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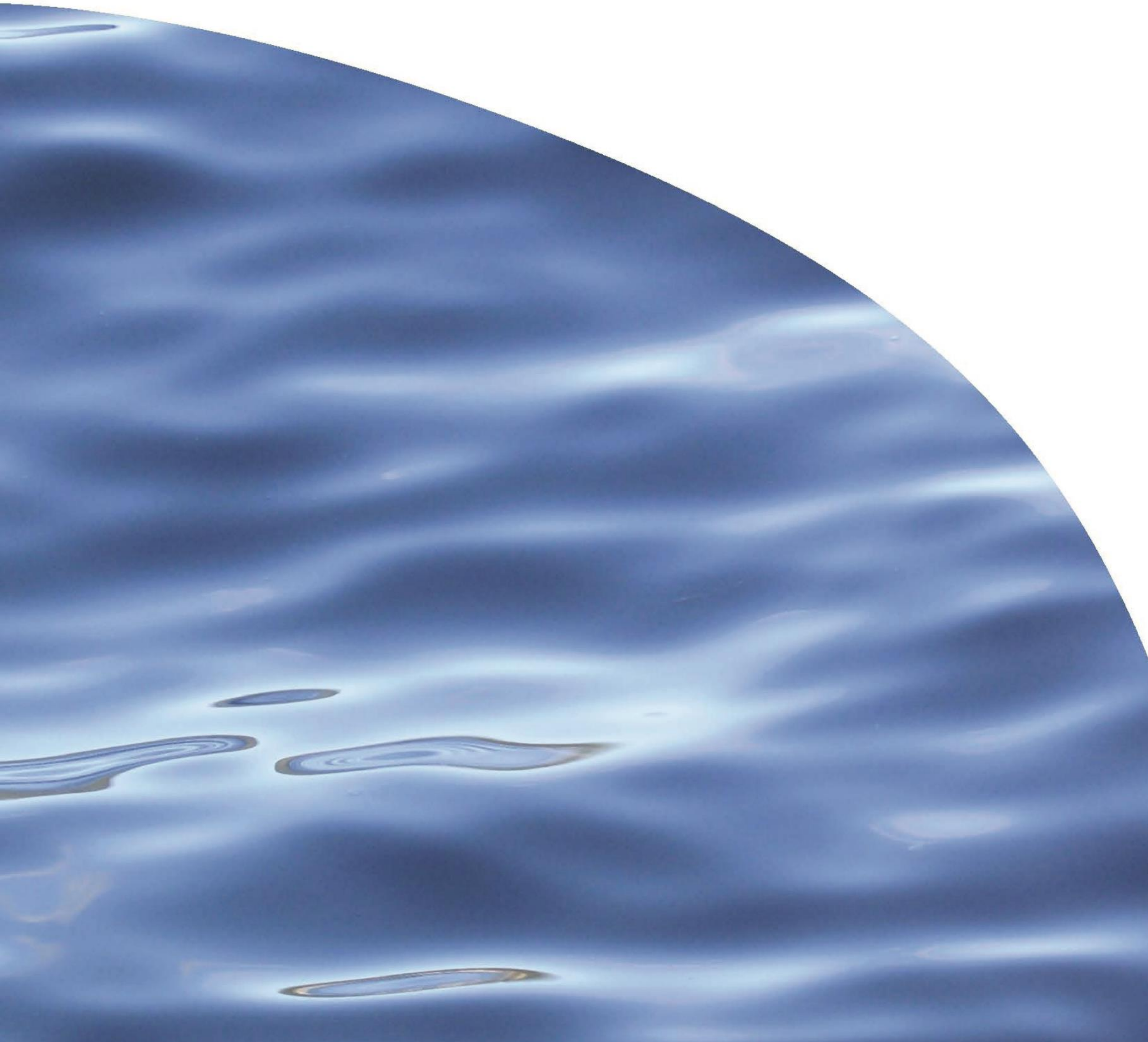
REVIEW OF THE POTENTIAL FOR LOW IMPACT SEAGRASS RESTORATION IN AOTEAROA NEW ZEALAND



REVIEW OF THE POTENTIAL FOR LOW IMPACT SEAGRASS RESTORATION IN AOTEAROA NEW ZEALAND

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EXECUTIVE SUMMARY

Seagrass meadows (*Zostera muelleri*) are one of Aotearoa New Zealand's most important coastal ecosystems. They provide crucial habitat for fish and other animals, mitigate climate change through carbon sequestration and improve coastal water quality by taking up nutrients and contaminants, and trapping sediments. Unfortunately, Aotearoa New Zealand, like many places worldwide, has experienced extensive seagrass decline, primarily due to physical disturbance and deteriorating water quality. Concern about the loss of seagrass meadows has prompted global efforts to facilitate natural recovery of, and actively restore, seagrass. In Aotearoa New Zealand, active seagrass restoration has been trialled by transplanting seagrass from wild donor meadows. However, removing plants from donor sites causes damage to wild seagrass meadows. Lower impact methods of seagrass restoration are available but have yet to be used in Aotearoa New Zealand.

The purpose of this report was to examine the potential for low impact forms of seagrass restoration in Aotearoa New Zealand, with a focus on the Whakatū/Nelson region. Seagrass restoration methods used internationally were reviewed. These included the facilitation of natural recovery through the reduction or removal of stressors (passive restoration) and active restoration using seagrass seeds or plants. Seagrass plants can be obtained by harvesting them from wild seagrass meadows, collecting naturally uprooted seagrass fragments, germinating seeds or propagating plants in a nursery. Plants are then transplanted as intact sods of seagrass or bare-rooted plants, with various methods available to fix them to the seafloor and/or shelter them from environmental conditions and animals.

We then evaluated the potential for low impact forms of seagrass restoration in Aotearoa New Zealand. Seed-based methods have not been trialled in Aotearoa New Zealand because it was previously thought that flowering (required for seed production) in *Z. muelleri* occurred infrequently. However, recent research indicates flowering might be more common than realised, although seagrass seeds have been rarely encountered. Restoration using naturally uprooted seagrass fragments, germinated seeds or seagrass plants propagated in nurseries is likely to have little to no impact on wild seagrass populations. However, these approaches often involve growing seagrass in nurseries, which is expected to cost more than harvesting wild plants from donor meadows.

To improve restoration success and to conduct restoration in an ethical manner, a suite of decisions needs to be made prior to commencing restoration activities. We review available decision-making frameworks and provide information to support some of these decisions. This information includes the environmental requirements, natural recovery timeframes and genetic diversity of *Z. muelleri*. We also highlight the role of local iwi in restoration efforts.

Within the Whakatū/Nelson region, a high-level assessment indicated that Te Tai o Aorere/Nelson Haven, Kokorua Inlet and Wakapuaka/Delaware Inlet show the most potential for seagrass restoration. The next steps to assess feasibility of low impact seagrass restoration include further trial of seed-based (e.g., identifying a source of seeds, germinating seeds and cultivating seedlings in nurseries) and plant-based (e.g., cultivating and propagating seagrass plants in nurseries) restoration methods. Steps to initiate decision making for seagrass restoration in the Whakatū/Nelson region include developing a decision-making framework and partnerships with local iwi, and identification of suitable restoration sites and methods.

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1. INTRODUCTION

Seagrasses are flowering marine plants that form extensive meadows in both intertidal and shallow subtidal coastal environments. These meadows commonly occur in sheltered, soft-sediment areas, away from strong currents and wave action (e.g., estuaries; Hemminga & Duarte 2000; Spalding et al. 2003). Aotearoa New Zealand has only one species of seagrass, *Zostera muelleri*[1] (Figure 1). It colonises intertidal areas and grows subtidally where water clarity permits (Turner & Schwarz 2006).

Figure 1. A subtidal seagrass meadow at Great Mercury Island (left) and an intertidal seagrass meadow at Shakespeare Bay (right), Aotearoa New Zealand (Photos: Dana Clark and Anna Berthelsen, Cawthron Institute).

Seagrass meadows are one of the world's most valuable coastal ecosystems (Costanza et al. 1997), offering an array of ecosystem services that benefit society and the environment (Nordlund et al. 2016). A single meadow creates a variety of habitats (e.g., on and between the plants, within the sediment and overlying water column) amid an often homogenous soft-sediment seafloor (Spalding et al. 2003). As such, they are hotspots of biodiversity, providing food, shelter and refuge for a range of organisms (e.g., macroinvertebrates, epiphytes, fish, birds; Hemminga & Duarte 2000; Figure 2). As one of the most productive ecosystems on Earth (Duarte & Chiscano 1999), seagrass meadows help to regulate the climate by transforming carbon dioxide into biomass as they grow (Duarte et al. 2010; Lavery et al. 2013). Additional carbon storage capacity arises from the ability of seagrass meadows to trap

[1] Previously known as *Z. capricorni* or *Z. novaezelandiae* (Jacobs et al. 2006) and commonly referred to as eelgrass, karepō, nana, rehia and rimurehia.

carbon-rich sediments from adjacent habitats. By stabilising sediments and taking up nutrients and contaminants (e.g., metals; Birch et al. 2018) from the water as they grow, seagrasses also act as natural filters, thereby improving or maintaining coastal water quality (Short & Short 1984).

Figure 2. Bubble shell hiding within a seagrass meadow at Slipper Island, Aotearoa New Zealand (Photo: Dana Clark, Cawthron Institute).

Seagrass meadows are often used as indicators of ecological health, due to their sensitivity to a range of stressors (e.g., Martínez-Crego et al. 2008). The sheltered coastal areas that seagrasses occupy are often close to areas heavily used by humans, making this species particularly exposed to anthropogenic threats. As a result, seagrass meadows are disappearing at a rapid rate worldwide, with at least 29% of global seagrass area lost already (Waycott et al. 2009). Aotearoa New Zealand is no exception; significant declines in seagrass extent occurred during the 1920s and 1970s (Inglis 2003) and continue today in many places (e.g., Park 2016). Consequently, the conservation status of *Z. muelleri* in Aotearoa New Zealand is 'At Risk – Declining' (de Lange et al. 2018).

Seagrass loss is often the result of poor water quality driven by human activities (e.g., coastal development and poor land management; Cullen-Unsworth & Unsworth 2013). Increased sediment and nutrient loads can reduce water clarity and stimulate phytoplankton, macroalgae and epiphytes that compete with seagrass for light (Short & Wyllie-Echeverria 1996). The release of toxic compounds in coastal waters (e.g., metals, herbicides, pesticides) can also inhibit seagrass growth and photosynthesis (Bester 2000; Macinnis-Ng & Ralph 2002; McMahon et al. 2005; Ambo-Rappe et al. 2011). Activities such as dredging, coastal development, anchoring, scouring from moorings, land reclamation and vehicle impact can cause physical damage to seagrass meadows (e.g., Walker et al. 1989; Erftemeijer & Robin Lewis 2006; Ceccherelli et al. 2007; Šunde et al. 2017; Figure 3). Other threats to seagrass include severe storms, overgrazing and/or competition from natural or introduced species, and fungal wasting disease (Matheson et al. 2009). Climate change could also place further pressure on seagrass meadows through exposure to more frequent/intense storm events, poorer water quality, extreme temperatures, and a reduction in suitable habitat due to sea level rise and coastal squeeze (Unsworth et al. 2019).

