 1987, 1988; Thomsen et al. 2005), early trials had variable success with damaged salt marshes slow to recover (Thomsen et al. 2005).

Methods and approaches used to assess the potential success of salt marsh species for transplanting in restoration programs include: field surveys and transplants selecting species based on salinity characteristics (e.g., Partridge and Wilson 1988); quantifying how environmental factors including soil characteristics affect survival, growth and reproduction (e.g., Keddy 1990); evaluating the effects of environmental tolerance of individual species and communities using laboratory and field microcosms (e.g., Weiher and Keddy 1995; Callaway et al. 1997; Fraser and Keddy 1997; Trnka and Zedler 2000); and experimental field plots testing combinations of species and/or genetic variation to naturally occurring variability across tides, rainfall and climate (e.g., Thomsen et al. 2005; Bernik 2015).

A-NZ trials

Field trials were used to assess the restoration potential of *J. maritimus*, *L. similis* and *Schoenoplectus pungens* within an established salt marsh near Christchurch (Thomsen et al. 2005). These trials assessed the effects of soil type, species, and plant source (commercial nursery stocks vs natural marsh stock). Plant biomass was unaffected by soil type, despite a minor increase in reclamation soil salinity during a November drought (Thomsen et al. 2005). There were some differences between species, with *S. pungens* failing to regenerate following seasonal die-back. Although *L. similis* and *J. maritimus* both survived well, *L. similis* produced more biomass than *J. maritimus*. Natural stocks were found to be hardier than nursery-sourced stock, so either splitting of natural stocks (destructive) or hardening of nursery stocks to the site was recommended (Thomsen et al. 2005).

Case studies:

Salt marsh restoration trials were initiated the Maketū Estuary (Bay of Plenty) in 1990 (commissioned by the Department of Conservation (DOC) and carried out by the NZ Forest Research Institute (NZFRI) and the Indigenous Forest Management section of NZFRI; (Bergin 1994) (see: Maketū case study box below, Figure 3-1). These showed that survival of *J. maritimus* and *L. similis* was dependent on clump-size at sheltered locations, with larger clumps achieving canopy closure sooner than smaller clumps. Complete vegetation cover was achieved in 3.5 years by planting at close spacing (4 plants/m^2) and using large sized transplants (100 x 100 x 150mm). Transplantation success of *J. maritimus* and *L. similis* in the Makatū Estuary varied with exposure to wind and waves. *Juncus maritimus* did not survive at the more exposed site (Bergin 1994).

Case study: 1

Success at Maketū: In 1956 the Kaituna River flow was diverted away from the river mouth to reduce flooding on the surrounding land. Concerned Tangata whenua and residents had been calling for a rediversion since 1979 to improve the health of the Maketū estuary. In 2009, a portion of the Maketū River was rediverted to improve the quality of the ecosystem, including widening of Ford's Cut and the removal of stop-banks beginning in 2018 (Figure 3-1). This allowed the estuary to expand, re-establishing Papahikahawai Island (now "Te Pā Ika"), that was recently replanted with wetland species. This has restored the farmland back into a nature reserve, and it is becoming a haven for native wildlife. This is a joint project between the landowners, Bay of Plenty Regional Council, Ngā Whenua Rāhui, tangata whenua, and the local community.

Since 2018, monitoring of macrofauna in the former reclaimed farmland of Te Pā Ika has shown recruitment of around three invertebrate species in each sample taken, with polychaete worms, amphipods and crabs being the predominant groups present at two monitoring sites (Park 2020). Although sediment mud levels were quite high (28-46%) the moderate levels of nutrients and organic content (TOC) present were lower than some nearby estuary sites that are still affected by nuisance macroalgae.

Case study: 1 cont…

Figure 3-1: Kaituna River redivision and Maketū Estuary enhancement plan (above)[2]. Community planting day[3] (below).

[2] https://www.boprc.govt.nz/our-projects/kaituna-river-rediversion-and-maketu-estuary-enhancement

[3] https://sustainablecoastlines.org/event/te-pa-ika-community-planting-day-1/

The recent successful small-scale restoration of salt marsh at the entrance to Waikawa Estuary in Marlborough (Stevens 2021) and proposed options for restoring 6,500 m^2 of intertidal in the Waimeha Estuary adjacent to Orchard Stream (Stevens, draft report) by Salt Ecology provide excellent advice and recommendations. Restoration of salt marsh at the entrance to Waikawa Estuary in Marlborough began in 2020 (Stevens and Robertson 2016). Salt marsh plantings included sea rush, *J. krausii,* jointed rush/oioi *L. similis,* glasswort *Sarcocornia quinqueflora* and knobby club rush *Ficinia nodosa.* verall, intertidal plantings survived well, despite some of the sea rush and oioi being periodically smothered by terrestrial debris/leaf litter, whereas on the larger glasswort divots survived. (see: Waikawa case study box below, Figure 3-2).

Case study: 2

Success at Waikawa: The successful small-scale Waikawa restoration project was some four-years in the making, following a 2016 broad scale habitat mapping report which recommended replanting estuary salt marsh and margin vegetation to improve ecological values that had been significantly degraded by historical habitat modification (Stevens and Robertson 2016). The aim of the restoration project was to redress some of the past habitat losses following the planned dredging of the Waikawa Stream to increase flood capacity, by utilising the fill to reshape the shoreline. The existing upper margin was widened, and a gently sloping shore profile created, to dissipate wave energy and allow replanting of intertidal and terrestrial salt marsh, with the aims of increasing ecological biodiversity and resilience and improving amenity values (Figure 3-2).

Early surveys included soil assessments to check for contaminants and suitable grain size (low mud content) for planting and help with the design phase. Consultation was then undertaken with local tangata whenua, Te Ātiawa-o-Te Waka-a-Māui Iwi (mana whenua and mana moana in Waikawa Bay), and after agreement, in partnership with iwi/hapū/tangatawhenua and Council, a draft plan was then shown to adjoining landowners and other stakeholders. The draft plan also helped inform cost estimates, the resource consent application, and identify areas for improvement.

During the construction phase of the Waikawa salt marsh restoration, stakeholder and team communication and supervision was important in minimising impacts of restoration work (e.g., assigning designated vehicle tracks) and fostering collaborative long-term goals for success. Initial weed spraying was recommended for both the Waikawa and the Waimeha Estuary projects, to suppress competitive species, especially during the establishment phase, and to reduce subsequent maintenance. Marking of terrestrial plants with bamboo stakes and the use of EmGuard plant guards also helps in reducing subsequent maintenance costs. Thomsen et al. (2005) recommended rabbit control to reduce herbivory of that pest. Adequate planning is very important as some plants may need to be grown from seed to be used in combination with nursery stocks and locally sourced donor plants (e.g., glasswort for Waikawa). Other recommendations for the Wiameha project included site preparation and reshaping (gravel, sand, soil additions, sills), planting (schedule, sourcing, methods, trained staff/volunteers, plant protection, fertiliser, mulch), and maintenance plans (Stevens, Salt Ecology, draft. report).

Case study: 2 cont…

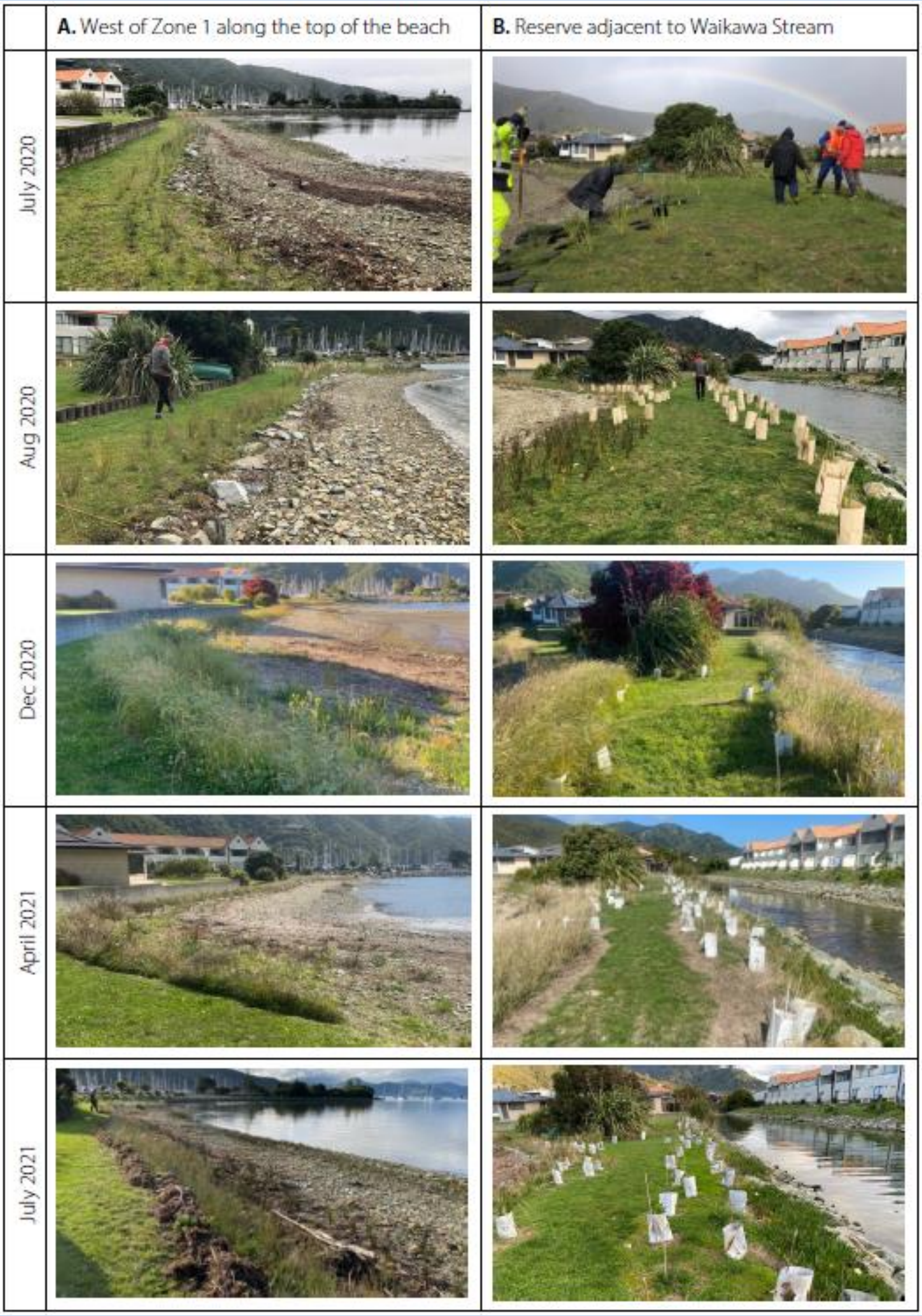

Figure 3-2: Changes in salt marsh over 12 months since planting in Waikawa Estuary. Source: Fig.3 from (Stevens 2021).

A similar example of success stemming from traditional knowledge and innovative science coming together comes from the freshwater marsh restoration Whakaora Te Ahuriri (Ahuriri Lagoon, Canterbury), renowned formerly as a significant mahinga kai for Ngāi Tahu, led by the Te Waihora Co-Governance Group[4]. The project provides another excellent example of a culturally led project, employing collaboration and consensus, to construct a wetland at landscape scale to give life back to a waterway.

3.1.3 New techniques, innovations

An innovative approach achieved a 60-fold increase in recruitment of the pioneering salt marsh genus *Salicornia* in the Eastern Scheldt estuary of the Netherlands (Fivash et al. 2021). Bed forms were made from porous artificial structures that produced a sheltered hydrodynamic environment in which suspended sediment and seeds preferentially settled (Fivash et al. 2021). *Salicornia* recruits grew to be on average three times greater in mass inside the structures than in the neighbouring sediment. The success of the structures was attributed to microscale wave attenuation that enhanced seed retention, suppressed mortality, and accelerated growth rates.

Similarly, efforts to restore seagrass, *Zostera marina,* at multiple locations (Finland, Sweden, UK, USA), found that mimicking the properties of key emergent traits (above-ground plants) greatly enhanced restoration success (van der Heide et al. 2021). Simulation of dense root mats or vegetation canopies with biodegradable structural mimics had varying success depending on the degree of exposure. In exposed environments, seagrass survival was enhanced. However, the positive effects of the mimics decreased and turned negative under benign conditions, and in extremely exposed environments the mimics insufficiently reduced physical stress.

An innovative company BESE, based in the Netherlands, have developed biodegradable products including mesh bags (for shellfish reef construction), reef paste (in-situ reef building), tiny reef modules, cable ties (e.g., for attaching plants, seagrass, sponge explants), and BESE-elements[5]. The latter "elements" have been used in wetland and shellfish restorations, among other use cases, to stabilise soft sediments during the colonisation phase[6].

[4] https://www.youtube.com/watch?app=desktop&v=a4TBaw0G3_o
[5] https://www.bese-products.com/biodegradable-products/
[6] https://www.bese-products.com/case-studies/

Figure 3-3: Artificial structure used in the field experiment at Texel in the Netherlands. The structure supported high densities of salt-marsh recruits, predominantly *Salicornia* sp., and recruits were considerably larger than those found in the nearby unmodified mudflat (source: Fivash et al. 2021)**.**

3.2 Urban/industrial infrastructure (seawalls, groins etc.).

Table 3-2: Summary of the salient points from the review of coastal urban/industrial infrastructure, with relevance to restoration activities in Te Tauihu. The remainder of Section 3.2 details the information behind this table.

1. Potential use or value, suitable locations to try in Te Tauihu?
▪ Habitat creation, enhancement, or restoration ▪ Where to try next: locations where intertidal habitats have been drained, modified/removed, or affected by coastal squeeze. During maintenance or replacement of existing infrastructure (seawalls, port structures, erosion barriers)
2. Status and why the habitat is important
▪ Status: following drainage, reclamations, and coastal developments, many seawalls, groins etc. have been installed using 'hard' materials with few living elements incorporated ▪ 'Soft/living' options can provide artificial/natural buffer between land and sea, protecting against large storms and tides. Reduces sediment erosion and runoff ▪ 'Soft/living' options can provide habitat for plants/algae, habitat/food for invertebrates, fish, and birds ▪ 'Soft/living' options reduce maintenance cost
3. Main, most recent, or successful techniques and methods used in restoration actions
▪ Softening edges, terraces, living setbacks ▪ Provision of sills or offshore artificial reef elements to reduce erosion during establishment ▪ Retrofit or designed elements that provide habitat for intertidal/subtidal species
4. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions
▪ Shellfish/shell or artificial reef restoration elements installed offshore to attenuate waves/ reduce erosion
5. Major barriers or deal breakers that prevent scaling-up restoration
▪ Availability of experienced practitioners/subject experts ▪ Cost/benefit analysis
6. Timescales for restoration
▪ Months to years
7. Other point/key message relevant to the habitat
▪ Softening edges can enhance biodiversity and resilience
8. Examples of where this has been tried before, likelihood of success
▪ Successful example of installation of living terraces (plantings between gabion baskets) in the lower Maitai River ✓ **Likelihood of success: High** ✓

3.2.1 Why should soft or living urban/industrial infrastructure be used?

Coastal built structures or infrastructure serve a range of purposes including coastal protection (e.g., seawalls, breakwaters, groynes), boating or recreational activities (e.g., marinas, wharves, pontoons) and the enhancement of fisheries yield (e.g., see artificial reefs, Section 3.5.1 below). When infrastructure is designed and built to include complexity, living elements, and 'soft' techniques, they can increase ecological value and resilience (e.g., Figure 3-4). An optimal approach is to use a few modifying species (such as mussel beds, oyster beds, and vegetation) that alter and soften the physical structure of the environment. For example, incorporating reef-forming shellfish that have the unique ability to trap and stabilize sediments in intertidal areas in the construction of dams or dikes can raise soil levels that can attenuate tides, at low-cost. Such self-sustaining approaches are low maintenance and require minimum reinforcement and less financial assistance.

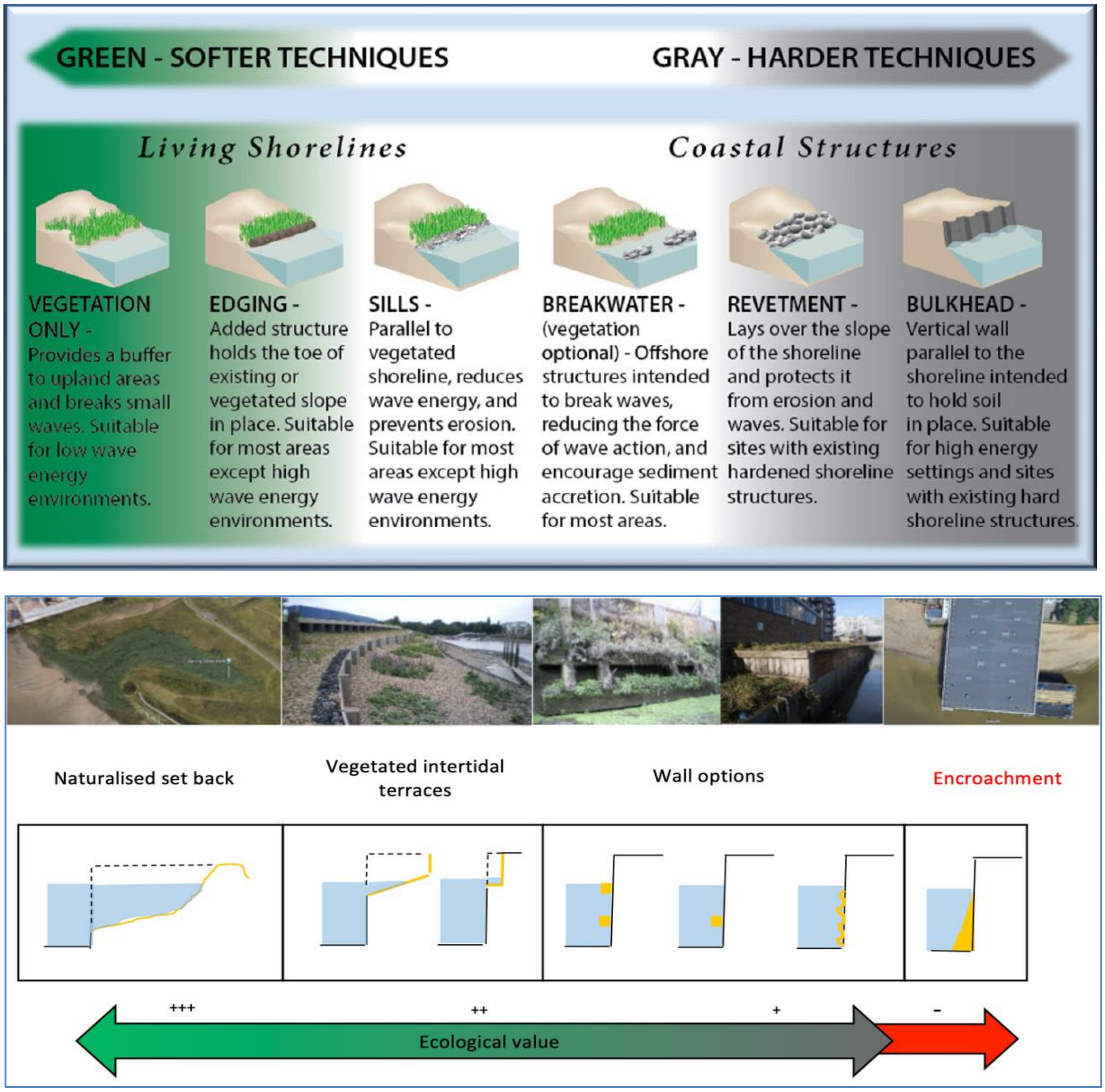

Figure 3-4: Diagram of continuum of living shorelines to hard or "gray" infrastructure (above). (source: http://sagecoast.org/docs/SAGE_LivingShorelineBrochure_Print.pdf), and estuary edge softening design principles (below) (source: https://www.estuaryedges.co.uk/design-principles/).

3.2.2 Successful techniques

When coastlines in urban and industrial areas require protection or conservation, especially in areas prone to erosion, the discipline of “ecological engineering” can be used. Ecological engineering seeks to improve new or existing infrastructure “hard” approaches by utilising environmentally benign materials and ecofriendly designs. These can incorporate “soft” replacements of buildings and infrastructure with natural habitats such as salt marshes, mangroves, or oyster reefs, and a combination of hybrid ecological engineering where natural habitats or vegetation are made to coexist with built infrastructure (Chapman and Underwood 2011). The best designs provide a balance between the natural ecosystem and ecosystem engineering services (Rajkhowa et al. 2021).

Ecological engineering approaches have had varying success, as evidenced by a meta-analysis and qualitative review of 109 studies to compare the efficacy of common eco-engineering approaches like increasing texture, crevices, pits, holes, elevations, and habitat-forming taxa (Strain et al. 2018). All interventions, except one, increased the abundance or number of species of $\geq$1 of four functional groups (sessile algae and invertebrates, mobile invertebrates, benthic sessile algae and mobile invertebrates combined, fish), but effects varied in magnitude among groups and habitat settings. Interventions that provided moisture and shade in the intertidal had the greatest effect on the richness of sessile and mobile organisms, while water-retaining features had the greatest effect on the richness of fish. In the intertidal, the taxa that responded most strongly to a particular type of intervention were those whose body size most closely matched the dimensions of the unit of intervention. However, shelter for new recruits from predators and other environmental stressors such as waves was provided by small-scale depressions. It was concluded that developing site-specific approaches that match the target taxa and dominant stressors were most effective. To maximise niche diversity, the use of a range of approaches applied simultaneously was recommended (Strain et al. 2018). Examples of living seawalls developed and tested in Australia can be seen in the case study box below (Figure 3-5).

As part of a collaborative[7] UK/France study that was predicated upon the central theory that structural complexity is one of the key processes driving biological diversity (Badgley et al. 2017), habitat structural complexity was compared between artificial coastal structures versus natural rocky shorelines, across a range of spatial scales from 1 mm to 10s of metres (Lawrence et al. 2021). The authors found natural shorelines were typically more structurally complex than artificial ones and offered greater variation between locations. Habitat complexity of the artificial structure varied considerably: seawalls were ca. 20–40% less complex than natural rocky shorelines, whereas rock seawalls provided low habitat complexity at the smallest and largest scales (ca. 20–50%). As a solution, retrofit or 'bolt-on' eco-engineering designs provide a means of modifying existing artificial structures to increase structural complexity. Retrofit designs include: small- to medium-scale (i.e. cm-scale) habitat features found in natural habitats (Strain et al. 2018) like drilled or cast pits, grooves and ridges (Hall et al. 2018; Barnett et al. 2020); drilled, cast or bolt-on rock pools and holes, 10–50 cm in depth or width (Evans et al. 2015; Morris et al. 2017; Hall et al. 2019). Combining large-scale units with medium-scale habitat features (e.g., rock pools) and fine-scale surface manipulation (e.g., texture or grooves) across multiple scales could restore structural complexities lost during the alteration of natural shorelines, without increasing structure footprints or compromising the function of the engineered structures (Lawrence et al. 2021).

Sources of NbS guidance on restoration methods (including living shorelines, infrastructure resilience, green-gray infrastructure, and coastal and riverine flood and erosion risk management) include the U.S. Army Corps of Engineers (USACE) Engineering With Nature® (EWN) Initiative[8], and the Thames, U.K. website Estuarine Edges[9] (see: case study box below, Figure 3-6) that outlines design principles and shows real world before and after examples. Another useful repository of technical guidance documents from the USA is from SAGE: a "Systems Approach to Geomorphic Engineering" that addresses ongoing and future coastal dynamic landscape change and threats and supports coastal transformation by integrating green and gray solutions to contribute to the resiliency of our communities, ecosystems and shorelines"[10].

[7] MARINEFF Project: http://marineff-project.eu/en/
[8] https://ewn.erdc.dren.mil/?page_id=3348
[9] https://www.estuaryedges.co.uk/
[10] http://sagecoast.org/info/sci-eng.html

Case studies:

Case study: 3

Success in Sydney: In Sydney, Australia, an experiment that tested different tile designs affixed to seawalls (complex and flat, 2.5 cm, 5 cm deep vertical or horizontal crevices) with some tiles seeded with the native oyster (*Saccostrea glomerata*) (Vozzo et al. 2021). Positive effects were found after 12 months (e.g., Figure 3-5). Complex oyster-seeded tiles supported a greater abundance of suspension feeding taxa, with richness and diversity increasing with complexity. Particle removal rates also increased following shellfish seeding. Their results suggest that the addition of complexity and filter feeders to marine artificial structures could potentially be used to enhance both biodiversity and particle or sediment removal rates.

Figure 3-5: Examples of living seawalls designed in Australia and tested across three continents. Source: https://www.livingseawalls.com.au/

Case study: 4

Successes in the Thames, UK: An exemplar of 'how to' soften estuarine edges - replacing brick, concrete, and metal tidal walls with a variety of habitats - can be seen at the 'Estuary Edges' project based on the Thames River, UK. Some seventeen projects are on display (www.estuaryedges.co.uk), with design principles covering a wide variety of topics including: setbacks/creek erosion management; intertidal vegetated terraces; vertical wall options; encroachment; masterplanning; archaeology and heritage; education, aesthetics and art; wildlife, planting and greenspace; fish; safety and navigation; monitoring and maintenance; litter; geomorphology; sustainability and adaptability; and use of timber[11]. Examples include terraced wall designs with inundation-appropriate plantings.

In a 1998 example, an existing sheet piled wall was replaced with terraces that were created between a newly constructed wall and the foreshore using gabions, with surfaces near horizontal (Figure 3-6). The growing medium was initially protected under coir matting where a variety of salt marsh plants were planted. Substrate particle size distribution of the growing medium was closely matched to the existing foreshore-type, for both stability and habitat value. As a result, a greater density and variety of species were associated with the significant development of vegetation cover on the terraces since 1998, especially plant and fish. However, invertebrate assemblage data suggested the site was impacted as the lower terraces were eroded downwards.

