Technical Committee - 1 August 2018 Attachments

Minutes of a meeting of the Technical Committee held in the Edinburgh Room, Municipal Chambers, The Octagon on Wednesday 13 June 2018, commencing at 10:30 am

Membership

Cr Andrew Noone	*(Chairperson)*
Cr Ella Lawton	*(Deputy Chairperson)*
Cr Graeme Bell	
Cr Doug Brown	
Cr Michael Deaker	
Cr Carmen Hope	
Cr Trevor Kempton	
Cr Michael Laws	
Cr Sam Neill	
Cr Gretchen Robertson	
Cr Bryan Scott	
Cr Stephen Woodhead	

Welcome

Cr Noone welcomed Councillors, members of the public and staff to the meeting.

1. APOLOGIES

Resolution

That the apologies for Cr Woodhead be accepted.

Moved: Cr Noone
Seconded: Cr Robertson
CARRIED

2. LEAVE OF ABSENCE
Leave of Absence was noted for Cr Kempton.

3. ATTENDANCE
Sarah Gardner	*(CEO)*
Nick Donnelly	*(DCS)*
Tanya Winter	*(DPPRM)*
Sian Sutton	*(DSHE)*
Gavin Palmer	*(DEHS)*
Scott MacLean	*(DEMO)*
Sally Giddens	*(DPC)*
Ian McCabe	*(Executive Officer)*
Petra Hunting	*(interim Committee Secretary)*

4. CONFIRMATION OF AGENDA
The agenda was confirmed as tabled.

5. CONFLICT OF INTEREST
No conflicts of interest were advised.

6. PUBLIC FORUM
No public forum was held.

7. PRESENTATIONS
No presentations were held.

8. CONFIRMATION OF MINUTES

Resolution

That the minutes of the meeting held on 2 May 2018 be received and confirmed as a true and accurate record.

Moved: Cr Noone
Seconded: Cr Lawton
CARRIED

9. ACTIONS

Status report on the resolutions of the Technical Committee.

Report No.	Meeting	Resolution	Status
10.1 Central Otago STED site no. 2	2/5/18	*That CODC, ORC and NZTA agree the criteria, and that Council requests for the Central Otago District Council to formally advise their preferred site, which satisfies*	In process

		the agreed criteria for the second new STEDS in Central Otago, by 31 July 2018. *That staff keep the elected arm informed as to the process of this request (councillors are kept informed and included in all communications in relation to the STED site options).*	

10. MATTERS FOR COUNCIL DECISION

10.1. Shag/Waihemo River and Waianakarua River Morphology and Riparian Management Strategies - Council Committee Hearing

The Chair welcomed Ellyse Gore, Senior Natural Hazards Analyst and Jean-Luc Payan, Acting Manager Resource Science to the meeting.

Cr Laws left the meeting at 03:52 pm and returned at 03:54pm.

Resolution

a) That the report be received; and
b) That the Shag/Waihemo River and Waianakarua River morphology and riparian management strategies are endorsed.

Moved: Cr Deaker
Seconded: Cr Lawton
CARRIED

10.2. Leith Flood Protection Scheme Dundas Stage Programme

The design of the Dundas Street stage of the Leith Flood Protection Scheme is nearing completion. Construction contract documents are being prepared and discussions with landholders and other stakeholders are progressing.

The programme for construction of this stage of the scheme is to invite tenders in July and to award a contract in September, subject to favourable tenders being received. Construction would take place over Summer 2018/19. The 2018-28 Draft Long Term Plan is based on this programme.

This stage completes the flood protection component of the scheme. Importantly, other stages already constructed become fully effective once this stage is commissioned as it is upstream of those other stages. Completion of this stage will be a significant milestone.

There was a lengthy discussion held on the request from University of Otago to delay commencement of the Dundas Street stage until after December 2019, to avoid construction works that may impact on access to and enjoyment of the university's 150^{th} celebrations during 2019. It was noted approving this request would cause a delay of two years before construction could recommence in the summer of 2020/21.

The valuable partnership and relationship ORC and University of Otago share was acknowledged. It was agreed the longer construction is delayed, the greater the risk to the community of a flood event occurring without the infrastructure in place to contain it, as there is currently a 4% likelihood of a flood occurring in the next two years.

It was agreed to amend the motion to decline the request and continue the project as planned, with ORC staff to liaise with the university on ways to avoid or minimise disruption at the university.

Resolution

a) That this report is received and noted;
b) The request by the University of Otago for ORC to delay construction of the Dundas Street stage of the Leith Flood Protection Scheme is declined.

Moved: Cr Robertson
Seconded: Cr Neill
CARRIED

11. MATTERS FOR NOTING

11.1. An assessment of the Clean Heat Clean Air program's effectiveness
Cr Noone welcomed Deborah Mills, Environmental Scientist, to the meeting.

The purpose of the paper is to provide an assessment of the Clean Heat Clean Air (CHCA) programme's effectiveness. CHCA was designed to provide a financial incentive for homeowners to upgrade their solid-fuel burners to cleaner heating options. A paper presented to the Finance and Corporate Committee on 13 September 2017 detailed the financial status of the programme. This paper examines the environmental outcomes related to work done for the CHCA initiative, fulfilling an annual plan target in the Air Management Planning project (A4 – Air Quality).

There was a discussion on the reported outcomes, whether there is an ongoing dialogue with peers around the country on similar schemes to achieve these aims and confirmed a future collaborative effort of many local and national government authorities is planned. It was noted the government are expected to release updated timelines at the end of 2019.

Cr Scott left the meeting at 04:22 pm and returned at 04:23pm

Resolution

a) That this report be received.
b) That this report be used to inform the review of ongoing financial incentives for Air Quality, proposed for 2018/2019 in the 2018-2028 Draft Long-Term Plan.

Moved: Cr Lawton
Seconded: Cr Hope
CARRIED

11.2. Director's Report on Progress
Resolution

a) *That the report be received and noted.*

Moved: Cr Scott
Seconded: Cr Lawton
CARRIED

11.3. Lake Hayes Restoration
Cr Noone welcomed Ben Mackey, Acting Manager Natural Hazards, to the meeting.

Three remediation options to improve the water quality in Lake Hayes are currently being assessed by Otago Regional Council. At this stage, Council has made no decision on whether it would pursue any particular option and public consultation on technical and funding options is still to be completed.

There was a discussion on the various implementation issues to be resolved successfully for each option, as well as the fit with other policies in ORC's Regional Plan: Water.

Resolution

This report is received and noted.

Moved: Cr Scott
Seconded: Cr Laws
CARRIED

12. NOTICES OF MOTION

No Notices of Motion were advised.

13. CLOSURE

The meeting was declared closed at 04:55pm.

Chairperson

Sub-program	Priority Ranking	Associated costs	Justification	Lead agency
1) Is *Lindavia intermedia* a native or non-native species?				
i) Investigation of cell genetics (microsatellite analysis) of NZ and overseas *L. intermedia* populations	**High - Immediate**	***Currently funded by ORC. To be delivered by end of Jun 17.***	This work will indicate if *L. intermedia* has recently arrived in NZ and should be considered an invasive species.	**ORC** *Landcare* *In progress*
ii) Comprehensive examination of NZ diatom samples, collections, reports	**High - Immediate**	**$11K** for detailed assessment of 3 separate catalogued collections Delivery 3 to 6 months.	To determine if previous '*Cyclotella*' identifications are in fact *Lindavia*. To help isolate the length of time the diatom has been present in NZ.	**ORC** *Done* *NIWA* *C Kilroy*
iii) Historical dynamics of *L. intermedia* in NZ lakes from which it has been reported using paleolimnological diatom analysis of dated sediment cores.	**High - Immediate**	4 priority lakes in Otago **$56K.** ($14K per lake). Delivery 6 to 9 months for Otago's 4 priority lakes. Estimated 10 lakes needed to be cored across Otago, Southland, Canterbury and Hawke's Bay	This work will allow a precise estimate of the time that *L. intermedia* has been present in NZ and will complement the microsatellite work currently being undertaken in (i) above.	**ORC** *In progress* *Uni Otago* *Landcare*

