Consequences of Inaction

Potential ramifications of delaying proposed nutrient source reductions for New Zealand rivers, lakes, and estuaries

Prepared for Ministry for the Environment

March 2020

Prepared by:
Elizabeth Graham
Ben Woodward
Bruce Dudley
Leigh Stevens
Piet Verburg
John Zeldis
Deborah Hofstra
Fleur Matheson
Sandy Elliott

For any information regarding this report please contact:

Sandy Elliott
Principal Scientist
Catchment Processes
+64-3-859 1839
sandy.elliott@niwa.co.nz

National Institute of Water & Atmospheric Research Ltd
PO Box 11115
Hamilton 3251

Phone +64 7 856 7026

NIWA CLIENT REPORT No: 2020046HN
Report date: March 2020
NIWA Project: MFE20203

Quality Assurance Statement		
	Reviewed by:	Clive Howard-Williams
	Formatting checked by:	Alison Bartley
	Approved for release by:	Scott Larned

Contents

Figures

Executive summary

Background and scope

The Government has developed the Essential Freshwater Package (the Package) to 'improve and maintain sustainable outcomes from freshwater management.' To support Ministers' decision-making on the package, the Ministry is undertaking impact assessments to better understand the environmental, economic, social and cultural impacts of the proposed regulations on freshwater quality, and how rivers are used and enjoyed. In support of these assessments, the Ministry requested NIWA to provide 'descriptive information on the ecological implications of delaying introduction of mitigation measures, in terms of irreversibility or increased recovery times following interventions, for rivers, lakes and estuaries.' Following further consultation with the Ministry, it was agreed that this would be addressed by answering the following questions for each of those three classes of ecosystem:

If nutrients were to be reduced at a later date, will leaving nutrient loads/concentrations as they currently are (rather than reducing them):

1. Increase the time necessary for remediation and/or recovery?
2. Make it harder to employ other remediation options?
3. Change the remediation methods required?

In addition:

4. How might ecosystem characteristics affect the three issues above?

Wetlands and groundwater ecosystems were excluded from the scope of the assessment, although discussion of nutrient lags in groundwater was included in an additional question:

5. Will delays in reduction in the emissions of nutrients at source (e.g., leaching through the soil profile) will result in greater delays in reductions of loadings to rivers, lakes, and estuaries?

The key findings were:

- Delays in reducing external sources of nutrient inputs will increase the subsequent time for recovery or rivers, lakes and estuaries because:
  - The stocks of nutrients stored in sediments will increase as long as inputs from external sources continue, and the release of nutrients from these internal storages will continue after inputs from external sources are reduced. This applies particularly to lakes and poorly-flushed estuaries, but also can apply to rivers.
  - Native seed banks in lakes become depleted over time (seeds are broken down or buried too deep in the sediment to germinate), thereby delaying or preventing the natural recovery of native macrophytes.
  - Feedbacks such as the enhancement of sediment deposition by river macrophytes and estuarine algae exacerbating nutrient storage effects and prolonging recovery, become more likely with duration of high source loading.

- Risks of development of ecological states that are resistant to rehabilitation (e.g., phytoplankton-dominated shallow lakes, deep lakes with high trophic levels, streams dominated by degradation-tolerant species that block re-establishment of more desirable biota) increase the longer that source reductions are delayed.

- Delaying reduction of nutrient loads will make remediation more difficult because:
  - Seed banks in lakes will be depleted, requiring more planting rather than natural regeneration.
  - Competitive exclusion by degradation-tolerant riverine biota are likely to make remediation efforts less effective and may require additional interventions targeting those species.
  - Additional remediation options may also be required to mitigate feedback effects, such as sediment trapping by macrophytes and estuarine algae – these may not have been necessary if nutrient management action were taken earlier.
  - For lakes, reductions of external loading may need to be reduced lower to achieve a desired trophic state, compared with loadings would be required the achieve that stated if the lake had not been degraded in the first place.
- The degree of degradation and remediation required will vary according to characteristics of the ecosystem:
  - For rivers, the hydrological regime will affect flushing of accumulated sediments and biomass. Smaller systems subject to frequent disturbance will likely recover faster, while poorly-flushed depositional environments are more likely to have delayed responses to remediation.
  - For estuaries, systems with finer sediment, lower flushing power, and lower tidal replacement rate will be more resistant to recovery.
  - For lakes, deep lakes that have elevated trophic states will take longer to respond and will be difficult to remediate due to their large volume and high residence times, while shallow lakes that have flipped to a phytoplankton-dominated state will also be difficult to remediate due to the ecological resilience of the phytoplankton dominated state.
- Delays in reducing nitrogen leaching will result in increased peak loading to streams and protracted recovery for groundwater systems that have yet not responded fully to past increases in loading. This situation is more likely for systems with long groundwater lag times and where denitrification is minimal. This is known to occur in some locations, such as the catchment of Lake Rotorua, but will not be the general case. The spatial occurrence of these conditions is not known at a national scale.

1 Introduction

The Government has developed the Essential Freshwater Package[1] (the Package) to 'improve and maintain sustainable outcomes from freshwater management.' To support Ministers' decision-making on the package, the Ministry is undertaking impact assessments to better understand the environmental, economic, social and cultural impacts of the proposed regulations on freshwater quality, and how rivers are used and enjoyed. In support of these assessments, the Ministry requested NIWA to provide 'descriptive information on the ecological implications of delaying introduction of mitigation measures, in terms of irreversibility or increased recovery times following interventions, for rivers, lakes and estuaries.' Following further consultation with the Ministry, it was agreed that this would be addressed by answering the following questions for each ecosystem:

If nutrients were to be reduced at a later date, will leaving nutrient loads/concentrations as they currently are (rather than reducing them):

1. Increase the time necessary for remediation and/or recovery?
2. Make it harder to employ other remediation options?
3. Change the remediation methods required?