Figure 3-6: Vertical sheet wall replacement with gabion terraces at Greenwich Peninsula Terraces North West. (source: https://www.estuaryedges.co.uk/case-studies/greenwich-peninsula-terraces-north-west/)

[11] https://www.estuaryedges.co.uk/design-principles/

Case study: 4 cont…

In another example, a 'naturalised setback' design (Figure 3-7) at Barking Creek, Creekmouth performed at the highest tier level for restored invertebrate assemblages, and now supports a juvenile bass (*Micropterus salmoides*) nursery as well as flounder (*Platichthys flesus*), and European eel (*Anguilla anguilla*).

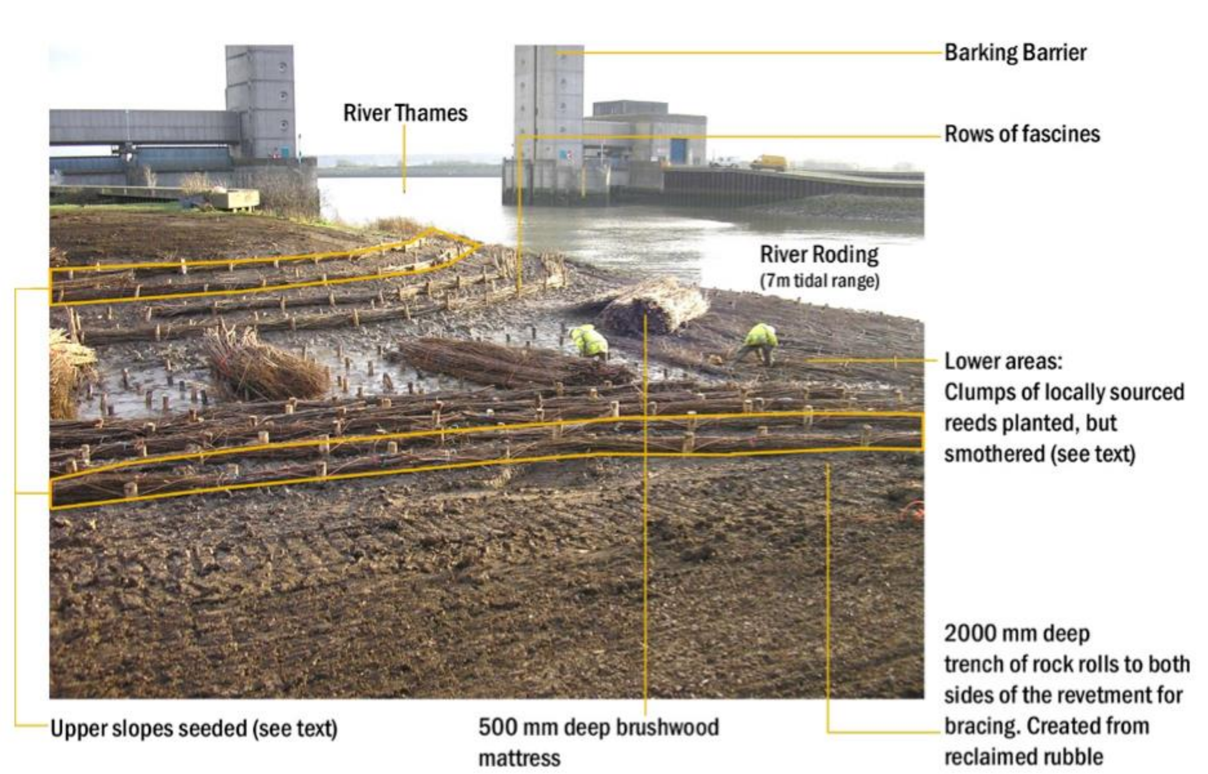

Figure 3-7: Barking Creek, Creekmouth vegetated intertidal terrace construction in 2006 (above) and Google 3D image of the results from 2018 (below). Source: https://www.estuaryedges.co.uk/case-studies/barking-creek-creekmouth/.

Case study: 5

Figure 3-8: Successful example of installation of living terraces (plantings between gabion baskets) in the lower Maitai River. Submerged gabion baskets are shown by dashed line in the bottom image

3.2.3 New techniques, innovations

Biodegradable artificial reefs

As an alternative to installing intertidal or shallow subtidal sills (e.g., Stevens 2021) to reduce sediment resuspension and/or shoreline erosion and providing protection for salt marsh plantings (or shellfish reefs, seagrass beds), the use of artificial biodegradable reefs has been proposed to attenuate local wave conditions to enhance restoration success (Marin-Diaz et al. 2021). These reefs can be made of living or dead material (e.g., oyster shells and live oysters, Figure 3-9).

Artificial reefs constructed from biodegradable potato-waste ('BESE' structures, 630 m long) installed on the exposed tidal flats of the Dutch Wadden Sea attenuated ca. 30% of the wave height in water depths below 0.5 m (Marin-Diaz et al. 2021). Local sediment accretions up to 11 cm deep were recorded, but the effect was limited to within 10 m of the landward edge of the structures. Sediment scouring of up to 10 cm deep was also found within some plots, potentially from increased turbulence. Marin-Diaz et al. (2021) concluded that while biodegradable artificial reefs have the potential to attenuate waves and trap sediment on tidal flats, exposed sites would require more resistant structures to provide longer-term benefits (e.g., Figure 3-9).

While retrofit or bolt-on living seawalls can be more costly to produce, can require greater maintenance efforts and monitoring, they often provide unaccounted benefits (Michael Allis, NIWA, pers. comm.). Benefits include the protective structure provided from the living seawalls (prolonged life) through to reducing wave effects. Additional costs can be offset by enhanced coastal amenities (more rockpools and natural features) and aesthetics (prettier than hard engineered shorelines). There is also the ecology/engineering paradox, where the ecological value (when fully colonised) will outweigh the protective value and, at the end of life of the structure, its replacement is impaired by the need to consider the effects on the enhanced (relative to a traditional structure) ecological values. The paradox is that the ecology wouldn't have been there to start with (at least at the same density/value).

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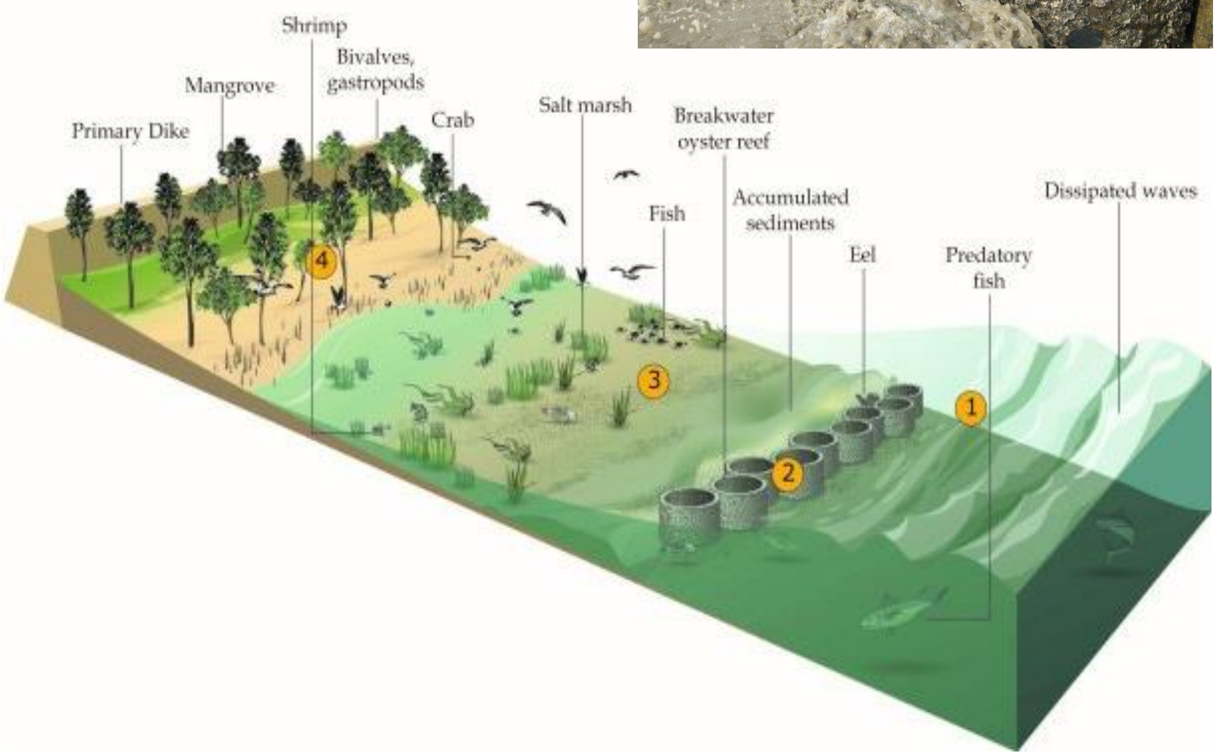


13

Figure 3-9: Examples of living oyster reefs installed to buffer coastal areas to reduce effects of erosion and sea-level rise (top)[21]. Schematic of types of living and hard coastal buffers (Source: Chowdhury 2019), Wageningen University, The Netherlands).

[12] https://www.wur.nl/en/Dossiers/file/Building-with-Nature-2.htm
[13] https://www.wur.nl/en/Research-Results/Research-Institutes/marine-research/show-marine/Oyster-reefs-used-to-counter-large-waves.htm

Floating marsh islands

With the desire in A-NZ to use wetlands as a means of mitigating pollution (Thomsen et al. 2005), a potential innovative solution for salt marsh restoration challenged by the shrinking of land-margins with sea-level rise, is to create floating marsh islands[14] (Figure 3-10). Such floating structures could also help in mitigating low-to moderate shoreline erosion in sheltered locations. Floating islands help clean up pollution and protect water resources as they enhance nitrogen removal capacity by more than 50%, most by denitrification – in which microbes break down nitrogen rather than its removal via plant uptake[15]. As an additional benefit, the islands provide habitat for wildlife, including ground nesting birds potentially reducing predation of their eggs.

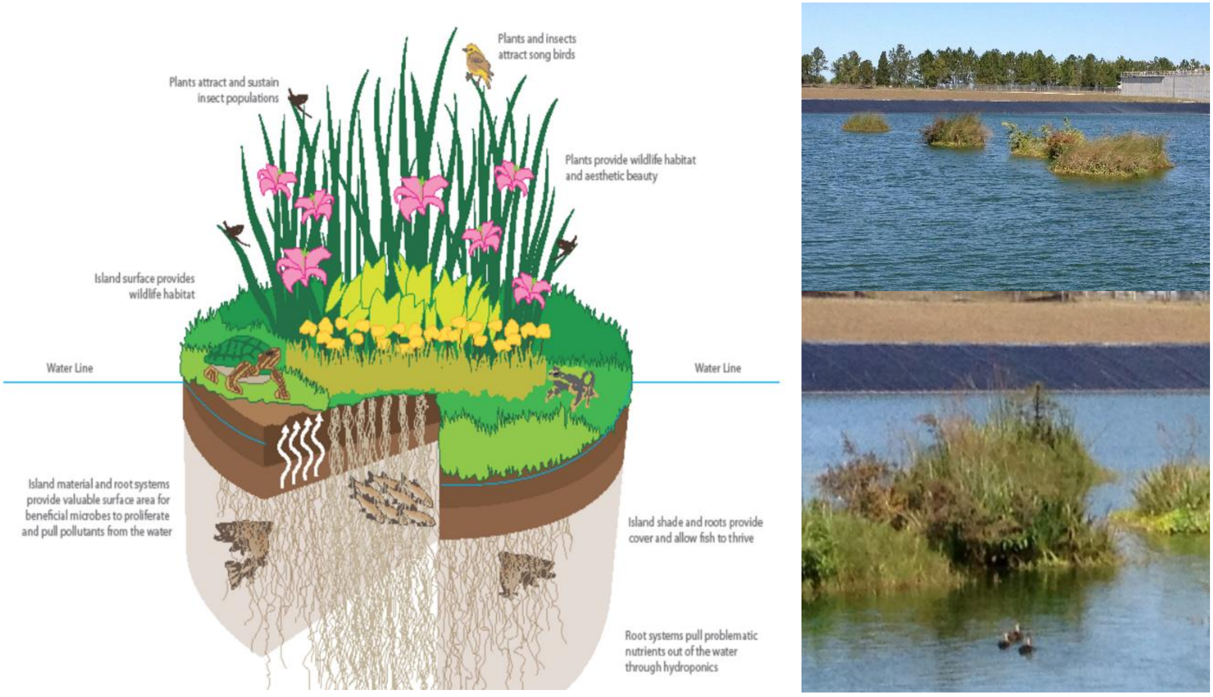


Figure 3-10: Floating islands as a potential mitigation solution to nutrient and chemical pollution[16]. The islands also provide refuge and habitat for wildlife.

[14] https://baysoundings.com/man-made-floating-islands-help-reduce-pollution/#
[15] https://stormwater.wef.org/2015/09/floating-treatment-wetlands-show-promise-pond-retrofit/
[16] https://baysoundings.com/man-made-floating-islands-help-reduce-pollution/#iLightbox[gallery8846]/0

3.3 Seagrass (eelgrass) *Zostera muelleri*, Horse's mane weed *Ruppia sp*.

Table 3-3: Summary of the salient points from the review of seagrass and horse's mane weed with relevance to restoration activities in Te Tauihu. The remainder of Section 3.3 details the information behind this table.

1. Potential use or value, suitable locations to try in Te Tauihu?

- Habitat creation, enhancement, or restoration
- Where to try next: inshore of the sills proposed for Waimeha Inlet salt marsh restoration (Stevens, draft report), Havelock Estuary, Mahau Sound, Head of Kenepuru Sound

2. Status and why the habitat is important

- Status: reported as "at risk-declining" DOC (De Lange et al. 2018)
- Nature based solution (NbS)
- Provides habitat and food for invertebrates, fish, and birds
- Prevents sediment erosion, resuspension
- Sequesters carbon and sediment, reduces eutrophication

3. Main, most recent, or successful techniques and methods used in restoration actions

- Transplantation of sods or small cores recommended to minimise impacts of extractions on existing beds, loss of transplanted material, and to simplify logistics

4. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions

- Reliable method for production of seed, so that reliance on vegetative transplants from existing beds is reduced
- Seagrass mycorrhizae and biome research, showing beneficial effects from biome transplants
- Shellfish facilitation experiments
- Shells + seagrass plant combinations

5. Major barriers or deal breakers that prevent scaling-up restoration

- Suitable sediment (grain size, pollution), turbidity/suspended solids concentrations (light) water quality (nutrients, pollution, phytoplankton) conditions
- Availability of adequate donor beds for sods/cores, or seed for restoration
- Adequate methods to overcome adverse thresholds/hysteresis: e.g., mitigating wave climate thresholds to stabilise sediments and prevent resuspension
- Muddiness of sediment affecting rhizome growth potential and shoot production
- Climate change: sea-level rise, coastal squeeze and increasing temperatures, marine heat waves, and water turbidity
- Coastal development, vehicles on tidal flats, swing moorings, propeller scarring
- Combination of factors above (multiple stressors, e.g., chemical pollution + silt, resuspended sediment + phytoplankton)

6.	**Timescales for restoration**

- 12-24 months: seagrass is very fast growing, and restoration success can occur within several years if environmental conditions are favourable

7.	**Other point/key message relevant to the habitat**

- Very important 'blue carbon' initiative to help reduce effects of climate change
- Subtidal seagrass beds are important juvenile fish nurseries

8.	**Examples of where this has been tried before, likelihood of success?**

- Successful core transplant method tested at Takahiwai and McDonald Bank, Whangarei Harbour ✓

Likelihood of success: Moderate to high for intertidal seagrass if stressors causing initial losses have been removed/reduced and donor plants available ✓

3.3.1 Why restore seagrass/horse's mane weed?

In New Zealand, seagrass *Zostera muelleri* is listed as "at risk – declining" (10-70%) and the horse's mane weed *Ruppia sp.* is listed as "at risk – naturally uncommon" by DOC (De Lange et al. 2018). Seagrasses are vulnerable to environmental change through global and local threats such as rising ocean temperatures and sea-levels, coastal development and pollution from sewage outfalls and agriculture, with 30% of seagrass meadows being lost worldwide over the last 50 years (Waycott et al. 2009; Tan et al. 2020).

Seagrass and horse's mane weed meadows provide valuable habitat in the form of shelter and food for marine invertebrates and fishes, and foraging grounds for certain shorebirds (Matheson et al. 2009; Drake et al. 2011). Dense meadows of these macrophytes can stabilise the seabed and reduce erosion and enhance water quality. Seagrass leaves trap fine sediments and reduce particle loads in the water by slowing water movement and encouraging particle deposition (Heiss et al. 2000; Bryan et al. 2007) which improves the water clarity. Seagrass plants absorb nutrients from the water and seabed and release oxygen from their leaves and roots, which is beneficial for other biota and stimulates nutrient cycling. They have also been found to reduce the presence of harmful strains of bacterial *Vibrio* loading by up to 63%, but studies have yet to discover how (Reusch et al. 2021).

Seagrass is ranked among the most significant organic carbon sinks on earth (Serrano et al. 2019a). In temperate coastal meadows, soil organic carbon stocks were enhanced by low hydrodynamic exposure, high mud and carbon content of seagrass, low to moderate solar radiation, and low human pressure (Mazarrasa et al. 2021). Decaying seagrass is decomposed by bacteria and fed on by small marine animals (particularly snails but also worms, bivalves, and crustaceans), both within and adjacent to seagrass meadows (Hailes 2006; Taylor and Brown 2006), supporting the marine food web (Woods and Schiel 1997; Hailes 2006; LeDuc et al. 2006). The small crustaceans and worms that live in seagrass meadows (van Houte-Howes et al. 2004) are important sources of food for wading birds (such as the South Island oyster catcher, pied stilt, royal spoonbill, bar-tailed godwit) and fish such as mullet, stargazers and juvenile flatfish (Inglis, GJ 2003). Snapper recruits also appear to utilise seagrass as refugia from high current flows that they periodically leave to feed on pelagic zooplankton higher in the water column (Parsons et al. 2018).

In northern A-NZ, subtidal seagrass meadows provide unmatched juvenile fish and invertebrate habitat. Where water clarity permits, seagrass can grow down to 2-3 m below low tide where it provides important nursery habitat for many fish species. These include snapper and leatherjacket juveniles, as well as mullet, trevally, garfish, parore, spotties, pipefish and triplefins that are often abundant in subtidal seagrass meadows in particular, but also reside in intertidal meadows when the tide is in (Morrison et al. 2007; Parsons et al. 2013; Parsons et al. 2014; Parsons et al. 2016; Morrison 2021). As a component of habitat, subtidal seagrass blade density is an important driver of small fish abundances, with the numbers of trevally, triplefins and juvenile snapper increasing with blade density (Morrison 2021). Abundance of invertebrates such as shrimps and crabs also increase with seagrass density (Morrison et al. unpubl. Data; Morrison 2021). In East Northland subtidal seagrass beds, juvenile snapper were recorded at densities of 159 per 100 m^2 and estimates of numbers associated with seagrass in the Pārengarenga and Rangaunu Harbours were 1.081 million and 1.886 million fish, respectively (Morrison et al. 2014a; Morrison et al. 2014b; Morrison et al. 2014c; Morrison et al. 2019). Juvenile snapper also appear to grow faster in seagrass meadows, weighing 1.45 and 1.87 times heavier after 40 and 70 days respectively (Stewart 2018; Morrison 2021). In Port Phillip Bay, Australia a hectare of seagrass was estimated to produce 110 – 1,080 kg^{-1} of fish per year (Jänes et al. 2021).

The comeback of extensive beds of high-value native macrophytes in the south-east of Big Lagoon, Wairau, including *Ruppia sp.,* was recently reported, covering an area of 199 ha or 16.2% of the subtidal area (Roberts et al. 2021). Although the brackish Big Lagoon was reported to have lost much of this habitat prior to 2015 (Knox 1983), the renewed presence of this rare habitat appeared promising. However, the extensive mats of filamentous green algae (*Cyanophyceae* and *Cladophora*) growing epiphytically on vegetation as well as on sediments indicates the estuary is still under stress (Roberts et al. 2021).

Pressures

Stressors affecting seagrass beds are many and can include: high sedimentation rates, turbidity and chemical pollution (Zabarte-Maeztu 2021); physical damage from dredging, coastal developments, mooring scour, vehicle and foot traffic (Clark and Berthelsen 2021); severe storms, overgrazing and/or competition of fouling species and a fungal wasting disease (Matheson et al. 2009). Eutrophication can also cause the proliferation of phytoplankton and harmful algae, reducing light levels available to seagrass for photosynthesis, and/or fuelling growth of competitive or fouling species including macroalgae (Kemp et al. 2005).

3.3.2 Successful techniques

Seagrass restoration is a rapidly maturing discipline with increasing rates of success, but improved restoration practices are needed to enhance the success of future programmes (Tan et al. 2020). The first step in successful restoration is to establish the physical, biological, and chemical stressors responsible for the loss and/or lack of recovery of historic seagrass meadows (Clark and Berthelsen 2021). Decisions then need to be made on whether such factors have been remedied or reduced to ensure the conditions for restoration are adequate for passive (natural recovery) or active restoration (seeding, transplanting). For example, hydrodynamic conditions can play a role in the success of re-establishing meadows once they have disappeared, so mitigation measures to consider include whether waves or currents need to be buffered to enable survival of plants. Attempts to attenuate waves have included the use of artificial structures mimicking seagrass to provide shelter in the meadow establishment phase, although they can also shade out transplants hindering their growth (F. Matheson, NIWA pers. comm., (van der Heide et al. 2021).

Following a disturbance, seagrass typically recovers via asexual reproduction from horizontal rhizome growth (Duarte and Sand-Jensen 1990; Xu et al. 2018). Sexual reproduction in seagrass plays a vital role in colonizing new habitats and in areas of large-scale decline or destruction (Lee et al. 2007; Xu et al. 2021).

Seagrass transplants

Restoration of seagrass generally involves planting seed and/or seedlings, or transplanting plant rhizomes, shoots or cores sourced from intact seagrass meadows. A successful core transplant method tested at Takahiwai and McDonald Bank, Whangarei Harbour using dormant winter transplants of small cores, and larger plots, both restored populations of *Z. muelleri* with full recovery of donor meadows (see: case study box below, Figure 3-11, Figure 3-12).

Other less successful methods (cf. sods or cores) trialled include transplanting: seeds of *Z. marina* (Marion and Orth 2010; Orth and McGlathery 2012; Infantes et al. 2016; Xu et al. 2021); *Posidonia oceanica* seedlings (Infantes et al. 2011); *Z. marina* rhizome fragments (Davis and Short 1997); and adult shoots (*P. australis* - Meehan and West 2002; *Z. marina* - Eriander et al. 2016).

To aid establishment of transplants, various innovative anchoring systems have been tested including rods and pegs (metal, bamboo, wood), biodegradable mesh (Kidder et al. 2015), hessian bags (Tanner 2015), and attaching *Z. marina* shoots to oyster shells (Lee and Park 2008). The latter achieved success in subtidal beds without the need for SCUBA (Lee and Park 2008) (Figure 3-13). However, planting of adult plants with intact rhizomes and sods seems to have the highest success rate (van der Heide et al. 2007). Mechanical planting has also been tried (Fishman et al. 2004) as well as sediment fertilization (Balestri and Lardicci 2006). Fertilization can be effective in areas that are nutrient-depleted, but fertiliser inhibits plant growth at high nutrient levels (Peralta et al. 2003). The use of un-treated iron nails is thought to enhance survival because addition of iron into a well-oxidized seagrass rhizosphere increases the absorption capacity for phosphorus and reduces sulphide toxicity (Holmer et al. 2005; Ruiz-Halpern et al. 2008).

Case study: 6

Successes in Whangarei Harbour: Illustrating the flow-on benefits of reducing long-term stressors in the Whangarei Harbour, the successful re-establishment of *Zostera marina* seagrass meadows has occurred in the Harbour since 2008 (Matheson et al. 2017; 2022). The earlier study compared the use of mid intertidal zone seagrass transplants of: i) intact entire 'sods' (including sediment, Figure 3-11), ii) unanchored 'sprigs' and iii) sprigs amongst 'mats' of artificial plants. Sods and sprigs proved equally effective with plant cover increasing from <1 to 63%, but artificial mats were ultimately not successful. Cover across the wider transplant site increased from 10% to 46% (biomass from 58 to 321 g m^{-2}). The success of these experiments was attributed largely to increased water clarity resulting from management of urban and industrial discharges and reduced dredging of shipping channels to access the Portland Cement Works in the upper Harbour.

Further success was achieved with the use of small cores transplanted at McDonald Bank in Whangarei Harbour between 2012-2016 that have re-established and are still expanding (Matheson et al. 2022) (see: Table 3-3 above). Both small cores and larger plots transplanted in winter began to spread after 12–18 months, eventually developing patches ranging in size from 5 to 68 m^2; the donor plots (from which the transplanted cores were originally sourced) recovered completely within 10 months. The use of small cores is recommended to minimise impacts of their extractions and the loss of transplanted material, and to simplify logistics.

Figure 3-11: Sods of seagrass being transplanted in Whangarei Harbour (left) and four years later (Photos: Jacquie Reed, Crispin Middleton[17].

