Remediation Options for Southland Estuaries

Prepared for Environment Southland

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Frontispiece: Digging up eutrophic sediments in a Jacobs River Estuary Gracilaria field (photo: David Plew, NIWA).

Executive summary

In January 2019, Environment Southland (ES) requested a report from NIWA that investigates remediation options for Southland estuaries. Several estuaries across Southland are in a degraded or threatened state as a result of intensive land use, reclamation of land, contaminants and habitat loss. As part of its objective setting and Coastal Plan review processes, ES wishes to understand what can be done to 1) remediate already heavily impacted estuaries and 2) protect resilience in other estuaries in varied conditions including those still in good ecological health.

This report is provided for Environment Southland's scientists and policy makers. It investigates eight remediation options that cover a range of potential actions for Southland's estuaries and their marginal habitats. For each remediation option, it identifies the environmental issues addressed, the benefits and feasibility of applying the option, and the likelihood of success with respect to restorative targets. It identifies knowledge gaps, potential ecological side-effects and qualitative costs involved.

The list of remediation options to be addressed was finalised in a scoping meeting between the project team and ES staff at project inception (meeting notes are given in an Appendix of this report). The options pertain specifically to remediation in the estuary and its margin, meaning that options limiting catchment resource use (for example, limits on diffuse-source catchment contaminant loads) were not considered. However, it was in scope to indicate when an option's success was contingent on catchment remediation, including load reduction. Cases are also discussed when an option may be applicable to estuaries experiencing challenges due to excessive loads, whilst other options could be used to enhance the resilience of estuaries, or parts of estuaries, without excessive loads.

The work was undertaken as a desktop exercise. The results for each option are outlined below.

<u>Option 1: Removal of macroalgal biomass</u>

The environmental issue addressed by this option is macroalgal eutrophication (excessive growth) impacts in Southland Estuaries. The option described is physical removal of the macroalgae. The benefits of applying this option are to reduce smothering of benthic habitat, improve sediment health, remove noxious odour, and improve estuary amenity. It was concluded that routine removal of macroalgae in heavily eutrophic estuaries (i.e., New River Estuary (NRE) and Jacobs River Estuary (JRE)) was unfeasible, even to achieve marginal improvement in trophic state (Estuary Trophic Index (ETI) band rating changed from D to C). This was because of the enormous spatial extents and tonnages of eutrophic growths, and high likelihood of regrowth unless catchment loads were reduced. The situation is less severe or non-existent in other Southland estuaries examined (Fortrose, Haldane, Waikawa, Freshwater). For NRE and JRE, some partial solutions may be helpful (removal of overwintering young plants that provide the nursery for the following summer outgrowths) or targeting selected areas before they develop into persistent eutrophic areas (Gross Eutrophic Zones: where effects high algal biomass and depleted oxygen in sediments are highly detrimental). Incidences where natural, wind-driven events had removed growths were described, restoring ecological function in JRE. Destructive side-effects of physical removal were considered likely and removal costs are likely to be high. Algal growth experimental and modelling research would be beneficial to evaluate efficacy of the option.

<u>Option 2: Removal of degraded sediments</u>

The environmental issue addressed by this option is sediment eutrophication and muddiness and loss of attendant ecosystem services. The option described is physical removal of eutrophic sediments. The benefits of applying this option are to lessen muddiness, improve nutrient status, improve sediment oxygen and sulphide status for biota, increase clarity and improve estuary amenity. It was concluded that complete removal was unfeasible in heavily impacted estuaries, even to achieve marginal improvement in trophic state (Estuary Trophic Index (ETI) band rating changed from D to C), because of the spatial extent and depth of eutrophic sediments, requiring removal of huge tonnages of sediment (for NRE, JRE, Waikawa especially, but also Haldane). Based on overseas examples, hydraulic interventions (drainage channel deepening, low pressure sluicing) were considered possible, and could be most effective in estuaries with higher natural river flushing power, i.e., Fortrose, JRE, but would need to be continuously applied if estuary sediment loads were not reduced. This option could interact synergistically (positively) with macroalgal removal because of the sediment-trapping tendency of macroalgal beds and the nutrient-enriched quality of fine sediment. Research on hydraulics and ecological interactions (with macroalgae) would be beneficial. Very destructive side-effects of applying this option are considered likely and costs are likely to be very high.

Option 3: Restoration of seagrass beds

The environmental issue addressed by this option is loss of large amounts of seagrass beds and attendant estuary habitat and ecosystem services that has occurred in Southland estuaries (NRE, JRE) over recent decades. The option described is restoration of seagrass beds by transplanting. The benefits of applying this option are to improve habitat for important ecosystem components such as young fish, and to improve estuarine biogeochemical ecosystem services (e.g., denitrification (gaseous N loss), nutrient sequestration). Based on information from other New Zealand and overseas sites, the option is considered feasible if other estuary conditions are remediated (sediment deposition and nutrient concentrations reduced) in Southland estuaries that are currently eutrophic. Estuaries in moderate health with historic seagrass beds could also benefit if their habitats are appropriate, however all mainland Southland estuaries are exhibiting either high nutrient concentrations or muddiness (or both) so it will be necessary to carefully consider the within-estuary distributions (zonation) of nutrient concentrations and muddiness in these estuaries when planning outplanting programmes. Experimental out-planting research would be beneficial. There is potential for synergistic interactions with cockle restoration (next option) with no detrimental side-effects. Costs could be relatively low.

Option 4: Cockle bed restoration

The environmental issue addressed by this option is loss of cockle beds in Southland estuaries through eutrophication / sedimentation effects, with impacts on supply of kaimoana and attendant ecosystem services provided by healthy macroinvertebrate communities. The option described is restoration of these beds by transplanting. The benefits of this option are re-establishing kaimoana resources, improving natural amenity and helping to restore natural ecosystems. Based on information from other New Zealand and overseas sites, the option is considered feasible if other estuary conditions are remediated (sediment muddiness and oxygen / sulphide conditions improved) which, in the case of eutrophic Southland estuaries, would require reduced catchment loads. Reseeding success is unlikely in parts of estuaries that are currently highly eutrophic and muddy, and NIWA reseeding guidelines indicate that sandy substrates in stable (non-highly sedimentary) habitats with good planktonic food supply and relatively high salinity are ideal. This would preclude the muddy backwater areas of NRE, JRE, Fortrose, Waikawa, and Haldane estuaries as well as the more exposed Gross Eutrophic Zones's in those estuaries. However, all those estuaries could be considered for reseeding if and where their fine-scale habitats are appropriate. Experimental cockle bed restoration research would be beneficial, along with gathering information on historic cockle populations in Southland estuaries, to determine prospective reseeding sites and provide a baseline for historic cockle abundances. Destructive side effects are unlikely. Cockle bed restoration has potential for (mainly) positive synergistic interactions with seagrass bed restoration (intersects with seagrass restoration option, above) and it has negligible detrimental side-effects and relatively low cost.

Option 5: Restoration of estuary riparian margins

The environmental issue addressed by this option is loss of estuary edge habitats (salt marsh) and their ecosystem services in Southland estuaries, through historic reclamation and drainage. The option described is restoration of these habitats by re-establishing natural state cover and improving habitat connectivity, through retirement of previously reclaimed or intensively drained land. Based on New Zealand (including Southland) and overseas examples, the option is considered feasible, subject to success of land retirement initiatives and de-reclamation efforts. Subsequent planting programmes are feasible at relatively low cost. Preliminary ETI rating criteria were used to describe extents of historic salt marsh habitat loss across Southland estuaries. In contrast to the unmodified Freshwater Estuary, all mainland estuaries have had significant areas previously drained and reclaimed, which could potentially be returned to salt marsh habitat (retired). The expected improvements that such retirement would achieve could be gauged using preliminary ratings presented in the report. NRE, Fortrose and Waikawa estuaries are most in need of restoration, in terms of losses since the 2000 NEMP baseline. NRE has the largest area of potential restoration, with over 1200 ha of low-lying land previously reclaimed. Restoration efforts must recognise future sea-level rise and potential for habitats to develop inland. Land retirement may be costly and legally complex, inferring the value of spatial planning research. There are no detrimental ecological side-effects, but there will be both positive and negative social side-effects. This option intersects with the Waituna Lagoon option, below.

<u>Option 6: Modification to Waituna Lagoon mouth opening regime to improve estuary resilience</u>

The environmental issue addressed by this option is estuary eutrophication and loss of estuary habitat and ecosystem services in Waituna Lagoon, a RAMSAR wetland of international significance and a place of great significance to Ngāi Tahu. The issue arises from a mouth-opening regime designed to maintain artificially low water levels that allow drainage of adjacent land. The option described is to promote healthy lagoon salinity, water quality and fish passage by implementing controlled closures of the lagoon mouth, and to reclaim or retire farm lands from low-lying margins of the lagoon to allow lagoon openings to be driven by ecological, cultural and recreational requirements rather than the pressure to drain land adjacent to the lagoon. This option is considered feasible through retirement of low-lying lagoon margin farmland to give greater freedom to prioritise the lagoon environment rather than land drainage when making decisions regarding openings. This option intersects strongly with the previous option (Restoration of estuary riparian margins). Retiring farmland will have economic and social implications that will need to be balanced with the environmental benefits of optimised openings and lagoon levels (including future sea-level rise), again inferring the value of spatial planning research. Several studies are available on control structure design and merits, and on merits of private vs public ownership of surround lands for Waituna Lagoon. The example of Te Waihora/Ellesmere (Canterbury) provides a useful case study for examining the social implications of this option.

<u>Option 7: Partial diversion of Oreti River from New River Estuary</u>

The environmental issue addressed by this option is NRE eutrophication and sedimentation driven by inputs to the estuary by the Oreti River, with attendant effects on eutrophication and sediment health attributes described in previous options. The option described is to divert ca 43% of the total volume of the Oreti River's flows to the sea (Oreti Beach) through a 2.2 km cutting to the coast (the volume limited by geomorphological considerations). This option was investigated using existing results of NIWA Delft3D / Delwaq water quality modelling results. The diversion would result in 50% reduction in suspended sediment load to NRE so there could be benefits in terms of improved sediment deposition rate for seagrass and macroinvertebrate communities. However, it would cause only 11-19% reduction in total nitrogen (TN), insufficient to change the NRE's very poor ETI trophic rating (Band D). Major risks associated with river diversion include increased salinity and downcutting/bank erosion in the river upstream of the diversion, silting up of the Oreti River / NRE downstream of the diversion and detrimental effects of sediment / nutrient dispersal on the Oreti Beach coast. Similar implications likely exist for the Mataura River/Fortrose system, should that river be wholly or partially diverted before entering the estuary. New Zealand and overseas case studies of river diversions and estuary interventions show some positive outcomes but also show that unexpected and detrimental side effects are common. Further studies and modelling would be required to investigate these risks. Costs would be very high with high risk of major detrimental side effects.

Option 8: Diversion of effluent from the Invercargill wastewater treatment plant from New River Estuary

The environmental issue addressed by this option is NRE eutrophication, driven by inputs of effluent from the Invercargill wastewater treatment plant (WTP) to the estuary. The option described is to remove the effluent from the estuary via an ocean outfall (or by other means, such as land-based disposal) to reduce the total nutrient load to the estuary. This option was investigated using existing results of NIWA Delft3D / Delwaq water quality modelling. Depending on the part of the estuary considered, the benefits would be reductions of 3-46% of the dissolved inorganic nitrogen (DIN) concentrations, and 19-76% of dissolved reactive phosphorus (DRP) concentrations in summer in NRE (when macroalgal growth is maximal). The reductions were greatest in the worst-affected (Gross Eutrophic Zone) areas of the estuary, where there would be up to 48% reduction in DIN and 76% reduction in DRP in summer. Benefits in terms of improved ETI trophic condition are likely (i.e., likely to shift to less eutrophic ETI condition band), depending on estuary zone, especially if combined with moderate improvement in catchment-derived loads. Further studies/modelling would be required to investigate this including the possibility that the large reductions of DRP concentration could be a major driver of macroalgal growth limitation. This option would require expensive infrastructure upgrades (for example, Invercargill WTP ocean outfall construction) but would have only beneficial side-effects within NRE, as were observed in the Avon-Heathcote/Ihutai example (Canterbury). Effects on the coastal environment of outfall effluent dispersal would need investigation.

The table below summarises the findings for each option including the environmental issue addressed, the benefits of applying the option, and the likelihood of success and feasibility, including synergistic interactions, side-effects, qualitative costs and needs for further investigation.

Summary of remediation options for Southland Estuaries. For each option, first discussed is the environmental issue addressed, the benefits potentially accruing by applying the option, likelihood of success, and feasibility of applying the option (including logistics, side effects, and qualitative cost) and necessity of catchment remediation for success of option. GEZ's: Gross Eutrophic Zones.

Option	Environmental issue	Benefits	Likelihood of success/feasibility	Catchment remediation
Removal of macroalgal biomass	Macroalgal eutrophication impacts	Reduce smothering of benthic habitat, improve sediment health, remove noxious odour, improve estuary amenity	Complete removal unfeasible, some partial solutions may work (winter removal, target selected incipient GEZ's). Destructive side effects of removal likely and costs likely to be high. Synergistic with fine sediment accumulation. Algal growth experimental and modelling research would be beneficial	In parts of estuaries with current or incipient GEZ's, removal would need to be continuously applied if catchment nutrient and sediment loads are not reduced, for estuaries exceeding trophic limits (NRE, JRE, Fortrose)
Removal of degraded sediments	Sediment eutrophication and muddiness, loss of ecosystem services	Reduce muddiness and sediment nutrient levels, improve sediment oxygen and sulphide status for biota, increase clarity, improve estuary amenity	Complete removal unfeasible, hydraulic interventions (drainage channel deepening, low pressure sluicing) possible. Could interact synergistically (positively) with macroalgal removal. Research on hydraulics and interactions with macroalgae would be beneficial. Very destructive side effects likely. Costs likely to be very high	Removal would need to be continuously applied, if catchment nutrient and sediment loads are not reduced for estuaries exceeding trophic limits (NRE, JRE, Waikawa, Haldane)
Restoration of seagrass beds	Loss of estuary habitat and ecosystem services	Improve habitat for important ecosystem components (recruits), improve biogeochemical ecosystem services (e.g., denitrification, nutrient sequestration)	Appropriate for estuaries with significant historical seagrass beds (NRE, JRE) that have lost them. Feasible if other estuary conditions remediated (sediment deposition and nutrient concentrations reduced). Experimental out-planting research would be beneficial. Potential for synergistic interactions with cockle restoration. No detrimental side-effects	For estuaries (or parts of estuaries) with historic seagrass beds, catchment load sediment and nutrient remediation would be required where seagrass trophic limits exceeded (NRE, JRE, Fortrose)

Option	Environmental issue	Benefits	Likelihood of success/feasibility	Catchment remediation
Cockle bed restoration	Loss of kaimoana and ecosystem services	Establish kaimoana sources, improve natural amenity and help restore natural ecosystems	Appropriate for parts of NRE, JRE, Waikawa, Haldane and Fortrose where fine-scale habitat conditions suitable. Reseeding success is unlikely in parts of estuaries that are currently highly eutrophic and muddy; NIWA reseeding guidelines indicate that sandy substrates in stable (non-highly sedimentary) habitats with good planktonic food supply and relatively high salinity are ideal. Experimental cockle bed restoration research would be beneficial, along with information on historic cockle distributions and abundance. Potential for synergistic interactions with seagrass restoration. Negligible detrimental side-effects, relatively low cost	Catchment load sediment and nutrient remediation would be required where conditions are insufficient for successful cockle bed restoration (muddy backwater areas of NRE, JRE, Fortrose, Waikawa, and Haldane estuaries as well as the more exposed GEZ's in those estuaries)
Restoration of estuary riparian margins	Loss of estuary edge habitat and ecosystem services	Regain natural ecosystems, habitats and ecosystem services, providing improved biodiversity, habitat connectivity, flood mitigation, sediment retention and carbon and nutrient uptake benefits	Appropriate for estuaries with significant historical riparian margins. NRE, Fortrose and Waikawa estuaries are most in need of restoration, in terms of losses since the 2000 baseline. Subject to success of land retirement and de-reclamation efforts. Sensitive to sea level rise and potential to move inland. Land retirement likely to be costly and legally complex. Planting programmes feasible at relatively low cost. Spatial planning research would be beneficial. No detrimental ecological side-effects but both positive and negative social side effects	For estuaries with significant riparian margin loss, retirement of land in catchment will be required

Option	Environmental issue	Benefits	Likelihood of success/feasibility	Catchment remediation
Modification to Waituna Lagoon mouth opening regime to improve estuary resilience	Waituna Lagoon eutrophication, loss of estuary habitat and ecosystem services	Openings prioritised for management of lagoon salinity, water quality and fish passage, with land drainage a lower priority	Retiring low-lying lagoon margin farmland would give greater freedom to prioritise lagoon environment rather than land drainage when making decisions regarding openings. Controlled closure / opening of the lagoon would prevent high salinities associated with prolonged openings and allow water quality control. Control structure design/location research is available. Land retirement likely to be costly and legally complex with both positive and negative social side effects	Retirement of land in catchment likely to be required to remove priority for land drainage
Partial diversion of Oreti River	NRE sedimentation and eutrophication	50% reduction in suspended sediment load to New River Estuary. 10-11% reduction in DIN concentration. Benefits in terms of improved ETI trophic condition negligible or minor	Reduction in Oreti River sediment inputs could interact positively with macroalgal, sedimentation and seagrass conditions, but nutrient reductions unlikely to improve NRE nutrient trophic state significantly. Major risks associated with river diversion including increased salinity and downcutting/bank erosion upstream of the diversion and silting up of the Oreti River/NRE downstream. Similar considerations apply for cutting the Mataura River to the sea before it enters Fortrose Estuary. Case studies of river diversions show that unexpected and detrimental side effects are common. Further studies/modelling would be required to investigate these risks. Synergistic interaction with Invercargill WTP diversion Option. Costs very high with high risk of major detrimental side effects	Reduction of catchment loads would augment benefits of diversion, but significantly improved trophic outcomes for NRE would require substantial catchment improvement. Similar considerations likely for Mataura / Fortrose estuary system

Option	Environmental issue	Benefits	Likelihood of success/feasibility	Catchment remediation
Diversion of effluent from the Invercargill wastewater treatment plant from NRE	NRE eutrophication	Reductions of 3-46% of DIN, 19-76% of DRP concentrations in summer in NRE. Up to 48% reduction in DIN and 76% reduction in DRP in GEZ's of NRE. Benefits in terms of improved ETI trophic condition likely depending on estuary zone	Realistic potential for improvement in estuary trophic state (likely to shift to less eutrophic ETI condition band), especially if combined with moderate improvement in catchment-derived loads. Would interact positively macroalgal, seagrass and cockle Options. Further studies / modelling would be required to investigate these possibilities. Synergistic with Oreti River diversion Option. Would require expensive infrastructure upgrades (for example, WTP ocean outfall construction). Would have only positive environmental side-effects within NRE	Moderate reduction of catchment nutrient loads would augment benefits of diversion, potentially leading to improved trophic outcomes for NRE

1 Introduction

In January 2019, Environment Southland requested a proposal from NIWA to investigate remediation options for Southland estuaries. Several estuaries across Southland are in a degraded or threatened state as a result of intensive land use, reclamation of land, contaminants and habitat loss. The National Policy Statement for Freshwater Management (NPSFM) directs regional councils to set state objectives for freshwater bodies in their region and set limits on resource use to meet those objectives. As part of the objective setting process, Environment Southland (ES) has included estuaries in the freshwater management units and therefore state objectives will also be set for estuaries, with the aim to 'maintain or improve' estuary state. Also, recent ES policy-based workshops have related to the Coastal Plan review process, where questions arising have asked what can be done to 1) remediate already heavily impacted estuaries and 2) protect resilience in estuaries in varied conditions including those still in good ecological health.

Part of the objective setting process will be to set achievable state objectives, in consultation with the community; this will require possible outcomes of state to be presented for different limit options. Several estuaries across Southland are in a state of poor ecosystem health and therefore remediation will likely be needed, in addition to limiting resource use. At present, ES has little information on remediation options that would be suitable for Southland estuaries. This is likely to be pertinent to the conversations around the objective setting process and what will be required to achieve a desired objective.

This report is provided for ES's scientists and policy makers, for managing the current situation wherein several Southland estuaries are degraded or threatened, or where advice is needed on maintaining resilience of estuaries currently in good ecological health. It outlines what is known about estuary remediation including the challenges of legacy effects, in addition to identifying key knowledge gaps. It is structured as an 'Optioneering' initiative, to investigate a range of potential remediation actions for Southland's estuaries and their marginal habitats, to identify restorative targets, and advise on their viability.

Remediation can focus on two areas: the catchment and the estuary and its margin. The objective setting process will address the limiting of resource use within in catchment, therefore, this report specifically pertains to remediation in the estuary and its margin. Two levels of remediation are considered in the report, being those which will:

1. enhance the resilience of estuaries in varied conditions, including estuaries that are in good ecosystem health and require protection, and
2. improve estuaries in poor ecosystem health.

2 Methods

Following proposal acceptance and contracting in April 2019, in June 2019 a project scoping meeting was held with the project team and Environment Southland, to discuss and confirm the list of remediation options which would be assessed. The meeting discussed the implications and available information regarding the options, and approaches to the reporting. Minutes of the scoping meeting are provided in Appendix A.

The list of options agreed upon in the workshop and considered in the report are as follows.

1. Removal of macroalgal biomass.
2. Removal of degraded sediments.
3. Restoration of seagrass beds.
4. Cockle bed restoration.
5. Restoration of estuary riparian margins.
6. Management of Waituna Lagoon to improve estuary resilience.
7. Partial diversion of Oreti River from New River Estuary.
8. Diversion of effluent from the Invercargill wastewater treatment plant from New River Estuary.

For each option, the objective is to provide a high-level overview relevant to setting state objectives, to highlight the challenges faced and to consider relevant case studies and knowledge gaps. The work is undertaken as a desktop exercise. As noted above, the work focuses on remediation in the estuary and its margin, meaning that remediation options limiting catchment resource use (for example, limits on diffuse-source catchment contaminant loads) are not considered. However, it is in scope to point out where an option will likely not be viable without catchment remediation. It points out where an option is probably not pragmatic or has associated unacceptable side-effects and/or very high qualitative costs, and/or is unlikely to succeed without accompanying reductions in catchment-derived loads of nutrients and/or sediments. It is usually not prescriptive in terms of detailed advice, for example exactly where in a particular estuary an option should be applied, but it does provide underpinning information and reference materials which will assist in making such decisions.

The options considered involve several estuary ecological indicators that are in Tool 2 of the Estuary Trophic Index (ETI) (Robertson et al. 2017; Zeldis et al. 2017b; Plew et al. 2018a). Tool 2 allows, for certain key indicators of estuary ecological health, an assessment of the degree of remediation required to achieve a range of outcomes. An example would be “what extent of physical removal of macroalgae would be required to change a macroalgal eutrophication rating of ‘D’ (very high eutrophication) to ‘C’ (high eutrophication)?” Similar assessments are considered for other indicators, including removal of muddy, eutrophic sediment to maintain healthier sandy substrates, and restoration of riparian habitat. Similarly, for the options that are intended to reduce nutrient concentrations in Southland estuaries, the nutrient reductions are rated in terms of their projected effects on macroalgal eutrophication. This is done using resources of Tool 1 of the ETI (Zeldis et al. 2017a; Plew et al. 2018a; Plew et al. 2018b; Plew et al. 2019). This approach provides policy makers with a concise view of the targets remediation actions would need to meet, to achieve desirable ecological health outcomes.

3 Results and Discussion

Below we evaluate the options considered. For each option, we consider first the issue it addresses, and then describe its potential benefits and feasibility, including side-effects, knowledge gaps and research needs.

3.1 Removal of macroalgal biomass

3.1.1 Environmental issue

The issue addressed by this option is excessive growths of opportunistic nuisance macroalgae in Southland estuaries. The option described here is the removal of macroalgae from Southland estuaries to control its biomass and associated ecological impacts of eutrophication. The presence of opportunistic macroalgae is a primary symptom of estuary eutrophication (nutrient-driven enrichment). Opportunistic macroalgae are highly effective at utilizing excess nitrogen, enabling them to out-compete other seaweed species and, at nuisance levels, can form mats on the estuary surface which adversely impact underlying sediments and fauna, other algae, fish, birds, seagrass, and salt marsh. Decaying macroalgae can accumulate subtidally and on shorelines causing oxygen depletion and nuisance odours and conditions. The greater the macroalgal cover, biomass, persistence, and extent of burial of algal material within sediments, the greater the subsequent impacts. Macroalgal biomass and spatial extent are thus key indicators of degraded ecological health in Southland's lagoon, riverine and coastal lake estuaries.

Blooms of opportunistic macroalgae in New Zealand (NZ) estuaries principally contain species of green algae *Ulva* (this includes taxa formerly known as *Enteromorpha*) and *Cladophora*, red algae (*Gracilaria*), and brown algae (e.g., *Ectocarpus, Pilayella, Bachelotia*). These bloom-forming species are a natural component of intertidal ecosystems (Adams 1994) and they only grow to bloom proportions when nutrient levels are elevated (Sutula 2011) and where sufficient light for growth reaches macroalgal beds. Consequently, they generally only reach nuisance conditions in shallow estuaries, or at the margins of deeper estuaries.

The macroalgal growth response to nutrient loads generally increases with decreased tidal mixing (dilution) rates with the sea (Painting et al. 2007; Plew et al. 2019), either of the whole estuary (as is often the case for many NZ short residence time estuaries), or part of the estuary (e.g., a poorly diluted upper estuary arm where nutrient-rich muds accumulate), or in 'backwaters' where drifting suspended macroalgae can accumulate (e.g., Avon-Heathcote Estuary: (Bolton-Ritchie and Main 2005). There is some evidence this response may also be attenuated by the presence of fringing salt marsh, due to reductions in nutrient loading through processes such as denitrification (Valiela et al. 1997). Other factors that can influence the expression of macroalgal growth are the presence of suitable attachment strata, and physical and hydrodynamic conditions e.g., temperature (desiccation), fetch (wind driven waves), and currents (scouring) as seen in the Avon-Heathcote Estuary (Hawes and Smith 1995) where *Ulva* spp and *Gracilaria* have been the dominant nuisance algae.