In addition:

4. How might ecosystem characteristics affect the three points above?

Wetlands and groundwater ecosystems were excluded from the scope of the assessment, although discussion of nutrient lags in groundwater was included in an additional question:

5. Will delays in reduction in the emissions of nutrients at source (e.g., leaching through the soil profile) will result in greater delays in reductions of loadings to rivers, lakes, and estuaries?

[1] Ministry for the Environment and Ministry for Primary Industries (2018) Essential Freshwater: Healthy Water, Fairly Allocated. *Wellington: Ministry for the Environment and Ministry for Primary Industries*: 56.

2 Rivers and streams

2.1 River and stream responses to unchanged nutrient loading

In this section we provide an overview of the likely consequences of inaction in reducing current nutrient concentrations/inputs of nutrient loads to New Zealand's rivers and streams.

Question 1. Increased time for recovery?

The time necessary for remediation and/or recovery of river and stream ecological health is likely to increase if nutrient concentrations in New Zealand's rivers and streams remain at current levels. This prediction is based on two distinct but related mechanisms.

The first mechanism is **nutrient cycling, spiralling, and storage**. As nutrients are transported downstream, they cycle between dissolved inorganic and organic forms in the water column, particulate forms adsorbed to sediment, and biomass (typically algae and macrophytes) (Newbold et al. 1981). Nutrients stored in sediments, algae, and macrophytes will serve as future sources of nutrients to the water column even after inputs are reduced (McDowell 2015). This was observed in the Tukituki river following wastewater treatment plant upgrades. The river was suspected to be phosphorus-limited prior to the upgrades, which reduced dissolved reactive phosphorus inputs to the river by over 90%. However, algal biomass remained moderately high and DIN continued to be reduced to near-detection limit 40-50 km downstream, indicating that sufficient phosphorus remained available for nitrogen uptake (Depree et al. 2016). It was subsequently discovered that algal photosynthesis increased pH each day to an extent that chemically immobilised phosphorus was released from sediments in bioavailable form (Wilcock et al. 2016). Increased storage of nutrients in sediments also favours the accrual of macrophyte biomass, which in turn enhances sedimentation within macrophyte beds, further increasing nutrient stores and therefore increasing the time and effort needed for restoration and recovery.

The second mechanism is **nutrient effects on stream biota via food web impacts**. Increased nutrient availability typically increases growth and biomass of algae and macrophytes in rivers and streams by the mechanisms discussed above. Increased nutrient availability can also affect detrital processing by stimulating leaf decomposition and altering carbon retention rates (Rosemond et al. 2015, Dodds and Smith 2016). Changes in nutrient ratios of both algae and detrital material affects the quality of those resources as a food source for stream organisms (Evans-White et al. 2009, Dodds and Smith 2016). Changes in resource quantity and quality then have flow-on effects on food web structure as well as on overall community composition and biodiversity (Graham 2013). Furthermore, invertebrate communities in eutrophic (nutrient rich) systems often become dominated by highly competitive and degradation-tolerant species which can prevent other more sensitive species from re-establishing once abiotic conditions have improved (Lake et al. 2007, Graham 2013). This phenomenon is known as 'biological resistance to restoration' (Warburton et al. 2018). Increased macrophyte densities can also influence stream communities and food web structure by providing habitat, refuge from predation for invertebrates and small fish, and substrate for growth of epiphytic algae (Collier et al. 1999, O'Hare et al. 2010). Associated changes in trait composition of invertebrate communities in macrophyte beds could similarly lead to restoration resistance. Therefore, leaving nutrient concentrations (and mass loading) at current levels may enable further 'biological resistance' to develop, which will increase recovery times (or possibly prevent recovery entirely) once restoration efforts commence.

Questions 2 and 3. Increased difficulty of remediation? Additional types of remediation needed?

The development of biological resistance to restoration will make remediation less effective. For example, a study conducted in lowland agricultural streams in the Canterbury region found that eutrophic streams had much greater numbers of grazing invertebrates, predominately *Potamopyrgus* snails. Because of their hard shells, these snails were less preferred as prey by fish and predatory invertebrates. As a result, biomass increased at the base of the food web but not at the top, creating a 'trophic' bottleneck effect (Graham et al. 2015). Attempts to "re-balance" the food web by restoration of detrital resources further increased snail populations, but did not increase the abundances of taxa characteristic of less eutrophic streams (Graham 2013). Therefore, it is likely that in these situations additional restoration actions targeted at reducing biological resistance, such as increased flow disturbance to remove benthic grazers, will be needed before other restoration measures, such as improved habitat and increased resource availability, will have the desired effect (Lake et al. 2007, Warburton et al. 2018, Barrett et al. 2019).

Biogeochemical resistance to restoration can also occur; for example, fine sediments and increased benthic algal biomass can reduce the vertical water fluxes within the hyporheic zone – the resulting alterations to fundamental biogeochemical processes is likely to impair the ability of streams and rivers to clean themselves (Boulton 2007, Mendoza-Lera and Mutz 2013).

Question 4. Influence of ecosystem characteristics.