Figure 3. Photographs showing scouring of subtidal seagrass surrounding swing moorings at Slipper Island (left), and vehicle tracks on intertidal seagrass in Wakapuaka/ Delaware Inlet (right), Aotearoa New Zealand (Photos: Dana Clark and Anna Berthelsen, Cawthron Institute).

Concern about the loss of seagrass meadows has prompted efforts globally to facilitate natural recovery and actively restore seagrass (Orth & McGlathery 2012; Paulo et al. 2019; Tan et al. 2020; Boudouresque et al. 2021). In Aotearoa New Zealand, improvements in environmental conditions have led to natural recovery of seagrass meadows in some places (e.g., Whangarei Harbour; Matheson et al. 2017; Morrison 2021). Active restoration efforts have also been trialled by transplanting seagrass from wild donor meadows. However, removing plants from donor sites places pressure on the wild seagrass meadows that remain. Increasingly, communities and environmental managers are becoming less comfortable with the idea of removing plants from these meadows. Restoration using seeds, or transplantation of naturally uprooted seagrass fragments, germinated seeds or seagrass propagated in a nursery could allow for large scale efforts without damaging wild seagrass meadows.

Nelson City Council (NCC) is exploring options for seagrass restoration in their region. Decline in seagrass meadow extent has been recorded (or inferred) in the four main estuaries in Whakatū/Nelson (Stevens & Robertson 2015; Stevens & Forrest 2019a, 2019b; Stevens et al. 2020). To avoid placing further pressure on remaining seagrass meadows, NCC is interested in the potential to use low impact methods (i.e., those that cause no or minimal damage to wild seagrass) to actively restore seagrass meadows in the region. Thus, the purpose of this report is to examine the potential to apply these low impact seagrass restoration methods in Aotearoa New Zealand, with a focus on the Whakatū/Nelson region:

- We first summarise seagrass restoration methods used internationally and in Aotearoa New Zealand.
- We describe in further detail previous active restoration efforts and information on flowering and seed production for *Z. muelleri* in Aotearoa New Zealand and evaluate the potential for low impact seagrass restoration.
- We review decision making frameworks for restoration and provide information to support some of these decisions in Aotearoa New Zealand. This information includes the environmental requirements, natural recovery timeframes and genetic diversity of *Z. muelleri*, and iwi considerations relevant to seagrass restoration.
- We then evaluate the potential to apply low impact seagrass restoration methods in Whakatū/Nelson estuaries, followed by an outline of the next steps to achieve this goal.

2. SEAGRASS RESTORATION METHODS

Ecosystem restoration as defined here can take two forms: passive restoration where the reduction or removal of an anthropogenic stressor(s) enables the system to naturally recover, or active restoration where humans directly facilitate recovery following the reduction or removal of an anthropogenic stressor(s) (Elliott et al. 2007). In the case of active restoration, reseeding or transplanting is used to establish seagrass at a restoration site. Because the recovery of impacted seagrass via natural processes can take longer than a decade, active restoration methods could be an essential component of a restoration strategy (Balestri et al. 2011). Below, we summarise passive and active seagrass restoration methods based on information in the literature, and comment on method success, resources required and likely impact to wild seagrass populations and the wider environment. Restoration methods used should cater for the seagrass species (and ideally population) that is being targeted. This is because species of seagrass can vary widely in their biological characteristics (e.g., reproductive traits; Orth et al. 2000; Kendrick et al. 2012), especially at the genus or higher level. To ensure that the information summarised is most relevant, we focus on species of the genus *Zostera* as this is the genus of seagrass present in Aotearoa New Zealand. However, examples of restoration efforts relating to other seagrass taxa are provided where they offer additional useful information. Information specific to Aotearoa New Zealand is discussed in Section 3.

2.1. Facilitating natural recovery (passive restoration)

Reducing or removing stressors impacting seagrass populations can passively facilitate natural recovery of seagrass (Paling et al. 2009). The benefits of this can extend beyond seagrass health to the wider ecosystem and human wellbeing (e.g., through improved water quality). Once environmental conditions are suitable for seagrass (refer Section 4.2.1 for further details), natural recovery can occur through (1) the expansion of the meadow due to rhizome growth, (2) the influx of naturally uprooted vegetative fragments from nearby meadows and (3) the germination of seeds produced by plants within the meadow or from nearby meadows (Boudouresque et al. 2021). As well as being a method of restoration in its own right, passive restoration is also a prerequisite for active restoration as the cause of seagrass decline needs to be established and reduced (or removed altogether) for active restoration to be successful (van Katwijk et al. 2009; refer to Section 4).

Actions taken to alleviate or remove physical stressors include the installation of environmentally friendly vessel moorings that reduce damage to seagrass (Ferretto et al. 2021), boating restrictions to reduce seagrass scarring from propellors (Dawes et al. 1997), and the recent banning of anchoring by large ships on seagrass meadows in the French Riviera (Boudouresque et al. 2021, https://medposidonianetwork.com/). Improvements in water quality, at both localised (e.g., around a wastewater pipe) and larger (i.e., estuary- or bay-wide) scales can also facilitate seagrass recovery. For example, reduced catchment nutrient loads, sewage treatment and disuse of sewage outfalls has led to the natural recovery of seagrass meadows (including *Zostera* species) in Europe, the Mediterranean, South Africa and the USA (Bryars & Neverauskas 2004; Leschen et al. 2010; Riemann et al. 2016; Boudouresque et al. 2021).

Passive restoration efforts worldwide have had varying levels of success (Tan et al. 2020). Poor success has been associated with factors such as failure to ensure the original stressor was alleviated, abiotic and biotic interactions (Moksnes et al. 2008; Valdemarsen et al. 2010) and propagule supply limitations (Orth et al. 1994; Kendrick et al. 2012). Timeframes for natural recovery of *Zostera* can be relatively slow, especially if large spatial scales are involved (refer Section 4.2.2). The financial cost and effort of passive restoration can vary substantially. Alleviating (or removing) local-scale impacts to seagrass meadows is likely to cost less than implementing larger-scale changes. For example, prohibiting vehicle access to an estuary is likely to be less expensive than improving water quality through catchment-scale land management changes, although in some cases both may need to happen for natural recovery to occur. It can also be more difficult to get support (e.g., from a societal or political perspective) for large-scale changes. In some cases, passive seagrass restoration is might not be possible, for example, if the original loss was due to land reclamation for development.

Promoting positive biological interactions could help to improve seagrass restoration success (Tan et al. 2020), both for natural recovery via passive restoration as well as active restoration. For seagrass, biological interactions can occur between microbial communities, plants, shellfish and coral reefs (Tan et al. 2020) as well as herbivores (Tol et al. 2021). For example, in a highly exposed habitat, seagrass planted within a mussel bed had better survival than transplants located 60 m away from the mussel bed (Bos & van Katwijk 2007). Negative biological interactions can also occur. For example, *Zostera* meadows can be replaced by mangroves (Cann et al. 2009) and cordgrass (Cottet et al. 2007; Roberts & Pullin 2008), although environmental conditions likely play a role in facilitating this in some cases.

2.2. Active restoration using seagrass seeds

Sexual reproduction via flowering and seed production represents an important mechanism for maintaining dispersal, connectivity, and genetic diversity in seagrass populations (Kendrick et al. 2012). Seagrass seeds collected from wild seagrass populations and spread or planted directly onto a restoration site is a low impact restoration technique (Boudouresque et al. 2021). Furthermore, large scale seed-based methods could provide an economically feasible source of plant material (Busch et al. 2010; Statton et al. 2013), although seed handling and storage can require considerable effort (Pickerell et al. 2006).The various steps in seed-based restoration, which include seed collection, processing and storage, followed by spreading or planting, are described in more detail below.

2.2.1. Collecting, processing and storing seeds from the wild

Seed-based restoration first requires the collection of seeds from donor meadows. Seagrass seeds attached to floating shoots can also disperse (Stafford-Bell et al. 2016), therefore, beach-cast wrack may be a potential source of seeds for restoration. Seagrass flowering and seed production has various stages (e.g., Figure 4), following seasonal patterns controlled by light, temperature and nutrients (Dos Santos & Matheson 2017). An in-depth understanding of sexual reproduction for the target seagrass population is important given the variation that can occur between species and populations (Kendrick et al. 2012; Dos Santos & Matheson 2017). For *Z. marina*, the collection of flowering shoots with maturing seeds for restoration has traditionally been carried out by hand (e.g., via snorkelling or SCUBA; Busch et al. 2010; Orth & McGlathery 2012). Mechanical harvesting of *Z. marina* seeds with a barge can be used to improve collection efficiency and still has a low impact on the donor meadow (Marion & Orth 2010). However, collection rates depend on seagrass productivity and it is currently unknown if mechanical harvesting is worthwhile for Australian and New Zealand species (Tan et al. 2020). Once collected, flowering shoots can be suspended above tanks until the seeds are shed, which is followed by processing to separate the seeds from leaf and stem matter (Marion & Orth 2010; Tanner & Parham 2010).

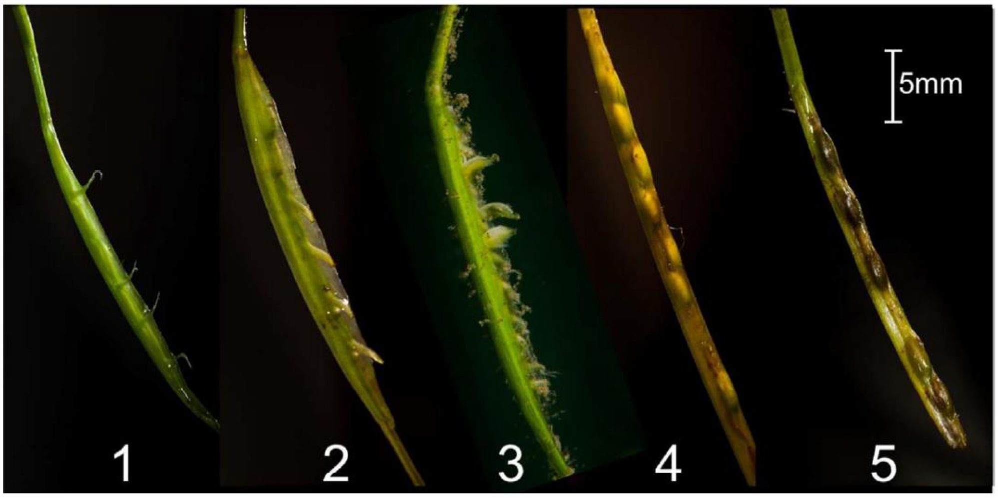

Figure 4. Flowering stages of *Zostera marina*: 1) styles erect from the spadix[2], 2) styles bend back after pollination, 3) pollen is released from the anthers[3], 4) seed maturation, 5) seeds are released. Image taken from Infantes and Moksnes (2018).

[2] A spike of minute flowers closely arranged around a fleshy axis and typically enclosed in a modified leaf called a spathe.
[3] Part of the stamen (pollen producing part of the flower) that produces the pollen.