Macroalgal blooms of *Gracilaria* commonly occur on muddy, intertidal flats in the mid- to upper estuary where salinity-induced flocculation and hydrodynamic sediment deposition is encouraged, there is little water motion, light is not limiting to growth, and exposure to elevated water column and sediment nutrient concentrations are greatest (Aldridge and Trimmer 2009; Longphuirt et al. 2015; Robertson et al. 2017). Such locations form the major *Gracilaria* production areas of Chile, New Zealand, Malaysia, Thailand, the Philippines, Indonesia and China (Santelices and Doty 1989). The success of *Gracilaria* in these relatively harsh environmental conditions of excessive muddiness, frequent fresh-water dilutions, high nutrient regimes, very low water motion, regular exposure to air, high temperatures, burial in sediment, and often anoxic or sulphide rich sediments, reflects the unique survival characteristics of this red alga. *Gracilaria* is now a dominant nuisance macroalgal species causing problems in Southland Estuaries.

High macroalgal cover (>50% cover) or density (>500 g wet weight m^{-2}) can lead to the development of gross eutrophic zones (GEZs) (Robertson et al. 2016a) in an estuary when they combine with high sediment mud contents and poor oxygenation. These areas are commonly associated with elevated nutrient and total organic carbon concentrations, and the displacement of invertebrates sensitive to organic enrichment and muds (Robertson et al. 2015; Robertson et al. 2016b). The areas most commonly first affected are natural deposition and settlement areas, often in the upper estuary. Because of the highly undesirable and often rapidly escalating decline in estuary quality associated with GEZs, even relatively small changes from baseline conditions should be evaluated as a priority.

In Southland estuaries, temporal trends of increasing GEZ extent indicate changes in catchment land-use management (i.e., reduced nutrient and sediment inputs) are likely to be needed. Figure 3-1 presents estimated TN estuary areal nitrogen loads for four Southland estuaries from 1996-2015 (Environment Southland data) compared to the threshold above which nuisance macroalgae problems have commonly begun to occur in NZ lagoon-type estuaries (SIDEs: Shallow, intertidal-dominated estuaries) (Robertson et al. 2017). It indicates that macroalgae issues are expected to be substantial in New River Estuary (NRE) and Jacobs River Estuary (JRE) but are not likely in Waikawa and Haldane Estuaries. Recent (2018) estimates for Fortrose Estuary (D. Plew, NIWA, pers. comm.) made using nutrient load data from ES indicate very high nutrient areal loads (2400 mg TN/m^2/d), indicating potential for macroalgal problems in Fortrose Estuary.

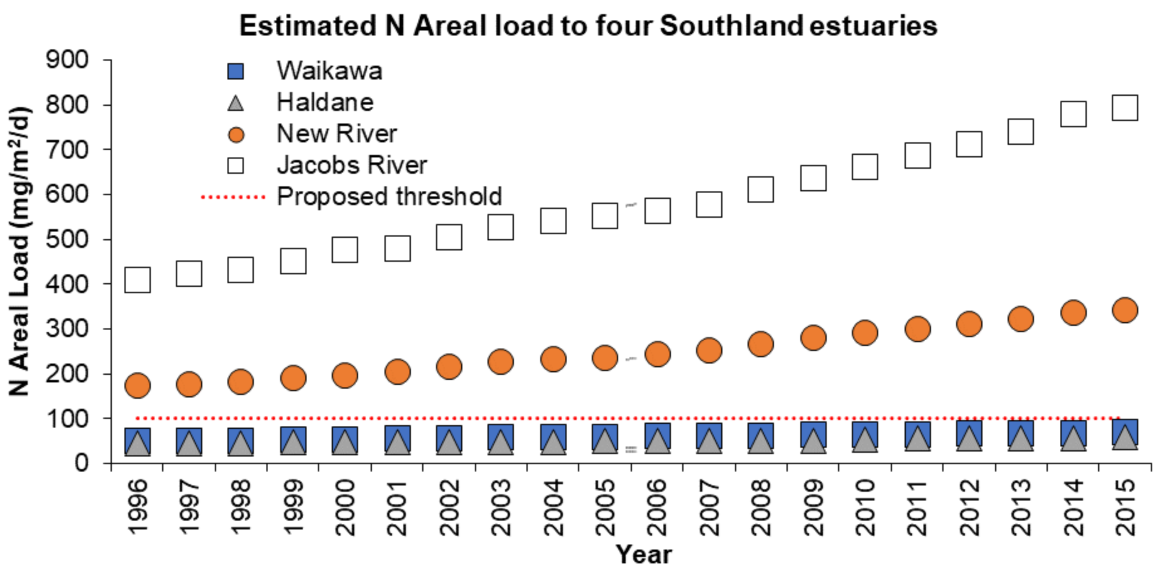

Figure 3-1: Estimated TN areal nitrogen load for four Southland estuaries from 1996-2015 (Environment Southland data). The dotted line is the threshold above which nuisance macroalgae problems have commonly begun to occur in NZ Shallow, Intertidal-Dominated Estuaries (SIDEs).

To assess the extent of macroalgae in Southland estuaries, monitoring commenced as part of development of the National Estuary Monitoring Protocol (NEMP) (Robertson et al. 2002) in 1999. This initially recorded the presence of macroalgae where it was a dominant feature in the estuary (e.g., percent cover >50%). This was augmented in subsequent monitoring with the Opportunistic Macroalgal Blooming Tool (OMBT) from the United Kingdom Water Framework Directive (WFD-UKTAG 2014), to more comprehensively track changes. This is a key part of ETI Tool 2 (Robertson et al. 2016a; Zeldis et al. 2017b), that enables quantifiable measurement of macroalgal cover and biomass. Alongside other metrics (e.g., sediment oxygenation and mud content), these have been used to assess the extent of GEZs. Figure 3-2 shows the relationship between Nitrogen (N) areal load and hectares of intertidal area classified as GEZs, and changes over time, examined in 25 cases for NZ SIDE estuaries from 2001-2016 (Robertson et al. 2017). Results show a significant and expanding problem of macroalgal-driven GEZs in estuaries which exceed the recommended nutrient load threshold.

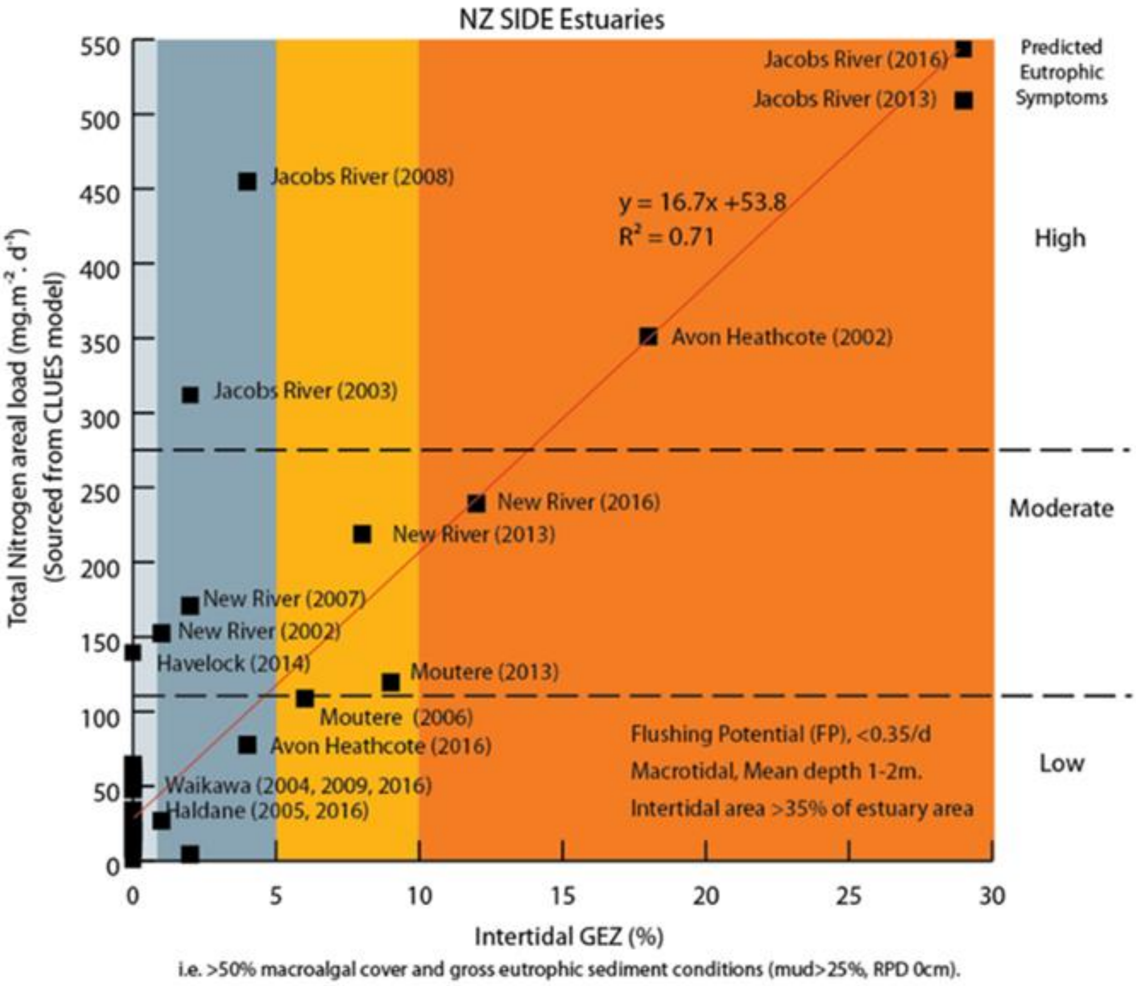

Figure 3-2: Relationship between Nitrogen (N) areal load and hectares of intertidal area classified as Gross Eutrophic Zones (GEZ's) for 25 cases in NZ SIDE estuaries 2001-2016 (Robertson et al. 2017). Colours ranging from light blue to orange indicate Estuary Trophic Index (ETI) trophic bands from A (minimal eutrophication) to D (very high eutrophication).

Recent work has scored NZ estuaries with respect to their susceptibility to macroalgal eutrophication based on potential nutrient concentrations and flushing times (Plew et al. 2019) using ETI Tool 1 methods (Zeldis et al. 2017a; Plew et al. 2018b). This work has identified NRE and JRE as being susceptible to very high eutrophication. Figure 3-3 shows that in NRE the extent of GEZ in the estuary was close to the ETI Band C/D (moderate/poor) threshold in 2001, but GEZs have expanded exponentially since that time, most developing in the upper reaches of the estuary in relatively sheltered deposition zones (Figure 3-4).

The extent and rate of growth of nuisance macroalgae in NRE (primarily *Gracilaria*) is unprecedented in NZ and has resulted in very significant and widespread estuary degradation including benthic smothering and sediment anoxia which have displaced most estuarine animals, shellfish and seagrass from GEZs. The presence of anoxic sediments also results in the release of nutrients previously bound in the sediments. These nutrients will predominantly be released in the form of ammonium, which is readily available to fuel macroalgal growth (Robertson and Savage 2018), thus establishing a cycle of increasing habitat deterioration that is likely to be difficult to reverse. In extreme cases sediment condition deteriorates to such an extent that macroalgae can no longer survive, a situation now evident in parts of NRE. The western Waihopai Arm now has such a large extent of rotting macroalgae, and high level of hydrogen sulphide in sediments, that there may be human health risks from any prolonged exposure in this part of the estuary. Elsewhere in Southland, significant macroalgal problems are also evident in JRE. Localised problems are also beginning to develop in Fortrose (Toetoes) Estuary, which is a river mouth-type estuary (SSRTRE: Shallow, short residence time tidal river). No significant macroalgal issues are currently evident in Waikawa, Haldane or Freshwater estuaries.

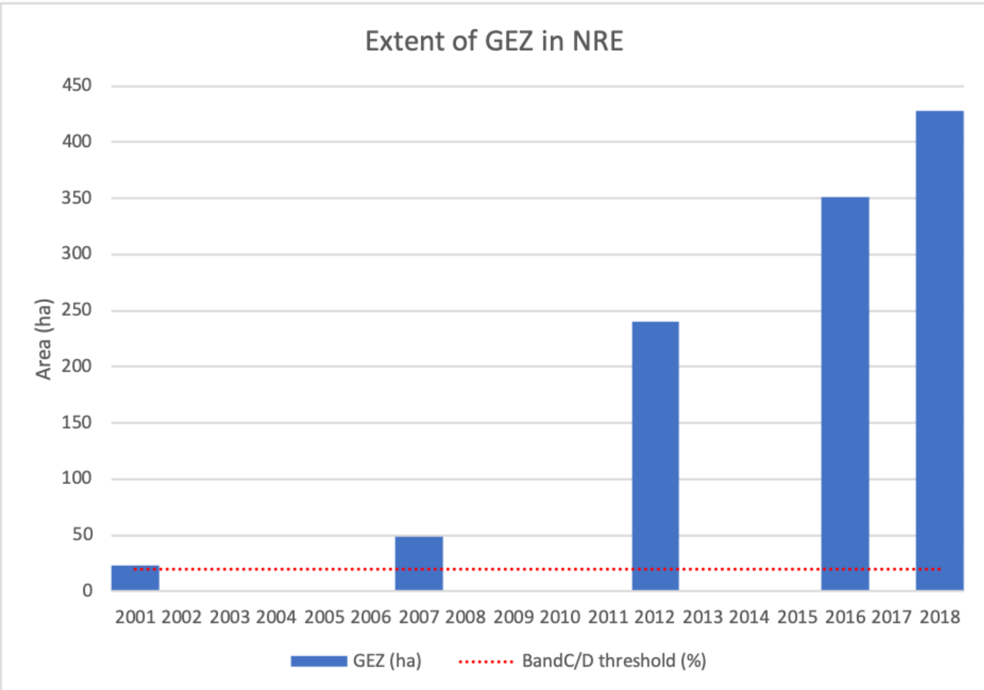

Figure 3-3: Temporal changes in the spatial extent of GEZs (ha) in New River Estuary, 2001-2018.

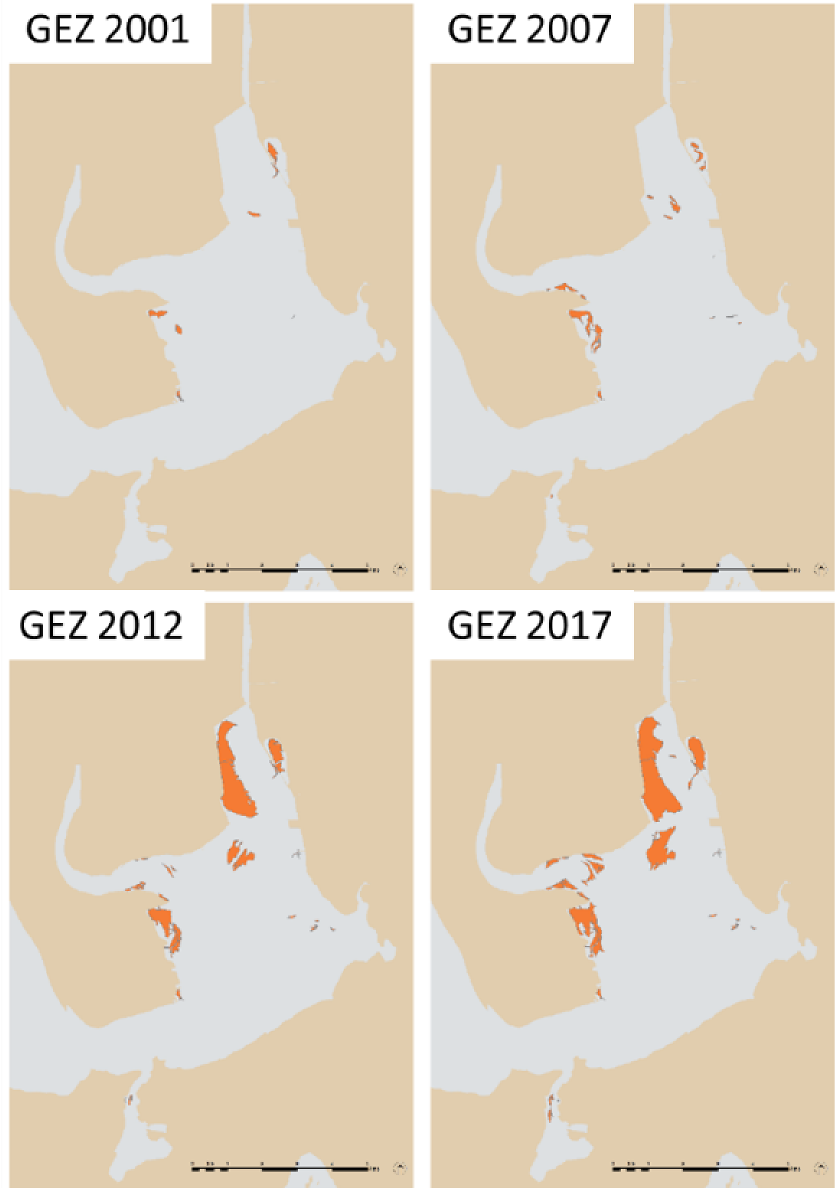

Figure 3-4: Location and extent of gross eutrophic zones in New River Estuary in 2001, 2007, 2012 and 2016.

For Southland Estuaries there is thus a strong correlation between nutrient load and GEZ expression (Robertson et al. 2017). Nationally, there is a strong correlation between estimated potential nutrient concentrations and macroalgal EQR[1] expression (Plew et al. 2019). These findings indicate that significant reductions in nutrient inputs would be required to prevent and potentially reverse macroalgal growth in NRE, JRE and Fortrose/Toetoes. However, because of the expected lag time between agreeing upon and achieving nutrient reductions, there may be potential benefits in the removal of excessive macroalgal growths from problem estuaries in the interim to mitigate against current impacts and avoid further degradation.

[1] Ecological Quality Rating (EQR), is a combined metric based on both macroalgal biomass and spatial measures. EQR is calculated from observations of % cover of available intertidal habitat, affected area with > 5% macroalgae cover, average biomass, and % cover with algae > 3 cm deep (Robertson et al. 2016a).

3.1.2 Benefits and feasibility

Employing the option of physical removal of macroalgae from Southland estuaries to control its biomass and associated ecological impacts could reduce the volume of algal material currently causing adverse effects through smothering and addition of organic matter to sediments. It would also likely reduce the trapping of fine sediments from the catchment that cause muddy, high nutrient conditions to accrue in estuaries and which contribute significantly to estuary degradation. At a localised scale, algal removal may prevent sediments from becoming strongly anoxic as a result of smothering and decay from excessive macroalgal accumulation. This would reduce the displacement of sensitive benthic communities including bivalves, crabs and other macrofauna that maintain sediment oxygenation through bioturbation of sediments and would favour the formation of desirable habitats such as shellfish and seagrass beds. Reducing the extent of macroalgal accumulation would also reduce the availability of reproductive macroalgal spores and may reduce the rate at which new algal beds become established. Decreased biomasses of macroalgae will also reduce noxious odours and will enhance human use (amenity) of the estuary.

Here we consider the feasibility of large-scale removal of *Gracilaria* from ES regional estuaries. Table 3-1 summarises key metrics used to assess macroalgal growth and related estuary condition including the extent of GEZs (ha and %), the extent of the estuary with biomass above 500 g/m^2 wet weight (which indicates potential adverse problems are developing), mean algal biomass, and the current state of the estuary in relation to its assessed ETI score and band rating (Robertson et al. 2016a; Zeldis et al. 2017b). The latter metrics are likely to inform National Objective Framework (NOF) criteria, with Councils required to avoid further degradation within bands and ideally improve the state of the estuary. As an exercise to evaluate the degree of remediation of macroalgal eutrophication required to achieve a range of ETI score outcomes graded from A (minimal eutrophication) to D (very high eutrophication), here we evaluate, as a minimum target, the extent of removal needed to achieve a rating of Band C. This is the minimum score that allows the estuary to continue to function in a healthy manner and prevent the loss of high value habitat like seagrass. The exercise assumes that:

- GEZ area was the sole criteria driving the ETI score
- Band C (≥5 to < 20 ha or ≥5 to <10% GEZ) is the minimum target for estuary condition
- removing excess macroalgae will directly reduce the extent of GEZ habitat
- there was no continuing growth of algae during the removal period.

We estimate the likely magnitude of change needed to move from Band D to Band C for NRE, JRE and Fortrose estaries by determining how much macroalgae would need to be removed from the estuary to change bands. While this is very much an oversimplification of needs and ignores legacy effects of current degradation, it nevertheless provides a starting point to assess the scale of potential change needed. Also given are predicted reductions in GEZ area to improve state to the next ETI Band for Waikawa Estuary (Freshwater and Haldane are currently in Band A). Note that TN areal load estimates are derived from NIWA's CLUES model run under default settings and are substantially lower than estimates calculated by ES (see Figure 3-1).

Table 3-1: Summary of key metrics used to assess macroalgal eutrophication in selected Southland estuaries.

Estuary	N Areal Load mg/m2/d (CLUES default)	Ha GEZ	% GEZ	OMBT EQR	Ha biomass >500g/m2 w.w.	Mean biomass of affected area (g/m2 w.w.)	Ha with >50% algal cover	Current ETI Score	Current ETI Band	Predicted GEZ reduction (ha) to reach improved ETI Band	Tonnes (w.w.) requiring removal to meet improved ETI Band	Tonnes/day to remove to achieve improved ETI Band in 1 year
NRE (2018)	259	428	13	0.284	678	3160	596	0.96	D	408	12893	35
JRE (2018)	474	144	30	0.245	146	3966	138	0.88	D	124	4918	13
Fortrose (2018)	1711	9	3.7	0.453	15	4155	16	0.75	D	2.4	101	0.3
Haldane (2016)	na	0	0	0.9	0	0	0	0.23	A	0	0	0
Waikawa (2016)	na	0.7	<1	0.75	0.7	509	3.8	0.57	B	0.3	2	0.004
Freshwater (2013)*	na	0	0	na	na	200	350	na	A	0	0	0

*biomass estimated

Based on the assumptions outlined above, Table 3-1 shows that to shift from Band D to Band C would require a reduction in GEZ area of 408 ha in NRE and 124 ha in JRE. The situation is less severe in Fortrose Estuary, minor in Waikawa Estuary and non-existent in Haldane and Freshwater Estuaries where macroalgae are either absent or present in much lower biomass.

If the mean macroalgal biomass in each estuary is multiplied by the GEZ area requiring reduction, a very rough estimate of the biomass requiring removal is 12,800 and 4,900 tonnes (T) wet weight (w.w.) respectively in NRE and JRE. To achieve this reduction in 1 year would require the daily removal of 35 T (w.w.) in NRE and 13 T (w.w.) for JRE, assuming no regrowth.

Fortrose/Toetoes Estuary remains much closer to the Band C/D threshold than NRE and JRE because strong river flows regularly uproot and flush macroalgae from the estuary. Consequently, prior to 2016 it had not exhibited accumulations of persistent macroalgae despite very high nutrient loads. However, persistent beds have established since 2016. To return the estuary to Band C based on GEZ area would require ca 100 T w.w. to be removed from 2.4 ha, (or ca 300 kg/d for 1 year).

These volumes are clearly very substantial and highlight the proliferation of excessive macroalgal growth in NRE and JRE, and to a lesser extent Fortrose/Toetoes. Furthermore, it should be realized that there is a high likelihood of rapid regrowth unless catchment nutrient loading is reduced and sediment nutrient sources are not remediated (Robertson and Savage 2018). This is examined further in the next Option (Removal of degraded sediments).

The side-effects of macroalgal removal include direct physical disturbance of estuary sediment from removal activities with effects on accompanying macrobenthos, as well as consequent changes to hydrological regimes from alterations to flow patterns through changed bed heights of intertidal flats (at present extensive growths restrict tidal drainage). Harvesting of macroalgae will also result in the release of fine sediments trapped in macroalgal beds reducing water clarity and potentially creating deposition impacts elsewhere in the estuary or on the coast. However, the release of fine sediments may alleviate current sediment anoxia by resuspending and aerating degraded beds and allowing bioturbating species to re-establish (see Removal of degraded sediments Option). This effect has been recently observed in JRE where the natural removal of dense macroalgal beds has occurred, likely as a direct consequence of wind driven waves washing into the Aparima Arm (Leigh Stevens, Salt Ecology, pers. obs.). The expansion of macroalgae in this part of the estuary over the past decade had resulted in previously firm sand-dominated sediments trapping fine soft muds that became progressively less oxygenated over time as a result of algal smothering and the increased cohesion in sediments. Following the wind-driven removal of macroalgae, the residual soft muds were also relatively quickly flushed from the estuary. This re-mobilisation of degraded muddy sediments has led to improvements in oxygenation, and a return of sediments to a healthier sand-dominated state, like the condition they were in prior to being smothered.

Other issues associated with large-scale removal will include effects on biota associated with macroalgal beds (predominantly amphipods and crabs) which would be impacted by algal removal. Also, disposal of macroalgae removed from the estuary would also require careful consideration.