[17] Zostera Restoration in NZ — Seagrass Restoration network

Case study: 6 cont…

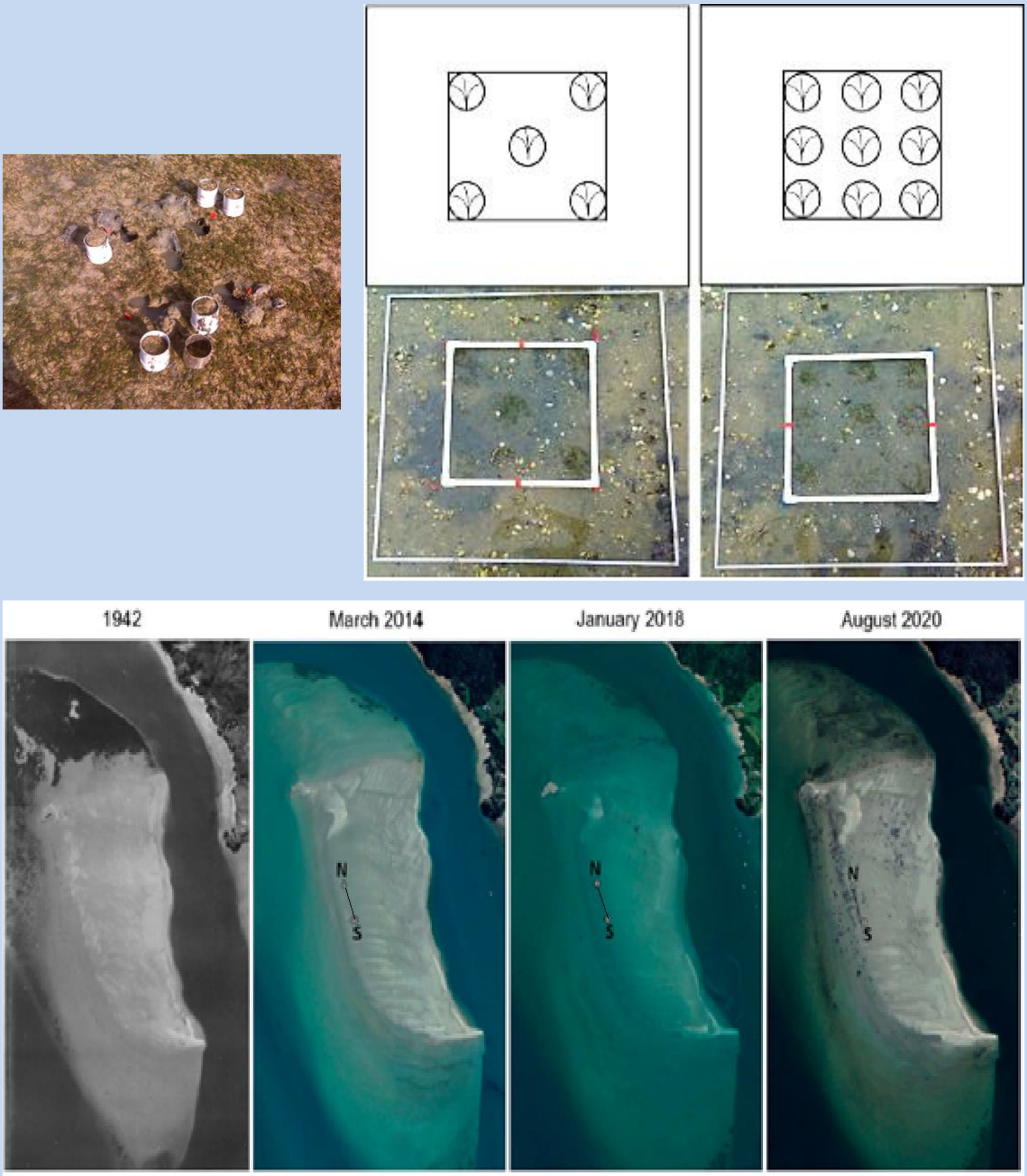


Figure 3-12: 5× 0.1 m cores (top left and centre) and 9 × 0.1 m diameter cores (right) Source: (Matheson, Fleur E et al. 2022). Aerial photograph time-series of McDonald bank (bottom). In the 1942 photo the dark patches on the northern tip and western side are seagrass beds. In the 2014–2020 photos, the location of the line of transplants is indicated with the northern (N) and southern (S) ends marked. The transplants are visible in the January 2018 and August 2020 photos. The 2014–2020 aerial photos were sourced from Google Earth (Image © 2022 Maxar Technologies, CNES/Airbus). Source: Matheson et al. (2022).

Figure 3-13: The innovative combination of seagrass plants and waste shell, used to restore subtidal beds without the need for expensive divers. Source: Lee and Park (2008).

Seagrass plants appear to grow best with conspecifics (peaking at moderate to high patch density) or in association with Neptune's necklace *Hormosira banksia*, perhaps because they reduce desiccation during low tide periods (Ramage and Schiel 1998; Dos Santos and Matheson 2017).

In South Australia, beds of *Ruppia tuberosa* are being restored by scraping up seeds, bagging the sediment, then redistributing the sediment at restoration sites[18].

Biosecurity risks associated with potential to transport toxic algal cysts and other pest species when moving sediments and plants should always be considered. Cultural aspects of transplants should also be approved by relevant iwi/hapū/tangatawhenua.

3.3.3 *Zostera* seed production

Propagation from seed has not yet been trialled for *Z. muelleri* in A-NZ due to lack of seed source (Clark and Berthelsen 2021; Zabarte-Maeztu et al. 2021b). Reports of seagrass producing seed in A-NZ are rare, but due to the inconspicuous nature of the flowering shoots that emerge near the sediment surface (inflorescence, see: Final A4 Seagrass Guide.indd (niwa.co.nz) flowering may be more common than previously thought (Dos Santos and Matheson 2017). It is hypothesised that *Zostera* is adapted to high stress but low disturbance whereby it has little reliance on sexual reproduction, but rather relies on vegetative growth and storage of biomass as rhizomes to overwinter underground (Zabarte-Maeztu et al. 2021b).

Flowering is more abundant in patches with high cover and high biomass (Dos Santos and Matheson 2017; Zabarte-Maeztu et al. 2021b), supporting the hypothesis that flowering is energetically expensive. In Kaikoura between 1994-95, flowering and seed production was reported from October to June with peak flowering between January to March (Ramage and Schiel 1998). Flowering and seed production have also been noted at multiple North Island sites (Bay of Islands, Whangarei, Kaipara, Porirua, Raglan (Matheson et al. 2010; Dos Santos and Matheson 2017). Large branching florets were produced lower on the shore in Kaikoura, especially in association with tidal pools and creeks where plant biomass was greater above and below ground. In Tauranga Harbour flowering was observed at one site high on the shore associated with a freshwater seep (Dos Santos and Matheson 2017). More recently, flowering has been observed in Golden Bay and in the Marlborough Sounds (Clark and Berthelsen 2021). The incidence of flowering in *Z. muelleri* has recently been

[18] https://seagrassrestorationnetwork.com/ruppia-restoration-in-sa

updated to seven locations in A-NZ, supporting the hypothesis that the cryptic nature of the flowers have prevented more widespread reporting (Dos Santos and Matheson 2017; Zabarte-Maeztu et al. 2021b). Encouragingly, three of the locations were from Te Tauihu: Golden Bay, Abel Tasman and Ngakuta Bay in Queen Charlotte Sound (Clark and Berthelsen 2021; Zabarte-Maeztu et al. 2021b). Seed has not yet been found. Flowering plants typically have longer and wider leaves (Ramage and Schiel 1998; Dos Santos and Matheson 2017).

Monoecious[19] pollination may occur during tidal exposure and, when submerged, the long filiform pollen strands produced by male plants come in contact with the female pistils where fertilisation takes place. It is likely that an unknown combination of multiple factors - including suitable sediment and nutrient regimes, moderate to high intra-specific competition, lower salinity (stimulating rhizome growth in winter), fluctuating but not cold temperatures (>15°C), irradiance (~300 µE m^{-2} s^{-1}) and photoperiod (13-14h light) - interact to induce flowering and allow *Zostera* to set seed in A-NZ.

Potential methods for collecting, processing, storing and planting of seagrass seed in Te Tauihu was recently reviewed by (Clark and Berthelsen 2021). This showed that information on the requirements for *Z. muelleri* seed germination and early phase of seedling survival are not currently available.

A recent study of the related species *Z. marina* from China compared multiple factors for planting, including light, seed planting depth and sand content (Xu et al. 2021). They found that seed planting depth was optimum at 2cm depth for any combination of sediment type. However, to increase seedling survival by planting seeds deeper at 5 and 10cm depth, germination was favoured by use of 100% sand at 5cm sediment depth; germination did not occur when planted at 10cm. Xu et al. (2021) hypothesised the differences were due to increased oxygen availability that declines with sediment depth and reduced mean grain size, despite it being reported that anoxic conditions can increase seed germination (Moore et al. 1993). Unsurprisingly, light levels, manipulated by growing seedlings at different water depths affected seedling survival below a threshold of 4-36% surface irradiance (Ralph et al. 2007). At the end of the experiment, seedling establishment was maximal in pots containing 100% sand. If sufficient seed of *Z. muelleri* can be obtained locally in Te Tauihu, inclusion of sand could be similarly trialled in conjunction with dispenser injection seeding systems tested in the Dutch Wadden Sea (Tan et al. 2020).

3.3.4 Evaluating conditions for successful seagrass restoration

Fine sediment, chemical properties

There appears a multifaceted challenge to successfully restoring seagrass due to multiple stressors acting on above and below-ground plant structures (Zabarte-Maeztu 2021). A study comparing sediment and light conditions across a gradient of 'present', 'potential' and 'historic' seagrass distribution in Pāuatahanui Inlet attributed seagrass loss to pollution with fine sediments exerting multiple stresses, particularly deoxygenation of the root zone (Zabarte-Maeztu et al. 2020). Substrate mud and organic content, and ammonium concentrations were higher at historic sites. They inferred that the higher mud and organic content reduced oxygen diffusivity, affecting iron–sulphide interactions that can affect the release of toxic heavy metals such as zinc, lead, ferrous iron and copper that are chemically bound to fine sediment particles. Factors potentially mitigating the apparently healthy plants presently growing in sediments with high mud content were the co-occurrence of bivalves such as tuangi/cockle (*Austrovenus stutchburyi*) and the wedge shell (*Macomona liliana*) that aerate the sediment through their bioturbation activities, reducing the

[19] A plant having both the male and female reproductive organs in the same individual

redox potential (oxygen diffusivity) of surface sediments. At 'historic' Pāuatahanui Inlet sites, light levels alone did not appear to explain the failure of seagrass to re-establish. Interestingly, the substrate mud content range (13–23%) in Pāuatahanui Inlet was higher than the 13% silt (3.9 < 63 µm particle size) threshold previously determined for this species in Tauranga Harbour (Park et al. 1994) suggesting that *Z. muelleri* has a wider tolerance to mud than previously thought or that other unmeasured factors were at play in the Tauranga study.

Sediment and light requirements

Another experiment that compared the combined effects of very low light levels and mud content of sediments (20% vs 42% mud content) found that low oxygen levels and greater nutrient content of muddier sediment affected the below-ground biomass of the rhizome of the *Z. meulleri* plants, regardless of light levels (Zabarte-Maeztu et al. 2021a). Those effects translated to reduced number of shoots produced above-ground, but the biomass of the rhizomes was reduced in high mud treatments. Again, those authors concluded that including light levels alone may not be sufficient to assess habitat suitability and substrate interactions also need to be included, as low light levels were an additional stressor along with adverse mud content at the scale of the rhizosphere. The results suggested that seagrass inhabiting muddy substrates has an increased light requirement to deal with adverse rhizosphere conditions, including sediment oxygenation (Zabarte-Maeztu 2021). A further experiment comparing the seagrass production of intertidal seagrass under submerged and emerged conditions, found that when emerged net photosynthesis was 25 times greater than when plants were submerged. Those results support previous studies reporting emerged photosynthesis during low tide as a mechanism that helps *Z. muelleri* to restore a degraded submerged light climate (Zabarte-Maeztu 2021).

Overcoming an alternative stable state to improve success

Challenging the desire to use passive restoration methods as a management option (e.g., see: Clark and Berthelsen 2021), the ability for seagrass to recover from adverse conditions has been shown in model simulations to suffer from hysteresis. The resilience of the alternative turbid state of the growing environment must first be overcome before positive feedback from the recovering seagrass stands can bind and prevent the resuspension of sediment, thereby reducing turbidity (van der Heide et al. 2007). Potentially demonstrating recovery from an alternative stable state, the success of restoration trials in Whangarei were thought to have been helped by decadal scale recovery of growing conditions followed (mainly) by cessation of (i) past significant sediment discharges to the harbour (by Portland Cement Works) and (ii) dumping of dredge spoil on tidal flats (F. Matheson pers. comm.). Subsequent research by Zabarte-Maeztu et al. (2021) and the successful transplanting trial at Takahiwai to McDonald Bank in 2012-2016 shows seagrass restoration is feasible in places where suitable growing conditions have been re-established, and at locations where there is no longer a bottleneck in propagule supply from remnant meadows nearby that delays or prevents a natural recolonisation process. Further supporting the sustainability of active restoration methods via sod transplants, the mitigation of negative environmental conditions in the Whangarei Harbour also allowed the donor beds to recover within 9 months (Matheson et al. 2017) rendering transplantation via sods a sustainable option for that harbour.

Decision support framework & iwi partnerships

The ultimate aim to enable successful seagrass restoration across A-NZ would be to develop a decision support framework that enables step by step instructions and troubleshooting for practitioners. Clark and Berthelsen (2021) reviewed information on the environmental requirements, natural recovery timeframes and genetics of *Z. muelleri* and the role that iwi/hapū/tangatawhenua and mātauranga Māori could play in restoration initiatives, building on earlier efforts to develop a decision framework by (Schwarz et al. 2005; see section 4 of Clark and Berthelsen 2021). Iwi should be valuable partners in seagrass and other forms of restoration because Māori hold a connected world view, Te Ao Māori that centres the connected nature of te taiao (the natural world/ecology) that encompasses active kaitiakitanga or guardianship. Potential estuaries recommended to trial passive or low impact seagrass restoration were the Te Tai o Aorere/Nelson Haven, Kokorua Inlet and Wakapuaka/Delaware Inlet in the Nelson/Whakatū region.

3.3.5 New techniques, innovations

Combining species

By combining the restoration of two of more species, success may be increased, especially in areas where environmental conditions are marginal. Failures in seagrass restoration have been attributed to factors including high turbidity (Thorhaug 1985; Eriander et al. 2016), fouling and smothering by filamentous algal loads, drifting algae (Gustafsson and Boström 2014), and wave exposure (Infantes et al. 2011). 'Facilitation cascades' have been described whereby one species enhances the colonisation success of another species (Wall et al. 2008; Maxwell et al. 2017; Gribben et al. 2019).

Filter feeding by shellfish has been shown to enhance seagrass and macroalgal restoration success as they increase water clarity by removing competitive phytoplankton and suspended sediments that shade the water column and enhance plant growth by releasing beneficial bio-deposits and soluble nutrients (Dame and Libes 1993; Reusch et al. 1994; Peterson and Heck 2001a; Peterson and Heck 2001b; Carroll et al. 2008). One of the largest scale examples of this is blue mussels facilitating seagrass restoration in the Wadden Sea by improving water quality and increasing the likelihood of seed settlement and shoot survival, enhancing bed recovery (e.g., van Katwijk et al. 2009).

To illustrate the complexities posed in marine restoration, Kemp et al. (2005) reported on restoration of seagrass beds in Chesapeake Bay in the United States has been hampered by a range of non-linear ecological feedback mechanisms involving light, sedimentation, nutrient dynamics, and species interactions. In an unmodified seabed, enhanced particle trapping, and sediment binding associated with plants (seagrass, sediment binding microalgae) help to maintain relatively clear water columns, allowing more light to support increasing benthic photosynthesis. Restoration or degradation can be driven either way by positive-feedback mechanisms. Nutrient enrichment creates turbidity, leading to the decline of benthic plants, allowing more resuspension, decreasing light further stressing benthic plants. Also, nutrient-enhancement stimulates phytoplankton growth that shades the seabed and upon death and subsequent sinking supports increased benthic respiration leading to anoxia. Sediment chemical pathways in the absence of oxygen cause more efficient benthic recycling of nitrogen and phosphorus which supports further algal blooms and so on. Initial seagrass restoration efforts failed without parallel oyster population restoration. Oysters are reported to provide negative feedback control on eutrophication by reducing phytoplankton biomass and increasing water quality thereby facilitating seagrass bed restoration (Fulford et al. 2007).

Microbiome

An exciting new avenue of study is the microbiome associated with seagrass species. Recent studies have identified a *Phyllobacterium* sp. which may be involved in nitrogen cycling in the seagrass ecosystem (Ettinger and Eisen 2020). If microbes are important, then they could either be cultured for inoculation or transplanted together to promote positive feedbacks to enhance seagrass restoration success (Suykerbuyk et al. 2016).

Genetics

Other recommendations to increase seagrass restoration success include ensuring high genetic diversity in the transplanted population (Jahnke et al. 2015; Evans et al. 2018; Allcock et al. 2022) and planting on a large scale (van Katwijk et al. 2016).

3.3.6 Protection of seagrass restoration plots

The persistence and restoration potential of seagrass can be impacted by waterfowl grazing including black swans (*Cygnus atratus*) and Canada geese (*Branta canadensis*). Swans can consume an average of 394 g dry mass (DM) swan^{-1} d^{-1}. In Tauranga Harbour grazing (at its most intense) removed 19–20% of the average seagrass biomass, causing substantial decline (43–69%) in plant biomass in the subsequent growing season (Dos Santos et al. 2012). Grazing by Canada geese may also contribute to seagrass decline, especially in summer months when pastures are less nutritious (Ferries 2021). A meta-analysis revealed strong top-down effects of grazing on *Zostera*, and ecological linkages between seagrass and waterfowl that may influence the spatial structure, composition, and functioning of the seagrass ecosystem (Kollars et al. 2017).

3.4 Shellfish beds: green lipped mussels, flat oyster, rock oyster, scallops and horse mussels

Table 3-4: Summary of the salient points from the review of shellfish beds with relevance to restoration activities in Te Tauihu. The remainder of Section 3.4 details the information behind this table.

1. Potential use or value, suitable locations to try in Te Tauihu?
▪ Habitat creation, enhancement, or restoration ▪ Where to try next: – Flat oysters: Tasman & Golden Bays, Pelorus Sound soft sediment habitats (outside trawl/dredge fisheries) – Green-lipped mussels: Tasman & Golden Bays, East Bay, Queen Charlotte Sound across historic distributions (outside trawl/dredge fisheries) – Scallops: on sandy or fine gravel habitats where sedimentation and disturbance is minimal, atop mud habitat if water quality amenable (minimal resuspension) – Horse mussels: across historic distributions where sedimentation and disturbance is minimal
2. Status and why the habitat is important
▪ Status: populations declining in Te Tauihu: scallop & oyster fisheries closed ▪ Nature based solution (NbS) ▪ Culturally, recreationally, commercially and ecologically important ▪ Provide habitat and food for invertebrates and fish ▪ Provide biogenic habitat structure and settlement surfaces in soft sediments ▪ Sequester sediment and carbon, reduce eutrophication
3. Main, most recent, or successful techniques and methods used in restoration actions
Flat oysters: ▪ Wild spat catching (shell, packing tape, wooden sticks), brooding females held in hatchery to produce larvae to settle on various substrates ▪ Shell or rock enhancement above/on muddy sediment Green-lipped mussels: ▪ Wild spat catching (filamentous substrata; native plant fibres e.g., harakeke), hatchery production of spat ▪ Relaying farmed mussels (sub-adults, adults) intertidally and subtidally, minimise de-clumping, inspect/remove fouling/non indigenous species Scallops: ▪ Wild spat catching in suspended mesh bags ▪ Recruits (~5-50 mm) released to seabed ▪ Secondary settlers (fallen from spat catching gear) dredged up and relayed

Horse mussels:

- Wild spat catching in suspended mesh bags (overseas)
- Hatchery production of spat
- Manual transplanting (overseas)

Cockles, pipi, tuatua:

- Wild collection of individuals for transplanting

Hatchery production of spat

4. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions

- Seagrass facilitation experiments (cockles, mussels, horse-mussels [subtidal])
- Seagrass & macroalgal restoration (scallops, mussels, oysters) across former range (e.g., *Z. muelleri, Cystophora, Carpophullum, Ecklonia*)
- Aggregating conspecifics to increase wild spawning success

5. Major barriers or deal breakers that prevent scaling-up restoration

- Suitable sediment and water quality conditions
- Spatial management or elimination of bottom impact fishing methods
- Waste shell deployment to stabilise soft sediments and reduce resuspension
- Multiple stressors (suspended sediment, elevated temperature, reduced pH, low oxygen), extreme weather events (storms, reduced salinity, warming, marine heat waves)

6. Timescales for restoration success

- 24-36 months (species specific)

7. Other point/key message relevant to the habitat – factors affecting success

- Sedimentation
- Poor water quality, high turbidity, high suspended sediment concentrations, low phytoplankton abundance
- Coastal acidification
- Removal, burial, by bottom contact fishing methods (dredging, trawling)
- Resuspension of legacy sediment (fishing, dredging, extreme weather events)
- Predators (e.g., 11-arm starfish), disease, parasites, pests

8. Examples of where this has been tried before, likelihood of success?

- Successful small plot restoration of green-lipped mussels (4 out of 5 sites, 85% survival) in Pelorus/Te Hoiere, and Hauraki Gulf
- Limited flat oyster enhancement success using waste scallop shell in Tasman Bay (green-lipped mussel also more abundant at one site)
- Rock oysters (*Saccostrea glomerata*) and flat oysters (*Ostrea angasi*) successfully restored in Australia
- Cockle transplants in Whangarei Harbour

- Likelihood of success:
  - Green-lipped mussels – high ✓
  - Flat oysters – **low** (*Bonamia*) ✗
  - scallops, horse mussels – **low** (predation, turbidity, seabed disturbance) ✗
  - cockles – high ✓

A community shellfish restoration guide tailored for cockle/tuangi *Austrovenus stutchburyi* was developed by NIWA in 2009 that outlines the steps and instructions to evaluating drivers for restoration, establishing a vision, values, goals, and understanding the issues faced (Cummings and May 2009)[20]. Cockle/tuangi were also successfully transplanted at two sites in Whangarei Harbour, with survival of different size classes context dependent (related to height on shoreline, hydrographic climate and connectivity) (Cummings et al. 2007; Hewitt and Cummings 2013).

This section of the review focusses mostly on bed or reef-forming epibenthic shellfish species that can form biogenic habitats. It does not include infaunal species that can play a facilitatory role with algal species including seagrass (e.g., see Section: 3-2).

3.4.1 Why restore shellfish?

A more in-depth summary of the benefits of benthic shellfish beds including green-lipped mussels *Perna canaliculus* (GLM), flat oyster *Ostrea chilensis*, scallop *Pectin novaezelandiae*, and horse mussels *Atrina zelandica* can be found in (McLeod 2009; Handley and Brown 2012; Handley 2015; Michael et al. 2015). Summarised briefly, studies of remnant soft-sediment mussel populations, bivalve ecological studies, and restoration research here and overseas demonstrate that shellfish living on soft-sediments are an integral part of soft-sediment ecosystems and provide significant ecosystem services as live aggregations and by providing dead calcareous shell. These services include:

- Bentho-pelagic coupling, via filtering water and recycling nutrients and oxygen between the sediments and the water column
- Filtration capacity: clearing the water-column of suspended sediments and phytoplankton, potentially mitigating eutrophication and soil erosion;
- Deposition of particulate matter and excretion of nutrients which fertilise benthic primary production (micro-phytobenthos and macro-algae);
- Aggregations stabilise sediments, add biogenic (incl. interstitial) habitat complexity, provide settlement surfaces for other species (including species potentially important to mussel and scallop settlement). These factors increase species diversity, secondary-producer biomass and productivity, and potentially enhance supported biomass of fishes (see below);
- Feedback mechanisms that enhance restoration of benthic primary producers like diatoms, macro-algae and submerged aquatic vegetation (e.g., seagrass);
- Sequestration of nutrients including carbon and nitrogen;

[20] http://www.niwa.co.nz/our-science/freshwater/research-projects/all/restoration-of-estuarine-shellfish-habitat/active-shellfish-reseeding/Restoring-Shellfish-beds-FINAL.pdf

- Dead shells contributing to the carbonate budget of coastal sediments, sequestering nitrogen and carbon, and buffering coastal waters from acidification arising from climate change.

Reef or bed forming shellfish are considered ecosystem engineers as their presence benefits and facilitates the co-occurrence of many other species. In shallow water bodies they can exert strong 'top-down' control of phytoplankton populations (Officer et al. 1982; Alpine and Cloern 1992) (Thompson et al. 2000) increasing light levels at the seabed and facilitating the shift in productivity/energy between the water column and the seabed. In a study of remnant GLM beds McLeod (2009) made estimates of lost secondary small invertebrate productivity following mussel biogenic reef loss in the Firth of Thames, and found the associated small mobile invertebrate assemblages had four times the average density, seven times the biomass, six times the productivity, and greater species richness than bare sediment areas. Morrison et al. (2014a) concluded that based on those estimates, a strong cascading effect to epibenthic carnivores such as fish (including snapper) was highly likely.

Horse mussels

Horse mussel (*Atrina zelandica*) beds provide settlement surface for a diverse range of species including sponges, macro-algae, bryozoans, filter feeding bivalves, and soft corals, and habitat for sea cucumbers, hermit crabs (Morrison et al. 2014a), polychaete worms, ascidians and bryozoans (Figure 3-14). Horse mussels can also co-occur with other diverse habitats such as dog cockles, scallops, maerl, bryozoans, sponges, hydroids and macro-algae. In northern A-NZ horse mussel beds provide a nursery function for juvenile snapper and trevally, as well as supporting other small fishes such as triplefins (Morrison and Carbines 2006; Parsons et al. 2020). Usmar (2010) reported a 10–30-fold difference in snapper associated with artificial horse mussel structures deployed in Mahurangi Harbour as compared with bare sediment.