The seeds of most seagrass species (including *Zostera*) have a dormant stage (Orth et al. 2000), where they spend time in the sediment before germinating. For example, along the mid-Atlantic coast of North America, *Z. marina* seeds are shed during spring and summer but generally do not germinate until the following autumn (Tanner & Parham 2010). For species with seed dormancy, collected seeds can be processed and then stored for a period of time so that spreading or planting aligns with the season in which seed germination naturally occurs. For *Z. marina* restoration in North America, seeds were processed by separating them from the senescing shoots and then stored in flow-through or recirculating seawater tanks until spreading (Moore et al. 1993; Marion & Orth 2010; Tanner & Parham 2010; Orth & McGlathery 2012). Ideal seed storage time has been studied for various *Zostera* species; for example, it was recommended that storage of *Z. marina* shoots should not exceed 40 days to allow optimal release of viable seeds (Infantes & Moksnes 2018). The environmental conditions (e.g., salinity and temperature) under which *Zostera* seeds are stored can influence the length of time that they stay viable during storage (Tanner & Parham 2010; Yue et al. 2019). Viable seeds can also be sorted prior to spreading to improve germination success. Viability of *Zostera* seeds has been determined by assessing their firmness and fall velocity (Marion & Orth 2010; Orth & McGlathery 2012), by using a vertical flume (Infantes & Moksnes 2018) or by using tetrazolium[4] (Conacher et al. 1994). Seed germination (and seedling recruitment) can also be enhanced *in situ* (refer Section 2.3.1).

2.2.2. Spreading and planting seeds

Once seagrass seeds are collected, processed and stored, they can then be spread or planted at the restoration site where they can germinate and develop into plants. Seeds have been traditionally spread by hand, which has proven to be successful even at large scales. For example, 10 million *Z. marina* seeds were broadcast by hand from a moving boat into four coastal lagoons over two decades in a successful restoration project in Virginia, USA (Orth et al. 2020). Not all seed-based restoration efforts are successful though. A Swedish trial, where reproductive shoots were stored within mesh bags, recorded very low *Z. marina* seedling establishment, although high seedling growth rate occurred in those that did survive (Eriander et al. 2016). A study on dispersal found that seed-based restoration of *Z. muelleri* was more likely to succeed in areas of existing seagrass rather than in disturbed areas where seagrass is no longer present (Macreadie et al. 2014).

Various other techniques have been trialled for planting *Z. marina* seeds, with differing levels of success. These include using planting seeds into peat pots, broadcasting seeds and then covering the area with burlap, placing seeds in burlap bags and then burying the bags in the field, deploying seed-bearing shoots in buoys, and using a mechanical planter (Harwell & Orth 1999; Orth et al. 2006; Pickerell et al. 2006; Orth et al. 2009; Marion & Orth 2010; Orth & McGlathery 2012). Dispenser injection

[4] Tetrazolium is a chemical commonly used to check seed viability.

seeding, where seeds and sediment are mixed and injected into the substrate (Tan et al. 2020) could be a useful method to test in Aotearoa New Zealand. This approach was less labour intensive than other seeding techniques and could be more successful in intertidal areas because the seeds are less likely to get washed away.

Seed-based restoration can provide an opportunity for community groups to work together, for example, in Western Australia, fishers, powerboat owners and scientists have together sown over 200,000 viable *Posidonia australis* seeds (Sinclair et al. 2021).

2.3. Active restoration using seagrass plants

Vegetative development via rhizomes is the main mechanism for seagrass expansion (Marbà & Duarte 1998). Thus, active seagrass restoration often involves planting seagrass at restoration sites with the goal of establishment and expansion via clonal reproduction. Seagrass plants for active restoration can be obtained by 1) harvesting wild plants from donor sites, 2) collecting naturally uprooted seagrass fragments, 3) germinating seeds, or 4) propagating plants in a land-based nursery (i.e., land-based laboratory aquaria/tanks or aquaculture systems). These plants are then transplanted to a restoration site (sometimes following storage or cultivation in a nursery) using a variety of methods.

2.3.1. Obtaining seagrass plants for transplantation

Harvesting wild plants from donor sites

Seagrass plants can be obtained from donor sites in the wild, by digging up intact clods[5] of seagrass (including shoots, rhizomes, leaf blades and roots) or by collecting 'bare-rooted' plants[6] (rhizome sections with leaf-bearing shoots and rhizomes/root nodes; Ganassin & Gibbs 2008). Taking seagrass from wild meadows is the most common method of obtaining seagrass plants for restoration (Ferretto et al. 2021) and has been used globally in restoration efforts for a range of seagrass species (Fonseca et al. 1998; Paling et al. 2001; Tan et al. 2020; Boudouresque et al. 2021). Clods are generally preferred over bare-rooted plants because the root and rhizome systems remain intact and the original sediment accompanies the plants to the new location (Fonseca et al. 1998). Furthermore, clods provide greater anchorage than bare-rooted plants (Paling et al. 2001).

Obtaining seagrass plants from wild meadows has the potential to impact donor populations through physical damage that causes fragmentation of the meadow and inhibition of natural re-growth (Bull et al. 2004). Physical disturbance could also

[5] Similar terms to 'clods' include plugs, cores, sods, matte, blocks and turf. Terminology may depend on the size and shape of the sediment clod.
[6] Similar terms to 'bare-rooted plants' include sprigs, shoots, cuttings and rhizomes. Terminology may depend on the specific part of the plant being used.

increase seagrass susceptibility to diseases (e.g., fungal wasting disease *Labyrinthula*; Turner & Schwarz 2006). Due to the potential for negative impacts, the conservation status of some seagrass species restricts this method of collection in certain countries (e.g., Australia, Europe; Coles & Fortes 2001; Boudouresque et al. 2012). Harvesting wild plants from donor meadows is generally more expensive than collecting seeds because it involves more effort (Fishman et al. 2004; Tanner & Parham 2010). Furthermore, it restricts, rather than increases, genetic diversity in restored populations (Williams 2001).

Collecting naturally uprooted seagrass fragments

Bare-rooted seagrass plants that have detached naturally (commonly referred to as ‘seagrass fragments’) due to natural (e.g., storms, strong winds) or anthropogenic (e.g., anchoring, dredging) processes can be collected and used as material for seagrass restoration, thereby avoiding the need to damage donor meadows. They are often found washed ashore on beaches adjacent to seagrass meadows and are inexpensive to collect. Collection success can be increased by identifying favourable collection conditions based on factors such as winds and tides (e.g., Australia, Europe; Ferretto et al. 2021). Balestri et al. (2011) calculated that up to 150 fragments could be collected by one person in an hour following an intense storm. This approach is conducive to citizen-scientist engagement (e.g., Ferretto et al. 2021), which can increase the number of fragments collected, reduce restoration costs, raise awareness and create a sense of community ownership in restoration programmes.

The length of time that seagrass fragments remain viable after detachment depends on the seagrass species and the environmental conditions (e.g., temperature, UV damage). In an Australian study, 50% of *Z. muelleri* fragments remained buoyant after 5 weeks of floating and most of these were viable (Stafford-Bell et al. 2015). *Z. marina* fragments have been shown to remain viable for 12 weeks, although their reestablishment capacity is significantly reduced after six weeks (Ewanchuk & Williams 1996). To be capable of producing new shoots and roots, seagrass fragments should ideally have at least one shoot attached to a rhizome, with larger fragments expected to grow faster because they have more meristems[7] (Figure 5; Balestri et al. 2011). An Australian study found that 3-11% of beach-cast *Z. muelleri* fragments were capable of regrowth (Stafford-Bell et al. 2015). Fragments must be collected before they dry out and kept in suitable conditions until planting (refer Section 2.3.2). Although storage of fragments can increase handling costs and reduce the survival of fragments, this process naturally selects the most viable fragments for transplantation and means fragments do not need to be re-planted on the same day as collection, allowing for transplantation in larger batches during a favourable season (Balestri et al. 2011; Ferretto et al. 2021).

[7] A region of plant tissue, found chiefly at the growing tips of roots and shoots and in the cambium, consisting of actively dividing cells forming new tissue.

Figure 5. Fragment of *Zostera muelleri* (Photo: The Marine Life Database 2020).

Following collection, fragments can then be propagated in a laboratory or nursery or directly planted at a restoration site. As the survival of stored fragments often decreases over time, fragments planted within a few months of collection may have the greatest success (Balestri et al. 2011). Direct transplantation of fragments has been successful for *Z. muelleri*. For example, Zhou et al. (2014) reported > 95% survival after three months and no significant differences in shoot density, height or aboveground biomass compared to a wild meadow after three years. However, in a different study 100% of transplanted *Z. nigricaulis* fragments died within 100 days, possibly because of the timing of the transplantation (Thomson et al. 2015).

Germinating seeds

An alternative to directly spreading or planting seagrass seeds is to germinate seeds under cultured conditions and then on-grow the seedlings (Figure 6) prior to transplantation. Benefits of this approach include reducing physical damage to donor meadows (compared with harvesting wild plants), increasing plant survival rates, and facilitating genetic diversity in the restored seagrass population (Tan et al. 2020). Barriers to using this method include limited seed availability for some species (Orth et al. 2000) and the resources required to maintain a nursey system (refer Section 2.3.1) and harvest the plants in preparation for transplantation (Tanner & Parham 2010). Germinating *Z. muelleri* seeds under cultured conditions has also received little attention overseas, although in Australia the practicality of germinating seeds of this species has been recently assessed (Tan et al. 2020). However, large-scale seedling culture has been successfully demonstrated for *Z. marina*, where 1500 shoots were grown from seed and transplanted to a restoration site in Chesapeake Bay, USA (Tanner & Parham 2010). It took between 70 to 100 days to germinate the seeds and grow the plants to a size large enough to transplant (Tanner & Parham 2010), with cultured plants performing better at the restoration site than plants taken from wild donor meadows (Tanner et al. 2010).

On smaller scales, seed germination and seedling growth and survival in cultured conditions has been demonstrated in experimental trials for several *Zostera* species (Abe et al. 2009; Tanner & Parham 2010; Kishima et al. 2011; Pan et al. 2011; Wang et al. 2016; Xu et al. 2016; Cumming et al. 2017). Germination was found to be influenced by several environmental variables including salinity, temperature, dissolved oxygen, sediment type, and seed burial depth (Moore et al. 1993; Orth et al. 2000; Abe et al. 2010; Kishima et al. 2011; Wang et al. 2016; Cumming et al. 2017). Survival and growth of cultured seedlings was also influenced by temperature and salinity, as well as light and sediment nutrient enrichment (Abe et al. 2010; Xu et al. 2016).

Figure 6. The morphology of *Zostera marina* seedlings. A, seed; B, after 1 d; C, after 2 d; D, after 1 week (arrows show cotyledon[8] and leaf); E, after 2 weeks; F, after 3 weeks; G, after 4 weeks; and H, after 5 weeks. (Photos: Pan et al. 2011).

[8] Embryonic leaf in seed-bearing plants (the first leaves to appear from a germinating seed).

Actions can also be taken to enhance natural seagrass germination and seedling survival in the wild or to collect germinating seeds and transplant them once they develop into plants. Methods trialled include laying a sheet, bag or fibre membrane on the seafloor to help prevent loss of seeds and seedlings (Seddon et al. 2005; Tanner 2015; Sousa et al. 2017).