The physical characteristics of rivers and streams can have a strong influence on the degree to which nutrient levels affect algal and macrophyte growth, and on the potential for development of restoration-resistant communities. Topography, slope, substrate, hydrological regime, temperature, degree of shading by riparian vegetation, and consumption by other organisms all interact to determine algae and macrophyte growth in streams (Biggs 2000, Matheson et al. 2012a, Burrell et al. 2014). Floods and associated high water velocities will remove accrued ("built-up") algal and macrophyte biomass; the degree of scouring or sloughing will vary according to substrate type, size and mobility (Biggs 2000, Riis and Biggs 2003). Shading will reduce the amount of light available for photosynthesis, thereby limiting algal and macrophyte growth even when nutrient concentrations are high (Matheson et al. 2012b, Burrell et al. 2014, Kankanamge et al. 2019, Mouton et al. 2019). The development of restoration-resistant communities will likewise depend upon the trait composition of the local species pool, which will vary naturally according to altitude, temperature, and hydrological disturbance regime, as well as in response to anthropogenic stressors such as eutrophication (Bonada et al. 2007a, Bonada et al. 2007b, Lake et al. 2007, Bruno et al. 2019). Therefore, both the physical characteristics and biotic community composition of river and stream ecosystems will determine how they respond to nutrient impacts and remediation efforts.

3 Estuaries

3.1 Estuary responses to unchanged nutrient loading

In this section we provide an overview of the likely consequences of inaction in reducing current nutrient concentrations/inputs of nutrient loads to New Zealand estuaries.

Question 1. Increased time for recovery?

The time necessary for remediation and/or recovery of estuaries is likely to increase if nutrient loads to New Zealand's estuaries are kept at current levels. The first reason for this is that physical and chemical conditions in many estuaries are degrading under current nutrient loads. This degradation causes feedbacks that hinder subsequent recovery. For example, the combination of high nutrient and high sediment loads from rivers causes the accumulation of nutrient-rich, oxygen-poor, fine sediments in New Zealand estuaries (Robertson et al. 2016, Zeldis et al. 2019a).

Dense beds of opportunistic seaweed (macroalgae) can flourish under high nutrient input conditions, and increasing density of macroalgae enhances fine-sediment trapping (Zeldis et al. 2019a). These sediments provide an additional source of the nutrients that stimulated algal growth. In turn, high algal biomass displaces and hinders the recovery of other biological communities after nutrient loads from rivers are reduced (Robertson et al. 2015). Figure 3-1 shows an example of this process. In recent years, areas of the New River Estuary in Southland have degraded from sandy, well oxygenated sediments to progressively anoxic fine sediments with almost continuous macroalgal cover (areas termed 'Gross Eutrophic Zones' (GEZs); Figure 3-2). Due to low oxygen availability in the accumulated sediments, conditions in these areas are now unsuitable for sustaining valued estuarine species such as cockles (Zeldis et al. 2019a) and seagrasses (Figure 3-3). The progressive nature of degradation of this estuary indicates that should catchment loading be reduced now, the estuary will likely take considerably longer to revert to a healthy state that would have been the case if nutrient and sediment loads had been reduced when excessive nutrient inputs were first identified as a problem, over a decade ago. Other estuaries in New Zealand that are subject to elevated sediment and nutrient inputs are on similar trajectories of degradation (Plew et al. 2018b, Robertson and Savage 2018).

Figure 3-1: Photographs illustrating the change in sediment trapping and retention following the establishment of persistent beds of macroalgae (*Gracilaria chilensis*). These photographs were taken at Bushy Point, New River Estuary (Southland) 2007, 2012 and 2016 (Zeldis et al. 2019a).

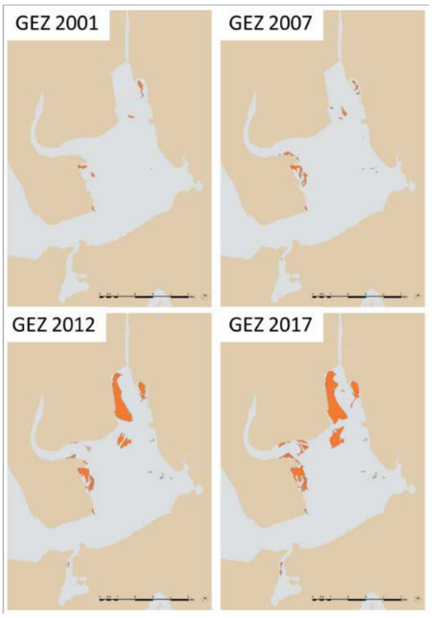

Figure 3-2: Time series of distributions of areas of anoxic fine sediment and continuous macroalgal cover (areas termed 'Gross eutrophic Zones' (GEZ)) in the New River Estuary. (Robertson et al. 2017).

The fact that biological communities are degrading under current nutrient loads means that further degradation will increase recovery times, because recovery of biological communities depends on the availability of suitable habitat (as described above), and on connectivity between sub-populations of biota (Duarte et al. 2015). As population densities and ranges of desirable species decrease and fragment in response to habitat loss, the availability of propagules and juveniles for future repopulation decreases. The loss of seagrass beds in the New River Estuary exemplifies this process (Figure 3-3). Seagrass beds are recognised as important habitats for juvenile stages of many commercially and culturally valuable New Zealand fish species (Morrison et al. 2009, Morrison et al. 2014). These seagrass beds have decreased markedly in the large Southland estuaries (e.g., up to 94% reduction in Waihopai Arm of New River Estuary (Figure 3-3); (Stevens 2018, Zeldis et al. 2019a).