Sub-program	Priority Ranking	Associated costs	Justification	Lead agency
2 What are the drivers of(A) *L. intermedia* dominance in lakes and				
2A i) Literature review of shifts in lake phytoplankton to increased dominance by (*Lindavia*-like) centric diatoms (e.g., climate connection)	**High - Immediate**	$3K – if aligned with 2B i).	This would increase our understanding of shifts and drivers of phytoplankton community structure to one dominated by centric diatoms and provide extremely valuable information to the NZ context.	**ORC** *University of Otago*
2A ii) Are historical *L. intermedia* dynamics correlated to environmental drivers in our lakes?	**Medium - Medium term**	$219K Delivery 3 years [Note: This work is covered in the University of Otago MBIE bid.]	As with 2B ii) this work-stream is extensive and likely best delivered through a University and a number of postgraduate and post-doctoral research programs.	**Catchments Otago / Uni. Of Otago / CRIs / support from RC's**
2A iii) Are proliferations of *Didymo* and *L. intermedia* in South Island waters related to a common driver or species incursion?	**Medium - Medium term**	$19K minimum Delivery difficult to estimate	If the timing and spread of these two incursions are coherent, then that would provide evidence of a common incursion (both place and time) and support management of future incursions and responses.	**Catchments Otago / Uni. Of Otago / CRIs / support from RC's**

Sub-program	Priority Ranking	Associated costs	Justification	Lead agency
2) What are the drivers of (B) polysaccharide overproduction by *L. intermedia*?				
2B i) Comprehensive literature review on diatom polysaccharide overproduction from similar situations overseas	**High - Immediate**	$10K Delivery 3 to 6 months	Seen as a top priority and would increase our current understanding of TEP production and the lake snow phenomenon. A straightforward exercise that hasn't been undertaken to date.	**ORC** *Uni Otago* *contract*
2c) Study of the relationships between diatom polysaccharide overproduction and (1) nutrient availability, (2) climate warming, and (3) grazing pressure.	**High - Medium term**	Year 1: $204K Year 2: $211K Year 3: $198K Delivery 3 years [Note: This work is covered in the University of Otago MBIE bid.]	As with 2A ii) this work-stream is extensive and likely best delivered through a University and a number of postgraduate and post-doctoral research programs.	**Catchments Otago / Uni. Of Otago / CRIs / support from RC's**

<table>
<tr><th>Sub-program</th><th>Priority Ranking</th><th>Associated costs</th><th>Justification</th><th>Lead agency</th></tr>
<tr><td colspan="5">3) Can we develop technologies for effective sampling and monitoring of L. intermedia and lake snow?</td></tr>
<tr><td>i) The development of new sensor technology to monitor in situ polysaccharide concentrations in lakes.</td><td>High - Medium term</td><td rowspan="3">$300K per year for three years
-
Part of an MBIE Smart Ideas bid – decision on success due Sept 2017.
Bid successful
In progress</td><td rowspan="3">Capacity to monitor the abundance and spatial variability of lake snow is critical to understanding the environmental drivers that lead to lake snow production. At present these techniques do not exist.</td><td>Landcare Research / Uni. Of Otago / Support from ORC</td></tr>
<tr><td>ii) The development of cost-effective and efficient methods for quantitatively sampling lake snow in lakes (at different depths).</td><td>High - Medium term</td><td>Landcare Research / Uni. Of Otago / Support from ORC</td></tr>
<tr><td>iii) Can DNA methods be developed for the sensitive detection of L. intermedia in lakes?</td><td>Medium - Medium term</td><td>Landcare Research / Cawthron / support from RC’s</td></tr>
</table>

Sub-program	**Priority Ranking**	Associated costs	Justification	**Lead agency**
4) How might the spread of *L. intermedia* between lakes be stopped or slowed?				
i) Are the BNZ Didymo sanitation methods adequate for the disinfection of *L. intermedia*?	**High - Immediate**	Currently contracted by MPI who have engaged NIWA to review the effectiveness of Check – Clean – Dry on *Lindavia*	MPI are reviewing their Check/Clean/Dry campaign and how effective it is for other pest species.	**MPI / NIWA** **?**

Sub-program	**Priority Ranking**	Associated costs	Justification	**Lead agency**
5) Supporting citizen science				
	High - Medium term	$10K	Links to 3.	**ORC** *Aspiring Environmental* *In progress*

Lower Waitaki Plains Aquifer

Summary of the Groundwater Quality Monitoring (July 2016 - January 2018)

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ISBN [To be added]

Report writer: Frederika Mourot, Resource Scientist Groundwater
Reviewed by: Scott Wilson, Hydrogeologist Lincoln Agritech

Published August 2018

Overview

Background

The Otago Regional Council (ORC) is responsible for managing Otago's water resources on behalf of the Crown and the community. This includes an obligation to manage drinking water at the source in accordance with the Resource Management (National Environmental Standard for Sources of Human Drinking Water) Regulations 2007. This investigation presents the context and activities on the Lower Waitaki Plains and studies the impact of land use intensification on groundwater quality.

Why is water quality management needed?

Substances that can affect groundwater quality tend not to be visible. Often the only signs that groundwater quality is becoming affected by contaminants are long-term trends observed in regular analysis of bore water. Once water in an aquifer becomes contaminated, it can take years or even decades for the contaminants to be flushed out. It is important, therefore, to look periodically for any discernible trends and to forecast the potential of contamination using scientific techniques.

What this study found

The Lower Waitaki Plains Aquifer quality is demonstrating the significant impact of the land use on the plains, with a cumulative effect, as we travel along the groundwater flow paths towards the coast.

While the nitrate nitrogen concentrations are below the NZ Drinking Water Standards Maximum Acceptable Value (MAV), recurrent *Escherichia coli* (*E. coli*) concentrations above the NZ Drinking Water Standards MAV have been observed.

What should be done next?

The proposals and recommendations from this report should be discussed with the local community and other stakeholders and considered in the context of relevant statutory requirements for managing groundwater.

Technical summary

The Lower Waitaki Plains Aquifer is mainly unconfined and hosted in quaternary river deposits made of gravel, sand and silt and overlying relatively impervious tertiary sediments.
The aquifer is inferred to be recharged by rainfall, irrigation returns, high flow events from the Waitaki River and foothill creeks. The main groundwater flow direction appears to be following the topography and is from the west/south-west towards the north-east/east.
The predominant activity over the Plains is dairy farming, covering approximately 70% of the area in 2015. The Lower Waitaki Irrigation Scheme delivers water to more than 200 stakeholders and extends over 20,000 hectares (ha).
The ORC has been monitoring the groundwater quality on five groundwater bores since July 2016. The purpose of this report is to present the analysis of the data collected up to January 2018, with a focus on nitrate-nitrogen (nitrate-N) and *Escherichia coli (E. coli)* concentrations.

The groundwater is of calcium-magnesium-bicarbonate type, which is characteristic of immature, relatively freshly recharged groundwater. The results of the age dating sampling show mean residence times ranging between 2 and 7 year old.

Nitrate concentrations

An increase in the nitrate-N concentrations from approximately 2 to 5 mg/l, correlated to the elevation of the water table, has been observed between 1997 and 2003 on the aquifer long-term monitoring bore. The exact reason for this increase is not clear. This concentration rise could be linked to increased drainage caused by the development of the irrigation scheme completed in 1982 (increase in irrigation), providing less opportunity for pasture uptake and consequently more nitrates to be flushed into the aquifer.

The data collected with the extended monitoring network between July 2016 and January 2018 show that the nitrate-N median concentrations range between 0.99 and 6.6 mg/l within the aquifer between the most up-gradient bore and the most down-gradient one.

