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Shallow lakes restoration review: A literature review

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Prepared by:
Jonathan Abell
(Ecofish Research Ltd)

For:
Waikato Regional Council
Private Bag 3038
Waikato Mail Centre
HAMILTON 3240

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Peer reviewed by:
Paula Reeves (Waikato Regional Council)
Tracie Dean-Speirs (Department of Conservation)

Date June 2018

Approved for release by:
Dominique Noiton

Date July 2018

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Abstract

Shallow lakes are important resources to the Waikato region. These ecosystems have generally been degraded due to multiple causes, particularly eutrophication which is increased productivity due to pollution by the nutrients nitrogen and phosphorus. Waikato Regional Council has developed a Shallow Lakes Management Plan to guide the restoration of shallow lakes in the region. Identifying cost-effective approaches to improve water quality in shallow lakes that have been degraded by eutrophication requires good understanding of the range of lake restoration methods that are part of the 'toolbox' available to decision makers and managers. This literature review is therefore intended to provide context to support environmental management of shallow Waikato lakes. The objective of this review is to synthesise and evaluate the current literature relating to approaches to restore shallow lakes degraded by anthropogenic eutrophication. Particular focus is given to quantifying the performance of individual approaches based on the results of robust monitoring studies described in the primary scientific literature. The review considers studies undertaken worldwide, with particular focus on lakes in temperate regions that have been degraded due to nutrient pollution from agricultural land, with implications for Waikato lakes highlighted where relevant.

In general, shallow lakes can be resistant to lake restoration efforts due to high internal loads, structural complexity and hysteresis in how water quality responds to changes in nutrient loads. In the Waikato, shallow lakes present a particular challenge as many lakes have historically received extremely high external loads, meaning that lake bed sediments are highly enriched with nutrients that fuel internal loading. The widespread presence of benthivorous invasive fish species such as common carp (*Cyprinus carpio*) also presents a particular challenge as they excrete nutrients and re-suspend sediments. Further, the predominantly diffuse nature of agricultural nutrient sources presents technical, economic and political challenges to reducing external loads. The rate of response to lake restoration can vary considerably among shallow lakes, highlighting the need to develop an adequate understanding of the limnology of a specific lake when projecting how water quality will respond to restoration.

There is a variety of lake restoration techniques that have been successfully applied to remediate the symptoms of eutrophication; broadly, these may be grouped into four categories: 1) controlling external loads; 2) controlling internal loads, 3) biomanipulation, and; 4) hydrologic manipulations. These have varying relevance to Waikato shallow lakes.

Controlling external loads is fundamental to the successful and sustainable restoration of eutrophic shallow lakes. Insufficient reductions to external loads is a major cause of lake restoration failure and the majority of successful case studies entailed external load reductions of at least 50%. Implementing best management practices in association with existing land uses (e.g., planting riparian buffers on farmland) can achieve detectible reductions in external nutrient loads; however, experiences elsewhere show that this alone is insufficient to cause significant improvements in water quality of lakes that are moderately to highly impaired, such as Waikato shallow lakes. Successfully achieving the necessary external load reductions requires the appropriate socio-political framework, including: leadership by a dedicated water management agency; engagement with motivated local groups; quantitative load reduction targets, and; regulation. Participatory tools such as structured decision making may help managers and communities to explore trade-offs among catchment activities and restoration methods.

Methods to control internal loads have been widely adopted to support lake restoration and can be important for overcoming inertia caused by internal loading in shallow lakes that have highly nutrient-enriched sediments. The main methods are: dredging, sediment capping and the addition of materials to adsorb phosphorus and flocculate organic matter. Actions to control internal loads are likely to be required to achieve even moderately ambitious lake restoration targets in many Waikato shallow lakes, particularly shallow riverine lakes that have historically received large inputs of nutrient-enriched sediments from floodplains. Research in northern temperate lakes has shown that, following large external load reductions, it typically takes 10–15 years for internal loads to decline sufficiently for lake restoration targets to be met. However, the highly enriched status (supertrophic or hypertrophic) of many Waikato shallow lakes indicates that this lag time will be at least at the upper end of this range, meaning that controls on internal loads are necessary to achieve substantial improvements in water quality of many lakes in less than one or two decades. The applicability of individual methods depends on numerous lake-specific factors and requires assessing ecological risks and cultural sensitivities. Logistical and financial challenges are substantial in large shallow lakes.

Biomanipulation has proven to be successful in many temperate lakes in the northern hemisphere; however, characteristics of lake food webs mean that biomanipulation of fish communities has more limited applicability in New Zealand. Of the range of biomanipulation techniques that are established, the control of benthivorous fish seems to have greatest applicability to shallow lakes in the Waikato, given the high biomass of invasive fish such as carp in many lakes. Despite this, experiences to date indicate that fish removal requires high, persistent effort and can only make a minor contribution (at best) to restoration of shallow, hypertrophic lakes in the Waikato. Nonetheless, there is an established effect pathway between benthivorous pest fish and poor water quality, meaning than pest fish control has a role as a restoration tool in shallow Waikato lakes, particularly in very shallow lakes with thick deposits of fine sediments (e.g., many riverine lakes) and lakes where opportunities for ongoing re-colonisation from connected waterbodies are limited. Planktivorous pest fish control may be applicable to a subset of lakes. The promotion of kākahi (*Echyridella menziesi*) may also be applicable to lakes that have suitable physicochemical conditions; however, this requires further investigation.

Hydrologic manipulations, notably inflow diversion or augmentation, can be highly successful in the right circumstances, although the applicability of these techniques is highly dependent on the local geography. There may be opportunities to use these techniques in shallow Waikato lakes, although this needs to be evaluated on a lake-by-lake basis. Manipulating water levels also has potential to achieve more modest outcomes, although again this depends on lake-specific factors such as the feasibility to install a control structure and the characteristics of riparian and emergent vegetation communities.

Approaches and results of multiple case studies are described and evaluated. Impediments to lake restoration are also discussed. In particular, climate change poses a major challenge to achieving lake water quality objectives due to a range of mechanisms; most notably, warmer temperatures are expected to promote the proliferation of cyanobacteria and can enhance internal loading. It will therefore be increasingly important for lake and catchment managers to consider ways to adapt to climate change to build resilience and support multiple objectives. For example, adoption of landscape-scale floodwater management can contribute to controlling phosphorus loads, while also supporting flood risk management and erosion control objectives

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1 Introduction

Shallow lakes are important resources to the Waikato region that provide a range of ecological, social, cultural, spiritual, and economic values. Waikato Regional Council (WRC) recognises that these ecosystems have generally been degraded due to multiple causes (Waikato Regional Council 2016a). In particular, shallow lakes in the region have been highly impacted by eutrophication (increased productivity due to nutrient pollution), which has impaired lake values (Hamilton *et al.* 2010; Waikato Regional Council 2014a)

To achieve the best use of fresh water in the region, WRC is implementing a new Waikato Freshwater Strategy (Waikato Regional Council 2017a) that includes adopting a longer-term (50+ years) view and implementing “smarter methods” to change the way that freshwater resources are managed. WRC has developed a Shallow Lakes Management Plan (two volumes) that specifies key management issues, policy objectives, and background information about 71 shallow lakes (Waikato Regional Council 2014a,b). An objective of this plan is to establish appropriate objectives, limits and targets for the future management and enhancement of shallow lakes. This requires good understanding of the range of lake restoration methods that are part of the ‘toolbox’ available to decision makers and managers. This knowledge should be informed by the best available science and, where possible, based on the results of robust and quantitative monitoring of comparable lakes.

To support implementation of the Shallow Lakes Management Plan, WRC has commissioned this literature review on the restoration of shallow lakes. The objective of this review is to synthesise and evaluate the current literature relating to approaches to restore shallow lakes degraded by anthropogenic eutrophication. This review aims to address the following research questions:

1. What methods are available to address eutrophication in shallow lakes?
2. What is the performance of individual lake restoration methods, as quantified by robust monitoring? How relevant are these results to Waikato shallow lakes?
3. What shallow lake restoration projects have been deemed “successful”? What were the restoration targets and how were they achieved?
4. Are there successful examples of shallow lake restoration that solely involved implementing best management practices (BMPs) to control diffuse nutrient pollution in the catchment? How relevant are these to the Waikato region?
5. What were the major drivers of successful lake restoration projects (e.g., legislation, industry-led initiatives, voluntary action by community groups)?
6. Are there examples of lake restoration projects that have successfully addressed multiple pressures in combination (e.g., invasive fish and macrophytes)? If so, how has this been achieved?

2 Overview of Waikato shallow lakes

2.1 Setting

There are approximately 70 to 100 lakes in the Waikato region that are 'shallow', based on the criterion that they have a maximum depth of less than 10 m (Waikato Regional Council 2014a; DoC 2017)[1]. Although depth alone is an imperfect criterion to define a 'shallow' lake (Kirillin and Shatwell 2016), this criterion provides a suitable estimate of the number of lakes that are "functionally shallow" (Padisák and Reynolds 2003), in that they do not persistently stratify over seasonal timescales, i.e., shallow lakes are polymictic rather than meromictic, monomictic or dimictic[2]. This physical instability leads to a common set of limnological features among shallow lakes, including non-linear responses to nutrient enrichment (Scheffer 2004; discussed further in Section 3).

Waikato shallow lakes can be categorised based on geomorphic types that share common characteristics (Hamilton *et al.* 2010; Waikato Regional Council 2014a). The most common geomorphic types are peat, dune and riverine (Table 1). There are a small number of volcanic and karst lakes (Table 1), while the FENZ database also identifies three shallow shoreline lakes and three shallow reservoirs (dam-formed lakes) in the Waikato Region (DoC 2017).

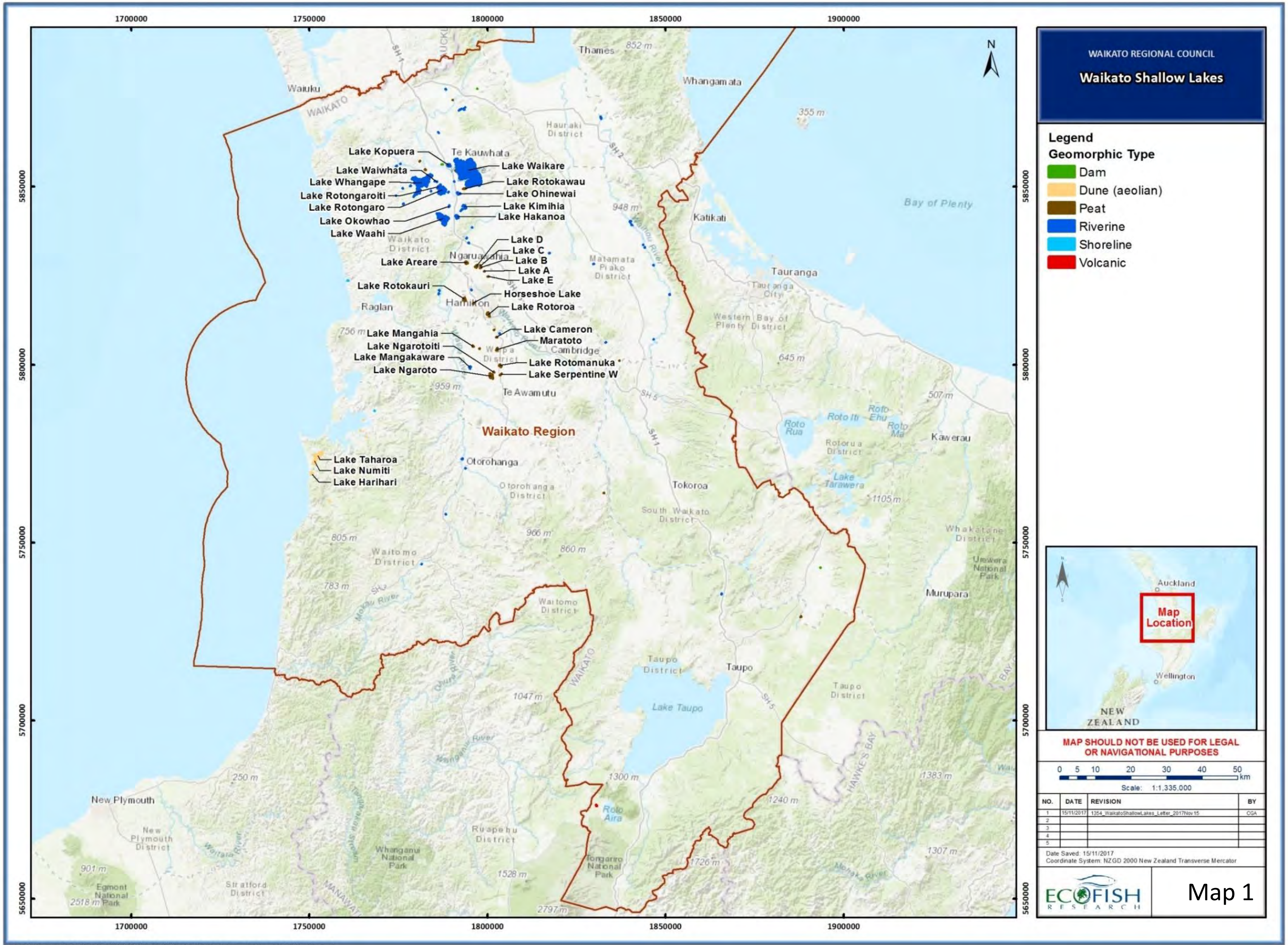
Table 1. Overview of five main geomorphic groups of Waikato shallow lakes. Based on WRC (2014a).

Geomorphic group	*n* ‡	Examples	Characteristics
Peat	37	Rotokauri Rotoroa Ngaroto	Internationally important ecosystems Naturally present in wetlands Acidic Naturally nutrient poor High tannin content; highly coloured Not widely present elsewhere in NZ
Dune	14	Taharoa Numiti Parangi	Formed by wind-blown sand deposition Predominantly on the west coast, e.g., Kawhia-Aotea Harbour
Riverine	14	Whangape Waahi Waikare	Lowland ecosystems that were naturally highly influenced by riverine processes, e.g., fluvial flooding Includes some of the largest shallow lakes (e.g., Lake Waikare); wind-driven resuspension can be an important process due to high fetch
Volcanic	3	Rotopounamu Ngahewa Tutaeinanga	Located in the Taupō Volcanic Zone < 10 ha
Karst	2	Koraha Disappear	Formed in limestone landscapes Lake Disappear forms intermittently after wet weather

‡ Based on WRC (2014a), which reports values based on lakes > 1 ha with maximum depth < 10 m. The FENZ database (DoC 2017) also lists three shoreline lakes and three reservoirs.

[1] WRC (2014b) states that there are 71 shallow lakes > 1 ha in the Waikato region; the Freshwater Ecosystems of New Zealand database (DoC 2017) lists 101 lakes with maximum depth < 10 m in Waikato Regional Council's jurisdiction, although maximum depths for many of these lakes were modelled, not measured.

[2] For comparison, in their national assessment, Larned *et al.* (2015) consider lakes with maximum depth < 5 m to mix "frequently" during the year, while lakes with maximum depth of 5–15 m to mix "occasionally" through the summer.



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2.2 Current water quality

This section provides an overview of the water quality of shallow lakes in the Waikato. This information provides context for the literature review and is not intended to provide a comprehensive summary of recent monitoring data.

Waikato Regional Council conducts water quality monitoring of eight shallow lakes bimonthly (lakes Waahi, Whangape, Waikare, Serpentine North, Serpentine East, Maratoto, Rotomānuka, Hakanoa). These lakes have been monitored consistently for over 10 years, providing information about water quality trends at these sites. Lake water quality has also been monitored as part of WRC's Shallow Lake Health Indicators programme, which has involved sampling up to 17 lakes three times per year (December, March and May). This programme includes lakes that are monitored consistently, in addition to lakes that are sampled annually to extend spatial coverage (Waikato Regional Council 2014b). The lake Submerged Plant Index (LakeSPI) is also monitored occasionally for a larger sample of lakes to provide a measure of the ecological health of the submerged macrophyte community. This assessment has now been repeated over multiple years on several lakes (e.g., Edwards *et al.* 2007, 2010).

Lake trophic status is evaluated using the Trophic Level Index (TLI; Burns *et al.* 1999), which is calculated based on mean measurements of Secchi depth (a measure of clarity), and concentrations of nitrogen, phosphorus and chlorophyll *a* (chl *a*; a metric of algal biomass). In general, shallow lakes in the Waikato region have enriched productivity due to anthropogenic eutrophication. For example, based on analysis of five-year TLI values, 18 of the 20 shallow Waikato lakes with sufficient data have a trophic status of 'eutrophic' or higher, with the majority (14 lakes) assigned a trophic status of either 'supertrophic' or 'hypertrophic' (Figure 1). These trophic states correspond to average chl *a* and total nitrogen (TN) concentrations that exceed the "national bottom lines" established by the New Zealand Government (2017a). These lakes represent some of the most nutrient-polluted lakes in the country, e.g., note that the TLI value for Lake Mangahia in Figure 1 (7.2 TLI units) exceeds the highest value of 7.0 that was envisioned when the TLI system was developed (Burns *et al.* 1999). Further, the trophic status of many Waikato shallow lakes is comparable with highly degraded lakes in the USA and Europe (Verberg *et al.* 2010) that have been subjected to pressures from intensive agriculture for a longer duration than lakes in the Waikato region.

Sufficient data to evaluate trends in trophic state are only available for a few lakes. WRC (Waikato Regional Council 2014b) presents trend information for 12 lakes, based on monitoring conducted during discrete periods during 1993–2012; this analysis showed that water quality improved during specific periods in two lakes (Hakanoa and Whangape), declined during specific periods in four lakes (Waikare, Waahi, Rotopiko East, Rotomanuka), and exhibited no change during the periods analysed in the remaining six lakes.

During 2004–2016, NIWA measured LakeSPI in 62 Waikato lakes, of which 44 were shallow based on the 10 m maximum depth criterion (Burton and de Winton 2016). Almost all lakes had undergone a significant decline in LakeSPI relative to a 'pristine' (1900) state. One shallow lake was in 'excellent' condition (Koraha), three shallow lakes were in 'high' condition (Serpentine North, Serpentine East and Rotopounamu) and three shallow lakes were in 'moderate' condition (Harihari, Rototapu and Puketi). The remaining 37 (84% of shallow lakes) were classified as either poor (LakeSPI Index ≤ 20%) or unvegetated (LakeSPI Index = 0%), in which case the lakes were completely de-vegetated of submerged rooted macrophytes and had therefore 'flipped' to a phytoplankton-dominated state. The prevalence of lakes in unsatisfactory condition was higher in the Waikato than in New Zealand generally.

The relatively eutrophic state of shallow Waikato lakes is further illustrated by **Error! Reference ource not found.**, which compares data for the 20 shallow Waikato lakes presented in Figure 1 with data for lakes from three other geographic regions. This figure is intended to provide context for the literature review, which draws extensively on studies in Europe and the USA. The figure is not intended to quantify water quality degradation in the Waikato and this international comparison fails to consider the extent to which lakes have changed from baseline or reference conditions. For example the Waikato lakes are exclusively shallow whereas the other lakes are not; this confounds the analysis because shallow lakes are typically naturally more productive than deep lakes (Section 3). Nonetheless, the comparison highlights the high productivity of shallow Waikato lakes; for example, the median value of the mean TN concentrations for the shallow Waikato lakes (1.503 mg/L) was greater than the mean TN concentration of approximately 85% of the lakes in EU countries, 66% of the USA lakes and 90% of the other New Zealand lakes in the sample.