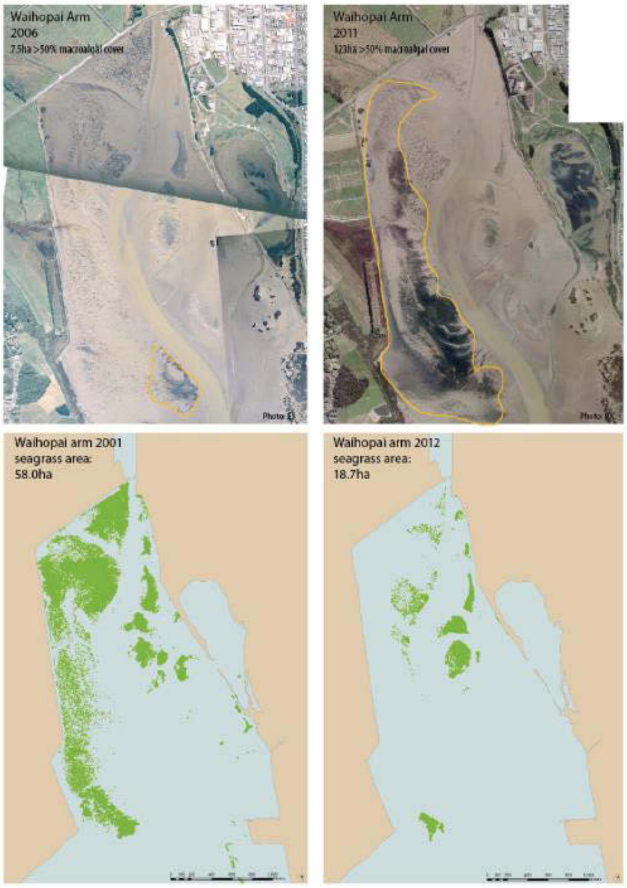

Figure 3-3: Aerial photos showing changes in macroalgal and seagrass cover in the Waihopai Arm, New River Estuary. Macroalgal cover 2006 and 2011 (upper), with corresponding seagrass cover in in the Waihopai Arm, 2001 and 2012 (lower) (Zeldis et al. 2019a). The area of high coverage of nuisance macroalgae are indicated by the orange line in the top right panel.

Question 2. Increased difficulty of remediation?

Estuary remediation options other than management of catchment nutrient loads are likely to become progressively more difficult to implement, and less effective, if reduction in nutrient loads is deferred. Potential options for remediating highly eutrophic (nutrient enriched) estuaries include dredging and removal of anoxic sediments, physical removal of excess macroalgae, re-seeding of seagrass beds, re-seeding of macroinvertebrates such as cockles and pipi, and improvements in point sources of nutrients (e.g., improving wastewater treatment, and diverting effluent discharge from within estuaries to offshore sites (Zeldis et al. 2019a). Manual removal of sediments and macroalgae are expensive options and are unlikely to be viable in estuaries where nutrient and sediment loads are not reduced because degraded conditions (e.g., macroalgal eutrophication, sediment degradation) are likely to return in the time between sediment removal and the implementation of load reductions. Manual sediment removal after load reductions may also be prohibitively expensive as areas of eutrophic fine sediment expand (Zeldis et al. 2019a). Re-seeding of seagrass beds, and re-seeding of macroinvertebrates such as cockles and pipi will be less viable if nutrient loading is deferred because the success of these measures requires that suitable habitat is available for their survival and reproduction (i.e., the reversal of the current trend of habitat loss for these species due to eutrophication).

Deferral of nutrient load reductions may also undermine the effectiveness of measures aimed at reducing the impact of point sources of nutrients (e.g., improvement in treatment efficacy and diversion of effluent discharge points in estuaries to offshore sites). An example of this can be seen in Ihutai/ Avon-Heathcote Estuary in Christchurch, where Christchurch wastewater was diverted from the estuary to an offshore outfall at a capital cost of ~$NZ80M, but nutrient inputs from tributary contributing rivers were not reduced as well. Although the capital expenditure has substantially improved macroalgal eutrophication, the recovery has been incomplete as a consequence of ongoing input of substantial nutrient loads from the Avon and Heathcote Rivers (Barr et al. 2019, Zeldis et al. 2019b).

Question 3. Additional types of remediation needed?

Deferral of nutrient load reductions is not likely to change the range of viable remediation methods available for *already* degraded estuaries. Although most of the remediation options currently available for improvement of the trophic state of estuaries (other than reductions in nutrient and sediment loads) are onerous and impractical at a large scale (reviewed in Zeldis et al. (2019a)), tangible benefits may be achieved if remediation is implemented before eutrophication becomes widely established within sediments. Deferral of nutrient load reductions is likely to result in increasing degradation of sediment condition, which will likely compromise future remediation effectiveness and extend recovery periods. For example, improving the trophic state of the New River Estuary from band D (very high eutrophication) to band C (high eutrophication) on the Estuary Trophic Index scale (Robertson et al. 2016, Zeldis et al. 2017) would require the removal of around 2100 tonnes of fine sediment per day and 35 tonnes of algae per day for an entire year (Zeldis et al. 2017, Zeldis et al. 2019a). Unless this reduction in contaminant load was continuously achieved, it is likely that the estuary would revert to degraded states. In addition, estuary remediation based around re-seeding of desirable ecological populations (such as seagrass and cockle beds) would likely remain non-viable unless underlying habitats were suitably remediated (Zeldis et al. 2019a).

Question 4. Influence of ecosystem characteristics.

Physical characteristics of estuaries affect their susceptibility to eutrophication (i.e., rates of degradation under current nutrient loads), as well as their recovery times following reductions in nutrient loads (Borja et al. 2010, Duarte et al. 2015, Plew et al. 2018a). Characteristics of estuaries that strongly influence recovery times from eutrophic conditions are sediment texture, flushing power, and tidal replacement (dilution of estuary water by ocean water) (Borja et al. 2010, Zeldis et al. 2019b). For example, coastal lakes such as Te Waihora/ Lake Ellesmere are likely to have very low tidal replacement, low flushing power and fine sediment texture, leading to slow recovery from eutrophic conditions following nutrient loading reductions. In contrast, well-flushed river estuaries tend to show comparatively little sign of eutrophication under high nutrient loads, and are likely to recover quickly from eutrophic conditions (Borja et al. 2010, Plew et al. 2018a). Shallow, intertidally-dominated lagoons (such as New River Estuary and Avon Heathcote Estuary) will likely have intermediate recovery potential depending on their size and shape, sediment characteristics, organic matter turnover rates, and the effectiveness of tidal replacement (Borja et al. 2010, Zeldis et al. 2019b). Extensive areas of most intertidally-dominated estuaries will exhibit few obvious symptoms of excessive eutrophication, with poorly flushed deposition zones or sheltered arms remaining most susceptible to persistent degradation.