E. coli concentrations

Because the aquifer is identified as having a value of human use without treatment in Schedule 3A of the Regional Plan: Water, understanding the trends in *E. coli* concentrations is also important. The NZ Drinking Water Standard recommends a Maximum Acceptable Value (MAV) of < 1 cfu/ 100 ml.

According to the long-term monitoring data, recurrent faecal contamination events with *E. coli* concentrations ranging between 1 and 150 cfu/100 ml have been detected over the last 20 years. Similarly, *E. coli* concentrations above the NZ Drinking Water and ranging between 2 and 73 cfu/100 ml have been recorded on the new monitoring bores. High concentrations were observed during the summer of 2017, when the water table was relatively high due to irrigation activity.

There are potentially multiple causes for these elevated concentrations, including sub-standard bore head protections, border dyke irrigation over macro-porous soils, or/and sub-standard effluent management practices. While some measures have already been undertaken to communicate the microbial contaminations, further analysis such as PCR markers, faecal sterol analysis, and fluorescent whitening agent analysis will be required to determine the predominant sources of contamination.

In addition, further understanding of groundwater pathways and processes in the Lower Waitaki Plains is required for the appropriate management of the aquifer water quality. In particular, investigations highlighting the interactions between surface water and groundwater, identifying sources of the nitrate, and an assessment of the current state of the groundwater eco-dependent systems and conditions to preserve their values are needed. The development of an integrated flow and transport model (soil/groundwater/surface water), which could be based on previous work (ORC, 2011), would provide a better understanding of the aquifer processes and enable optimisation of the water resource management.

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1. Context of the study

In June 2000, Otago Regional Council (ORC) published "Lower Waitaki Groundwater Investigation", which summarised the findings of a year long investigation carried out in 1999. The purpose of this study was to enhance the level of knowledge for the quality of groundwater and Lower Waitaki Alluvium.

Based on the information collected, the main recommendations of this report were to:

- Improve the knowledge of the groundwater quantity by:
  - ✓ Developing a better comprehension of the geological context, knowledge of the aquifer geometry and characteristics;
  - ✓ Installing groundwater level and irrigation systems monitoring sites;
  - ✓ Developing a coupled soil moisture/groundwater model.
- Continue and complement groundwater quality monitoring by:
  - ✓ Monitoring selected existing sites seasonally;
  - ✓ Adding sites slightly up gradient of the contaminated areas;
  - ✓ Further investigating the source of faecal contamination;
  - ✓ Analysing for pesticides and herbicides in bores used for horticultural purposes.
- Analyse management practices in the vicinity of the sites with elevated contaminants and ensure that the bore head protection is upgraded.
- Encourage community awareness and education regarding quantitative and qualitative groundwater issues.

In 2014, as part of mediation on plan change 6A (Water Quality) of the Regional Plan: Water (the Plan), the ORC, Waitaki Irrigators Collective Limited, and Lower Waitaki Irrigation Company Limited (LWIC) signed a Memorandum of Understanding (MOU). It was agreed in the MOU that the ORC will monitor trends of groundwater quality in the Waitaki Plains and share this information with LWIC and WIC.

The ORC has been monitoring the groundwater quality on five groundwater bores since July 2016. The purpose of this report is to present the analysis of the data collected up to January 2018, with a focus on nitrate-nitrogen (nitrate-N) and *Escherichia coli* (*E. coli*) concentrations.

This report summarises:

- The main features of the area of the study;
- The investigations carried out;
- The results and interpretation of the data collected between July 2016 and January 2018;
- The recommended additional investigations.

2. Setting and background information

The Lower Waitaki Plains are located near the east coast of north Otago, approximately 20 km north of Oamaru (Figure 2-1).

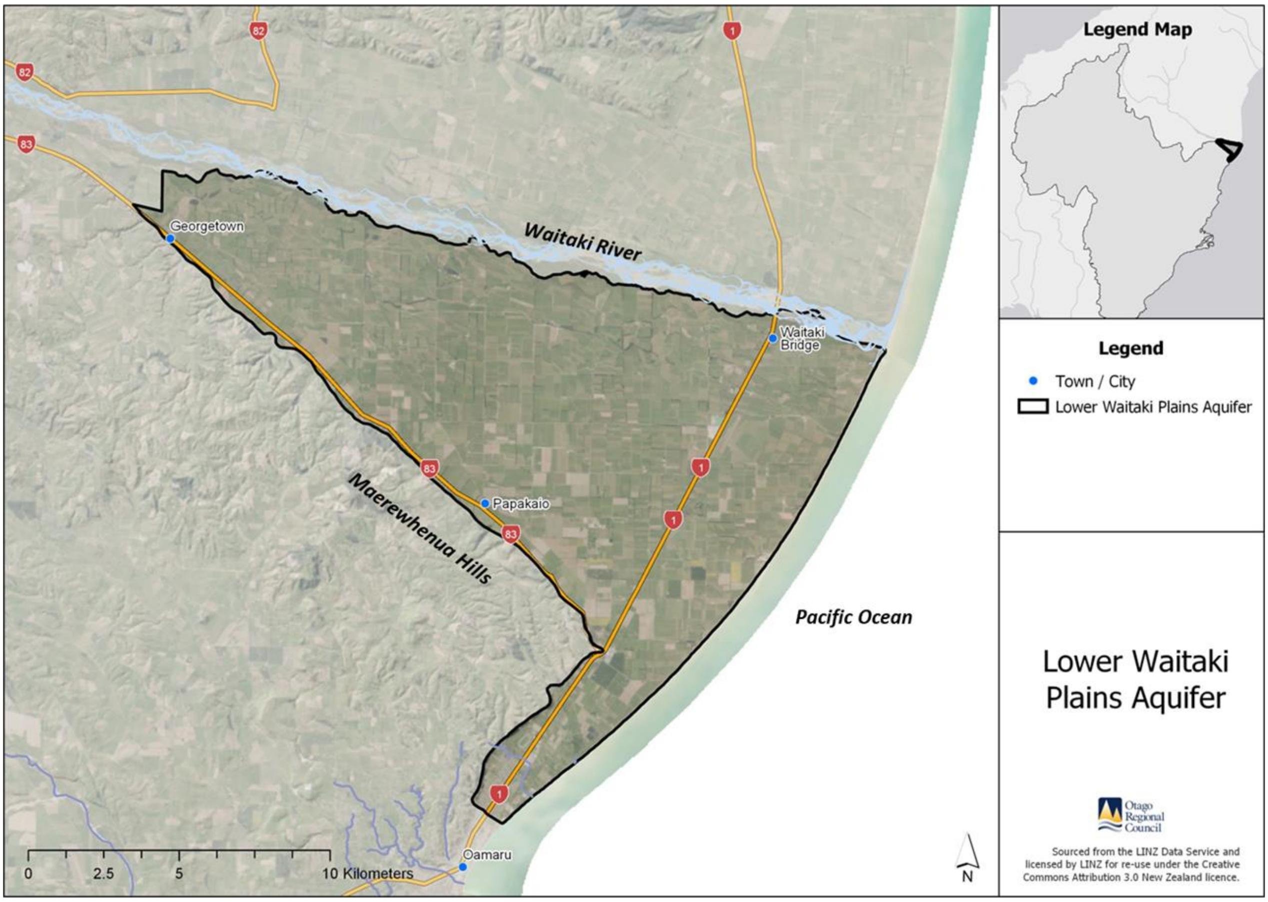

Figure 2-1 Location of the Lower Waitaki Plains and Aquifer

The following sections will introduce the main features of the area regarding the climate, the main surface water bodies, the soils, the groundwater protection zones, the land use, the irrigation, and the geology.

2.1. Rainfall and evaporation

The knowledge of local rainfall and evaporation patterns is a major input to characterise the aquifer water balance. Further details about this water balance are provided later in the report (section 3.6). Two local sites, Oamaru Airport AWS and Windsor EWS NIWA stations, provide rainfall and evaporation records to determine the Lower Waitaki Plains climate conditions.