Because of the very large volumes of removed macroalgae involved in NRE and JRE, and associated poor sediment conditions in GEZs, a selective approach to the removal of algae from key parts of the estuary may be the best way to optimise impact mitigation or prevent the expansion of problem areas. In other words, implement a strategy that targets high biomass growths in areas currently exhibiting low impacts for priority removal to prevent persistent problems from establishing, and existing degraded areas from expanding. For example, in NRE, estimates for 2018 (Stevens 2018a) indicated that ca 250 ha of dense macroalgal beds were growing on sediments that remained in relatively good condition, predominantly on the lower reaches of Bushy Point. In previous years such areas have quickly degraded and developed into GEZs because of smothering algae. Also, Table 3-1 indicates that in the less-impacted Fortrose Estuary, such a strategy of targeting incipient GEZ areas could be viable. Removing or reducing the cover of macroalgae from these areas has a relatively high likelihood of preventing GEZs from developing.

Another aspect of the removal of macroalgae is that regrowth could be slowed, or removal options could be more successful, if the overwintering biomass (reproductive source) is targeted. Hawes and O'Brien (2000) concluded that in the Avon-Heathcote Estuary the overwintering *Ulva* biomass was an important determinate of biomass of subsequent seasonal growth. Whether this is the *ca*se for *Gracilaria* in Southland Estuaries is a knowledge gap that could be explored further, potentially by experimental removals and algal growth modelling.

A possible option to consider is whether there is a potential commercial market for *Gracilaria* in NZ, as there is elsewhere in the world. At present, there is a moratorium on new approvals for the commercial harvest of *Gracilaria* in NZ, but it is a commercially valuable species, and at a local scale the managed ongoing removal from the estuary as a commercial enterprise may be feasible, potentially offsetting removal costs.

It is also worth considering the efficacy of *Gracilaria* harvest in relation to how much of the N-load is absorbed by the harvested biomass in to Southland estuaries. Duarte and Krause-Jensen (2018) described research showing the efficacy of nutrient removal by Chinese cultured algal harvesting relative to catchment areal nutrient inputs: 1 ha of seaweed aquaculture removed nutrients equivalent to the nitrogen inputs to 17.8 ha of catchment and phosphorus inputs to 127 ha of catchment. Using NRE as an example, these figures show that using *Gracilaria* removal as a mechanism for mitigating nitrogen loading to NRE would be ineffectual, given the vast size of the Oreti River catchment (*ca*. 350,000 ha). It can also be asked if complete *Gracilaria* removal would lessen the N availability in the estuary, by removing them as a potential source of regenerated nutrients upon their senescence and breakdown. However, calculations (not shown) show that the N masses stored in the *Gracilaria* biomasses given above in NRE (12,800 T w.w.) or JRE (4,900 T w.w.) are equivalent to only about 6 days of N load to those estuaries (based on the 2015 loads shown in Figure 3-1). Therefore, such removals would not constitute an effective control of N availability in the estuaries.

In any event, as described by Duarte and Krause-Jensen (2018), large-scale removals would need careful consideration to minimize damage to desirable habitats and ecosystem services. The side effects of algal removal, which have been briefly covered above, could be substantial but need to be considered in the context of the current adverse impacts that are occurring. It is reasonable to apply a Net Environmental Benefit Approach (NEBA) in assessing the likely effects where any active intervention may reduce the substantial impacts of current excessive macroalgal growth. In other words, while there will be localized sediment disturbance and resuspension because of algal removal, the longer-term benefits may outweigh these environmental costs. It is clear from the tonnage figures in Table 3-1, however, that large-scale removal would have very high monetary costs.

There have been anecdotal reports of removal of nuisance macroalgae from the Avon-Heathcote estuary in the early 2000's to lessen eutrophication impact (primarily from Rockinghorse Road area) by Christchurch City Council (J. Zeldis NIWA pers. comm.). High biomasses of algae persisted in the estuary through this period, however (Barr et al. 2019). Ultimately, significant remediation of macroalgal eutrophication was not achieved until there was diversion of Christchurch City wastewater out of the estuary to an ocean outfall (Bolton-Ritchie 2015; Barr et al. 2019; Zeldis et al. 2019). This indicates that substantial nutrient load reduction will be necessary in eutrophic Southland estuaries to allow their recovery from eutrophication.

In summary, macroalgal removal could reduce smothering of benthic habitat, improve sediment health, remove noxious odour, and improve estuary amenity. However, routine large-scale removal of macroalgae in heavily eutrophic estuaries (i.e., NRE and JRE) is unfeasible, even to achieve marginal improvement in trophic state. Other Southland estuaries (Fortrose, Haldane, Waikawa, Freshwater) have less severe problems. Some partial solutions may exist for parts of heavily impacted estuaries (removal of overwintering young plants, or targeting incipient GEZ's) or less-impacted estuaries, but the primary solution is to reduce nutrient loading. Physical removal is likely to have destructive side-effects and be expensive. Algal growth experimental and modelling research would be beneficial to evaluate efficacy of the option.

3.2 Removal of degraded sediments

3.2.1 Environmental Issue

The issue addressed by this option is degradation of the estuarine sedimentary (substrate) environment by fine sediment retention and eutrophication. The option described here is the physical removal of degraded sediments from Southland estuaries to reduce their ecological impacts. The estuary response to excessive fine sediment loads is an increase in soft muddy areas, elevated sedimentation rates, and high sediment mud content - particularly in mid-upper estuary (backwater) deposition zones. If nutrient loads are excessive, these muddy areas may have opportunistic macroalgal blooms and associated elevated organic content and sulphides, low sediment oxygenation (i.e., low redox potential) and a depressed condition of sediment-associated invertebrate communities (Robertson et al. 2015; Robertson et al. 2017). There is a positive feedback between fine sediment and nutrient loading and eutrophication, with fine, muddy sediments and high nutrient loading favouring macroalgal outgrowth, and macroalgae in turn trapping more fine sediment (Figure 3-5).

Figure 3-5: Photographs illustrating the change in sediment trapping and retention following the establishment of persistent *Gracilaria* beds at Bushy Point, NRE 2007, 2012 and 2016 (Robertson et al. 2017).

High muddiness results in reduced water clarity, with associated impacts on seagrass, fish and bird life, and human uses. Studies indicate that NZ estuaries with less fine sediment impact tend to be more favourable for healthy macrofaunal communities (Robertson et al. 2016b; Clark et al. 2019) and NZ estuarine sediments with less mud content have been shown to be resilient to eutrophication impacts once the stressor is removed (Zeldis et al. 2019).

The spatial extent of mud-dominated sediments in Southland estuaries has been monitored using the NEMP (Robertson et al. 2002) since 1999, with earlier studies providing context on changes prior to that time (Blakely 1973; Thoms 1981). Figure 3-6 shows the current percentage of intertidal mud-dominated substrate (excluding salt marsh) in selected Southland estuaries is well above the recommended ETI Band D rating (Robertson et al. 2016a) in four of the six estuaries shown. Temporal changes since 2002 (data not presented) indicate a relatively stable overall extent of mud-dominated substrate, reflecting historical impacts on the estuaries and highlighting that natural deposition zones are likely maintained to a large extent by hydrodynamic processes. Notwithstanding, in conjunction with the expansion of dense macroalgal beds in NRE and JRE, there have been corresponding increases in the extent of mud of 254 ha (34%) and 9 ha (5%) respectively. There have also been very significant increases in the rate of sediment deposition and accumulation within beds. Monitoring of vertical sediment accrual within the upper Waihopai Arm (NRE) from 2007-2018 recorded an average increase in the bed height of the estuary of 17 mm/year (ES data). While this increase is not due solely to fine sediment deposition and is due in part to the very large increase in organic material because of algal growth, this shift in bed height is very high compared to the rate of sediment accumulation in most NZ estuaries under natural state conditions which is well below 1 mm/yr (Townsend and Lohrer 2015), and recently modelled estimates for the greater NRE of 2.9mm/yr (Hicks et al. 2019).

Consequences of the combined rapid build-up of sediment and macroalgae also include the displacement of seagrass beds (58 ha of seagrass in the Waihopai Arm in 2001 reduced to less than 5 ha in 2018: see Restoration of Seagrass option, below). There have also been changes in drainage patterns with nutrient-rich surface waters now pooling at low tide on the intertidal flats rather than draining freely, and a significant degradation of the macroinvertebrate community, marked by a transition to almost exclusively surface dwelling algal scavengers and the loss of high valued infauna, particularly shellfish (Robertson et al. 2017).

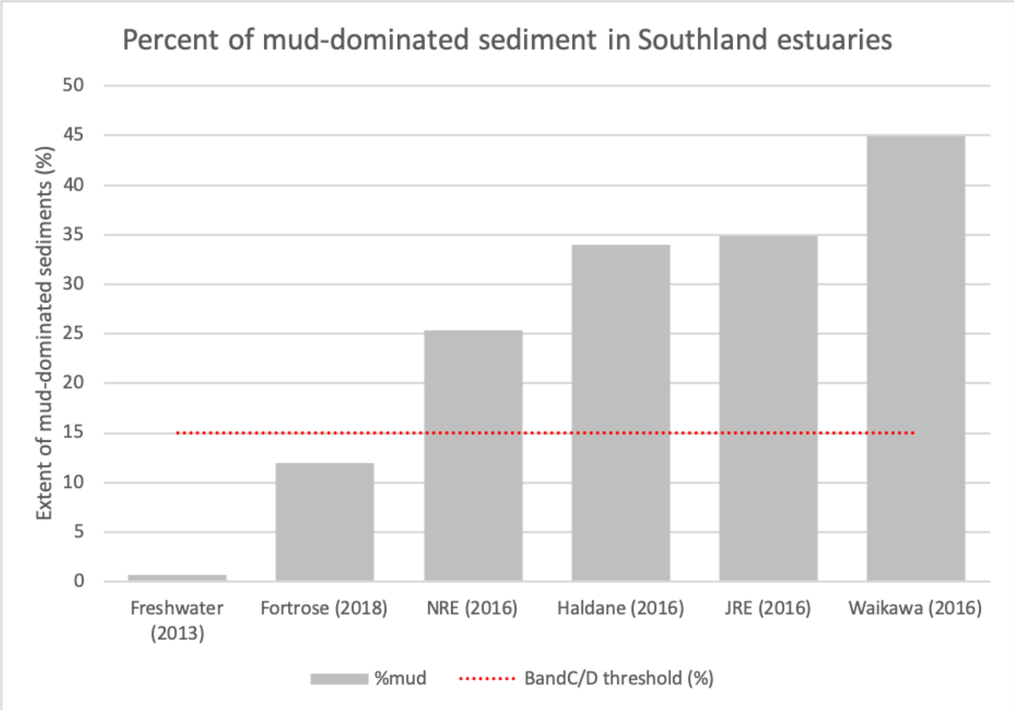

Figure 3-6: Spatial extent of mud-dominated intertidal substrate in selected Southland estuaries The red dotted line indicates the ETI Band C-D threshold for extent of mud-dominated sediments (Robertson et al. 2016a).

3.2.2 Benefits and feasibility

The key benefit of sediment removal is that reducing deposits of excessive fine sediment would facilitate the return of estuary sediments to a more natural state. This will help improve water clarity, create more favourable conditions for species sensitive to elevated mud content, like filter-feeding shellfish to live in (Robertson et al. 2015; Robertson et al. 2016b), increase interstitial sediment oxygenation potential, and reduce the retention of porewater nutrients in sediments (Robertson and Savage 2018; Zeldis et al. 2019) in areas where macroalgal growths are known to proliferate. Decreased muddiness will also enhance human use (amenity) of the estuary.

Table 3-2 summarises data on the spatial extent of mud in selected Southland estuaries, the percent cover in the intertidal zone (excluding salt marsh areas), and the current state of the estuary in relation to its assessed ETI rating criteria and band (data from Robertson et al. (2017)). The ETI measures are likely to inform NOF criteria with Councils required to avoid further degradation within bands, and ideally improve the state of the estuary. As a minimum target, estuaries should achieve a rating of band C if they are to continue to function in a healthy manner and prevent the loss of high value habitat like seagrass.

In terms of restoration, assuming that:

- Band C (≥5 to < 15 ha mud-dominated sediment) is the minimum target for estuary condition,
- removing excess mud will directly reduce the extent of mud-dominated habitat, and

- that mean mud depth is a nominal 25 cm across the estuary,

then the likely magnitude of change needed to move from Band D to Band C can be very roughly assessed by determining how much sediment would need to be removed from the estuary to change bands. While this is very much an oversimplification of needs and ignores legacy effects of current degradation, it nevertheless provides a starting point to assess the scale of potential change needed.

Based on the assumptions above and Table 3-2 the results show that to shift from Band D to Band C would require a substantial reduction in mud in Waikawa, JRE, Haldane and NRE. Fortrose/Toetoes, currently in Band C, would require a significant reduction to move to Band B. Freshwater Estuary is currently in Band A.

Using NRE as an example, if a nominal average mud depth of 25 cm is assumed, and a sediment density of 1,000 kg/m^3 is applied (both likely to be underestimates of the true values) then a very rough estimate of the mud requiring removal can be calculated. Table 3-2 indicates that the removal of a minimum of 762,500 T would be needed in NRE, with an additional 3,000,000 T if the deeper sediment present in the upper Waihopai Arm and Daffodil Bay is factored in (300 ha at an average depth of 1 m). To achieve the lower value reduction in 1 year would require the daily removal of 2090 T of sediment (150-200 truck-loads per day every day of the year). The upper estimate would necessitate *ca* 850 truck-loads per day every day of the year. Both assume no continuing deposition of mud-dominated sediment. Clearly these figures indicate there is currently a significant issue with mud-dominated sediments in NRE.

Table 3-2: Summary of key metrics used to assess muddiness in selected Southland estuaries and predicted change in spatial area to improve state to the next ETI Band.

	Intertidal area (excluding saltmarsh)	Intertidal Mud (ha)	Intertidal Mud %	Current ETI Band	Reduction in mud extent (ha) to reach improved ETI Band	Tonnes requiring removal to meet improved ETI Band if mean mud depth is 10cm and mud = 1kg/mm/m^2	Tonnes/day to remove to achieve improved ETI Band in 1 year
NRE (2016)	2944	747	25	D	305	762500	2089
JRE (2016)	498	174	35	D	100	248750	682
Fortrose (2018)	243	32	13	C	19	46250	127
Haldane (2016)	175	60	34	D	34	84250	231
Waikawa (2016)	563	253	45	D	169	422500	1158
Freshwater (2013)	638	5	1	A	0	0	0

The mechanical removal of the volume of sediment indicated above will require substantial effort and have associated side-effects like benthic dredging operations, in terms of both removal and disposal. The major side-effect of large-scale mechanical removal is the disturbance of estuary surface over many hectares. One option for reducing the existing impact of highly eutrophic sediments (those exhibiting extremely anoxic and sulphide-rich sediments) with less disturbance would be to facilitate incremental natural removal from the estuary by enhancing sediment resuspension on the outgoing tide. This might be achieved by increasing the flows of tidal water draining from eutrophic areas through removing surface macroalgae and creating additional or deepening existing drainage channels. Duarte and Krause-Jensen (2018) describe several overseas examples wherein such eco-hydrological interventions have led to improved water quality (see Partial diversion of Oreti River Option, below). It would need to be established through research, however, whether they would lead to increased sediment mobilisation.

Another technique which could be considered is low-pressure flushing, as used in oil spill remediation, to resuspend and flush surface sediments with minimal disturbance of underlying sediment. This essentially involves pumping and releasing seawater through multiple outlets along the upper edge of the shore to wash sediment to the low tide channels for discharge to sea. These processes mimic those naturally occurring in tidal river dominated estuaries like Fortrose/Toetoes where strong river and flood flows limit the settlement and accumulation of muds in intertidal areas. In this regard, it is possible to rank the hydraulic character of Southland estuaries in terms of the extent to which they are dominated by river forcing (Figure 3-7: N. Ward Environment Southland pers. comm.). On this scale, JRE would be between Fortrose Estuary and NRE, for example. The analysis also shows that winter and spring are the most energetic seasons in terms of flushing power.

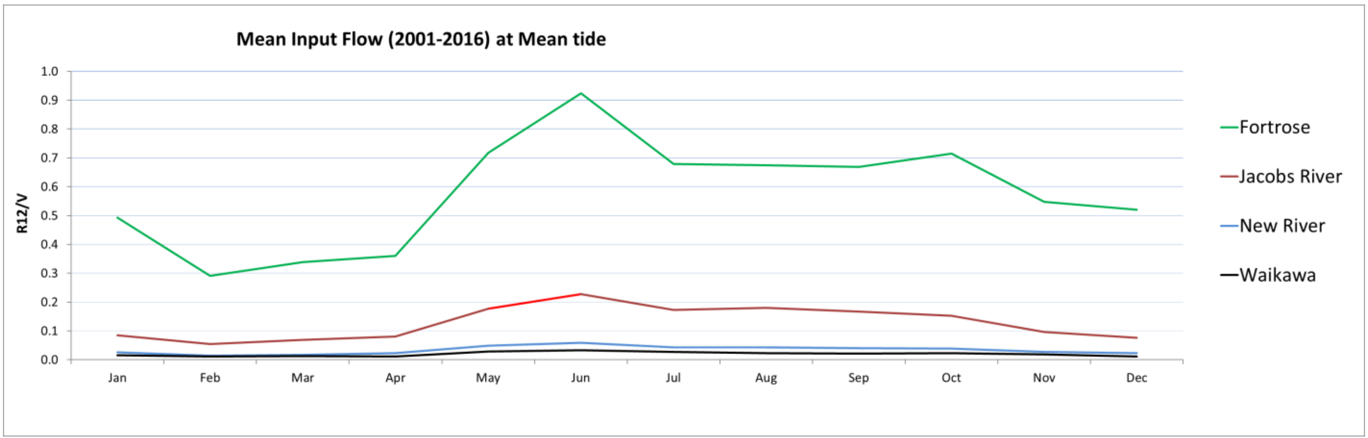

Figure 3-7: The balance of flushing power from river inputs to tidal inputs for four Southland Estuaries. The parameter shown is R12/V where R12 is the total mean volume of fresh water flowing into the estuary (of volume V) during a tidal cycle (12.4 h), calculated from monthly river flows into the estuary. Hydrodynamic processes of estuaries with a large R12/V ratio are dominated by river forcing.

A key consideration of such hydraulic interventions is that they do not address the root causes of the problem. Unless interventions are continued, estuaries will infill and revert to muddy conditions unless fine sediment loading is reduced. There is also a likelihood of return to muddy conditions unless nutrient loading is reduced or macroalgae are otherwise removed, because of the connection of macroalgal biomass accumulation with fine sediment accumulation, described above. This connection is a knowledge gap that could be tested in the field using experimental trials, wherein experimental removal of macroalgae could be tested in terms of its effects on underlying sediment muddiness.

As noted by Duarte and Krause-Jensen (2018), "…eco-hydrological interventions need be considered with care, using models to predict possible responses, to avoid negative experiences due to ill planned interventions". With sediment dredging or channelling, there is real potential for simply shifting the problem further downstream in the estuary or into the nearshore coastal environment. However, downstream impacts may still be more favourable for the environment that the existing situation. This is important in the context of the Resource Management Act: balancing benefits against impacts. There will be a balance of positive and negative environmental outcomes associated with options, and while some may seem very destructive, they could have net-positive environmental outcomes.

In summary, physical removal of eutrophic sediments would benefit estuaries by lessening muddiness, improve sediment nutrient, oxygen and sulphide status for biota, increase clarity and estuary amenity. However, complete removal is unfeasible in heavily impacted estuaries, even to achieve marginal improvement in trophic state, because of the huge tonnages of degraded sediment involved (for NRE, JRE and Waikawa Estuary especially, but also Haldane). Hydraulic interventions (drainage channel deepening, low pressure sluicing) may be possible, especially in estuaries with higher natural river flushing power (Fortrose, JRE), but would need to be continuously applied if estuary sediment loads were not reduced. This option could interact synergistically (positively) with macroalgal removal. Research on hydraulics and ecological interactions (with macroalgae) would be beneficial. Very destructive side-effects of applying this option are considered likely and costs are likely to be very high.

3.3 Restoration of seagrass beds

3.3.1 Environmental issue

The issue addressed by this option is the loss of seagrass beds in Southland estuaries with attendant loss of their ecosystem services. The option described here is restoration of seagrass beds in estuaries where such beds have been lost through eutrophication/sedimentation impacts. Seagrasses (*Zostera muelleri* in NZ estuaries) are vascular, rooted estuarine macrophytes that are key ecological components of historically healthy Southland estuaries (Robertson et al. 2016a). Seagrasses play an important role in NZ estuary ecology and are well-documented as keystone species that can reliably be used as indicators of estuary health. They provide high value habitat for a wide range of biota and their presence in good condition generally indicates low/moderate nutrient and mud inputs and good water quality. Seagrass beds are well-known as providers of key ecosystem services including wave attenuation, increased water clarity and denitrification and carbon sequestration (Reynolds et al. 2016; Duarte and Krause-Jensen 2018).

In some shallow NZ tidal lagoons, seagrass loss is associated with smothering by excessive macroalgal cover (in association with increased organic enrichment of sediments, low water clarity, poor oxygenation and increased muddiness) (Robertson et al. 2017; Stevens 2018a). Time-series surveys of NRE (Stevens 2018a) (Figure 3-8) and JRE (Robertson et al. 2017) have shown alarming die-offs of these valuable ecosystems in the last 20 years. While masked to some extent by seagrass beds in the lower well-flushed parts of the estuaries in good condition, seagrass losses have been significant in NRE, JRE and Fortrose/Toetoes since ca. 2002. In NRE there was 40% reduction in seagrass over the whole estuary from 2001-2016 (Robertson et al. 2017) with remaining cover just 56 ha (2% of the intertidal area) in 2018. In JRE there was a 31% reduction from 2003-2016 with remaining cover now just 24 ha (5% of the intertidal area). Fortrose/Toetoes had a 33% reduction from 2003-2016, with 0.2 ha remaining (0.1% of the intertidal area) in 2018 (note, however, that seagrass beds were never extensive in Fortrose). Haldane has no significant seagrass beds, and in Waikawa there has been a small reported increase in seagrass cover, most likely due to improved mapping resolution. The largely pristine Freshwater Estuary on Stewart Island has very extensive seagrass beds (315 ha, 55% of the intertidal area) which have not diminished over the same period, providing strong support that the recent seagrass losses apparent in NRE, JRE and Fortrose/Toetoes are occuring as a direct consequence of documented catchment intensification and estuary degradation.

By far the most extensive seagrass losses in Southland estuaries have come from areas directly affected by excessive macroalgal growth and the deposition of mud-dominated sediments. For example, Stevens (2018a) reported a 94% reduction in dense seagrass in the Waihopai Arm in NRE from 2001-2018, attributed primarily to smothering by fine sediments and nuisance macroalgal growths that initially established in 2007. Within JRE there was a >80% loss from the highly eutrophic Pourakino Arm between 2003 and 2016, and Fortrose/Toetoes showed similar percentage losses as a consequence of smothering by macroalgae and fine sediment in the northern embayment by Titiroa Stream (Robertson et al. 2017; Stevens and Robertson 2017).

Losses of seagrass beds have been documented in other parts of NZ. In Porirua Harbour (Greater Wellington) there has been an estimated 40% loss of seagrass beds relative to historical extent, with the largest losses (>30 ha) from inner areas of Pāuatahanui Inlet (Matheson and Wadhwa 2012). These inner areas receive runoff from the catchment via significant streams inflows, and are characterized by lower salinity, current speeds and wave activity, and are subject to higher suspended sediment and nutrient concentrations and sediment mud content than outer areas of the inlet where seagrass beds still persist (Matheson and Wadhwa 2012; Zabarte-Maeztu et al. 2019; Zabarte-Maeztu et al. in prep.). In contrast to Southland estuaries, macroalgal growths occur in the harbour but are not prolific or consistent (Stevens and Robertson 2016).

In Nelson Haven (Tasman), although seagrass has reduced significantly since 1840, (ca 126 ha, 50% loss) primarily due to reclamation, seagrass remains a prominent estuary feature (136 ha, 15% of the intertidal area) (Stevens and Forrest 2019). There has been an increase of ca 17 ha of seagrass from 2009 to 2019 which coincides with low macroalgal growth and a reported 78 ha reduction in mud extent since 2009. The data currently available do not allow the increase in seagrass to be attributed to a specific cause (Stevens and Forrest 2019). The estuary has relatively good water clarity/light climate due to having few significant freshwater inflows, and with the upper reaches of the main river in the catchment having a water supply area in native forest, and a supply dam (that traps sediment). Consequently, predicted catchment sediment inputs are low.

A threshold of 23% mud (silt + clay) content, above which seagrass (*Zostera muelleri*) does not occur, has been suggested by recent work in Porirua Harbour (Zabarte-Maeztu et al. 2019). A silt threshold in surficial sediments of 13% was indicated for Tauranga Harbour by Park and Donald (1994). Above this threshold, the authors considered that it was unlikely for seagrass to be present. Work in the USA (Chesapeake Bay) has observed the preferred sediment mud content for (the larger species, *Zostera marina*) seagrasses is 0.4%–30% mud content (Batuik et al. 2000) although Kemp et al. (2004) widened this range to 70%. However, it is also noted that narrower thresholds reported for Porirua and Tauranga Harbours do not concur with the documented extent of seagrass beds in the Waihopai Arm of NRE between 2001 and 2016, where seagrass beds were growing in sediments with measured mud contents ca 50-90%. Despite the high mud content these seagrass beds did not become displaced until they were overgrown with macroalgae (initially *Ulva* and then *Gracilaria*) (Figure 3-8) (L. Stevens, Salt Ecology pers. obs.) Robertson et al. (2017) and Stevens (2018c) report a similar situation in Westhaven Inlet (Tasman) with very extensive seagrass beds growing in sediments with a high (>25%) mud content for long periods (1990-2013) before undergoing a catastrophic reduction in extent from 2013-2016. The cause for that decline is unclear but appears unrelated to catchment land use changes or macroalgal impacts. These results indicate that mud alone is not necessarily the sole determinant of seagrass presence.