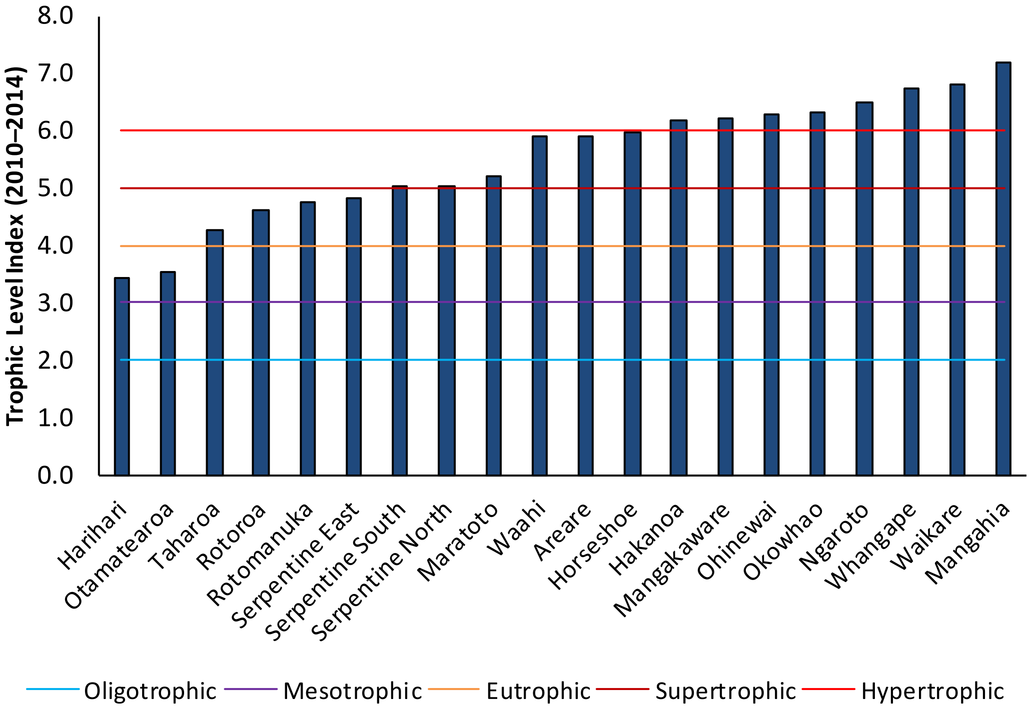

Figure 1. **Five-year Trophic Level Index value for 20 shallow Waikato lakes.**

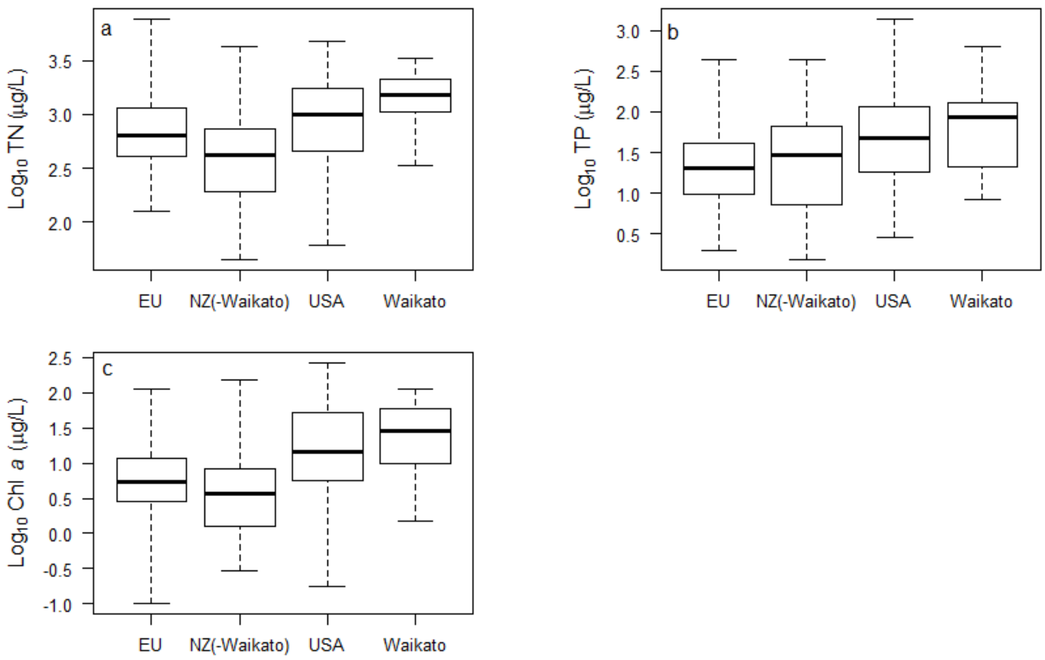

Figure 2. Box and whisker plots comparing mean concentrations of total nitrogen (TN; a), total phosphorus (TP; b) and chlorophyll *a* (Chl *a*; c) among lakes from the following regions: European Union countries (*n* = 689), New Zealand (excluding the Waikato; *n* = 110), the USA (*n* = 311) and the Waikato (20 shallow lakes presented in Figure 1). Data sources for the first three categories are described in detail in Abell et al (2010)**. Note $\log_{10}$-transformed y-axes.**

Current approaches to restoration

This review considers national and international studies of lake restoration; however, it is useful to summarise the main restoration methods that have been trialled or proposed for Waikato shallow lakes to provide context. Table 2 presents an overview of the main methods that have been trialled although it is not exhaustive and readers should refer to the Shallow Lakes Management Plan (Waikato Regional Council 2014a,b)and associated information for further details of the range of lake restoration actions that WRC and its partners have undertaken in recent decades.

To date, restoration measures have predominantly focussed on controlling external nutrient loads from agricultural land using methods that are broadly classed as 'BMPs' and include riparian fencing, restoring wetlands, installing constructed treatment systems (CTS) such as silt traps on farm drains, and riparian planting. Research to develop these methods has been undertaken at numerous sites in the Waikato. Notably, monitoring and research has been conducted since 1995 by NIWA in the Toenepi Stream catchment (Tanner *et al.* 2005), which is one of five Best Practice Dairying Catchments that are located throughout New Zealand (Ministry for the Environment 2009).

Pest fish control has been an active area of research in the Waikato region, partly facilitated by the Invasive Fish research programme within the Lake Ecosystem Restoration New Zealand group at the University of Waikato. Pest fish control is primarily intended to improve water quality by removing non-indigenous benthivorous fish such as common carp (*Cyprinus carpio*) that suspend bed sediments (discussed further in Section 5.4.1). Successful methods include: trap netting, boat electrofishing and pod traps (pyramid-shaped nets equipped with feeder stations to attract pest fish) (Hicks *et al.* 2015). Pest fish control programmes have been undertaken at lakes that include the Rotopiko lakes, Lake Rotomanuka, Lake Rotoroa and Lake Ohinewai.

The use of flocculants to reduce internal nutrient loads in lakes is a well-established restoration method internationally (e.g., Welch and Cooke 1999) but field applications of this method have not been undertaken in Waikato region. A pilot study to assess the applicability of methods to reduce internal loads in four peat lakes (Ngāroto, Kainui, Rotomanuka and Cameron) identified opportunities to use certain materials but recommended that external loads be first reduced (Faithfull *et al.* 2005). A recommendation has been made to investigate the feasibility of dosing aluminium sulphate to a lake inflow for Lake Waahi and Lake Ngāroto (WRC 2017b).

The use of constructed floating wetlands is also being studied at lakes Kaituna, Rotopiko, Koramatua, Areare, Rotomanuka and Ruatuna (WRC 2014b). The technique may have potential to reduce nutrient concentrations in the water column by plant uptake (reviewed in Section 5.4.4).

Table 2. An overview of actions that have been implemented to restore water quality in individual Waikato shallow lakes (Waikato Regional Council 2017b). This table is not exhaustive.

Lake	Restoration action (implemented)
Lake Rotomānuka	Constructed Treatment Systems (CTS) – sediment basin, silt traps, infiltration wetlands. Six constructed.
Lake Mangakaware	Install CTS on two largest drains Install stock exclusion fencing on reserve boundary
Lake Ngāroto	CTS installed on some inflows Whole Farm Plans prepared for eight farms in the catchment
Lake Rotopotaka, Lake Pataka and Lake Posa	Wetland enhancement at each lake: -weed control -planting -fencing
Lake Rotopiko	Pest fish removal (weir installation)
Lake Areare	CTS installed in lake inflows (silt traps and constructed wetlands)
Lake Kimihia	Weed control and planting
Lake Ohinewai	Removal of stock from District Council Reserve Pest fish removal, including installing a carp exclusion barrier Revegetation
Lake Te Kapa and Lake Waiwhata	Fencing of wetlands Weed control
Lake Okowhao	Stock exclusion fencing around lake
Lake Whangape	Wetland fencing (partial)
Lake Waahi	Riparian fencing and planting

3 Eutrophication in shallow lakes

3.1 Nutrient sources

A major pressure to shallow lakes is eutrophication, which is increased productivity associated with elevated nutrient loading. Nitrogen and phosphorus are the key nutrients of concern (Vollenweider 1971; Gluckman 2017). A range of human activities can increase the external loads of nitrogen and phosphorus to shallow lakes; these include: municipal wastewater disposal, use of septic tanks, and agriculture (Smith et al. 1998). Globally, agriculture and urban areas are the primary sources of nitrogen and phosphorus to freshwaters, with agriculture dominating in rural areas (Carpenter *et al.* 1998).

In New Zealand, the proportion of high intensity pastural land is a strong predictor of nutrient concentrations in lakes, with the water quality of shallow lakes being particularly sensitive to catchment land use (Abell *et al.* 2011b). The main sources of nitrogen to catchments are nitrogen fixed by legumes in pasture and fertiliser (Parfitt *et al.* 2006), while the major source of phosphorus is fertiliser applied to pasture (Parfitt *et al.* 2008). In the Waikato, these agricultural sources account for 73% of nitrogen inputs and 90% of phosphorus inputs, with total inputs of both nutrients greater than inputs in any other region (Parfitt *et al.* 2006, 2008).

External nutrient loads are transported to lakes via lake inflows, including groundwater and surface water, with a minor component (most notably for nitrogen) also transported to lake surfaces via atmospheric deposition. Typically, a high proportion of nitrogen transport in catchments is as nitrate transported in sub-surface pathways through soil or groundwater aquifers (Petry *et al.* 2002). By contrast, a major proportion of the phosphorus load to lakes comprises phosphorus bound to soil particles that is transported in surface waters (e.g., perennial streams or overland flow), although sub-surface fluxes of phosphorus can be significant, particularly in agricultural catchments with tile drainage (McDowell *et al.* 2004; King *et al.* 2015). The relative loads transported in surface streams (ephemeral and perennial) and groundwater are highly dependent on catchment geomorphology.

3.2 Sediment fluxes

Lake bed sediments exert an important control on water quality in shallow lakes. Sediments act as a sink for nutrients originating from the catchment, which may then be recycled within the lake (internal loading). In shallow lakes, the link between sediment nutrient release and phytoplankton growth is direct because vertical stratification is absent or only intermittent, resulting in direct transfer of nutrients from the sediment-water interface to the euphotic zone.

In particular, internal loads of P can be high in shallow lakes (Søndergaard *et al.* 2003), although internal loading of nitrogen (as ammonium) can also be substantial (Burger *et al.* 2007; Mátyás *et al.* 2009). Internal loads may exceed external loads on an annual basis in lakes that have highly nutrient-enriched sediments due to a legacy of high external loads. For example, sediment release rates measured in Lake Rotorua (Bay of Plenty; mean depth = 11 m) demonstrated that bottom sediments were the dominant source of nitrogen and phosphorus on an annual basis, at least during the mid-2000s (Burger *et al.* 2007). Similarly, a 26-year study of internal loading in Lake Säkylän Pyhäjärvi (Finland; mean depth = 5.5 m) estimated that the internal phosphorus load was approximately 60% of the average external load, although the internal load was dominant in years with high water temperatures and low inflow (Nürnberg *et al.* 2012). Such high relative internal loads are likely to be exceptional; a study of four hypereutrophic and shallow (maximum depth = 4.3–4.9 m) agricultural reservoirs in Nebraska, USA measured internal loads to be only 4–12% of the total annual phosphorus load, although internal loads were highest during the summer when they exceeded external loads in two of the lakes, potentially exacerbating and prolonging algal blooms (Song *et al.* 2017). Similarly, see Section 6.4 for discussion of how lake water quality improvements occurred atypically fast (1–2 years) during restoration of hypereutrophic Lake Apopka, indicating a low contribution from internal loading.

The mechanisms by which nutrients can be released from sediments can vary considerably among lakes. In particular wind-induced sediment resuspension rates can be high in shallow lakes due to high benthic shear stresses during windy conditions (Hamilton and Mitchell 1997). Resuspension mobilises nutrients and increases total suspended sediment (TSS) concentrations, reducing water clarity. Lake Waikare is an example of a shallow Waikato lake that has high inorganic suspended sediment concentrations caused by resuspension of clay-sized particles (Allan 2016a). Other important release mechanisms include: 1) anoxia-mediated release of P from redox-sensitive iron compounds; 2) macrophyte senescence, and; 3) bioturbation (Welch and Cooke 1995). Bioturbation by invasive fish is discussed in Section 5.4.1 below.

Physicochemical characteristics exert a major control on nutrient release from the sediments. In addition to sediment nutrient concentrations, sediment fluxes are also highly dependent on redox conditions and temperature (Søndergaard *et al.* 2003; Nürnberg *et al.* 2012). The range in the relative potential contribution of internal loading to lake productivity highlights the importance of having an adequate understanding of the limnological characteristics of a lake, particularly those of the bed sediments, when framing expectations regarding how water quality in an individual lake will respond to restoration.

Climate change is expected to exacerbate internal loading in many regions due to changes to these physicochemical factors. Although there is uncertainty about how changes to lake processes due to climate change will interact (Jeppesen *et al.* 2007b), warmer temperatures are expected to increase internal loading due to increased mineralisation rates and reduced dissolved oxygen concentrations in warmer waters (Jeppesen *et al.* 2009b; Nürnberg *et al.* 2012). This is discussed further in Section 7.6.

3.3 Alternative stable states

In undisturbed catchments, water clarity in shallow lakes is generally high. This allows sufficient light to penetrate the water column for rooted macrophytes to widely colonise and maintain an important role in structuring the ecosystem (Scheffer 2004; Bakker *et al.* 2013). Rooted macrophytes serve to maintain desirable clear water conditions by up-taking dissolved nutrients from the water column, stabilising bed sediments, providing refuge for invertebrate grazers and minimising wave action (Scheffer 2004; Phillips *et al.* 2016). Prior to disturbance, it is expected that submerged macrophytes were prevalent in shallow lakes in the Waikato region, with the caveat that naturally elevated dissolved organic carbon concentrations reduce the euphotic depth to some extent in peat lakes, affecting the depth that macrophytes occur. In general, clear macrophyte-dominated conditions therefore represent an ecologically desirable and stable state in shallow lakes.

Increased nutrient loads to shallow lakes promote the growth of phytoplankton (suspended algae). Increased phytoplankton biomass can be associated with a range of water quality problems, including odour issues and proliferation of nuisance phytoplankton species (e.g., Cyanophyta spp.) that may produce toxins and have low nutritional quality for aquatic life (Carpenter *et al.* 1998; Reynolds 2006). Further, proliferation of phytoplankton biomass increases turbidity, leading to decreased penetration of light (photosynthetic active radiation) and a decline in the distribution and biomass of rooted macrophytes. Once a critical turbidity threshold is reached, a lake that has a primary producer community dominated by macrophytes may 'flip' to an alternative stable state that is dominated by phytoplankton (Figure 3). Since first proposed (Scheffer *et al.* 1993), this theory of alternative stable states has been developed to reflect that the critical turbidity threshold is mediated by factors such as lake size, and that 'flipping' occurs along a continuum of gradual changes in the dominant communities, e.g., from green algae to cyanobacteria (Scheffer and van Nes 2007). Further, it has been recognised that macrophyte collapse can (briefly) precede phytoplankton proliferation in cases where factors such as temporarily high waterfowl grazing or uprooting during storm events damage macrophyte beds, resulting in a shift to phytoplankton dominance due to removal of the stabilising influence of macrophytes (Phillips *et al.* 2016). Nonetheless, the theory of alternative stable states is critical for understanding how water quality in shallow lakes responds to pressures and restoration (discussed further in Section 7). Schallenberg et al. (2009) catalogued New Zealand lakes that had changed from a macrophyte-dominated clear water state to a devegetated, turbid state. They identified 37 lakes that had undergone regime shifts, all of which had maximum depth <20 m, highlighting that such shifts are a feature of relatively shallow lakes. Fifteen (41%) of the lakes were in the Waikato region; these had maximum depths of 1.8–8.0 m. Lakes that had undergone regime shifts were shown to be positively correlated with the proportion of pasture in lake catchments (an indicator of nutrient loading), the presence of an invasive macrophyte (*Egeria densa*) and the presence of five exotic fish species.

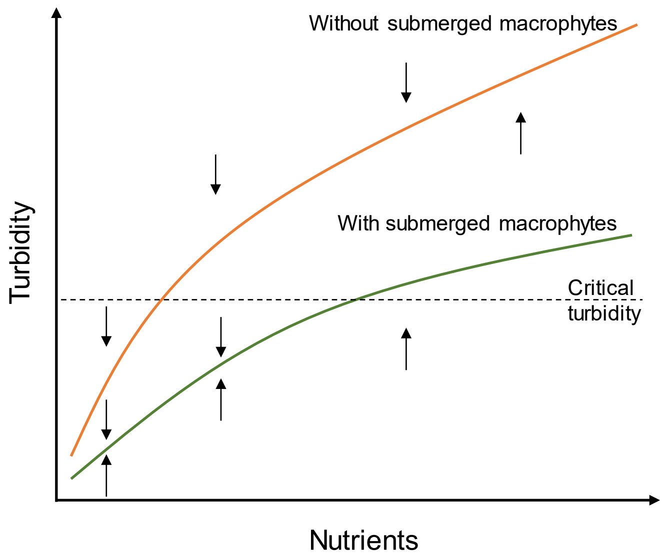

Figure 3. Alternative stable states in shallow lakes. In temperate latitudes, shallow lakes tend to exist in either a clear macrophyte-dominated state, or a turbid phytoplankton-dominated state. Once a critical turbidity threshold is reached, lakes are susceptible to 'flip' to an alternative stable state. Redrawn from Scheffer et al. (1993).

3.4 Invasive fish

Colonization by non-indigenous invasive fish is a major ecological stressor to lakes in the North Island, with shallow lakes in the Waikato particularly affected (Rowe 2007; Collier and Grainger 2015). Key invasive fish species in the Waikato include common carp, brown bullhead catfish (*Ameiurus nebulosus*), rudd (*Scardinius erythrophthalmus*), tench (*Tinca tinca*) and gambusia (*Gambusia affinis*) (Rowe and Graynoth 2002). Invasive fish have widely colonised the region, although they are particularly prevalent in riverine lakes within the Waikato River floodplain (e.g., lakes Whangape, Ohinewai and Waahi) because fish can move easily between lakes during high water periods (Daniel 2009). Peat lakes such as Lake Ngāroto (Berry and Dresser 2012) are also widely affected.

Invasive fish introductions can exacerbate eutrophication, although it can be difficult to identify the key mechanisms involved because they vary among fish species, and multiple invasive species are typically present in affected lakes (Rowe 2007). Koi (common) carp are present at particularly high biomass in many Waikato lakes, e.g., 308 kg/ha estimated in Lake Ohinewai in 2011 (Tempero and Hicks 2017). Therefore, it is likely that bioturbation (Havens 1991) or direct excretion (Morgan and Hicks 2013) by this benthivorous species is an important contributor to nutrient cycling in affected lakes, with the caveat that other internal loading processes may still be at least as important for maintaining high nutrient concentrations in affected lakes (Bajer and Sorensen 2015). Increased TSS concentrations due to bioturbation also likely accelerates the deterioration of submerged macrophyte communities by reducing light penetration. Invasive fish introductions could also potentially exacerbate the symptoms of eutrophication by top down effects, such as zooplankton predation by gambusia or macrophyte grazing by rudd. Such top down controls have been shown to be important in shallow lakes in Denmark (Jeppesen *et al.* 1997), although their significance in New Zealand is uncertain (Rowe and Graynoth 2002; Rowe 2007). Details of biomanipulation of fish communities to restore eutrophic lakes, and its applicability to Waikato lakes, are discussed further in Section 5.4.1.

4 Drivers of lake restoration

4.1 International

Regulation is an important driver of lake restoration actions in many countries, notably Europe and the USA. In Europe, the Water Framework Directive (2000/60/EC) has required the ecological quality of waterbodies to be determined, including shallow lakes (Moss *et al.* 2003). The directive requires defining reference (undisturbed) conditions for habitat types (ecotypes), in addition to different grades (good, moderate, poor and bad), with the requirement that waterbodies are restored to 'good' ecological status. Member states are required to transpose the requirements of the Directive into national legislation, thus creating a regulatory driver. In the USA, the Clean Water Act regulates the discharge of pollutants to the waterbodies and the use of water quality standards for surface waters. The act includes four tools to manage diffuse pollution: water quality standards, total maximum daily loads, National Pollution Discharge Elimination System permits, and support for a Nonpoint Source Program (reviewed by McDowell *et al.* 2016).

Although, national regulation has been an important driver of lake restoration activities elsewhere, compliance with legislation alone is often inadequate for achieving ecological and water quality aspirations in many watersheds. For example, the implementation of the Water Framework Directive has been criticized for lacking ambition and failing to define and manage ecological quality in a way that recognises the inherent variability among waterbodies (Moss 2007a). Similarly, the Clean Water Act has been identified as a weak driver of the ecological restoration of waterbodies (Palmer and Ruhl 2015). Critics have claimed that the success of the Clean Water Act for addressing non-point nutrient sources has not matched its success with addressing point sources, and that the act is an insufficient driver for implementing in-lake restoration actions such as flocculation (NALMS 2014). Gross and Hagy (2017) reviewed 16 worldwide case studies that involved restoring waterbodies (including seven lakes) by reducing nutrient loading. The authors identified three drivers (which they term 'antecedents') for undertaking nutrient management: 'public crisis', 'government mandate', and 'existing networks with funding incentives'. 'Public crisis' was identified as the most common driver, which is characterised by local citizens demanding action in response to a highly public ecological crisis. The authors concluded that the four most important attributes of successful cases were: 1) leadership by a dedicated watershed management agency; 2) governance through a bottom-up collaborative process; 3) a strategy that set numeric targets based on a specific ecological goal, and; 4) comprehensive actions to reduce nutrient loads from all sources (e.g., not just point sources).