Monthly rainfall and evaporation totals averaged for Windsor EWS station, located approximately 18.4 km from Hilderthorpe, and monthly rainfall totals averaged for Oamaru Airport AWS station, located over the lower part of the Waitaki Plains are summarised in Table 2.1.

Table 2.1 Average rainfall and evaporation totals for the the closest LWP climate stations

	Windsor EWS (November 2000 - March 2018)		Oamaru AWS (April 1982 - March 2018)
Month	**Potential Evaporation* (mm)**	**Rainfall (mm)**	**Rainfall (mm)**
January	121.9	50.2	50.4
February	92.6	48.1	47.6
March	75.1	33.6	44.4
April	43.1	43.6	41
May	26.2	49.3	48.7
June	20.4	40.6	42.8
July	21.7	39.1	40.7
August	31.7	43.3	54.1
September	58.1	26.8	31.8
October	83.6	43.2	39.4
November	105.8	43.3	45.1
December	119.0	55.8	50.6
Annual	**799.2**	**516.9**	**486.2**

* *estimated with Penman's potential evaporation equation*

Based on the data from Oamaru Airport AWS station, the mean annual rainfall for the Lower Waitaki Plains is approximatively 486 mm with an average monthly fall of 45 mm.

The variation in monthly rainfall is minor, with August being the wettest month (average of 54 mm) and September the driest (average of 32 mm).

The rainfall data on Windsor EWS station are quite similar (monthly average of 43 mm), however December is the wettest month for this site.

Potential evaporation exceeds the rainfall from September to March (highlighted in blue in Table 2.1), with more than twice the rainfall values during December and January. This indicates that without the irrigation scheme the area would be naturally dry, with extensive soil moisture deficits.

2.2. Hydrology

In the study area, the main hydrological features are the Waitaki River along the northern margin and creeks flowing from the Maerewhenua Hills in the south, as shown in Figure 2-2.

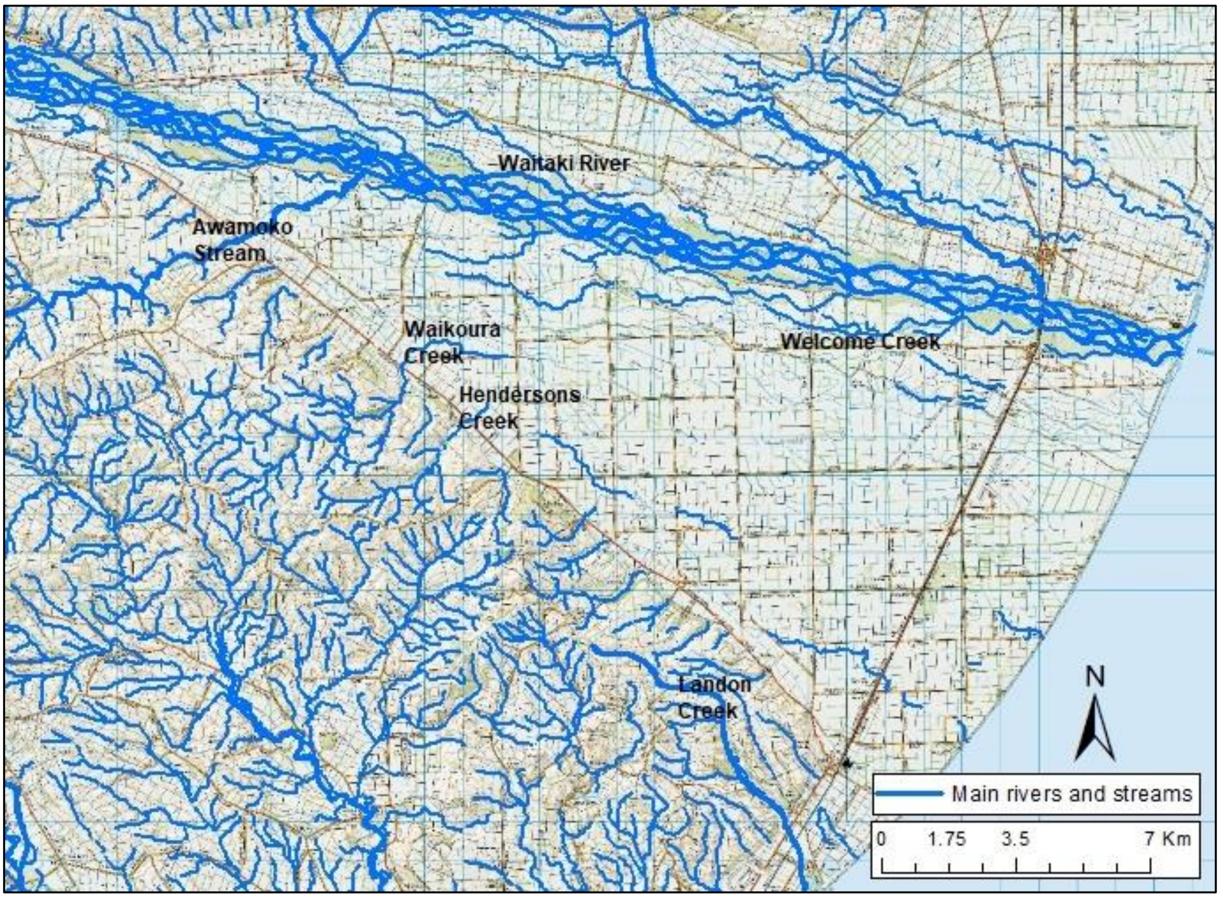

Figure 2-2 Main surface water bodies located near the Lower Waitaki Plains area

Welcome Creek is the major spring outflow of the aquifer (detailed location map provided in Appendix A). The Awamoko, and Waikoura–Henderson streams flow along a southwest - northeast direction, across the area. However, they are generally dry as flow from these streams leaks to the aquifer within 100 to 300 metres of their confluence with the plain.

2.3. Soils

For the purpose of the study, the emphasis has been given to the drainage characteristics of the soils, as the aquifer vulnerability will be highly linked to this feature.

Figure 2-3 outlines the main soil drainage categories in the Lower Waitaki Plains, with globally three main soil types:

- Well drained soils mainly made of (potentially gravelly and stony) sandy loams, covering approximately half of the plains in the north along the Waitaki River margin (such as Rangitata sandy loam on the lower terraces near the river, and Stewart silty loam on the higher terraces);
- Moderately well drained soils mainly made of silty loams (intermediate soils such as Ngapara and Darnley silty loams), covering almost the southern half part of the plains;
- Imperfectly drained soils mainly made of silty loams (heavier soils such as Pukeuri silty loam), encountered in a limited number of places over the plains such as Hilderthorpe and Pukeuri areas.