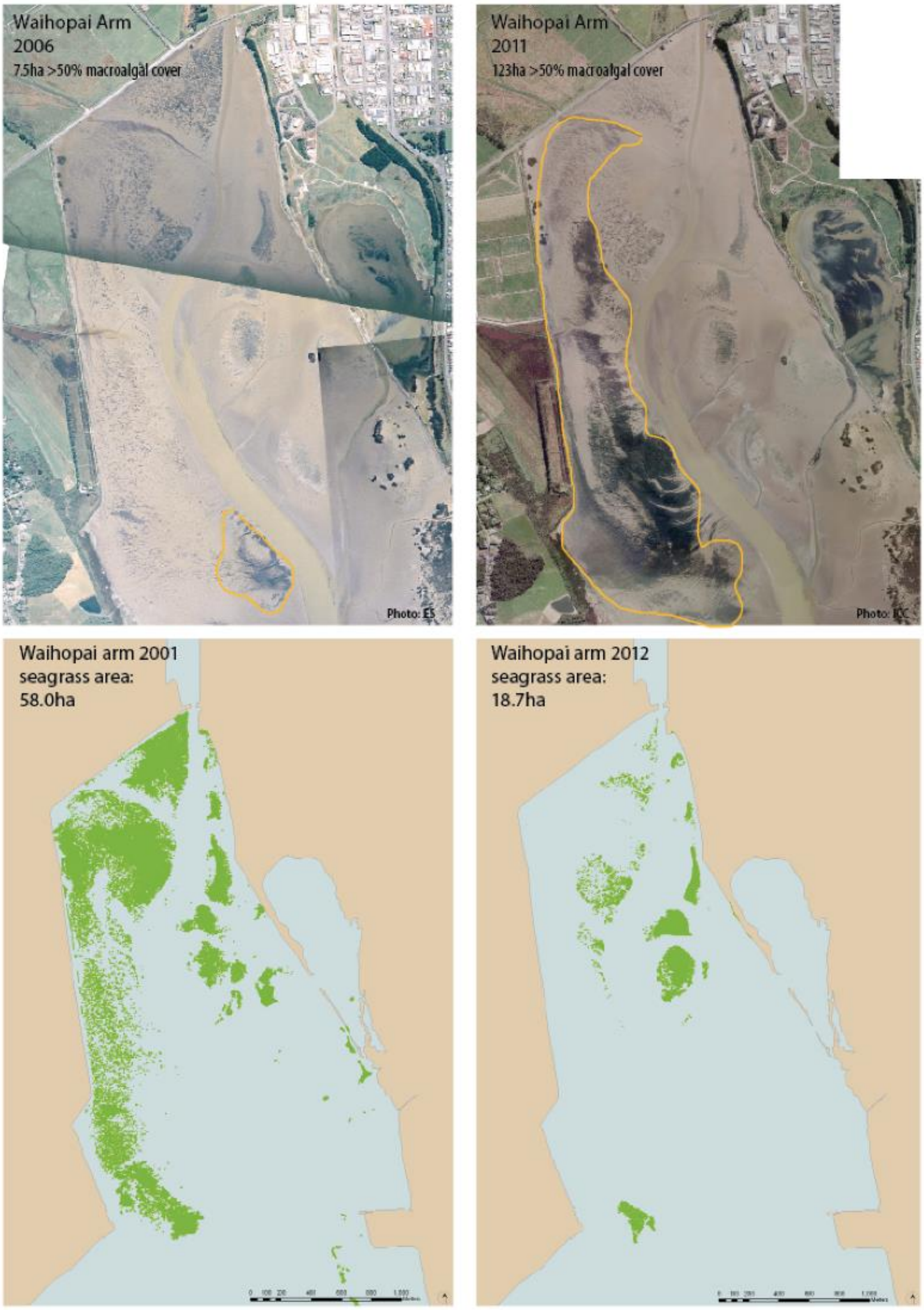

Figure 3-8: Aerial photos showing changes in macroalgal cover in the Waihopai Arm, New River Estuary, 2006 and 2011 (upper), with corresponding seagrass cover in the Waihopai Arm, 2001 and 2012 (lower) (Robertson et al. 2017). General coverage of nuisance macroalgae is indicated by the yellow line (upper, right).

3.3.2 Benefits and feasibility

Efforts for restoration of seagrass beds in NZ have included out-planting programmes and reductions of sediment and nutrient discharges. In Whangarei Harbour there have been positive outcomes from seagrass restoration efforts (Matheson et al. 2017b), upon reductions in sediment discharges associated with cessation of industrial sediment discharges and dumping of dredge spoil (see Morrison (2003) and Reed et al. (2004) for further details of the seagrass decline in this harbour). There has been substantial seagrass recovery through time, with anecdotal water quality improvement over two decades. Transplanting in 2008 and 2012 has worked well. Seagrass has been re-established at two former locations and there has been an overall estimated 40% recovery in the harbour (Matheson et al. 2017b).

In Porirua Harbour, seagrass transplanting trials in 2015, involving the Regional Council, NIWA, Guardians of Pāuatahanui Inlet and Ngāti Toa, proceeded on the basis that a preliminary assessment of seagrass restoration potential indicated a potentially suitable light climate for plant re-establishment at Ratio Point in the upper Pauatahanui Inlet (Matheson and Wadhwa 2012). However, the two small transplanting trials that were subsequently carried out were ultimately unsuccessful, with transplanted specimens not surviving more than 14 months (Matheson et al. 2017a; Matheson et al. in prep.). The more intensive monitoring of light climate over the two-year period of the trials (2015 to 2017) showed that the seagrass transplant site experienced seasonally lower light than the donor site in the outer harbour (with up to 20 days in winter with light below critical compensation levels (Matheson et al. in prep.). Subsequent (PhD) research in the harbour is exploring the interaction between seasonally low light levels and increased anoxia of sediments as a result of mud deposition (and resuspension) as causative in seagrass decline (Zabarte-Maeztu et al. 2019; Zabarte-Maeztu et al. in prep.).

In the Avon-Heathcote Estuary (Canterbury), work by Gibson and Marsden (2016) showed rapid improvement in seagrass extent (40%) following the diversion of Christchurch Wastewater Treatment Plant effluent from the estuary, over a four-year period post-diversion (2015 census) (Barr et al. 2019; Zeldis et al. 2019). This is similar to seagrass responses seen in other overseas estuaries recovering from eutrophication (e.g., Cardoso et al. (2007)).

There have also been reports (Duarte and Krause-Jensen 2018) that hydraulic interventions in lagoons to enhance circulation have facilitated the development of seagrass beds (described in Partial diversion of Oreti River Option). In addition, Reynolds et al. (2016) described very successful seagrass broadcast seeding operations in Virginia coastal bays (USA). It was estimated that the seagrass bed restoration achieved in 10 years was equivalent to more than 100 years of naturally-occurring seagrass areal coverage. It was estimated also that the restored seagrass beds denitrified 170 tons of nitrogen per year and sequestered 630 tons of carbon per year in the sediment, signifying both the success of restoration, and the value in terms of those ecosystem services.

In summary, efforts for seagrass restoration via transplanting efforts have been most successful when accompanied by other restorative activities in their estuaries, primarily nutrient and sediment load attenuation. For this reason, and considering the data on nutrient loads (Table 3-1) and muddiness (Table 3-2) showing all mainland Southland estuaries to sustain either high nutrient concentrations or muddiness (or both), it will be necessary to carefully consider the distributions (zonation) of nutrient concentrations and muddiness within Southland estuaries in planning outplanting programmes. It is likely that the muddy backwater areas of NRE, JRE, Fortrose, Waikawa, and Haldane estuaries as well as the more exposed GEZ's in those estuaries would be precluded. From a planning point of view, modelling presented later (Oreti River and Invercargill WTP diversion Options) for NRE provides a useful example of how potential outplanting sites within individual estuaries could be identified. Broadscale mapping of Southland Estuaries (Robertson et al. 2017; Stevens and Robertson 2017; Stevens 2018a) should also be consulted. As part of this, it would also be important to gather information on historic seagrass bed distributions in Southland estuaries, to determine accurate zonation of prospective outplanting sites. This would also provide a baseline for historic seagrass distributions in the estuaries. Such planning would help avoid areas that are less likely to support successful outplanting. Such seagrass outplanting would have negligible detrimental side-effects, and incur relatively low cost.

3.4 Cockle bed restoration

3.4.1 Environmental issue

The issue addressed by this option is the loss of cockle beds in Southland estuaries through eutrophication/sedimentation effects. The option described here is restoration of these beds by transplantation. Surveys of sites in Southland Estuaries (NRE, JRE, Waikawa: Robertson et al. (2017)) have shown associations of healthy shellfish beds (including cockles) with clean sandy sediments (their estuary sub-habitats A and B, but not C). These shellfish contribute important ecological functional and structural attributes, by acting to mix and irrigate sediments (bioturbation) enhancing nutrient and oxygen fluxes and by influencing the types of other macroinvertebrate species present. The species present include those both tolerant and not tolerant of muds and enrichment, indicating a functionally balanced community (Robertson et al. 2016b). In contrast, surveys have shown the muddy eutrophic sites were the least diverse, had dominant species characterised by high mud and organic enrichment tolerances, especially small, low biomass, surface scavengers or infaunal deposit feeders. Pipi, cockles and the wedge shell were all absent. The findings support the proposition of Robertson et al. (2015) and Braeckman et al. (2014) who showed that functional macrobenthic diversity and biogeochemical cycling was poor in cohesive, muddy sediments, and rich in fine sandy sediments at a number of marine subtidal sites.

3.4.2 Benefits and feasibility

Re-established, healthy cockle beds could support a source of kaimoana for Southland estuaries. Furthermore, re-established beds would help restore natural ecosystems, because of the ecological, functional and structural attributes they contribute, described above. These include synergistic interactions between cockle beds and seagrass in Kaipara Harbour (F. Matheson, NIWA, unpubl. data), where they frequently occur together. Cockles, and other suspension-feeding bivalves potentially have beneficial effects on seagrass beds including filtration of suspended particles (improving water clarity and light penetration) and excreting nutrients into the sediment, increasing nutrient availability (Reusch et al. 1994; Reusch and Williams 1998; Peterson and Heck 2001). Furthermore, they have symbiotic bacteria that can oxidise sulphide; a phytotoxin that typically accumulates in the organic matter rich sediments of seagrass beds (Van de Heide et al. 2012). Other benefits of bivalves include increased structural complexity of the beds and providing a refuge for invertebrate grazers that can control epiphytic loads and further enhance light supply (Orth and van Montfrans 1984; Peterson and Heck 2001). In turn, seagrass beds can create favourable conditions for bivalves by oxygenating sediments and the water column (through root oxygen release and leaf photosynthesis, respectively), reducing intertidal desiccation stress through shade and water entrapment, and providing a refuge from predation (Tu Do et al. 2011).

Some negative interactive effects are also possible. Bivalves depend on water flow for the supply of edible particles so reduction of water flow within seagrass beds may potentially limit food supply and growth (Irlandi 1997), and high bivalve densities may impair rhizome elongation (Reusch and Williams 1998). In Kaipara Harbour, denser seagrass patches (higher biomass) tended to contain fewer large cockles but had higher numbers of very small (<1 mm) cockles and wedge shells (F. Matheson and J. Hewitt, unpubl. data).

In Whangarei Harbour, research to develop a successful method for re-seeding cockles (tuangi) was carried out from 2003 to 2008, at sites adjacent to subsequent seagrass restoration trials at Takahiwai. Cockles (25-32 mm in diameter were seeded into 30 x 30 cm plots at two densities (222 and 832 individuals per m^2). The trial also tested the effect of caging but found that this did not enhance results. The project had some success (Cummings et al. 2007) and restoration guidelines were produced based in part on the trial results (see NIWA website: https://www.niwa.co.nz/our-science/freshwater/research-projects/all/restoration-of-estuarine-shellfish-habitat/active-shellfish-reseeding). The trials showed that 30% of re-seeded stock was retained after 12 months and overall abundance was increased compared to before. The re-seeding trial sites had sediments composed primarily of fine sand (>90%) and low organic content (<2.11%) (Cummings et al. 2007).

Cockles can occur in habitats with sediment varying from coarse sand to more than 90% fine sediment, and with salinities ranging from fully saline coastal water to 14 ppt (Marsden and Pilkington 1995). However, populations from sites with fine sediments, contaminants and/or low salinity may be smaller in size with reduced growth rates (Marsden and Adkins 2010). Sites experiencing reduced oxygen levels or increased sediment loads are unlikely to be suitable for cockle re-seeding (Marsden and Adkins 2010). For successful cockle transplantation (Adkins 2012) recommended “large scale, un-caged placement of 25-30 mm length cockles in the mid-low tide region of areas with stable, but not necessarily uncontaminated substrate, moderate salinity and temperature and with a reliable nutrient supply”.

In summary, the available information suggests that cockle reseeding success is unlikely in parts of estuaries that are currently highly eutrophic and muddy. The NIWA reseeding guidelines (see URL above) indicate that sandy substrates in stable (non-highly sedimentary) habitats with good planktonic food supply and relatively high salinity are ideal. This would preclude the muddy backwater areas of NRE, JRE, Fortrose, Waikawa, and Haldane estuaries as well as the more exposed GEZ’s in those estuaries. The appropriatness of Southland estuary habitat could be rated using the fine-scale criteria of Robertson et al. (2017) and broadscale mapping (Robertson et al. 2017; Stevens and Robertson 2017; Stevens 2018a). As part of this, it would also be important to gather information on historic cockle populations in Southland estuaries, to determine accurate zonation of prospective reseeding sites. This would also provide a baseline for historic cockle distributions in the estuaries.

Reseeding trials in Southland estuaries should be attempted initially on an experimental, research basis and their effectiveness could be scored using the NZ-AMBI macrobenthic indicator in Tool 2 of the ETI (Robertson et al. 2015; Robertson et al. 2016b). However, because this index integrates across macrobenthic communities, not single species, it would index general macroinvertebrate health responses to re-establishment of shellfish beds (i.e., it is not specific to cockle restoration). While this means it would not provide ETI Tool 2 rating bands specific for cockles, the NZ-AMBI would be a valuable indicator for assessing the ecosystem benefits of cockle bed restoration. Such reseeding will have negligible detrimental side-effects, and incur relatively low cost.

3.5 Restoration of estuary riparian margins

3.5.1 Environmental issue

The issue addressed by this option is the loss of estuary-fringing habitats and the ecosystem services they provide. The options described here are strategies toward restoration of riparian margin and salt marsh habitat by re-establishing natural state cover and improving habitat connectivity. This includes retirement of previously reclaimed or intensively drained land in estuary-adjacent areas, returning them to functional estuary margin.

Salt marsh (vegetation able to tolerate saline conditions where terrestrial plants are unable to survive) is important as it is highly productive habitat that naturally filters and assimilates sediment and nutrients, acts as a buffer that protects against introduced grasses and weeds, and provides important habitat for a variety of species including fish and birds. Salt marsh is also vulnerable to increased nutrient inputs, particularly nitrogen. Added nutrients stimulate salt marsh growth but, if excessive, may lower dissolved oxygen levels, change food web dynamics, alter community composition and stimulate the growth of algae and weeds (Deegan 2002; Pennings et al. 2002). In addition, although the Water and Soil Conservation Act (1967) and the Resource Management Act (1991) introduced wide-ranging controls over the destruction of salt marshes and other wetlands, since 1967 the legacy of detrimental salt marsh impacts remains visible in the undersized culverts below roads, railways and stopbanks that prevent adequate salt-water flow into these environments, and drainage and reclamation. The reduced salinity alters the plant community and facilitates the spread of the invasive species (e.g., reed *Phragmites australis*), which out-compete other salt marsh vegetation. Because of its lower habitat value for many species, biodiversity is reduced in areas where *Phragmites* becomes dominant. The combination of these factors has resulted in widespread and ongoing loss of wetlands, including salt marsh, throughout Southland including Awarua RAMSAR area (Robertson et al. 2019).

A salt marsh is classified as being the intertidal area of fine sediment that has been transported by water and is stabilized by vegetation (Boorman et al. 1998). Extensive salt marshes tend to be present if the coastal plain is gently sloping and wide (Friedrichs and Perry 2001). In general, marsh grasses cannot survive below mean tide level (the midway point between MLW and MHW) and are outcompeted by terrestrial plants above spring high tide (Pethick 1984). There are usually three distinct communities in NZ estuaries;

- a “salt marsh ribbonwood/rush” community consisting of a mix of salt marsh ribbonwood (*Plagianthus divaricans*) and rushes;
- a “rushland/sedge” community consisting of primarily searush (*Juncus kraussii*), oioi (*Apodasmia similis*) and three-square (*Schoenoplectus pungens*);
- a “salt meadow” community consisting of small herb-like plants including, sea primrose (*Samolus repens*), remuremu (*Selliera radicans*), glasswort (*Sarcocornia quinqueflora*) and in more brackish areas batchelor’s button (*Cotula coronapifolia*), leptinella (*Leptinella doica*), slender clubrush (*Isolepis cernua*) and arrow grass (*Triglochin striata*).

In many areas there is also a “weed” community at the upper tidal margin consisting of extensive patches of introduced iceplant (*Carpobrotus edulis*), gorse and various introduced grasses.

Salt marsh is one of the most productive environments on earth and provide important nursery grounds, wildlife habitat and nutrition for associated marine food webs. These dynamic ecosystems provide additional benefits for humans including flood and erosion control, water quality improvements, opportunities for recreation and for atmospheric gas regulation – salt marshes tend to be “carbon sinks,” since carbon dioxide is absorbed in the photosynthesis carried out by the prolific plant growth and subsequent burial.

Tidal salt marshes can respond rapidly to physical stressors, and their condition is often a dynamic balance between relative sea-level rise, sediment supply and the frequency/duration of inundation (Friedrichs and Perry 2001). However, if sea level rises too much, or the sediment supply or inundation through flooding is excessive, then the balance can be upset, and the salt marsh is lost, or its condition deteriorates. This balance varies between different types of estuaries but their response centres around how each reacts to sediment inputs and inundation (the latter is particularly important in the face of predicted accelerated sea-level rise through global warming).

Sedimentation within salt marshes is relatively high – approximately 5 times that of adjacent unvegetated flats (Eisma and Dijkema 1997) with most of the sediment depositing close to the sediment source (e.g., tidal creek) or spread evenly if sourced from the main body of the estuary. Sedimentation rates increase with grass stem density and because most NZ salt marsh plants tend to grow in dense stands e.g., searush (*Juncus kraussii*) and oioi (*Apodasmia similis*), sedimentation rates in NZ salt marsh are expected to be relatively high. The increase in sedimentation and subsurface plant growth results in an elevation of bed level for most NZ estuaries.

The vulnerability to inundation of salt marsh habitat in tidal lagoon estuaries of NZ is mainly from sea-level rise. There are two processes by which sea level can increase relative to the marsh surface: (1) sea level rises because of increases in the volume of the oceans, and (2) the marsh surface sinks (subsides) because of soil compaction and other geologic processes. Under current conditions, the majority of marsh environments tend to keep pace with sea level changes due to sedimentation and subsurface plant growth (Bartholdy 2000) and can respond rapidly to changing conditions. However, under an accelerated rate of sea-level rise it is expected that bed elevation through sedimentation will lag further behind relative sea-level rise and plant stress will increase until the plants die, the soil volume collapses, and the marsh becomes submerged. The vulnerability to salt marsh decline is expected to vary between estuaries with different tidal ranges. The most vulnerable are the microtidal estuaries (those with a tidal range of less than 2 m) because a relatively small increase in sea level or decrease in sedimentation rate can submerge the marsh vegetation to a level that is too stressful for survival. Conversely, when sedimentation is high, microtidal marshes will expand seaward more quickly than systems in higher tidal ranges. This is because it takes relatively little upward growth to significantly reduce submersion, causing available suspended sediment to be deposited further seaward.

Ecological ratings such as applied in Stevens (2018b) are used here to assess current state and indicate the degree of marginal habitat restoration required to meet ecological health criteria. This is relevant to decision making in the Regional Coastal Plan which is charged with estimating ‘Net Environmental Benefit’. While out of scope for the present report, the report’s narrative around ecosystem service benefits of healthy salt marsh/riparian condition will be useful in estimating benefit.

NEMP results obtained ca. 2000 enabled the extent of salt marsh in Southland estuaries to be defined with respect to that baseline period, with subsequent monitoring tracking changes since then. However, because the most significant salt marsh losses have generally occurred well prior to this time (Robertson et al. 2019), estimates have also been made of historical salt marsh extent based on aerial photographs or maps (where available), or expert judgement. Although there is considerable uncertainty with such estimates, they provide a context for the percentage of salt marsh that now remains. Subjective indicator thresholds used in previous broad scale mapping assessments (e.g., (Stevens 2018b) have been proposed to help guide assessment of current condition and past losses as follows:

Broad scale Indicator	Unit	Very Good	Good	Moderate	Poor
Salt marsh extent	% of intertidal	≥ 20	≥ 10-20	≥ 5-10	0-5
Historical salt marsh	% remaining	≥ 80-100	≥ 60-80	≥ 40-60	< 40

The following ratings have also been used to indicate the scale of changes in salt marsh extent: Slight 0 to <5%, Small ≥5 to <10%, Moderate ≥10 to <20%, and Large ≥20%. None of these preliminary ratings are currently included in NOF criteria and further work is needed at an estuary-specific level if they are to be used to determine estuary status or to set management targets. However, they do provide initial guidance as to the likely magnitude of change that may be needed.

Table 3-3 presents a summary of salt marsh extent and preliminary ratings applied for a selection of Southland estuaries. The results show that all estuaries except for Freshwater Estuary on Stewart Island are rated 'poor' for the percentage of historical cover remaining, reflecting large past losses due to land clearance, reclamation and drainage. Recent losses (since ca. 2000) range from 0-63 ha. However, these changes require some explanation. For NRE, the 63 ha loss in salt marsh includes the targeted eradication of 112 ha of the invasive cord grass *Spartina*, followed by growth of ~50 ha, predominantly herbfield growing on the residual *Spartina* root masses. This highlights that salt marsh will readily re-establish in areas where conditions are favourable for growth. In JRE, Fortrose/Toetoes and Waikawa, losses are a combination of natural erosion by wind-driven waves undercutting and washing out rushland, and small areas of ongoing drainage and conversion to pasture. There has been no significant change in Haldane, and no change is expected to have occurred in Freshwater Estuary.

Table 3-3: Summary of key metrics used to assess salt marsh in selected Southland estuaries and predicted change in spatial area to improve state to the next ETI rating Band.

Estuary	Saltmarsh area (ha)	Saltmarsh area (%)	Rating (Saltmarsh %)	Loss since NEMP baseline (ha)	Loss since NEMP baseline (%)	Estimated % historical remaining	Rating (% Historical remaining)	Saltmarsh Loss since NEMP baseline	Predicted increase (ha) to reach improved Rating
NRE (2013)	461	10.1	Good	63	12	<40*	Poor	Moderate	449
JRE (2013)	76	13.2	Good	2	3	<40*	Poor	Slight	39
Fortrose (2017)	74	23.3	Very Good	9	11	<40*	Poor	Moderate	0
Haldane (2005)	10	5.4	Moderate	0	0	<40*	Poor	Slight	9
Waikawa (2009)	5	0.9	Poor	5	51	<40*	Poor	Very High	52
Freshwater (2013)	40	5.9	Moderate	0	0*	≥80*	Very Good	Slight	28

* estimated value

The results also indicate a problem with the current rating criteria. Despite very large salt marsh losses in Fortrose/Toetoes, the estuary currently sits in the 'very good' band due to the high percentage of salt marsh remaining compared to the estuary area. Being classed in the top band means it is not possible to change to an improved band, however, there is significant scope for improving salt marsh in this estuary. There are also natural limits on the potential for salt marsh expansion in some estuaries. Freshwater Estuary is largely unmodified, yet because of the often steep and rocky to surrounding landforms, it is essentially at 100% of its maximum salt marsh capacity and increases in cover are unlikely.

In contrast to the unmodified Freshwater Estuary, all the listed mainland estuaries have had significant areas previously drained and reclaimed (e.g., Figure 3-9), which could potentially be returned to salt marsh habitat (retired) in the future, were that considered an appropriate management option. The degree that such retirement would achieve positive outcomes for each Southland estuary described in Table 3-3 could be gauged using the rating criteria presented above, while remaining cognisant of the preliminary status of the ratings. On that basis, NRE, Fortrose and Waikawa estuaries are most in need of restoration, in terms of losses since the 2000 NEMP baseline. NRE has the largest area of potential, with over 1200 ha of low-lying land previously reclaimed adjacent to the estuary (Figure 3-10).

Figure 3-9: Aerial photo of drainage and conversion of salt marsh to pasture in the Aparima Arm of JRE, ca 2013.

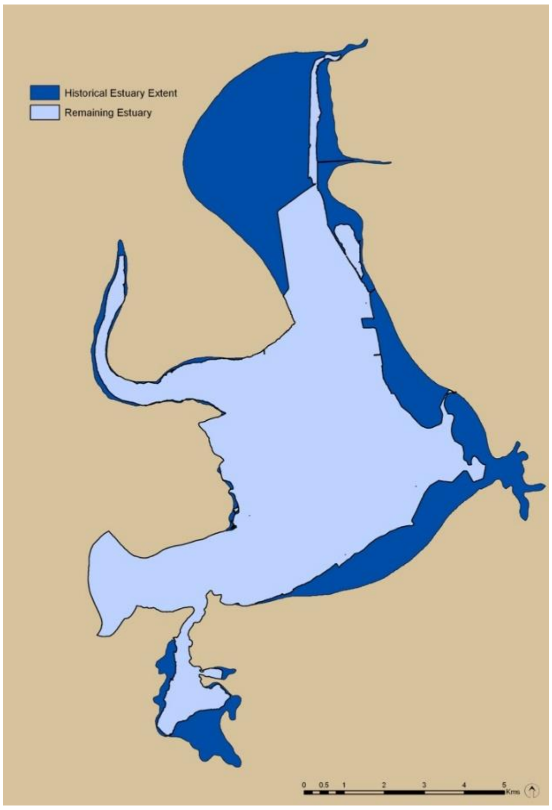

Figure 3-10: Estimate of the possible historical extent of New River Estuary based on land contours and historical maps. Source: Wriggle Coastal Management – unpublished.