Propagating seagrass plants

Seagrass shoots can be propagated vegetatively from adult plants that have either been transplanted from wild donor sites or grown in a nursery from seeds or naturally uprooted vegetative fragments. Conceivably, propagation of seagrass plants in land-based nurseries (refer Section 2.3.2) could provide a continuous supply of seagrass plants for low-impact restoration, as occurs for terrestrial plants. This could also create opportunities to selectively breed cultivars with faster growth rates, greater tolerance to environmental stressors (e.g., turbidity, nutrient loading, climate change impacts) or disease resilience. For example, Tanner and Parham (2010) observed a 19-fold increase in the number of shoots from *Z. marina* plants grown in the nursery over six months. By harvesting shoots and replanting these into the nursery at lower densities, they were able to maintain seagrass in a nursery system for four years. Similarly, dividing *P. oceanica* fragments grown in a nursery for three years doubled the number of transplants for restoration (Balestri et al. 2011). Emerging propagation techniques (e.g., micropropagation and callus induction) that are commonly used for terrestrial crops are also beginning to be developed for seagrasses (Hamill & Sumby 2002; Ailstock & Shafer 2006; Tigani 2007; O'Brien 2019). These approaches offer the potential to rapidly propagate disease-free plants from limited donor material and require less space than traditional methods of propagation (O'Brien 2019). Although propagation (via traditional or emerging methods) has the potential to provide a continuous supply of seagrass plants for restoration, this clonal method of reproduction is likely to result in low genetic diversity (and thus resilience) in the restored population. Genetic diversity can be maintained by starting clones from seeds, or plants/fragments collected from several seagrass populations or populations with high genetic diversity (Tanner & Parham 2010; refer Section 4.1.3).

2.3.2. Nursery systems for storage, cultivation and propagation of seagrass

Donor plants harvested from wild seagrass meadows, naturally uprooted vegetative fragments or germinated seeds can be stored, cultivated and/or propagated in nurseries before transplanting to restoration sites. Storage means the temporary housing of seagrass plants until enough material is collected for planting or until planting conditions are favourable (e.g., a season with high seagrass growth and/or calm weather conditions). Plants can also be cultivated in a nursery until they reach a larger size to increase their chances of post-transplant survival. Alternatively, adult plants can be propagated to increase the amount of seagrass material for restoration.

Nurseries can range from simple aquaria that fluctuate with the ambient environmental conditions (e.g., Cade 2016; Figure 7) to sophisticated flow-through systems with options to control environmental variables (e.g., light, temperature; Zabarte-Maeztu et al. 2021). Although *Z. muelleri* fragments can be stored floating in water before transplanting (Stafford-Bell et al. 2015; Cade 2016), planting seagrass into a sand substrate is likely to be required for long-term cultivation. Sediment composition may be important, with growth of nursery-cultivated *Z. marina* and *P. australis* seedlings and fragments improved when sediments were finer (Tanner & Parham 2010; Statton et al. 2013). When the sediment is not naturally rich in organic matter, the addition of refractory organic matter (e.g., seagrass detritus) may also enhance plant growth (Statton et al. 2013). However, the addition of seagrass detritus to sediments that already contain organic matter (1.0-1.3% sediment dry weight) did not improve the growth of *Z. marina* seedlings (Tanner & Parham 2010) and inhibited growth of *Halophila ovalis* plants (Kilminster et al. 2006). Similarly the addition of rapidly decomposing forms of organic matter (e.g., sucrose, fish feed and algae) could have a negative effect on seagrass growth and survival (Statton et al. 2013). Exogenous application of rooting hormones and fertiliser has been used to enhance shoot production and rhizome growth (e.g., Fonseca et al. 1994; Sheridan et al. 1998; Balestri & Bertini 2003; Tanner & Parham 2010) and could be trialled for nursery cultivated *Z. muelleri*.

Figure 7. Growing seagrass (*Zostera muelleri*) in a simple nursery system in Aotearoa New Zealand (Photos: Cade 2016).

Key considerations for nursery establishment include:

- seawater should be filtered
- seawater temperature remains within natural range for the local seagrass cultivar
- seawater salinity remains within natural range for the seagrass species
- sufficient light and nutrients are available for photosynthesis
- water remains oxygenated

- algal growth is controlled
- suitable substrate is used if planting seagrass.

Several studies have demonstrated that it is possible to store or cultivate seagrass in land-based nurseries (e.g., Balestri & Bertini 2003; Tanner & Parham 2010; Statton et al. 2013; Cade 2016; Ferretto et al. 2021; Zabarte-Maeztu et al. 2021), in some cases for as long as two to three years (e.g., Meinesz et al. 1991; Meinesz et al. 1993; Balestri et al. 2011). For example, large-scale germination and culture of *Z. marina* provided 15,000 seedlings that were planted out at a restoration site three months after germination (Tanner & Parham 2010). Nursery-grown seedlings were found to have higher survival rates, taller shoots and greater lateral expansion than plants transplanted from wild seagrass meadows (Tanner & Parham 2010). Similarly, Balestri et al. (2011) demonstrated that naturally uprooted *P. oceanica* seagrass fragments cultivated in a nursery for three years maintained their capacity to grow and re-establish when transplanted back into the wild.

One downside to this approach is that maintaining plants in a nursery could be expensive. This cost, and the need to develop successful germination and cultivation methods for many species, is likely the reason that seagrasses are not generally commercially propagated for restoration (Shafer & Bergstrom 2008). The cost will depend on the growing requirements of the seagrass species and will be higher where electricity is required to power heat pumps, lights and flow-through water systems. There are also large space requirements along with high labour costs associated with planting and harvesting plants, frequent cleaning of the tanks and water changes. Costs could be reduced if seagrass is able to be grown using a simple nursery system (e.g., static tanks kept outdoors and exposed to ambient changes in light and temperature) and maintained by community groups. The addition of invertebrate grazers to tanks could be a cost-effective way to control excessive algal growth (e.g., Seddon et al. 2005).

2.3.3. Transplanting seagrass plants

Once obtained, seagrass plants need to be transplanted to a restoration site. A variety of transplantation techniques exist, ranging from simple direct planting to the more complex use of stakes or structures to anchor and shelter the plants. Transplantation of seagrass plants generally costs more than spreading or planting seeds because there is more effort involved (Fishman et al. 2004; Tanner & Parham 2010).

Planting seagrass clods or plants in the seafloor

The least destructive form of transplantation is to plant seagrass clods or bare-rooted plants directly into the seafloor sediments without the addition of stakes or other artificial structures (Campbell 2002; Boudouresque et al. 2021). Transplantation has been used for *Zostera* species worldwide using relatively small clods or bare-rooted plants, with mixed success. It was successful in a Swedish transplantation study where survival and growth of unanchored *Z. marina* plants was higher than for those that were anchored (Eriander et al. 2016). Conversely, in an Australian trial the survival of transplanted *Z. capricorni*[9] was low and high costs were incurred along with damage to the donor plots (McLennan & Sumpton 2005).

Paulo et al. (2019) identified that the size of the transplanted plot can influence restoration success with only larger plots (up to 103 m^2) of transplanted *Z. marina* persisting over time along the Portuguese coastline. Larger scale (i.e., 43–1020 m^2) seagrass transplantation has also been trialled. Relatively large clods (e.g., ~ 1–3 m^2 in area, 40–60 cm deep) of seagrass (none of these *Zostera* species) were transplanted using balloons (Sánchez Lizaso et al. 2009), boat-cranes and barges (Bedini et al. 2020), underwater harvesting and planting machines (Paling et al. 2001) and mechanised boats and other equipment (Uhrin et al. 2009; Descamp et al. 2017), with mixed success.

Fixing plants to the seafloor

Fixing seagrass plants to the seafloor using some form of anchoring system (e.g., stakes, staples, pegs, wire, nails) can help reduce the impacts of water motion (e.g., from storms and waves) on transplants thereby increasing survival success. Absence of a sufficient anchoring system for transplants is a common reason for poor restoration success (West et al. 1990; Bastyan & Cambridge 2008). Anchoring plants can be relatively inexpensive, but it does require additional time to attach fixtures (Fonseca et al. 1998). Various *Zostera* species have been successfully anchored to the seafloor using staples (Fonseca et al. 1998), bamboo sticks (Davis & Short 1997), wire anchors (West et al. 1990), oyster shells (Lee & Park 2008), stones (Zhou et al. 2014), and iron nails (Lange et al. in review, referenced in Tan et al. 2020). However, depending on the hydrodynamic environment, anchors may still not be strong enough to hold plants in position over the longer term (West et al. 1990). The use of anchoring devices introduces artificial material into the environment (Fonseca et al. 1998), hence natural or biodegradable options are desirable (e.g., bamboo stakes, stone or oyster shells as described above, or the starch-based anchoring system described in Scannavino et al. 2014).

Use of structures

In-water structures can improve seagrass restoration success by stabilising the sediment and providing protection from water motion and herbivores (Tan et al. 2020; Piazzi et al. 2021). The range of structures and material used for this purpose include grids, mats, frames, and cages made of artificial (metal, concrete, plastic) or biodegradable materials. Structures can be cost effective or relatively expensive, depending on the technique used (Piazzi et al. 2021).

[9] Synonymous with *Zostera muelleri*.

Several structures have been trialled using *Zostera* species. For *Z. muelleri,* a study in Australia demonstrated that frames to anchor seagrass plants, mats to protect from bioturbating animals and cages to exclude herbivorous fish were the most successful techniques of those investigated, although overall success of these techniques was site dependent (Wendländer et al. 2020). Weighted wire frames that lie on the seafloor and protect plants from bioturbation and uprooting have also been successfully trialled for *Z. marina* in the USA (Short et al. 2002). There are various examples for other seagrass species. For example, mats (degradable and non-degradable) were used to enhance the survival of *P. oceanica*, minimise the loss of transplanting plots and facilitate colonisation of other organisms (Piazzi et al. 2021) and wire mesh cages were used to protect transplants (*Cymodocea nodosa*) from fish grazing (Boudouresque et al. 2021). However, surrounding seagrass units with plastic mesh did not improve plant survival in a high energy wave environment (van Keulen et al. 2003). Concrete structures have also been used overseas in attempts to restore *P. oceanica*, although this method is destructive to the environment given that the plants may die but the cement could endure for centuries (Boudouresque et al. 2021).

Artificial seagrass that mimics the physical properties of living seagrass can also improve survival of seagrass transplants. For example, artificial seagrass improved hydrodynamic conditions for *Z. marina* by reducing wave height and current velocity (Carus et al. 2020). For other seagrass species, this method has also been shown to reduce damage to seagrass from herbivory (Tuya et al. 2017) and increase transplant survival by stabilising sediment grain size (although not sediment erosion or accretion; Campbell & Paling 2003).

Artificial structures can pollute the environment, and efforts have been made to develop more sustainable options by using biodegradable materials (Tan et al. 2020). Examples include hessian bags, mesh made from natural fibres, paper mesh and coconut fiber pots (Gobert et al. 2005; Tanner 2015; Ward et al. 2020; Ferretto et al. 2021). Artificial seagrass made from biodegradable materials is also being developed (SeaArt 2020).

3. SEAGRASS RESTORATION IN AOTEAROA

3.1. Previous active restoration efforts

Active seagrass restoration has been trialled in four estuaries/harbours in Aotearoa New Zealand: Manukau (Turner 1995), Whangarei (Matheson 2016; Matheson et al. 2017), Porirua (Zabarte-Maeztu 2021) and the Avon-Heathcote estuary (Gibson & Marsden 2016). These studies relied on the transplantation of wild seagrass plants and had varying degrees of success. The first restoration trial began in 1993 in Manukau Harbour (Turner 1995), as an attempt to rehabilitate seagrass following large losses between the 1960s and 80s (Inglis 2003). Seagrass was transplanted both as intact sods and as bare-rooted plants anchored with staples. Unfortunately, restoration was unsuccessful, with a decline in survival occurring after six months following the onset of autumn storms (Turner 1995).