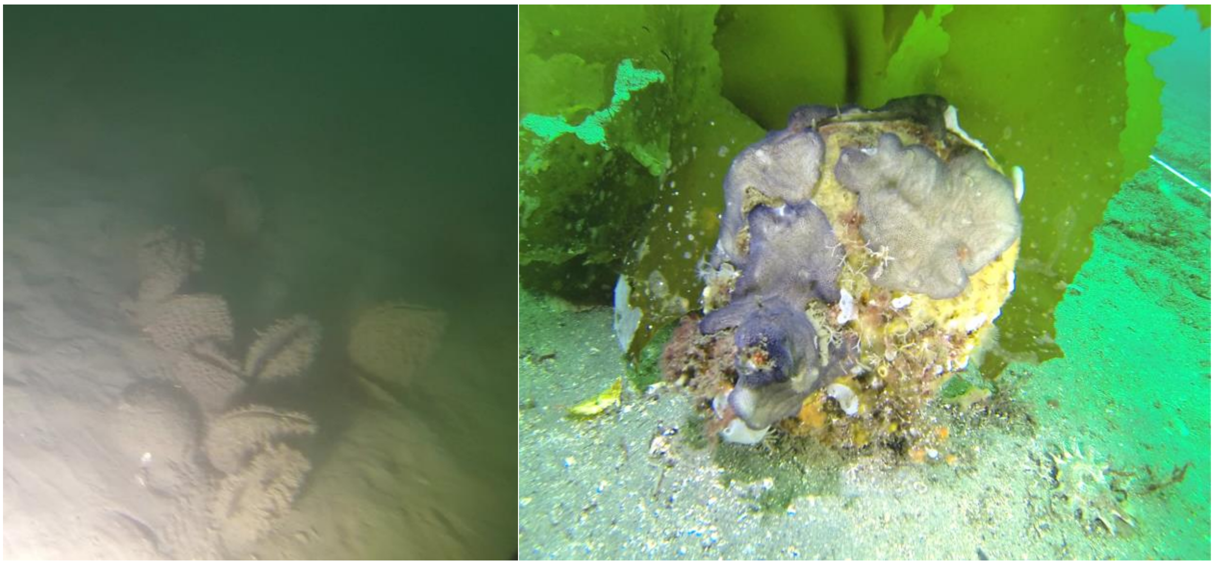


Figure 3-14: Left: a remnant but highly sedimented horse-mussel and scallop bed recently discovered inside/offshore of the Horoirangi Marine Reserve, Tasman Bay. (Photo: Louis Olsen). Right: in contrast, a single horse-mussel colonised by a myriad of flora and fauna providing habitat and ecosystem services in a pristine location in Fiordland (Photo: Sean Handley).

Oysters

Justifications cited for oyster restoration overseas, that come from many sources including economics, conservation, and ecological reasons, are highly relevant in the A-NZ situation. Globally approximately 85% of oyster reefs have been lost worldwide (Beck et al. 2011; Bersoza Hernández et al. 2018). Stressors associated with loss of oyster habitat include disease, habitat destruction, and eutrophication (Jackson 2008).

The economic gains estimated from oyster restoration include the ecosystem services of the American eastern oyster, Crassostrea virginica, (excluding harvesting) was conservatively estimated between $5,500 and $99,000USD per hectare, recouping costs of restoration within 2–14 years (Grabowski et al. 2012). Non-market ecosystem service factors include increasing water quality, provision of habitat for fish, shoreline stabilisation and erosion protection (Coen et al. 2007; North et al. 2010; Grabowski et al. 2012; Gamble 2016). Demonstrating the ecological benefits, restored Australian rock oyster reefs had fivefold higher density and biomass of larger (>2 mm) macroinvertebrates and almost fivefold higher productivity than that of adjacent bare sediments (McLeod et al. 2020). The productivity of infaunal communities was twice as high under oyster reefs than in adjacent bare sediments. As oyster reefs support extremely biodiverse and productive communities and can ameliorate the environmental stress experienced by associated communities, oyster restoration provides an ecosystem-based strategy for assisting the adaptation of marine biodiversity to a changing climate and intensive human encroachment (McAfee et al. 2020).

Negating the hypothesis that restored oyster reefs simply aggregate fish rather than increasing their biomass, a study of restored oyster reefs in Pumistone Passage, Queensland, quantified the effect of restored oyster reefs on new fish populations (Gilby et al. 2021). They found restoration significantly enhanced the diversity and abundance of fish assemblages and the density of harvestable fish at the oyster restoration site by 3.8, 10.7, and 16.4 times, respectively. Restored shellfish reefs significantly enhanced fish abundance, diversity, and the overall carrying capacity rather than simply aggregating them to restoration sites.

Green-lipped mussels

The background and impetus for green-lipped mussel (GLM) restoration in Te Tauihu has been previously reviewed (see: Handley and Brown 2012; Handley 2015; Handley 2017).

3.4.2 Successful techniques: green-lipped mussels/kūtai, including waste shell

With larger scale declines in wild GLM populations in the Hauraki Gulf, most GLM restoration research has occurred in the Auckland region (e.g., McLeod 2009; McLeod et al. 2011b; McLeod et al. 2014; Wilcox 2017). Nevertheless, mussel restoration trials have recently been carried out in Te Tauihu, in Pelorus Sound/Te Hoiere (see case study 7, Figure 3-15, Figure 3-16). There has also been recent promising success using replicated small-scale plots that proved efficient at testing mussel survival/habitat suitability across historic, green-lipped mussel range (Benjamin et al., UoA, *submitted*).

Case study: 7

Promising, green-lipped mussel restoration success in Pelorus/Te Hoiere: The use of small-scale mussel restoration plots across historic gradients of mussel distribution in the inner Pelorus Sound have proved successful using 1.5m x 1.5m deployments (Benjamin et al., *submitted*) (Figure 3-16). This University of Auckland restoration project was funded by the Sustainable Farming Fund, ten marine farming companies, The Nature Conservancy and NIWA. Average survival of mussels after a two-year period was 85% at 4 out of 5 sites, compared with 26.2% reported in a similar study in the Hauraki Gulf (Benjamin, et al., *submitted*.; Wilcox 2017).

A potential contributory factor in the greater success of the Pelorus based experiments is the relatively low-impact method used to harvest and deploy mussels in the Pelorus experiments. In Auckland, HG restoration practitioners' de-clump, clean and soak the mussels in freshwater prior to re-laying them at restoration sites. The soaking is required to fulfil biosecurity requirements to prevent the spread of non-indigenous species (NIS). Following the advice from long-time mussel farmer, Vaughan Ellis (pers. comm.), the Pelorus mussels were not de-clumped. This was permissible because they were being deployed in the same region that they were sourced from, in the presence of known invasive species (e.g., *Undaria, Styella*) that could be removed during harvesting, thereby avoiding the necessity to further stress the mussels by soaking them in freshwater. This may have afforded considerable advantage to the mussels that have shown the ability to quickly re-organise themselves in a manner that allowed them to stay in attached clumps that soon formed byssal attached matts (Figure 3-15). Similar high survival has, however, been observed in the Hauraki Gulf mussels at some locations (Jen Hillman, pers. comm.).

The habit of GLMs to form byssal matts forming a 'crust' on the seabed was earlier recorded by marine biologist and mussel farmer, Graeme Clarke (Clarke 2014; e.g., Figure 3-15). It is thought the stress associated with biosecurity measures and removal of byssus material produced by the mussels, along with high predation from 11-arm starfish, may contribute to their reduced survival in the Hauraki Gulf. Eleven-arm starfish appear to be the largest predator impacting subtidal restoration success, where the inner Pelorus Sound is experiencing fewer starfish compared with the mid-Pelorus Sound. Experiments testing suitability of seeding beds with sub-adult mussels versus adults are currently underway in intertidal and subtidal locations in Kenepuru Sound, with summer heat clearly impacting survival in the intertidal zones (Benjamin, UoA., pers. comm.). Success with similar trials in the Hauraki Gulf were hindered by high predation rates of mobile predators (Alder et al. 2021).

Case study: 7 cont…

Figure 3-15: Green lipped mussels removed from a restoration plot. The mussels have formed a mattress by attaching with byssus atop the soft sediment making them difficult to prise apart. Photo: Emilee Benjamin, Sean Handley.

Case study: 7 cont…

Figure 3-16: Green-lipped mussel restoration trials, Pelorus Sound (photos: courtesy twitter @EmileeBenjamin_). Bags of harvested mussels in transit (top left), newly deployed mussels inside a plastic quadrat (top, middle), mussels deployed in an intertidal plot (top right), divers quantifying survival of deployed mussels (bottom, left), blue cod and spotties swarming over newly deployed mussels that have oriented themselves upward into the water column (bottom, middle), predatory 11-arm starfish, common triplefin and limpet in a mussel plot (bottom, right).

GLM settlement

While it appears restoration of mussel beds is feasible by deploying adult GLMs at many locations of historic distribution, factors affecting long term persistence of beds and innate recruitment of GLM juveniles to adult beds have not yet been resolved (Handley 2016). GLMs have a similar life-cycle to Mytilus edulis in the US; they first settle as larvae on taller reproductive shoots of eelgrass (Zostera marina) that provide a refuge from predators during metamorphosis, before migrating to adult mussel beds when they are of a size that prevents them being cannibalised by adult mussels. Experiments have been initiated in Kenepuru Sound to identify natural settlement associations with local seaweeds (Toone, in prep.).

Waste shells to enhance restoration success

Initial results of an experiment designed to test the efficacy of using waste shell to stabilise soft sediments to enhance GLM survival, at publishing of this report, do not show significant differences with bare sediment control sites (Benjamin, in prep.). In a follow-up survey of waste scallop shell enhancement trials in Tasman Bay for flat oysters (see below), some sites showed elevated levels of GLM recruitment as compared with control sites (Brown et al. 2014). Waste shell therefore shows some promise as a potential tool in shellfish restoration.

Predation

Mortality of GLMs at sites on the outer range of former wild populations appear to be more vulnerable to predation from 11-arm starfish *Coscinasterias muricata* (Benjamin, in prep.). It is unknown if predation levels are linked to the increased prevalence of mussel farms that have been developed since the 1970s, as hypothesised by Inglis and Gust (2003). To enhance survivorship of restored mussel beds, regular control of 11-arm starfish is advised.

3.4.3 Successful techniques: flat oysters/tio, *Ostrea chilensis*

The decline of flat oyster *Ostrea chilensis* stocks in Nelson Bays is summarised in (Michael et al. 2015) and the restoration potential via the use of shell enhancement has been documented in the Doctoral thesis of Brown (2011).

Hatchery settlement of larvae onto waste shell

Restoration of the flat oyster could most easily be achieved by taking advantage of its unique life cycle. Flat oysters are a protandrous hermaphrodite that brood their larvae on the gill of the female oyster for an incubation period lasting from 2 to 9 weeks (Millar and Hollis 1963; Cranfield 1968; Westerskov 1980; Toro and Morande 1998). As a result, fully developed larvae, that have been brooded inside the oyster on its gill, are released and either settle in the immediate vicinity of their parent or drift in the plankton for up to 20 days (Cranfield 1968; Stead 1971; Westerskov 1980; DiSalvo et al. 1983). In the aquaculture production of *O. chilensis*, this phenomenon has been taken advantage of by placing settlement surfaces in tanks holding brooding oysters (e.g., plastic packaging tape, Handley pers. observ.). Once the oyster larvae have settled on the tape and grown to a size that can survive in mesh trays, they are flexed off for 'on-growing'. This simple method could also be used to supply flat oyster juveniles for restoration trials, after a period of nursery growth.

Aggregating shellfish may also be an effective strategy to enhance restoration success for shellfish species that have been exploited below densities that are effective for spawning and reproduction, resulting in recruitment failure (ICES 2002; Dare et al. 2004). Brown (2011) considered that the spat settlement density was strongly related to background adult oyster density, and recruitment to the fishery was likely to be limited in part by the low oyster densities in Tasman Bay. Therefore, to enhance settlement success during shell enhancement trials in Tasman Bay, scallop shell piles were stocked with adult oysters (see: flat oyster case study below, Figure 3-17 and Figure 3-18). Oysters in Tasman Bay brood larvae between November (2004 and 2005, 17%) and December (2006, 23%), with an estimated 55 to 78 % of adult oysters incubating larvae over an entire breeding season (Brown et al. 2010). As such, the timing of waste shell deployment is a critical factor to population enhancement success, as available settlement substrate decreased by 82% in the five months after deployment due to fouling by numerous invertebrates and sedimentation.

Restoration of the flat oyster is likely to be affected by mass mortality due to infection by the protozoan, *Bonamia exitiosa* (Doonan and Cranfield 1992) and more recently an incursion of *Bonamia ostreae* (Lane et al. 2016). Restoration of a mixture of shellfish species may reduce infection if their filtration reduces the exposure to the oysters to *Bonamia*.

Case study: 8

Limited success of flat oyster habitat enhancement in Tasman Bay: Fisheries scale waste scallop shell habitat enhancement trials in 2008 and 2009 by the Challenger Dredge Oyster Co. Ltd. and Brown (2011) found the survival of flat oysters recruited to enhanced shell habitat was generally very low, with a maximum 0.4 m^{-2} oysters after 3.41 years (Figure 3-17, Figure 3-18). This equated to an increase in relative density of commercial sized oysters from ~0.01 m^{-2} prior to enhancement, to ~0.14 m^{-2} at the end of the experiment as compared with stocks that were below threshold densities for commercial dredge fishing (0.02 m^{-2}). Peaks in mortality occurred within experimental plots when oysters were less than one year old, and three years old. Oyster larvae do not settle on surfaces contaminated with fine-grained sediment so the continued >10-fold sediment deposition rates and resuspension of fine-grained sediments by storms and contact fishing methods (Handley et al. 2017a; Handley et al. 2020a; Handley et al. 2020c) is expected to negatively affect shell enhancement success unless shell piles are stocked with brooding oysters or laying of shell is timed to coincide with known larval supply.

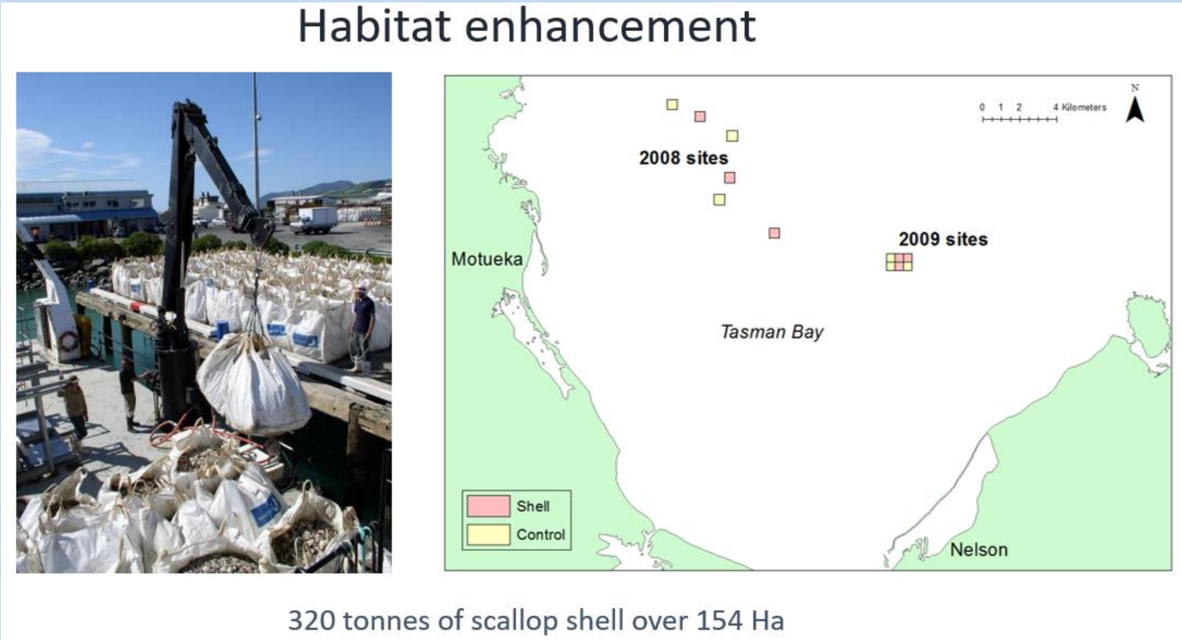


Figure 3-17: Fisheries scale waste shell (scallops) enhancement trials to test flat oyster recruitment were laid in 2008 and 2009 in Tasman Bay. Habitat enhancement was implemented by the Challenger Dredge Oyster Management Co. Ltd. Source: Brown (2011).

Case study: 8 cont…

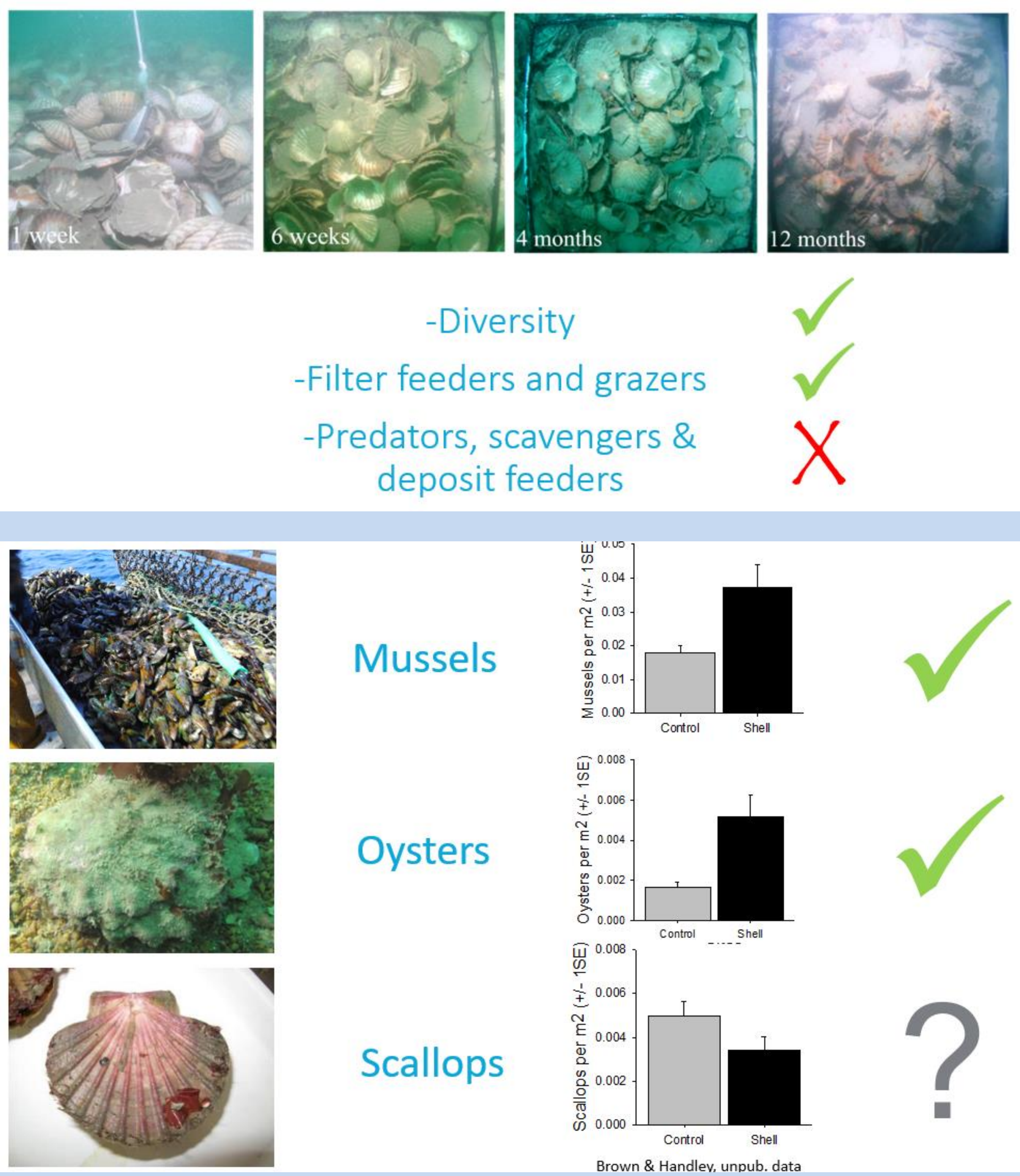
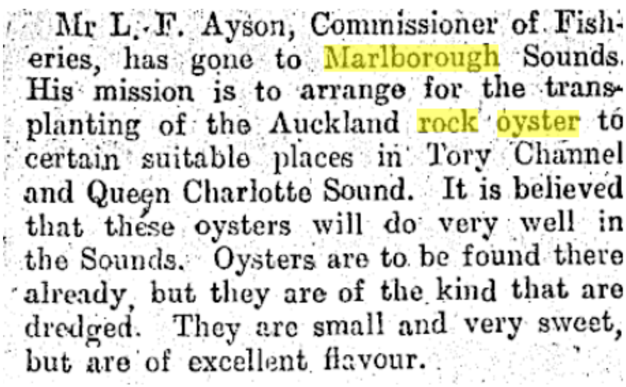


Figure 3-18: After 12 months waste scallop shell became colonised by fouling filter feeders including green-lipped mussels and flat oysters, and grazers, increasing species diversity (top). Four years later, mussel and oyster populations were greater at the 2008 site (bottom) (see also map in Fig. 3-14).

3.4.4 Potential techniques: rock oysters/tio, *Saccostrea glomerata,* Pacific oyster *Magellana gigas*

The majority of international shellfish restoration literature covers oysters, especially *Crassostrea virginica* in the US (Toone et al. 2021). As a result, there have been guides published on shellfish restoration that focus on methods for subtidal or shallow intertidal species, especially the placement of bags or structures of shell seeded with oyster spat prior to deployment (e.g., Beck et al. 2009; Baggett et al. 2013).

In Te Tauihu the invasive Japanese oyster *Magellana* (*Crassostrea*) *gigas* colonised foreshores during the 1970s (Jenkins and Meredyth-Young 1979), and is now cultured commercially in Pelorus Sound, Admiralty Bay, Croisilles Harbour and in Golden Bay (Handley pers. observ.). Historically, the native rock oyster *Saccostrea glomerata*, endemic to the North Island, appears to have been introduced to the South Island in 1915 to extend the North Island fishery[21]. It is not known if the rock oyster continues to persist in Tory Channel, Queen Charlotte Sound, or at other locations within Te Tauihu.


Mr L. F. Ayson, Commissioner of Fisheries, has gone to Marlborough Sounds. His mission is to arrange for the transplanting of the Auckland rock oyster to certain suitable places in Tory Channel and Queen Charlotte Sound. It is believed that these oysters will do very well in the Sounds. Oysters are to be found there already, but they are of the kind that are dredged. They are small and very sweet, but are of excellent flavour.

7

In Australia, loss of oyster reef habitat (*S. glomerata*[22], *Ostrea angasi*) has been of considerable conservation interest, and the focus of their restoration efforts (see: case study box below, Figure 3-19) (Gillies et al. 2015; Diggles and Sain 2016; Gillies et al. 2020; McLeod et al. 2020). Methods developed and tested include stocking artificial rock or block reef habitats, and seeding weathered oyster shells, placed in bags and sacrificial gabion wire baskets (Figure 3-19). Similar methods are also being used in the US including in the billion oyster project in New York to protect against storm surges and hurricane damage that foster restoration of other habitats[23] (Figure 3-20). These methods may be successful in A-NZ and are worth investigating.

With projected sea-level rise and for warmer coastal waters (Law et al. 2018; Lawrence et al. 2018), intertidal and shallow subtidal oyster reef restoration practices (using Pacific or rock oysters) could form living armour along coastal areas exposed to erosion from wave action, especially in locations affected by boat traffic wake (see Ferry-wake section of Handley 2016; e.g., Figure 3-9, Section 3.2.3).

[21] https://paperspast.natlib.govt.nz/newspapers/GEST19151116.2.14?items_per_page=10&page=3&query=%22rock+oyster%22+Marlborough&snippet=true&sort_by=byDA&type=ARTICLE
[22] The Sydney rock oyster and the native northland rock oyster are the same species
[23] Video of how oyster reefs protect against hurricane damage https://youtu.be/UcN6RXT7qpw

Case study: 9

Rock oyster restoration success in Queensland, Australia:

0 Months	6 Months	12 Months	18 Months
			
New mild steel wire has surface rust and no sign of structural degradation. Recycled oyster shells are clean with no accretion.	6-month-old ROB showing more surface rust and recruitment of small oysters on the recycled shells. Up to 20,000 spats have been recorded on individual ROBs at this stage.	This ROB is 12 months old. The integrity of the structure is still intact, but we see signs of degradation. Rock Oysters (*Saccostrea glomerata)* have started to encrust the wire mesh.	This ROB is 18 months old. The integrity of the structure is starting to fail but the steel is mostly in place. Rock Oysters are now growing over the mesh and encasing the steel. The oysters have formed a fused together clump of living shell that mimics remnant shellfish reefs in Moreton Bay.

Figure 3-19: Queensland rock oyster restoration trials using gabion basket "ROB" (Robust Oyster Basket) with timeline showing oyster growth and mesh degradation. (images courtesy: Robbie Porter, oz.fish.org).