3.5.2 Benefits and feasibility

The most effective way to maintain salt marsh value and realise their ecological benefit is to prevent avoidable loss in existing areas by limiting reclamation or by reversing it (retirement or de-reclamation). Restoring salt marsh is relatively straightforward where there are suitable habitat areas for planting, and many of the key species are readily available in commercial nurseries or can be grown from seed.

For restoration, an important consideration is the need to provide capacity for salt marsh to migrate in response to sea-level rise. Because most plants are unable to survive prolonged inundation, they require intertidal areas that allow enough wetting and drying. Sea-level rise will see salt marsh preferentially grow further inland from its current location to facilitate this. However, where barriers to this migration are present such as seawalls, migration is constrained and there is a high probability that salt marsh losses will increase where plants become more frequently inundated. To avoid this undesirable outcome, it is important to identify areas where migration can occur. Ironically, but not surprisingly, this is often those areas surrounding the estuary which have been previously drained and reclaimed. Where hard infrastructure is in place (e.g., roads and buildings) there is little scope for salt marsh expansion or migration. However, many areas are low-productivity and flood-prone pastures that offer substantial potential for a return to more natural features. Such changes could provide significant improvement in biodiversity, and provide flood mitigation, sediment retention and nutrient uptake benefits. An important development that should be made would therefore be to map future extent of sea-level rise for all Southland estuaries, overlaid with maps of ownership, land-use etc., i.e., utilize spatial planning tools.

Restoration planting also offers benefits through improved connectivity of existing estuary margin habitat with the estuary and surrounding landscape. This will provide (or re-instate) wildlife corridors and habitat for a range of estuary edge species like banded rail, fernbird and bittern, help prevent invasions of terrestrial weeds, and provide increased filtering, trapping and assimilation of nutrients and sediments. Assessments of current marginal wetland habitats such as Robertson et al. (2019) for Waituna Lagoon, Awarua Wetland and for NRE could be used in this context. Reports and public information including restoration activities relating to Waituna Lagoon are available on the Whakamana te Waituna website under the resources tab: https://www.waituna.org.nz/.

New Zealand case studies on estuarine riparian restoration include constructed wetlands for Te Waihora / Lake Ellesmere a large, low-lying coastal lagoon in Canterbury (Tanner et al. 2015). An assessment for Te Waihora predicted that a total of 593 ha of suitably-designed surface-flow wetland would reduce the annual nitrogen loads in all the major surface inflows to the lake by 20% and 1,782 ha of wetland to reduce the annual load by 40%. These wetlands were also predicted to decrease TP loads by 11–35% and 25–76% respectively and to concurrently achieve substantial reductions in sediment and microbial loads. This represents approximately 3 and 9% of the lake’s surface area (ca. 20,000 ha). Locating the wetlands on the edge of the lake, in river/stream riparian zones, or targeting farm run-off was deemed most feasible and lowest risk of adverse effects (e.g., affecting fish passage), as opposed to using remnant natural wetlands for this purpose. In-lake floating treatment wetlands (FTW – rafts of emergent plants) were a second option explored for nutrient load reduction at Te Waihora. Predictive work determined that to achieve the desired 20% and 40% reduction of surface inflow nitrogen loads, around 440 or 880 ha of FTW would be needed. However, based on wave climate modelling, only 72 ha of lake area was predicted to be suitable for FTW deployment.

In the Waimea Inlet near Nelson, recent planting of 800-1000 *Juncus kraussii* (searush) plants in the upper intertidal zone of a previously modified arm was undertaken as a trial to assess the success rate of infill planting. Rushes were purpose-grown from seed in a commercial nursery and planted out after six months to extend small beds of existing rushland and establish new beds adjacent to terrestrial margin plants previously planted by Nelson City Council. Initial indications are of a high survival rate. Similar restoration planting is scheduled for April 2020 as part of offset mitigation of stream dredging for flood control purposes at Waikawa Estuary (coastal Marlborough).

There is activity being undertaken by Department of Conservation on Waituna creek: 'Waituna creek transformation plan': https://www.livingwater.net.nz/catchment/waituna-lagoon/lower-waituna-creek-transformation-project/. There is a clear association of land retirement at Waituna Lagoon with the issues discussed below, in the 'Modification to Waituna Lagoon mouth opening regime to improve estuary resilience' Option.

Overseas case study examples of riparian remediation include those from the UK of 'managed realignment' of coastal defences to provide environmental benefits through re-creation of natural habitats or replace habitats lost due to reclamation or sea level rise. One such example is Paull Holme Strays on the north side of the Humber Estuary, where 80 ha of salt marsh and mudflats were created in 2003 by breaching existing flood defence embankments and realigning them to landward (Manson and Pinnington 2012). The scheme successfully restored habitat and attracted wading birds although there were some unexpected issues caused by rapid sedimentation in the restored area due to the shelter from the residual parts of the breached flood defences and high suspended sediment concentrations present in the Humber Estuary. Key differences between this type of habitat restoration in the UK and that being discussed for Southland are the higher tidal ranges, and increased presence of flood defences in the UK. This means that often removing flood defences is enough to initiate transformation of low-lying areas back to functional estuary/salt marsh.

Another overseas case study is that of the upper estuary of the Oka River in Urdaibai Biosphere Reserve (Biscay, Basque Country, northern Spain). https://climate-adapt.eea.europa.eu/metadata/case-studies/restoration-of-the-oka-river2019s-upper-estuary-part-of-the-urdaibai-biosphere-reserve/#challenges_anchor. Due to modification, the estuary's natural functions have been lost. The region is also threatened by climate change, with projected temperature rises and drops in annual rainfall, an increase in the intensity and frequency of extreme rainfall events as well as sea-level rise. It is predicted that the already fragile ecosystem could experience more damage from increased river flows, flooding, erosion of banks, beaches and estuary marshes, sediment flow changes and silting up of tidal channels. Riparian restoration initiatives for this system include removing dykes and other barriers to restore the intertidal function of the river channel, restoring previously existing marshland, building new embankments to create lagoons that will become intertidal zones, construction of walkways and bridges, development of environmental education tools and eradication of invasive species. Most of these initiatives are underway, with an accompanying cost of €2.5 million. Limitations have included lack of participation by all affected municipalities in the area, and legal issues with landowners.

In summary, restoration of estuary riparian margins will reverse loss of estuary edge habitat and ecosystem services, by regaining natural ecosystems. This will improve estuary biodiversity, habitat connectivity, flood mitigation, sediment retention and carbon and nutrient uptake benefits and potentially enhance resilience to sea-level rise. Among Southland estuaries, largest positive outcomes could potentially accrue for NRE (especially), Fortrose and Waikawa estuaries which have had largest riparian losses since the 2000 NEMP baseline. Success of restoration will depend on success of land retirement and de-reclamation efforts. Riparian restoration will be sensitive to sea-level rise and potential to move inland. Because the areas to be restored are often currently under alternate uses, land retirement is likely to be costly and legally complex. On the other hand, planting of retired areas is feasible at relatively low cost. The interaction of positive ecological outcomes with the costs and legal complexities indicates that spatial planning research would be beneficial in support of riparian restoration initiatives.

3.6 Management of Waituna Lagoon to improve estuary resilience

3.6.1 Environmental issue

The issue addressed by this option is the loss of ecosystem services in Waituna Lagoon, caused by a mouth opening regime designed to maintain artificially low water levels that allow drainage of adjacent land. The option described here is to promote ecological health by implementing controlled closures of the lagoon mouth, and to reclaim or retire farm lands from low-lying margins of the lagoon to allow lagoon openings to be independent of land drainage requirements.

Waituna Lagoon is a RAMSAR wetland of international significance and it is a place of great significance to Ngāi Tahu. It is also a scientific reserve administered by Department of Conservation (DOC). The native aquatic plants of the lagoon (primarily two species of *Ruppia*, *R. megacarpa*, *R. polycarpa*) are key contributors to its ecological condition. Issues in Waituna include the occurrence of phytoplankton blooms from nutrient enrichment and sedimentation (Schallenberg et al. 2010) with effects on aquatic vegetation (e.g., *Ruppia*) (Thompson and Ryder 2003; de Winton and Taumoepeau 2017). The lagoon receives runoff from an intensively farmed catchment and its water quality status is classed as eutrophic. There are concerns that ongoing nutrient enrichment could eventually change the lake to a devegetated, phytoplankton dominated state. The artificial lagoon levels have detrimental effects in terms of high salinity affecting macrophyte germination, increased wave action during open (shallow) lagoon periods (particularly on growing plant tips during the spring/summer period) and the exposure of lake bed sediments (dewatering).

This option considers artificial lagoon mouth management which can have a large impact on water quality and lagoon level in Waituna (Schallenberg et al. 2010), and it considers effects of prioritising ecological values when making decisions about mouth openings and lagoon levels (as opposed to prioritising land drainage). This option intersects strongly with the previous option (Restoration of estuary riparian margins).

3.6.2 Benefits and feasibility

The Lagoon has no permanent natural outlet and current consents allow it to be artificially opened to alleviate flood risk to low-lying land. The current consent for the opening of Waituna Lagoon permits the lagoon to be opened at 2.2m (a.m.s.l.) during the spring/summer period (Sept-March) and 2m for the remainder of the year. The current 5 year consent also allows for opening the lagoon in event of prolonged algal blooms. In addition to promoting land drainage, this practice has a benefit of diluting/flushing nutrients from the lagoon (Schallenberg et al. 2010). Ongoing monitoring of the lagoon indicates that the status of the aquatic vegetation is sensitive to the timing of opening: closure over the spring-summer period allows stable conditions for growth of *Ruppia*, and a winter opening is recommended to allow for flushing out of nutrients before the onset of the plant growing season to minimize phytoplankton and macroalgal growth (LagoonTechnicalGroup 2013; de Winton and Taumoepeau 2017; De Winton 2019).

Ramifications and practicalities of mouth opening and closure of Waituna Lagoon have been examined (Thompson and Ryder 2003; Schallenberg et al. 2010; Measures and Horrell 2013; de Winton and Taumoepeau 2017), with advice on optimal times and effects of opening with respect to adjacent wetland inundation and *Ruppia* bed health and phytoplankton. Closure of the lagoon is at present achieved naturally by long-shore transport but prolonged openings can result in detrimentally high lagoon salinity. Therefore, it has been considered desirable to investigate flexible closures and openings of the mouth (Measures and Horrell 2013) to provide an added level of control on lagoon salinity and nutrient status. There have been several detailed option investigations for control structure design for this purpose (see weblinks below for details of control structure studies[2]).

The Waituna Partners' Working Group has considered the issues around mechanical control of Waituna Lagoon water levels based on the values of partner agencies and the community (provided in review comments by DOC). It was concluded that control structures that provide flexible control of lagoon level had a mixture of negative, neutral and beneficial attributes across a range of goals and performance measures. These results would need to be addressed in decisions regarding implementation of control structures in Waituna Lagoon.

Land retirement in the low-lying parts of the Waituna Lagoon catchment would enable re-establishment of wetland ecology. Land retirement would reduce the imperative of Waituna openings for flood control and offer the opportunity to prioritise ecological restoration. Retiring farmland will have economic and social implications that will need to be balanced with the environmental benefits of optimised openings and lagoon levels. There are moves toward land retirement around Waituna (N. Ward, Environment Southland, pers. comm.) and it is intended that land identified as susceptible to drainage damage be gradually retired. It is intended this will ultimately allow consented lagoon openings based on primarily ecological outcomes. There are no site-specific or direct reports available yet, on this process (Tyron Strongman, Environment Southland, pers. comm.). Similar to their evaluations of water level control structures, the Waituna Partners' Working Group has considered the issues around land management, again based on the values of partner agencies and the community (provided in review comments by DOC). This assessment assumed lagoon water levels would be controlled and evaluated effects on goals and performance measures of private ownership vs public investment in surrounding lands.

Relevant areas where land drainage is influenced by lagoon water levels have been mapped by Walsh et al. (2016) (Figure 3-11). Such spatial planning research that maps important variables in Waituna Lagoon management (including sea-level rise, land ownership, land-use etc.) should be a priority.

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[2] https://www.waituna.org.nz/repository/libraries/id:1ytnyjmap17q9s20wg7s/hierarchy/Waituna%20resources/Lagoon%20management/2015%2002%20Tuckey%20Waituna%20Lagoon%20Pre-feasibility%20Engineering%20Scoping%20for%20Lagoon%20Closings%20and%20Openings.pdf

https://www.waituna.org.nz/repository/libraries/id:1ytnyjmap17q9s20wg7s/hierarchy/Waituna%20resources/Lagoon%20management/2015%2004%20Engineering%20Options%20for%20Managing%20Waituna%20Lagoon%20Water%20Levels%20and%20Values%20%5Bfactsheet%5D.pdf

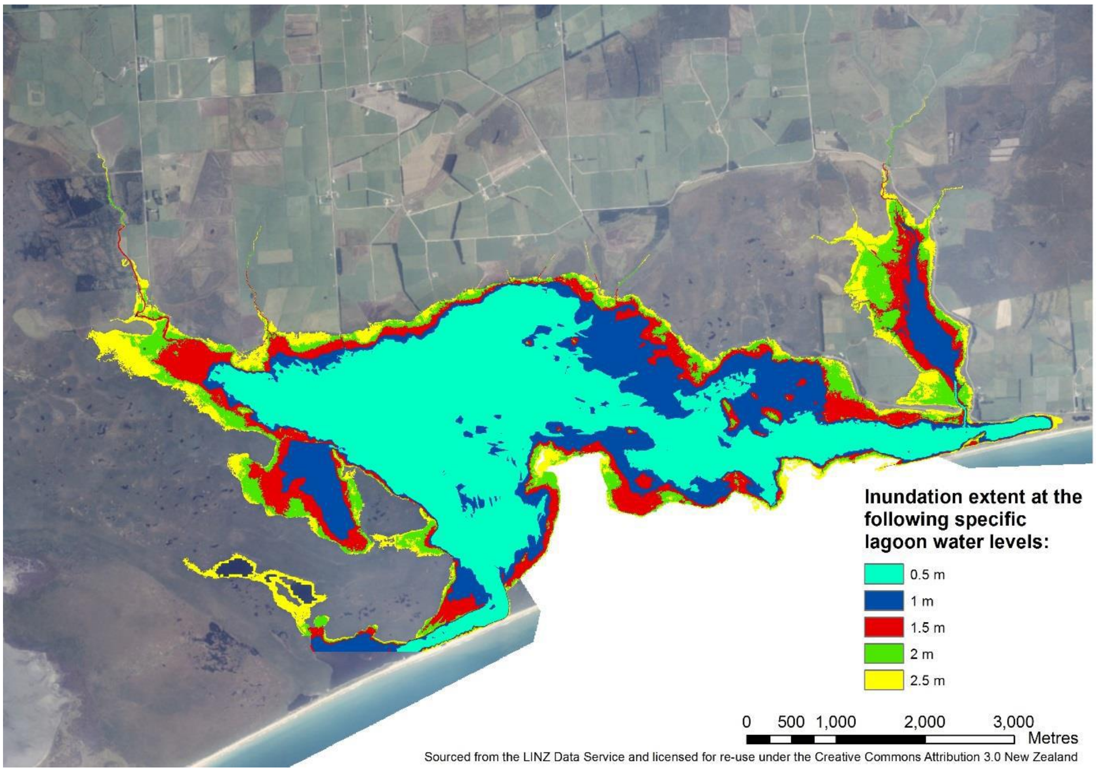

Figure 3-11: Potential inundation extent for Waituna Lagoon levels at 0.5 m, 1.0 m, 1.5 m, 2.0 m and 2.5 m. Figure reproduced from Walsh et al. (2016).

A relevant case study example for changing priorities of artificial lake openings is that of Te Waihora/Lake Ellesmere, a low-lying coastal lagoon in Canterbury with many similarities to Waituna Lagoon. Like Waituna Lagoon, Te Waihora is regularly artificially opened, with openings having a controlling effect on lagoon level. In the past Te Waihora openings were conducted solely for land drainage, and the cost of openings was funded by targeted rates charged to landowners. In recent years the decision-making process regarding lake openings has changed to include more consideration of environmental and cultural needs, including fish passage and water quality considerations. In line with this shift in priorities the funding for lake openings has now changed, with a greater contribution from general rates and only 50% of the cost paid by landowner targeted rates. Te Rūnanga o Ngāi Tahu and Environment Canterbury jointly hold the resource consent for Te Waihora lake openings. Decisions regarding openings are made jointly using a documented decision making 'protocol' which involves technical advice from experts and consultation with Rūnanga, local government, environmental groups, and landowner representatives[3]. The Te Waihora case study highlights that to shift the priorities regarding lake openings towards cultural and environmental values, the funding and decision-making processes for lake openings have also changed.

To summarise, water quality problems in Waituna Lagoon arise from a mouth-opening regime designed to maintain artificially low water levels that allow drainage of adjacent land. Healthy lagoon salinity, water quality and fish passage could be implemented with controlled closures of the lagoon mouth, and to reclaim or retire farm lands from low-lying margins of the lagoon to allow lagoon openings to be independent of land drainage requirements. Retirement of low-lying lagoon margin farmland will give greater freedom to prioritise the lagoon environment rather than land drainage when making decisions regarding openings and closures. Several studies are available on control

[3] https://tewaihora.org/understanding-the-lake-opening-process/

structure design and merits of private vs public ownership of surround lands for Waituna Lagoon. Retiring farmland will have economic and social implications that will need to be balanced with the environmental benefits of optimised lagoon levels, inferring the value of spatial planning research.

3.7 Partial diversion of Oreti River from New River Estuary

3.7.1 Environmental issue

The issue addressed by this option is the supply of nutrients and sediment to the NRE from the Oreti River, which is the major source of these contaminants to the estuary. Potentially, ameliorating these loads could stem eutrophication and sedimentation impacts in the estuary. The option described here is the partial diversion of Oreti River high flows through a cutting to the coast, where the river comes close to the coast, west of the NRE. This idea is not new, having been considered previously as a means of managing flood risk in the lower part of the Oreti River catchment (Gibb 1985). While this option is not strictly 'within the estuary' it addresses the major contributor of nutrient and sediment supply as a point source which could potentially be remediated and is therefore considered in scope of this project. This option is considered in detail here for the Oreti River / NRE system, primarily because of the good amount of existing knowledge and modelling pertaining to Oreti River flood control and loading implications for the highly impacted NRE. However, consideration is also given to the Mataura River, as it also comes close to the coast before entering the Fortrose estuary.

3.7.2 Benefits and feasibility

The Oreti River flows within 2.2 km of the open coast at a point approximately 8km upstream of where the river enters the NRE. The most feasible location for a diversion channel would likely be from this point directly southwest to Oreti Beach. The cut would be approximately 7.5 km shorter than the current channel through the estuary to the coast. As such the channel would be steeper and would tend to capture most of the river flow (full diversion), allowing the old channel to silt-up and close. An engineered control structure could be constructed on the diversion channel to ensure that under normal flow conditions a significant proportion of the river flow continues to be routed through the original channel. The control structure would then be opened during high flows to allow flood water to be diverted through the new channel (partial diversion).

Because of siltation, full diversion would have much greater impacts than partial diversion. Full diversion would substantially change the hydraulics of the estuary and have large implications for fish passage. Full diversion would also tend to induce downcutting of the river bed in the lower reaches of the Oreti River upstream of the cut, with potential implications for bank erosion and river morphology. The cut would likely allow saline water to propagate further upstream with potential associated effects on riparian vegetation and groundwater. Partial diversion would mitigate many of the biggest impacts associated with diversion as under normal conditions the changes to tidal and river hydraulics would be minimized. Partial diversion would, however, require the additional expense of a control structure and still allow some sediments and nutrients from the Oreti River to enter the estuary. It is likely that partial diversion is more feasible than full diversion due to the risk of unintended consequences associated with full diversion. For this reason, we have assumed partial diversion for our analysis of the potential benefits and feasibility.

Benefits and feasibility of diversion of the Oreti River was investigated here using the Delft / Delwaq water quality modelling results of Measures (2016). A diversion would intercept flood flows from both the Oreti and Makarewa Rivers (their confluence is upstream of the potential diversion location). Together these two rivers account for the vast majority of sediment delivered to the NRE

(Table 3-4), as shown by recent estimates of the suspended sediment load from Hicks et al. (2019) (of the order of 10 times the sediment delivery of either the Waihopai (Table 3-4), Waikiwi or other small streams (not shown). Of this suspended sediment most is carried at high flows. Figure 3-12 shows the proportion of total sediment load carried at a range of different flows in the Oreti River. If we assume that the diversion channel would take 70% of flow during high flow events (assumed to be at flows which are exceeded 10% of the time), and 30% of the flow at other times, it would divert approximately 43% of the total water volume, but 54% of suspended sediment load from the Oreti and Makarewa Rivers. This is equivalent to 52% of the total suspended sediment load to the estuary. This could be beneficial, considering interactions of macroalgal outgrowth and sediment retention described in the macroalgal (section 3.1), sediment (section 3.2) and seagrass (section 3.3) options discussed previously.

Table 3-4: Suspended sediment loads entering the New River Estuary. Data from Hicks et al. (2019).

River/site	Annual suspended sediment load (t/year)	Catchment area (km^2)
Oreti at Wallacetown	106687	2151
Makarewa at Counsell Road	71588	1016
Waihopai at Kennington	7572	162

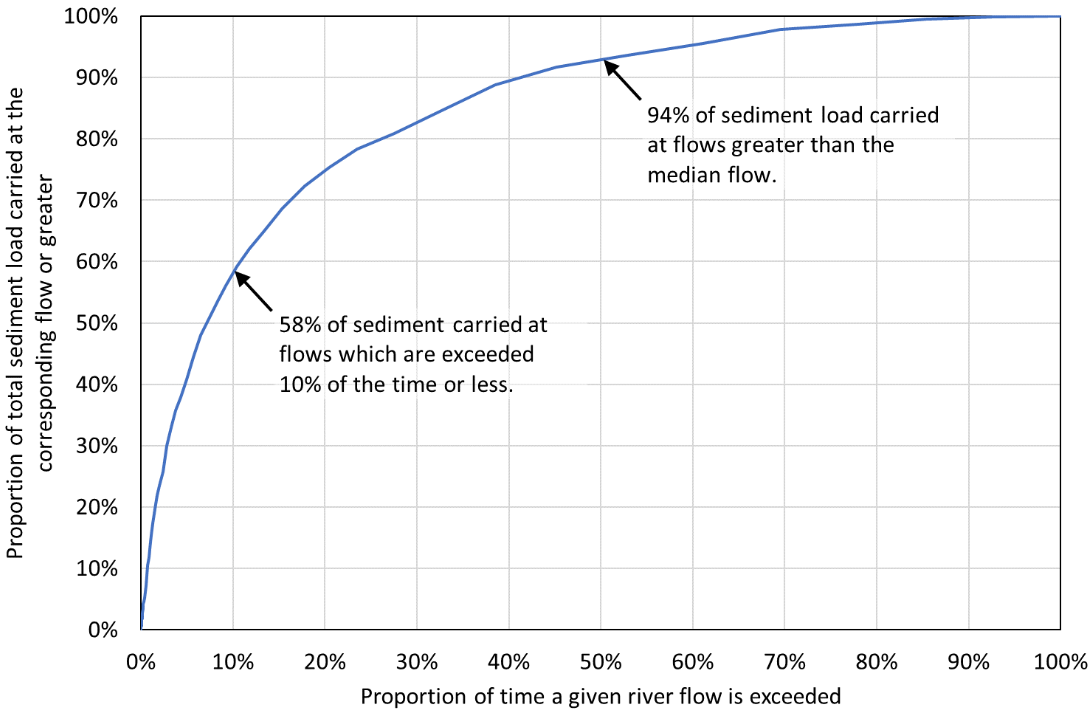

Figure 3-12: Proportion of total suspended sediment load carried at a range of different flows for the Oreti River. Based on unpublished NIWA data based on suspended sediment monitoring on the Oreti River at Tamaroa (just upstream of the Makarewa confluence). Data were collected as part of research conducted under NIWA's Managing Mud research program (Haddadchi et al. 2019).

Whilst diversion would significantly reduce the sediment loads to the NRE, it would also reduce the scouring effect of floods. This means that although the sediment load reduces, the proportion of sediment retained in the estuary is likely to increase in areas which were previously scoured by flows from the Oreti River (cf Figure 3-7). It is difficult to assess the significance of this effect without detailed modelling. It is likely that the Waihopai Arm, which currently has major issues with deposition of Oreti sediment but is not scoured by flood flows from the Oreti would likely see a significant reduction in fine sediment loading. Areas of the estuary downstream of the Oreti would see reduced loads but increased trap efficiency of the remaining loading, so the benefits are less certain.

The proportional reduction in total annual nutrient loads to the estuary from the Oreti River is likely to be at least the same as the proportion of water volume diverted (43%), with a higher proportion diverted if river nutrient concentrations are correlated to river flow. Total nitrogen concentrations in the Oreti River are only slightly higher at high flows, with seasonal variability in concentration more significant than flow related variability (Figure 3-13a). Phosphorus concentrations (both total and dissolved (DRP)) are correlated to flow (Figure 3-13b and c). As a diversion would divert a higher proportion of river flow at high flows it is likely to be more effective at reducing phosphorus loading than nitrogen loading.