4 Lakes

4.1 Lake responses to unchanged nutrient loading

In this section we provide an overview of the likely consequences of inaction in reducing current levels of nutrient loading to New Zealand's lakes.

Question 1. Increased time for recovery?

The longer elevated loads of catchment derived nutrient (external loading) are allowed to enter lakes, the longer it will take to complete the recovery phase after nutrient loads are reduced (Søndergaard et al. 1993, Søndergaard et al. 2003), as explained below.

A large fraction of the sediment and nutrient loads entering lakes are stored within the water column, sediment and biomass. These nutrients cycled between forms (particulate, dissolved, organic and inorganic) and internal recycling can cause elevated trophic states (an indexed measure of total nitrogen, total phosphorus, algal biomass and Secchi depth in a lake water column). Lake sediments in particular store large quantities of 'legacy' nutrients that are released to the water column through multiple processes (Boström et al. 1982, Søndergaard et al. 2003). The breakdown of sediment organic matter causes ammoniacal-N to be released, and nitrate-N can also be released under oxygenated water column conditions (Gibbs et al. 2011). If near-bed oxygen concentrations become depleted, sediment-bound phosphorus is released in the form of dissolved reactive phosphorus (DRP) (Boström et al. 1982, Søndergaard et al. 2003).

High rates of nutrient release from sediments can fuel planktonic algal blooms. High rates of photosynthesis during such blooms consume CO_2, increasing water column pH, which can in turn cause further DRP release from the sediments and stimulate more algal growth. Furthermore, upon the collapse and sedimentation of an algal bloom, the decaying algal biomass consumes oxygen, which can cause or maintain anoxia in the bottom water column, resulting in more DRP release in a positive feedback loop (Burger et al. 2008). Bed sediment disturbance in shallow lakes by wind or biota can also cause the water column to be nutrient enriched.

Shallow lakes can flip between clear-water, macrophyte dominated states and turbid-water, phytoplankton dominated states (Sayer et al. 2010). In most cases, the preferred state is the macrophyte dominated state, in which the water clarity is higher and water quality better than in the phytoplankton-dominated state. Macrophytes rely on high water transparency for photosynthesis, and in the macrophyte-dominated state, they stabilize bed sediments and reduce resuspension. In turbid water, macrophytes are outcompeted for light by phytoplankton (Sand-Jensen and Søndergaard 1981). Under conditions of high nutrient loads and low clarity, macrophytes may disappear altogether. Improvements in water quality are then impeded by the lack of sediment stabilization. In other words, for macrophytes to re-establish, water quality may need to improve to a better state than the lake was in before it shifted to the phytoplankton dominated state (Hilt et al. 2013). The positive feedback described here make turbid, phytoplankton dominated lakes highly resistant to recovery, as discussed in the next section.

Questions 2 and 3. Increased difficulty of remediation? Additional types of remediation needed?

Several factors may impair the recovery of degraded shallow lakes to a clear, (native) macrophyte-dominated ecological state. These include poor water clarity, pest fish disturbance, herbivory, competition from invasive macrophytes, low density bed sediment, depauperate seedbanks, and internal cycling of nutrients. To reduce internal phosphorus supplies and break the positive feedback discussed above, lakes can be "geo-engineered" by adding substances that clear the water column (e.g., flocculants), and/or reduce the flux of phosphorus from sediments (sediment capping agents) (Hickey and Gibbs 2009). The dose required for these agents to be effective depends on the phosphorus load within the lake and the amount of material in the water column that needs to be cleared. Therefore, leaving current phosphorus inputs unchanged will require greater amount of geo-engineering products than would be required if the inputs were reduced immediately.

The recovery of macrophytes in shallow lakes depends in part on the size and viability of the seedbank (dormant seeds in bed sediments) and the input of propagules from external sources via tributaries, waterfowl and other vectors. Seedbanks decline over time, especially in algal dominated lakes and in lakes with frequent sediment disturbances (de Winton and Clayton 1996). If seedbanks are greatly depleted and external supplies are small, manual replanting may be required. Therefore, maintaining current nutrient loads/concentrations will cause further decline to seedbanks, making it harder and more expensive to re-establish native vegetation.

In deep lakes, the role of native macrophytes on susceptibility to degradation is small. Instead, susceptibility to degradation is strongly related to hypolimnetic de-oxygenation and associated nutrient release from bed sediments. In-lake remediation measures are then primarily flocculation, sediment capping, and biomass removal. These measures are costly for large lakes, and the cost increases as nutrient stocks accumulate. Aeration and artificial mixing are used in some lakes at risk of hypolimnetic de-oxygenation, but these methods are not very effective for large lakes (although they are effective for reducing cyanobacterial blooms in water supply reservoirs).

As with shallow lakes, deep lakes that have shifted to a degraded ecological regime may require nutrient loads to be reduced to a very low levels before recovery commences, compared with what would have been required if the lakes were not degraded in the first place (e.g., Janse et al. 2008, Hilt et al. 2013). The longer reductions in nutrient inputs are delayed, the more likely it is that such additional source load reduction will be needed.

Question 4. Influence of ecosystem characteristics.