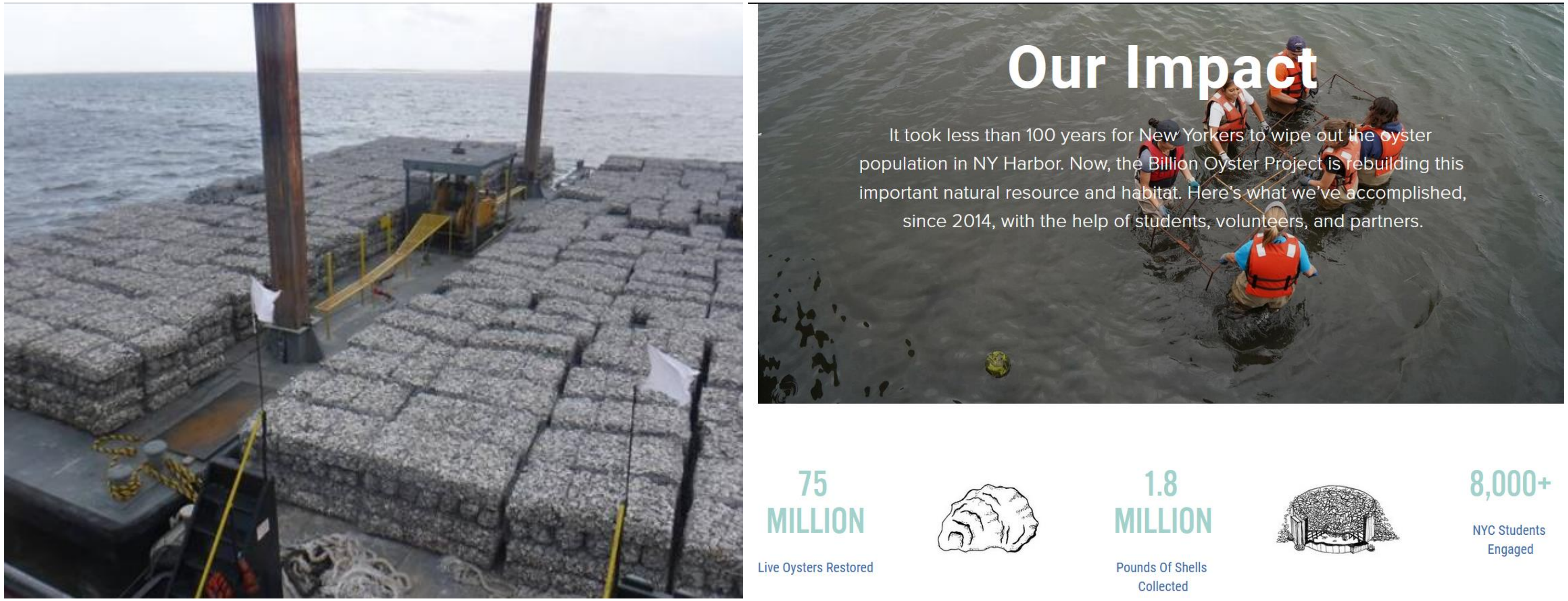


Figure 3-20: Large scale use of oyster shells in gabions. The shellfish reefs were restored near Soundview Park in the Bronx during the Billion Oyster project in New York Harbour (https://www.billionoysterproject.org/, Courtesy Ben Diggles, Diggsfish.com).

3.4.5 Successful techniques: scallop/tipa

The restoration of scallops *Pecten novaezelandiae* in Te Tauihu has been a hot topic since the collapse and lack of recovery of commercial beds in Nelson Bays since 2016, not least because of the cultural value and the lost economic opportunity estimated at ca.$90M/annum (Michael et al. 2015). The history of the fishery extends back to the mapping of stock in the early 1960s (Choat 1960; Tunbridge 1968) in conjunction with exploitation of GLM and flat oyster stocks. Following the decline of the fishery in the 1980s, a scallop enhancement programme was developed using Japanese methods of capturing wild larvae that settle and form spat on artificial collectors (fine mesh spat bags) and releasing them into commercial fishery areas that are closed until the scallops reach legal size (termed 'primary enhancement'; Tuck and Williams 2012).

Spat that have fallen off the spat bags and associated ropes can also be dredged up using fine mesh liners and relocated to commercial fishery areas (termed 'secondary enhancement'). Although the scallop enhancement program appeared to have a significant effect on survey catches (Tuck and Williams 2012), the subsequent decline of scallop stocks in Nelson Bays (Williams et al. 2014) and the Marlborough Sounds (Williams et al. 2019) suggests something in the growing environment has changed, including the increasing prevalence of easily resuspended soft sediment (Handley et al. 2020c). Tests of survivorship of spat caught in spat bags showed them to be robust to periods of aerial exposure, further implicating issues at the seabed rather than changes in scallop spat enhancement methods (Handley 2016).

The culture of scallops in aquaculture systems has been tried in A-NZ using wild caught spat by several commercial Companies including Kiwi Mussels, Aqua King, Sanford, and Talleys, but none have been successful to date due to high mortality and issues with labour costs and fouling rendering aquaculture uneconomic (Handley pers. observ.). The use of wild spat catching methods could be used to provide spat for local restoration trials (within bounds of biosecurity rules regarding moving spat within and between regions in Te Tauihu). The ongoing success of spat catching at the 'ring road' spat catching sites in Golden Bay is likely due to eddies or gyres established by northwest to westerly winds flows, with these wind driven circulations thought critical to concentrating and retaining larvae in the areas where spat catching gear is deployed (Michael et al. 2015). Analysis of the marine farming spat catching results from the Marlborough Sounds might provide insight into potential scallop spat catching sites. Anecdotally, scallop larvae can settle on marine farm structures, especially anchor warps. These structures later shed spat when they have reached a larger size, and some appear to survive on the seabed, especially in sandy locations (Handley, pers. observ.).

The long-term persistence of restored scallop beds, like other shellfish species discussed herein, may be contingent on lost or dwindling settlement substrata for larvae as primary settlers. Scallop larvae settle and attach themselves with byssus threads to filamentous materials such as seagrass (*Zostera* sp.) debris, filamentous red, and brown algae (including *Cystophora retroflexa*), sea fans (hydroids), horse mussels (*Atrina zelandica*), and shell fragments (Bull 1976; K. Michael, NIWA, unpub. data). Therefore, the identification and mitigation of stressors responsible for wider system and habitat decline and tipping points would be highly beneficial to shellfish restoration (Hewitt and Thrush 2019; Handley et al. 2020b). Three studies are underway in this respect: (i) using experimental and modelling techniques to assess the cumulative effects of a range of physical, biological, and ecological stressors (including fishing) on scallops and scallop habitat in the Marlborough Sounds (Project: ZBD2020-09, Drew Lohrer, NIWA, pers. comm.); (ii) species-habitat associations for the New Zealand scallop (Project: SCA201703, James Williams, NIWA, pers. comm.); and (iii) Ecosystem Based

Management of shellfish in Marlborough Sounds (Project: SUSS22303, Vonda Cummings, NIWA, pers. comm.).

3.4.6 Establishing techniques: horse mussels

There are no studies published on the culture or methods for restoration of horse mussels *Atrina zelandica* (Cameron Hay, pers. comm., B. Skelton, pers. comm.). Historical accounts and Ecologically Significant Marine Sites Monitoring indicate that horse mussels were an important component of soft sediment seabed communities in Te Tauihu, forming extensive beds in parts of Tasman Bay and the Marlborough Sounds (Davidson et al. 2011; Handley et al. unpub. data). Anecdotally, they appear to undergo boom and bust cycles of recruitment and mortality.

While there have been no attempts to restore *A. zelandica* beds in A-NZ, several studies have successfully transplanted live *A. zelandica* to different locations, in order to understand how suspended sediment concentrations affect their structure and function and the surrounding seafloor community. *A. zelandica* have a natural distribution threshold that is controlled by suspended sediment concentrations, which negatively influence their condition (health; Ellis et al. 2002). Laboratory and transplantation experiments show that *A. zelandica* are sensitive to suspended sediment levels as low as 80 mg/l, with adverse suspended sediment concentrations potentially explaining the loss or distribution in locations exposed to high sediment loadings (Ellis et al. 2002). Given *A. zelandica* add complexity to soft sediment habitats, modifying boundary flow conditions and strongly influencing local macrobenthic community composition (Cummings et al. 1998; Green et al. 1998; Cummings et al. 2001), they would be an aspirational candidate for restoration if they could be cultured and/or enhanced in areas of suitable sedimentary climate.

Related taxonomic and morphologically similar species have been cultured overseas including: *Atrina maura* in Mexico (Robles-Mungaray et al. 1996; Leal-Soto et al. 2011), *Pinna pectinata* in China (Guo et al. 1987), *P. rugosa* (California) and *P. bicolor* (Australia) spat have been collected in mesh bags in the similar method used for scallops (Cendejas et al. 1985; Sumpton et al. 1990; Beer and Southgate 2006). The University of Auckland (UoA) have funded a Post-Doctoral position to investigate spat catching and hatchery production of *A. zelandica* (B. Skelton, UoA, pers. comm.). *Atrina* have had their gonads manually stripped, gametes fertilised, and larvae cultured for 12 days, but not to settlement yet (B. Skelton, UoA, pers. comm.).

3.4.7 New techniques, innovations

Reestablishment of habitat modifiers often hinges on reinforcing feedbacks generated by traits that emerge when individuals naturally aggregate, that results in density or patch size-dependent establishment thresholds (Temmink et al. 2021). Therefore, to overcome establishment thresholds that limit natural or innate recovery, adult or juvenile habitat-forming species are often transplanted in clumped designs (e.g., cockles; Cummings et al. 2007), or stress-mitigating structures can be installed (e.g., Figure 3-21). However, restoration approaches that focus on a single life stage do not address the potential that bottlenecks that may limit survival can occur at other life stages that are hard to control. To overcome such barriers, ‘life cycle informed restoration’ was tested in experiments in intertidal soft-sediment systems in Florida and the Netherlands where oysters and mussels act as reef-building habitat modifiers. Experiments used both biodegradable structures for oysters and coir rope for mussels. Those temporary structures successfully facilitated larval recruitment and enhanced post-settlement survival by lowering predation through provision of habitat complexity (Temmink et al. 2021).

Other innovative methods that deserve attention are the previously mentioned aggregating of shellfish (Section 3.4.3) to ensuring they are at adequate densities to effectively synchronously spawn and fertilise gametes. Also, analysis of hydrodynamic conditions and the use of particle tracking models might facilitate restoration or spat catching success in areas where eddies occur, to predict where larvae may travel (e.g., Lundquist et al. 2009). Because historic shellfish beds in Te Tauihu did not form patches comprising single species or monocultures, but rather appeared to comprise mixed species distributions (Handley 2006; Handley and Brown 2012; Handley et al. 2019a), multi-species restoration efforts might be more successful.

Case study: 10

Success in overcoming 'establishment thresholds':

Figure 3-21: Temporary biodegradable structures can be used to overcome establishment thresholds, reduce stress, and provide habitat complexity increasing restoration success. Structure used to aid plant establishment (left, source: Fivash et al. 2021), biodegradable BESE-elements (centre, source: www.bese-products.com), ROB gabion wire baskets (right, image courtesy: Robbie Porter, oz.fish.org).

As marine farms afford protection from mobile fishing gear, they may also be a valuable tool in shellfish bed restoration. Mussels are often shed from mussel farms throughout farming regions, but only in very rare cases do mussels survive directly beneath mussel farms (Handley, pers. observ., R. Davidson, pers. comm.). Such rare cases include the seabed below spat holding farms in Forsyth Bay (R. Davidson, pers. comm.) that may have resulted from spat being released from the farm above. Because elevated numbers of eleven-arm starfish *C. muricata* are often found beneath farms, their population was hypothesised to be enhanced by increasing numbers of mussel farms (Inglis and Gust 2003). Their detrimental effects as predators of restored GLM beds in Pelorus Sound have been documented (Handley pers. observ., Benjamin, et al. *submitted*), and removal of these predators should help in shellfish restoration efforts.

Not discounting the benefits marine farms can provide (e.g., Figure 3-22, (Stenton-Dozey and Broekhuizen 2019; Theuerkauf et al. 2021), the temporary use of marine farms to raise juvenile shellfish (e.g., the 'ring-road' spat catching sites in Golden Bay) could also be used on a rotational basis to seed areas of seabed formerly denuded of benthic shellfish.

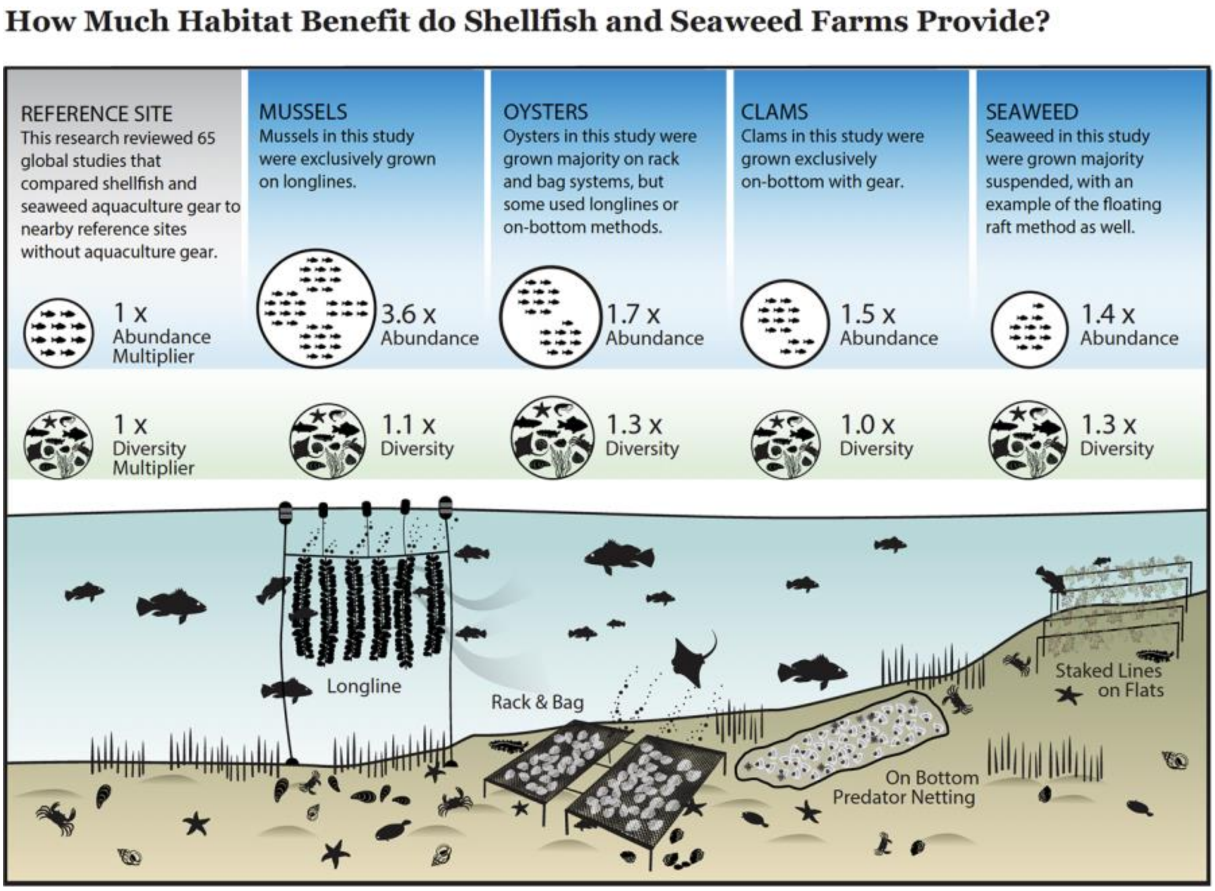


Figure 3-22: Ecosystem benefits provide for by shellfish and seaweed farms. Infographic provided by The Nature Conservancy.

Other built structures have also been used to collect and hang shellfish off, in efforts to restore filtration in modified locations like marinas and to culture shellfish to enable restoration of rare shellfish species. For example, 38 seeded mussel lines were attached to Te Wananga at the ferry basin, Te Waitemata (Waitemata Harbour), aimed at illustrating how Māori practices can provide natural solutions to problems[24]. "*A single mature mussel can filter up to 150 to 200 litres of seawater a day taking in phytoplankton for nourishment as well as removing pollutants, and effectively act as bio-indicators of aquatic health, helping us monitor for any unwanted invasive species in the water*." Another two examples are the Vertical Oyster Gardens (VOG) developed in heavily polluted Tampa Bay in the US[25], and restoration efforts in Pumistone Passage, Queensland Australia where oyster gardens are being used to revive the native rock oyster *S. glomerata* (Diggles and Sain 2016) Figure 3-23). "Oyster gardens" could perhaps be used to reinvigorate the native flat oyster populations in Te Tauihu, or mussel lines be used to monitor pollution and invasive species, like in the Waitemata.

[24] https://www.stuff.co.nz/environment/300311159/mussel-ropes-to-filter-seawater-at-new-downtown-auckland-space-te-wnanga
[25] https://tampabaywatch.org/restoration/oyster-communities/vertical-oyster-gardens/

Figure 3-23: Examples of "oyster gardens" used to reduce pollution, habitat loss and provide stocks for restoration of rare species. Above: "Vertical Oyster Gardens" deployed in Tampa Bay, U.S., and native rock "oyster gardens" used in Queensland, Australia (Diggles and Sain 2016).

3.5 Artificial reef structures, wrecks

Table 3-5: Summary of the salient points from the review of artificial reefs/wrecks with relevance to restoration activities in Te Tauihu. The remainder of Section 3.5 details the information behind this table.

1. Potential use, suitable locations to try in Te Tauihu?

- Habitat creation or enhancement
- Where to try:
  - Artificial reefs: Degraded soft sediment habitats or locations requiring shoreline erosion protection for habitats, properties/amenities (e.g., eroded shorelines on properties in Kenepuru Sound, Tory Channel).
  - Sinking of wrecks: Atop degraded soft sediment habitats to enhance and aggregate recreational or commercial fish, or to deter trawling/dredging (fishing exclusion zones)

2. Status and why the habitat is important

- Status: multiple wrecks are present, but no artificial reefs installed in Te Tauihu
- Increasing in prevalence/use as coastal protection (urbanisation, industrial developments, infrastructure, sea-level rise)
- Provides habitat for marine algae, invertebrates and fish, enhancing recreational and commercial fish stocks
- Many forms/options (e.g., wharves, marinas, groins, softening shoreline/infrastructure edges (see Section 3.2), designer artificial reefs, wrecks)
- Fisheries benefits additive (can change species composition)
- SCUBA Dive attractions, increased diver safety and diverting damage of natural reefs

3. Main, most recent, or successful techniques and methods used in restoration actions

- Reef elements designed to provide a mix of micro- to macro-scale rugosity, refugia and habitat
- Natural products included in concrete blends (e.g., oyster shell, hemp, blast furnace waste)
- Scuttling of derelict/unwanted vessels

4. New technique, approach, or technological innovation that could make a difference in up-scaling restoration actions

- Sustainable/ecologically designed formulations (concrete)
- Mass production methods vs local community led/constructed initiatives

5. Major barriers or deal breakers that prevent scaling-up restoration

- Public acceptance, perceptions, advocacy of benefits
- Testing/validation of mimics that are tailored to provide context dependent habitat
- Temporary biodegradable elements vs permanent engineered structures
- Shifting baselines (lack of knowledge of what has been lost)
- Wrecks: Public/cultural acceptance, perceptions, advocacy of benefits
- Testing/validation of ecological/fisheries benefits of existing wrecks
- Could result in recreational depletion of top predators (e.g., sharks)

6.	Timescales for restoration
▪	Design/context/location dependent (from immediate provision of refugia to long-term colonisation and species succession)
▪	Wrecks: immediate to months to decades
7.	**Other point/key message relevant to the habitat – factors affecting success**
▪	Positive effects depend on site/context and scale (micro- and macro-scale combinations best)
8.	**Examples of where this has been tried before, likelihood of success?**
▪	Artificial reefs: Port of Napier, Pania Reef; numerous overseas successful case studies of deployments
▪	Successful wrecks: e.g., HMNZS Canterbury, Bay of Islands; the Hippalos, Blumine Island; Mikhail Lermontov, Port Gore
▪	Likelihood of success: **high** ✓

3.5.1 Why install artificial reefs, wrecks?

The European Artificial Reef Research Network (EARRN) defines Artificial Reefs (AR) as `submerged structures placed on the seabed deliberately, to mimic some characteristics of natural reefs' (Pickering et al. 1999). Artificial reefs are a contentious issue in restoration, with most used to create a new ecosystem not already present and their use justified as part of physical protection, mitigation, or enhancement (Papadopoulou et al. 2017). That said, artificial reefs have a long human history of some 3,000 years in the Mediterranean and have been constructed from artificial and natural materials to provide settlement surfaces for biofouling species and habitat in the form of rugosity, crevices, overhangs and caves that also provide refugia for mobile species (Seaman 2000; Ramm et al. 2021; Vivier et al. 2021). They have a multitude of forms, from unintentional shipwrecks, scuttled vessels, discarded vehicles and machinery, stone and rock structures, to designer-built box or ball structures. With the global degradation of coastal habitats such as salt marshes, coral reefs, mangroves, seagrass and shellfish reefs, the provision of ARs can provide space for colonisation surfaces and complex physical habitat for benthic invertebrates, provide prey for consumers and increase fisheries productivity especially in habitat-limited locations (Sherman et al. 2002; Lemoine et al. 2019; Vivier et al. 2021). Other benefits include recreational diving, aquaculture, environmental aid and scientific experimentation (Seaman and Jensen 2000; Ramm et al. 2021), but also protection from bottom trawling, promotion of conservation, and bio-filtration structures (Jensen and Spanier 2004; Ramm et al. 2021).

Reef structures improve fish production

Of all the objectives ARs/wrecks have been used for, a world-wide meta-analysis of 162 ARs found fisheries enhancement projects showed the "highest of efficiencies" (Vivier et al. 2021). Artificial reefs sequentially deployed in southern Portugal were modelled to increase fish abundance and the carrying capacity of the environment by 35% for the two-banded seabream *Diplodus vulgaris* (Roa-Ureta et al. 2019). As was the case with shellfish reefs (see Section 3.4.1), the attraction hypothesis was rejected over the production hypothesis by model calculations that resulted in semi-industrial fishing harvest rates increasing 3-fold. The authors of the study concluded that new production created by ARs spilled over in significant amounts to the wider areas of the surrounding continental shelf, enhancing fishable stocks. A study in Brazil that tested the attraction versus production effect on fish of a reef complex installed in 1996 found that 10 main species collected in the region were more abundant in the AR than at control locations (da Costa et al. 2022). Furthermore, there were more juvenile fish, and adults with larger gonads associated with ARs, leading them to conclude that the reef complex is probably functioning as a fish attractor as well as enhancing production. In an Australian study of the effects of artificial reef installations in a degraded estuary in Botany Bay, Sydney, the authors found increases in abundance across multiple seasons and years of three species of Sparidae fishes important to fisheries (Folpp et al. 2020). They consider that the increased abundance was 'produced' by increasing the carrying capacity in the estuaries by providing refuge that would otherwise be unavailable, rather than being attracted to ARs because natural rocky-reef sites showed no changes in Sparid abundance.

ARs for erosion protection

ARs have been deployed as coastal protection devices in response to sea-level rise. For example, 10,600 AR modules were deployed in two layers to protect Vaan Island in the Gulf of Mannar, southeast India were used to protect the beach from erosion and to enhance biodiversity in a coastline affected by severe climate change impacts (Mathews et al. 2021). Similarly, AR deployments following hurricane damage in the Caribbean resulted in fish biomass 4.6 times higher than reference sites covered by bare sand (Hylkema et al. 2020). See also Section 3.2.3 above with respect to principals and designs of urban/industrial infrastructure.

Benefits to tourism, recreation, safety

A major benefit encouraging the use of shipwrecks is their attraction for tourism and SCUBA diving (Kirkbride-Smith et al. 2013; Lima et al. 2019; GBSDC. 2020). Dive wrecks can contribute millions of dollars to local economies. For example, in Sydney an economic analysis for a proposed wreck deployment estimated benefits between $12.4M and $48.6M over five years, with a cost : benefit ratio of between 2.1 and 4.8 (GBSDC 2020). Wreck diving was one of the fastest growing recreational activities during the 1980s and 1990s (Davis et al. 1995) as divers seek new challenges including wreck diving (Cater 2007). As the dive community has increased in size and matured, there has been an increased interest in and subsequent demand for wreck diving opportunities (Edney 2006). As the potential economic benefits of wreck diving are increasingly recognised by governments and communities, increasing numbers of vessels have been sunk to form artificial reefs, particularly in Australia and North America, (Seaman and Jensen 2000; Dowling and Nichol 2001; Pendleton 2005; Stolk et al. 2007).

Given inexperienced divers can resuspend sediment on top of organisms like corals, topple corals, touch sensitive corals or other organisms thus causing harm (Polak and Shashar 2012; Tynyakov et al. 2017), there has been a movement towards teaching divers how to dive safely on artificial reefs rather than natural ones in order to protect natural reefs (Kirkbride-Smith et al. 2013). As many purposefully sunken ships are modified and cleaned prior to their sinking, wrecks are considered safer options for training and skill developmental opportunities (Edney and Spennemann 2015).

3.5.2 Optimum conditions/design of modular ARs

AR size, shape and orientation

If the aim is to mimic adjacent natural reefs, the structural design of ARs is very important. The most efficient shape was cylindrical, but size had a larger influence especially for AR volumes higher than

1000 m^3 (Bortone et al. 2011). Larger ARs were significantly correlated with maximum effectiveness due to the creation of upwelling that promotes primary production and, hence, the local fishery (Bortone et al. 2011). An increase of surface heterogeneity and complexity promotes biodiversity and facilitates colonization (Boaventura et al. 2006; Loke and Todd 2016). Large holes and orientation to currents and/or light were also important determinants in the holding capacity of fish of reproductive ages, and colonisation and benthic biodiversity respectively (Boaventura et al. 2006; Bortone et al. 2011; Loke and Todd 2016). This is because vertical surfaces host communities with different species presence, composition, and densities in comparison to horizontal surfaces (Perkol-Finkel et al. 2018).

In a study that compared different AR designs (reef balls, layered cakes, and piles of basaltic rock, Figure 3-24), fish abundance was greatest (3.8 times) on layered cakes, followed by rock piles then reef balls (Hylkema et al. 2020). Three-dimensional modelling revealed that although layered cakes had a smaller gross volume, shelter volume and total surface area than reef balls, layered cakes provided more small shelter sites important to fish. For cost however, rock pile plots (e.g., Figure 3-27), with intermediate performance, were 4–10 times cheaper to construct. For maximising enhancement of fish, they concluded more effort should be applied to deploying ARs with higher shelter density.