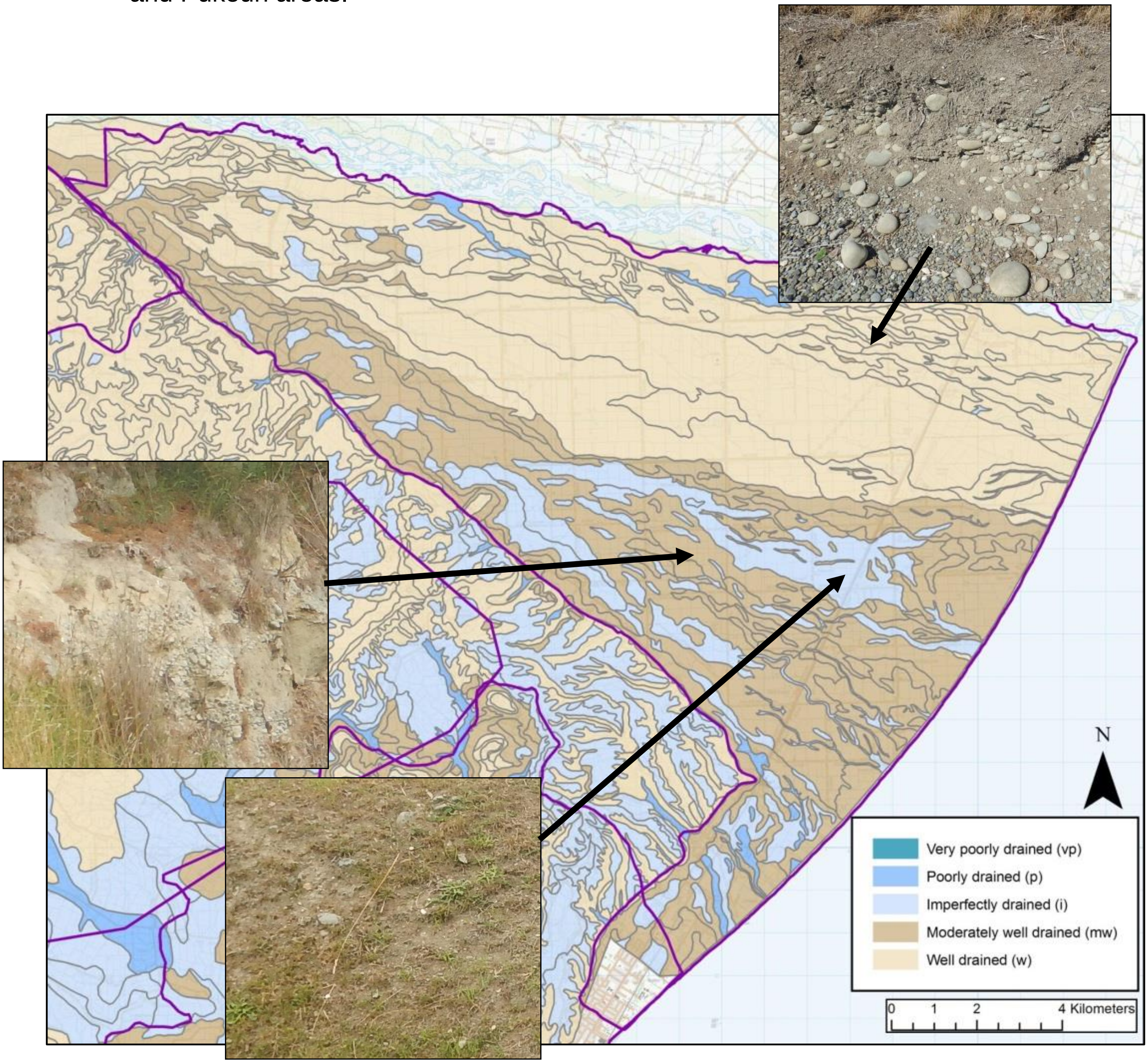

Figure 2-3 Soil drainage categories in the Lower Waitaki Plains (From Lancare Research S-Maps)

Soil drainage plays an important role in determining the aquifer vulnerability, as well as the suitability of the land for various land use practices, as shown in the following sections.

2.4. Aquifer Protection zone

Recognising the low water holding capacities of soils, a large proportion of the lower Waitaki Plain is listed as a groundwater protection zone in the Regional Plan: Water (C-series map).

The delineation of this protection zone, based on the aquifer vulnerability also considers the depth of the water table and is shown in Figure 2-4.

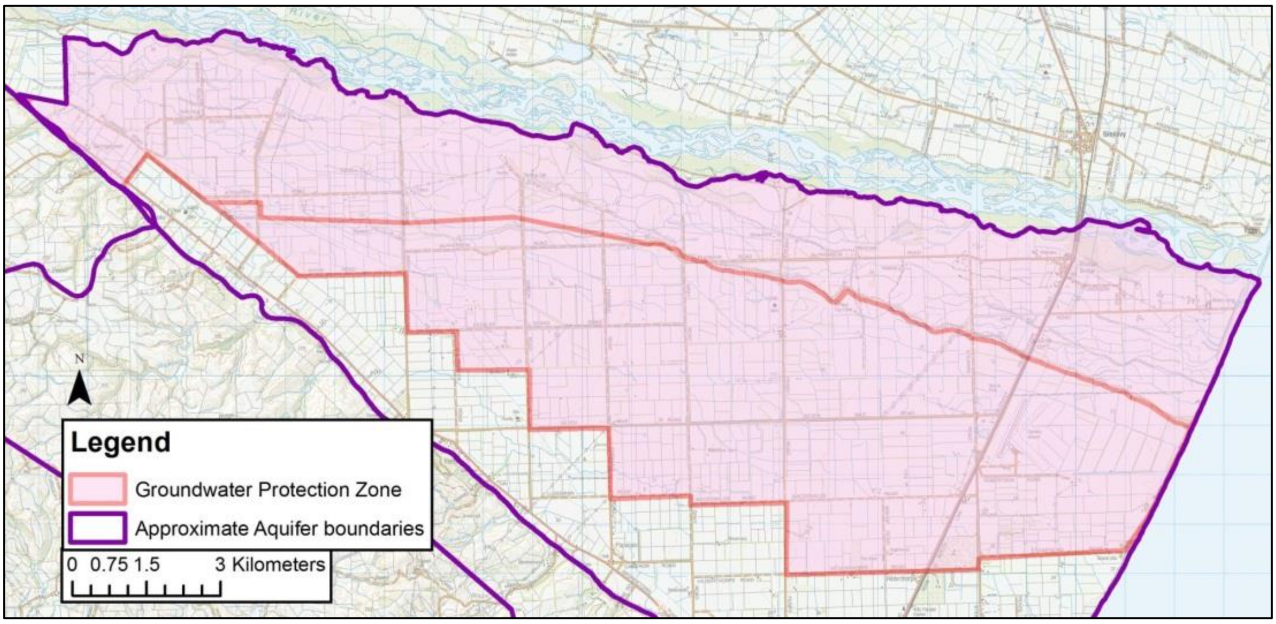

Figure 2-4 Groundwater Protection zone as delineated in the Regional Plan : Water

2.5. Land use

With approximately 54 dairy farms milking around 36,000 cows, dairy farming is currently the predominant activity in the Lower Waitaki Plains (covering approximately 70% of the area in 2015).

The development of this land use is linked to the commissioning of the Lower Waitaki Irrigation Scheme. Table 2.2 presents the evolution of the main land uses over more than 45 years and Figure 2-5 gives an idea of their distribution through the plains in 2016.

Table 2.2 Changes in land use in the Lower Waitaki Plains since 1969 (from LWIC 2018)

Landuse types	Area in hectares				
	1969	1984	1994	1998	2015
Pastoral	13,950	10,500	5,000	3,000	5,500
Cropping	1,975	3,000	3,000	2,000	500
Dairying*	75	2,500	8,000	11,000	14,000
Total	**16,000**	**16,000**	**16,000**	**16,000**	**20,000**

* *Dairy support and runoff blocks have been categorised as pastoral land use when there was a clear distinction*