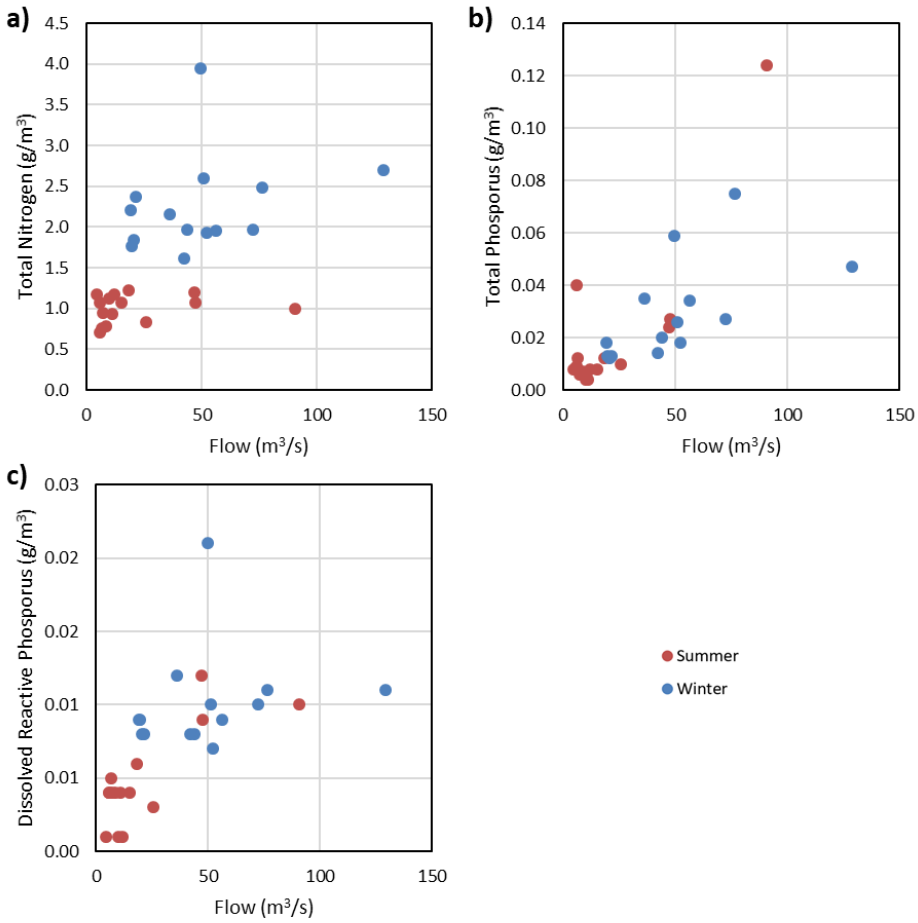

Figure 3-13: Seasonal relationship between nutrient concentrations and flow in the Oreti River at Wallacetown. Data from Environment Southland sampling between 2011 and 2016. Summer data are from December to February, Winter data from June to August.

While a diversion would result in substantial reductions to total nutrient loading this would not necessarily equate to as significant a decline in estuarine nutrient concentration. This is because the bulk of the diversion occurs during high flows, which only occur for a short proportion of the time. During normal and low flow conditions nutrient loading from the rivers would only be reduced 30% (equivalent to the proportion of flow being diverted). Furthermore, the concentration of incoming water would not be reduced, only it's flow rate, so reduction in estuarine nutrient concentrations would depend on increased dilution of the Oreti River water with other, lower nutrient sources of water. Given that all other sources of freshwater to the estuary have similar or higher concentrations of nutrients compared to the Oreti and Makarewa, seawater is the only lower nutrient source of water which could increase dilution.

An indication of the potential for increased dilution can be obtained by re-analysing the results of the summer and winter simulations of the NRE conducted by Measures (2016). The winter simulation had a total inflow from the Oreti and Makarewa of 54.2 m^3/s and the summer simulation 22.3 m^3/s, 59% lower. Using the results of these simulations it is possible to estimate the reduction in nutrient concentrations within different zones of the estuary (Figure 3-14) associated with this 59% change in Oreti/Makarewa flow. Assuming that the change associated with a 30% reduction in flow (under normal conditions with the diversion in place) is proportional (approximately half) of the reductions associated with the modelled 59% reduction in flow gives the reductions shown in Table 3-5. The analysis shows that the diversion would result in an approximately 11-12% reduction in DIN across the estuary, with some variability between zones. There is little reduction in DRP (slight increase in summer) as the concentration in the Oreti and Makarewa rivers is similar to the coastal seawater and the Invercargill wastewater treatment plant is a larger source of DRP to the estuary than the rivers (see section 3.8 for discussion of diversion of effluent from the Invercargill wastewater treatment plant).

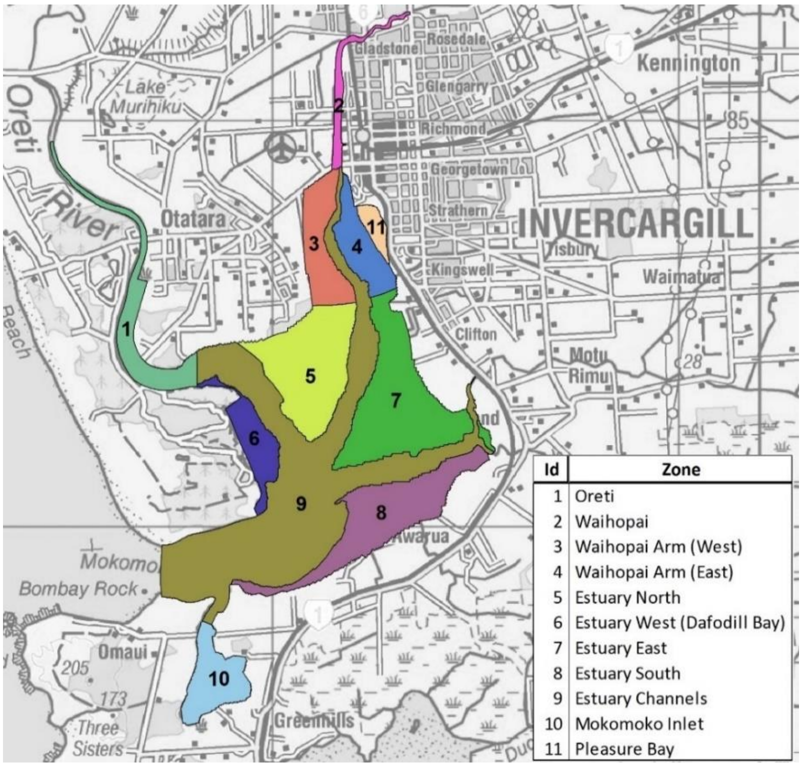

Figure 3-14: Location of estuary 'zones' used for analysis of the New River Estuary model results. Reproduced from Measures (2016).

Table 3-5: Estimated current and diverted nutrient reductions achieved by a 30% reduction in flow from the Oreti River and % change achieved. Results indicate nutrient reductions during median flow conditions in summer and winter, based on a re-analysis of model results from Measures (2016). Zone headings in red are those with greatest extent of Gross Eutrophic Zones (GEZ's: refer to Figure 3-3 and text).

			Predicted average nutrient concentration in each zone (g/m^3)											
			Zone 1	Zone 2	Zone 3	Zone 4	Zone 5	Zone 6	Zone 7	Zone 8	Zone 9	Zone 10	Zone 11	Whole
Summer	TN	Current	0.894	1.350	0.433	0.439	0.253	0.124	0.406	0.439	0.251	0.121	0.460	0.345
		Diverted	0.858	1.364	0.381	0.389	0.214	0.100	0.365	0.404	0.216	0.103	0.399	0.305
		Change	-4%	+1%	-12%	-11%	-15%	-19%	-10%	-8%	-14%	-15%	-13%	-12%
	DIN	Current	0.678	1.030	0.333	0.336	0.197	0.098	0.325	0.362	0.202	0.098	0.355	0.272
		Diverted	0.650	1.040	0.294	0.299	0.168	0.080	0.294	0.336	0.175	0.084	0.310	0.242
		Change	-4%	+1%	-12%	-11%	-15%	-18%	-10%	-7%	-13%	-14%	-13%	-11%
	DRP	Current	0.006	0.017	0.021	0.021	0.018	0.015	0.037	0.054	0.026	0.019	0.025	0.025
		Diverted	0.007	0.016	0.022	0.022	0.019	0.015	0.038	0.054	0.027	0.020	0.026	0.026
		Change	+10%	-1%	+4%	+4%	+4%	+3%	+2%	+1%	+2%	+2%	+4%	+3%
Winter	TN	Current	2.190	3.741	1.876	2.012	1.004	0.525	1.205	0.927	0.830	0.438	2.126	1.052
		Diverted	2.105	3.771	1.731	1.874	0.883	0.444	1.093	0.840	0.724	0.384	1.951	0.941
		Change	-4%	+1%	-8%	-7%	-12%	-15%	-9%	-9%	-13%	-12%	-8%	-11%
	DIN	Current	1.823	3.194	1.576	1.687	0.841	0.441	1.005	0.763	0.695	0.369	1.789	0.879
		Diverted	1.754	3.218	1.458	1.575	0.744	0.376	0.914	0.692	0.609	0.325	1.647	0.789
		Change	-4%	+1%	-7%	-7%	-12%	-15%	-9%	-9%	-12%	-12%	-8%	-10%
	DRP	Current	0.012	0.020	0.043	0.051	0.025	0.018	0.048	0.037	0.025	0.022	0.083	0.025
		Diverted	0.012	0.020	0.044	0.051	0.025	0.018	0.048	0.037	0.025	0.022	0.083	0.025
		Change	+1%	-0%	+0%	+0%	+0%	+0%	+0%	+0%	+0%	+0%	+0%	+0%

Using the same re-analysis of model results it is possible to estimate the effects of partial diversion on estuary salinity as shown in Table 3-6. This indicates that the changes in salinity during normal flow conditions would be relatively small.

Table 3-6: Estimated changes in time-averaged estuary salinity resulting from 30% reduction in flow from the Oreti River. Shown are estimated changes in salinity during median flow conditions in summer and winter, based on a re-analysis of model results from Measures (2016).

Oreti/Makarewa		Salinity (ppt)											
	flow	Zone 1	Zone 2	Zone 3	Zone 4	Zone 5	Zone 6	Zone 7	Zone 8	Zone 9	Zone 10	Zone 11	Whole
Summer	median	3.0	9.1	19.6	19.7	24.4	28.8	24.1	27.2	26.6	30.7	18.3	23.4
	-30%	4.4	8.6	21.9	22.0	26.4	30.1	25.9	28.7	28.3	31.6	21.2	25.2
Winter	median	2.1	1.7	14.4	13.6	22.7	28.3	21.9	24.9	25.1	30.1	13.5	21.8
	-30%	3.5	1.2	16.7	15.9	24.6	29.6	23.8	26.4	26.8	31.0	16.4	23.6

The effects of the diversion channel delivering high loading of suspended sediment and nitrogen to the open coast during flood are uncertain. Due to the energetic wave environment and high dilution the sediments and nutrients are likely to disperse rapidly, however further analysis would be required to identify any potential issues, including potential for deposition of fine sediments.

Longshore transport of beach sand on the coast where the diversion channel emerges may constrict or deflect the mouth of the channel. Detailed analysis would be required to investigate what proportion of flow needs to be diverted under normal flow conditions to maintain an open channel (we have assumed 30% is enough). If groynes or similar structures are required to maintain an open channel, then their design would need to consider their impacts on longterm shoreline/beach morphology as well as their cost.

These results indicate that a 30% diversion of Oreti River flow from the NRE, predicted to cause a 12% reduction of potential TN concentration across the whole estuary (Table 3-5), would achieve relatively little in terms of eutrophication remediation. This is because the current potential TN concentration in the NRE is very high – ~0.65 g TN/m^3 (estimated from the Delwaq results: Table 3-5) averaged across summer and winter[4]. This is in ETI macroalgae EQR band D (strongly impacted) with potential TN concentration being far higher than the upper (C : D) threshold of EQR band C (0.32 g TN/m^3) (Plew et al. 2018a; Plew et al. 2019).

However, considering only summer, when *Gracilaria* outgrowths are greatest and not-light limited (as they are in winter), the Delwaq modelling (Table 3-5) shows considerably lower TN across the estuary (ca 0.34 g/m^3). Focusing on the estuary areas with largest extents of GEZs (Figure 3-4 and Table 3-5) where *Gracilaria* outgrowths are greatest (corresponding to zones 3, 4, 5 and 6 in Figure 3-14), summer NRE TN concentrations are ~0.10-0.39 g/m^3 (Table 3-5) after the Oreti River diversion. For zones 3 and 4 these are again higher than the EQR band C:D threshold but considerably closer to it. For zone 5, the Delwaq estimate is lower (0.25) but its GEZ is close to zone 3 (Figure 3-4) and likely affected by its poor conditions. Thus, for the worst-affected areas of the NRE, the forecasted summer reductions from Oreti River diversion remain insufficient to shift eutrophication conditions toward a significantly improved state.

Another potential candidate for Southland estuary river diversion could be the Mataura River which drains to Fortrose Estuary. Fortrose Estuary currently sustains high areal nutrient loading (D. Plew, NIWA pers. comm.) and high nutrient concentrations, especially in its backwater Titiroa Arm. Although there is no detailed hydraulic or nutrient modelling available for this system, some initial considerations can be made. Similar to the Oreti River case, assuming a cut to the coast is made in the Mataura River just before it enters Fortrose Estuary, it is likely that the resulting tidal exchange with the sea would increase salinity in the Mataura River some (unknown) distance landward, effectively transferring the eutrophication problem into that new area, and would also have implications for riparian habitat and groundwater. Furthermore, full diversion would substantially change the hydraulics of Fortrose Estuary, with implications for siltation, fish passage, and potentially downcutting of the river bed in the lower reaches of the Mataura River. As with Oreti River diversion, partial diversion would mitigate these impacts but would require the additional expense of a control structure and still allow some sediments and nutrients from the Mataura River to enter the estuary. Thus, similar to the Oreti / NRE system, the realised benefits in terms of nutrient reduction in Fortrose Estuary may not be cost-effective. These issues would need detailed examination, including detailed hydraulic modelling of proposed diversion options for the Mataura / Fortrose system. Finally, for both systems, impacts of the 'new' nutrient and sediment loading to the coastal area adjacent to the cuttings would need to be forecast.

Thus, for two of the three Southland estuaries most heavily impacted by nutrient loading (NRE and Fortrose: Table 3-1), there are indications that eco-hydrological interventions (cuttings to the sea) may have either little restorative impact and/or important deleterious side effects. Again, as noted by Duarte and Krause-Jensen (2018): "such eco-hydrological interventions need be considered with

[4] Note this is close to annual average estimates made using CLUES Estuary (Plew et al. 2018b)): 0.644 g/m^3.

care, using models to predict possible responses, to avoid negative experiences due to ill planned interventions".

There are several examples of New Zealand rivers where diversion channels have been constructed, although all were for flood control rather than ecological reasons. The Heathcote River (Christchurch) had a 500 m flood diversion channel constructed in 1986, known as the Woolston Cut. This channel shortcut across a meander in the river channel making a steeper, more direct path for floodwater. Whilst the cut was successful at reducing flood risk it had several unintended side effects, including propagation of salinity further up the channel leading to the deaths of riparian trees and bank erosion. To mitigate these effects a control structure (the Woolston Barrage) was constructed on the cut to direct water through the original channel except during floods, when it was opened to allow the diversion channel to operate.

The Wairau River (Blenheim) had a 5 km diversion channel constructed in 1963 to alleviate flood risk upstream. The diversion cuts from the river approximately 13 km upstream of the Wairau lagoon directly to the coast at Cloudy Bay. The channel is effective at conveying floodwater but has resulted in sedimentation within the original course of the channel, reducing the flood capacity of that channel, and has reduced the tidal prism of the Wairau Lagoon mouth, allowing longshore transport to constrict the mouth resulting in perched water levels at low tide. To mitigate these issues an erodible gravel embankment has been used since 2009 at the bifurcation between the river and diversion channel. The embankment ensures that 70% of flow is retained in the original river course during normal flows and small floods, helping to flush sediment from the lower river. During large floods the embankment erodes and the diversion operates to its full capacity (Christensen and Doscher 2010).

The Kaituna Cut (Bay of Plenty) was constructed in 1956 for land drainage. Whilst effective at reducing water levels the cut captured almost the entire flow of the river and had major impacts on the health of the Maketū Estuary, which the river used to flow through. Water quality has declined and the estuary has become clogged with algae and coastal sediment impacting natural habitats of species such as pipi and making it difficult for boats to enter or leave the harbour. Due to land drainage and development, 95% of the natural estuarine wetland habitat has been lost. Major works are being undertaken by Bay of Plenty Regional Council to "re-divert" the river through a new channel and increase the flow through the Maketū Estuary. There has been extensive community consultation and planning, and acquisition of 45 ha of adjacent pastoral land is planned. New construction work includes an inlet structure, moving and upgrading stop banks, and creating new wetlands, at a total cost of at least $NZ17 million (https://www.boprc.govt.nz/our-projects/kaituna-river-rediversion-and-maketū-estuary-enhancement/).

There are also overseas examples of where hydrological interventions have been employed to improve ecological conditions in estuaries (Duarte and Krause-Jensen 2018). These include the Mondego estuary (Lillebø et al. 2005) (Portugal), which was eutrophic for some decades and water circulation mainly depended on tides and on a highly nutrient-loaded small freshwater tributary. After 1998, this freshwater was diverted to a more tidally-dominated part of the estuary and an internal estuarine channel was deepened to improve tidal exchange. Biomass of nuisance macroalgae was reduced by one order of magnitude due to nutrient reduction and nutrient limitation in the system. After the mitigation seagrass and macroinvertebrate health were improved (Lillebø et al. 2005; Cardoso et al. 2007), although seagrass areal cover remained much less than its original state (Cardoso et al. 2010). This demonstrated the non-linear recovery rates of estuarine systems discussed by Duarte et al. (2009), wherein systems may resist full return to original conditions upon remediation.

The Peel-Harvey estuary system (southwest Australia) (Humphries and Robinson 1995) has been the subject of intensive investigation and management because of worsening algal blooms and their impacts. An integrated catchment management scheme of voluntary reduction in fertiliser use and construction of a new channel from the estuary to the sea was implemented. There was some uncertainty whether the cut should be built, because it was predicted that it would not be sufficient without the projected reductions in catchment fertiliser use being implemented. However, upon implementation, these measures have been successful in reducing phosphorus pollution and algal blooms, although there have been subsequent upward pressures on land use (Humphries and Robinson 1995).

Due to coastal reconstruction in response to flooding, the former tidal inlet Lake Veere (south-western Netherlands) was rendered a stagnant brackish lake in 1961 (Wijnhoven et al. 2010), followed by continuous degradation in macrobenthic and associated abiotic conditions. Restoration of hydraulic exchange between the tidal marine Eastern Scheldt and Lake Veere in 2004 allowing improved water quality (transparency, nutrients, and oxygen). Initial indications of macrofaunal improvements occurred by 2009, showing that restoration following improved abiotic conditions can take several years.

Changes in ecosystem states in a coastal lagoon were elicited in the nutrient-stressed Ringkøbing Fjord (Denmark) (Petersen et al. 2008), via a change in sluice management. There was a shift from a turbid state into a clear-water state, caused by high recruitment and population growth of plankton-feeding clams. The change was not caused by reduced nutrient loading or concentration, which did not occur in the estuary as a result of the sluice managment. Biomass of rooted benthic vegetation (*Ruppia cirrhosa* and *Zostera marina*) reduced because of salinity increase, and *Ulva lactuca* increased in the system, possibly because nutrient pollution in the system was not solved by the management action.

Sand build-up surrounding Mont Saint-Michel (Brittany, NW France) has threatened the island nature of the abbey by potentially connecting it permantly to the mainland by 2040 http://www.projetmontsaintmichel.com/en/why_act/objectives.html. To avoid this, in 2009 a new dam was built on the Couesnon River draining to the Moidrey Cove surrounding the Mont, to regulate the river's hydraulic capacity and flushing power in Moidrey Cove. This was combined with clearance of the Couesnon River channel and retirement of a car park on the causeway to the Mont, to further improve the hydraulics. In 2017 a report considered that the remediation was successful, and will achieve its aims by 2025. The cost was over €200 million, and there have been concerns about ongoing costs to maintain the reclaimed maritime character of the site.

These NZ and overseas case studies point out the balance between catchment nutrient and sediment delivery and removal by flushing may be improved by interventions, while also pointing out knowledge gaps that are addressable and specific to the functionality of individual systems. The biological responses may be unpredictable due to non-linear recovery. Some of the examples also show that the interventions can have detrimental consequences and also that their degree of success may not be obvious without detailed prior examination (such as shown by modelling of the Oreti Diversion). Finally, it was clear that some interventions would not succeed without concomitant catchment load reduction, which adds risk to the intervention implementation. Interventions would therefore need to be carefully explored for individual Southland estuaries using detailed hydrodynamic and nutrient modelling tools such as demonstrated here for the Oreti / NRE system, combined with ecological and land-management policy study and forecasting.

In summary, modelling of partial diversion of the Oreti River's flows to the sea (Oreti Beach) showed a 50% reduction in suspended sediment load to NRE, potentially yielding improved sediment deposition rate for seagrass and macroinvertebrate communities. However, it would cause only 11-19% reduction in total nitrogen (TN), insufficient to change the NRE's very poor trophic rating. There are major risks associated with river diversion including increased salinity and downcutting/bank erosion in the river upstream of the diversion and silting up of the Oreti River / NRE downstream of the diversion, and effects of sediment and nutrient dispersal on the Oreti Beach coastal environment. Many of the same implications likely exist for the Mataura River, should it be wholly or partially diverted before entering Fortrose Estuary. New Zealand and overseas case studies of river diversions show that unexpected and detrimental side effects are common and costs are likely to be very high. Further studies/modelling would be required to investigate these risks.

3.8 Diversion of effluent from the Invercargill wastewater treatment plant from New River Estuary

3.8.1 Issue

The issue addressed by this option is the supply of nutrients to the NRE from the Invercargill Wastewater Treatment Plant (WTP), which contributes nutrients to the NRE and is partially responsible for its eutrophication. The option described here is the removal of this wastewater stream, for example through an ocean outfall or by other means. Although this option is specific to NRE among Southland estuaries, it is considered an important one to address because of the poor state of the NRE and the unique role it has in Invercargill wastewater disposal.

3.8.2 Benefits and feasibility

The Invercargill WTP discharges treated wastewater into the NRE. The discharged nutrients contribute a significant proportion of the total nutrient loading to the estuary (Table 3-7), particularly of dissolved reactive phosphorus (DRP), and especially during summer, when nutrients concentrations and flows in rivers are generally lower. However, the total load does not provide the full picture as wastewater is only discharged from one hour before high tide until two hours after high tide, to ensure that it gets rapidly transported to sea on the outgoing tide, reducing its impact on the estuary. Delft/Delwaq hydrodynamic modelling of the estuary (Measures 2016) shows that the concentration of nutrients within different parts of the estuary is influenced to different extents by the wastewater treatment discharge. The study broke the estuary up into the same 11 zones used in the Oreti Diversion option (Figure 3-14) and quantified the impact of the wastewater treatment works on mean nutrient concentrations in each zone (Table 3-8).

Table 3-7: Proportions of New River Estuary nutrient load originating from the Invercargill Wastewater Treatment Plant. Estimates based on the median summer and winter river/wastewater flow and nutrient conditions.

	TN	DIN	DRP
Summer	20%	22%	89%
Winter	4%	4%	56%

Table 3-8: Concentrations of New River Estuary nutrients originating from the Invercargill Wastewater Treatment Plant (WTP). Estimates for each estuary zone (refer Figure 3-14), based on a re-analysis of model results from Measures (2016). Zone headings in red are those with greatest extent of Gross Eutrophic Zones (GEZ's: refer Figure 3-4 and text).

		Source	Average nutrient concentration in each zone (g/m^3)										
			Zone 1	Zone 2	Zone 3	Zone 4	Zone 5	Zone 6	Zone 7	Zone 8	Zone 9	Zone 10	Zone 11
Summer	TN	WTP	0.02	0.23	0.27	0.3	0.08	0.02	0.27	0.17	0.07	0.07	0.44
		Other	0.89	1.31	0.37	0.37	0.22	0.12	0.23	0.17	0.17	0.09	0.36
		WTP %	2%	15%	42%	45%	28%	17%	53%	50%	29%	42%	55%
	DIN	WTP	0.02	0.2	0.23	0.26	0.07	0.02	0.23	0.15	0.06	0.06	0.38
		Other	0.68	1.00	0.28	0.28	0.17	0.10	0.18	0.13	0.13	0.07	0.27
		WTP %	3%	17%	46%	48%	30%	18%	56%	53%	31%	44%	58%
	DRP	WTP	0.003	0.033	0.038	0.042	0.012	0.004	0.038	0.024	0.010	0.009	0.062
		Other	0.006	0.012	0.013	0.013	0.014	0.015	0.014	0.015	0.015	0.015	0.013
		WTP %	32%	72%	75%	76%	46%	19%	73%	62%	40%	38%	83%
Winter	TN	WTP	0.01	0.08	0.23	0.26	0.09	0.03	0.25	0.18	0.07	0.07	0.51
		Other	2.20	3.66	1.66	1.74	0.92	0.50	0.96	0.76	0.74	0.38	1.63
		WTP %	0%	2%	12%	13%	9%	5%	20%	19%	9%	15%	24%
	DIN	WTP	0.01	0.07	0.2	0.23	0.08	0.02	0.21	0.15	0.06	0.06	0.44
		Other	1.83	3.12	1.39	1.45	0.77	0.42	0.79	0.62	0.62	0.32	1.36
		WTP %	0%	2%	12%	14%	9%	5%	21%	20%	9%	15%	24%
	DRP	WTP	0.001	0.012	0.032	0.037	0.013	0.004	0.035	0.025	0.010	0.009	0.072
		Other	0.012	0.009	0.012	0.012	0.013	0.014	0.013	0.013	0.013	0.014	0.012
		WTP %	9%	58%	73%	76%	49%	22%	73%	65%	43%	41%	86%

Like the option for Oreti River diversion, these results for diversion of the Invercargill WTP from the NRE can be evaluated with respect to the degree of eutrophication remediation expected. When examined spatially, Table 3-8 shows reductions ranging from 17 to 45% of summer TN concentrations, with largest reductions in zones with largest GEZs (zones 3, 4 and 5). Thus, it is expected that diversion of Invercargill WTP effluent would have a more substantial effect on shifting ETI TN bands meaningfully (potentially through EQR band D:C threshold) than diversion of Oreti River discharge. The band C:B threshold (to moderate eutrophication (Plew et al. 2018a)) is 200 mg TN/m^3, so it is unlikely that the diversion would achieve that level of TN in the zones with greatest GEZs (zones 3 and 4) but could do so in zone 5.