Like Manukau Harbour, Whangarei Harbour suffered extensive losses of seagrass in the 1960s and 70s (Reed et al. 2004). Deteriorating water clarity and sediment smothering caused by capital dredging operations and discharges from a cement factory were likely the primary drivers of this loss (Reed et al. 2004). Seagrass transplantation was trialled here in 2008 and 2012 following a feasibility study that showed that water quality and sediment properties were suitable for restoration (Reed et al. 2005). In the 2008 trial, three transplantation methods were tested: intact sods, unanchored bare-rooted plants and bare-rooted plants planted within mats of artificial seagrass (Matheson et al. 2017). Transplanting sods and bare-rooted plants was equally effective, suggesting no benefit to keeping the root and rhizome system intact and no negative effects of additional handling or lack of anchoring device using the bare-rooted plant method. Conversely, planting within the mats was unsuccessful, likely due to light limitation arising from the artificial canopy and associated biofouling, and restrictions on macrofaunal bioturbation caused by the mesh base (Matheson et al. 2017). A more than four-fold increase in seagrass extent was observed across the 1 ha site in the two years post-transplantation (Matheson et al. 2017). In 2012, four planting configurations, ranging from a cluster of five small cores to larger plots of 0.5 m^2, were tested at a second site in the harbour (Matheson 2016). All configurations were equally successful four years post-transplant. Seagrass in Whangarei Harbour has now increased by 600 ha (Tan et al. 2020) to reoccupy 40% of its previous range (Matheson et al. 2017). However, it is likely that seagrass extent would have increased without intervention following improvements in environmental conditions (passive restoration, refer Section 2.1), making it difficult to assess the contribution of active transplantation to this expansion. Nevertheless, it demonstrates that it is possible to successfully transplant seagrass in Aotearoa New Zealand if the environmental conditions are suitable.

In Porirua Harbour, seagrass extent has declined by 40% since 1980 (Matheson & Wadhwa 2012). These losses have been linked to substrate muddiness and consequent unfavourable rhizosphere conditions as well as low irradiance and smothering by sediment during storms (Zabarte-Maeztu et al. 2020; Zabarte-Maeztu 2021). Elevated water column nitrate, sediment contamination and smothering by sea lettuce (*Ulva* spp.) and a filamentous green alga (*Chaetomorpha ligustica*) have also been proposed as drivers of this change (Matheson & Wadhwa 2012; Zabarte-Maeztu 2021). In 2015 and 2019, two seagrass transplant trials were carried out in Porirua Harbour (Zabarte-Maeztu et al. 2020; Zabarte-Maeztu 2021). Intact seagrass sods transplanted on two occasions in 2015 did not survive more than 14 months (Tan et al. 2020), likely due to low light availability, smothering by fine sediment and unfavourable substrate conditions (Morrison 2021; Zabarte-Maeztu 2021). Seagrass transplantation using intact sods and bare-rooted plants anchored with staples was tested in 2019 (Zabarte-Maeztu 2021). After 45 days, seagrass retention ranged from 20–100%, with transplantation using sods the preferred method because seagrass retention was high where environmental conditions were favourable, and it was easier to relocate the transplanted seagrass. Zabarte-Maeztu (2021) found it difficult to recognise the transplanted bare-rooted plants amongst the surrounding seagrass (in control treatments) and challenging to stabilise them using staples in the dynamic intertidal zone.

Extensive loss of seagrass occurred in Ihutai/Avon Heathcote estuary prior to 1929 with almost no seagrass remaining by 1952 (Inglis 2003). Since then, the extent of seagrass bed has fluctuated and was reported to cover 52 km^2 in 2016 (Gibson & Marsden 2016). A transplant experiment was carried out in the estuary in 2016, using intact sods harvested from two donor sites (Figure 8; Gibson & Marsden 2016). However, after three months only four of the 15 transplanted sods were still alive. The rest had either washed away, been smothered with sea lettuce (*Ulva* spp.) or died after bleaching. Gibson and Marsden (2016) recommended that seagrass transplantation is undertaken in the spring when seagrass growth has resumed.

Figure 8. Intact sods being transported (left) and *in-situ* at the restoration site (right) in the Avon-Heathcote seagrass transplant experiment (Photos: Gibson & Marsden 2016).

3.2. Information relevant to seed-based restoration potential

Seed-based restoration (refer Section 2.2) has not been trialled in Aotearoa New Zealand because it was previously thought that sexual reproduction for *Z. muelleri* occurred infrequently (Turner & Schwarz 2006). For example, extensive sampling within Raglan Harbour found no evidence of a sediment seed bank (Cade 2016). However, recent research has documented flowering by *Z. muelleri* (Figure 9) in several locations across the country, indicating that seed-based methods may be an option for restoration if viable seeds can be found.

Figure 9. Seagrass flowering in *Zostera muelleri* showing protruding stigma[10] to catch pollen for fertilisation (Photo: Tracey Burton, from de Kock et al. 2016).

Seagrass flowering and/or seeding has now been documented in both the South Island (Ramage & Schiel 1998; Ismail 2001; Zabarte-Maeztu 2021) and North Island (Matheson et al. 2010; de Kock et al. 2016; Dos Santos & Matheson 2017). Flowers were recently recorded in the Top of the South (Te Tau Ihu) from Golden Bay (Tasman) to Ngakuta Bay in Queen Charlotte Sound, although no seeds were found (pers comm., Zabarte-Maeztu, NIWA, 10 September 2021). Flowering has also been observed at Kaikōura and Otago. The reproductive periodicity and output of seagrass on two intertidal reefs in Kaikōura was studied over two years (Ramage & Schiel 1998). The number of flowering shoots was found to peak during March (late summer), with no flowering shoots present during winter (July to September). Flowering shoots were more abundant in the low intertidal zone in comparison to the

[10] Part of a plant that is specially adapted to catch pollen

middle and upper zones, as well as in seagrass patches associated with tidepools. Seeds were found over summer (December to March), and appeared to germinate the following spring (September). Ramage and Schiel (1999) also reported that seagrass seedlings recruited into small areas of sediment in tidepools on intertidal reef platforms in Kaikōura during spring (although few survived through summer due to removal by wave action). Overall, they determined that successful recruitment by seedlings may be rare or at least episodic at this location. A few flowering shoots were also documented at an intertidal meadow in Otago Harbour during summer (December and March) in 1997 and 1998. However, flowering was not followed in detail during this study, with the data limited only to plants in experimental plots. Germinating seagrass seedlings (on cockles in the intertidal) have also been observed in Ihutai/Avon Heathcote estuary in Christchurch (Gibson & Marsden 2016).

In the North Island, the abundance of flowering shoots was quantified at four intertidal sites in Tauranga Harbour every month over two and a half years by Dos Santos and Matheson (2017). Low densities of flowering shoots were present annually in spring and summer at some of the sites. Flowering shoots were associated with higher plant cover and biomass, and larger leaf size. Dos Santos and Matheson (2017) looked for seeds at one site but none were found, potentially due to the relatively low sampling effort and/or low flowering density or the possibility that low genetic diversity in the local seagrass population precluded successful fertilisation. Seagrass flowering was also recorded monthly (November to January) at seven sites Raglan Harbour (de Kock et al. 2016). Like the Tauranga study, flowers were associated with higher seagrass cover and biomass and larger leaf size. They also found that flowers were more common where in lower salinity areas and near seeps, streams and tide pools. Despite the presence of flowers, no seeds were found. A single seagrass flowering shoot was also discovered during a subtidal survey in the Bay of Islands (Matheson et al. 2010).

Local iwi may also have mātauranga (pūrākau, tohu, whakapapa) relating to seagrass flowering and seed production in their rohe (refer Section 4.2.4). Restoration programmes should be sure to incorporate this information.

3.3. Potential for low impact forms of seagrass restoration

Up until now, seagrass restoration efforts in Aotearoa New Zealand have relied on harvesting wild plants from donor sites and/or passive restoration through the reduction or removal of stressors. Other methods of obtaining seagrass material for restoration (i.e., collecting seeds or naturally uprooted seagrass fragments, germinating seeds or propagating seagrass plants in nurseries) show potential but have not yet been trialled here. The pros and cons of applying different restoration methods in Aotearoa New Zealand are summarised in Table 1 and discussed below.

3.3.1. Passive restoration

Water quality is one of the key threats to seagrass in Aotearoa New Zealand (Turner & Schwarz 2006), so the reduction of sediment and nutrient inputs to the coastal environment provides opportunities for passive seagrass restoration. For example, seagrass extent increased substantially within four years after wastewater diversion in the Ihutai/Avon Heathcote estuary in Christchurch (Gibson & Marsden 2016; Zeldis et al. 2020). Passive restoration leading to increased water clarity in Whangarei Harbour was also critical to the success of active seagrass restoration in 2008 and 2012 (Matheson et al. 2017). Efforts to improve water quality through voluntary and regulatory (e.g., National Policy Statement for Freshwater Management; NPS-FM 2014) actions are already occurring in Aotearoa New Zealand, but there is scope to increase work in this space. In addition to improvements in water quality, there is also opportunity to facilitate natural recovery on local scales by reducing physical impacts such as those caused by vehicle and boating activities. Passive restoration to facilitate natural recovery should be considered prior to active restoration, as the establishment of appropriate environmental conditions is a prerequisite for successful active restoration. However, natural recovery on large spatial scales can be slow; often taking decades (Vaudrey et al. 2010; refer Section 4.4.2.). Active restoration offers the potential to accelerate natural recovery rates.

3.3.2. Seed-based restoration

Seed-based restoration relies on a source of seeds, ideally local. Although flowers have now been observed in several seagrass meadows across Aotearoa New Zealand, seeds have only been found in one location (Ramage & Schiel 1998). If a suitable source of seeds could be found, seed-based restoration is likely to be feasible in Aotearoa New Zealand, given the success of this approach for other species of *Zostera* (e.g., *Z. marina* in the United States of America; Orth et al. 2020). To the best of our knowledge, seed-based restoration has not yet been carried out for *Z. muelleri*. However, the practicality of collecting, storing, and germinating *Z. muelleri* seeds was recently assessed in Australia (Tan et al. 2020), providing useful guidance for Aotearoa New Zealand. Spreading or planting seeds is a low cost, low effort and low impact form of restoration.

3.3.3. Obtaining plants for transplanting

Wild seagrass harvested from donor sites has been successfully transplanted in Aotearoa New Zealand when environmental conditions were suitable for establishment (e.g., Matheson et al. 2017). Although higher impact than the other forms of restoration methods discussed, donor sites in Whangarei Harbour and Avon-Heathcote estuary recovered relatively quickly (Gibson & Marsden 2016; Matheson et al. 2017; refer Section 4.2.2 for more information). However, harvesting wild seagrass for transplantation may not be a sustainable restoration method for large-scale projects, or where donor meadows are small, fragmented or under pressure, especially given the conservation status of *Z. muelleri* (de Lange et al. 2018). It is likely to become increasingly difficult to justify the removal of seagrass from donor site, particularly if lower impact methods of restoration can be developed in Aotearoa New Zealand.