The morphology of lakes (i.e., area, shape, volume) affects the response of lakes to nutrient loads and will affect responses to delayed nutrient load reduction. Deep lakes typically have longer residence times; while they take longer to degrade, they also take longer to recover from high nutrient loads. Most of the lakes in New Zealand have residence times of < 10 years, so the consequences of delays in delayed nutrient load reduction for these lakes will be less severe than very large and deep lakes such as Lakes Taupo and Wakatipu. In addition to the effects of long residence times, deep lakes may resist recovery due to stratification. In lakes with anoxic bottom water, loads of settled nutrients will be recycled into the upper water column, driving growth of phytoplankton, which then settles back into the bottom water.

As discussed earlier, shallow lakes which have flipped from clear, macrophyte dominated states to turbid algal-dominated states are also resistant to recovery. In these lakes, rapid wind-driven mixing through the entire water column leads to high sediment resuspension and bed instability, both of which impede macrophyte recovery.

5 Groundwater lag times

Nitrogen that is leached from the soil profile can, in some circumstances, take decades to migrate into and through groundwater. In turn, nitrogen-enriched groundwater may take decades to move from the site of enrichment to the site of discharge to a surface water body. These delays are termed lag times. If leaching increases over time (due to land use intensification or conversion to intensive land uses), loading to stream and lakes may take decades to increase to the point where it fully reflects historical increases. Therefore, currently-observed loads concentrations may be less than subsequent loads and concentrations, even if leaching rates remain basically constant. This is often termed the 'load to come'. Similarly, streams may take decades to respond to *reductions* in leaching due to the same time lags.

Lag times have implications for the consequences of delays in reducing leaching. If there has been an increase in leaching, then loading to the stream will gradually increase, until it plateaus out. If the stream has not already fully adjusted to historical increases in leaching, and leaching sources remain the same, then source loading to the stream will increase. If the leaching were to reduce, however, then loading to the stream would start to reduce, and the peak loading to the stream would not reach the same level as if the leaching continued. Hence delaying reductions in leaching will increase the peak rate of loading into the stream, and consequently increase the peak stream concentration. A second effect of delaying reductions in leaching for systems that have not yet fully adjusted would be to protract the time to respond to reductions in leaching due to the time lags noted above.

Impacts of groundwater lag times have been studied in systems where groundwater is known to have long residence times. For example, in the Lake Rotorua catchment, groundwater lags range from 14 to 170 years (Rutherford et al. 2019). If leaching remains at current levels, loading to the lake is estimated to increase by about 15% from current levels (Rutherford et al. 2019). In the Hamurana Spring which feeds Lake Rotorua, there is a long residence time, and loading in the stream could ultimately double from current levels (Morgenstern et al. 2015). The implications of delays in reductions in source loads were not examined in those studies, though. Similarly, the upper Waikato catchment and northeastern Lake Taupo areas are, like Lake Rotorua, also in the Central Volcanic Zone and have porous aquifers with long residence times, and the load to come has been incorporated into future concentration projections. Long groundwater lag times are also characteristic of coastal alluvial plains such as the Canterbury and Heretaunga Plains; lakes and rivers in these areas are likely be increasingly nutrient enriched in the future due to the leaching from intensified agriculture in the inland plains.

In the Rotorua case, it was argued that there will be little loss of nitrate through denitrification because even the deep groundwater remains oxic. However, in other systems, deeper older groundwater becomes anoxic and nitrogen removed through denitrification (e.g., Woodward et al. 2013). In those other cases, the load to come, and implications of delaying nutrient reductions, will be less important. There is uncertainty about where such conditions occur and their role, because it is difficult to characterise subsurface pathways and nutrient processing.

The Central Volcanic Zone does not represent a general case. For example, in the Aparima catchment in Southland, the load to come is not considered to be an important factor, because there are predominantly shallow alluvial aquifers with residence times less than a decade. Such systems will have generally responded fully to historical increases in leaching, and the implications of delays in implementing source reductions will be of less concern, insofar as the effects of groundwater lags are concerned.

While the principles of the implications groundwater lags are clear, it is difficult to quantify the implications accurately nationally because we do not have a national picture of groundwater lags (although work is underway by GNS to develop such maps), nor are the variations of denitrification along groundwater pathways known nationally.

6 References

Barr, N., Zeldis, J., Scheuer, K., Schiel, D. (2019) Macroalgal Bioindicators of Recovery from Eutrophication in a Tidal Lagoon Following Wastewater Diversion and Earthquake Disturbance. *Estuaries and Coasts*, 43: 240-255.

Barrett, I., McIntosh, A.R., Febria, C., Warburton, H. (2019) Negative resistance and resilience: a mesocosm experiment demonstrating consequences for biological recovery in restoration. *New Zealand Ecological Society Conference,* Lincoln, New Zealand.

Biggs, B. (2000) New Zealand periphyton guideline: Detecting, monitoring and managing nutrient enrichment in streams. *Ministry for the Environment*, Wellington, New Zealand.

Bonada, N., Doledec, S., Statzner, B. (2007a) Taxonomic and biological trait differences of stream macroinvertebrate communities between mediterranean and temperate regions: implications for future climatic scenarios. *Global Change Biology*, 13: 1658-1671.

Bonada, N., Rieradevall, M., Prat, N. (2007b) Macroinvertebrate community structure and biological traits related to flow permanence in a Mediterranean river network. *Hydrobiologia,* 589: 91-106.

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

Boström, B., Jansson, M., Forsberg, C. (1982) Phosphorus release from lake sediments. *Archiv für Hydrobiologie-Beiheft Ergebnisse der Limnologie,* 18: 5e59.

Boulton, A.J. (2007) Hyporheic rehabilitation in rivers: restoring vertical connectivity. *Freshwater Biology,* 52: 632-650.