Thus, voluntary instruments, in addition to mandatory measures, have been identified as an important component of the policy mix required to promote successful freshwater restoration. This was also highlighted by McDowell et al (2016) in an international review of policies for phosphorus control. The importance of management actions initiated voluntarily by local groups (i.e., bottom-up governance) is partly due to the ability to use local ecological knowledge to support adaptive management approaches that are often required to address the specific mix of issues that cause the ecological impairment of individual lakes (Olsson and Folke 2001). The importance of such bottom-up or grass-roots governance is also supported by a sociopolitical study of stream restoration in the USA, which highlighted the importance of demographic factors as a driver of the occurrence of ecological restoration (Stanford *et al.* 2018). Specifically, the study showed that restoration projects were clustered in areas with a high population of affluent and well-educated citizens, and that restoration was highly dependent on the presence of a local restoration organisation. This finding supports the observation that local watershed groups often have an instrumental role in promoting and overseeing lake restoration projects, e.g., Friends of Lake Tahoe in the USA.

4.2 National

4.2.1 Policy drivers

Water quality is currently a pressing issue in New Zealand. In particular, the subject has received heightened expert scrutiny in the last decade (e.g., Land and Water Forum 2010, 2015; Gluckman 2017; Ministry for the Environment & Stats NZ 2017) and it was a prominent issue in the 2017 general election. There has been growing interest in understanding how water quality in New Zealand compares with other countries (Verberg *et al.* 2010; OECD 2017), which has contributed to increased awareness of how the adverse water quality effects of agricultural intensification represent environmental externalities that compromise the nation's 'clean and green' brand (Abell et al. 2011a; Foote et al. 2015).

The mix of broad policy instruments to address nutrient pollution in New Zealand lake catchments is shown conceptually in Figure 4. Command and control regulation is discussed in Section 4.2.2 below, while Table 2 presents examples of voluntary actions that have been undertaken by landowners (albeit often with financial assistance from the regional council). Economic instruments have received limited attention to date in the context of shallow lake restoration although, as noted below, the development of economic tools to manage freshwater quality has been identified as a national priority (Ministry for the Environment 2013). Underpinning these tools is the role of education and engagement to inform stakeholders such as landowners about the issues and promote change. Regional Nutrient Management Indicators to assist landowners with benchmarking nutrient losses (DairyNZ and FertResearch 2017) are one example of an educational tool.

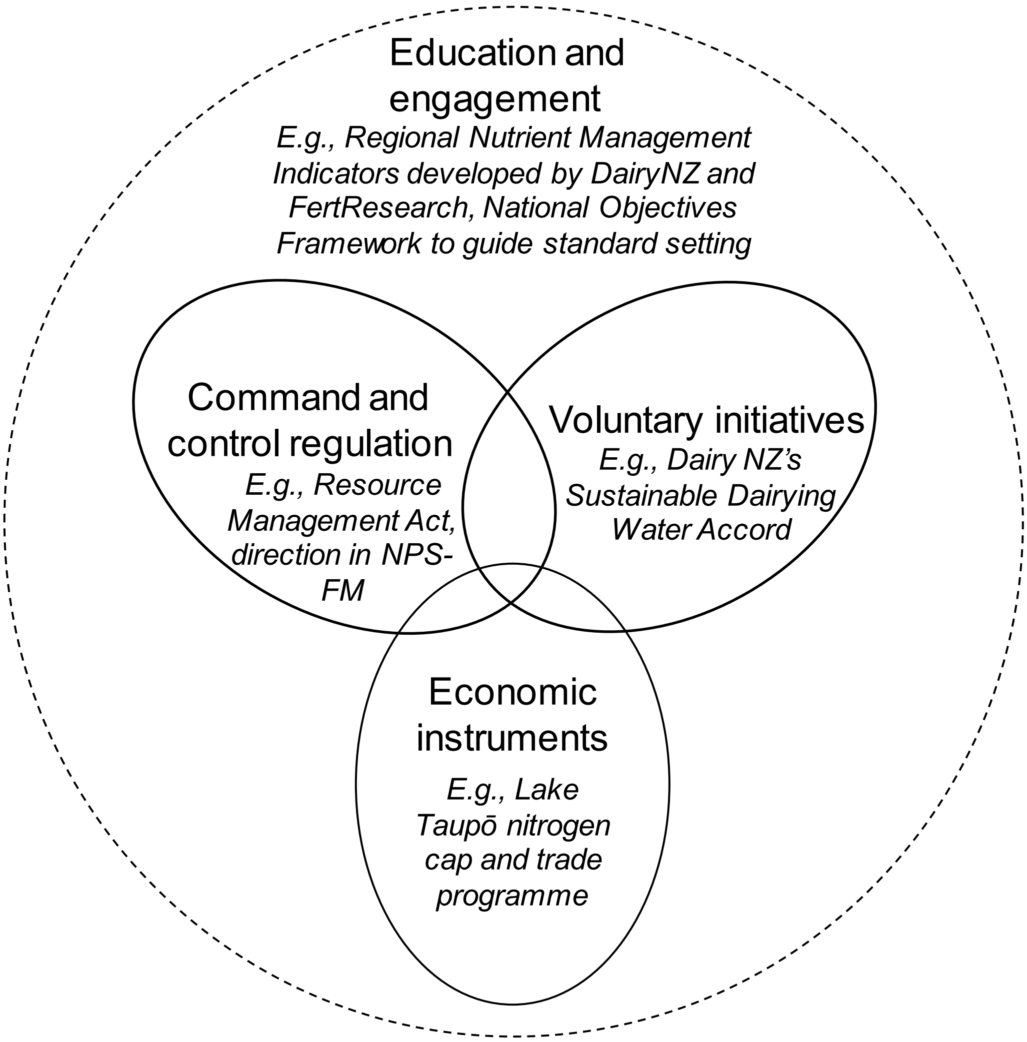

Figure 4. Broad policy instruments to control nutrient pollution in catchments, underpinned by policies to promote education and engagement. Adapted from Abell et al. (2011a).

In 2013, the New Zealand Government set out an approach to reform national freshwater management (Ministry for the Environment 2013). This requires regional councils to set objectives, guided by a National Objectives Framework. The approach emphasises managing water quality within limits and supports the development of new tools to do this, e.g., transfer, offsetting or pricing mechanisms. The National Policy Statement for Freshwater Management (NPS-FM; New Zealand Government 2017a) provides further direction regarding this reform (discussed below).

In 2017, the New Zealand Government published 'Clean Water', which provided an update on freshwater management reform (New Zealand Government 2017b). This document proposed a target of 90% of large[3] rivers and lakes swimmable by 2040, with 'swimmable' defined based on compliance with a National Bottom Line of an 80th percentile value for cyanobacteria volume of < 1.8 mm^3/L. The document also included requirements for stock exclusion from waterways, including the requirement that all dairy cattle on milking platforms be excluded from waterways > 1 m wide after 1 July 2017.

4.2.2 Legislation

The Resource Management Act (New Zealand Government 1991) is New Zealand's principal piece of environmental legislation, designed to provide for sustainable management of environmental resources. As described in WRC (2014a), Section 30 of the Act sets out the functions of regional councils that include developing objectives, policies and methods to maintain and enhance water bodies and water quality. Section 6 of the Act outlines matters of national importance that need to be recognised and provided for in relation to managing the use, development, and protection of natural and physical resources including the natural character of wetlands and lakes. The Act requires councils to consider the intrinsic value of ecosystems, enhancing the quality of the environment and the principles of the Treaty of Waitangi.

The NPS-FM provides direction to regional councils on undertaking their responsibilities for managing fresh water under the Resource Management Act. The NPS-FM requires regional councils to set objectives for the state of freshwaters and to set limits on resource use to meet these objectives. With regard to lakes, the NPS-FM defines categories (Attribute States) for the following water quality variables phytoplankton biomass (chl *a* concentration), TN, total phosphorus (TP), *E. coli*, ammonia and planktonic cyanobacteria biovolume. These Attribute States are designed to assist regional councils with setting objectives. In addition, a National Bottom Line has been set for each water quality variable that is the minimum acceptable state, below which action is required to improve water quality. Such bottom lines are intended to provide clarity about the minimum extent of restoration that is required to remediate water quality in failing waterbodies (New Zealand Government 2017b). Water quality in many shallow lakes in the Waikato region fails to comply with these National Bottom Lines. As an indication of this, note that the National Bottom Line for chl *a* is an annual median concentration of 12 µg/L, which is an equivalent value to the upper bound for annual average chl *a* in the 'eutrophic' category that is exceeded by 18 of the 20 lakes represented in Figure 1.

In the Waikato, treaty settlement legislation has been a major driver of freshwater restoration. In particular, the Waikato River Clean-Up Trust is an important driver, as discussed in Section 4.3.3.

4.2.3 Funding

Public funding has been an important national driver of lake restoration in New Zealand (Hamilton *et al.* 2016) and the New Zealand government has proposed to invest NZ$100 million over 10 years to improve freshwater quality in surface and ground waters through the Freshwater Improvement Fund (New Zealand Government 2016). In the Waikato, two projects have so far been funded through this fund. First, approximately $1 million has been allocated over five years to undertake restoration actions in Lake Whangape that include fencing, planting,

[3] For lakes, this is defined as those with a perimeter >1.5 km.

invasive plant containment and implementation of a Kaitiaki Monitoring Framework. Second, approximately $750,000 has been allocated over five years to improve water quality along the Pūniu River (Ministry for the Environment 2018). Funding for lake restoration in the Waikato has also been provided by a range of other agencies, including the Department for Conservation, Fish and Game New Zealand and multiple iwi groups.

4.3 Regional

4.3.1 Waikato Freshwater Strategy

Waikato Regional Council adopted the Waikato Freshwater Strategy in June 2017 (Waikato Regional Council 2017a). The Waikato Freshwater Strategy is in line with the Vision and Strategy for the Waikato River, which was developed as part of the Waikato River Settlement between the Crown and Waikato-Tainui, Te Ture Whaimana o Te Awa o Waikato.

The goal of the strategy is to "Achieve the best use of fresh water through time via better allocation systems using new methods based on better information." The strategy recognises that freshwaters are degrading due to the legacy effects of agricultural land use, while freshwater demands are increasing. A key element of the strategy is the recognition that WRC needs access to a wider range of policy tools beyond the legislative measures available under the Resource Management Act (New Zealand Government 1991). In particular, the plan recognises that economic instruments (e.g., water pricing) should have a greater role in water management.

4.3.2 Proposed Waikato Regional Plan Change

The Waikato Freshwater Strategy identifies opportunities to improve freshwater management when revising the Operative Waikato Regional Plan. The Proposed Waikato Regional Plan Change 1 (Waikato and Waipā river catchments) (Waikato Regional Council 2016b) developed from the Healthy Rivers: Plan for Change/Wai Ora: He Rautaki Whakapaipai project. The plan change is designed to meet the requirements of the New Zealand Policy Statement for Freshwater (New Zealand Government 2017a)and the Waikato Iwi Settlement Act (Office of Treaty Settlements 2010).

The proposed change to the Operative Waikato Regional Plan is designed to manage discharges of nitrogen, phosphorus, sediment and microbial pathogens to land and water. The plan change includes a suite of policies that are designed to manage diffuse and point sources of these pollutants within prescribed Freshwater Management Units. Policy 14 relates to Lakes Freshwater Management Units. The policy requires WRC to:

"*Restore and protect lakes by 2096 through the implementation of a tailored lake-by-lake approach, guided by Lake Catchment Plans prepared over the next 10 years, which will include collecting and using data and information to support the management of activities in the lakes Freshwater Management Units*".

The plan change defines separate targets for four lake classes (dune, riverine, volcanic and peat) in relation to the following water quality variables: chl *a*, TN, TP, *E. coli* and clarity. These targets are to be used in decision making processes, although they are not intended to be used directly as receiving water compliance standards.

4.3.3 Waikato River Clean-Up Trust

The national government has directed NZ$210 million into the Waikato River Clean-Up Trust (New Zealand Government 2016). The trust was established following a treaty settlement process which recognised that the Waikato River has been degraded following European settlement, severely compromising the ability of Waikato River iwi to exercise kaitiakitanga or conduct their tikanga and kawa (Waikato River Authority 2018). This trust is applicable to all lakes and wetlands in the Waikato River catchment and funds projects that include those that are directly focused on shallow lake restoration (Waikato River Authority 2017). The sole trustee of the trust is the Waikato River Authority who determines which projects receive contestable funding each year. Current funding priorities include: investigating the impacts of koi carp removal from lakes; using innovative technologies to improve lake water quality, and; enhancing water quality at lakes Waahi, Waikare, Whangape and Rotoroa (Waikato River Authority 2018).

4.3.4 Shallow Lake Management Plan

The Shallow Lake Management Plan is a non-statutory tool for implementing WRC's responsibilities for shallow lake management. The plan includes two volumes. Volume 1 (Waikato Regional Council 2014a) provides details about the management context; legislative and policy drivers; the interagency agreements to shallow lake management in the Waikato and Waipā Districts; progress to date, and; objectives for shallow lake restoration. Appendix 1 of this volume provides an overview of legislative and policy drivers and the reader is referred to this appendix for further information. Volume 2 of the Shallow Lake Management Plan (Waikato Regional Council 2014b) summarises the current status and management recommendations for individual shallow lakes in the region.

The Shallow Lakes Management Plan includes nine policy objectives that are reproduced below:

Policy & planning objectives

1. Appropriate objectives, targets and limits are established for the future management and enhancement of shallow lakes
2. Water levels of shallow lakes and associated wetland margins are adequate to support hydrological and ecological processes and functions, and maintain or enhance the values associated with these.
3. The hydrology of shallow lakes (and their associated wetland margins) is protected from the effects of further wetland drainage.
4. Shallow lakes and their associated wetland margins are protected from the effects of stock access.

Information & monitoring objectives

5. Sufficient information is collected by WRC to assess and rank the biodiversity (SNA) values of all shallow lakes, and this information is analysed, reported, and used as the basis of effective lake management programmes.
6. Sufficient information is collected from shallow lakes to assess and report upon their condition (water quality and ecological health), and to assess the effectiveness of WRC's policy and planning framework and shallow lake management programmes.

7. WRC's lake level setting programme is underpinned by quality information, to ensure that shallow lake water levels are adequate to support hydrological and ecological processes and functions, and the values identified at these lakes.

Lake restoration & rehabilitation objectives

8. WRC supports the development, testing and implementation of methods and techniques to maintain and/or enhance the values of shallow lakes.
9. In conjunction with co-management partners, other agencies, stakeholders and landowners, WRC develops and implements integrated management and restoration programmes to protect and enhance priority shallow lakes, or valued aspects of these sites.

5 Techniques to address eutrophication in shallow lakes

5.1 Overview

This section describes the main techniques that have been used to restore shallow lakes degraded by eutrophication. The review focuses on approaches to improve water quality in shallow lakes that have undergone a shift to a phytoplankton-dominated state, associated with a substantial decline or complete loss of submerged macrophytes (see Section 3.3).

Broadly, these techniques may be grouped into four categories: 1) controlling external loads; 2) controlling internal loads, 3) biomanipulation, and; 4) hydrologic manipulations (Figure 5). From an ecological perspective, the first two categories involve 'bottom-up' control by reducing the concentrations of nutrients that drive primary production. Biomanipulation can either involve 'top-down' control (e.g., biomanipulation of zooplankton grazing rates) or bottom-up control (e.g., reducing benthivorous fish biomass to reduce sediment resuspension). Hydrologic manipulations can involve reducing external loads, increasing flushing, increasing depth, or modifying the water level regime to benefit riparian communities and wetlands. The applicability of emerging technologies is also discussed.

A summary of all methods is presented in Table 3, which includes examples of the application of individual techniques. Where relevant, cross-references are also provided to relevant long-term monitoring studies described in Section 5.5.

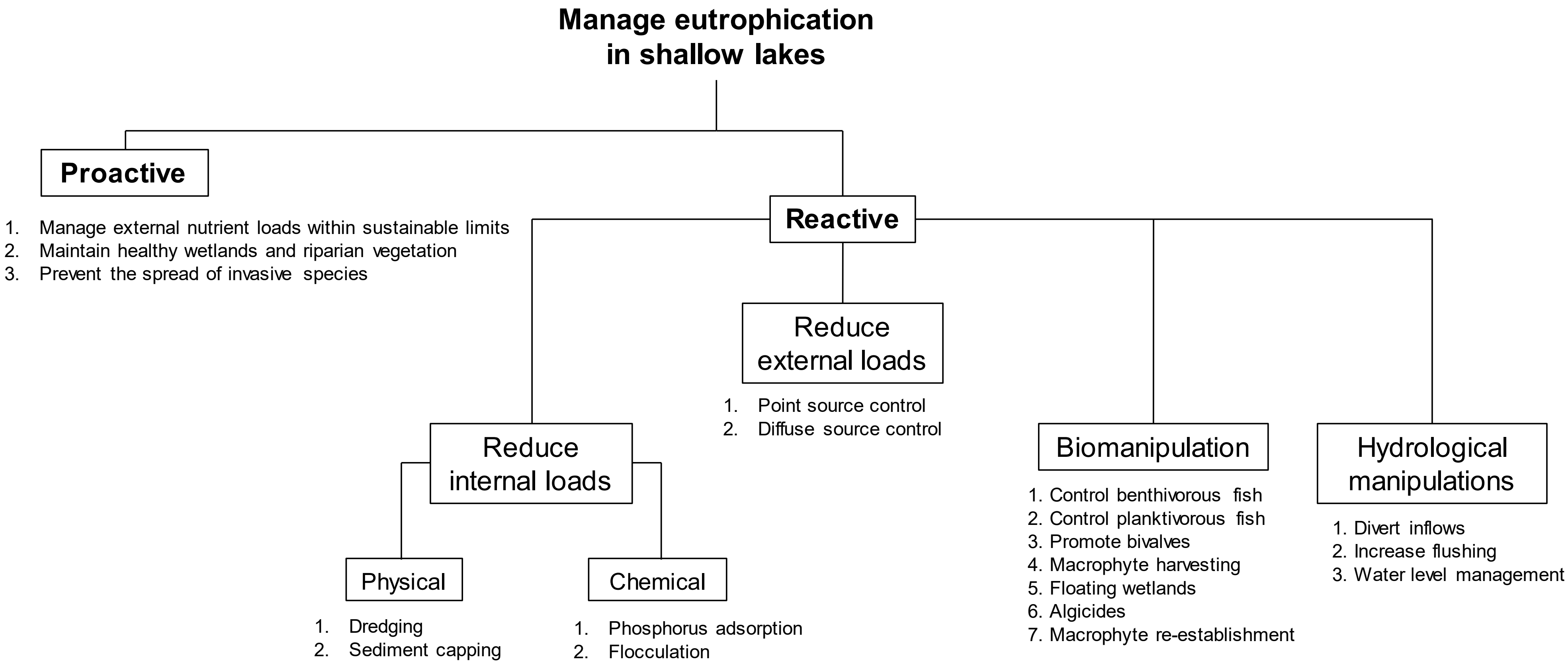

Figure 5. Overview of established methods to control eutrophication in shallow lakes. Updated and modified from Singh (1982).

Table 3. Summary of restoration techniques to address eutrophication in shallow lakes.

Group	Restoration Action	Purpose	Application	Example	Advantages	Disadvantages	References
Reduce external nutrient loads	Diffuse and point source control	Minimise nutrient loading	Essential component of a sustainable lake restoration strategy to control eutrophication	◦Lake Müggelsee, Germany ◦Lake Peipsi, Estonia/Russia ◦Loch Leven, Scotland ◦City Park Lake, Louisiana, USA	◦Addresses the root cause	◦Sufficient reductions typically require major economic costs, e.g., due to land use change or improved wastewater treatment	Jeppesen et al. (2005) Ruley & Rusch (2002)
Reduce internal nutrient loads (physical)	Dredging	Reduce internal loading by removing nutrient-enriched sediments	Best suited to small lakes and/or iconic lakes due to the high costs	◦City Park Lake, Louisiana, USA ◦Lake Kraenepoel, Belgium	◦Directly removes nutrients ◦Increases depth	◦Expensive ◦Disposal of dredgings can be difficult	Peterson (1979, 1981) Van Wichelen et al. (2007)
	Sediment capping (passive)	Reduce internal load by creating a physical barrier between benthic sediments and the water column	Generally suited to smaller lakes with high internal loads	◦Taihu Lake, PRC (one embayment)	◦May be opportunities to use inexpensive local soil/sand	◦Advese effects to benthic biota such as mussels ◦Adding foreign materials to waterbodies is culturally sensitive in NZ	Xu et al. (2012)
Reduce internal nutrient loads (chemical)	Phosphorus inactivation/ flocculation	Reduce concentrations of dissolved nutrients (primarily P) by adsorption. May be combined with using a flocculant to remove organic material.	Generally suited to smaller lakes with high internal loads	◦Minneapolis Chain of Lakes, USA	◦Potentially rapid improvements ◦Cost-effective (internal) load reductions ◦Well-established	◦Reduced efficacy in shallow lakes due to sediment resuspension ◦Adding foreign materials to waterbodies is culturally sensitive in NZ ◦Metal toxicity needs to be considered ◦Not a sustainable solution alone	Welch et al. (1988) Huser et al. (2016)
Biomanipulation	Fish removal (zooplanktivorous)	Increase cladocean zooplankton biomass → reduce phytoplankton biomass	May be applicable to lakes with invasive zooplanktivores, e.g., juvenile perch	◦Lake Vaeng, Denmark	◦Established method in northern hemisphere lakes with abundant zooplanktivores (e.g., roach)	◦Limited applicability to NZ lakes due to few zooplanktivorous fish and depauperate zooplankton communites	Søndergaard et al. (2008) Burns et al. (2014)
	Fish removal (benthivorous)	Reduce bioturbation and nutrient excretion	Applicable to lakes with high biomass of invasive fish such as common carp and tench	◦Lake Ohinewai, Waikato, NZ	◦Removing invasive fish can also support biodiversity objectives	◦High, ongoing effort required to maintain low biomass ◦Results are inconsistent	Søndergaard et al. (2008) Tempero & Hicks (2017)
	Promote bivalves	Increase filtration rates and phytoplankton grazing	Untrialled, although potentially suitable for lakes that are very shallow (relatively low volume) and oligo- mesotrophic (more suitable physicochemical habitat conditions)	◦Lake Faarup, Denmark (following an undesired invasion by zebra mussels)	◦Promote native biodiversity if kākahi are used in NZ	◦Limited abundance of host fish for larval development is a constraint ◦Habitat conditions may be unsuitable in lakes that are the greatest priorities for restoration	Jeppesen et al. (2012) Burns et al. (2014)

Table 3 continued.