The use of naturally uprooted seagrass fragments for seagrass restoration has not been trialled in Aotearoa New Zealand but fragments of *Z. muelleri* (as beach-cast wrack or floating in the water column) commonly occur adjacent to seagrass meadows. Further work will be required to identify what environmental conditions are most conducive to the generation of these fragments for a particular location. An Aotearoa New Zealand study demonstrated that it is possible to store *Z. muelleri* fragments for at least one month prior to transplanting, with neither the length of time post-detachment nor the detachment season affecting rhizome or shoot growth six weeks after planting in aquaria (Cade 2016). However, the authors noted that further evidence is required to rule out a seasonal effect (i.e., fragments detached from plants experiencing high growth rates and productivity during summer may have higher survival rates; Cade 2016). We recommend trialling this method in Aotearoa New Zealand because it offers a low impact alternative to harvesting wild plants and community engagement could be used to reduce costs and foster engagement.

Germinating seeds and transplanting seedlings also places minimal pressure on wild seagrass populations but requires more effort, and therefore has higher costs, than spreading seeds directly at a restoration site. Although germination in cultured conditions has not yet been demonstrated for *Z. muelleri*, it has been successful for other *Zostera* species, indicating this restoration method may be possible in Aotearoa New Zealand. Germination should be trialled if seeds can be found.

Propagating seagrass in a nursery offers the potential to provide a continuous supply of seagrass plants for low-impact restoration. Although propagation has not yet been trialled in Aotearoa New Zealand, several short-term experiments have demonstrated that it is possible to grow *Z. muelleri* in a nursery for up to six weeks (Ramage & Schiel 1998; Ismail 2001; Cade 2016; Zabarte-Maeztu et al. 2021). It is likely that propagation of *Z. muelleri* in a nursery would be possible, given its success for *Z. marina* (Tanner & Parham 2010). However, this method of seagrass restoration is likely to be the most expensive because it requires long-term cultivation in a nursery.

3.3.4. Transplanting

For plant-based active restoration, planting intact sods of seagrass is likely to be the most successful method of transplantation in Aotearoa New Zealand. Although Matheson et al. (2017) recorded equal success using unanchored bare-rooted plants, Zabarte-Maeztu (2021) found it challenging to relocate bare-rooted plants and stabilise them using staples. Transplantation of intact sods of seagrass is unlikely be possible when seagrass material is obtained from germinated seeds, naturally uprooted vegetive fragments or propagated plants. If plants require anchoring or shelter from environmental conditions or animals, then biodegradable or natural materials should be used.

Table 1. Summary of the pros and cons of different seagrass restoration methods for application in Aotearoa New Zealand.

Restoration method	Pros	Cons
Passive (facilitating natural recovery)	• Low impact • Could be low cost (e.g., removing localised impacts) • Potential for wider ecosystem benefits • Potential for wider human health benefits	• Could be slow • Could be very expensive (e.g., catchment-scale land management changes) with additional barriers to overcome (e.g., societal, political) • Unlikely in some situations (e.g., if seagrass loss was due to reclamation)
Active		
Seed-based		
Collecting, processing and storing seeds	• Low impact • Low cost • Moderate effort • Genetically diverse	• Untrialled in NZ • Seeds have only been identified in one location in NZ • Usually requires a storage facility
Spreading/planting seeds	• Low impact • Lower effort than planting clods or bare-rooted plants • Genetically diverse	• Untrialled in NZ • Seeds have only been identified in one location in NZ
Plant-based		
Harvesting wild plants from donor sites	• Demonstrated success in NZ	• Higher impact than other methods • High effort • Low genetic diversity • Unsustainable at large scales
Collecting naturally uprooted fragments	• Low impact • Low effort	• Untrialled in NZ (but see Cade 2016) • Source of material unreliable • Low genetic diversity • Requires storage facility

Restoration method	Pros	Cons
Germinating seeds	• Low impact • Supports genetic diversity	• Untrialled in NZ • Seeds have only been identified in one location in NZ • High effort • Requires a nursery • Potential for high cost
Propagating of seagrass plants	• Low impact • Potential to supply a continuous supply of seagrass material	• Untrialled in NZ • High effort • Requires a nursery • Potential for high cost • Low genetic diversity
Transplantation		
Planting clods or bare-rooted plants	• Demonstrated success in NZ	• Higher effort than spreading/planting seeds
Fixing plants to the seafloor	• More likely to withstand disturbance	• Addition of artificial material/litter into the environment (unless biodegradable/natural materials are used)
Use of structures	• More likely to withstand disturbance	• Addition of artificial material/litter into the environment (unless biodegradable/natural materials are used) • Structures could damage seagrass

4. DECISION MAKING FOR SEAGRASS RESTORATION

4.1. Decision making frameworks and considerations

To improve restoration success and to conduct restoration in an ethical manner, a suite of decisions needs to be addressed prior to commencing restoration activities. These decisions include identifying a suitable restoration location, determining what restoration method is most likely to succeed, identifying a source of donor material, and various environmental, ethical, financial and legal considerations. Although gathering the information required to make these decisions may take substantial effort, it is important for the reasons indicated above. Unfortunately, comprehensive decision-making processes are often neglected in seagrass restoration (Boudouresque et al. 2021).

Various frameworks have been developed to guide seagrass restoration decisions. In Aotearoa New Zealand, a decision-making framework was created for a *Z. muelleri* restoration trial in Whangarei Harbour (Schwarz et al. 2005). It included flow charts with the necessary steps for (1) identifying a restoration site and requirements for environmental enhancement, (2) conducting a trial, and (3) monitoring the success of the trial, with other considerations discussed in the report text. Frameworks developed overseas encompass additional aspects to the Schwarz et al. (2005) framework. Despite being developed for other seagrass species, many of these frameworks could be applicable to decision making in an Aotearoa New Zealand context bearing in mind the biological differences between seagrass species where relevant. For example, a comprehensive five-step program for was developed for *P. australis* restoration in Western Australia (Campbell 2002). Planning processes were broken into five steps (1) objective setting, (2) site selection, (3) transplant unit and technique, (4) habitat enhancement and (5) review of objectives. Campbell (2002) also emphasises the importance of cost:benefit analyses. Another example of decision-making frameworks is the code of conduct and decision-making strategy for *P. oceanica* restoration developed in France (Boudouresque et al. 2021). The code of conduct (Appendix 1) considers factors such as whether there are wild meadows nearby, the reason for restoration, experimental planting success, the risk to wild meadows and the larger seagrass management strategy for the region. The decision-making strategy (Appendix 2) outlines local- and regional-scale decisions relating to site selection as well as consideration of investment trade-offs. Boudouresque et al. (2021) also discuss in further detail ‘why’, ‘where’ and ‘when’ to conduct seagrass restoration.

Various overseas studies outline decision making considerations that are relevant to seagrass restoration in Aotearoa New Zealand. These either build on information in existing frameworks or outline topics not currently included in these frameworks. Considerations relating to site conditions and restoration methods, such as the importance of removing seagrass threats prior to replanting, proximity to (and recovery of) donor meadows, site-specific planting techniques and size of the restoration effort size are discussed by van Katwijk et al. (2016). Seagrass genetics are also important to consider, as seagrass populations may be adapted to local condition, and therefore sourcing seagrass seeds or plants from a nearby population (i.e., eco-sourcing) may be the best restoration approach (Jones et al. 2008). It is also important to have enough genetic variation in seagrass donor material to avoid genetic isolation and increase survivability under future conditions (van Katwijk et al. 2009; Tan et al. 2020). Future conditions at a restoration site are also important to consider (Tan et al. 2020 and references within), given that restoration success depends on whether seagrass (plants or seeds) can persist on a long-term basis (refer Section 4.2.1 for seagrass environmental requirements). For example, future impacts of climate change and other anthropogenic activities should be kept in mind when making restoration-related decisions. It is also important to work with traditional owners, as indigenous cultures have observed and managed natural environments over long time periods and can give important insights, observations, and interpretations relating to the state of these environments (Tan et al. 2020; refer Section 4.2.4). Other people with ties to an area may also provide useful local knowledge (Unsworth et al. 2019).

4.2. Information to support seagrass restoration decisions in Aotearoa New Zealand

Below we provide information to support decision-making processes for seagrass restoration in Aotearoa New Zealand. This includes information on the environmental requirements, natural recovery timeframes, genetics of *Z. muelleri* and the role that iwi could play in restoration initiatives.

4.2.1. Key environmental requirements

An important precursor to successful seagrass restoration is the identification of a suitable site for restoration. If seagrass does not currently exist in a particular location, it is important to establish the reason for its absence. Where environmental conditions are not suitable for the seagrass species, restoration is likely to fail. Simply relying on the historical distribution of seagrass meadows is insufficient because even if the source of stress has been removed, negative feedbacks may prevent the reestablishment of suitable environmental conditions (e.g., Kemp et al. 2005; van Der Heide et al. 2011). Below we outline current knowledge on the environmental requirements of *Z. muelleri* plants in Aotearoa New Zealand (Table 2). These environmental requirements can be used to check the suitability of a potential restoration site or to model areas that could have a high chance of restoration success (e.g., Davis et al. 2016; Kuusemäe et al. 2016; Vanglis 2016). In addition to these environmental requirements, seagrass need to be free of physical impacts that could cause damage to them. Environmental requirements of other stages of the lifecycle (e.g., seeds, seedlings) could differ; consideration of these in future work could be useful to facilitate sexual reproduction.

Seagrass meadows occur along the length of Aotearoa New Zealand (Turner & Schwarz 2006), suggesting that temperature is not a limiting factor for establishment. They generally grow in the intertidal zone, but can be found at depths of at least 8 m in some places (Clark & Crossett 2019). The upper depth distribution is determined by the risk of seagrass drying out at low tide, photoinhibition due to high light irradiance, wave exposure and associated turbidity, and reductions in salinity from freshwater input (Turner & Schwarz 2006). The lower depth limit is primarily governed by light availability (Hemminga & Duarte 2000). Light availability is a critical determinant of whether seagrass will grow and persist at a site (Dennison et al. 1993). Studies in Aotearoa New Zealand indicate that *Z. muelleri* requires at least 10–50 mol m^{-2} d^{-1} of photosynthetically available radiation (PAR) or water clarity of more than 0.5 m (Table 2). These high light requirements suggest restoration success will be more likely in intertidal areas. High sedimentation rates can also reduce light availability as well as cause direct mortality to seagrass via smothering (Sorensen 2020). Thus, restoration attempts may not be successful where sedimentation rates are more than 2–3 mm yr^{-1} (Table 2).

Seagrasses use nutrients in the sediment porewater and the water column to support their growth. Nutrient concentrations need to be sufficient for photosynthesis but not so high that they stimulate algal growth, outcompeting the seagrass for light (Turner & Schwarz 2006), or cause direct physiological effects that inhibit seagrass growth (Burkholder et al. 1992; van Katwijk et al. 1997; Table 2). Robertson and Savage (2020) suggest that once nutrient loadings surpass 50 mg N m^{-2} d^{-1}, seagrass extent will be severely reduced or absent. However, the resilience of *Z. muelleri* meadows to nutrient enrichment is highly site-dependent (Gladstone-Gallagher et al. 2018). To the best of our knowledge, nutrient thresholds have not been established for *Z. muelleri.* However, water column nitrate concentrations of 80–170 µL^{-1} have been shown to cause toxicity symptoms in a closely related species, *Z. marina* (Burkholder et al. 1994).