Bruno, D., Belmar, O., Maire, A. Morel, A., Dumont, B., Datry, T. (2019) Structural and functional responses of invertebrate communities to climate change and flow regulation in alpine catchments. *Global Change Biology,* 25: 1612-1628.

Burger, D.F., Hamilton, D.P., Pilditch, C.A. (2008) Modelling the relative importance of internal and external nutrient loads on water column nutrient concentrations and phytoplankton biomass in a shallow polymictic lake. *Ecological Modelling*, 211: 411-423.

Burrell, T.K., O'Brien, J.M., Graham, S.E., Simon, K.S., Harding, J.S., McIntosh, A.R. (2014) Riparian shading mitigates stream eutrophication in agricultural catchments. *Freshwater Science*, 33: 73-84.

Collier, K.J., Champion, P.D., Croker, G.E. (1999) Patch-and reach-scale dynamics of a macrophyte-invertebrate system in a New Zealand lowland stream. *Hydrobiologia,* 392: 89-97.

de Winton, M.D., Clayton, J.S. (1996) The impact of invasive submerged weed species on seed banks in lake sediments. *Aquatic Botany*, 53: 31-45.

Depree, C., Quinn, J.J., Rutherford, K., Wilcock, B., Young, R. (2016) Response of the 'P-Limited' Tukituki River to a reduction in P-loading from Waipukurau and Waipawa WWTPs: River brings is own P to the party'? *New Zealand Freshwater Sciences Society Conference,* Invercargill, New Zealand.

Dodds, W.K., Smith, V.H. (2016) Nitrogen, phosphorus, and eutrophication in streams. *Inland Waters*, 6: 155-164.

Duarte, C.M., Borja, A., Carstensen, J., Elliott, M., Krause-Jensen, D., Marbà, N. (2015) Paradigms in the Recovery of Estuarine and Coastal Ecosystems. *Estuaries and Coasts,* 38: 1202-1212.

Evans-White, M.A., Dodds, W.K., Huggins, D.G., Baker, D.S. (2009) Thresholds in macroinvertebrate biodiversity and stoichiometry across water-quality gradients in Central Plains (USA) streams. *Journal of the North American Benthological Society*, 28: 855-868.

Gibbs, M., Hickey, C., Özkundakci, D. (2011) Sustainability assessment and comparison of efficacy of four P-inactivation agents for managing internal phosphorus loads in lakes: sediment incubations. *Hydrobiologia*, 658: 253-275.

Graham, S.E. (2013) Mechanisms and mitigation of food web change in stream ecosystems. *PhD Dissertation*, Canterbury University.

Graham, S.E., O'Brien, J.M., Burrell, T.K., McIntosh, A.R. (2015) Aquatic macrophytes alter productivity-richness relationships in eutrophic stream food webs. *Ecosphere,* 6: 1-18.

Hickey, C.W., Gibbs, M.M. (2009) Lake sediment phosphorus release management—decision support and risk assessment framework. *New Zealand Journal of Marine and Freshwater Research*, 43: 819-856.

Hilt, S., Köhler, J., Adrian, R., Monaghan, M.T., Sayer, C.D. (2013) Clear, crashing, turbid and back–long-term changes in macrophyte assemblages in a shallow lake. *Freshwater Biology*, 58: 2027-2036.

Janse, J.H., Domis, L.N.D.S., Scheffer, M., Lijklema, L. Van Liere, L., Klinge, M., Mooij, W.M. (2008) Critical phosphorus loading of different types of shallow lakes and the consequences for management estimated with the ecosystem model PCLake. *Limnologica*, 38: 203-219.

Kankanamge, C.E., Matheson, F.E., Riis, T. (2019) Shading constrains the growth of invasive submerged macrophytes in streams. *Aquatic Botany*, 158: 103125.

Lake, P.S., Bond, N., Reich, P. (2007) Linking ecological theory with stream restoration. *Freshwater Biology*, 52: 597-615.

Matheson, F., Quinn, J., Hickey, C. (2012a) Review of the New Zealand in-stream plant and nutrient guidelines and development of an extended decision making framework: Phases 1 and 2 final report. *NIWA Client Report* No. HAM2012-081, prepared for Ministry of Science & Innovation Envirolink Fund: 127.

Matheson, F.E., Quinn, J.M., Martin, M.L. (2012b) Effects of irradiance on diel and seasonal patterns of nutrient uptake by stream periphyton. *Freshwater Biology,* 57: 1617-1630.

McDowell, R.W. (2015) Relationship between Sediment Chemistry, Equilibrium Phosphorus Concentrations, and Phosphorus Concentrations at Baseflow in Rivers of the New Zealand National River Water Quality Network. *Journal of Environmental Quality,* 44: 921-929.

Mendoza-Lera, C., Mutz, M. (2013) Microbial activity and sediment disturbance modulate the vertical water flux in sandy sediments. *Freshwater Science*, 32: 26-38.

Morgenstern, U., Daughney, C., Leonard, G., Gordon, D., Donath, F., Reeves, R. (2015) Using groundwater age and hydrochemistry to understand sources and dynamics of nutrient contamination through the catchment into Lake Rotorua, New Zealand. *Hydrology and Earth System Sciences,* 19: 803-822.

Morrison, M., Jones, E.G., Consalvey, M., Berkenbusch, K. (2014) Linking marine fisheries species to biogenic habitats in New Zealand: a review and synthesis of knowledge. *New Zealand Aquatic Environment and Biodiversity Report,* 130.

Morrison, M.A., Lowe, M., Parsons, D., Usmar, N., McLeod, I. (2009) A review of land-based effects on coastal fisheries and supporting biodiversity in New Zealand. *New Zealand Aquatic Environment and Biodiversity Report,* 37.