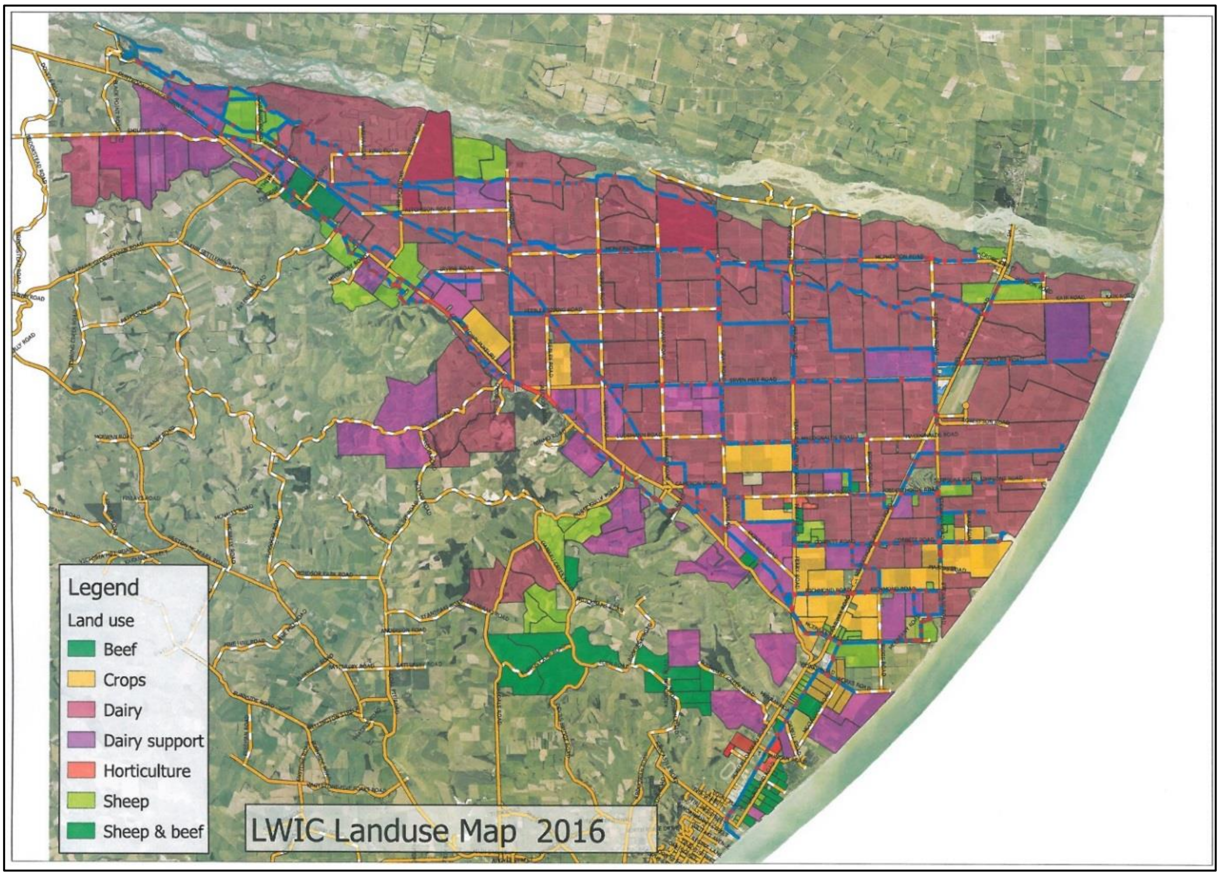

Figure 2-5 Land uses over the Lower Waitaki Plains (from LWIC 2016)

As presented in section 2.3, most of the soils along the river terraces and in the northern half of the plains are free draining. This means that they are not suited to cereal cropping and horticulture activities, which require heavier soils and are located in the southern margin of the valley. As a result, intensive pastoral farming now occupies much of the northern plain area.

2.6. Irrigation

The construction of the Lower Waitaki Irrigation Scheme started in November 1970 and was completed in March 1982. The whole scheme originally irrigated 16,000 hectares and served 170 farms.

The Scheme draws water from the Waitaki River at Black Point at the upper edge of the Waitaki Plain and a managed water race network delivers this water through the lower plains and down to Oamaru township for agricultural, commercial, and domestic uses.

The Scheme distributes approximately 1.2 million m^3 per day at peak operation. Table 2.3 shows the breakdown of the consented volumes and different water uses.

Table 2.3 Lower Irrigation Scheme Water use breakdown (from LWIC February 2018)

<table>
<tr><th colspan="3" rowspan="2"></th><th colspan="2">Average volumes for peak season</th></tr>
<tr><th>Instantaneous rate (l/s)</th><th>Daily rate (m^3 / 24h)</th></tr>
<tr><td colspan="3">LWIC consented Take</td><td>18,780</td><td>1,622,592</td></tr>
<tr><td colspan="3">LWIC physical take (infrastructure restrictive)</td><td>17,200</td><td>1,486,080</td></tr>
<tr><td rowspan="12">Water distributed</td><td rowspan="2">Irrigation</td><td>Border dyke</td><td>7,000</td><td>604,800</td></tr>
<tr><td>Spray</td><td>6,000</td><td>518,400</td></tr>
<tr><td colspan="2">Sub-total</td><td>13,000</td><td>1,123,200</td></tr>
<tr><td rowspan="6">Commercial and Domestic</td><td>WDC Redcastle Rd</td><td>360</td><td>31,104</td></tr>
<tr><td>Awamoko</td><td>6</td><td>518</td></tr>
<tr><td>Alliance</td><td>150</td><td>12,960</td></tr>
<tr><td>Oamaru Shingle</td><td>120</td><td>10,368</td></tr>
<tr><td>Road Metals co</td><td>120</td><td>10,368</td></tr>
<tr><td>Stock water estimated</td><td>200</td><td>17,280</td></tr>
<tr><td colspan="2">Sub-total</td><td>956</td><td>82,598</td></tr>
<tr><td colspan="2">Total</td><td>13,956</td><td>1,205,798</td></tr>
<tr><td colspan="3">Difference between Physical Take and Total of water distributed (losses: by wash, leakage and evaporation)</td><td>3,244</td><td>280,282</td></tr>
</table>

According to the information provided by LWIC and summarised in Table 2.3, the main water use during the peak season is for irrigation with up to 75.5% of the volumes, followed by the volume linked to by-wash, leakage and evaporation losses which represents approximately 19% of the take. These losses are indicated in the previous table by the difference between the physical take volume and the actual volumes taken. The water supplied for commercial and domestic purposes only represents 5.5% of the water use.

Figure 2-6 Irrigation race and center pivot in the Lower Waitaki Plains (February 2016)

The Scheme delivers water to more than 200 shareholders and covers 20,000 hectares on the Lower Waitaki Plains. The water is supplied to the farm offtakes under gravity through a distribution network made up of 200 kilometres of open canals (see Figure 2-6) and 12.5 kilometres of siphons and pipework. The detail of the irrigation network is given in Figure 2-7. The current irrigation methods are spray and border dyke covering 11,000 and 9,000 hectares, respectively. Figure 2-8 provides details about the irrigation methods distribution in 2016.

Across the plain, there is a trend towards spray irrigation across all the different land uses, with an increasing number of farmers converting from border dyke methods to pivots, fixed grid and other spray irrigation methods.