In Aotearoa New Zealand, *Z. muelleri* prefers muddy-sandy sediments that contain a low level of organic matter (Table 2). However, seagrass will not grow where nutrient porewater or hydrogen sulphide levels are too high (Table 2). Seagrasses can bioaccumulate metals and other contaminants from both the water column and sediment but limited information on toxicity thresholds is available for *Z. muelleri* (Matheson & Wadhwa 2012; Zabarte-Maeztu 2021).

Seagrass meadows are commonly found in sheltered areas. This distribution is likely controlled by the impacts that currents and waves have on meadow development and spread (e.g., transportation of seeds and vegetative propagules, uprooting of seedlings, and damage to mature plants) and the erosion, transport and deposition of sediment, nutrients or organic matter (Turner & Schwarz 2006). Previous restoration attempts in Aotearoa New Zealand have failed due to storm damage (e.g., Turner 1995; Zabarte-Maeztu 2021) demonstrating the importance of choosing a sheltered location for restoration, particularly in intertidal areas. Fonseca et al. (1998) caution against attempting seagrass restoration in areas where current speeds exceed 0.5 m s^{-1}.

Table 2. Optimal environmental conditions for *Zostera muelleri* plants based on studies in Aotearoa New Zealand (modified from Schwarz et al. 2005; Zabarte-Maeztu 2021). Seagrass also needs to be free of physical impacts that could cause damage.

Parameter	Too little	Suggested optimum	Too much	Source
Water depth		0.5–2.0 m		(Schwarz et al. 2005)
			> 8 m	(Clark & Crossett 2019)
Tidal exposure (for intertidal meadows)	6 hours	2–5 hours	0-1 hour	(Schwarz et al. 2005)
Wave and current exposure		Currents speeds < 0.5 m s^{-1}*		(Fonseca et al. 1998)
Light (PAR)		25 mol m^{-2} d^{-1}		(Matheson et al. 2017)
	< 2.4 mol m^{-2} d^{-1}	10-35 mol m^{-2} d^{-1}	> 70 mol m^{-2} d^{-1}	(Zabarte-Maeztu 2021)
	< 2.1 mol m^{-2} d^{-1}	10-50 mol m^{-2} d^{-1}	Unlikely	(Bulmer et al. 2016)
Sedimentation rate		1-2 mm yr^{-1}	> 3 mm yr^{-1}	(Zabarte-Maeztu 2021)
Catchment nutrient loading			> 50 mg N m^{-2} d^{-1}	(Robertson & Savage 2020)
Sediment mud content		8–23%		(Zabarte-Maeztu 2021)
		< 13%	> 13%	(Park & Donald 1994)
	< 5%	5–22%	> 23%	(Zabarte-Maeztu 2021)
	Clay/silt	Silt/sand	Sand	(Schwarz et al. 2005)
Sediment organic matter	< 0.5%	0.5–3.0%	> 5%	(Zabarte-Maeztu 2021)
Sediment porewater nutrients		Ammonium < 30 µM	Ammonium > 70 µM	(Zabarte-Maeztu 2021)
Sediment porewater hydrogen sulphide		0–9.0 µM	10 µM	(Zabarte-Maeztu 2021)
Water clarity	< 0.5 m Secchi	> 0.5 m Secchi		(Schwarz et al. 2005)
Water column nutrients		Ammonium 3–62 µL^{-1}		(Matheson et al. 2017)
		Nitrate 0.5–12 µL^{-1}	> 80 µL^{-1}*	(Matheson et al. 2017), (Burkholder et al. 1994)
		Phosphate 3–13 µL^{-1}		(Matheson et al. 2017)
Temperature	Frosts			(Schwarz et al. 2005)
Salinity	15 ppt	29 ppt	36 ppt	(Matheson et al. 2017)

* Data from a study on *Zostera marina* carried out in the United States of America.

4.2.2. Timeframes for natural recovery

Understanding timeframes for natural seagrass recovery after a disturbance is important for developing realistic restoration goals (Vaudrey et al. 2010; Cunha et al. 2012), as is understanding the trade-offs between passive and active restoration and, evaluating the speed at which donor meadows will recover from harvesting. Seagrass recovery timeframes depend on the type of disturbance and timeframe over which it occurred (Turner & Schwarz 2006). Furthermore, recovery will only occur once environmental conditions are suitable (refer Section 4.2.1). In some cases, the potential for recolonisation is limited by dispersal capability (i.e., where there are no wild seagrass meadows nearby; Turner & Schwarz 2006). Other factors that may inhibit natural seagrass recovery include physical disturbance (e.g., from wind, currents) and biological stressors (e.g., herbivore grazing; Wendländer et al. 2020).

Natural recovery can occur over relatively large spatial scales (e.g., populations or meadows), for example, in association with catchment-level water quality improvements. Timeframes for natural recovery of *Zostera* species over larger scales (e.g., from a few to 1000s of hectares) are generally longer than five years and are often more than a decade, although more rapid recovery has been documented at smaller scales (e.g., 1000s of m^2) (Vaudrey et al. 2010; Matheson et al. 2017). As discussed in Section 3.1, it took more than 50 years before *Z. muelleri* meadows began recovering from an historical increase (and then decrease) of fine sediment inputs to Whangarei Harbour (Morrison 2021). Slowly improving conditions are suspected to be the reason it took this length of time. However, *Z. muelleri* cover increased relatively rapidly (from 1000 m^2 to 4600 m^2) over two years at a transplanting site in Whangarei Harbour (Matheson et al. 2017). This expansion largely took place over a relatively short period (from mid-spring to mid-summer). While some of it could be attributed to active transplantation (up to 4.5 m^2), much of it occurred beyond the original transplant site (Matheson et al. 2017). Similar timeframes for large-scale recovery of *Z. marina* have been reported, with areas hectares in size taking more than five years (and often more than a decade) to recover (Vaudrey et al. 2010), although recovery of a smaller area (0.8 ha) of *Z. marina* occurred over three years (Vaudrey et al. 2010). In Tauranga Harbour (Aotearoa New Zealand), damage (presumably at scales less than 1000 m in distance; more likely to be 10s or 100s) to a seagrass bed caused by laying a pipeline was evident two decades later (Park 1999).

At smaller spatial scales (i.e., < 0.25 m^2), *Zostera* species are reported to have relatively fast natural recovery due to high growth rates, and in some cases the production of seed (Fonseca et al. 1998; Duarte et al. 2006). These scales are relevant to very localised areas of physical disturbance, such as those caused by the harvesting of plants (bare-rooted plants/clods) from donor plots. *Zostera muelleri* in harvested donor plots from Whangarei Harbour (10 cm^2, 15 cm deep) and Ihutai/Avon Heathcote estuary (0.22 m^2, 10 cm deep) recovered within nine and three months, respectively, likely through vegetative recolonisation (Matheson et al. 2017). Relatively rapid recovery at small scales has been documented by various other studies for both *Z. muelleri* (Peterken & Conacher 1997; Macreadie et al. 2014; Dos Santos & Matheson 2017) and *Z. marina* (Rasheed 1999; Zhang et al. 2020).

4.2.3. Seagrass genetics

Understanding genetic diversity and connectivity of seagrass populations is a critical step in seagrass restoration. Knowledge of both dispersal mechanisms and seagrass reproduction contributes to this understanding. For example, ocean currents can disperse seagrass fragments and seeds over relatively long distances, while vegetative growth expands from the existing seagrass meadow (Tan et al. 2020). In Aotearoa New Zealand, *Z. muelleri* typically has restricted gene flow between meadows (Sorensen 2020). Seagrass genetics differ between the North and South Islands and between the East and West coasts, with these patterns found to be associated with ocean currents (Jones et al. 2008). Some genetic separation was also detected between sites within estuaries, although *Z. muelleri* was found to be genetically similar at scales of metres and kilometres within these estuaries (Jones et al. 2008). Diversity between seagrass populations in Aotearoa New Zealand is likely related to dispersal of seagrass fragments rather than seeds (Cade 2016; Sorensen 2020). Fragments of *Z. muelleri* can be buoyant for a relatively long period, with the timeframe potentially dependent on water temperature (Stafford-Bell et al. 2016; Weatherall et al. 2016; refer Section 2.3.1). Hydrodynamic modelling tools could be used to track where seagrass seeds and fragments might disperse to (Jackson et al. 2021)

Given that seagrass populations in Aotearoa New Zealand are genetically diverse, it has been traditionally recommended that future restoration efforts use locally sourced seagrass material (Jones et al. 2008). However, increasing the genetic diversity in a population can also improve resilience, for example, to changing environmental conditions associated with climate change (reviewed in Tan et al. 2020). It may be beneficial therefore to include seagrass material from non-local (and potentially pre-adapted) sources or mix seeds from multiple sources to improve fitness, particularly for small populations with low genetic diversity. If *Z. muelleri* could be successfully germinated or propagated in land-based nurseries there could also be opportunities to selectively breed cultivars for certain traits (e.g., faster growth rates, greater tolerance to environmental stressors or disease resilience). However, the ecological and ethical implications of transplanting selectively bred plants into the wild would need to be considered first.

Although not strictly genetics-related, biosecurity also should be considered if seagrass material is being transferred between locations (e.g., the spread of introduced plants/animals or potential pathogens). For example, the fungal wasting disease *Labyrinthula* has been associated with declines in seagrass health in

Aotearoa New Zealand (Morrison 2021), although it can also occur in healthy seagrass meadows (Matheson et al. 2009; Clark & Crossett 2019).

4.2.4. Iwi considerations

Seagrass decline, and the loss of whānau/hapū/iwi resources and mātauranga (knowledge systems) associated with this habitat, is a particular issue for Māori across Aotearoa New Zealand. These meadows provide habitat and mahinga kai areas for taonga species and are a critical component of a healthy marine environment. To maximise seagrass restoration success, initiatives should be co-designed with mana whenua hapū/iwi to utilise their mātauranga and understand their desires and priorities for restoration. Drawing on mātauranga-a-whānau/hapū/iwi of historical seagrass distributions, and the values, uses, associations, and importance of seagrass meadows will help guide restoration efforts. This knowledge, which often extends further back in time than scientific observations, can be useful to counter 'shifting baselines', providing insight into what has been lost and aspirations for what could be restored. Local whānau/hapū/iwi may have witnessed the stressors that led to seagrass loss and could offer their view on whether these stressors have been sufficiently reduced or removed to allow seagrass to recolonise an area. Whānau/hapū/iwi may also hold mātauranga surrounding the flowering and seeding of seagrass in Aotearoa New Zealand or views on the cultural appropriateness of transferring seagrass plants between locations.

The Wai 262 claim relating to indigenous flora and fauna and Māori cultural intellectual property (Waitangi Tribunal 2011) may have implications for the propagation and transplantation of seagrass. The way that Wai 262 will be managed by the New Zealand government is still evolving (Wheeler et al. 2021). It will be important to develop a shared understanding of what this means for mana whenua in restoration locations. For example, whānau/hapū/iwi may have views on the cultural appropriateness of transferring seagrass plants between locations. Previous seagrass restoration projects in Aotearoa New Zealand have been initiated by, carried out in partnership with, and/or supported by, local iwi (e.g., Whangarei Harbour kaitiaki roopu, Reed et al. 2004; Ngāti Toa in Porirua, Matheson & Wadhwa 2012).