Group	Restoration Action	Purpose	Application	Example	Advantages	Disadvantages	References
Biomanipulation (continued)	Macrophyte harvesting	Remove nutrients present in plant tissues	Very shallow (low volume) lakes with high abundance of invasive macrophytes	◦Lake Rotoehu, Bay of Plenty, NZ	◦Removing invasive plants can promote native plant biodiversity ◦Plants could provide a resource (e.g., feedstock), pending research and development	◦High, ongoing effort required to maintain low biomass ◦Nutrient removal expected to be minor compared with external loads	Carpenter & Adams (1978) Quillam et al. (2015)
	Floating wetlands	Uptake dissolved nutrients. Potentially also increase denitrification and settling.	Small lakes, embayments and drains where high coverage is feasible	◦Lake Kaituna, Waikato, NZ	◦May provide additional habitat values ◦Can provide a visual focus for lake restoration efforts	◦Field trials that demonstrate use to manage eutrophication are lacking ◦Not applicable to restore medium-large lakes ◦Plant harvesting necessary for optimum performance	Pavlineri et al. (2017)
	Algicides	Directly reduce phytoplankton biomass	May be suitable as an emergency measure	◦Cazenovia Lake, New York, USA	◦Effective at causing rapid short-term declines in phytoplankton biomass with sufficiently high doses	◦Toxic effects on other biota ◦Sediment contamination ◦Culturally insensitive in NZ ◦Not generally recommended as a lake restoration method	Effler et al. (1980) Fan et al. (2013)
	Macrophyte re-establishment	Promote re-establishment of macrophytes by planting founder colonies and/or protecting plants with exclosures and wave buffers	Suitable for lakes that have experienced improved clarity but macrophyte re-establishment is hindered by lack of viable seeds/propagules or grazing	◦Delta Marsh, Manitoba, Canada	◦Can yield improved macrophyte growth in some areas	◦Only suitable for lakes that have already been partially restored and have suitable light conditions	Evelsizer & Turner (2006)
Hydrologic alterations	Inflow diversion	Reduce external loads	Applicable to lakes for which external loads are dominated by a single surface inflow, and there is a suitable receiving waterbody nearby	◦Lake Rotoiti, Bay of Plenty, NZ	◦Step-change reductions in external loads	◦Potential ecological impacts to receiving waterbody ◦High capital costs ◦Feasiblity depends on local hydrology and not possible for most lakes	Hamilton & Dada (2016)
	Increase flushing	Dilute poor quality lake water with higher quality water	Applicable to lakes for which there is a suitable donor waterbody nearby	◦West Lake, PRC	◦Major improvement in water quality possible	◦Potential ecological impacts to donor waterbody ◦High capital costs ◦Feasiblity depends on local hydrology and not possible for most lakes	Jin et al. (2015)
	Water level management	◦Increasing depth can reduce sediment resuspension ◦May restore riparian vegetation, depending on the hydrologic regime	Very shallow lakes or lakes where the riparian vegetation communities are impaired due to the existing hydrologic regime	◦Volkerak-Zoommeer lake system, Netherlands	◦Can improve habitat for plants and wildfowl	◦Can only improve water quality indirectly ◦Land tenure can be a constraint to increasing lake level ◦Not a primary method to reduce trophic status	Gulati & van Donk (2002)

5.2 Controlling external nutrient loads

Anthropogenic eutrophication of lakes is caused by excess nutrient loading from the surrounding catchment, either from point sources such as municipal wastewater discharges, or diffuse sources such as leaching from pasture (Vollenweider 1971; Smith 2003) (Section 3.1). By corollary, the reduction of such external nutrient loading is therefore an important component of strategies to restore lakes that have been impaired by eutrophication. The control of phosphorus has been identified as particularly important, partly because it is most commonly the limiting nutrient in temperate lakes (Schindler 2012). However, it is also necessary to control nitrogen loads (Lewis 2011; Paerl *et al.* 2011), particularly in New Zealand where primary nitrogen limitation of phytoplankton biomass accumulation is relatively common (Abell *et al.* 2010). In shallow lakes, the rationale for controlling loads of nitrogen in addition to phosphorus is further supported by studies that showed that submerged macrophytes fail to colonise when nitrogen concentrations exceed threshold levels, regardless of phosphorus concentration (González Sagrario *et al.* 2005; Jeppesen *et al.* 2007b). Concerted efforts to reduce external nutrient loads typically result in water quality improvements following a lag period as sediment phosphorus concentrations and associated internal loads decline; see Section 6.2 for discussion of a detailed review of data from 35 lakes that were subject to external load controls.

Developing successful plans to control external nutrient loads requires knowledge of where, when and how nutrient losses are occurring from the catchment as many control techniques are specific to individual nutrients, sources and pathways (Hamilton et al. 2016). Preparing a lake nutrient budget is an important step in developing a plan to control external nutrient loads (Ministry for the Environment 2002). The accuracy of a budget will depend on the availability of good quality data regarding catchment hydrology, land use and land management practices (e.g., fertiliser application rates). Catchment models can be used to develop a nutrient budget based on current practices and, potentially, provide projections for alternate scenarios. In New Zealand, the Catchment Land Use for Environmental Sustainability (CLUES) model can be used to estimate annual loads of TN and TP to waterbodies, with a sub-catchment resolution of 0.5 km^2 (Elliott *et al.* 2016). CLUES can be used to simulate a range of land use/management scenarios and also estimate multiple socioeconomic indicators, thus providing a tool to evaluate catchment load reduction options. In the Waikato, nutrient budgets have been prepared for a limited number of lakes, e.g., Lake Waiwhakareke (Duggan 2012). The increasing adoption of whole farm system planning could support the development of accurate catchment nutrient budgets; farm system plans have been funded in lake catchments that include Kaituna, Tunawhakaheke, Mangakaware and Rotomanuka (Waikato Regional Council 2014b). Despite this, it can be a challenge to accurately quantify variability in external loads that arise from short-term (~hours) variability in discharge and water quality, particularly in low-order streams that are highly influenced by storm events, e.g., as demonstrated by high frequency sampling undertaken at Lake Mangakaware and the Rotopiko lakes (Tempero and Hamilton 2014).

In some catchments, point sources can account for a major component of external nutrient loads and therefore controlling these can provide substantial nutrient load reductions. For example, municipal wastewater was diverted from Lake Rotorua (Bay of Plenty, New Zealand) in 1991 (Rutherford *et al.* 1996), which had previously contributed approximately half of the external phosphorus load and a quarter of the external nitrogen load to the lake (Rutherford *et al.* 1989). However, in the majority of New Zealand lake catchments, diffuse pollution from agricultural sources is the primary concern for lake managers (Ministry for the Environment

2002; Gluckman 2017). A detailed review of the available techniques to manage catchment nutrient loads is beyond the scope of this review; instead, a summary of key measures to reduce external nutrient loads from pasture is presented in Table 4. Readers are referred to the cited literature for further details, e.g., McDowell and Nash (2012) present details of a range of phosphorus control measures, including expected effectiveness (% TP decrease) and approximate costs. A particular focus of research in the Waikato region has been the use of constructed wetlands to treat sub-surface drainage in pastural catchments. This has demonstrated that these systems can successfully attenuate nutrient fluxes, although performance can vary depending on maturity of the wetland and climatic conditions (Tanner *et al.* 2005; discussed further as an in-lake treatment in Section 5.4.4).

Table 4. Examples of actions to reduce nitrogen (N) and phosphorus (P) loads from diffuse sources in pastoral catchments. Based on McDowell et al. (2012) and Cherry et al. (2008).

Nutrient	Strategy	
	Manage nutrients at source	*Edge of field/lake treatments*
Nitrogen and phosphorus	Fertiliser management (e.g., avoid applying at high risk times)	Riparian fencing/planting
	Livestock management (e.g., reduce stocking rates)	Constructed wetlands
	Effluent management (e.g., adopt batch storage)	Restore natural wetlands
Primarily phosphorus	Soil P testing	Grass buffer strips
	Low solubility P fertiliser	Sediment traps
	Floodwater/irrigation management	Detainment bunds
Primarily nitrogen	Nitrification inhibitors (Di and Cameron 2006)	Denitrification beds (Schipper *et al.* 2010)

5.3 Controlling internal nutrient loads

5.3.1 Overview

Internal loading – i.e., the release of nutrients from lake sediments to the water column – can be substantial in shallow lakes (see Section 3.2). Consequently, numerous restoration techniques have been identified to control internal loads. Broadly, this category can be sub-divided into chemical and physical techniques (Figure 5; Table 3). Such techniques are typically applied to supplement actions to reduce external loads.

Hypolimnetic withdrawal (e.g., Nürnberg *et al.* 1987) and hypolimnetic aeration/oxygenation (e.g., Ashley 1985) can also be effective methods to reduce internal loads in lakes; however, these techniques are best-suited to deep lakes or reservoirs that exhibit stable stratification. For example, Bormans et al. (2016) identify hypolimnetic aeration/oxygenation as unsuitable in lakes shallower than 15 m, and conclude that hypolimnetic withdrawal should be avoided in shallow lakes due to the need to replace the discharged water. These techniques are therefore not discussed here. Hickey and Gibbs (2009) provide a useful decision support framework in the form of a flow diagram that can be used to identify appropriate techniques to reduce internal loads in a specific lake. The framework also provides guidance for assessing ecological risks associated with restoration options.

5.3.2 Dredging

Dredging involves removing lake bed sediments and therefore has the advantage of directly removing nutrients from a lake that have accumulated due to a legacy of high external loading. In shallow lakes, dredging is a primary method to physically control internal loads (Bormans *et al.* 2016).

Dredging can substantially reduce sediment nutrient releases; for example, a six-month sediment incubation experiment using samples from hypertrophic Taihu Lake (mean depth = 2 m; PRC) showed that maximum phosphorus release was reduced approximately 200-fold from 3,048 µg/m^2/d to 14.4 µg/m^2/d (Zhong *et al.* 2008). A separate study at Taihu Lake confirmed that reduced phosphorus release due to dredging results in reduced soluble reactive phosphorus (SRP) concentrations in the water column; field sampling there showed that peak surface water SRP concentrations at dredged sites (~50–55 µg/L) were slightly lower than peak concentrations at an un-dredged site (62 µg/L) (Cao *et al.* 2007). This reduction in water column SRP concentrations was markedly lower than indicated by the laboratory experiment, although there was presumably some mixing between dredged and un-dredged areas.

Removing nutrient-rich sediments is the main mechanism by which dredging reduces internal loads (Peterson 1981) and reduced internal loads due to dredging can result in stark ecological improvements in shallow lakes, e.g., see case studies of City Park Lake and Lake Kraenepoel in Section 5.5. Increasing the lake depth will also reduce the rate of wind-driven re-suspension. This was demonstrated in a modelling study of a eutrophic peat lake in the Netherlands (mean depth = 1.77 m), which showed that increasing the depth to 12 m in localised areas that cover 10% of the lake bed would reduce suspended particulate organic sediment by 25% and improve clarity (Penning *et al.* 2010). One further advantage of dredging is the potential to reduce the prevalence of cyanobacteria by the direct removal of akinetes (resting cells) from the sediments (Tsujimura and Okubo 2003). To date, dredging has had limited use in New Zealand (Hamilton and Dada 2016). This may reflect the high costs, meaning that it is most feasible for small lakes and those that are in high-use urban areas, as opposed to larger lakes in rural areas that are more typically the focus of restoration efforts in New Zealand, and for which it is more difficult to justify the costs. Dredging has been used at Lake Oranga (0.69 ha; Waikato University campus) and its use has been evaluated for Lake Rotorua (Bay of Plenty), for which the estimated cost is at least NZ$84 million (Hamilton and Dada 2016).

Potential adverse effects associated with mobilisation of suspended sediments should be evaluated when considering dredging options. Dredging can adversely affect benthic organisms such as kākahi (*Echyridella menziesi*) Depending on particle size and density, sediments suspended during the dredging process may also remain in the water column for extended periods (~days–weeks). This can temporarily reduce clarity and mobilise contaminants present in the sediments such as organochlorine pesticides or polychlorinated biphenyl (PCB) compounds (Peterson 1979). The presence of contaminants can pose difficulties for disposal of dredged sediments. The 'dredge-skim' method can potentially alleviate the need to dispose of contaminated sediments; this involves pushing the upper layer of sediments to one side of the lake during dredging and only removing the deeper, potentially less-contaminated material (Ruley and Rusch 2002). In cases where sediments are uncontaminated, the removal of nutrient-rich sediments can provide resource to amend agricultural soils (Peterson 1981).

As with other methods to control internal loads, dredging is best undertaken in conjunction with external load reduction as failure to reduce external nutrient loads will provide only temporary improvements before the surficial sediment layer becomes enriched with nutrients again (Kleeberg and Kohl 1999). Nutrient and sediment mass balances should be undertaken when evaluating the applicability of dredging to a particular lake to inform the likely success (based on internal load) and the expected longevity of water quality improvements (based on sediment composition and sedimentation rate) (Peterson 1981).

5.3.3 Sediment capping

Benthic sediment nutrient release can be substantially reduced by capping sediments with inert materials such as sand or gravel (Theis 1979). This creates a physical barrier between organic-rich sediments and the water column, thus reducing sediment oxygen demand, resuspension and nutrient efflux. Active capping materials can further reduce sediment release by adsorbing dissolved nutrients (see Section 5.3.4 below). The thickness of applied material should ideally be several cm, which means that costs can be prohibitive in larger lakes, although costs can be minimised by targeting application to deeper areas where existing sediment thickness is greatest (Hickey and Gibbs 2009).

Experiments with sediment cores collected from Taihu Lake (mean depth = 2 m; PRC) demonstrated that effective capping agents were sand and locally-sourced inorganic soils that had been combined with chitosan, which is a polysaccharide flocculant (Pan *et al.* 2012). The best results were obtained by applying the modified soil to flocculate phytoplankton cells, followed by a 2 cm capping layer of soil or sand. Concentrations of TP were reduced by 94% after 20 days in the treatments relative to a control, reflecting near zero phosphorus flux from the sediment in the treatments compared with 27 mg P/m^2/d in the control. Sand and modified soil were equally effective as capping agents during incubation experiments, although separate resuspension (stirring) experiments demonstrated that sand was superior for reducing simulated resuspension. These results were supported by a separate study conducted in situ at Taihu Lake, which demonstrated that SRP sediment flux was reduced by ~50% 18-months after capping with clean local soils (Xu *et al.* 2012). Adsorption by aluminium and iron in the soil was identified as important.

5.3.4 Phosphorus adsorption and flocculation

Adding materials to lakes to adsorb dissolved phosphorus is a well-established method for improving water quality in eutrophic lakes. Phosphorus adsorption has been used worldwide since at least the late 1960s, most notably in the USA (e.g., Welch *et al.* 1988; Welch and Schrieve 1994). The method involves applying materials to lakes to adsorb SRP and therefore reduce the bioavailability of phosphorus in the lake. Materials often have an additional flocculating property, which causes the removal of suspended sediments from the water column that contain particulate nutrients.

The most frequently used products have been aluminium salts, notably aluminium sulphate ('alum'). Alum reduces internal phosphorus loading due to adsorption of SRP to aluminium hydroxide to form a stable complex that is unavailable for phytoplankton growth (Hickey and Gibbs 2009):

$$Al_2(SO_4)_{3_{xH_2O}} + 2PO_4^{2-} \rightarrow 2AlPO_4 \downarrow + 3SO_4^{2-} + xH_2O \qquad \text{Eq.1}$$

A wide range of materials have been applied to lakes (Table 5). In addition to aluminium, calcium iron, lanthanum and organic material have all been used as the chemical agent to adsorb P; however, research has now confirmed aluminium and the rare earth element lanthanum as the "treatment binders of choice" (Wagner 2017). A constraint of iron is that iron-bound phosphorus may be released in soluble form under anoxic conditions (Welch *et al.* 1988) and addition of calcium is generally only suitable for hardwater lakes (Wagner 2017). Both aluminium and lanthanum can be combined with other products to enhance efficacy and create engineered solid-phase phosphorus sorbents (Lürling *et al.* 2016). For example, aluminium can be combined with the aluminosilicate mineral zeolite (Z2G1 or marketed as 'Aqual-P™') to provide a product that has high phosphorus adsorption capacity, but can also adsorb nitrogen in the form of ammonium (Gibbs *et al.* 2011). Similarly, Phoslock™ is a product consisting of bentonite clay modified with lanthanum that has been trialled in New Zealand and has the advantage that it aids flocculation, enhancing the removal of organic material (Hickey and Gibbs 2009). Flocculation can also be enhanced by adding a separate flocculent such as polyaluminium chloride (also see discussion of chitosan above). Such dual treatment has been highly successful for precipitating developing algal blooms and restoring small stratifying lakes from a eutrophic/hypertrophic state to an oligo/mesotrophic state (Lürling and Oosterhout 2013), although the efficacy is likely to be lower in shallow lakes as sedimented material is more readily reintroduced back into the water column. Unlike alum, modified zeolite and lanthanum modified bentonite products both settle rapidly, meaning that they can also form an effective physical sediment capping layer, in addition to adsorbing SRP in the water column (Zamparas and Zacharias 2014).

Materials are typically sprayed onto the lake surface with the objective of achieving an even application (Wagner 2017), although phosphorus adsorbing materials can also be used to bind phosphorus from external sources, e.g., by deploying in agricultural drains (McDowell and Nash 2012). It is important to consider site-specific constraints when identifying appropriate products and application strategies (Lürling *et al.* 2016). In particular, pH is an important consideration due to potential metal toxicity issues. For alum applications, it is necessary to buffer the lake to maintain pH > 6.5 to avoid the formation of highly toxic Al^{3+} ions (Hickey and Gibbs 2009). For lanthanum applications, studies to date have shown that concentrations generally remain below acute toxicological threshold of different aquatic organisms, except for very low alkalinity waters (Copetti *et al.* 2016).

Although phosphorus adsorbing materials have been widely applied to shallow lakes, the efficacy of such products is generally lower compared to applications in deep stratified lakes. For example, a study of alum treatments in nine polymictic (i.e., relatively shallow) and 12 dimictic (i.e., relatively deep) lakes showed that it is reasonable to expect alum treatment in polymictic lakes to be effective for 10 years, whereas 15 years longevity is reasonable for dimictic lakes (Welch and Cooke 1999). Of the nine polymictic lakes studied, alum was effective in six lakes, with internal phosphorus loading rates reduced by an average of 66% and chl *a* concentrations reduced by an average of 40% after 5–11 years. Alum was believed to be ineffective in three of the lakes due to the influence of macrophytes, which may have caused uneven floc distribution or enhanced phosphorus release due to plant decay. Similarly, Huser et al. (2016b) undertook a review of 83 lakes that had been treated with aluminium salts and showed that shallow, polymictic lakes had an average treatment longevity (based on TP concentrations) of 4.6 years (range: 0 to 14 years), compared to an average treatment longevity for dimictic lakes of 15 years (range: 0 to 45 years). Other factors that influenced treatment longevity included: aluminium dose, the magnitude of external loads and the presence of moderate to high densities of benthivorous fish such as carp, which reduced treatment efficacy, likely due to bioturbation of benthic sediments. The negative effect of benthivorous fish was greatest in shallow lakes. Alum loading rates also tend to be lower in shallow lakes because the alkalinity available to buffer acid-producing reactions is limited due to lower lake volume.