An Australian study compared fish abundance, diversity, and community composition on custom-designed reef structures (CDARs) versus Reef Balls (RBs) with nearby natural reefs in Port Phillip Bay (Komyakova et al. 2019). They found that AR design influenced fish species richness and community composition, with community structure converging closer to natural reef with time on more complex reefs. Although densities on both AR designs were markedly lower than natural reefs at some locations, fish species richness on the custom-designed reef structures was, on average, two times higher than natural or Reef Ball reefs. But there were large dissimilarities in fish community composition among reef types across all locations and years.

Modular ARs come in a large range of shapes and sizes, available for purchase directly, or with designs and moulds available (e.g., Figure 3-25, Figure 3-26) with some designed specifically as fish attraction devices (see: NSW case study box, Figure 3-28).

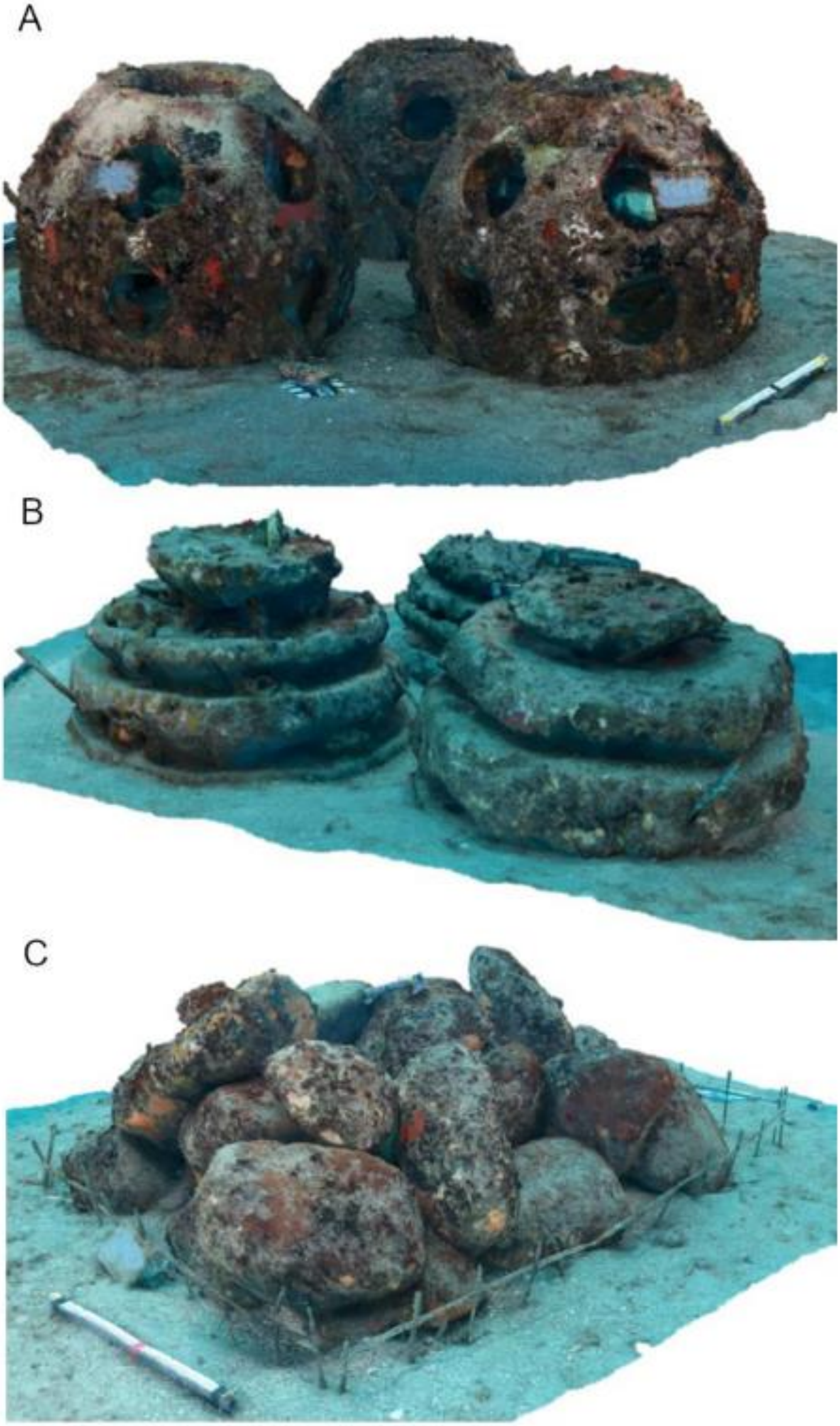


Figure 3-24: Three different types of artificial reef: reef balls A), layered cakes B) and rock pile C) plot. Each plot covers approximately 2 m^2 seafloor area. Source: Hylkema et al. (2020).

Figure 3-25: Diverse examples of use-cases and locations where reef balls have been installed. Sources: http://www.reefball.org/, https://reefball.wixsite.com/habitat, http://www.reefball.com/

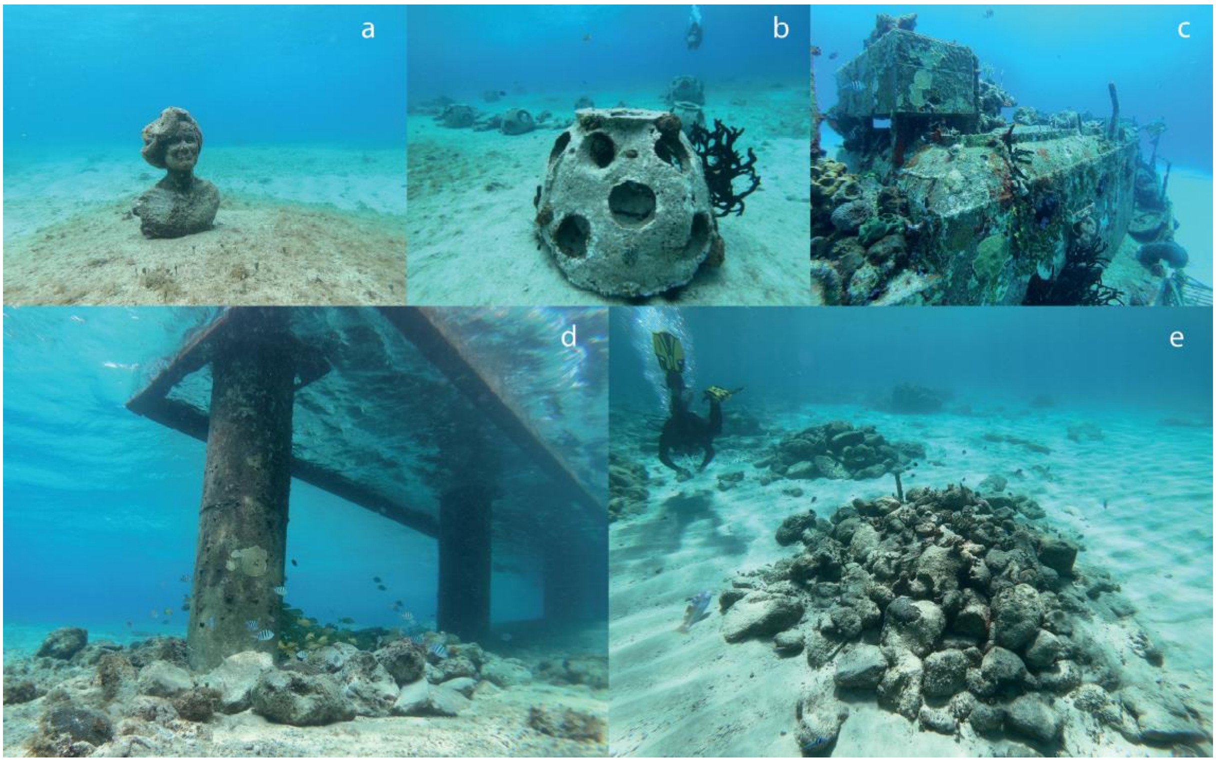


Figure 3-26: Examples of different artificial reefs: a) artworks, b) prefabricated modules, c) sunken artifacts, d) infrastructure, and e) traditional structures. Photographs: SCT Mexico (2019) Source: (Tickell et al. 2019).

Modular AR construction materials

A meta-analysis of AR design, objectives and effectiveness, reported that inert materials like concrete associated with biomimetic designs (i.e., mimicking nature) showed the greatest ecological benefits of reefs to the local environment with the highest efficiencies from ARs deployed to enhance fisheries (Vivier et al. 2021). This result reflected the high rates of fixation/attraction and colonisation associated with concrete structures.

As the use of concrete has a high carbon-footprint, more sustainable solutions are now being used. For example, ECOncrete® includes the addition of marine products including oyster shells and other environmentally sensitive technologies, reducing the environmental footprint of ARs (Walles et al. 2016; Perkol-Finkel et al. 2018; Georges et al. 2021; Vivier et al. 2021). Other initiatives include testing recycled ground granulated blast-furnace slag (GGBS), and partial replacement of coarse aggregate with hemp fibres and recycled shell material that reduced the carbon footprint and incorporated carbon storage (Dennis et al. 2018). After 12 months the hemp and shell concrete supported significantly more cover by living organisms than the standard GGBS control blend. Taxon richness, especially of mobile fauna, and the overall species pool were also higher on the hemp concrete. These alternative materials were considered to be of equal or better habitat suitability compared to ordinary GGBS based concrete (Dennis et al. 2018).

A similar study comparing different blends of alternative waste products including GGBS described the optimum percentage replacement of cement with 5% Homra (fly ash or clay- brick wastes; El-Gamal and Selim 2017). In a trial comparing different proportions of oyster shell used in concrete, 20% oyster shell (see paper for formulation) showed a better acclimatisation[26] of the microphytobenthos than in other concrete blends, despite a lower colonization on this concrete (Georges et al. 2021). Their results suggested that the accumulation of biofilm plays a role as protective barrier against the action of chloride ions in seawater, that affect the strength of the concrete.

Case study: 11

Success at Pania Reef, Napier Port: As part of the 2019 "6 Wharf" redevelopment of the Port of Napier, an AR built from re-purposed limestone was installed 1.4 km northeast of Pania Reef at the "*Gwen B"* shipwreck site. This reef consisted of ca.15,000 m^3 piles of rock and boulders added to the seafloor. Prior to the port development, Napier Port partnered with mana whenua hapū to develop a Marine Cultural Health Programme (MCHP) to protect, monitor and assess (before, during and after) the cultural health of the marine environment, in particular Pānia Reef, during the 6 Wharf project[27]. The health of the reef was monitored at 15-minute intervals through the deployment of monitoring buoys linked to an interactive website and dashboard. The monitoring buoys record, and the data is displayed via a web portal[28]. Tide, current speed/direction, and turbidity, are presented so that dredging, and spoil relocation could be adaptively managed[29]. By 2021 the boulder habitat provides overhangs, services and rugosity that have been colonised by encrusting and mobile species including fish and rock lobster (Figure 3-27).

Figure 3-27: Artificial reef built from re-purposed limestone from the Port of Napier, installed northeast of Pania Reef. (Photographs courtesy of Robyn Dunmore, Cawthron Institute).

[26] Acclimatization is a process where continuous exposure of a microbial population to a chemical results in a more rapid transformation (biodegradation) of the chemical than initially observed
[27] https://marineculturalhealth.co.nz/panias-reef-focal-point-in-port-plan/
[28] https://6wharf.vdvcloud.com/vdv.php/dashboard/361
[29] https://www.napierport.co.nz/our-business/our-future/6-wharf/building-in-partnership/marine-cultural-health-programme/

Case study: 12

Successes in New South Wales, Australia: The NSW Department of Primary Industries artificial reef program is building both estuarine and offshore reefs (9 sites) to enhance recreational fishing using innovative and state of the art designs[30]. The reefs are designed to deflect currents to create eddies and upwellings to provide intricate habitats for a variety of fish species (Figure 3-28). The project is funded from the NSW Recreational Fishing Trust with the reefs designed and located for the benefit of recreational fishers. The structures attract species such as kingfish, trevallies, snapper, morwongs tailor, salmon, mulloway and leatherjacket.

Figure 3-28: Examples of steel and concrete artificial reefs deployed by NSW Department of Primary Industries to enhance recreational angling. Source: https://www.dpi.nsw.gov.au/fishing/recreational/resources/artificial-reef.

[30] https://www.dpi.nsw.gov.au/fishing/recreational/resources/artificial-reef

Wrecks as artificial reefs

Sunken vessels have traditionally been considered beneficial to the marine environment as they provide new habitat and a refuge for fish and benthic communities (Pickering et al. 1999; Morrison 2018). In New Zealand, sunken wrecks have demonstrated that they can be used to establish viable fish and benthic communities (see case study boxes below, Figure 3-30 -Figure 3-32; Forever 2012a; Booth 2020). Like ARs, sunken vessels are quickly colonised by encrusting flora and fauna that then provide beneficial habitat for fish and mobile invertebrates. As with modular ARs providing new habitat, the benefits of shipwrecks appear to be additive rather than merely attracting mobile species away from natural reefs. The scuttling or intentional deployment of wrecks documented in a standardised database recorded a total of 1907 vessels, of which 1739 (91%) were in the USA, 64 in Australia, and 6 in A-NZ (Ilieva et al. 2019; Figure 3-29).

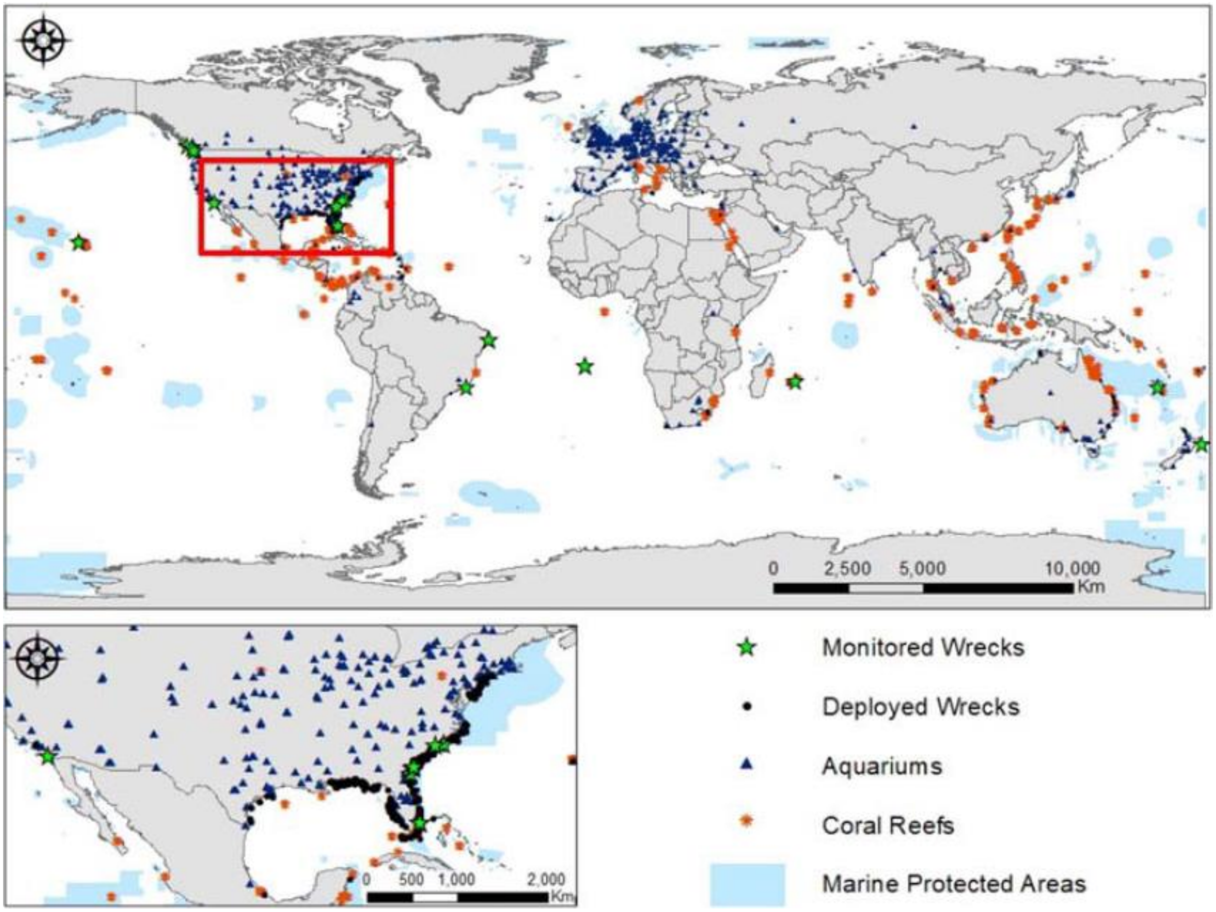


Figure 3-29: Global distribution of monitored and deployed wrecks, aquariums, coral reefs and marine protected areas. Source: Ilieva et al. (2019).

Case study: 13

Success, HMNZS Canterbury: was scuttled in Maunganui Bay (Deep Water Cove) in the Bay of Islands on 3rd November 2007. She lies in 38 m of water, a depth more suited to advanced or technically trained divers that can use mixed gases (e.g., Enriched nitrox). The wreck was sold to Te Rawhiti Enterprises (the local Hapū) for one dollar (the same amount that the Canterbury Trust paid the New Zealand Navy) on 15 July 2008. A Section 186A Rahui was established in Maunganui Bay in November 2010, three years after the scuttling of the frigate. The HMNZS Canterbury Artificial Reef, with its high degree of vertical surfaces, differs from surrounding natural reefs of Maunganui Bay that are characterised by horizontal and/or slightly inclined surfaces.

The process of colonisation by algae, invertebrates and fishes continues to take place, and may not have yet reached its climax state (Figure 3-30). Comparative surveys undertaken in 2011 vs 2012 showed large increases in sponge and tubeworm cover on all surfaces, with decreases in filamentous algae and lithothamnion (encrusting red alga) 'paint' (Forever 2012b). Densities of an essentially unfished reef-associated generalist indicator species, the leatherjacket *Parika scaber*, along with the planktivorous two-spot demoiselle *Chromis dispilus* and the sought-after generalist snapper *Pagurus auratus* were all lower around the Canterbury in 2012 than in 2011 (Booth 2020). A crude estimate of snapper associated with the Canterbury in 2012 was 118 g per 100m^{-2} compared with 157 g in 2011 (Forever 2012b). In surveys of natural reefs in Maunganui Bay, mean numbers of fish and mean species richness were much the same in 2012 as they were in 2011 (Forever 2012b). The main issues of concern by Northland Regional Council are the stability of the vessel, and signs of non-indigenous species (NIS) of which none have been discovered to date.

Figure 3-30: HMNZS Canterbury circa 1996[31] (top) and view from the bridge ca.2020[32] (bottom).

[31] HMNZS Canterbury (F421) - Wikipedia
[32] https://www.divenow.co.nz/go-diving/canterbury-wreck

Case study: 14

Accidental success, wreck of Hippolas, Bluemine Island: During the recent multibeam acoustic survey of Queen Charlotte Sound (Watson et al. 2020) 10 ship wrecks were surveyed, some of them discovered for the first time during the survey. Those discoveries led to the idea of sinking old ships, mooring blocks or mussel-shell structures off the shores of Marlborough to help encourage reef growth and fish life[33]. For example, the Hippalos that sunk near Blumine Island in 1909 is now colonized by a wide range of invertebrates and provides habitat for rock lobster and fish aggregations (Figure 3-31).

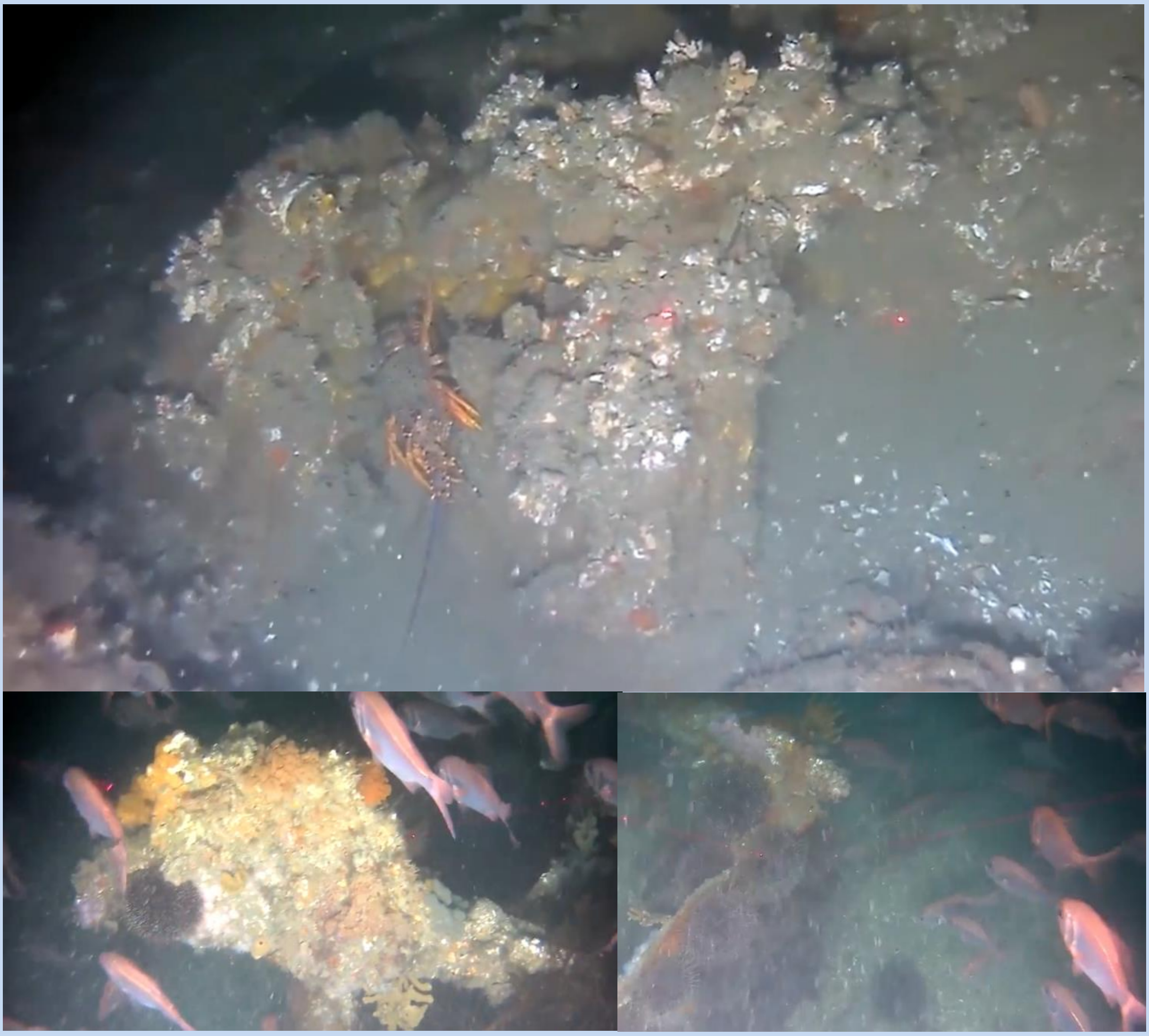


Figure 3-31: The wreck of the Hippalos encrusted with invertebrates and attracting fish aggregations. Source: NIWA[38].

[33] https://www.stuff.co.nz/national/107760738/artificial-reef-strategy-floated-for-marlborough-sounds

Case study: 15

Accidental success, Mikhail Lermontov, Port Gore: The accidental sinking of the Russian cruise liner in Port Gore, Mikhail Lermontov in 1986, after hitting rocks off Cape Jackson, has provided habitat for invertebrates, seaweed, and fish (Figure 3-32). In a survey for juvenile fish and their habitats in the Marlborough Sounds, the Lermontov wreck was the only place where juvenile tarakihi were observed (Tara Anderson, NIWA, pers. comm.).

Figure 3-32: "Boxfish" ROV footage of encrusting organisms, seaweed, and fish associated with the wreck of the Mikhail Lermontov, Port Gore (above, below; Source: https://www.boxfish.nz/tourism/ms-lermontov-shipwreck/). Sidescan sonar image of the wreck (insert, upper right; Source, NIWA).

Differences in biological responses from modular ARs, wrecks and natural reefs

One of the criticisms of ARs is that they don't function the same as natural reefs, despite ARs having demonstrated utility in supporting and increasing invertebrate biodiversity in degraded locations. A literature review and meta-analysis of 39 studies of fish communities associated with ARs/wrecks versus naturally occurring reference reefs (rocky reefs and coral reefs) revealed that, across reef ecosystems, ARs/wrecks support comparable levels of fish density, biomass, species richness, and diversity to natural reefs (Paxton et al. 2020b). However, nuances in fish communities were associated with the geographic setting (ocean basin, latitude zone) and artificial reef material.

In a study that compared AR/wreck effects on rock and ground fishes in temperate British Columbia, ARs did not consistently replicate the habitat of natural reefs and multiple variables were found important in the conservation success of an AR/wreck (Bulger 2019). To enhance rockfish abundance, Bulger (2019) recommended AR planning should consider depth, conservation status, rugosity, and reef age. Whereas, to maximize groundfish species richness conservation status, relief, reef size, and interactions between depth and reef age should be considered. In California, where 50 years of artisanal and recreational fishing have removed most of the largest and most vulnerable fish species, the highest species richness was found in the artificial reefs, with total of 83 species, of which 21 species were exclusive (Sánchez-Caballero et al. 2021). However, in that case contrary to the species richness, the total fish abundance was 20% higher at the natural reefs.