Mouton, T.L., Matheson, F.E., Stephenson, F., Champion, P.D., Wadhwa, S., Hamer, M.P., Catlin, A., Riis, T. (2019) Environmental filtering of native and non-native stream macrophyte assemblages by habitat disturbances in an agricultural landscape. *Science of the Total Environment*, 659: 1370-1381.

Newbold, J.D., Elwood, J.W., O'Neill, R.V., Winkle, W.V. (1981) Measuring nutrient spiralling in streams. *Canadian Journal of Fisheries and Aquatic Sciences,* 38: 860-863.

O'Hare, M.T., Clarke, R.T., Bowes, M.J., Cailes, C., Henville, P., Bissett, N., McGahey, C., Neal, M. (2010) Eutrophication impacts on a river macrophyte. *Aquatic Botany*, 92: 173-178.

Plew, D., Dudley, B.D., Shankar, U., Zeldis, J. (2018a) Assessment of the eutrophication susceptibility of New Zealand Estuaries. *NIWA Client Report,* No. 2018206CH, prepared for Ministry for the Environment: 61.

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

Riis, T., Biggs, B.J.F. (2003) Hydrologic and hydraulic control of macrophyte establishment and performance in streams. *Limnology and Oceanography,* 48: 1488-1497.

Robertson, B., Savage, C. (2018) Mud-entrained macroalgae utilise porewater and overlying water column nutrients to grow in a eutrophic intertidal estuary. *Biogeochemistry,* 139: 53-68.

Robertson, B.M., Stevens, L., Robertson, B., Zeldis, J., Green, M., Madarasz-Smith, A., Plew, D., Storey, R., Oliver, M. (2016) NZ Estuary Trophic Index Screening Tool 2. *Determining Monitoring Indicators and Assessing Estuary Trophic State.* Prepared for Envirolink Tools Project: Estuarine Trophic Index, Contract: C01X1420: 68.

Robertson, B.P., Gardner, J.P.A., Savage, C. (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.

Rosemond, A.D., Benstead, J.P., Bumpers, P.M., Gulis, V., Kominoski, J.S., Manning, D.W.P., Suberkropp, K., Wallace, J.B. (2015) Experimental nutrient additions accelerate terrestrial carbon loss from stream ecosystems. *Science,* 347: 1142-1145.

Rutherford, J.C., Palliser, C.C., MacCormick, A. (2019) Eutrophication In Lake Rotorua. 2. Using ROTAN and OVERSEER to model historic, present and future nitrogen loads. *New Zealand Journal of Marine and Freshwater Research*, 53: 128-161.

Sand-Jensen, K., Søndergaard, M. (1981) Phytoplankton and epiphyte development and their shading effect on submerged macrophytes in lakes of different nutrient status. *Internationale Revue der gesamten Hydrobiologie und Hydrographie,* 66: 529-552.

Sayer, C.D., Burgess, A., Kari, K., Davidson, T.A., Peglar, S., Yang, H., Rose, N. (2010) Long-term dynamics of submerged macrophytes and algae in a small and shallow, eutrophic lake: implications for the stability of macrophyte-dominance. *Freshwater Biology*, 55: 565-583.

Schindler, D.W., Hecky, R., Findlay, D., Stainton, M., Parker, B., Paterson, M., Beaty, K., Lyng, M., Kasian, S. (2008) Eutrophication of lakes cannot be controlled by reducing nitrogen input: results of a 37-year whole-ecosystem experiment. *Proceedings of the National Academy of Sciences*, 105: 11254-11258.

Søndergaard, M., Jensen, J.P., Jeppesen, E. (2003) Role of sediment and internal loading of phosphorus in shallow lakes. *Hydrobiologia,* 506: 135-145.

Søndergaard, M., Kristensen, P., Jeppesen, E. (1993) Eight years of internal phosphorus loading and changes in the sediment phosphorus profile of Lake Søbygaard, Denmark. Pages 345-356 in: *Proceedings of the Third International Workshop on Phosphorus in Sediments.* Springer.

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

Warburton, H.C., Febria, K., Hogsden, E., Graham, J., Harding, A. McIntosh, A., Barrett, I. (2018) Resilience isn't always healthy: disturbing degraded communities to reverse the effects of environmental filtering. *Society for Freshwater Science Conference*, Salt Lake City, Utah.

Wilcock, B., Quinn, J., Rutherford, K., Depree, C., McDowell, R., Young, R. (2016) Phosphorus, periphyton, photosynthesis, and pH. *New Zealand Freshwater Sciences Society Conference*, Invercargill, New Zealand.

Woodward, S.J., Stenger, R., Bidwell, V.J. (2013) Dynamic analysis of stream flow and water chemistry to infer subsurface water and nitrate fluxes in a lowland dairying catchment. *Journal of Hydrology,* 505: 299-311.

Zeldis, J., Measures, R., Stevens, L., Matheson, F., Dudley, B. (2019a) Remediation Options for Southland Estuaries*. NIWA Client Report* No. 2019344CH, prepared for Environment Southland, December 2019: 73.

Zeldis, J., Whitehead, A., Plew, D., Madarasz-Smith, A., Oliver, M., Stevens, L., Robertson, B., Storey, R., Burge, O., Dudley, B. (2017) The New Zealand Estuary Trophic Index (ETI) Tools: Tool 2 - Assessing Estuary Trophic State using Measured Trophic Indicators. Prepared for Ministry of Business, Innovation and Employment Envirolink Tools, *Contract* C01X1420.

Zeldis, J.R., Depree, C., Gongol, C., South, P.M., Marriner, A., Schiel, D.R. (2019b) Trophic Indicators of Ecological Resilience in a Tidal Lagoon Estuary Following Wastewater Diversion and Earthquake Disturbance. *Estuaries and Coasts*: 1-17.