The use of phosphorus adsorption materials as an effective method to control eutrophication has been widely advocated, particularly in the USA (Huser *et al.* 2016a; Wagner 2017; Osgood 2017). The use of such materials is advocated because they have been proven to be effective, reliable and cost-effective; for example, based on their study of four urban lakes in the USA (see Section 6.9), Huser et al. (2016a) concluded that in-lake alum treatment was on average 50 times more cost-effective than catchment-based measures to reduce storm water nutrient loads.

The addition of foreign materials to waterbodies is a culturally sensitive issue in New Zealand and this needs to be considered prior to any application (Hamilton and Dada 2016).

Table 5. Materials used to adsorb phosphorus in lakes (Douglas *et al.* 2016).

Group	Examples
Minerals, soils and sands	Carbonates, e.g., limestone Soil Sand
Natural or synthetically produced materials	Aluminosilicate minerals , e.g., allophane Fe- and Al-oxides and (oxy)hydroxides, e.g., goethite, gibbsite Hydrotalcites
Modified clay materials	Expanded/thermally treated clay aggregates Rare earth modified clays, zeolites and soils
Mining, mineral processing and industrial by-products	Red mud from metal refining Slags Neutralised used acid Coal fly ash

5.4 Biomanipulation

5.4.1 Fish

Manipulation of fish communities is a well-established lake restoration technique. Broadly, applications involve: 1) removal of planktivorous fish to reduce grazing pressure on zooplankton (cladocera), which feed on phytoplankton (top-down control), and/or; 2) removal of benthivorous fish to reduce internal nutrient loads via bioturbation (bottom-up control).

Control of planktivorous fish has been applied extensively in Europe where it has been demonstrated to be an effective restoration technique in shallow eutrophic lakes with abundant planktivorous fish such as roach (*Rutilus rutilus*) and rudd (e.g., Jeppesen *et al.* 1990). The objective of the technique is to reduce planktivorous fish biomass in turbid lakes sufficiently so that clear water conditions return and macrophytes re-establish. Ideally, external load reductions are undertaken simultaneously. The technique is therefore most applicable to shallow turbid lakes where the primary contributor to turbidity is algae (e.g., not inorganic sediment) and zooplankton biomass is maintained at low levels by fish predation. Planktivorous fish removal is ideally conducted in the winter (prior to the spring zooplankton bloom) and supplemented by stocking with native piscivorous fish (Perrow *et al.* 1997). Depite good results of planktivorous fish removal in the northern hemisphere (see case study in Section 6.3), the technique is generally less applicable in New Zealand lakes due to differences in lake food webs. A review of the applicability of plankivorous fish control to improve water quality in New Zealand lakes concluded that use of the technique was constrained by a lack of obligate piscivores, few planktivorous fish species, and a depauperate cladoceran zooplankton fauna (Burns 1998). This review was subsequently revisited by Burns et al. (2014) who concluded that, although the earlier constraints still remain, changes had since occurred that increased the suitability of the technique for a subset of lakes. Specifically, colonisation of some lakes by non-indigenous large-bodied cladoceran zooplankton (*Daphnia* spp.) has increased potential phytoplankton grazing rates, while colonisation by invasive fish species has altered interactions at higher trophic levels. The authors concluded that there is scope for sucessful use of biomanipulation of planktivorous fish in certain cases, following a thorough evaluation of a range of lake specific factors. The chances of success were considered highest in shallow lakes with maximum depth < 6 m to enhance the potential for macrophyte re-establishment. The presence of high dissolved organic matter (e.g., in peat lakes) was also identified as a desirable factor to provide food for protozoans. Notwithstanding the earlier constraints identified by Burns (1998), the changes that have since occurred (Burns *et al.* 2014) are particularly applicable to Waikato shallow lakes. Specifically, numerous shallow Waikato lakes (e.g., Waiwhakareke, Ruatuna, Waahi) have been colonised by non-indigenous *Daphnia* species that are highly efficient grazers of phytoplankton (Duggan 2017). Although the effects of these invasions are not well understood (Duggan 2017), maintaining a high abundance of these species has the potential to reduce phytoplankton biomass (notwithstanding potential concerns regarding adverse effects to ecological integrity due to proliferation of non-indigenous taxa). For example, colonisation of a deep mine pit lake in the Waikato (Lake Puketirini) by the large-bodied and efficient cladoceran filter feeder *D. dentifera* was associated with a stark improvement in water clarity, with Secchi depth increasing from ~2 m to > 6m over a two-year study (Balvert *et al.* 2009). The deep lake (maximum depth = 64 m) has unusual physical characteristics (meromictic), although the authors concluded that the improvement was caused by the *Daphnia* invasion and was unrelated to the physical features. Furthermore, Waikato shallow lakes have been widely colonised by invasive fish species (see Section 3.4), some of which are plankivores, e.g., juvenile perch (*Perca fluviatilis*). Therefore, control of these invasive fish species may reduce zooplankton grazing pressure in some lakes, supporting lake restoration objectives.

Control of benthivorous fish can reduce internal nutrient loads by reducing bioturbation of benthic sediments and direct nutrient excretion by fish. Mejer et al. (1990) describe an experiment in two shallow lakes in the Netherlands that demonstrated that removal of benthivorous fish can yield substantial improvements in water quality, at least in the short-term. Both lakes are divided into hydrologically separated basins. In each lake, 2000 kg of benthivorous fish (mainly common carp and bream *Abramis brama*) were removed from one basin, while the other basin was left as a control. Piscivorous fish were also stocked into one lake but this was unnecessary in the second as a high biomass (30 kg/ha) of pike (*Esox lucius*) remained. In the first lake (mean depth = 1.0 m, surface area = 0.14 km^2), fish biomass was reduced from 600–700 kg/ha to 120 kg/ha in the first year, although this increased to 350 kg/ha after two years. In the second lake (mean depth = 1.5 m, surface area = 0.2 km^2), fish biomass was reduced from 550–650 kg/ha to 145 kg/ha in the first year, although this increased to 300 kg/ha after one year. Water quality monitoring in the one or two years following the fish removal showed that chl *a* concentrations significantly decreased from 60–100 µg/L in the control basins to 5–20 µg/L in the experimental basins. Concentrations of TSS decreased by ~50% in the experimental basins; concentrations of nutrient fractions were also typically (but not consistently) lower in the experimental basins. In a similar study, Bajer and Sorensen (2015) describe the reduction of common carp from 300 kg/ha to 40 kg/ha in Lake Susan (maximum depth = 5.1 m, surface area = 0.35 km^2, USA). Monitoring over three years showed some dramatic improvements in water quality and aquatic health: macrophyte coverage increased from 5% to >45% and maximum TSS concentrations decreased by ~50%. However, reductions in chl *a* concentrations were subtle (small decline in spring concentrations only) and TP concentrations were unchanged, suggesting that internal loading was not reduced. The authors concluded that removal of carp can yield substantial improvements in water clarity; however, the contribution of carp to internal loading was minor.

As described in Section 3.4, colonisation of invasive fish is a pernicious issue facing Waikato waterbodies, with benthivorous fish such as brown bullhead catfish, tench and, most notably, common carp, now prevalent in many Waikato shallow lakes (Collier and Grainger 2015). Control of benthivorous fish has been identified as a potential lake restoration tool in the region, with a major research programme undertaken at the Lake Ecosystem Restoration NZ group at the University of Waikato to study invasive fish control. A particular focus of study has been Lake Ohinewai (maximum depth = 4.5 m, surface area = 16.8 ha), which is a shallow and eutrophic riverine lake north of Huntly (Map 1). A large-scale removal of invasive fish was undertaken at the lake during 2011–2016, as described by Tempero and Hicks (2017). The removal was high effort, involving boat electrofishing, fyke net deployment and installation of a one-way barrier at the outlet. This caused a large decline in fish biomass between 2011 and 2014, although biomass rebounded in 2016 (no sampling was undertaken in 2015). For example, the estimated biomass of koi carp (the dominant species, based on biomass) was 308 kg/ha in 2011, 14 kg/ha in 2014 and 94 kg/ha in 2016. Despite the large reductions in biomass, water quality monitoring generally showed no improvement in trophic status during a five-year period. Secchi depth, nutrients and TSS concentrations showed no improvement, while mean annual chl *a* concentration declined between 2011 and 2014, although this returned to pre-treatment levels in 2016. Although the authors acknowledge that the water quality monitoring programme had limitations, they conclude that “during the period of low carp abundance there was no convincing evidence for a corresponding improvement in water quality and zooplankton and phytoplankton community composition” (Tempero and Hicks 2017). This disappointing result is supported by a modelling study that showed that near-complete removal of koi carp from Lake Ohinewai could only yield a small (3%) reduction in TLI from 6.45 to 6.38, with substantial reductions in external loads instead required to achieve a reduction of ~1 TLI unit (Allan 2016b).

In summary, biomanipulation of fish communities is an established restoration method overseas, particularly in shallow European lakes that have high biomass of zooplanktivorous (e.g., roach) and benthivorous (e.g., bream) fish; zooplankton species that are efficient grazers, and; remnant macrophyte communities that can rapidly re-establish if clear water conditions are restored for at least for 1–2 years. In New Zealand, characteristics of lake food webs mean that biomanipulation of fish communities has limited potential to ameliorate eutrophication (Burns 1998; Burns *et al.* 2014), although the high biomass of invasive fish in many Waikato shallow lakes mean that control of these species has potential to improve water quality, predominantly via bottom-up processes. Despite this, the results of work at Lake Ohinewai indicate that invasive fish control requires high, persistent effort and can only make a minor contribution (at best) to restoration of shallow, hypertrophic lakes in the Waikato.

5.4.2 Bivalves

Mussels are effective filter feeders and enhancing mussel density has been identified as a potential method to reduce phytoplankton biomass in lakes overseas (Gulati *et al.* 2008). Zebra mussel (*Dreissena polymorpha*) has been identified as a species with potential to improve water quality in European lakes due to high clearance rates (Reeders and Bij de Vaate 1990). Jeppesen et al. (2012) describe the effects on water quality of colonisation by zebra mussels in Lake Faarup (surface area = 0.99 km^2; mean depth =5.6 m; Denmark). Mussels were first observed in 1993 and were recorded at a density of 1300/m^2 in 2000. From 1996, a major decline was observed in concentrations of TN, TP and chl *a*, in addition to an increase in Secchi depth. For example, annual peak summer chl *a* regularly exceeded 100 µg/L prior to 1996, but generally remained < 50 µg/L during 1998–2006. External nutrient loads from the predominantly agricultural catchment did not decline and therefore the improvement in water quality was attributed to the mussel invasion. Despite their potential as effective grazers, Jeppesen et al. (2012) note that full-scale mussel stocking experiments had not yet been undertaken. The potential for unforeseen adverse effects is an important consideration with introducing non-indigenous species; unintended zebra mussel introductions have resulted in well-documented adverse effects such as competition with native species and infrastructure fouling (Roberts 1990).

In New Zealand, biomanipulation of mussels has been considered as a potential lake restoration, with focus directed at native species. In this regard, the study of Lake Tuakitoto in Otago (surface area = 1.2 km^2; mean depth = 0.7 m) by Ogilvie and Mitchell (1995) is frequently cited, which showed that the native kākahi (*Echyridella menziesii*) population could filter the volume of the lake in 32 hours, causing chl *a* concentrations to be ~90% lower than would be predicted based on a standard chl *a*–TP regression equation. Phillips (2007) evaluated the potential for kākahi to suppress phytoplankton growth in the Te Arawa lakes, Rotorua. For the smallest shallowest lake that was considered (Lake Ngahewa; surface area = 0.1 km^2, mean depth = 3.5 m), the author concluded that a density of 46 mussels/m^2 would be required to cause a "continuous decline in phytoplankton abundance". This density was in the range that has been measured in other Te Arawa lakes (5–160/m^2), indicating that such densities are biologically feasible. The potential for biomanipulation was considered greatest in shallowest lakes where mussel populations could be augmented by seeding mussel beds, or installing caged mussels, potentially combined with improvements in habitat. However, the review identified numerous challenges, most significantly the limited abundance of host fish for larval development and the poor habitat conditions (e.g., low dissolved oxygen) present in eutrophic lakes that are higher priorities for restoration. In situ trials are therefore required before the viability of biomanipulation mussel densities to improve water quality in New Zealand lakes can be confirmed (Phillips 2007; Burns *et al.* 2014)

5.4.3 Macrophyte harvesting

Aquatic macrophyte harvesting is undertaken by lake managers for a variety of purposes; notably, to reduce competition with native flora from invasive macrophyte species (Boylen *et al.* 1996), and to reduce macrophyte density to benefit recreational users such as boaters (van Nes *et al.* 2002). In addition, macrophyte harvesting can potentially control eutrophication, reflecting the large stores of nutrients that can be present in plant tissues. For example, analysis of Eurasian milfoil (*Myriophyllum spicatum*) shoots in eutrophic Lake Wingra (surface area = 1.3 km^2, mean depth = 2.4 m, USA) showed that during peak biomass (late August), the nitrogen and phosphorus contents of the macrophyte tissues in the lake were equivalent to 16% and 37%, respectively, of the net annual external loads of these nutrients (Carpenter and Adams 1977).

Harvesting macrophytes can therefore directly remove nutrients from a lake; however, the potential for this action to subsequently reduce the trophic status of a lake is variable. In a follow-up study, Carpenter and Adams (1978) examined the potential role of Eurasian milfoil harvesting in a restoration strategy for Lake Wingra. Their analysis showed that repeated harvesting could remove 22% of the inorganic phosphorus present in the upper 12 cm of littoral sediments, equivalent to 1.4 g phosphorus/ m^2. However, the authors estimated that at least 10–15 years of harvesting would be required to achieve this because 0.5–1.0 g phosphorus/m^2 is replenished due to sedimentation. It was concluded that harvesting alone could not counteract the effects of high external loads from urban runoff, although harvesting could contribute to lake restoration as part of a larger strategy that included external load reductions. Peterson et al. (1974) demonstrated a more discouraging result for Lake Sallie (surface area = 4.9 km^2, mean depth = 5.6 m, USA), where a large-scale programme to harvest macrophytes was estimated to only remove 1.37% of the annual TP load to the lake.

Quillam et al. (2015) reviewed the topic, with specific consideration of whether macrophyte harvesting could “close the loop” in agricultural catchments by removing nutrients exported to eutrophic lakes from agricultural sources. The authors concluded that macrophyte harvesting could potentially offset agricultural nutrient loads to eutrophic lakes; however, based on costs, the technique has limited viability solely as a lake restoration option. Instead, the approach has promise providing that markets can be developed for the harvested plants, e.g., for biofuel, feedstock or compost, supported by further research.

In New Zealand, a macrophyte harvesting programme has now been undertaken for multiple years to remove prolific accumulations of invasive hornwort (*Ceratophyllum demersum*) from Lake Rotoehu (surface area = 7.9 km^2, mean depth = 8.0 m, Bay of Plenty) (Burns *et al.* 2009; Hamilton *et al.* 2016). This action is a key component of the action plan for the lake, which identifies removal of 1000 t of hornwort a year as a feasible way to remove 2,400 kg of nitrogen and 320 kg of phosphorus annually (BoPRC 2007). Costs of this action are considered to provide economic value at $22/kg of nitrogen removed and $165/kg of phosphorus removed (BoPRC 2007). The water quality of Lake Rotoehu has fluctuated since the hornwort removal programme began, with no clear indication of improvement (Te-Arawa Lakes Trust 2017). However, it is challenging to separate the response to the harvesting from the effects of climatic variability and other changes in the catchment.

5.4.4 Floating wetlands

The construction of floating wetlands has been proposed as a lake restoration technique and involves installing floating mats that contain emergent vegetation such as rushes (e.g., *Juncus* spp.) or sedges (e.g., *Schoenoplectus* spp.) (Pavlineri *et al.* 2017). Floating wetlands can primarily remove nutrients from lake water via uptake into plant tissues, although denitrification in root mats may also contribute to nitrogen removal (Zhang *et al.* 2014; Pavlineri *et al.* 2017). Increased settling of organic sediments beneath the wetlands may also contribute to nutrient removal as wetlands can reduce wave action (Sukias *et al.* 2010).

The use of constructed floating wetlands has yet to be established as a proven technique to restore lakes that have been degraded by eutrophication. Instead, the use of constructed floating wetlands is more established for treating storm water ponds (e.g., Tanner and Headley 2011) or domestic wastewater in developing countries (Zhang *et al.* 2014). For example, Pavlineri et al. (2017) reviewed published studies of the application of constructed floating wetlands, which yielded a database of 43 studies, only five of which related to lake applications. Of these, all studies were conducted in mesocosms (i.e., not in situ applications) with the exception of one study conducted over 16-days in a very small (0.6 ha) pond at a Chinese university (Zhao *et al.* 2012). However, the use of floating wetlands to restore eutrophic lakes has recently been an active area of research, particularly in New Zealand. Sukias et al. (2010) undertook trials to examine the potential for constructed floating wetlands to reduce nutrient concentrations in mesocosms with water chemistry consistent with three eutrophic Te Arawa lakes (Rotorua, Bay of Plenty). Mats were planted with three species of native emergent plants. Vegetation was allowed to become established prior to the trial, resulting in plants that were 0.5–0.8 m high, with roots extending ~0.5 m below the mats. Water quality sampling for a duration of 7-days demonstrated the following large reductions in dissolved nutrients: ammonium (82–99%), oxidised nitrogen (38–83%) and dissolved reactive phosphorus (57–84%). The authors concluded that the results of the study demonstrate that constructed floating wetlands can be used to manage lake eutrophication, although they acknowledged that the results were likely to be biased high as the trials were short-term and conducted in mid-summer. The authors also acknowledged that, unlike the mesocosms (fully covered by the wetlands), it is impractical to cover more than 10% of the surface area of a lake with floating wetlands. The authors recommended longer-term field studies; these have recently been conducted in some shallow Waikato lakes, including Lake Kaituna (WRC 2014b).

Once plants are established, vegetation harvesting is required to maintain performance and therefore this technique is largely an extension of macrophyte harvesting described above (Section 5.4.3). Relative to constructed riparian wetlands, floating wetlands are insensitive to changes in water level which makes them particularly suitable for shallow lakes with dynamic hydrology (Pavlineri *et al.* 2017). They may also provide additional habitat values (e.g., for wildfowl) and serve as visual focal points that help to engage communities in lake restoration efforts. The constraints with covering sufficiently large areas with wetlands mean that overall levels of nutrient removal in a full-scale application are expected to be modest (Hamilton and Dada 2016) and therefore, where appropriate, this technique is expected to be a supplementary tool applied in specific areas such as embayments (Sukias *et al.* 2010), rather than a standalone solution to restore a lake. A disadvantage is that large-scale applications may detract from aesthetic values for some lake users who enjoy views of open water (Hamilton and Dada 2016). Also, as described above, wetlands will impede wave action which could reduce sediment resuspension; however, it is plausible that this reduced mixing could also prolong stratification in polymictic lakes, potentially enhancing internal loading mediated by anoxia.

5.4.5 Algicides

Algicides have historically been used to directly control phytoplankton biomass in lakes (Cooke *et al.* 2005). In particular, copper sulphate ($CuSO_4$) has historically been widely used, although other algicides such as hydrogen peroxide (H_2O_2) and potassium permanganate ($KMnO_4$) have also been applied to lakes (Fan *et al.* 2013). However, despite these precedents, algicides have limited application as a lake restoration method for lakes where protecting and enhancing aquatic health is an objective due to the potential for toxic effects on non-target organisms such as zooplankton, molluscs and fish, in addition to causing contamination of bed sediments (Cooke *et al.* 2005; Stroom and Kardinaal 2016). Low dose applications can potentially limit such effects (Stroom and Kardinaal 2016), although applications of copper sulphate at low doses have been shown to be ineffective for controlling phytoplankton (Effler *et al.* 1980). More broadly, the use of algicides can be expensive and only provide a short-term solution. Applying chemicals to waterbodies is also particularly culturally sensitive in New Zealand.

Based on the above, algicides are considered to have limited applicability as a restoration method for Waikato shallow lakes, although a possible exception could be to manage biosecurity emergencies (de Winton *et al.* 2013).

5.4.6 Macrophyte re-establishment

Rooted macrophytes are an important component of healthy shallow lake ecosystems that serve to maintain desirable clear water conditions. Following a shift to a turbid phytoplankton-dominated state, it can be challenging to promote macrophyte re-establishment, despite subsequent improvement to water clarity. Two key challenges are lack of viable propagules (e.g., because the historical seed bank has been buried in new sediment) and grazing by wildfowl or herbivorous fish (Bakker *et al.* 2013). To address this, macrophyte re-establishment can be promoted by planting and/or protection of macrophytes in enclosures. This can allow strategically located 'founder colonies' to become established that may then spread through the lake (Smart and Dick 1999).

In a review of options to re-establish macrophytes in Te Waihora (surface area = 145 km^2; mean depth = 3.0 m; Canterbury), Jellyman *et al*. (2009) identified that a combination of founder colony plantings, use of grazing exclosures and addition of wave baffles would have a moderate to high chance of success. The authors suggested planting robust propagules of whole plants at a density of 0.4–0.8 plants/m^2 in sheltered coves, protected by mesh enclosures and wave baffles such as logs or floating islands.