As seen with ARs, there is evidence that the fish communities associated with wrecks differ to those in adjacent reef habitat (Simon et al. 2013). Notably, when placed on a featureless seabed, shipwrecks function as key habitats, nurseries, and refugia for rare or absent species (Lengkeek et al. 2013). For example, in the Netherlands, juvenile and large Atlantic cod, linear skeleton shrimp, goldsinny wrasses and leopard spotted gobies were associated with shipwreck habitats. Furthermore, wrecks appear to change the trophic structure of fish communities and attract more large transient predators compared with similarly placed concrete ARs (Lemoine et al. 2019), although these changes differ with size and height of vessels (Paxton et al. 2020a).

Lemoine et al. (2019) recommended installing concrete modules to mimic rocky reefs or deploy ships to surpass natural reefs in enhancing fish abundance and biomass, but with different communities. In contrast, metal ships supported different fish communities than concrete modules and rocky reefs. Analyses revealed that these patterns reflect the 'footprint' and structural complexity of reef structures, and they concluded that managers should strategically deploy particular types of artificial reefs depending on defined objectives. Similarly, surveys of twenty-three reefs offshore of North Carolina, USA found that concrete modules performed similarly to rocky reefs, supporting similar fish abundance, biomass, and community composition (Lemoine et al. 2019). In contrast, metal ships supported different fish communities than concrete modules and rocky reefs. They recommended that concrete modules be deployed to mimic rocky reefs, whereas wrecks can create habitats that surpass natural reefs in fish abundance and biomass but with different communities.

Timeframes and succession

Timeframes of species colonisation and succession is also an important factor in biological responses to ARs/wrecks. It took more than five years to create a reef ecosystem analogous to natural reefs that provided adequate topographic complexity to attract fish, following deployment of AR modules in southeast India (Mathews et al. 2021). Hard corals started to flourish after ca.2 years, causing a decline in densities of other epibenthic categories. After five years of successional growth the modules supported a hard coral dominated assemblage (76.01/module) comprising 37 species belonging to 16 genera (from 2015 to 2020). Other categories that benefitted included molluscs, hydroids, sponges, ascidians, octocorals and echinoderms (40.44/module by 2020). Differences in fish communities between younger and older artificial reefs were also attributed to slow-paced structural changes over time in both biotic and abiotic aspects of wrecks affecting biomass density of most feeding guilds (Simon et al. 2013).

Influence of physical versus environmental factors

Structural (e.g., shape, height, vertical, material) and environmental factors (e.g., depth, exposure/shelter, sediment) can influence the community composition of reef fish associated with artificial reefs as compared with natural reefs, making them either similar or dissimilar. A study comparing assemblages colonising horizontal versus vertical surfaces and on natural and artificial surfaces reported that orientation may be of greater influence on the biological diversity of epibiota on subtidal reefs than whether reefs are natural or artificial (Knott et al. 2004). Sediment accumulation on low lying reef as compared with overhanging and vertical surfaces can influence encrusting species distributions (Handley et al. 2010; Handley and Page 2017). A study of a 1-year old scuttled shipwreck off the Eastern Australian coastline in 27 m of water described a rich assemblage of epifauna dominated by barnacles, sponges and bryozoans (Walker et al. 2007). Community structure varied significantly over small spatial scales of meters to tens of meters with depth, surface orientation and exposure being the major environmental drivers. Assemblages were substantially less diverse and abundant on the deepest part of the hull with residual antifouling paint, on sheltered surfaces inside the wreck, and on the sediment-laden horizontal surfaces.

Artificial reefs only appear to mimic adjacent natural reef communities if they possess structural features similar to those of the natural surroundings (Perkol-Finkel et al. 2006). Hydrodynamics and proximity to natural reefs with increasing wave exposure appears to make communities more similar (Pinto et al. 2021). In an exposed area, fouling community on a shipwreck was similar to the benthic community of the nearest reefs, with decreasing values of richness with distance. Sheltered area communities presented high dissimilarity. Structural complexity however does not always influence reef fish species abundance and composition, especially in areas of elevated human influence (Paxton et al. 2017). While structural complexity generally leads to increased species density, richness, and diversity, a study of 30 warm-temperate reefs in the southeastern US found that intermediate complexity maximizes fish abundance on natural and artificial reefs (Paxton et al. 2017). This challenges the current paradigm that fish abundance and other fish community metrics increase with increasing complexity. Artificial reefs of intermediate complexity maximised fish abundance, but human-made structures composed of low-lying concrete and metal ships differed in community types, with less complex, concrete structures supporting lower numbers of fishes. Metal ships protruding into the water column harboured higher numbers of fishes, including more pelagic species. Vessels protruded into the water column form pronounced peaks and valleys in their contours, characterized by greater variability than lower relief structures, such as concrete pipes. Deeper reefs supported more extra-large fishes whereas flat and complex natural reefs supported equivalent numbers of species. Complex artificial reefs (ships) supported more species than low complexity artificial reefs (concrete). Paxton et al. (2017) recommended that habitat-focused management efforts should include reefs representative of a wide-variety of structural complexities, including both the most topographically complex reefs and those that are low-lying pavements that offer ephemeral essential fish habitat on the continental shelf.

Comparisons of different materials placed on two historical shipwrecks (copper, brass, cast iron, carbon steel, pine, and oak) showed different responses with exposure (González-Duarte et al. 2018). At the exposed site, environmental conditions more strongly influenced biological succession than the material type, with pioneer colonisers dominating the communities in both sampling periods. At the more environmental stable site, the sessile community showed differences between sampling periods and among materials. Under more stable environmental conditions, material type showed a higher influence on the sessile community. Species that produce calcareous concretions developed on metallic panels, but were absent on wood panels, where the shipworm *Teredo navalis* was more abundant.

Unintended consequences of AR/wreck deployments

If the goal of AR deployments is to rehabilitate lost reef communities, the use of shipwrecks may not be fully appropriate. This is because shipwrecks can strongly change the trophic structure of fish communities and consequently energy flow from natural reefs (Simon et al. 2013) meaning they should not be considered surrogates for natural reefs (Sánchez-Caballero et al. 2021). Also, setting shipwrecks near natural reefs should be avoided as they differ in resources availability for many species, which may alter the community structure of natural habitats (Simon et al. 2013). Comparisons of fish assemblages over two metallic vessels, 5 and 105 years old, and two natural rocky reefs showed it was the substrate characteristics such as rugosity and benthic cover that influenced the trophic organisation of the communities.

One guild that is disproportionately imperilled by fishing are large predators that play important ecological roles. A comparative study of thirty artificial and natural reefs across North Carolina, USA, revealed large reef-associated predators were more dense on artificial than natural reefs (Paxton et al. 2020a). That pattern was associated with higher densities of transient predators (e.g., jacks, mackerel, barracuda, sharks) on artificial reefs, but not of resident predators (e.g., grouper, snapper). This effect was reported to relate to reef vertical extent, with shipwrecks hosting higher transient predator densities than concrete reefs. Taller artificial reefs had higher densities of transient predators, even when accounting for habitat area. A global literature review of high trophic level fishes on artificial and natural habitats suggests that the overall pattern of increased predator populations on artificial habitats is generalizable (Paxton et al. 2020a). In contrast, in California shipwrecks, protection from fishing provided refuge to commercially important fish species (such as Snappers, Triggerfish, Jacks, and Groupers) including threatened species, with differences in fish compositions on the natural and artificial reefs likely to be the outcome of differential fishing pressure (Sánchez-Caballero et al. 2021). The non-fished wrecks were considered a potential management strategy for reef restoration and enhancing fishing grounds (via spill-over).

3.5.3 Potential constraints of ARs and wrecks

Logistical and financial constraints can be a large factor in deciding whether to deploy purpose built ARs or sink a vessel (Lemoine et al. 2019). Common criticisms of AR planning include lacking assessment plans, objectives, and monitoring (Seaman 2000). Improperly designed and deployed ARs may displace valuable natural habitat or wrecks can facilitate pollution including oil spills or the establishment of invasive species (George et al. 2005; Sherman and Spieler 2006; Glasby et al. 2007; Bulger 2019). Without some analysis of the ecological performance of different types of AR/wrecks, there is a risk that deployments will fail to meet habitat restoration or supplementation goals (Bulger 2019; Lemoine et al. 2019).

Use in fisheries management/enhancement

The expectation for ARs to ‘produce’ more commercial fish biomass often ignores the role that hard habitat plays in a fish's life history and also that not all commercial fish species (for example many flatfish or pelagic species such as tuna) utilise rocky habitat (Jensen et al. 2000). There is also the issue of ‘shifting baselines’ (Pauly 1995) whereby gradual change to seafloor habitats occurs unrecorded over decades, and while the current political desire may be to maintain the seabed in a supposed 'pristine state', it ignores the ongoing physical impact that trawling, dredging and sedimentation has on benthic ecology. Artificial reefs should therefore not be considered a 'cure all' solution for fishermen and fisheries managers to protect habitats from trawling and dredging. For species such as lobsters that require refuge habitat, any increase in the numbers of individuals are likely to be proportional to the habitat's complexity and the availability of shelter and food (Jensen et al. 2000).

Public perceptions and values

Public perceptions of artificial reefs and their value can also differ. For example, engineering works to build coastal defence structures, harbours and dikes can be considered acceptable to environmental lobby groups and local government but the construction of artificial reefs which introduce hard habitat onto a previously impacted soft sediment habitats are often considered to be undesirable (Jensen et al. 2000).

Demonstrating different levels of acceptance between countries use of ARs and how coastal systems are modified, the percentage of coastal area modified by AR constructions in Europe were dwarfed by Japanese interventions that were estimated to have modified as much as 12% of its fishing grounds by 2000 (Stone et al. 1991; Simard 1995; Jensen et al. 2000). The context within which some of the Japanese developments have occurred is the ethos that because humans are part of the ecosystem, we can modify the environment to enhance societal and ecological outcomes. Japanese have defined this as “satoumi” or “marine and coastal landscapes that have been formed and maintained by prolonged interaction between humans and ecosystems”[34]. These “Japanese coastal socio-ecological production landscapes” (Uehara et al. 2020) are also integrally linked to the land-based equivalent of “satoyama”, meaning working together to realize societies in harmony with nature[35,36].

Potential hazards

With wrecks, there are obvious hazards from the release of bunker oils and other liquid contaminants from ships scuttled on purpose (e.g., Helton 2005). Before deployment, any vessel repurposed for sinking should include a risk assessment and cleaning management plan to remove any potentially harmful contaminants. This is because the release of metals such as lead, zinc, cadmium, nickel, and especially copper, can affect the fertilisation success of organisms such as corals (Reichelt-Brushett and Harrison 2005; Farrell 2021).

Persistent point source pollution has been measured following the accidental sinking of the MV Rena that struck the Astrolabe Reef (Otaiti) in 2011. Aqueous copper was found in the water immediately above the aft section of the wreck where the highest sedimentary load of copper was located (Hartland et al. 2019). That study found intermittent elevated concentrations of strong copper-

[34] https://ourworld.unu.edu/en/satoumi-the-link-between-humans-and-the-sea
[35] https://satoyama-initiative.org/about/
[36] E.g. https://youtu.be/yRlB6fwW98U

binding ligands 5-years later. Ligand binding strength was consistent with ligands actively produced by organisms in response to copper induced stress. At the contaminated site, species more robust to pollution like barnacles and sponges were the most abundant taxa present. While not all artificial reefs emit metals, more research needs to be conducted to determine the precise impact of different reef-building materials on the surrounding environment (Farrell 2021).

Exposure and wreck stability

Another important consideration in the scuttling of vessels for ARs is their stability in high energy wave locations. For example, the HMNZS Wellington that was scuttled as a dive attraction off Island Bay in November 2006, soon broke up after being hit by storms[37]. In Korea, modelling was used to predict the best depth for scuttling a ship at a proposed site (Kim et al. 2021). Their 2D and 3D numerical models coupled with a flume experiment were combined with the use of a 3D printed model of the proposed wreck to simulate wave pressure, scouring and movement of the structure.

[37] https://www.nzherald.co.nz/nz/sunken-frigate-further-broken-up/B3ADKL3BX52UCVJ7PMREEYVB3Y/

4 Discussion

To evaluate the best restoration options to restore and build resilience in the coastal marine areas within Te Tauihu, some questions are posed to guide decision making:

1. What are the key species, habitats and/or infrastructure that are currently degraded in Te Tauihu?
2. What species, habitats and/or infrastructure are threatened by climate change?
3. What species, habitats and/or infrastructure are not likely to be resilient or able to adapt to climate change stressors?
4. What species, habitats and/or infrastructure could be used to reduce climate change impacts?
5. What mechanisms are in place to foster and enable restoration in Te Tauihu?
6. What restoration options are 'shovel-ready' for implementation?
7. Is marine restoration economically viable?
8. What challenges might limit restoration success, and how can we overcome them?
9. How do we scale restoration so that it is effective?
10. "We don't know what we don't know", is this a cause for optimism?

In addressing these questions, the findings of each topic above (summarised in Tables 3-1 to 3-5) have also been considered.

4.1 What are the key species, habitats and/or infrastructure that are currently degraded in Te Tauihu?

In areal extent, the largest degraded habitats in Te Tauihu are shellfish beds (scallops, flat oysters, green-lipped mussels, e.g., Figure 4-1), that have been reduced to very low densities making them uneconomic to fish (e.g., flat oysters) or warranting fisheries closures (e.g., scallops). Horse mussels, important ecosystem engineers that historically formed extensive beds in Nelson Bays and in the Marlborough Sounds (Handley et al. 2019), have also undergone large scale decline, especially in Queen Charlotte Sound (Davidson et al. 2011). The major reasons for these declines include sedimentation, diminished water quality/clarity, overfishing and disturbance from contact fishing methods.

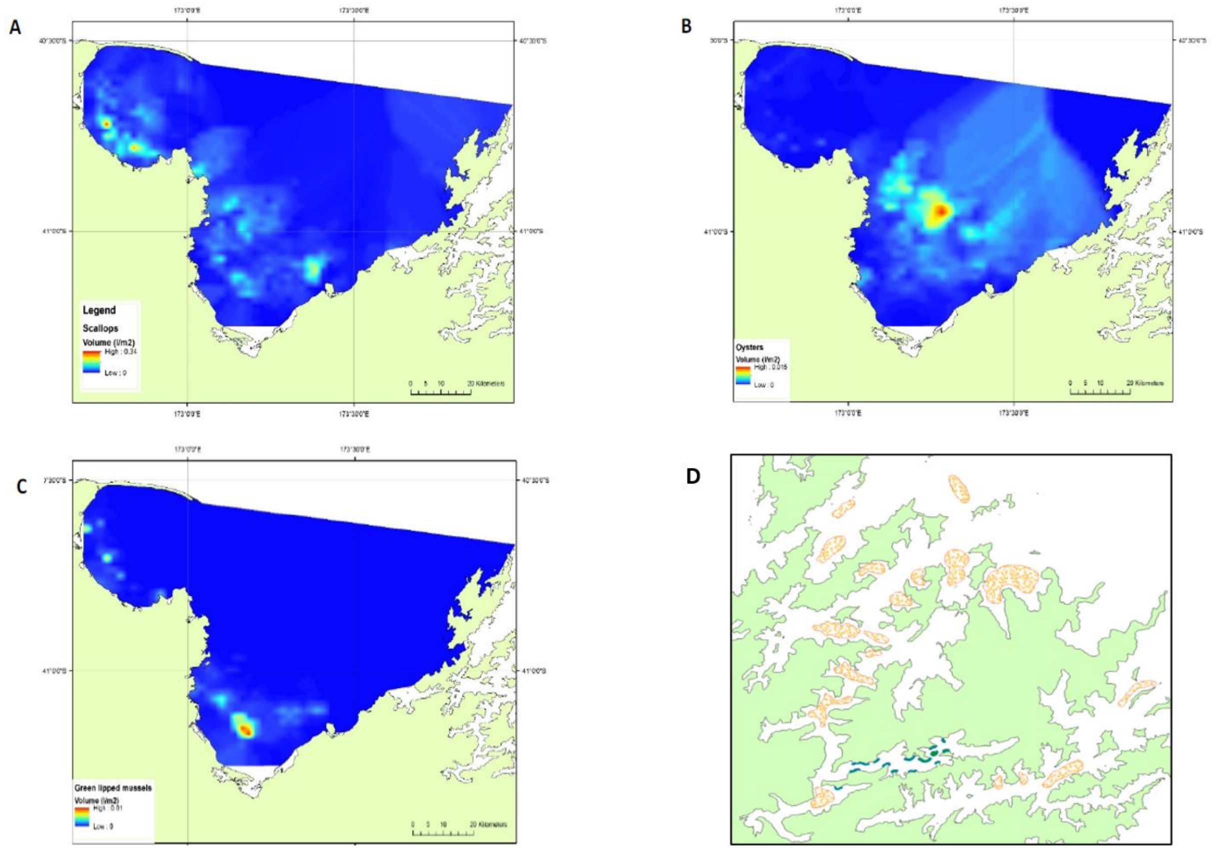


Figure 4-1: Recent and more historic distributions of scallops, flat oysters and green-lipped mussels. Krig interpolations of historic biomass estimates for Nelson Bays, collected between 1994 and 2012: A, scallops, B, flat oysters, and C, green-lipped mussels. Source: Handley et al. (2018). D, Historic distributions of scallop (orange polygons) and green-lipped mussel (green polygons) in the Marlborough Sounds. Source: Stead (1971b); Bull (unpub. map).

The ecosystem services formerly provided by these historic shellfish beds include filtration (of sediment, phytoplankton), habitat for other species (settlement surfaces, habitat complexity), food-chain (as food for predators), and contribution to biogeochemical processes (sediment-water column nutrient cycling, provision of carbonate to sediments).

At a similar spatial scale, seabed plants (macrophytes [e.g., red macroalgae] and microphytes [e.g., benthic diatoms]) have also likely declined, for the same reasons as given for shellfish. The concomitant loss of these two important habitats is likely to be reinforcing because of the feedback mechanisms between shellfish and benthic plants, as discussed earlier (also, see: Section 4-2 below).

Key message: Soft-sediment shellfish beds and plants are degraded across large areal extent in Te Tauihu

4.2 What species, habitats and/or infrastructure are threatened by climate change?

The Ministry for the Environment's National Climate Change Risk Assessment 2020 ranked two marine issues as the most urgent amongst A-NZ's 10 most significant climate change risks (MfE 2020). These are:

1. “Risks to coastal ecosystems, including the intertidal zone, estuaries, dunes, coastal lakes and wetlands, due to ongoing sea-level rise and extreme weather events”. *Top risk, with an urgency rating of “78”.*

2. “Risks to indigenous ecosystems and species from the enhanced spread, survival and establishment of invasive species due to climate change”. *Urgency rating of "73”.*

Therefore, sea-level rise, extreme weather events and threats from invasive species have been assessed as the most important stressors impacting high ranking marine systems under modelled climate change scenarios.

Sea-level rise and extreme weather are projected to impact shorelines prone to erosion, or highly modified steep catchments delivering high sediment loads under extreme weather events. Te Tauihu examples include the Pelorus Sound/Te Hoiere catchments comprising production forestry and farming (Handley et al. 2017a; Swales et al. 2021) and coastal areas with adjacent steep catchments with highly erodible soils like Kaiteretere, Mapua and the Abel Tasman Peninsula. Similarly, inundation from sea-level rise is projected to affect shorelines, increasing erosion, overtopping low lying areas of the coastline, especially in Nelson Bays. The predominant impact from coastal erosion and inundation will likely be an increase in sediment discharge from catchments and coastlines into marine areas. The sediment is likely to further reduce water quality and clarity degrading the mauri of coastal habitats, affecting especially primary producers, shellfish, and invertebrate communities with flow-on effects to fisheries/food production.

Key message: Primary producers (seabed plants, phytoplankton), shellfish, and invertebrate communities are expected to be impacted by climate change driven sea-level rise and extreme weather events

Catchment sediment erosion

Sediment is likely the most significant stressor in Te Tauihu, with sediment accumulation rates more than 10-fold higher today than pre-human baseline conditions (Handley et al. 2017a; Handley et al. 2020a; Swales et al. 2021). For example, “The Mahau Sound sediment accumulation rates are up to 90% higher (i.e., +1.8 mm yr^{-1}) than the recommended ANZECC[38] default guideline value of no more than 2 mm yr^{-1} above the natural annual sedimentation rates” (Swales et al. 2021). This sediment is highly likely to be impinging on and contributing to lack of recovery of benthic shellfish and algal (diatoms, macroalgae) habitats (see: Handley and Brown 2012; Handley et al. 2014; Michael et al. 2015; Tuck et al. 2017; Handley et al. 2020c). A recurring theme driving the more than ten-fold increase in the sediment accumulation rates over the past 100 years or so has been clear-felling and uniformity in land use along with soil disturbance from slips, road and track cutting, particularly on steep erosion prone country (Swales et al. 2021). Investigations into sediment sources (using isotope analysis) from different land-uses have identified that plantation forestry, farming, and subsoils from land disturbance (slips, tracks, roading) were significant sources of new sediment deposits in the Mahau Sound (Swales et al. 2021). Urlich (2020) recommended steep erosion-prone faces and gullies should be retired and protected from land-use practices that increase the erosion of topsoil and subsoil. To enable such land-protection, just-transition schemes (including carbon offsetting – see Section 4.7 below) could be co-created and implement to buy out forestry cutting rights or purchase of steep forest/farmland (Urlich and Handley 2020). With the use of soil source tracing studies in Te

[38] Australian and New Zealand Environment and Conservation Council (ANZECC) https://www.waterquality.gov.au/anz-guidelines/resources/previous-guidelines/anzecc-armcanz-2000

Tauihu (Gibbs and Woodward 2017; Handley et al. 2017a; Swales et al. 2021) that acquired 'soil source libraries', an alternative approach during restoration trials could be to capture and identify the sources of sediment incursions, to engender stakeholder ownership of the issue, reinforcing their social license to operate in a rapidly changing and unstable climate system.

Protecting soft sediment habitats from seafloor disturbance

As legacy sediments have been identified as contributing ~70% of sediment deposited in Pelorus Sound/Te Hoiere and Tasman/Golden Bays (Handley et al. 2017b; Swales et al. 2021) strategies are needed to protect and stabilise soft sediment habitats. One strategy, that could also help address climate change, is to co-create and implement just-transition schemes to enable commercial fishers to retire (e.g., Mahr 2011) or adopt non-contact fishing methods (Urlich and Handley 2020) (e.g., long-lining, trammel nets, mid-water trawls, precision harvesting[39]) to protect soft sediment habitats to rebuild resilience. Such measures could be combined with gifting of carbon credits to offset the reworking of soft sediments that may contribute loss of organic carbon with the disturbance of muddy habitats (Epstein et al. 2021). Given the scale of the benthic footprint of fishing (see: Tuck et al. 2017; Baird and Mules 2019) a strategy that protects soft sediment habitats could also help rebuild fish stocks that have been identified in this review to benefit significantly from restoration of habitats including shellfish, macroalgae and seagrass beds. If fishers were to transition to non-contact methods, that can selectively harvest fish of a higher quality (e.g., Jones 2020), then the reversal of fishers having to travel longer distances to catch a diminishing catch (Handley et al. 2019) could be achieved, softening the just-transition and leading to long-term economic and ecological gains. For harvest of fish species associated with the benthos (e.g., yellow belly flounder, gurnard, red cod), as an alternative to trammel nets that may create issues with bycatch and netting bans, fishing zoning or corridors could be developed through spatial planning to create areas acceptable or desirable to target soft sediment associated species (e.g., fishing and customary zones in the Great Barrier Reef, Australia; Day et al. 2019).

Key message: Restoring and protecting soft sediment habitats could rebuild resilience and increase fisheries yields over the long-term

4.3 What species, habitats and/or infrastructure are not likely to be resilient or able to adapt to climate change stressors?

Resilience and adaptability of key species/habitats is expected to depend on (i) the risks of future climate change to the species/environment, and (ii) a species might be more resilient to climate change if the habitat/environment it lives in is otherwise healthy. For example, on the first point, it will likely be risky to attempt restoration of species/habitats like the large brown macroalgae *Macrocystis* that is growing at the northern most range of its temperature tolerance in Te Tauihu because further warming is projected in coastal waters around A-NZ. As an example of species with compromised resilience, trying to restore rare rhodolith beds that are being shaded by high turbidity and buried by high rates of sediment deposition might be futile until sediment discharge from land is reduced or causes of resuspension of fine sediments have been reduced. Prioritising the restoration of rare species/habitats may therefore be risky until the general health of the ecosystem has been made more resilient by managing key stressors, but also recognising that this is expected to become more challenging with ensuing climate change (see 4.1 above). A risk assessment should therefore be part of early restoration planning process. Because high ranking climate change stressors (sea-level

[39] https://www.pmcsa.ac.nz/2021/02/21/precision-seafood-harvesting-tiaki/

rise, extreme weather, invasions) are likely to reduce the resilience of species there is a necessity for urgent concomitant action to reduce existing stress like sediment entering waterways (e.g., Handley et al. 2017a; Handley et al. 2020c; Swales et al. 2021) and its subsequent disturbance and resuspension (e.g., via trawling, dredging).

Key message: Risks associated with climate change are projected to be species/habitat dependent, and the general health of the environment is expected to affect their resilience to climate change

4.4 What species, habitats and/or infrastructure could be used to reduce climate change impacts?

Protecting existing blue carbon ecosystems (including mangrove forests, seagrass meadows, tidal marshes) will help to reduce impacts from climate change. At a global scale, it has been estimated that such protection could avoid emissions of 304 (141–466 ±95% CI) teragrams of CO_2 equivalent ($TgCO_2e$) per year (Macreadie et al. 2021). Large scale restoration of blue carbon ecosystems has an estimated removal of an extra 841 (621–1,064±95% CI) Tg CO_2e per year by 2030 (Macreadie et al. 2021). Together these estimates equate to ca.3% (0.5–0.8% from protection and 2.3–2.5% from restoration) of annual global greenhouse gas emissions, worth ca. USD14.54 billion dollars[40].