4.3. Seagrass restoration potential for Whakatū/Nelson estuaries

There are four main estuaries within the Whakatū/Nelson region: Te Tai o Aorere/Nelson Haven, Wakapuaka/Delaware Inlet, Waimeha/Waimea Inlet and Kokorua[11] Inlet. Historical decline in seagrass has been recorded (or inferred) for all these estuaries but the remaining seagrass meadows still cover relatively large areas (except for Kokorua). Below, we review changes in the extent of these seagrass meadows over time and the likely reasons for their decline. Based on this information

[11] Called Whangamoa Inlet by Gillespie (2013).

we provide a high-level assessment of the potential for seagrass restoration in these estuaries. However, more detailed information (e.g., Reed et al. 2004; Reed et al. 2005; Matheson & Wadhwa 2012) is required to formally evaluate whether environmental conditions (now and in the future) are suitable for seagrass restoration.

4.3.1. Waimeha/Waimea Inlet

Waimea Inlet is a large (3,910 ha) estuary with a heavily modified catchment dominated by exotic forest (33%) and pasture (20%; Stevens & Robertson 2017). This estuary has lost more than 60% of its intertidal seagrass since records began in 1990, with existing seagrass meadows now only covering 64 ha (2% of the intertidal area; Stevens et al. 2020). However, Waimea Inlet was already extensively modified by 1990, therefore, the true extent seagrass of loss is likely much higher (e.g., 15% of Nelson Haven is covered with seagrass). Most of this loss occurred prior to 1999 with seagrass extent relatively stable since then (Stevens et al. 2020). Low coverage of seagrass in Waimea is likely due to low water clarity forcing seagrass into areas of higher tidal elevation that are vulnerable to desiccation stress at low tide (Stevens et al. 2020). In addition, seagrass was lost following physical disturbance from the installation of a sewage pipe in 2011, while other areas of seagrass are impacted by fine sediment deposition (Stevens et al. 2020). The estuary is relatively muddy, largely as a result of historical sediment inputs, and has high nutrient and contaminant inputs in localised areas (Stevens & Robertson 2017; Stevens et al. 2020). As such, finding an area with suitable environmental conditions for seagrass restoration could be difficult.

4.3.2. Te Tai o Aorere/Nelson Haven

Nelson Haven is a large (1,242 ha) estuary adjacent to Nelson City. Despite its urban location, the catchment is dominated by native (54%) and exotic (33%) forest (Stevens & Robertson 2017). Nelson Haven has lost 50% of its intertidal seagrass since 1840. However, expansive seagrass meadows still exist, covering an area of 136 ha (15% of the intertidal area), and seagrass extent increased by 17 ha between 2009 and 2019 (Stevens & Forrest 2019b). Historical seagrass loss has been attributed to reclamation (114 ha of seagrass was lost between 1931 and 1979 when the port area was reclaimed) and changes in water quality (Gillespie et al. 2011a). However, recent increases in seagrass extent, an improvement in the extent of muddy sediments and a low eutrophication risk rating (Stevens & Forrest 2019b) suggest that environmental conditions in this estuary are becomingly increasingly suitable for seagrass colonisation. Active seagrass restoration could be used to accelerate this process. However, this will need to be weighed up against the costs of carrying out the restoration as expansion is likely occur through passive restoration alone. Nelson Haven has relatively good water clarity because it has few large freshwater inflows, and the main river has native forest in its upper catchment as well as a supply dam that traps sediment (Zeldis et al. 2019). Despite this, the estuary still suffers from elevated muddiness and local eutrophication and toxicity (Stevens & Robertson

2017). Future land-use changes (e.g., forest harvesting, road development, urban subdivision) could increase sediment loading to the estuary, making environmental conditions less suitable for seagrass.

4.3.3. Wakapuaka/Delaware Inlet

Delaware Inlet is a moderate-sized (335 ha) estuary that has been described as being 'relatively pristine' (Gillespie 2009), despite having an extensively modified catchment (42% exotic forest; Stevens & Robertson 2017). Intertidal seagrass extent has remained stable relative to 1983, covering 9.8 ha (2.5% of the intertidal area) of the estuary (Stevens & Forrest 2019a). However, subtidal seagrass extent appears to have declined since 1983 (Stevens & Forrest 2019a). There is some indication that seagrass loss may have occurred between 1983 and 2009, as Gillespie et al. (2011b) reported only 4.9 ha of seagrass in their survey (or 6.3 ha if patchier areas of seagrass were included in the total). However, these changes could be due to differences in survey methods (e.g., area mapped, or percent cover threshold required for an area to be classified as seagrass). Loss of seagrass has been attributed to increased muddiness near the Wakapuaka River Delta (Stevens & Forrest 2019a) as well as physical damage from vehicles in localised areas (Gillespie et al. 2011b; Šunde et al. 2017). Sedimentation is an issue for this estuary and forest harvesting in the catchment may exacerbate this (Stevens & Forrest 2019a). Eutrophication and toxicity also occur in localised areas (Stevens & Robertson 2017). Passive restoration of seagrass meadows damaged by vehicles in Delaware Inlet has a high chance of success if vehicle access is restricted or prohibited. Subtidal seagrass meadow restoration is unlikely to be successful unless sedimentation is significantly reduced.

4.3.4. Kokorua Inlet

Kokorua Inlet is a small (61 ha) estuary surrounded by exotic forest (45%), pasture (4%) and native forest (51%; Stevens & Robertson 2017). A recent survey of this estuary found no seagrass meadows (Stevens & Robertson 2015). Although seagrass has not been recorded here historically, Gillespie (2013) reported in his preliminary assessment of the environmental status of Kokorua Inlet that seagrass meadows were expected to be present in limited areas within the estuary. Stevens and Robertson (2015) suggest that the absence of seagrass is due to muddy sediments and low clarity arising from sedimentation. A lack of seagrass propagules may be also preventing seagrass reestablishment in parts of the estuary that are less affected by sedimentation (i.e., mud/sand areas near the mouth of the estuary). As such, Kokorua Inlet may benefit from active seagrass restoration efforts if areas exist with suitable environmental conditions. However, it is uncertain whether seagrass was historically present in this estuary and if so, the reason for its current absence. This information should be established before attempting seagrass restoration.

5. NEXT STEPS

Our review has identified several low impact seagrass restoration methods that could be trialled in Aotearoa New Zealand. However, there are several areas that would benefit from further investigation before applying these methods to a restoration project. We have listed these in the context of seagrass restoration in the Whakatū/Nelson region, but they could be applied elsewhere in Aotearoa New Zealand. Steps to initiate decision making for seagrass restoration in the Whakatū/Nelson region are also provided.

5.1. Trial of low impact seagrass restoration methods

- Seed-based active restoration:
    - Quantitative survey of seagrass seeds and flowering within the Whakatū/Nelson region.
    - Engagement with local whānau/hapū/iwi to see if they have mātauranga relating to seagrass flowering or seed production.
    - If seeds can be identified, germination of seeds and growth of seedlings in land-based nurseries and *in situ* within estuaries should be trialled.
    - Evaluate the potential costs associated with seed-based restoration approaches.
- Plant-based active restoration:
    - Identify favourable conditions and locations for the collection of naturally uprooted seagrass fragments in the Whakatū/Nelson region.
    - Trial growing naturally uprooted seagrass fragments in a cost-effective nursery system and compare their growth and survival to that of seagrass plants harvested from wild meadows.
    - Trial propagating seagrass plants in a cost-effective nursery system.
    - Evaluate the potential costs associated with nursery-based restoration approaches.

5.2. Steps to initiate decision making for seagrass restoration

- Develop a decision-making framework for seagrass restoration in the Whakatū/Nelson region. Engagement with local whānau/hapū/iwi will be important to ensure restoration guidelines are grounded in tikanga Māori and align with te ao Māori values.
- Gather information underpinning decision-making for seagrass restoration in the Whakatū/Nelson region:
    - The historic and current distribution of seagrass meadows in the Whakatū/Nelson region have been discussed in this report. Further information (particularly for Kokorua Inlet, where no records of seagrass exist) could be gathered by talking to local iwi and community members.

- This report provides some information on impacts to local seagrass meadows over time. Further information to determine whether the cause of seagrass loss has been removed or reduced could be gathered by conducting an in-depth review of available data as well as by talking to local iwi and community members. Evidence of natural recolonisation of nearby seagrass beds also indicates whether seagrass stressors have reduced in the general area.
- We also recommend conducting a quantitative survey of environmental conditions at potential restoration sites to ensure they are suitable for seagrass establishment.
- Collate or collect any other information required for decision making.

The decision-making framework for seagrass restoration in the Whakatū/Nelson region can be used to specify the location and methods for seagrass restoration. It is important to monitor the success of seagrass restoration, not only in terms of seagrass expansion, but also with regard to ecological functioning (e.g., by using proxies such as community composiiton, sediment grain-size and organic matter; see Turner 1995).

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8. APPENDICES

Appendix 1. A code of conduct for seagrass (*Posidonia oceanica*) restoration. Figure sourced from Boudouresque et al. (2021).

A code of conduct

To avoid techniques for transplanting *Posidonia oceanica* being used as a pretext for going ahead with the destruction of existing meadows, a code of good conduct has been proposed, at the request of the French Ministry of the Environment (Boudouresque et al., 1994; Boudouresque, 2001). The main principles are as follows:

1. The exact site and the biotope where the transplanting will be done must have been occupied previously by *P. oceanica*.
2. The causes of the disappearance of *P. oceanica* (pollution, trawling, anchorage, etc.) from the site where the transplanting will be done must have ceased to operate. Thus, before any transplanting is done, it must be demonstrated that the meadows or isolated clumps of *P. oceanica* that are nearest to the transplanting site have started a process of natural recolonization.
3. Transplanting must not be done near very extensive meadows. It is useless to add several dozen or hundreds of square metres (0.001 to 0.01 ha) to a meadow consisting of several hundreds or thousands of hectares.
4. Transplanting must not be done to compensate for the destruction of a meadow. To avoid such abuse, no transplanting must be done within a distance of 10 km from the site of deliberate destruction of a meadow (as part of coastal development) for a 10-year period.
5. However, transplanting on the exact site of the temporary destruction of a meadow may be possible, at least in the countries where the legal protection of *P. oceanica* is not opposed to this. This is the case when an open trench for an archaeological dig is covered over, or a pipe (or cable) crossing a meadow is buried.
6. With the exception of this special case above (point 5), any transplanting of *P. oceanica* must be undertaken after experimental transplanting of several hundred cuttings; scientific monitoring for at least 3 years must show that the experiment has been a success before a larger scale operation can be envisaged.
7. The removal of cuttings for transplanting must not endanger existing meadows. Therefore, it must be spread over a large area of meadow (less than 2 cuttings/m^2). The use of cuttings detached naturally, although giving less good results, or plantlets from seeds, can also be envisaged.
8. Lastly, transplanting must be done within an overall strategy of *P. oceanica* meadow management of the concerned region.

Appendix 2. A decision-making strategy for transplanting *Posidonia oceanica* and other seagrasses. The question–answer sequence first looks at the local level (the site of the anticipated transplanting) and then the regional level (a homogenous area, such as a bay). 'No' answers should lead to the project being abandoned. Figure and caption text sourced from Boudouresque et al. (2021).