Techniques to re-establish macrophytes have been used in some Waikato lakes; for example, Jellyman *et al*. (2009) reviewed the historical use of exclosures in Lake Rotoroa and Lake Whangape and noted that macrophytes persisted in cages for some time while other areas were devegetated. These results are supported by experimental results from elsewhere that have demonstrated preferential growth of macrophytes in exclosures designed to exclude fish (e.g., Evelsizer and Turner 2006).

Despite these applications, a key point is that this technique is only suitable for lakes that have already been subject to successful restoration actions that have increased clarity sufficiently to create suitable light conditions for macrophytes to re-colonise (Bakker *et al.* 2013). Therefore, this technique must be used in conjunction with other measures and it is not suitable for lakes that remain in a turbid phytoplankton-dominated state.

5.5 Hydrologic manipulations

Lake inflows and outflows can be manipulated to achieve hydrologic conditions that support restoration objectives. The technique greatly depends on individual lake characteristics but it can be highly successful in certain circumstances.

Diversion of lake inflows can substantially reduce external loads. For example, diversion of a stream (mean flow = 0.18 m^3/s) to Lake Tutira (surface area = 1.7 km^2; mean depth = 21 m; Hawke's Bay) was estimated to reduce the TP load to the lake by 76% (McColl 1978). At a much larger scale, the Ōhau Channel wall was constructed in 2008 and diverts water that previously flowed from eutrophic Lake Rotorua (surface area = 80 km^2; mean depth = 11 m) into Lake Rotoiti (surface area = 34 km^2; mean depth = 33 m) in the Bay of Plenty. The wall has been highly successful, reducing the three-year mean TLI from 4.45 prior to the diversion to 3.79 afterwards (Hamilton and Dada 2016). Inflow diversion is clearly only applicable in specific circumstances and it is important to consider effects on receiving waterbodies, in addition to potential hydrological effects on the lake, e.g., changes to water residence time.

Alternatively, lake inflows can be augmented to improve water quality by flushing a lake with relatively high-quality water. The technique has been applied at a large scale to multiple lakes, particularly in the People's Republic of China (Li *et al.* 2011; Liu *et al.* 2014; Jin *et al.* 2015) For example, 4.6 m^3/s of water is treated with alum and diverted from the Qiantang River into West Lake (surface area = 6.5 km^2; mean depth = 2.3 m; Zhejiang), resulting in a complete exchange of water in the lake every month and improved water quality (Jin *et al.* 2015). Temporarily allowing intrusion of marine water into coastal lakes can be effective for improving water quality in shallow barrier lakes that are naturally connected to the sea periodically. Analysis of data for 1983–2007 showed that periodic managed openings at Waituna Lagoon (surface area = 7.2 km^2; mean depth = 1.6 m; Southland) resulted in stark improvements in water quality, with rapid reductions in chl *a* concentrations (Schallenberg *et al.* 2010). By contrast, the study showed that managed openings of Te Waihora (surface area = 145 km^2; mean depth = 3.0 m; Canterbury) had only muted effects, likely because phytoplankton remained light-limited due to high turbidity in the lake. This contrast highlights the importance of baseline studies for informing lake restoration plans and lake ecosystem modelling can be particularly useful for predicting the effects of managing inflows. Hydrodynamic-ecological modelling showed that without the managed openings, algal biomass in Waituna Lagoon would be over twice as high (453 g C/m^2 compared with 192 g C/m^2) and therefore the artificial openings somewhat offset the influence of high external nutrient loads. Similarly, a modelling study of Wainono Lagoon (surface area = 3.7 km^2; maximum depth = 1.8 m; Canterbury) estimated that diversion of 1 m^3/s of high quality water from the Waitaki River would decrease TLI in the lake from 6.34 to 5.96 (Abell *et al.* 2014).

In shallow lakes, water level is a "master variable" that can have a dominant effect on ecological functioning, meaning that management operations that affect water level fluctuations can have additional substantial effects (Gulati *et al.* 2008). Restoration of natural water level regimes has been identified as an important shallow lake restoration tool in the Netherlands where, like the Waikato, water levels have historically been extensively managed in shallow lakes situated in agricultural regions, resulting in reduced fluctuations (Gulati and van Donk 2002; Coops and Hosper 2002; Coops *et al.* 2003) Specifically, restoring water level fluctuations has been identified as a tool to restore turbid eutrophic lakes by promoting the growth of emergent vegetation that favours variable water levels (e.g., *Phramites* spp.) and reducing erosion due to a less confined swash zone (Coops and Hosper 2002). In addition to increasing the magnitude of water level fluctuations, increasing the lake depth can also provide benefits. Increased depth will reduce sediment resuspension due to reduced benthic shear stress (Hamilton and Mitchell 1997) and it may support wetland restoration in some circumstances; for example, increased lake levels have been proposed at Lake Mangakaware (surface area = 0.13 km^2; maximum depth = 4.8 m; Waikato) to aid restoration of riparian wetlands (Waikato Regional Council 2014a). Conversely, lake drawdown has been used as a tool to increase the abundance and diversity of submerged macrophytes by increasing the area of the lake bed within the euphotic zone and stimulating germination (Hanson *et al.* 2016). WRC has a lake level setting programme and minimum lake levels have been identified for 15 lakes in the region, with multiple further lakes identified for the process (Waikato Regional Council 2014b). Land tenure can be a constraint to raising water levels.

5.6 Emerging technologies

The techniques described above encompass the range of established lake restoration methods to manage the symptoms of eutrophication that are described in peer-reviewed literature (e.g., Welch and Cooke 1995; Cooke *et al.* 2005; Søndergaard *et al.* 2007; Jeppesen *et al.* 2009a). However, given the size of the problem posed by eutrophication, it is unsurprising that new technologies are being developed to provide additional tools that may prove useful. Some of these technologies are supported in the 'grey' literature, although their successful use has yet to be robustly demonstrated and generally accepted by lake managers. Examples of such technologies include nanobubbles (Brooks 2012), ultrasound (Hicks and Bryant 2002), and new phosphorus adsorption products. When considering the use of such technologies, it is appropriate that careful assessment is made of potential ecological risks and that robust field trials are undertaken before any full-scale applications.

To support this, Spears et al. (2014) provide the six recommendations below for best practice in applying new 'geo-engineering' methods. Although they were developed to specifically apply to chemical products to manage internal loads, their tenor applies to evaluating other emerging technologies:

1. *"Protocols should be developed combining traditional long-term data with high frequency monitoring programmes to provide the process-based understanding necessary to assess the candidacy of lakes for this approach*
2. *Evidence of responses following treatment should be combined across case studies to provide a meta-analysis against which the generality of responses in individual case studies can be assessed.*
3. *An assessment of all reported target and nontarget effects associated with all available products should be conducted.*
4. *Decision support systems should be designed to assist material selection across a range of receiving water types.*
5. *Standard protocols for dose determination should be developed based on published literature and knowledge of the likely behaviour of the amendment material in the receiving water.*
6. *The application of materials should be assessed in the context of "dose−response" (i.e., repeated smaller doses) as opposed to the common "single pill" (i.e., single large dose) approach to minimize the risk of unintended consequences."*

It is also appropriate that proponents of proprietary products adopt or share the financial risks with applying new technologies that have not been fully field tested.

6 Case studies

6.1 Overview

This section presents key case studies that describe the outcomes of restoration in shallow lakes. All case studies are based on peer-reviewed articles in established scientific journals. The review focuses on examples involving long-term (>10 years) monitoring, although some case studies are presented that involve shorter monitoring periods in cases involving a robust programme to sample multiple variables over a shorter period. A summary of case studies is presented in Table 6.

Table 6. Lake restoration case studies.

Lake	Region	Area (km^2)	Mean depth (m)	Max. depth (m)	Watershed land use	Restoration action	Restoration period	Monitoring period	Restoration goal	Variables monitored	Restoration outcome	Reference
35 lakes (21 shallow)	Europe (n=32), USA (n=3)	0.0003–3 55,500	0.7–177	1.9–374	Variable	External load reduction (P reduction with/without N reduction)	Generally 1970s-1980s	5–35 years; mean duration: 16 years	Variable	Nutrients Chl *a* Secchi depth Phyto/zoo plankton Macrophyes Fish	◦ Annual mean in-lake [TP] ↓ in 86% of shallow lakes • Summer [TN] ↓ in 83% of shallow lakes • In-lake [TP] typically ↓ after 10–15 years, while in-lake [TN] ↓ after <5 years • [Chl *a*] ↓ in 71% of the shallow lakes • Secchi depth ↑ in 77% of shallow lakes	Jeppesen et al. (2005)
36 lakes	Denmark	0.02–8.6	0.8–4.3	1.1–10	UNK	Biomanipulation (removal of zooplanktivorous and benthivorous fish)	Generally 2–4 year periods in the 1980s and 1990s	Up to 20 years	Not stated	Nutrients Chl *a* Secchi depth TSS Phyto/zoo plankton Macrophyes (11 lakes) Fish	◦Limited and short-lived improvements when fish removal <200 kg/ha (9 lakes) ◦Substantial improvements when fish removal >200 kg/ha (26 lakes): nutrient concentrations ↓ by ~50%; chl *a* and TSS concentrations↓ by 50–70%; Secchi depth approximately doubled ◦The maximum depth of macrophytes↑ in nine lakes but did not change in two lakes	Søndergaard et al. (2008)
Barton Broad	Norfolk, England	0.72	1.4	≤2	Agricultural	Point source control Dredging	1977–1980: point source control 1997–2000: dredging	1977–2001	Not stated	Nutrients Secchi depth Chl *a* Phyto/zoo plankton	◦External phosphorus load ↓ from ~11 g P/m^2/d to ~4 g P/m^2/d ◦Large declines in annual [TP] occurred following load reductions although high internal loads persisted for ~15 years ◦Water quality improvements occurred more slowly in the summer than the spring due to reduced flushing and higher internal loading ◦Spring: median Secchi depth ↑ by ~100% after 15 years to ~0.8 m. Summer: median Secchi depth only ↑ from ~0.35 m to ~0.40 m after 15 years. Spring: [chl *a*] concentrations ↓ by ~50% after 5-years. Summer significant ↓ in [chl *a*] took 15-years. ◦Cyanobacteria abundance ↓ markedly after 15 years	Lau & Lane (2002); Geoff et al. (2005)
City Park Lake	Baton Rouge, Louisiana, USA	0.23	1.2	UNK	Urban (>80%)	Dredging Point source control	1983	1979–1984, 1990, 2000–2001	Not stated	Nutrients Chl *a* Secchi depth	Rapid ↑ in water clarity and ↓ frequency of algal blooms	Ruley & Rusch (2002)
Lake Apopka	Florida, USA	125	1.6	UNK	Agriculture since 1940s	External load reduction (convert farmland to wetlands)	Since 1993	1987–2004	Reduce external P load by 76% to 0.13 g P/m^2/year. Intended to achieve in-lake [TP] = 55 µg/L.	Nutrients Chl *a* TSS Secchi depth	◦In-lake [TP] started to ↓ within 2-years of load reductions ◦[Chl *a*] and [TSS] started to ↓ within ~4 years ◦After 10-11 years, [TP] and [Chl a] had ↓ by ~40-50% ◦Physico-chemical characteristics meant that internal loading was relatively low, which enhanced the rate of recovery	Coveney et al. (2005)
Lake Finjasjön	Sweden	11	3	13	Predominantly forested, with several towns and 10-15% agriculture	Point source control (1977) Dredging (1987-1992) Cyprinid reduction (1992-1994) Wetland creation (1990s)	1997-1999	1950-1995	Swimmable	Nutrients Phyto/zoo plankton	◦Dredging either failed or yielded a slow response due to insufficient removal of sediments ◦High effort cyprinid removal increased *Daphnia* spp. abundance. Phytoplankton biomass declined by ~50% and Secchi depth ~doubled in the subsequent two-years	Annadotter et al. (1999)
Lake Kraenepoel	Belgium	0.22	1	1.5	Agriculture Aquaculture (pre-1950s)	Dredging Biomanipulation (cyprinid removal) Inflow diversion	1999	1999–2001	Not stated	Nutrients Phyto/zoo plankton Macrophytes	◦Stark improvement in dredged basin: ↓ nutrients, e.g. mean [NO_3-N] and [SRP] ↓ 16-fold ↓ phytoplankton biomass and cyanobacteria ↑ macrophyte diversity and cover ◦Negligible improvement in un-dredged basin	Van Wichelen et al. (2007)

Table 6. Continued.

Lake	Region	Area (km^2)	Mean depth (m)	Max. depth (m)	Watershed land use	Restoration action	Restoration period	Monitoring period	Restoration goal	Variables monitored	Restoration outcome	Reference
Lake of the Isles	Minnesota, USA	44.2	2.7	UNK	Urban	Storm water treatment Alum treatment	1995–1997	1998–2014	Mean surface [TP] ≤ 40 µg/L	Nutrients Chl *a* Secchi depth	◦Epilimnetic [TP] ↓ from a 5-year pre-treatment average of 60 µg/L to 35 µg/L post-treatment (~40% reduction) ◦[Chl *a*] ↓ from a 5-year pre-treatment average of 39 µg/L to 17 µg/L post-treatment (~55% reduction)	Huser et al. (2016)
Lake Rotorua	Bay of Plenty, NZ	81	10.8 (poly-mictic)	53	Pastoral and urban	Point and diffuse source control Alum dosing to lake inflows Wastewater reticulation Constructed floating wetland	1991 (waste water diversion) Since 1990s - implement BMPs 2004 - 'Rule 11' enforcing land use controls Since 2006 - alum dosing in inflows	Since 2001 (intermittent prior to this)	TLI = 4.2	Nutrients Secchi depth Chl *a* Phytoplankton	◦Improved water quality since mid 2000s ◦TLI target achieved in 2012 ◦High inter-annual variability associated with climatic variability ◦Alum dosing has been demonstrated to be effective but is not expected to provide a long-term solution ◦Internal loading due to historic wastewater discharge remained high for at least ~15 years following diversion ◦Groundwater lags and P enriched soils cause inertia	Smith et al. (2016)
Lake Susan	Minnesota, USA	0.35	UNK	5.1	UNK	Biomanipulation (removal of common carp)	2008	2009–2011	Not stated	Nutrients Chl *a* TSS Secchi depth Macrophyte coverage	◦Macrophyte coverage ↑ from 5% to >45% ◦Maximum TSS concentrations ↓ from ~20 mg/L to 10 mg/L ◦Only subtle ↓ in chl *a* concentration ◦No change in TP concentrations	Bajer & Sorensen (2015)
Loch Leven	Scotland	13.3	3.9	25.5	Mainly agricultural (80%)	Point source control (primary)	Mainly early 1980s and 1990s Diffuse pollution measures have subsequently been implemented (buffer strips)	1968–2007	**1990s** - designed to support rooted macrophytes to a depth of 4.5 m: ◦mean annual [TP] = 40 µg/L ◦mean annual [chl *a*] = 15 µg/L ◦mean annual Secchi depth = 2.5 m **2000s** - based on EU Water Framework Directive: ◦mean annual [chl a] < 11 µg/L ◦geometric mean annual [TP] < 32 µg/L	Nutrients Secchi depth Chl *a* Phyto/zoo plankton	◦TP loads reduced by 60% ◦[TP] ↓ from ~90 µg/L in the 1970 tp <40 µg/L in the mid 2000s ◦[Chl *a*] ↓ and Secchi depth ↑ but targets not achieved	Carvalho et al. (2012) D'Arcy et al. (2006)

6.2 35 lakes: Reductions in external nutrient loads

Jeppesen et al. (2005) evaluated the effects of external nutrient load reductions by analysing monitoring data (duration: 5–35 years; mean duration: 16 years) for 35 northern hemisphere lakes that had been subjected to reduced external phosphorus loads, with additional measures to reduce external nitrogen loads in some cases. Measures to reduce external loads typically began in the 1970–1980s and comprised the sole lake restoration measure undertaken, with the exception of three lakes where fish removal had been subsequently undertaken. Twenty-one lakes are considered 'shallow' as they do not exhibit stable summer stratification. This comprehensive study involved 30 authors and considered lakes located in the USA (n = 3) and throughout Europe (n = 32), including numerous case studies that are prominent in the limnology literature, e.g., Lake Müggelsee (surface area: 730 ha, mean depth: 4.9 m; Germany), Lake Peipsi (surface area: 355,500 ha, mean depth: 7.1 m; Estonia/Russia), Lake Okeechobee (surface area: 173,000 ha, mean depth: 2.7 m; USA) and Loch Leven (surface area: 133 ha, mean depth: 3.9 m; Scotland).

The authors analysed changes in nutrient concentrations and Secchi depth in response to reduced external nutrient loads for all lakes. Changes in plankton, macrophyte and fish communities were also analysed for most lakes, depending on data availability. Key conclusions were:

- Annual mean in-lake TP concentrations declined in 86% of shallow lakes and all deep lakes;
- Summer TN concentrations declined in 83% of shallow lakes;
- For both deep and shallow lakes, in-lake TP concentrations typically showed a decline after 10–15 years, while in-lake TN concentrations showed a decline after <5 years. This difference in response was likely due to inertia caused by internal phosphorus loads, whereas denitrification minimised internal nitrogen loads;
- Chl *a* concentrations decreased in 71% of the shallow lakes;
- Secchi depth increased in 77% of shallow lakes;
- TN:TP typically increased, reflecting proportionally greater reductions of phosphorus compared with N;
- Phytoplankton community composition typically changed in shallow lakes, although a clear decline in cyanobacteria abundance was not observed;
- Zooplankton biomass decreased in shallow lakes;
- No clear pattern was observed in macrophyte response.

6.3 36 Danish lakes: Biomanipulation of fish communities

Søndergaard et al. (2008) reviewed the effects of lake restoration by fish removal in 36 Danish lakes that were shallow (mean depth: 0.8–4.3 m) and eutrophic–hypertrophic (mean chl *a* concentration = 26–208 µg/L). A total of 41–1360 kg fish/ha was removed from each lake during a 1–19-year period, although fish removal occurred during a 2–4 year period in most lakes. Planktivorous (roach) and benthivorous (bream) were removed from all lakes. Mean values of water quality measurements were compared between pre-treatment (typically 4–5 years) and post-treatment (up to 20 years) periods, with corrections made to account for external loading reductions that occurred coincidentally. Their results showed that fish removal generally produced substantial improvements in water quality, although effects were most pronounced when > 200 kg/ha of fish were removed over a relatively short period (~3 years). When fish removal exceeded this threshold (27 lakes), nutrient concentrations were reduced by ~50%, while chl *a* and TSS concentrations were reduced by 50–70% of the pre-treatment levels. Secchi depth approximately doubled indicating an improvement in water clarity. The maximum depth of macrophytes increased in nine lakes but did not change in two lakes, while the coverage of macrophytes increased in seven lakes but did not change in in three lakes. Effects were generally apparent for 2–6 years after the treatment, with return to pre-treatment conditions observed in some lakes after longer periods. Overall, Secchi depth and TSS concentrations were the variables that showed the greatest and most long-lasting improvement, suggesting that reduced sediment resuspension by benthivorous fish was an important and persistent effect. The authors caution that their results may "be restricted to shallow, nutrient rich Northern temperate lakes".

6.4 Lake Apopka, USA: reduction of external phosphorus load from agriculture

Lake Apopka (surface area: 12,500 ha; mean depth 1.6 m; Florida, USA) is a hypereutrophic lake that experienced a seven-fold increase in external phosphorus loading from 1940 to the late 1980s, as described by Coveney et al. (2005). The increase was primarily due to agricultural intensification on drained floodplain marshes, which resulted in mineralisation of peat soils and fertiliser application. The eutrophication of the lake was associated with a collapse of macrophyte communities in the late 1940s and a switch to a phytoplankton-dominated state with cyanobacteria dominant.

A major lake restoration programme commenced from around the early 1990s. The major component of this was retiring farms in the floodplain and restoring wetlands. Farms were purchased in two phases: 1988–1992 and 1996–1998. Between these periods, changes were also made to farming practices to reduce external loads, notably storm water management and reduced fertiliser use. The goal of the programme was to reduce external phosphorus loads to 0.13 g P /m^2 /year, which represented a 76% reduction from the mean load for 1968–92. This load reduction target was based on a model and was intended to meet an in-lake TP concentration target of 0.055 mg/L.

External TP loads declined from the start of the programme, although there was considerable variability among years due to hydrologic fluctuations that caused variability in phosphorus loads in surface runoff. Prior to restoration, the external phosphorus load was 0.56 g P/m^2 /year; this fell to 0.38 g P /m^2/year (-32%) 1–3 years after the start of the programme and 0.12 g P /m^2 /year (-79%) 7–9 years after the start of the programme. The lake was sampled at least bi-weekly at 3–10 sites from 1987 to 2004. In-lake TP concentrations began to decline within two years of the reduction in loading. Chl *a* and TSS concentrations also declined, although the response of these variables was slower. By 2003–2004 (10–11 years post restoration), mean in-lake TP concentration had declined by 54%, chl *a* concentration by 37%, TSS concentration by 35% and Secchi depth increased by 47%.