Therefore, any carbon accounting benefits would be additional to the economic and societal gains already estimated from marine restoration projects. A recent analysis of blue carbon in Hobson Bay, Victoria, Australia, valued the asset value (carbon storage and coastal protection) of their 108 ha of seagrasses, 251 ha of salt marshes and 2 ha of mangroves at $11.8M, $1.9M and $31.8M AUD respectively (Costa et al. 2021b). Further benefits from ecosystem services, including ongoing carbon sequestration, commercial and recreational fisheries and birdwatching, are valued at $\$750\ y^{-1}$, $\$136.240\ y^{-1}$ and $\$42,250\ y^{-1}$ AUD, respectively.

In Te Tauihu, blue carbon accounting and credit schemes could be used to analyse cost-benefits for spatial management scenarios using an ecosystem approach to fisheries management. Such schemes to improve the cover and health of blue carbon contributors (see Sections 3.1, 3.2, 3.3) would also improve coastal habitat and ecosystem resilience.

Key message: Restoration of wetlands, tidal marshes and seagrass beds offer opportunities to build resilience and reduce effects of climate change

4.5 What mechanisms are in place to foster and enable restoration in Te Tauihu?

Seddon et al. (2021) recommend that policymakers, practitioners, and researchers consider the synergies and trade-offs associated with Nature Based Solutions (NbS), and to follow four guiding principles to enable NbS to provide sustainable benefits to society:

1. "NbS are not a substitute for the rapid phase out of fossil fuels;
2. NbS involve a wide range of ecosystems on land and in the sea, not just forests;

[40] Nature based carbon credits currently worth ca. 12.7M/mt CO_2e: https://www.spglobal.com/platts/en/market-insights/latest-news/energy-transition/111121-cop26-voluntary-carbon-market-value-tops-1-bil-in-2021-ecosystem-marketplace

3. NbS are implemented with the full engagement and consent of Indigenous Peoples and local communities in a way that respects their cultural and ecological rights; and

4. NbS should be explicitly designed to provide measurable benefits for biodiversity."

These guiding principles, especially (3) in particular, are aligned with the spirit of working with iwi/hapū/tangata whenua in Te Tauihu with the Intergenerational Wellbeing Framework incorporating Te Taiao (The Natural World), and the desire to leave nature 'better off' for future generations requiring "*wide-scale change in behaviours and practices across society to reduce our environmental footprint*" (Wakatū 2020). Both the Wakatū strategy and the Kotahitanga Mō te Taiao Alliance strategy include visions of sustainable use of our natural resources, while reversing and restoring degraded natural heritage (KMTT 2020).

A recent restoration project in Australia demonstrated that social participation in the decision-making process can create public stewardship that led to an innovative solution to revive extinct oyster habitat at a coastal scale (Australia's first large-scale reef restoration) (McAfee et al. 2021b). They found that engaging community stakeholders in the decision-making process allowed social knowledge, prior scepticism and concerns to inform the restoration planning. This reduced the risk of community or political backlash to environmental decisions and built support for innovative solutions. They recommended three key components:

1. "a collaborative decision-making process through regular engagement with coastal residents and workers,

2. encouragement of collaborative innovation of solutions (co-designed) to benefit economic and social activities, and

3. engaging local communities on the region's forgotten socio-ecological history to reveal the opportunity to recover their natural history."

The latter to some extent in Nelson/Marlborough has been achieved through historical ecological reviews and Māori led strategies (e.g., Handley 2006; Michael et al. 2015; Handley 2016; KMTT 2020; Wakatū 2020). McAfee et al. (2021b) concluded that "ecological restoration at coastal-scales is a viable policy solution for the environment and society; illustrating how valuing the socio-ecological context with social engagement can rally support for building positive environmental legacies."

Local iwi strategies call for adoption of Māori values including Manākitanga, Kaitiakitanga, Matauranga Māori, Kotahitanga, Rangitiratanga, Mauri and Arohatia (KMTT 2020). To maximise restoration success through incorporation of Matauranga Māori, Councils and iwi should work together in partnership, to develop draft plans to consult with the wider community (e.g., Stevens et al, Waikawa draft report; Clark and Berthelsen 2021). Local whānau/hapu/iwi may have knowledge and history passed down through generations on the extent of losses of key species or habitats, as well as views as to the stressors to control before any restoration is attempted (Clark and Berthelsen 2021).

Key message: Partner with tangata whenua/iwi then involve local communities and stakeholders from the outset

4.6 What restoration options are 'shovel-ready' for implementation?

Restoration options 'shovel-ready' for implementation include salt marshes, seagrass, cockles/tuangi, green-lipped mussels, softening estuarine edges/infrastructure, and ARs (see tables at start of each section). Encouragingly, the success of marine restoration initiatives is continually growing. Of the total 498 publications reviewed for the MERCES European review, 50-70% of the studies were successful, whilst failure was linked to methodological aspects and overlooking important site characteristics and local threats (Papadopoulou et al. 2017).

Restoring salt marshes and seagrass beds to build resilience against erosion

Although salt marsh and seagrass restoration appear 'shovel-ready' for use at locations where key stressors have been reduced, estuarine margins and coastal areas are degraded through modifications, estuarine 'squeeze', and receive high loads of fine sediment (e.g., Waimeha Inlet, Moutere Inlet, Havelock Estuary, Mahau Sound, Head of Kenepuru Sound). At degraded sites, risk assessments should be carried out to identify key stressors and their source, and where possible addressed or reduced. At sites that are deemed by species experts to be marginal for survival, temporary (biodegradable mimics/structures, shell gabions) or permanent structures (ARs) can be used to protect plantings from resuspension, creating windows of opportunity to achieve effective restoration at scale. Strategies to test survival and restoration potential should include the use of small-scale plots deployed across historic species distributions (e.g., seagrass and green-lipped mussel; case studies: 6 & 7)

Onshore, in areas that have been affected by coastal erosion and slips, shoreline edges should be softened with the use of terraces and naturalised setbacks (e.g., case studies: 4 & 5) or with the development of buffer regions between land-stressors and the subtidal could be planted out with pioneering salt marsh species (e.g., case studies 1 & 2). Infrastructure projects or ARs, where possible should be constructed using carbon friendly materials and additives to create and enhance habitat that has dual purpose (e.g., case study 3), extending the life of the structure, and to create a diverse range of habitats at different scales to enable different species with variable niche or ecosystem roles to take up residence. If possible, seeding with shellfish can speed up colonisation and increase biodiversity, reduce turbidity, and their waste feed plants.

Key message: Salt marsh and seagrass restoration can increase resilience of estuaries and help sequester carbon

Restoring shellfish beds to address legacy sediment and resuspension

A strategy to increase resilience and address legacy sediments/sediment discharge from land arising from climate change/sea-level rise is based on the ability of shellfish to filter and sequester sediment (Green, Malcolm O et al. 1998; Kent et al. 2017); Handley, pers. observ.), sequester carbon (Lee, Hannah ZL et al. 2020) and armour sediments from resuspension (Ysebaert et al. 2009). With the increase in sediment accumulating in Te Tauihu, there has been a loss of filtration capacity from our waterways (see death assemblage studies: Handley et al. 2017; Handley et al. 2020c; Swales et al. 2021), especially with the dramatic decline and then closure of shellfish fisheries over the last two decades. The successful trials of green-lipped mussel restoration (>85% survival) on soft sediments in Pelorus Sound/Te Hoiere and the highly turbid Kenepuru Sound (Benjamin, in prep., Handley pers. observ.) demonstrate that green-lipped mussels can survive in areas where they were historically abundant but have not recovered without intervention.

Green-lipped mussels also stabilise sediments with their habit of forming a byssus-bound crust on top of soft sediment when deployed at high density (Case study 7; Clarke 2014; Figure 3-15). The greater abundance of red macroalgae at one location at Skiddaw as compared with control plots indicates that seabed plants may also benefit. The inconclusive results to date on testing the use of laying waste mussel shell to enhance mussel survival (Benjamin, UoA, unpub. data), at minimum, demonstrates that the shell does not hinder survival. The colonisation of waste scallop shell in Tasman Bay by filter-feeders including shellfish (Brown 2011; Brown et al. 2014), and the increase in biodiversity beneath mussel farms over soft-sediment habitats (Hartstein and Rowden 2004; Keeley et al. 2009; Stenton-Dozey and Broekhuizen 2019), further demonstrate the benefits of returning shell to soft sediment habitats. Currently, waste shell that goes to landfill could be deployed to buffer projected ocean acidification and provide rugosity to soft sediments vulnerable to erosion and resuspension. Shell could also be used to create ‘windows of opportunity’ (Fivash et al. 2021) to enable shellfish deployments or seagrass plantings to become established[41] or used as a tool in subtidal seagrass seeding (Lee and Park 2008).

Key message: Mussel restoration and waste shell return can stabilise soft sediments, increasing biodiversity and clear sediment from the water column

4.7 Is marine restoration economically viable?

In the compilation of this review, it became evident that there were parallels among the topics reporting potential economic returns that, if realised and reported on, could foster and enable more interest and action on restoration. For example, as a narrow example, focussing solely on the potential benefits to fisheries (ignoring unaccounted ecosystem services and carbon accounting), highlights include:

- Salt marshes: provision of ca. 90% of whitebait habitat (A-NZ)
- Seagrass: 159 juvenile snapper per 100 m^2 and weighing 1.45-1.87 times heavier
- Shellfish: 10-30 fold increase in snapper (associated with horse mussels, in A-NZ), USD$5,500 and USD $99,000/ha (*Crassostrea virginica,* in USA), 16.4 times more harvestable fish (shellfish reef, Queensland, Australia)
- ARs: 35% increase in two-banded seabream (Portugal), 4.6 times more fish biomass (Caribbean), increases in sparid fish (Botany Bay, Australia), increase in juveniles and reproductively mature fish (Brazil)
- Wrecks: more large transient predators were associated with wrecks (USA), nursery and key features increase diversity and density of fish (North Sea)

A global cost–benefit analysis to determine the net monetary value of ecosystem service benefits of restoring coral reef, mangrove, salt marsh, and seagrass ecosystems, reported that benefits outweighed costs (i.e., there were positive net benefits) (Stewart-Sinclair et al. 2021). Mean benefit-

[41] See also Handley, S., Brown, S. (2012) Feasibility of restoring Tasman Bay mussel beds.: 31. https://tasmanbayguardians.org.nz/wp-content/uploads/2018/11/Handley-and-Brown-2012-ELF12243-Feasibility-of-restoring-Tasman-Bay-mussel_FINAL-2.pdf
, Handley, S. (2016a) The history of benthic change in Pelorus Sound (Te Hoiere), Marlborough. *NIWA Client report, prepared for Marlborough District Council, NEL2015-018*: 66.
, Handley, S. (2017) Advice for mussel restoration trials in Pelorus Sound/Te Hoiere, Marlborough: 15. https://www.envirolink.govt.nz/assets/Envirolink/1713-MLDC120-Advice-for-mussel-restoration-trials-in-Pelors-Sound-Te-Hoiere-Marlborough.pdf for justifications for returning waste shell to soft sediment habitats.

to-cost ratios for ecosystem restoration were 8 to 10 times higher than prior studies of coral reef and seagrass restoration. Salt marsh restoration had the greatest net benefits, followed by mangroves; coral reef and seagrass ecosystems had the lowest net benefits. The review by Stewart-Sinclair et al. (2021) challenges the perception that blue restoration is expensive with low economic viability and encourages further targeted investment in marine restoration.

The inclusion of carbon accounting and carbon credit schemes[42] may also hold great value. There are however knowledge gaps of estimates on carbon storage and greenhouse gas fluxes following restoration that likely hinder the inclusion of blue carbon ecosystems into carbon accounting and crediting schemes. There is also large variability in carbon storage among and within blue carbon habitats at local scales (Thomas 2014; Green, Alix et al. 2018; Lewis et al. 2018) that hampers obtaining robust estimates of carbon storage at national to global scales (Duarte et al. 2013; Serrano et al. 2019b; Costa et al. 2021a).

Successful restoration of ecosystems and fisheries have been shown to provide both direct and indirect benefits: a reversal of the "shifting baselines syndrome" and a motivation to manage fisheries sustainably, diversification of local economies and fisheries, community building, an increased sense of local pride, a demographic broadening of the conservation community, and enhanced ecosystem services and recreational opportunities (McClenachan et al. 2015). Like the positive feedback mechanisms between oysters and seagrass beds, restoration of ecosystems also provides positive feedback between economic benefits and other social benefits, with local boosts in revenue enhancing community engagement and providing motivation for further restoration (McClenachan et al. 2015).

Key message: Marine restoration, habitat creation or habitat enhancement provides demonstrable economic, societal, ecological benefits, that once initiated can build further support

Recent discussions about the use of biodiversity offsetting and compensation during the design and implementation of marine infrastructure projects. Biodiversity offsetting seeks to balance the environmental impacts from development through the generation of measurable gains in biodiversity that compensate for loss. Biodiversity offsetting aims to create measurable conservation actions to compensate for significant residual adverse biodiversity impacts arising during a development (e.g., Alestra and Bell 2021; Yu et al. 2022). Biodiversity offsetting should however be approached with caution because the assumptions underpinning the estimation of offsets required to achieve “No Net Loss” or a “Net Gain” in biodiversity, must be at least equivalent or greater to the biodiversity losses from development, which is seldom the case (O’Brien 2020). Offsetting does not therefore appear to deal with redressing historic losses or designing infrastructure/restoration to enhance ecosystem resilience by incorporating species redundancy or multifunctionality (*sensu* Selkoe et al. 2015; Handley et al. 2020b).

[42] Note: The Nature Conservancy in A-NZ are currently working on feasibility assessment for blue carbon credits (Erik van Eyndhoven pers. comm. TNC.)

4.8 What challenges might limit success, and how can we overcome them?

Site selection

Failure in restoration, that is believed to be under-reported (Hobbs 2009; Knight 2009; Suding 2011), is most likely related to inadequate site selection, stochastic events, or human disturbance (Bayraktarov et al. 2016; Papadopoulou et al. 2017). To enable suitable site selection, existing data can be used to identify key species habitat requirements. Examples of existing data, more likely available in the Marlborough Sounds can include historical local knowledge of species and habitat distributions (e.g., Handley, S. 2016b; Handley, S. 2016a; Urlich and Handley 2020; Handley et al. 2019b; Handley et al. 2019), and more recent declines identified through MDC's Ecologically Significance Marine Sites (ESMS) program, and shellfish biomass surveys. Habitat suitability modelling approaches could then be used to improve site selection success for key species restoration trials (e.g., Anderson et al. 2020; Ribó et al. 2021; Smith et al. 2022). As an example of using existing data, the ESMS database could also be used to develop a GIS based threats risk assessment tool that utilises adjoining Council held land-use-overlays coupled with knowledge of stressors/threats[43] to help as a decision support system for managing threats to ESMS as well as identifying potential suitable locations for marine restoration initiatives.

> ***Key message: Use existing data where possible to evaluate potential of restoration sites***

Managing threats/stressors

Success of restoration activities requires us to first rehabilitate habitats by ceasing or mitigating key stressors that have caused species or system decline and prevented their natural recovery. "This can be perceived as preventing harmful activities through regulatory management (from controlling/banning specific activities to creating Marine Protected Areas) or removing/adding barriers in an intervention to protect an ecosystem from further harm" (Fivash et al. 2021).

Signals from central government support such action. For example: proposed changes to the NPS-IB recognise that "*The maintenance of indigenous biodiversity may also require the restoration or enhancement of ecosystems and habitats*" (Urlich 2021). There was also advice from the Office of the Prime Minister's Chief Science Advisor for changing fishing practices to protect seabed habitats that support and enhance fisheries (Gerrard 2021), and there were recent acknowledgements from the Minister for the Environment:

> *"We've lost 95 per cent of our scallops. We can't even reseed them now in the Marlborough Sounds because of the amount of sediment.*
>
> *"Some of that sediment came down 100 years ago, following colonisation, when there was so much forest clearance. But something has tipped things in recent decades with more clearance of land and increasing intensity of land use practices, deforestation, subdivisions, everything adding to it."*[44]

[43] E.g. NIWA is preferred provider to MPI to undertake a blue cod habitat risk assessment: BEN2021-05 that includes classifying habitat stressors in the Marlborough Sounds

[44] https://www.stuff.co.nz/environment/300422097/this-is-how-it-ends-how-producing-milk-turned-a-lake-bright-orange

Key message: Rehabilitate habitats by ceasing or reducing key stressors that have caused species or system decline and prevented their natural recovery

It is important to note that systems do not necessarily recover to their former state, and that the degree of recovery from its original state equates to levels of resilience (Elliott et al. 2007). For example, natural system changes due to climate variation, where temperature regime shifts result in warmer water species migrating into some areas, whereas colder water species migrate out of the same areas, or where invasive species displace native species with similar functional traits (Morrison 2021).

Key message: Rehabilitation may not fully return systems to historical conditions

Overcoming tipping points (hysteresis)

A significant challenge in restoration is overcoming a system that has been through a 'tipping point' into an alternate stable state, where innate recovery of desired species/habitat does not occur after the key stressor has been removed. In such cases, a strategy is required to overcome hysteresis or the lag preventing the system tipping back into its prior stable state. Relevant examples of hysteresis include: urchin barrens where control of urchins on barrens is difficult given positive density-dependent feedbacks that act to stabilize urchin populations (Ling et al. 2019) and seagrass restoration failing due to high suspended sediment shading out plantings (Zabarte-Maeztu 2021). In such instances, for large-scale restoration to be successful, the system state must be moved past threshold conditions that are critical. For example, restoring light availability in seagrass beds and algal systems (Diefenderfer et al. 2021). The use of temporary or biodegradable elements can be used to overcome establishment thresholds, but these may be costly and difficult to deploy at meaningful scales.

Key message: Restoration may not work if the habitat is no-longer suitable

Key message: Overcoming tipping points may involve lags (hysteresis) that may require interventions to reduce 'establishment thresholds' or providing 'windows of opportunity'

Encouragingly, overseas restoration efforts indicate that once restoration is initiated (e.g., shellfish restoration), benefits flow to other components (like seagrass and benthic microalgae), which in-turn reinforce and enhance broader restoration goals including stabilisation of soft sediments that help maintain water clarity (Kemp et al. 2005; Greening et al. 2014).

Key message: Develop an understanding of the system, and the multiple interacting factors within it that can affect your restoration target/goal

Key message: Plan and be prepared. Think through the biology, the ecology, the environment, and map out pathways, "what ifs"

Although most restoration remains primarily a single-species exercise where positive interactions are seldom incorporated into planning, accelerated habitat recovery can be achieved by restoring multiple species in combination (McAfee et al. 2021a). For example, experiments tested whether restoration of canopy-forming kelp (*Ecklonia radiata*) could accelerate the natural recruitment of oysters (*Ostrea angasi*) to rocky habitat monopolised by turf-forming algae. Results showed that turf algae inhibited oyster recruitment to the exposed surfaces of the reef, limiting their capacity to grow and form complex, three-dimensional habitat. Whereas, transplanted kelp (live kelp and synthetic mimics) reduced the biomass of turf and enhanced oyster recruitment, allowing 26 times more oyster recruitment (McAfee et al. 2021a).

Key message: Multi-species restoration appears to increase likelihood of success

Biosecurity risks

Biosecurity also poses a significant risk from restoration as identified by MfE's climate change risk assessment (2020). In a global analysis, hard coastal infrastructure has replaced more than half (52.9 ± 4.9%) of the coastline associated with 30 overseas urban centres, with a model forecasting a further 50–76% expansion of coastal infrastructure over a 25-year period that included effects of sea-level rise (Floerl et al. 2021). While hard structures can drive dramatic changes in functional profiles, productivity and ecosystem service provision of ecological communities (Scherner et al. 2013; Heery et al. 2017), they also provide habitat for invasive NIS organisms (Glasby et al. 2007; Airoldi et al. 2015). Floerl's et al. (2021) model for A-NZ, show the greatest absolute increases in coastal infrastructure is projected to be for breakwalls of 31–48% with highest relative increases projected for shipping wharves (125–191%) and jetties (119–197%). The design of such infrastructure should not only consider softening edges (terraces, natural plantings, offshore buffers using modular ARs, e.g., case studies: 2 to 5) to protect against large waves as extreme weather events intensify (e.g., Figure 3-9), but also utilise native species or enhance habitats to enable their coexistence.

Another biosecurity risk, previously discussed, that hampered mussel restoration in the Hauraki Gulf came from MPI raising concerns about the transport of unwanted NIS organisms around the Gulf with the adult mussels harvested from aquaculture farms for transplanting. MPI required suitable but prohibitively expensive cleaning treatment of these mussels before allowing their movement (Morrison 2021). "The counter-argument was made that the NIS organisms discussed were widespread and well-established in the Gulf already, and present on both the donor farms and the transplant sites (e.g., Mediterranean fan worm). This is true but does not account for the possibility that a significant new NIS organism could be introduced in the gulf at any time, and then be advantaged in its spread by mussel restoration transfers" (Morrison 2021). As was the case with mussel deployments in Pelorus Sound/Te Hoiere, the waterbodies exchanged or the distance mussels are transported from collection and re-laying sites should be minimised, with every effort made to remove conspicuous NIS during harvest.

Key message: Carry out a biosecurity risk assessment early-on and plan mitigation strategies

Key message: Successes may be site specific, rather than a one-size-fits-all

4.9 How do we scale restoration so that it is effective?

Technological solutions

Due to the large scales at which extreme weather events can occur, another challenge is implementing the potential very large-scale restoration response required. The scale of response might cover multiple ecosystems necessitating concerted management efforts (e.g., response to the Gulf of Mexico Deep Horizon oil spill, Rena oil spill in the Bay of Plenty) posing technological and innovation challenges as compared with terrestrial analogues, because of the working environment, particularly in deeper or offshore areas (Papadopoulou et al. 2017). Papadopoulou et al. (2017) report that:

> "*New technologies are becoming available or adaptable, with access to underwater vehicles or new materials for underwater work. Mechanical planters are already available for very shallow work. However large area coverage is still a major issue, as transplanting on hard bottoms is still labour intensive as for example corals need careful placement and orientation. In shallow, more accessible waters, volunteer engagement through citizen-science initiatives may be a significant way towards up-scaling restoration over wider areas, either from collection of fisheries bycatch (e.g., coral fragments for on-growing), volunteering equipment (small vessels), space (for nursery grounds), or time (for labour at any stage in the process). Also, the use of social media can enhance any kind of campaign organisation reaching wider distributions than has previously been possible*."

At large scales, because outcomes are more complex and synergistic than the additive impacts of individual restoration projects, the challenge is to identify the spatial extent of key stressors. Encouragingly the study of cumulative effects can help. Because cumulative effects, usually measured for ecosystem degradation, are also measurable during ecosystem restoration, collaborative understanding and management of cumulative effects are essential for the success of restoration at large scales (see Diefenderfer et al. 2021).

> ***Key message: When faced with large scale degradation, understanding and management of cumulative effects is essential***

At small scale, the use of shellfish gardening approaches (e.g., see Section 3.4.7) could be used to obtain adequate numbers of adult shellfish that are more likely to survive than juveniles when released. To restore benthic filtration, large-scale mussel restoration could be enabled and funded using a 'not-for-profit' entity created to grow and deploy mussels using temporary marine farm structures on-site where needed in Te Tauihu. A 'for-profit' model for this currently operates for spat catching by the Golden Bay and Tasman Bay Ring Road Spat Catching Permit Holders (e.g., FNZ 2018). Green lipped mussels could be cultured on-site, with a portion harvested to cover operational costs, and the remainder of the crop deployed to the seabed once any non-indigenous-species (NIS) organisms have been removed. The growing structures could then be moved to a new location or rotated across areas as necessary until a desired density of mussels and filtration have been achieved. Control of 11-arm starfish would be recommended to reduce mortality.

> ***Key message: To scale up restoration, explore emerging/existing planting technologies, and use existing marine industries to advantage***

4.10 "We don't know what we don't know", cause for optimism?

The results of the small scale subtidal GLM deployments in Pelorus Sound/Te Hoiere (Benjamin, et al. *submitted*) provided surprising results that challenged preconceived perceptions about where mussels would survive. The site where we thought mussels would survive best, which had the clearest water conditions for diving and dominated by rocky substratum, was the first site where the mussels were completely consumed by 11-arm starfish with 100% mortality. Ironically, the site with the lowest water clarity, least current flow, and muddiest seabed conditions, had the highest survivorship and mussel growth rates after 2 years. Because these results challenged our preconceived knowledge of how mussels would respond to restoration within their historic range[45], we have cause for optimism to try restoration of mussels and other species across their historic spatial extents. It also illustrates the value in using small scale plots to test and learn about the drivers of restoration success (Benjamin, et al. *submitted*; Gann et al. 2019; Fitzsimons et al. 2020). In A-NZ marine restoration, "learning-by-doing" is likely be an inevitable necessity until more local knowledge and capacity of subject experts are enabled.

The successes of salt marsh restoration in Waikawa and Maketū and the promising results of ongoing GLM restoration trials in the inner Pelorus Sound/Te Hoiere (Benjamin, in prep., Toone, in prep.) provide encouraging impetus to continue broadening the scope and scale of marine restoration in Te Tauihu.

Key message: There will be an element of "learning-by-doing", but early successes in Te Tauihu gives cause for optimism

[45] https://youtu.be/eNncmnKvZm8

5 Acknowledgements

Thanks to: Oliver Wade, Vicki Ambrose and Trevor James for supporting the Envirolink funding application; Steve Urlich (Lincoln University) for initial discussions on topics and providing lots of valuable reference material he had collected over the preceding years; Dana Clark, Robyn Dunmore, Oliver Fluer (Cawthron) for articles and photos; Emilee Benjamin, Brad Skelton (UoA) for shellfish advise; Fleur Matheson, Vonda Cummings, Carolyn Lundquist, Michael Allis (NIWA) for literature and advice; and Oliver Wade, Trevor James, Vonda Cummings and Darren King for review and comments on prior drafts.

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