A further consideration is that because the effects of the WTP diversion on DRP are considerably larger than for N (Table 3-8), it is possible that DRP reduction could be more effective than N reduction in shifting ETI trophic bands in NRE. *Gracilaria* growth in NRE is currently most likely not nutrient limited by either N or P supply as both are very high (Plew et al. 2019) (D. Plew, NIWA, pers. comm.). The finding that the WTP diversion would affect DRP much more than N, raises the possibility that the diversion would render *Gracilaria* growth P-limited (raise the estuary N:P molar ratio to greater than 30 (Plew et al. 2019)) above which macroalgal growth is typically P- rather than N-limited (Atkinson and Smith 1983). The degree to which the ratio is raised, and the degree to which the absolute concentration of P is reduced, could be used as guides to nominate ETI bands based on P and N, to assess the likelihood that the WTP diversion would change trophic rating for the NRE. This possibility requires further research including detailed examination of nutrient uptake physiology of *Gracilaria*[5]. Also, because the ETI bandings for macroalgae are based on estuary-wide macroalgal estimates and nutrient concentration data, revised bandings may be required for more localised predictions of macroalgal response.

[5] Such experimental data have been collected with recent NIWA experimental research (B. Dudley NIWA pers comm).

An important ramification of results from this Invercargill WTP diversion option and the Oreti River diversion option is that diversion of Invercargill WTP effluent *combined* with partial diversion of Oreti River flows could lead to substantially improved NRE water quality and trophic conditions. Another consideration is that WTP diversion, *combined* with improvements in Southland catchment runoff water quality (potentially achievable over time) could also do so. The latter combination may be seen by the community as a more appropriate way forward.

A case study relevant to Invercargill WTP diversion is that of the Christchurch WTP effluent diversion from the Avon-Heathcote Estuary (Ihutai) which occurred in March 2010 (Barr et al. 2019; Zeldis et al. 2019). Like the NRE, the Avon-Heathcote Estuary is a well-flushed tidal lagoon. Upon wastewater diversion, there were considerable improvements in a range of ecological indicators, including decreased water column and porewater nutrient concentrations, microphytobenthic and macroalgal biomasses and enrichment-affiliated macrobenthos. There was improved denitrification efficiency of the estuary nitrogen load, overall. The studies showed the Avon-Heathcote Estuary, despite receiving decades of heavy nutrient loading and eutrophication driven by poor water quality, responded rapidly to decreased loads and exhibited high ecological resilience (Zeldis et al. 2019). It was concluded that this resilience stemmed from the estuary's high tidal flushing and its coarse (sandy), well-irrigated sediments which did not store a legacy of eutrophication. This was compared with several overseas estuarine case studies (described in Zeldis et al. (2019)), showing that resilence to eutrophicaton and recovery rates upon wastewater removal can depend on attributes including variable flushing and sediment grain size (Borja et al. 2010). In this respect it is noted that the NRE also has coarse sediments over much of its area (Robertson et al. 2015), although not in its heavily impacted GEZ areas which, while historically sandy, are now very muddy and could sustain a legacy of N and P efflux. This possibility would need further research.

It is also notable that in the Avon-Heathcote case, the Christchurch WTP diversion elicited a 90% reduction in N load (Burge 2007; Zeldis et al. 2019); this would not happen in the case of Invercargill WTP diversion, where NRE catchment inputs (of N) would continue to be major unless also remediated. Also, it is noted that macroalgal eutrophication is still an issue in the Avon-Heathcote: it is apparent that its riverine loading is still sufficient to elicit outgrowths (Hawes and O'Brien 2000; Barr et al. 2019), (J. Zeldis pers. obs.) albeit at lower levels than prior to diversion. The ca. $NZ80M cost of the Christchurch WTP outfall indicates that the cost/benefit of such development for NRE would need careful consideration. Offshore (coastal) environmental effects of Invercargill WTP effluent dispersal via an outfall would also need to be considered. In the case of the Christchurch WTP diversion, hydrodynamic and ecological modelling predicted that detrimental effects of the effluent dispersal in Pegasus Bay in the near-field were unlikely, because of very high dilution and dispersal of treated effluent (Zeldis and Gall 1999; Spigel and Zeldis 2004).

In summary, removal of the WTP effluent would reduce concentrations of DIN by 3-46% and DRP by 19-76% in summer in NRE (when macroalgal growth is maximal), with the ranges depending on the estuary zone considered. The reductions were greatest in the worst-affected (GEZ) areas of the estuary, where there would be up to 48% reduction in DIN and 76% reduction in DRP in summer. Benefits in terms of improved ETI trophic condition are potentially achievable, depending on estuary zone, especially if combined with moderate improvement in catchment-derived loads. Further studies/modelling would be required to investigate this, including the possibility that the large reductions of DRP concentration could be a major driver of macroalgal growth limitation. This option would require expensive infrastructure upgrades but would have only beneficial side-effects within

NRE, as were observed in the Avon-Heathcote/Ihutai (Canterbury). Effects on the coastal environment of outfall effluent dispersal would need investigation.

4 Conclusion

This report has evaluated the issues, benefits and feasibilities of eight options for Southland estuary remediation. It has revealed that some of the options are probably not pragmatic or have associated unacceptable side-effects and very high qualitative costs, and/or are unlikely to succeed without accompanying reductions in catchment-derived loads of nutrients and/or sediments. Other options are found to be more viable in terms of likelihood of positive remediation outcomes and absence of deleterious side-effects, but with a wide range of qualitative potential costs. There are often sub-areas within estuaries where options are non-viable but other sub-areas where they may be. In some cases, synergistic interactions between the options are evident and there are cases where more than one option would be required to gain meaningful change.

Table 4-1 summarises the findings for each option including the environmental issue addressed, the benefits of applying the option, and the likelihood of success and feasibility, including synergistic interactions, side-effects, qualitative costs and needs for further investigation.

Table 4-1: Summary of remediation options for Southland Estuaries. For each option, first discussed is the environmental issue addressed, the benefits potentially accruing by applying the option, likelihood of success, and feasibility of applying the option (including logistics, side effects, and qualitative cost) and necessity of catchment remediation for success of option. GEZ's: Gross Eutrophic Zones. Note: this table is replicated in the Executive Summary

Option	Environmental issue	Benefits	Likelihood of success/feasibility	Catchment remediation
Removal of macroalgal biomass	Macroalgal eutrophication impacts	Reduce smothering of benthic habitat, improve sediment health, remove noxious odour, improve estuary amenity	Complete removal unfeasible, some partial solutions may work (winter removal, target selected incipient GEZ's). Destructive side effects of removal likely and costs likely to be high. Synergistic with fine sediment accumulation. Algal growth experimental and modelling research would be beneficial	In parts of estuaries with current or incipient GEZ's, removal would need to be continuously applied if catchment nutrient and sediment loads are not reduced, for estuaries exceeding trophic limits (NRE, JRE, Fortrose)
Removal of degraded sediments	Sediment eutrophication and muddiness, loss of ecosystem services	Reduce muddiness and sediment nutrient levels, improve sediment oxygen and sulphide status for biota, increase clarity, improve estuary amenity	Complete removal unfeasible, hydraulic interventions (drainage channel deepening, low pressure sluicing) possible. Could interact synergistically (positively) with macroalgal removal. Research on hydraulics and interactions with macroalgae would be beneficial. Very destructive side effects likely. Costs likely to be very high	Removal would need to be continuously applied, if catchment nutrient and sediment loads are not reduced for estuaries exceeding trophic limits (NRE, JRE, Waikawa, Haldane)
Restoration of seagrass beds	Loss of estuary habitat and ecosystem services	Improve habitat for important ecosystem components (recruits), improve biogeochemical ecosystem services (e.g., denitrification, nutrient sequestration)	Appropriate for estuaries with significant historical seagrass beds (NRE, JRE) that have lost them. Feasible if other estuary conditions remediated (sediment deposition and nutrient concentrations reduced). Experimental out-planting research would be beneficial. Potential for synergistic interactions with cockle restoration. No detrimental side-effects	For estuaries (or parts of estuaries) with historic seagrass beds, catchment load sediment and nutrient remediation would be required where seagrass trophic limits exceeded (NRE, JRE, Fortrose)

Option	Environmental issue	Benefits	Likelihood of success/feasibility	Catchment remediation
Cockle bed restoration	Loss of kaimoana and ecosystem services	Establish kaimoana sources, improve natural amenity and help restore natural ecosystems	Appropriate for parts of NRE, JRE, Waikawa, Haldane and Fortrose where fine-scale habitat conditions suitable. Reseeding success is unlikely in parts of estuaries that are currently highly eutrophic and muddy; NIWA reseeding guidelines indicate that sandy substrates in stable (non-highly sedimentary) habitats with good planktonic food supply and relatively high salinity are ideal. Experimental cockle bed restoration research would be beneficial, along with information on historic cockle distributions and abundance. Potential for synergistic interactions with seagrass restoration. Negligible detrimental side-effects, relatively low cost	Catchment load sediment and nutrient remediation would be required where conditions are insufficient for successful cockle bed restoration (muddy backwater areas of NRE, JRE, Fortrose, Waikawa, and Haldane estuaries as well as the more exposed GEZ's in those estuaries)
Restoration of estuary riparian margins	Loss of estuary edge habitat and ecosystem services	Regain natural ecosystems, habitats and ecosystem services, providing improved biodiversity, habitat connectivity, flood mitigation, sediment retention and carbon and nutrient uptake benefits	Appropriate for estuaries with significant historical riparian margins. NRE, Fortrose and Waikawa estuaries are most in need of restoration, in terms of losses since the 2000 baseline. Subject to success of land retirement and de-reclamation efforts. Sensitive to sea level rise and potential to move inland. Land retirement likely to be costly and legally complex. Planting programmes feasible at relatively low cost. Spatial planning research would be beneficial. No detrimental ecological side-effects but both positive and negative social side effects	For estuaries with significant riparian margin loss, retirement of land in catchment will be required

Option	Environmental issue	Benefits	Likelihood of success/feasibility	Catchment remediation
Modification to Waituna Lagoon mouth opening regime to improve estuary resilience	Waituna Lagoon eutrophication, loss of estuary habitat and ecosystem services	Openings prioritised for management of lagoon salinity, water quality and fish passage, with land drainage a lower priority	Retiring low-lying lagoon margin farmland would give greater freedom to prioritise lagoon environment rather than land drainage when making decisions regarding openings. Controlled closure / opening of the lagoon would prevent high salinities associated with prolonged openings and allow water quality control. Control structure design/location research is available. Land retirement likely to be costly and legally complex with both positive and negative social side effects	Retirement of land in catchment likely to be required to remove priority for land drainage
Partial diversion of Oreti River	NRE sedimentation and eutrophication	50% reduction in suspended sediment load to New River Estuary. 10-11% reduction in DIN concentration. Benefits in terms of improved ETI trophic condition negligible or minor	Reduction in Oreti River sediment inputs could interact positively with macroalgal, sedimentation and seagrass conditions, but nutrient reductions unlikely to improve NRE nutrient trophic state significantly. Major risks associated with river diversion including increased salinity and downcutting/bank erosion upstream of the diversion and silting up of the Oreti River/NRE downstream. Similar considerations apply for cutting the Mataura River to the sea before it enters Fortrose Estuary. Case studies of river diversions show that unexpected and detrimental side effects are common. Further studies/modelling would be required to investigate these risks. Synergistic interaction with Invercargill WTP diversion Option. Costs very high with high risk of major detrimental side effects	Reduction of catchment loads would augment benefits of diversion, but significantly improved trophic outcomes for NRE would require substantial catchment improvement. Similar considerations likely for Mataura / Fortrose estuary system

Option	Environmental issue	Benefits	Likelihood of success/feasibility	Catchment remediation
Diversion of effluent from the Invercargill wastewater treatment plant from NRE	NRE eutrophication	Reductions of 3-46% of DIN, 19-76% of DRP concentrations in summer in NRE. Up to 48% reduction in DIN and 76% reduction in DRP in GEZ's of NRE. Benefits in terms of improved ETI trophic condition likely depending on estuary zone	Realistic potential for improvement in estuary trophic state (likely to shift to less eutrophic ETI condition band), especially if combined with moderate improvement in catchment-derived loads. Would interact positively macroalgal, seagrass and cockle Options. Further studies / modelling would be required to investigate these possibilities. Synergistic with Oreti River diversion Option. Would require expensive infrastructure upgrades (for example, WTP ocean outfall construction). Would have only positive environmental side-effects within NRE	Moderate reduction of catchment nutrient loads would augment benefits of diversion, potentially leading to improved trophic outcomes for NRE

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6 References

Adams, N.M. 1994. *Seaweeds of New Zealand*. Canterbury University Press. Christchurch. 360 pp.

Adkins, S.C. 2012. Population biology and restoration of intertidal cockle beds University of Canterbury

Aldridge, J.N., and M. Trimmer. 2009. Modelling the distribution and growth of 'problem' green seaweed in the Medway estuary, UK. *Hydrobiologia* 629: 107-122.10.1007/s10750-009-9760-6

Atkinson, M.J., and S.V. Smith. 1983. C:N:P ratios of benthic marine plants. *Limnology and Oceanography* 28: 568-574

Barr, N., J. Zeldis, K. Scheuer, and D. Schiel. 2019. Macroalgal indicators of recovery from eutrophication in a tidal lagoon following wastewater diversion and earthquake disturbance. *Estuaries and Coasts (in press)*

Bartholdy, J. 2000. Process controlling import of fine-grained sediment to tidal areas: a simulation model. Coastal and Estuarine Environments: sedimentology, geomorphology and geoarchaeology. Geological Society Special Publication 13-30.

Batuik, R., P. Bergstrom, M. Kemp, E. Koch, L. Murray, C. Stevenson, R. Bartleson, V. Carter, N. Rybicki, J. Landwehr, C. Gallegos, L. Karrh, M. Naylor, D. Wilcox, K. Moore, M. Ailstock, and M. Teichberg. 2000. Chesapeake Bay Submerged Aquatic Vegetation Water Quality and Habitat-Based Requirements and Restoration Targets: A Second Technical Synthesis A Watershed Partnership.

Blakely, R.J. 1973. Sedimentation in the New River Estuary

Bolton-Ritchie, L. 2015. The sediments and biota within the Estuary of the Heathcote and Avon Rivers/Ihutai 2007-2013. Environment Canterbury Technical Report R15/46 102 pp.

Bolton-Ritchie, L., and M. Main. 2005. Nutrient water quality Avon-Heathcote Estuary/Ihutai: Inputs, concentrations and potential effects. Environment Canterbury Technical Report U05/71 84 pp.

Boorman, L.A., A. Garbutt, and D. Barratt. 1998. The role of vegetation in determining patterns of the accretion of salt marsh sediment. In *Sedimentary Processes in the Intertidal Zone*, ed. K.S. Black, D.M. Paterson and A. Cramp, 389-399. London: Geological Society.

Borja, Á., D. Dauer, M. Elliott, and C. Simenstad. 2010. Medium- and Long-term Recovery of Estuarine and Coastal Ecosystems: Patterns, Rates and Restoration Effectiveness. *Estuaries and Coasts* 33: 1249-1260

Braeckman, U., M.Y. Foshtomi, D. Van Gansbeke, F. Meysman, K. Soetaert, M. Vincx, and J. Vanaverbeke. 2014. Variable Importance of Macrofaunal Functional Biodiversity for Biogeochemical Cycling in Temperate Coastal Sediments. *Ecosystems* 17: 720-737.10.1007/s10021-014-9755-7

Burge, P.I. 2007. Sedimentation in the Avon-Heathcote Estuary/Ihutai – an analysis of past and present studies. 49 pp. https://drive.google.com/file/d/0B43syqhCcNE_ZTUyMGVhNGMtZjEzZC00MjE4LTg0N2QtZmY3NDdiOGRlYjk2/view?num=50&sort=name&layout=list&pli=1

Cardoso, P.G., M. Bankovic, D. Raffaelli, and M.A. Pardal. 2007. Polychaete assemblages as indicators of habitat recovery in a temperate estuary under eutrophication. *Estuarine, Coastal and Shelf Science* 71: 301-308.https://doi.org/10.1016/j.ecss.2006.08.002

Cardoso, P.G., S. Leston, T.F. Grilo, M.D. Bordalo, D. Crespo, D. Raffaelli, and M.A. Pardal. 2010. Implications of nutrient decline in the seagrass ecosystem success. *Marine Pollution Bulletin* 60: 601-608.https://doi.org/10.1016/j.marpolbul.2009.11.004

Christensen, K., and C. Doscher. 2010. The interaction of river engineering and geomorphology in the Lower Wairau River, Marlborough, New Zealand. *J Hydrol New Zeal* 49: 79–98

Clark, D.E., J.E. Hewitt, C.A. Pilditch, and J.I. Ellis. 2019. The development of a national approach to monitoring estuarine health based on multivariate analysis. *Marine Pollution Bulletin*: 110602.https://doi.org/10.1016/j.marpolbul.2019.110602

Cummings, V., J. Hewitt, J. Halliday, and G. Mackay. 2007. Optimizing the success of Austrovenus stutchburyi restoration: preliminary investigations in a New Zealand estuary. *Journal of Shellfish Research* 26: 89-100

De Winton, M. 2019. Vegetation status in Waituna Lagoon: summer 2019. NIWA Client Report 18 pp

de Winton, M., and A. Taumoepeau. 2017. Macrophyte monitoring in Waituna Lagoon - summer 2017. NIWA Client Report 2017381HN. 42 pp.

Deegan, L.A. 2002. Lessons Learned: The Effects of Nutrient Enrichment on the Support of Nekton by Seagrass and Salt Marsh Ecosystems. *Estuaries and Coasts (in press)* 25: 727-742

Duarte, C.M., D.J. Conley, J. Carstensen, and M. Sánchez-Camacho. 2009. Return to Neverland: Shifting Baselines Affect Eutrophication Restoration Targets. *Estuaries and Coasts* 32: 29-36.10.1007/s12237-008-9111-2

Duarte, C.M., and D. Krause-Jensen. 2018. Intervention Options to Accelerate Ecosystem Recovery From Coastal Eutrophication. *Frontiers in Marine Science* 5.10.3389/fmars.2018.00470

Eisma, D., and K.S. Dijkema. 1997. *The influence of salt marsh vegetation on sedimentation*. CRC Press. Boca Raton. 403-414.

Friedrichs, C.T., and J.E. Perry. 2001. Tidal Salt Marsh Morphodynamics: A Synthesis. *Journal of Coastal Research* 27: 7-37

Gibb, J.G. 1985. A Cut to the Sea? Predicted impacts of either a cut, or spillway to the sea to relieve flooding in the Taramoana Basin. Report to the Southland Catchment Board 17 pp.

Gibson, K., and I.D. Marsden. 2016. Seagrass *Zostera muelleri* in the Avon-Heathcote Estuary/Ihutai, summer 2015–2016. Estuarine Research Report 40 pp. University of Canterbury, Christchurch

Haddadchi, A., A. Willsman, and M. Hicks. 2019. Internal data report for suspended sediment measurements on the Oreti River by June 2019.

Hawes, I., and R.O. O'Brien. 2000. Contribution to the assessment of the effects of the Bromley oxidation ponds effluent on the Avon-Heathcote Estuary: modelling the growth of sea lettuce in the estuary. NIWA Client Report CHC00/19 57. NIWA.

Hawes, I., and R. Smith. 1995. Effect of current velocity on the detachment of thalli of *Ulva lactuca* (Chlorophyta) in a New Zealand estuary. *Journal of phycology* 31: 875-880

Hicks, M., A. Semadeni-Davies, A. Haddadchi, U. Shankar, and D. Plew. 2019. Updated sediment yield estimator for New Zealand. NIWA Client Report 2018341CH. NIWA Project MFE18502, Prepared for Ministry for the Environment 190 pp.

Humphries, R., and S. Robinson. 1995. Assessment of the success of the Peel-Harvey estuary system management strategy - a Western Australian attempt at integrated Catchment Management. *Water Science and Technology* 32: 255-264.10.2166/wst.1995.0619

Irlandi, E.A. 1997. Seagrass patch size and survivorship of an infaunal bivalve. *Oikos* 78: 511-518

Kemp, M.W., R. Batleson, P. Bergstrom, V. Carter, Brendan R., C.L. Gallegos, W. Hunley, L. Karrh, E.W. Koch, J.M. Landwehr, K. Moore, A.,, L. Murray, M. Naylor, N.B. Rybicki, S.J. Court, and D.J. Wilcox. 2004. Habitat requirements for submerged aquatic vegetation in Chesapeake Bay: Water quality, light regime, and physical-chemical factors. *Estuaries* 27: 363-377.10.1007/bf02803529

LagoonTechnicalGroup. 2013. Ecological guidelines for Waituna Lagoon.

Lillebø, A.I., J.M. Neto, I. Martins, T. Verdelhos, S. Leston, P. Cardoso, S. Ferreira, J. Marques, and M. Pardal. 2005. Management of a shallow temperate estuary to control eutrophication: The effect of hydrodynamics on the system’s nutrient loading. *Estuarine, Coastal and Shelf Science* 65: 697-707.10.1016/j.ecss.2005.07.009

Longphuirt, S., S. O'Boyle, R. Wilkes, T. Dabrowski, and D. Stengel. 2015. Influence of Hydrological Regime in Determining the Response of Macroalgal Blooms to Nutrient Loading in Two Irish Estuaries. *Estuaries and Coasts* 39.10.1007/s12237-015-0009-5

Manson, S., and N. Pinnington. 2012. Paull Holme Strays Managed Realignment: Measure analysis 31 in the framework of the Interreg IVB project TIDE https://www.tide-toolbox.eu/pdf/measures/Paull_Holme_Strays.pdf1211

Marsden, I.D., and S.C. Adkins. 2010. Current status of cockle bed restoration in New Zealand *Aquacult Int* 18: 83–97

Marsden, I.D., and R.M. Pilkington. 1995. Spatial and temporal variations in the condition of *Austrovenus stutchburyi* Findlay, 1927 (Bivalvia:Veneridae) from the Avon-Heathcote Estuary, Christchurch. *NZ Nat Sci* 22: 57–67

Matheson, F., M. Oliver, G. McKay, M. Diniss, and R. Griffiths. 2017a. Is light climate the main driver of seagrass growth and persistence? In New Zealand Coastal Society Conference. Tauranga.

Matheson, F., and S. Wadhwa. 2012. Seagrass in Porirua Harbour Preliminary assessment of restoration potential. NIWA Client Report HAM2012-037. 35pp.

Matheson, F., I. Zabarte-Maeztu, R. Davies-Colley, G. Mackay, C. Middleton, M. Oliver, M. Diniss, R. Griffiths, R. Eyre, and R. Ovenden. in prep. Effects of estuary muddiness on seagrass light climate and restoration. *Marine Pollution Bulletin*

Matheson, F.E., J. Reed, V.M. Dos Santos, G. Mackay, and V.J. Cummings. 2017b. Seagrass rehabilitation: successful transplants and evaluation of methods at different spatial scales. *New Zealand Journal of Marine and Freshwater Research* 51: 96-109.10.1080/00288330.2016.1265993

Measures, R. 2016. New River Estuary Hydrodynamic Modelling. Model build, calibration and tracer simulations. NIWA Client Report 2016077CH. 52 pp.

Measures, R., and G. Horrell. 2013. Opening and closure of Waituna Lagoon Review of recent investigations. NIWA Client Report CHC2013-039. 31 pp.

Morrison, M. 2003. A review of the natural marine features and ecology of Whangarei Harbour. NIWA Client Report 60 pp.

Orth, R.J., and J. van Montfrans. 1984. Epiphyte-seagrass relationships with an emphasis on the role of micrograzing: a review. *Aquatic Botany* 18: 43-69

Painting, S.J., M.J. Devlin, S.J. Malcolm, E.R. Parker, D.K. Mills, C. Mills, and K. Winpenny. 2007. Assessing the impact of nutrient enrichment in estuaries: susceptibility to eutrophication. *Marine Pollution Bulletin* 55: 74-90

Park, S., and R. Donald. 1994. Environment Bay of Plenty Tauranga Harbour Regional Plan Environmental Investigations. Ecology of Tauranga Harbour. Environment Bay of Plenty Environmental Report 94/8. 177 pp.

Pennings, S.C., L.E. Stanton, and J.S. Brewer. 2002. Nutrient Effects on the Composition of Salt Marsh Plant Communities Along the Southern Atlantic and Gulf Coasts of the United States. *Estuaries*: 1164–1173

Petersen, J.K., J.W. Hansen, M.B. Laursen, P. Clausen, J. Carstensen, and D.J. Conley. 2008. Regime shift in a coastal marine ecosystem. *Ecological Applications* 18: 497-510.10.1890/07-0752.1

Peterson, B.J., and K.L.J. Heck. 2001. Positive interactions between suspension-feeding bivalves and seagrass – a facultative mutualism. *Marine Ecology Progress Series* 213: 143-155

Pethick, J. 1984. *An Introduction to Coastal Geomorphology*. London.