The reduction of in-lake TP concentrations in response to reduced external loading occurred atypically fast (~1–2 years) in Lake Apopka, e.g., see Section 6.2. The authors attribute this response to physicochemical characteristics; notably: high proportion of SRP (~80%) in historic phosphorus loads from farmland, which minimised sediment enrichment; low iron content of the sediments, which minimised phosphorus retention, and; short duration of stratified periods (< 1 days), which minimised the occurrence of anoxia and associated internal loading.

6.5 Barton Broad, England: External load reduction and dredging

Barton Broad (surface area: 72 ha, mean depth: 1.4 m) is situated in the Norfolk Broads, which is a low elevation region in eastern England that is characterised by a network of wetlands and shallow lakes (‘broads’) set within pastoral and arable farmland. Eutrophication has been pervasive throughout the region since the 1950s (Lau and Lane 2002). Phillips et al. (2005) evaluate the performance of a restoration programme that commenced in 1977, based on analysis of a 24-year dataset (1977–2001). Point source control was undertaken during 1977–1980, involving treatment to remove phosphorus from upstream waste water discharges and a reduction in external phosphorus load from ~11 g P/m^2/d to ~4 g P/m^2/d. Recovery was then monitored for 17 years prior to initiating a dredging programme in 1997–2000 to reduce internal loading and increase lake depth.

Annual mean TP concentration declined in response to external load reduction, declining from ~0.3 mg/L to ~0.06–0.2 mg/L. Comparison of in-lake nutrient concentrations with concentrations in the single major inflow showed that it took ~15 years for the bed sediments to equilibrate to the reduced external loads. Dredging accelerated reductions in internal loading. Concentrations of chl *a* declined following the external load reduction, although there were marked seasonal differences in this response: spring concentrations declined by ~50% after 5-years and by ~80% after 15-years, while summer concentrations responded more slowly, with reductions (~50%) only apparent after 15-years. This difference was attributed to lower flushing rates and higher release rates of SRP from benthic sediments in summer, which supported accumulation of SRP from internal sources. Secchi depth in the spring approximately doubled but improvements in summer water clarity were more muted. Improved water quality was associated with a change in phytoplankton community composition, with a shift from centric to pennate diatoms and a marked decline in the relative abundance of cyanobacteria. Despite the improvement in water quality, macrophytes failed to re-establish after 20 years, possibly because the improvements in summer water clarity were only minor. The biomass of cladoceran zooplankton consistently declined through the restoration period; the authors contrast this with restoration projects elsewhere that have shown the opposite trend and they speculate that this may reflect the absence of macrophyte refuges and continued high abundance of planktivorous fish.

6.6 City Park Lake, USA: dredging and point source reduction

City Park Lake (surface area = 0.23 km^2, mean depth = 1.2 m, USA) was subject to dredging and point source control (repair of sewerage infrastructure) in 1983, as described by Ruley and Rusch (2002). Approximately 100,000 m^3 of sediments were dredged, increasing mean depth by ~0.3 m. The restoration yielded rapid improvements in water clarity, and the frequency of algal blooms and fish kills was greatly reduced for the remainder of the 1980s. Lake water subsequently declined again during the 1990s; however, this was primarily attributed to changes in hydrology.

6.7 Lake Finjasjön, Sweden: external load reduction, dredging and biomanipulation

Lake Finjasjön (surface area = 11 km^2, mean depth = 3.0 m, Sweden) was subject to multiple lake restoration methods to address eutrophication caused by discharge of sewage from the early 1900s, as described by Annadotter et al. (1999). In 1977, upgrades to wastewater treatment substantially reduced external phosphorus loads to the lake from ~30–40 t P/y in the early 1970s to ~5 t P/y. Despite this, water quality continued to decline, with mean summer chl *a* concentrations of ~60 µg/L in the early 1980s, compared with ~20 µg/L in the early 1970s. In response, dredging was undertaken in 1987–1992 across an area of 6.6 km^2 to remove 0.5 m of surficial sediments that had higher TP concentrations than the remaining 2-m-deep sediment layer. Despite a major cost (~GBP5 million), the dredging programme was deemed unsuccessful. This is because it failed to remove the remaining layer of sediment that was nutrient-enriched from legacy sewage inputs and continued to support high internal loading. During the late 1980s and early 1990s, the lake suffered major blooms of toxin-producing cyanobacteria, resulting in swimming being banned and incidents of liver damage in livestock. In response to ongoing poor water quality, a biomanipulation programme was initiated in 1992. The programme involved removing cyprinids using pelagic trawl nets deployed from one or two boats until 1994 when fish catches had declined to 20% of initial catches and a ratio of 1 planktivore to 1 piscivore was observed. External loads were also reduced by implementing additional wastewater treatment, designating 5-m-wide riparian buffers and constructing a 30 ha wetland. Plankton sampling (weekly or monthly) was undertaken in 1992–1995 to evaluate the success of the biomanipulation programme. This showed that the abundance of *Daphnia* spp. increased substantially accompanied by a reduction in mean chl *a* concentrations from 60–160 µg/L in 1900–1993 to ~40 µg/L in 1994–1995 and an approximate doubling of Secchi depth. The longevity of these improvements is uncertain. A nutrient budget for 1995 indicated that net internal loading had declined to near zero, suggesting a possible delayed effect of the dredging, which ceased three years earlier.

6.8 Lake Kraenepoel, Belgium: dredging, external load reduction and biomanipulation

Lake Kraenepoel (surface area = 0.22 km^2, mean depth = 1.0 m; Belgium) was subject to dredging, external load reduction (diversion of a small stream), and biomanipulation in 1999, as described by Van Wichelen et al. (2007). Biomanipulation involved draining the lake, removing all fish (predominantly benthivorous cyprinids) and then re-stocking with juvenile pike. Dredging was only undertaken in the northern basin of the lake, which is separated from the remainder of the lake by a dike. Dredging removed 24,600 m^3 of sediments. Water chemistry, zooplankton, phytoplankton and macrophytes were monitored for one year prior to the restoration and two years afterwards.

Post-restoration monitoring showed stark improvements in the northern basin (dredged) but not the southern basin (un-dredged). Mean dissolved nutrient concentrations in the northern basin declined five-fold for ammonium and 16-fold for nitrate and phosphate following the restoration. Phytoplankton biomass declined in both basins but to a greater extent in the northern basin where large cyanobacteria almost completely disappeared and cryptophytes became more dominant. The estimated grazing pressure of macro-zooplankton on phytoplankton increased significantly in the northern basin (but not the southern basin) after the restoration measures. The macrophyte community changed starkly in the northern basin as canopy-forming *Potamogeton* species were replaced by meadow-forming plants (e.g., *Littorelletea* spp.) that extended over a larger area but a similar or lower proportion of the lake volume. Several macrophyte species returned to the northern basin that had been absent for multiple years. By contrast, no macrophytes colonised the southern basin following restoration, with the exception of an expansion of filamentous green algae (e.g., *Ulothrix* spp.). Thus, this case study demonstrates short-term success following major lake restoration actions, although the longer-term performance is uncertain.

6.9 Loch Leven, Scotland: External load reductions

Loch Leven (surface area = 13.3 km^2, mean depth = 3.9 m, Scotland) is the largest shallow lake in Great Britain and provides a valuable case study of the long-term response to external load reductions, as described by Carvalho et al. (2012)[4]. Sampling has been undertaken every 1–2 weeks at the lake since 1968. The external load of TP was reduced from 20 t P/y in 1985 to 8 t P/y in 1995 and has since remained relatively consistent. The current load is approximately twice the estimated reference load for 1850 (D'Arcy *et al.* 2006). Nutrient load reductions were primarily achieved by:

- reductions in effluent from a woollen mill between the 1960s and 1989, and;
- tertiary treatment and effluent diversion works to reduce nutrient loads from treated municipal wastewater in the 1990s.

Buffer strips were also installed after 1995 to manage diffuse loads from agricultural sources but the effects of these are uncertain.

In the early 1990s, the following mean annual water quality targets were established to support the growth of rooted macrophytes to a depth of 4.5 m: TP concentrations < 40 µg/L, chl *a* concentrations < 15 µg/L, Secchi depth > 2.5 m. More stringent targets were set in the 2000s based on the EU Water Framework Directive with the aim to achieve good or moderate ecological status. These included annual mean chl *a* concentrations of ≤11 µg/L.

Monitoring data show an abrupt improvement in water quality in the early 1970s although, interestingly, this occurred prior to major load reductions and coincided with a return of populations of *Daphnia* spp. after being absent for 15–20 years. Since then, water quality has further improved, although trends have been non-linear, with substantial inter-annual variability, likely associated with climatic variability. In the mid-2000s, TP concentrations first achieved the 40 µg/L target, following a decline from ~90 µg/L in the 1970s. The Secchi depth and chl *a* concentration targets were not achieved in the 40-year monitoring period. The analysis showed strong climatic effects with the lowest chl *a* concentrations in wet years (presumably due to increased flushing rate) and a strong dependency of zooplankton grazing rate on warm spring temperatures.

[4] This paper is part of a special issue of *Hydrobiologia* that focuses on Loch Leven research and includes several other relevant papers that are cited in the overview provided by Carvalho et al. (2012).

6.10 Minneapolis Chain of Lakes, USA: storm water treatment and alum dosing

Four interconnected lakes (surface area: 44–139 ha, mean depth: 2.7–11 m; USA) were subject to external load reductions and alum treatment in 1995–1997, as described by Huser et al. (2016a). Bi-weekly monitoring has been undertaken since 1991 and the lakes provide a case study to examine the relative efficacy of external load reductions in comparison to in-lake measures to reduce internal loading. Further, one of the lakes is polymictic (Lake of the Isles, surface area = 44 ha, mean depth = 2.7 m) while the others are fully or partially dimictic; therefore, the case study also provides a comparison of responses between lakes with alternate mixing regimes.

At Lake of the Isles, in-pipe storm water treatments (continuous deflective separation units and grit chambers) were installed, followed by a whole lake alum treatment. Varying degrees of external load reductions and alum treatments were undertaken at the three deeper lakes. Monitoring confirmed that five-year average concentrations of chl *a* and TP were significantly reduced following the treatment: epilimnetic TP concentration declined from a pre-treatment average of 60 µg/L to 35 µg/L post-treatment (~40% reduction), while average chl *a* concentration declined from 39 µg/L to 17 µg/L post-treatment (~55% reduction). Similarly, average Secchi depth increased from 1.3 m to 2.1 m. The alum treatment was assessed to have provided an improvement for a period of approximately four years. The in-pipe storm water treatments were shown to have had low phosphorus removal efficiency of ~1%. The estimated average cost of phosphorus reduction was $27/kg of phosphorus for alum treatment compared with $1,368/kg of phosphorus for the storm water treatment. Based on this, the authors concluded that in-lake treatments are by far the most cost-effective means of improving water quality, with the caveat that external load reductions increase the longevity of alum treatments. The longevity of the alum treatment was lower in Lake of the Isles than the deeper lakes.

6.11 Lake Rotorua, Bay of Plenty, New Zealand: Reductions in external loads and alum dosing

Lake Rotorua is large (81 km^2), relatively shallow (mean depth = 11 m) and polymictic. The lake is eutrophic, with a TLI target of 4.2, which is based on estimated 1960s water quality (BoPRC 2009). The lake TLI is currently above the target but lake water quality has significantly improved since the early 2000s. The lake has been subject to numerous lake restoration methods and arguably represents the best-documented shallow lake restoration case study in New Zealand. Key insights from this case study include:

- Historic discharge of municipal wastewater to the lake from 1973–1991 exacerbated eutrophication and was a major cause of high internal loading. Wastewater was subsequently discharged to a land treatment system. Internal loading likely peaked in the mid-2000s when internal loads were estimated to exceed external loads on an annual basis (Burger *et al.* 2007);
- High groundwater transit times cause great inertia between changes to land management and lake water quality responses, largely due to lags in nitrate transport (Morgenstern *et al.* 2015);

- Inter-annual climatic variability can have a strong bearing on lake water quality and confound changes in external loads due to effects on stratification and associated internal loading (Rutherford *et al.* 1996);
- Lake management policy has often been reactive and failed to respond to lake water quality degradation in a timely manner, delaying lake restoration (Mueller *et al.* 2015);
- Catchment modelling is used in association with lake modelling to understand how land management will affect water quality. Modelling has shown that large (~25%) reductions in external loads of both nitrogen and phosphorus from predominantly agricultural sources are required to improve TLI from the middle to the lower end of eutrophic category, while climate change will exacerbate eutrophication (Hamilton *et al.* 2012);
- Alum dosing of lake inflows since 2006 has significantly improved lake water quality . (Smith *et al.* 2016), and;
- External loads from diffuse sources are being managed by assigning Nitrogen Discharge Allowances to landowners to manage nutrient loads within a 'sustainable' limit. Achieving these reductions requires land use changes such as retiring dairy farmland; employing BMPs alone on existing farmland is insufficient.

7 Impediments to shallow lake restoration

7.1 Overview

The lake restoration literature abounds with records of lake restoration projects that failed or were only partially successful. Søndergaard et al. (2007) reviewed the record of lake restoration projects in Denmark and the Netherlands, encompassing approximately 70 lakes, the majority of which were eutrophic and shallow (mean depths = 2.6 m and 1.5 m for the two countries). The majority of projects entailed removal of zooplanktivorous and benthivorous fish (56 lakes); alum treatment was applied to two lakes, while dredging was undertaken in eight lakes. The study showed that, although substantial improvements in lake water quality occurred in most lakes, long-term effects were less apparent, particularly in lakes that had been biomanipulated. Specific reasons for the failure of individual methods are listed in Table 7. The remainder of this section discusses some of the more general impediments to lake restoration.

Table 7. Impediments to the success of specific lake restoration techniques. Modified from Søndergaard et al. (2007).

Method	Reason
Reduce external nutrient loads	Insufficient load reductions
Reduce internal loads (physical)	Incomplete dredging Low P sorption of new sediment surface
Reduce internal loads (chemical)	Reduced P retention capacity of chemical (e.g., alum) over time Reduction/binding of ferric chloride by carbonate or sulphide
Biomanipulation (fish)	Insufficient biomass removed Rapid return of strong cohorts of zooplanktivorous fish High resuspension rate of loose sediment Continued high internal loading Instability due to low macrophye coverage

7.2 Insufficient reductions to external nutrient loads

Reducing external nutrient loads is fundamental to sustainably restoring lakes degraded by eutrophication; a failure to enact sufficient reductions to external nutrient loads is a major cause of poor lake restoration outcomes (Carpenter *et al.* 1999; Søndergaard *et al.* 2007; Gulati *et al.* 2008; Gross and Hagy 2017) The magnitudes of external load reductions required to achieve lake restoration goals are often under-estimated and managers should be wary of setting over-optimistic targets.

Carpenter et al. (1999) analysed management policies for lakes subject to varying degrees of eutrophication. They grouped lakes in terms of responses to external load reductions, classifying lakes as either "recoverable" (recovery is immediate and proportional to phosphorus load reductions), "hysteretic" (very high reduction in phosphorus loads are required for a period – see Section 7.4 below) and "irrecoverable" (external phosphorus load reductions alone are insufficient to restore a lake). Shallow lakes were identified as most likely to correspond to the latter two categories. Policy analysis showed that the effects of stochasticity (e.g., in weather), lags and uncertainty in limnological models means that, to maximise economic benefits of lake restoration, nutrient load reduction targets should be more ambitious than those indicated by empirical limnological models (e.g., TP–chl *a* regressions) alone. Causes of uncertainty in load reductions and response times are described further in Sections 7.2 and 7.4 below.

There is a growing recognition that the adoption of BMPs alone is insufficient to restore eutrophic lakes in catchments with high external nutrient loads from diffuse agricultural sources. Osgood (2017) reviewed the efficacy of BMPs to reduce external phosphorus loads to lakes in the USA, where BMPs are defined as "management techniques or landscape-modifying actions" such as detention basins, swales, constructed wetlands, filter strips or porous pavements. The author noted that phosphorus load reductions of >80% are required to restore most eutrophic lakes, yet the maximum possible phosphorus load reduction due to BMPs was ~50%, with reductions of <25% typically recorded in practice. The author concludes that the use of BMPs alone is "generally inadequate as a strategy to reverse eutrophication" and advocates for restoration plans to also focus on internal loads by including in-lake engineering or chemical treatments, as these provide faster improvements, typically at reduced cost.

7.3 Lags

Lags can delay improvements in lake water quality following implementing lake restoration actions. Lags can occur due to delays between implementing actions in the catchment and reductions to external loads to a lake associated with biogeochemical processes that govern nutrient transport (lags in the catchment response). Additionally, lags can occur due to inertia caused by persistent internal loading following restoration actions (lags in the lake response).

Lags in the catchment response can occur due to long groundwater transit times (primarily an issue for nitrogen) or due to enrichment of catchment soils with phosphorus, which results in persistently high external phosphorus loads (Hamilton 2011). Groundwater lags are a particularly issue for the management of Lake Rotorua (Section 6.9), for which groundwater mean residence times in sub-catchments are estimated to range from 30–145 years, thereby causing great inertia between catchment actions and changes to nitrate loading to the lake (Morgenstern *et al.* 2015). Lake Rotorua perhaps represents an extreme example of the potential effect of groundwater lags, although this issue is relevant to some lakes in the Waikato, particularly volcanic lakes that overlie the permeable geology of the Taupō Volcanic Zone. The ongoing erosion of soil enriched with phosphorus can also cause lags in catchment responses in the order of decades, particular in catchments dominated by high intensity agriculture (Carpenter 2005) such as many of those in the Waikato region. In extreme cases, lags exceeding 100 years are possible in catchments where groundwater percolation pathways are saturated with phosphorus from anthropogenic sources (Hamilton 2011).

As described in Section 3.2, lake bed sediments can sequester nutrients that can fuel internal loading, creating lags in the response of lake water quality to restoration actions. This can maintain poor water quality for years, even after external loads have been reduced (Søndergaard *et al.* 2003, 2013). Providing that sufficient external load reductions are maintained, then internal loads should decline and result in water quality improvements within 15 years (Jeppesen *et al.* 2005). The short lag time for the restoration of Lake Apopka (Section 6.4) perhaps indicates that lag times may be low in Waikato peat lakes with organic soils low in iron; however, the response may be substantially longer than 15 years in large hypereutrophic riverine lakes in the Waikato that have received high sediment loads, e.g., Lake Waikare (Lehmann *et al.* 2017). It is therefore necessary to consider such lags when framing expectations for lake restoration programmes; lags due to internal loading are a common cause of delayed responses to lake restoration, e.g., see case study of Lake Finjasjön (Section 6.7). Lags due to internal loading should also be considered when planning the best timing of restoration actions; for instance, Søndergaard et al. (2007) highlight that biomanipulation is likely to be most successful when internal loads are approaching equilibrium conditions.

Further insights into the rate of recovery are provided by McCrackin et al. (2016) who conducted a global meta-analysis that included data from 57 lakes, predominantly in the northern hemisphere. The analysis focused on recovery rate and recovery ‘completeness’, which was based on improvements relative to estimated baseline conditions. The researchers estimated that baseline conditions could be achieved 15 years (±7 years) after complete nutrient load reductions and 31 years (±13 years) after partial nutrient load reductions. Shallow and deep lakes were not distinguished in the analysis and therefore these estimates may be biased low for shallow lakes in general, due to the generally intransigent response of shallow lakes. Median recovery rate was lowest in agricultural catchments, perhaps reflecting the predominantly diffuse nature of these sources and the potential for soil enrichment discussed above.

7.4 Hysteresis

The model of alternative stable states in shallow lakes (Figure 3) indicates that to restore a turbid phytoplankton-dominated lake to a previous clear macrophyte-dominated state, it is necessary to reduce external nutrient loads to levels that are lower than when the lake originally ‘flipped’. This partly reflects the contribution of persistent internal loading described in Section 7.2 above. Furthermore, submerged macrophytes stabilise clear water conditions by reducing sediment resuspension, up-taking dissolved nutrients from the water column and providing refuge for zooplankton, which graze on phytoplankton (Scheffer *et al.* 1993). Thus, following a transition to a turbid phytoplankton-dominated state, it is necessary to reduce nutrient loads to a lower level to achieve the critical turbidity threshold required for macrophytes to re-establish (see different thresholds in Figure 3). The critical turbidity therefore depends on the direction of change, i.e., the turbidity–nutrient load relationship exhibits hysteresis. This impedes lake restoration as the restoration of clear water conditions requires relatively large external load reductions, the magnitudes of which are frequently underestimated (see Section 7.2). A corollary to this is that it is more cost-effective to take proactive measures to maintain water quality in clear macrophyte-dominated lakes, than to try and restore such lakes after they have transitioned to a turbid state (Figure 5).

7.5 Delayed macrophyte re-establishment

Despite achieving improvement in water clarity, it can still be challenging to succeed in re-establishing macrophytes in a lake that has previously experienced a collapse in macrophyte abundance. Bakker *et al*. (2013) identified the following impediments to re-establishing a diverse macrophyte community following shallow lake restoration: 1) lack of viable propagules; 2) grazing by wildfowl or herbivorous fish, and; 3) persistently high nitrogen conditions. The first two impediments may be overcome by planting and establishing protective enclosures (Section 5.4.6). The precise causes of poor re-establishment due to high nitrogen concentrations are unclear, although they certainly include shading due to phytoplankton and, potentially, periphyton growth (González Sagrario *et al.* 2005). Destabilisation of macrophyte beds due to a reduction in the root-to-shoot ratio caused by high nitrogen concentrations also seems to be a potential factor (Boar *et al.* 1989; Moss *et al.* 2013). González Sagrario (2005) identified a threshold TN concentration of 1–2 mg/L (regardless of TP concentration) at which collapse of submerged macrophytes can occur; however, the authors stress that this threshold may not be universal and depends on fish abundance. The threshold at which macrophyte beds re-establish is also expected to be lower than the threshold at which they collapse, due to the issue of hysteresis discussed above (Section 7.4).

7.6 Climate change

Climate change is predicted to exacerbate eutrophication (Paerl *et al.* 2011; Moss *et al.* 2011). This will increase the effort required to achieve desirable lake water quality outcomes (Jeppesen *et al.* 2009b). In some cases, this may also require lake water quality targets to be re-evaluated, e.g., in instances where targets are linked to an estimated reference condition that applies to a pre-anthropogenic climate change state.

Greater mean temperatures are predicted to directly increase phytoplankton biomass by directly increasing growth rates, particularly for cyanobacteria which are expected to generally become more dominant in temperate eutrophic lakes (Paerl *et al.* 2011; O'Neil *et al.* 2012). In addition, climate change is predicted to cause a range of more indirect effects that will reinforce the symptoms of eutrophication; these effects include: proliferation of planktivorous fish that depress zooplankton abundance; greater internal loading associated with deoxygenation caused by warmer temperatures; increased nutrient mineralisation; reduced lake depth during droughts, and; higher nutrient loads in storm flows due to more intense precipitation events (Moss 2011).

Modelling can be used to tease apart the relative responses of lake water quality to nutrient loading and climate change. Trolle (2011) used one-dimensional lake ecosystem models of three New Zealand lakes to simulate the effects of changes in nutrient loads in conjunction with projected increased water temperatures for 2100 associated with the IPCC A2 scenario, which entailed increases to regional mean air temperatures of 2.5–2.7°C. The results showed that TN, TP and chl *a* concentrations were generally greater for the climate change scenario, with increases comparable with those expected for increased external nutrient loading of 25–50%

under the current climate. Similarly, sensitivity analysis conducted as a part of a modelling study[5] of four representative Waikato shallow lakes (Rotomānuka, Ngāroto, Waahi and Waikare) showed that warming temperatures were expected to exacerbate eutrophication and hinder lake restoration by increasing internal loading and cyanobacteria growth rates, supporting a conclusion that "the prognosis for these shallow lakes is dire" (Lehmann *et al.* 2017).

7.7 Need for lake-specific approaches

The diverse nature of lake characteristics poses a challenge to adopting a centralised approach to lake restoration policy. Planning for lake restoration requires a good understanding of the limnological characteristics of a lake in order to develop an appropriate and individualised approach that accounts for factors such as lake type, nutrient sources and food-web structure (Hamilton et al. 2016). This requirement particularly applies to shallow lakes, which are generally more structurally complex than deep lakes (Moss *et al.* 2005) and can exhibit more variable responses to nutrient load reductions (Havens *et al.* 2001). Therefore, the need to collect information to sufficiently understanding lake characteristics can hinder the development of sound restoration plans.

Ecological and lake water quality decline in shallow lakes typically has multiple causes, implying that multiple actions are necessary for restoration (Carpenter and Lathrop 1999). While the control of external loads is a fundamental component of lake restoration strategies for lakes in general, it is nonetheless necessary to understand the specific details of the nutrient budget for an individual lake to identify appropriate local interventions that provide the most cost-effective controls on external nutrient loads (Withers et al. 2014). Other lake restoration methods are not necessarily applicable to all shallow lakes and therefore their appropriateness needs to be considered on a lake-by-lake basis. Individual methods may also provide complementary values that are in addition to reducing trophic status, e.g., improving habitat for ecologically important species or enhancing recreational values. This requires considering trade-offs when evaluating the applicability of different methods.

Decision support tools have been developed to help identify the most appropriate lake restoration methods for a particular lake, notably the framework developed by Hickey and Gibbs (2009). To guide lake restoration policy, such a framework could potentially be developed for shallow Waikato lakes that reflects the lake types in the region, as well as the specific priorities of WRC and local communities. Further, the formal process of structured decision making (Gregory *et al.* 2012) could be used to guide lake restoration at regional and watershed scales in the Waikato, providing a transparent, rigorous and defensible approach to support resource allocation (discussed further in Section 8.3).

7.8 Social and political challenges

This review has largely focused on the most appropriate methods for lake restoration. However, numerous social and political challenges create a gap between identifying an appropriate prescription (i.e., lake restoration actions) and achieving successful outcomes (Blomquist and Schlager 2005).

A fundamental problem in lake restoration is that those who cause diffuse pollution do not benefit from reduced pollution (Carpenter and Lathrop 1999). This requires strong political

[5] The results of this modelling study have yet to be validated, either based on comparisons with historical observations or further monitoring.

leadership and a range of policy instruments (Figure 4) to enact suitable controls and internalise the external costs of activities that generate high nutrient loads, i.e., to make polluters pay (Abell *et al.* 2011a). The development of novel policy instruments is being actively pursued in New Zealand (Section 4.2.1); however, successfully realising lake restoration plans remains a challenge. Duncan (2014) reviewed developments in diffuse pollution management in New Zealand and concluded that the focus on applying predictive models (e.g., Overseer) to quantify targets for regulatory compliance ("rule by numbers") has neglected the social-political component of environmental management. Although Duncan (2014) recognised the value of a quantitative framework for providing accountability, the author argued that there are epistemological, institutional and practical challenges that need to be overcome to effectively link science with practice. Examples of these challenges include: legal and administrative issues involved with updating models (epistemological), the lengthy and infrequent process for revising regional plans that prescribe environmental limits (institutional), and; the potential for land owners such as farmers to change how they provide inputs to nutrient budget models to comply with regulations (practical). Another challenge is the need to coordinate actions among multiple land owners and administrative agencies to achieve catchment-scale management; it is revealing that a key factor in the success of the restoration of Lake Finjasjön (Section 6.7) was identified as the fact that the lake catchment included a single municipality, which avoided the potential for contentious and protracted discussions among multiple government agencies about how to split restoration costs (Annadotter *et al.* 1999).

Mueller et al. (2015) highlighted the issue of long lags between water quality degradation and lake management, noting that fifty years elapsed between scientists documenting lake water quality declines and managers implementing formal regulation to control nutrient loads to Lake Rotorua (Bay of Plenty). They noted that high visibility of environmental degradation was an important driver of lake restoration, e.g., lake closures or health warnings acted as motivation for management responses. In this regard, shifting baselines will affect the public's perception of water quality degradation and the resulting management response (Gillon *et al.* 2016). Thus, the public pressure to restore a specific lake (and associated social licence for managers to enact punitive controls on polluting economic activities) is likely to be greatest for lakes that have experienced recent and sudden declines in water quality, relative to those that have experienced long and gradual declines. Local volunteer groups can have a large influence on the success of freshwater restoration projects (Stanford et al. 2018) and likely foster public participation and community 'buy-in' that is necessary to achieve successful restoration outcomes. Involving local stakeholder groups is particularly important when attempting to control nutrient pollution from diffuse sources as this often requires developing partnerships between multiple parties who need to feel included in the process (D'Arcy *et al.* 2006).

8 Conclusions

8.1 Overview

This review has considered the current scientific literature relating to the restoration of shallow lakes degraded by anthropogenic eutrophication. The review has considered studies undertaken worldwide, with particular focus on lakes in temperate regions that have been degraded due to nutrient pollution from agricultural land. Accordingly, this review can inform lake restoration policies to address degraded water quality in shallow Waikato lakes.

Conclusions in relation to the six research questions that were identified in the Introduction are presented in Section 8.2 below. Some options regarding next steps are then presented in Section 8.3.

8.2 Answers to research questions

1. *What methods are available to address eutrophication in shallow lakes?*

Reducing external nutrient loads is fundamental to the successful and sustainable restoration of eutrophic shallow lakes (Cooke *et al.* 2005; Jeppesen *et al.* 2007a; Moss 2007b). In addition, there is a range of supplementary methods that have proved successful; these can be broadly grouped into: controlling internal loads, biomanipulation and hydrologic manipulations, as summarised in Table 3 and Figure 5.

2. *What is the performance of individual lake restoration methods, as quantified by robust monitoring? How relevant are these results to Waikato shallow lakes?*

External load reductions were a consistent component of restoration programmes that resulted in water quality improvements. However, external nutrient load controls have generally only been successful when major reductions are achieved and underestimating the magnitude of reductions required is a major cause of lake restoration failure (Section 7.2). The generally large and diffuse nature of external loads poses a major challenge to the restoration of shallow Waikato lakes, as does the legacy of high nutrient loads in the bed sediments. This means that controlling external loads alone is unlikely to achieve successful outcomes such swimmable conditions (New Zealand Government 2017b) in the most degraded lakes within a timeframe of less than a 15 years.

Methods to control internal loads (Section 5.3) have been widely adopted to support lake restoration and can be important for overcoming inertia caused by internal loading in shallow lakes that have highly nutrient-enriched sediments. Controls on internal nutrient loads have been an important and necessary component of several lake restoration projects that achieved water quality improvements, e.g., see case studies of Lake Kraenepoel, City Park Lake and Lake of the Isles in Section 6. In particular, the targeted use of materials to adsorb dissolved phosphorus and flocculate organic material has been used successfully overseas but has received relatively little application in New Zealand. The use of these approaches could therefore be explored further for Waikato shallow lakes, with the understanding that the successful use of such products depends on numerous lake-specific factors and requires considering ecological risks and cultural sensitivities. This could build on previous work that identified potential to use phosphorus adsorption/flocculating materials in peat lakes as part of an integrated approach that also involved reductions of external loads by approximately 50% in catchments that are highly modified (Faithfull *et al.* 2005). Controls on internal loads are likely to be particularly necessary to restore water quality in many shallow riverine lakes in the Waikato that have historically received large inputs of nutrient-enriched sediments, although the large size of many of these lakes poses logistical and financial challenges. Successful application of this technique requires an understanding of sediment nutrient fluxes in a specific lake; this has been identified as a particular information gap for many Waikato lakes (Lehmann *et al.* 2017).

Biomanipulation has proven to be successful in many temperate lakes in the northern hemisphere (Section 6.3); however, characteristics of lake food webs mean that biomanipulation of fish communities has more limited applicability in New Zealand (Burns 1998; Burns *et al.* 2014). Of the range of biomanipulation techniques that are established, the control of benthivorous fish seems to have greatest applicability to shallow lakes in the Waikato, given the high biomass of invasive fish such as carp in many lakes. Despite this, the large-scale removal at Lake Ohinewai indicates that invasive fish control requires high, persistent effort and can only make a minor contribution (at best) to restoration of shallow, hypertrophic lakes in the Waikato (Tempero and Hicks 2017). Nonetheless, there is an established effect pathway between benthivorous pest fish and poor water quality, meaning than pest fish control has a role as a restoration tool in shallow Waikato lakes, particularly in very shallow lakes with thick deposits of fine sediments (e.g., many riverine lakes) and lakes where opportunities for ongoing re-colonisation from connected waterbodies are limited. Planktivorous fish control (e.g., juvenile perch) and, possibly, promotion of kākahi may be applicable to a subset of lakes, although the use of bivalves for lake restoration requires further study.

Hydrologic manipulations (Section 5.5), notably inflow diversion or augmentation, can be highly successful in the right circumstances, although the applicability of these techniques is highly dependent on the local geography. There may be opportunities to use these techniques in shallow Waikato lakes, although this needs to be evaluated on a lake-by-lake basis. Manipulating water levels also has potential to achieve more modest outcomes, although again this depends on lake-specific factors such as the feasibility to install a control structure and the characteristics of riparian and emergent vegetation communities.

3. *What shallow lake restoration projects have been deemed "successful"? What were the restoration targets and how were they achieved?*

Numerous examples of successful shallow lake restoration projects, including restoration targets, are described in Section 6 and summarised in Table 6. Successful projects typically involve:

- major controls on external nutrient loads;
- additional controls on internal nutrient loads;
- nutrient load reduction targets that address a range of sources;
- a lake restoration plan with quantitative targets;
- leadership by a dedicated water management agency;
- regulation to achieve nutrient load targets;
- a political framework that supports a catchment-scale approach to management;
- engagement with motivated local groups;
- good knowledge of the limnology of the lake; and
- a robust monitoring programme to monitor change.

4. *Are there successful examples of shallow lake restoration that solely involved implementing best management practices to control diffuse nutrient pollution in the catchment? How relevant are these to the Waikato region?*

No examples of successful lake restoration were identified that only involved implementing BMPs in conjunction with existing agricultural land use practices; the use of BMPs alone is generally regarded as inadequate to restore lakes that are substantially degraded due to eutrophication (e.g., Osgood 2017). Nonetheless, the use of BMPs can yield reductions to external nutrient loads (e.g., McDowell and Nash 2012) and are an important tool. Given the dominance of agriculture in the Waikato, the use of BMPs has an important role to manage nutrient loads in Waikato lake catchments where it is not desirable to pursue land use change, particularly given that it is more cost-effective to proactively prevent nutrient pollution from reaching a lake that to reactively try and restore degraded lakes. However, it is important to maintain realistic expectations regarding the limited potential of BMPs to cause substantial improvements in lake water quality, particularly in shallow lakes that have high internal nutrient loads.

5. *What were the major drivers of successful lake restoration projects (e.g., legislation, industry-led initiatives, voluntary action by community groups)?*

Legislation is an important driver of successful lake restoration because it provides a mandate for government agencies to enforce the strict controls on external loads that are necessary. The presence of an active local restoration group is also an important driver of success (Stanford *et al.* 2018), particularly when motivated by a perceived environmental crisis (Gross and Hagy 2017). As discussed in Section 7.8, addressing social and political challenges is an important requirement for successful lake restoration that is often overlooked. At the catchment scale, leadership by a dedicated agency and engagement with motivated local groups are important for success, particularly where it is necessary to coordinate controls on diffuse pollution among multiple landowners. At both the catchment and regional scale, structured decision making (Gregory *et al.* 2012) can provide a tool to help WRC to develop effective and science-based restoration plans in a transparent manner that promotes 'buy-in' and consensus among stakeholders.

6. *Are there examples of lake restoration projects that have successfully addressed multiple pressures in combination? If so, how has this been achieved?*

There are multiple examples of the use of invasive fish control that provide complementary benefits for both biodiversity and lake water quality values (e.g., carp control in Lake Susan, described in Section 5.4.1). This is particularly relevant to Waikato shallow lakes. Further, the available evidence indicates that several methods may only have a modest effect on the symptoms of eutrophication in highly degraded lakes; however, these methods may provide additional benefits that need to be considered when developing lake restoration plans. For example, small-scale wetland creation alone is not expected to provide major improvements in water quality in shallow Waikato lakes; however, this can provide considerable biodiversity and aesthetic co-benefits that should be considered when evaluating costs and benefits.

Climate change poses a major challenge to achieving lake water quality objectives (Section 7.6). While this clearly cannot be addressed at the lake catchment scale, it will be increasingly important for catchment managers to consider ways to adapt to climate change to build resilience and support multiple objectives. For example, greater use of landscape-scale floodwater management (e.g., detainment bunds and wetland creation) can contribute to controlling sediment and nutrient loads, while also supporting flood risk management and erosion control objectives.

8.3 Next steps to consider

It is relevant to consider potential next steps for WRC to make further progress with shallow lake restoration (see Section 4.3). Some options based on the literature review are presented below, recognising that these steps may already be underway by WRC as part of their ongoing commitment to restore and protect lakes. In summary, these are:

1. Continue to prioritise work to promote and enforce changes that will reduce external nutrient loads;
2. Use existing information to link the techniques described in this report with individual lakes where the techniques are suitable, and;
3. Consider the potential to use structured decision making to evaluate and prioritise lake restoration actions.

Controlling external nutrient loads is fundamental to managing environmental risks associated with eutrophication (Section 7.2). WRC recognises this and is actively working with partners to develop new tools to promote and enforce the adoption of sustainable nutrient management practices by land users (Waikato Regional Council 2017a). Recent national and regional policy reforms (Section 4) have provided a catalyst for this work and it is important that momentum is maintained, particularly given the magnitude of the challenge presented by Waikato shallow lakes (Section 2.2). Such work will be important to reduce nutrient pollution in catchments throught the region and ensure that social and economic policies are aligned with desired freshwater outcomes

In addition to reducing nutrient pollution generally, it is clear that targeted in-lake actions are required to achieve lake water quality goals in many shallow Waikato lakes that are moderately to severely degraded. In such lakes, catchment-based actions alone (e.g., implementing BMPs) are inadequate to meet restoration targets and actions are required to reduce internal loads. This review has considered the relevance of restoration actions to Waikato lakes; however, the need for lake restoration strategies to reflect the limnological characteristics of individual lakes (Section 7.7) means that it is not possible to provide definitive conclusions here about which restoration actions are 'best' for individual Waikato shallow lakes. To aid decision making, there is a need to review the characteristics of individual lakes to identify which options may be suitable. To provide high-level decision support, such a review need not be particularly onerous as there is good potential to use information that has already been compiled, e.g., in the Shallow Lakes Management Plan (Waikato Regional Council 2014a,b). It is desirable to use the information presented in this report (and in the cited references) to specify criteria used to define applicability, e.g., a maximum lake size below which dredging may be a realistic option. The envisaged outcome of this exercise would be a list of restoration techniques for each priority shallow lake that are applicable based on technical considerations (only) of characteristics such as: lake morphometry, fish communities, alkalinity, hydrology and degree of impairment. The latter characteristic could potentially be determined based on departure of current TLI from an estimated reference state. Ideally, the list will make some distinction between techniques that are highly suitable and those that are less suitable but still feasible.

The step described above will help to further identify which techniques are relevant (from a technical perspective) to individual lakes. However, when designing lake restoration plans, there is a need to consider broader social, economic and political factors (Section 7.8). These may include: funding, land tenure, cultural concerns about applying chemicals to waterbodies and co-benefits for wildlife such as waterfowl. This necessity calls for an approach to environmental decision making that integrates science-based knowledge, viewpoints of multiple groups and multi-criteria analysis. As part of the Shallow Lake Management Plan (Section 4.3.4), WRC has already defined an inter-agency approach and associated strategies for shallow lake restoration throughout the region. This may provide a fully satisfactory framework to guide prioritisation and decision making that is necessary to make further progress. However, supplementary approaches may be valuable to support strategic planning, or to inform complex decisions in high priority catchments. To this end, structured decision making could be adopted to guide decision making (Gregory *et al.* 2012). Structured decision making has been used for over two decades in British Columbia, Canada to explore trade-offs in water management. In New Zealand, the approach has yet to be generally adopted in the field of water resources management[6]; however, it has been used in other fields, e.g., to inform Hector's dolphin conservation (Conroy *et al.* 2008) or agricultural decisions (Dooley *et al.* 2010). Structured decision making is based on six planning steps that are designed to help parties manage the complexities of technical decisions and trade-offs between values (Gregory *et al.* 2012):

1. Clarify the decision context
2. Define objectives and measures
3. Develop alternatives
4. Estimate consequences
5. Evaluate trade-offs and select
6. Implement, monitor and review

This approach could help with lake restoration planning in at least two contexts: 1) at the regional scale as an internal tool to help WRC to allocate resources, and; 2) at the catchment scale to help multiple stakeholders to develop a restoration plan for a specific lake, e.g., a highly degraded lake that is a high priority for restoration and requires complex decisions about trade-offs to be made.

Step 1 involves defining the scope of the decision, e.g., the geographical boundaries and the techniques in the 'toolbox' that may be considered. Step 2 involves defining the 'things that matter' (e.g., issues such as specific elements of cost, environmental and social values), plus quantitative metrics that can be used to evaluate alternatives (e.g., capital and operational costs in NZ$, response time in years). If the objective is to prioritise lakes for restoration, then estimates of the magnitude that water quality has declined from a reference state (e.g., based on estimated change in TLI) could provide a particularly valuable metric. Step 3 involves identifying candidate management actions, i.e., the actions that are technically applicable to one or more lakes, as identified in the option described previously. Step 4 involves estimating consequences, typically using a table that is designed to present information about key trade-offs in a way that enables people with different levels of technical expertise to understand the effects and to participate in an informed way. Step 5 then involves evaluating trade-offs and selecting a preferred approach, with clear documentation of the rationale for decisions and associated uncertainties. Step 6 then involves implementation, with recognition that it will likely be necessary to use an adaptive management approach to manage uncertainties and fine-tune implementation protocols. Steps 2–5 are typically iterative, requiring multiple rounds to narrow the focus of the decision making. If WRC wishes to consider this approach, it may be useful for them to initially consult with other agencies that have worked through a structured decision making process (but not necessarily in a water management context) to inform their expectations.

[6] Although there is some overlap with participatory integrated watershed management approaches that have been used in catchments such as the Motueka (Fenemor *et al.* 2011).

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