Plew, D., B. Dudley, U. Shankar, and J. Zeldis. 2018a. Assessment of the eutrophication susceptibility of New Zealand Estuaries. NIWA Client Report 2018206CH 61 pp.

Plew, D., J.R. Zeldis, B.D. Dudley, A.L. Whitehead, L.M. Stevens, B.M. Robertson, and B.P. Robertson. 2019. Assessing the eutrophic susceptibility of New Zealand Estuaries. *Estuaries and Coasts (in review)*

Plew, D.R., J.R. Zeldis, U. Shankar, and A.H. Elliott. 2018b. Using Simple Dilution Models to Predict New Zealand Estuarine Water Quality. *Estuaries and Coasts* 41: 1643-1659.10.1007/s12237-018-0387-6

Reed, J., A. Schwarz, A. Gosai, and M. Morrison. 2004. Feasibility study to investigate the replenishment/ reinstatement of seagrass beds in Whangarei Harbour – Phase 1. NIWA Client Report.

Reusch, T.B.H., A.R.O. Chapman, and J.P. Groger. 1994. Blue mussels Mytilus edulis do not interfere with eelgrass Zostera marina but fertilise shoot growth through biodeposition. *Marine Ecology Progress Series* 108: 265-282

Reusch, T.B.H., and S.L. Williams. 1998. Variable responses of native eelgrass Zostera marina to a non-indigenous bivalve Musculista senhousia. *Oecologia* 113: 428-442

Reynolds, L.K., M. Waycott, K.J. McGlathery, and R.J. Orth. 2016. Ecosystem services returned through seagrass restoration. *Restoration Ecology* 24: 583-588.10.1111/rec.12360

Robertson, B., and C. Savage. 2018. Mud-entrained macroalgae utilise porewater and overlying water column nutrients to grow in a eutrophic intertidal estuary. *Biogeochemistry* 139: 53-68.10.1007/s10533-018-0454-x

Robertson, B., L. Stevens, B. Robertson, J. Zeldis, M. Green, A. Madarasz-Smith, D. Plew, R. Storey, T. Hume, and M. Oliver. 2016a. NZ Estuary Trophic Index Screening Tool 2. Determining Monitoring Indicators and Assessing Estuary Trophic State. Prepared for Envirolink Tools Project: Estuarine Trophic Index, MBIE/NIWA Contract No: C01X1420 68 pp.

Robertson, B.M., P.A. Gillespie, R.A. Asher, S. Frisk, N.B. Keeley, G.A. Hopkins, S.J. Thompson, and B.J. Tuckey. 2002. Estuarine Environmental Assessment and Monitoring: A National Protocol Sustainable Management Fund 5096 Part A. 93p. Part B. 159p. Part C. 140p plus field sheets.

Robertson, B.M., L.M. Stevens, N. Ward, and B.P. Robertson. 2017. Condition of Southland's Shallow, Intertidal Dominated Estuaries in Relation to Eutrophication and Sedimentation: Output 1: Data Analysis and Technical Assessment - Habitat Mapping, Vulnerability Assessment and Monitoring Recommendations Related to Issues of Eutrophication and Sedimentation. Report prepared by Wriggle Coastal Management for Environment Southland 172 pp.

Robertson, B.P., J.P.A. Gardner, and C. Savage. 2015. Macrobenthic–mud relations strengthen the foundation for benthic index development: A case study from shallow, temperate New Zealand estuaries. *Ecological Indicators* 58: 161-174.http://dx.doi.org/10.1016/j.ecolind.2015.05.039

Robertson, B.P., C. Savage, J.P.A. Gardner, B.M. Robertson, and L.M. Stevens. 2016b. Optimising a widely-used coastal health index through quantitative ecological group classifications and associated thresholds. *Ecological Indicators* 69: 595-605.http://dx.doi.org/10.1016/j.ecolind.2016.04.003

Robertson, H.A., A.G. Ausseil, B. Rance, H. Betts, and E. Pomeroy. 2019. Loss of wetlands since 1990 in Southland, New Zealand. *New Zealand Journal of Ecology* 43: 1-9

Santelices, B., and M.S. Doty. 1989. A review of Gracilaria farming. *Aquaculture* 78: 95–133

Schallenberg, M., S.T. Larned, S. Hayward, and C. Arbuckle. 2010. Contrasting effects of managed opening regimes on water quality in two intermittently closed and open coastal lakes. *Estuarine, Coastal and Shelf Science* 86: 587-597.10.1016/j.ecss.2009.11.001

Spigel, B., and J. Zeldis. 2004. Assessment of effects of Bromley sewage effluent on phytoplankton growth at the proposed Pegasus Bay ocean outfall site: Update and revisions. CHC2004-058. 22 pp.

Stevens, L., and B. Robertson. 2016. Porirua Harbour. Intertidal macroalgal monitoring 2015/16. Report prepared for Greater Wellington Regional Council 12 pp.

Stevens, L.M. 2018a. New River Estuary: 2018 Macroalgal Monitoring. Report prepared by Wriggle Coastal Management for Environment Southland 29 pp.

Stevens, L.M. 2018b. Tokomairiro Estuary: Broad Scale Habitat Mapping. 40 p. Report prepared by Wriggle Coastal Management for Otago Regional Council.

Stevens, L.M. 2018c. Whanganui Inlet : Mapping of Historical Seagrass Extent. 10 pp.
Stevens, L.M., and B.M. Forrest. 2019. Broad scale intertidal habitat mapping of Nelson Haven. Salt Ecology Report 022. 42 pp.
Stevens, L.M., and B.P. Robertson. 2017. Fortrose (Toetoes) Estuary 2016: Broad Scale Substrate, Macroalgae and Seagrass Mapping. Report prepared by Wriggle Coastal Management for Environment Southland 28 p.
Sutula, M. 2011. Review of Indicators for Development of Nutrient Numeric Endpoints in California Estuaries. Southern California Coastal Water Research Project Technical Report 646 269.
Tanner, C.C., J. Sukias, M. de Winton, S. Larned, D. Sutherland, and A. Hughes. 2015. Wetland Nutrient Attenuation Assessment for Inflows to Te Waihora / Lake Ellesmere. NIWA Client Report HAM2015-040. 68 pp.
Thompson, R.M., and G.R. Ryder. 2003. Waituna Lagoon: summary of existing knowledge and identification of knowledge gaps. Science for Conservation 37 pp. Department of Conservation.
Thoms, M.C. 1981. Sedimentation in the New River Estuary, Southland, University of Canterbury
Townsend, M., and D. Lohrer. 2015. ANZECC Guidance for Estuary Sedimentation. NIWA Client Report HAM2015-096. 45 pp.
Tu Do, V., X. de Montaudouin, N. Lavesque, H. Blanchet, and H. Guyard. 2011. Seagrass colonization: knock-on effects on zoobenthic community, populations and individual health. *Estuarine, Coastal and Shelf Science* 95: 458-469
Valiela, I., J. McClelland, P. Hauxwell, P. Behr, D. Hirsch, and K. Foreman. 1997. Macroalgal blooms in shallow estuaries: controls and ecophysiological and ecosystem consequences. *Limnology and Oceanography* 42
Van de Heide, T., L.L. Govers, J. de Fouw, H. Olff, M. van der Geest, M.M. van Katwijk, T. Piersma, J. van de Koppel, B.R. Silliman, A.J. Smolders, and J.A. van Gils. 2012. A three-stage symbiosis forms the foundation of seagrass ecosystems. *Science* 336: 1432-1434
Walsh, J., R. Measures, J. Bind, and A. Haddadchi. 2016. Waituna Lagoon level impacts on land drainage and inundation Investigation stages 1 and 2. NIWA Client Report CHC2016-010. 122 pp.
WFD-UKTAG. 2014. UKTAG Transitional and Coastal Water Assessment Method Macroalgae Opportunistic Macroalgal Blooming Tool. Water Framework Directive – United Kingdom Technical Advisory Group. http://www.wfduk.org/sites/default/files/Media/
Wijnhoven, S., V. Escaravage, E. Daemen, and H. Hummel. 2010. The Decline and Restoration of a Coastal Lagoon (Lake Veere) in the Dutch Delta. *Estuaries and Coasts* 33: 1261-1278.10.1007/s12237-009-9233-1
Zabarte-Maeztu, I., F. Matheson, I. Hawes, and M. Manley-Harris. 2019. Sediment-effect thresholds for the seagrass, Zostera muelleri: a case study in Porirua Harbour, New Zealand. In New Zealand Marine Sciences Society Conference. Dunedin.
Zabarte-Maeztu, I., F. Matheson, I. Hawes, and M. Manley-Harris. in prep. Multi-faceted effects of fine sediment on seagrass meadows and sediment grain size as an indicator of suitable growing conditions: a case study in Porirua Harbour, New Zealand. *Estuaries and Coasts*
Zeldis, J., and M. Gall. 1999. Assessment of Bromley Sewage effluent on phytoplankton growth at the proposed Pegasus Bay ocean outfall site. NIWA Client Report CHC99/30.
Zeldis, J., D. Plew, A. Whitehead, A. Madarasz-Smith, M. Oliver, L. Stevens, B. Robertson, O. Burge, and B. Dudley. 2017a. The New Zealand Estuary Trophic Index (ETI) Tools: Tool 1 - Determining Eutrophication Susceptibility https://shiny.niwa.co.nz/Estuaries-Screening-Tool-1/730
Zeldis, J., A. Whitehead, D. Plew, A. Madarasz-Smith, M. Oliver, L. Stevens, B. Robertson, R. Storey, O. Burge, and B. Dudley. 2017b. The New Zealand Estuary Trophic Index (ETI) Tools: Tool 2 - Assessing Estuary Trophic State using Measured Trophic Indicators https://shiny.niwa.co.nz/Estuaries-Screening-Tool-2/731

Zeldis, J.R., C. Depree, C. Gongol, P.M. South, A. Marriner, and D.R. Schiel. 2019. Trophic Indicators of Ecological Resilience in a Tidal Lagoon Estuary Following Wastewater Diversion and Earthquake Disturbance. *Estuaries and Coasts*.10.1007/s12237-019-00637-8

Appendix A Minutes of June 2019 workshop

Minutes from Scoping Workshop for Remediation Options for Southland Estuaries Project

20 June 2019

Minutes prepared by John Zeldis

Workshop attendees (by Skype): Keryn Roberts, Nick Ward (tba), Kathryn McLachlan (Environment Southland), John Zeldis, Bruce Dudley, Richard Measures, Fleur Matheson (NIWA), Leigh Stevens (Salt Ecology)

Purpose: The purpose of the scoping workshop was to confirm and discuss with Environment Southland the list of remediation options which would be assessed. We took the opportunity to discuss implications and available information regarding the options, and approaches to the reporting. The meeting started at 0945.

Requirements of ES Estuary Remediation Options project

- A report on remediation options for Southland estuaries. It will be targeted toward policy makers, outlining what we know about estuary remediation including the challenges of legacy effects in addition to identifying key knowledge gaps.
- The work will investigate a range of potential remediation options for Southland's estuaries and their marginal habitats, to identify restorative targets, and advise on their viability, including relevant case studies.
- The work will be high-level, and not involve new modelling or other detailed analyses.

Minutes

The meeting started with a discussion of over-arching issues related to the project. Keryn described recent policy-based workshops related to the Coastal Plan review process. A big question arising was what can be done to 1) remediate already heavily impacted estuaries and 2) protect resilience in those estuaries in varied conditions including those still in good ecological health.

Nick indicated that although recommendations around catchment load reduction are 'out of scope', it should be pointed out in the report when an option will likely not be viable without catchment load reduction.

Richard queried the level of specificity of the reporting – should it be specific on an estuary-by-estuary basis or more generic? Nick said where appropriate the applicability across NZ as a whole could be pointed out using cases from Southland or elsewhere, citing likely differences in response between Jacobs River and New River Estuary (NRE) as an example.

Related to this, here the discussion turned to the first option: Removal of macroalgal biomass. John pointed out that the report will be able to assess this solution relative to the size of the problem, especially for NRE where there is a wealth of time series information (Stevens 2018a) on macroalgal extent. Leigh discussed the observation from Jacobs River (Aparima Arm) where macroalgae were removed by natural flushing events, resulting in remobilization of eutrophic sediments and improvements in oxygenation, etc.

This demonstrated potential connectivity of the removal of macroalgal biomass option with the Restoration of sediments option. This raises the possibility that removal of algal biomass could also restore sediments. John pointed this out as a knowledge gap which could be tested in the field using experimental trials. Nick mentioned that the hydraulic character of different estuaries, such as parameterized in Plew et al. (2018), could be a way of ordering estuaries in terms of the balance of flushing power from river inputs to tidal inputs. On this scale, Jacobs R. would be 'halfway' between Fortrose Estuary and NRE, for example.

Leigh said that removal of the reproductive source of the algae (e.g., overwintering plants) could affect subsequent outgrowths and therefor be most effective, although the relationships are not known and were pointed out as a knowledge gap. A case study in point is modelling in Avon-Heathcote Estuary showing the importance of overwintering biomass on subsequent outgrowth (Hawes 2000). Connectivity of algal biomass and flushing effects on sediment health indicated cross-connections between algae- and sediment-related options. The Avon-Heathcote also provides a valuable case study in benefits of maintaining sandy (rather than muddy) sediments for estuary health and ecological resilience (Zeldis et al. 2019).

Physical dredging of sediments was discussed (note: moved up from later in the meeting). Leigh has a good appreciation of the extent and depth of degraded sediments (Robertson et al. 2017) – likely to be hundreds of hectares in extent and up to 1 m deep for historically modified sediments and up to 20-30 cm deep for recent, highly degraded sediments. Leigh suggested that eutrophic sediments could be physically mobilized/dispersed with high pressure hosing. This raised the statement (Nick/Keryn) that our discussions of options should mention that there will be a balance of positive and negative environmental outcomes associated with them and that while some may seem very destructive, they could have net-positive environmental outcomes. This is important in the context of the RMA.

The discussion then moved to the Restoration of seagrass cover option. Fleur described her Whangarei Hbr research case study (Matheson et al. 2017) showing positive outcomes for restoration upon reductions in sediment discharges, associated with cessation of industrial sediment discharges and dumping of dredge spoil. There has been substantial seagrass recovery through time, with anecdotal WQ improvement over two decades. Transplanting in 2008 and 2012 has worked well. Seagrass has been re-established at two locations and overall 40% recovery in the harbour (Matheson et al. 2017). In contrast, the Porirua Hbr case has not improved in terms of seagrass extent (Matheson and Wadhwa 2012), and while the light climate appears not at fault, there has been no improvement in WQ, suggesting the latter as causative. Fleur is overseeing a PhD project on Porirua Hbr where student (Inigo Zabarte-Maeztu) is exploring the multi-faceted effects of fine sediment (mud) on seagrass: i.e., on light climate, plant smothering and sediment chemistry (further details below). In the Avon-Heathcote case, work by Gibson and Marsden (2016) showed rapid improvement in seagrass extent (40%) following the diversion of Christchurch WTP effluent from the estuary. Richard raised the case of Te Waihora (Mary de Winton and Deb Hofstra) which has yet to show positive restoration responses – pointed out as a 'difficult environment' by Fleur due to system size, turbidity and wave climate. Leigh described Nelson Haven estuary where whilst water clarity/light climate is good, sediment settling on seagrass plants is preventing improved seagrass condition. Again, the cross-connections among the options was pointed out: relationships of sediment/macroalgal biomass and clarity, impacting seagrass.

Nick suggested that cockle bed restoration could be considered an option for estuary remediation/improved water quality. Leigh noted the Ruataniwha Inlet/Pakawau estuary as an example of cockle transplantation. Bruce mentioned Ngai Tahu efforts to restore cockle beds in Lyttleton Hbr, as yet unsuccessful (Andre Konia pers. comm.). Fleur also cited work by Vonda Cummings in Whangarei Hbr on cockle transplantation (ca. 2008), at sites adjacent to seagrass restoration trials, which had some success and restoration guidelines were produced (see NIWA website). Fleur and Judi Hewitt have done some (unpublished) work on cockle-seagrass associations in Kaipara Hbr which suggest they frequently occur together, although at very high densities one may exclude the other to some extent. Nick mentioned the use of macrobenthic indicators as integrators of multistressor thresholds, an example of which could be mud-macrobenthic health relationships (Robertson et al. 2015; Robertson et al. 2016). Leigh described interactions of cockles, seagrass and muddiness: too muddy causes small cockles, causes shallow bioturbation, causes no seagrass, implying the success of seagrass restoration is proportional to muddiness and that muddiness thresholds could predict success. Thresholds of 23 % mud have been suggested by PhD student (Inigo Zabarte-Maeztu) work in Porirua Harbour (to be presented at NZMSS conference in Dunedin), which concurs with Wriggle work but is higher than a threshold (13%) suggested in Tauranga Hbr (Steven Park).

The conversation then moved onto engineered options, including evaluation of the consequences of partial diversion of Oreti River. Nick indicated the effects on estuary dilution, and Richard said that along with nutrient loads the major effect would be on sediment load (especially into the Waihopai Arm of NRE) and on sediment and nutrient flushing power. Nick further pointed out the unknowns associated with longshore drift of sediment / contaminants out of the cutting at the coast. The seasonal impacts on NRE nutrient status under different Oreti R. flow conditions resulting are also unknown. There are therefore competing influences and effects and knowledge gaps that would be addressable with Richard's hydrodynamic and nutrient modelling (Measures 2016). Case studies that involve channel diversion for flood control include Wairau River (Blenheim) and Heathcote River (Christchurch). The former is known to have caused degraded conditions in the channel system. Leigh mentioned the case of the Makatu R. diversion (BoP) where flows have been restored to improve degraded conditions. These cases again point out the balance between sediment delivery and removal by flushing as a knowledge gap that is addressable and specific to the functionality of individual systems (Nick). There are no quick wins here without consequences (Richard).

Bruce asked if the option of Evaluation of the impact of diversion of effluent from the Invercargill wastewater treatment plant was in scope of the project. It is, and its effects would be relatively easy to assess, using Richard's modelling (both in 'back of envelope' for this report and using more detailed examination in future work). The case study of the Christchurch WTP diversion is useful in terms of the description of subsequent responses of Avon-Heathcote Estuary ecology wherein trophic indicators (Zeldis et al. 2019) macroalgal indices (Barr et al. 2019) and ETI Tool scores improved considerably. However, it was noted that the Avon-Heathcote case involved a 90% reduction in N load: this will not happen in the case of Invercargill WTP diversion, where catchment inputs will likely continue to dominate. Thus, the cost/benefit of such infrastructure would need careful consideration (in future work). Keryn pointed out that this option is specific to NRE and it is an important one for the study to address. The issue of additional loads of stormwater was raised by Nick, and the metals contamination component of this at least will be addressed by Jenni Gadd's report, in parallel with this one (due in August 2019). The diversion of the wastewater will also have negligible bearing on sediment loading issues, which are riverine (Keryn).

Restoration of Estuary margins, Salt marsh estimated natural state cover and Riparian margins and salt marsh restoration engineering solutions were considered next. Leigh said the reality of sea level rise (SLR) on any solutions involving reclamation would need to made clear and this suggests that restoration should proceed landward rather than seaward. The report could say what extent of restoration would be required to achieve degrees of improvement, but that these will need to be estuary specific (e.g., Stevens (2018b)). Narrative can be provided on the importance of surrounding salt marsh/wetlands for ecosystem services such as land stabilization, sediment and nutrient attenuation and adaptation to SLR along with knowledge gaps in relation to these services. There will be plenty of case studies available in the literature (Nick) including those specific to Southland (Robertson et al. 2019).

It was clear that the above options intersect strongly with the Assessment of engineered estuary mouth openings to improve resilience of less degraded Southland estuaries in need of protection. The case of opening and closure of Waituna Lagoon has been examined by Richard (Measures and Horrell 2013) and others (de Winton and Taumoepeau 2017; Schallenberg et al. 2010; Thompson and Ryder 2003) which can advise on optimal times and effects for opening, with respect to adjacent wetland inundation and Ruppia bed health and phytoplankton (Fleur). Other case studies include those for wetland construction bordering Te Waihora (Tanner et al. 2015).

Leigh noted that flood control issues weigh heavily on Waituna Lagoon opening scheduling, indicating the interaction of adjacent land-holdings with ecological performance of the lagoon. Richard noted similar social / hydrological dynamics for Te Waihora. Nick and Keryn provided important political background information here: there are proposals being discussed involving land purchase in the Waituna surrounding area to enable retirement and re-establishment of wetland ecology. This would reduce the imperative of Waituna openings for flood control and offer the opportunity for compromise between land use and ecological restoration. Nick will undertake to provide documentation on these efforts as a case study. Kathryn indicated the relevance of this to decision making in the Regional Coastal Plan which is charged with estimating 'Net Environmental Benefit'. While that is out of scope for the present report, the report's narrative around ecosystem service benefits of healthy salt marsh/riparian condition will be useful in estimating benefit.

The discussion concluded with higher level uses for the report. Nick mentioned its use in section 32 (RMA) reporting and best option evaluations: weighing up risks and benefits. Richard pointed out its value in 'managing expectations' of the options. John described it as a road map for further investigations, where needed. Richard suggested that the work not consider 'biological options' and 'engineered options' as distinct, as was made clear by the cross-connections between the options revealed by the morning's discussion. This was agreed.

John indicated he would prepare the workshop minutes and distribute.

The workshop adjourned at 12pm.

<u>Workshop References</u>

Barr, N., J. Zeldis, K. Scheuer, and D. Schiel. 2019. Macroalgal indicators of recovery from eutrophication in a tidal lagoon following wastewater diversion and earthquake disturbance. Estuaries and Coasts (in review).

de Winton, M., and A. Taumoepeau. 2017. Macrophyte monitoring in Waituna Lagoon - summer 2017. NIWA Client Report 2017381HN: 42 pp.

Gibson, K., and I.D. Marsden. 2016. Seagrass Zostera muelleri in the Avon-Heathcote Estuary/Ihutai, summer 2015–2016. Estuarine Research Report: 40 pp. Christchurch, New Zealand: University of Canterbury

Hawes, I., O'Brien R.O. 2000. Contribution to the assessment of the effects of the Bromley oxidation ponds effluent on the Avon-Heathcote Estuary: modelling the growth of sea lettuce in the estuary. . NIWA Client Report CHC00/19: 57. Christchurch: NIWA.

Matheson, F., and S. Wadhwa. 2012. Seagrass in Porirua Harbour Preliminary assessment of restoration potential. NIWA Client Report HAM2012-037: 35pp.

Matheson, F.E., J. Reed, V.M. Dos Santos, G. Mackay, and V.J. Cummings. 2017. Seagrass rehabilitation: successful transplants and evaluation of methods at different spatial scales. New Zealand Journal of Marine and Freshwater Research 51: 96-109.

Measures, R. 2016. New River Estuary Hydrodynamic Modelling. Model build, calibration and tracer simulations. NIWA Client Report 2016077CH: 52 pp.

Measures, R., and G. Horrell. 2013. Opening and closure of Waituna Lagoon Review of recent investigations. NIWA Client Report CHC2013-039: 31 pp.

Plew, D.R., J.R. Zeldis, U. Shankar, and A.H. Elliott. 2018. Using Simple Dilution Models to Predict New Zealand Estuarine Water Quality. Estuaries and Coasts 41: 1643-1659.

Robertson, B.M., L.M. Stevens, N. Ward, and B.P. Robertson. 2017. Condition of Southland's Shallow, Intertidal Dominated Estuaries in Relation to Eutrophication and Sedimentation: Output 1: Data Analysis and Technical Assessment - Habitat Mapping, Vulnerability Assessment and Monitoring Recommendations Related to Issues of Eutrophication and Sedimentation. Report prepared by Wriggle Coastal Management for Environment Southland: 172 pp.

Robertson, B.P., J.P.A. Gardner, and C. Savage. 2015. Macrobenthic–mud relations strengthen the foundation for benthic index development: A case study from shallow, temperate New Zealand estuaries. Ecological Indicators 58: 161-174.

Robertson, B.P., C. Savage, J.P.A. Gardner, B.M. Robertson, and L.M. Stevens. 2016. Optimising a widely-used coastal health index through quantitative ecological group classifications and associated thresholds. Ecological Indicators 69: 595-605.

Robertson, H.A., A.G. Ausseil, B. Rance, H. Betts, and E. Pomeroy. 2019. Loss of wetlands since 1990 in Southland, New Zealand. New Zealand Journal of Ecology 43: 1-9.

Schallenberg, M., S.T. Larned, S. Hayward, and C. Arbuckle. 2010. Contrasting effects of managed opening regimes on water quality in two intermittently closed and open coastal lakes. Estuarine, Coastal and Shelf Science 86: 587-597.

Stevens, L.M. 2018a. New River Estuary: 2018 Macroalgal Monitoring. Report prepared by Wriggle Coastal Management for Environment Southland: 29 pp.

Stevens, L.M. 2018b. Tokomairiro Estuary: Broad Scale Habitat Mapping. : 40 p. Report prepared by Wriggle Coastal Management for Otago Regional Council.

Tanner, C.C., J. Sukias, M. de Winton, S. Larned, D. Sutherland, and A. Hughes. 2015. Wetland Nutrient Attenuation Assessment for Inflows to Te Waihora / Lake Ellesmere. NIWA Client Report HAM2015-040: 68 pp.

Thompson, R.M., and G.R. Ryder. 2003. Waituna Lagoon: summary of existing knowledge and identification of knowledge gaps. Science for Conservation: 37 pp. Wellington: Department of Conservation.

Zeldis, J., C. Depree, C. Gongol, A. Marriner, P. South, and D. Schiel. 2019. Trophic indicators of ecological resilience in a tidal lagoon estuary following wastewater diversion and earthquake-driven disturbance. Estuaries and Coasts (in review).