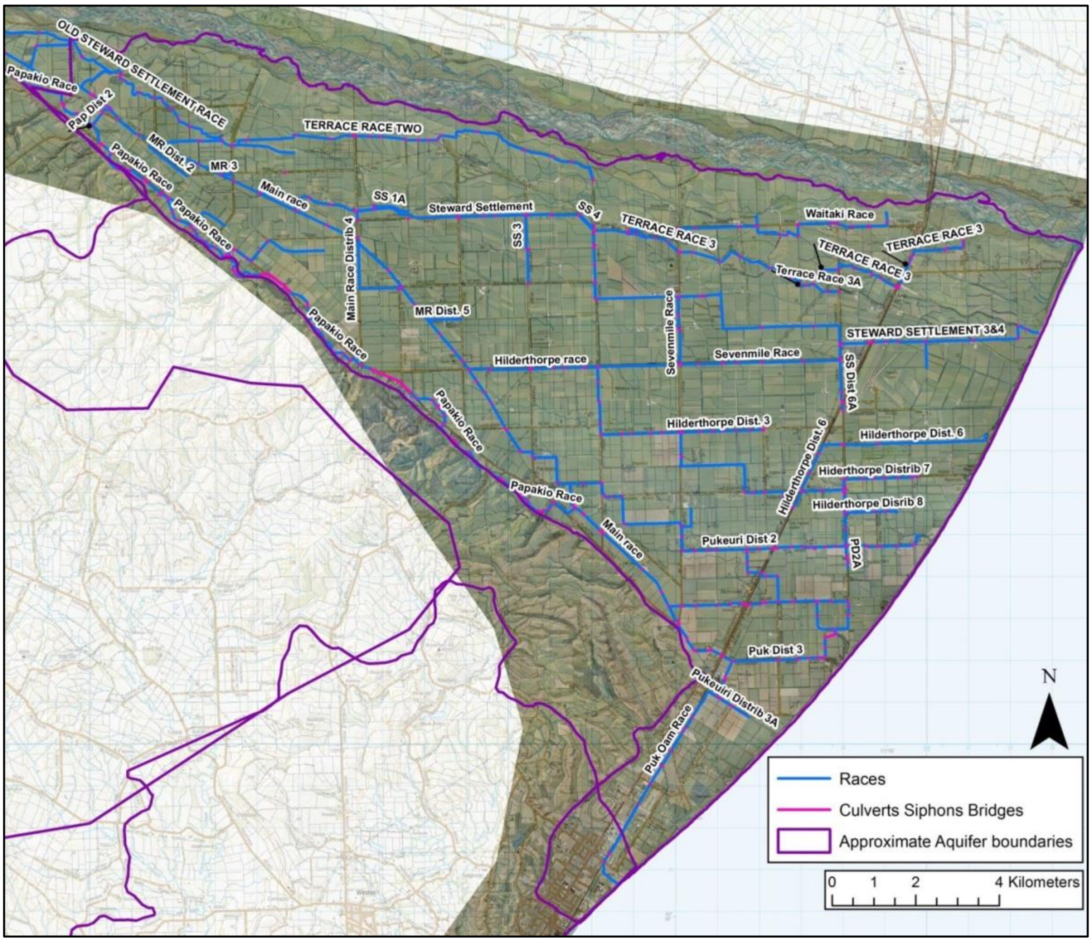

Figure 2-7 Lower Waitaki Irrigation Scheme (January 2018)

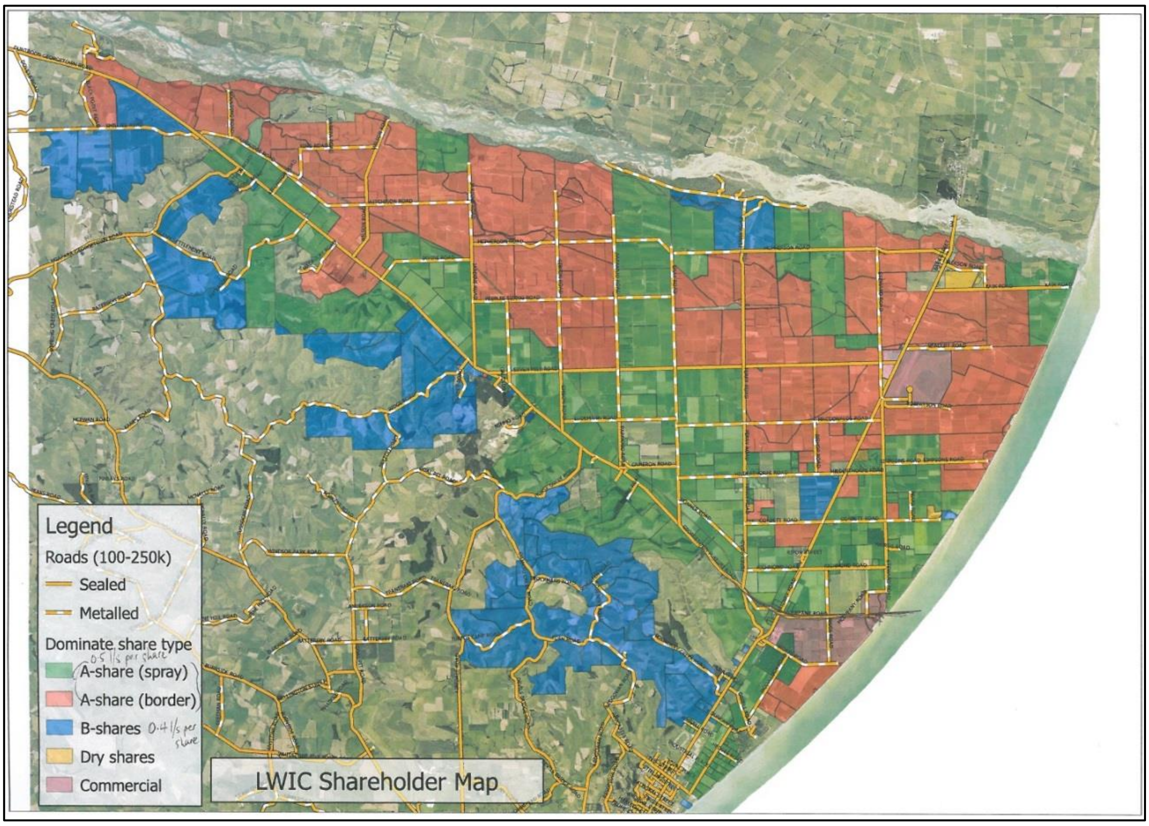

Figure 2-8 LWIC shareholder and irrigation uses and types (from LWIC 2016)

2.7. Geology

Figure 2-9 outlines the main geology units present in the Lower Waitaki Plains area:

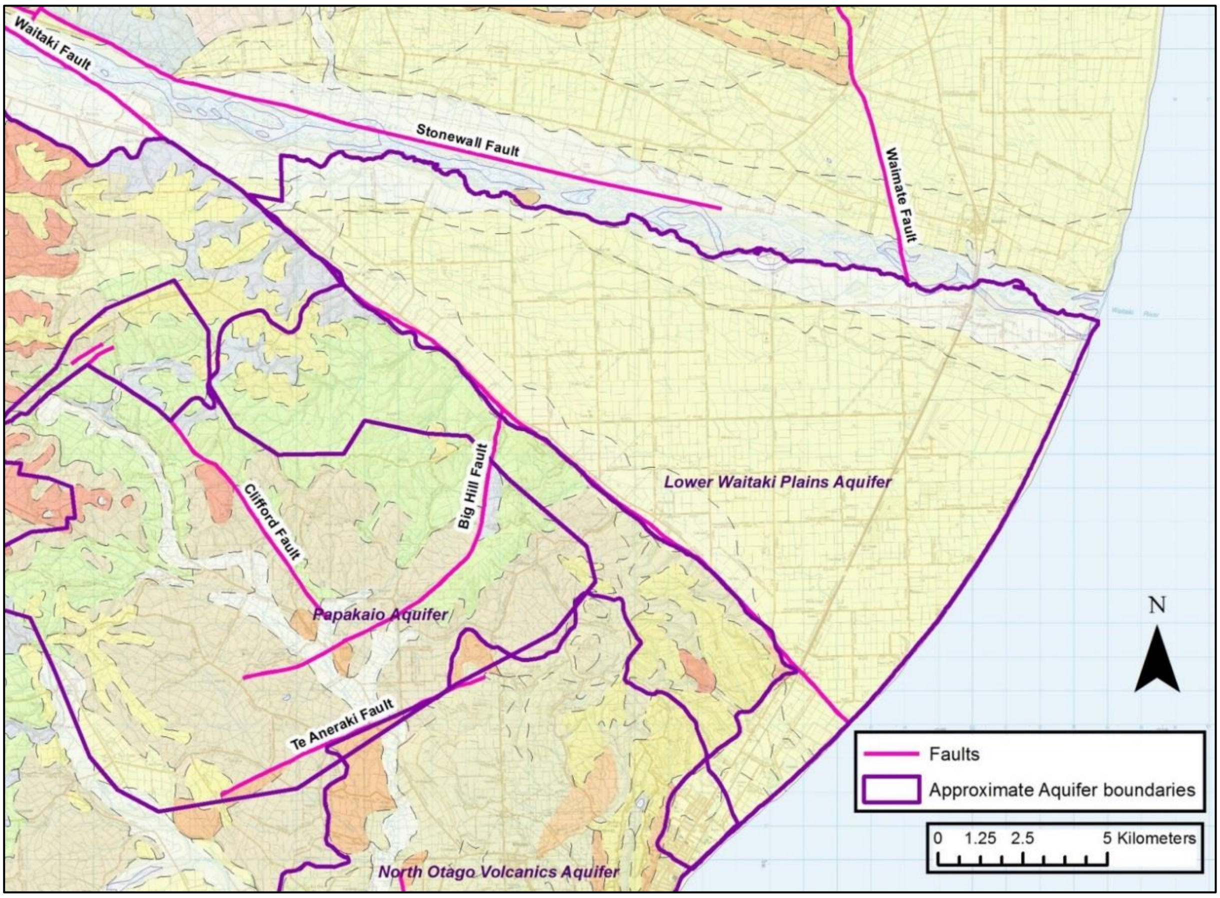

Figure 2-9 Geological Map of the Lower Waitaki Plains (extract from IGNS Q Map)

- Middle Pleistocene river deposits: moderately to highly weathered brown gravel in highly weathered sandy matrix, overlain by up to 3 loesses; clasts of greywacke.
- Late Pleistocene river deposits: gravel, sand and silt of low river terraces with patchy loess cover.
- Holocene river deposits: gravel, sand and mud of modern and postglacial flood plains.
- Late Cretaceous-Paleocene sedimentary rocks: non-marine quartz sand and conglomerate with clay matrix; lignite seams and carbonaceous mudstone; limonite and silica cemented.
- Basement (Eastern Province) metamorphic rocks: well foliated quartzo-feldspathic sandstone (greywacke) interbedded with mudstone (argillite).
- Paleocene sedimentary rocks: bryozoan grain-supported calcarenite with volcanogenic and marly layers.

The plains are mainly made of Holocene and Late Pleistocene unconsolidated gravels and sands. These materials were deposited during successive phases of glacial advance during the last glaciation period. Adjacent to the Waitaki River and to the coastline more recent sediments were deposited. The Plain alluvium are overlying consolidated Tertiary sedimentary rocks (known locally as the Rifle Butts Formation), consisting of various conglomerates, sandstones, siltstones and mudstones.

The Lower Waitaki Plains morphology is affected by the local fault system. Of particular interest, are the Stonewall Fault located along the Waitaki River and the Waitaki Fault located at the southern margin of the plains. The Maerewhenua Hills form the uplifted side of this northwest-southeast trending fault.

3. Groundwater hydrology

3.1. Aquifer geometry

The plain alluvial sand and gravel formation is hosting an unconfined aquifer, with potentially semi-confined conditions locally linked to the presence of silty and clayey lenses. The triangular shaped aquifer extends over approximately 19,793 ha, from a narrow stretch of land near Black Point to its widest margin along the coast.

The purple line represented in Figure 2-9 delineates the approximate aquifer boundaries.

The vertical extent is more uncertain as only a few bore logs attest that the Tertiary sedimentary rock basement has been tapped during drilling.

Figure 3-1 sketches the topography of the area and gives the location of the drilled boreholes with their total depth.

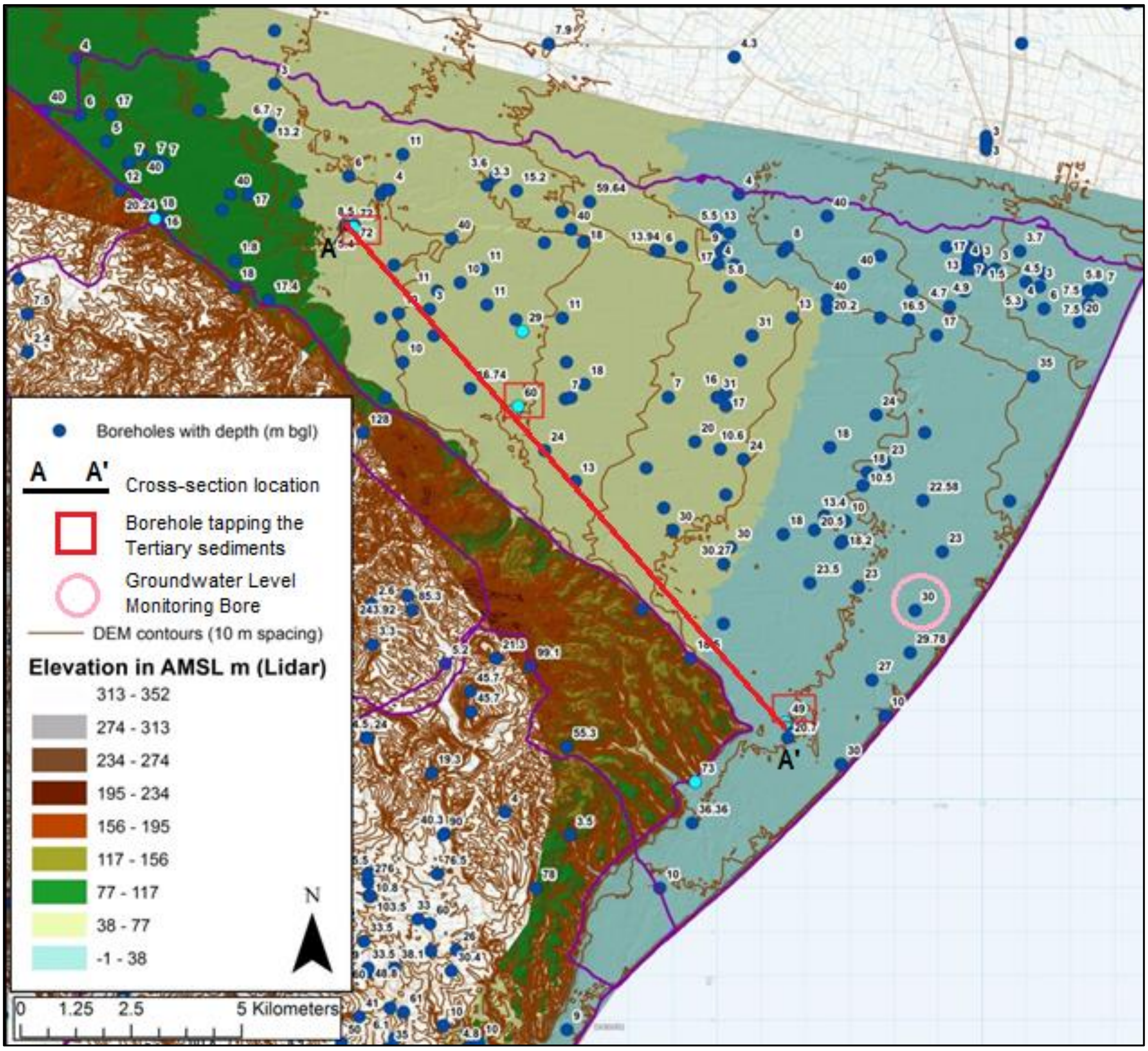

Figure 3-1 Boreholes location with their total depth (in m below ground level) and land elevation (in AMSL m)

According to the data currently available, only six borehole drillings reached the Tertiary sediments, described as “blue and grey silt and sandstone”, “grey brown siltstone”, “grey mudstone”, “mudstone” at depths ranging between 13 m and 45.80 m below ground level for the western and eastern parts respectively. These boreholes are highlighted in light blue in Figure 3-1.

The deepest bore with a borelog available is located near Pukeuri (to the east of the Maerewhenua Hills) and shows that below the “grey mudstone”, the “Oamaru stone” (limestone) was identified between 59 to 73 m below ground level.

An approximate vertical cross section of the plains, along a north west - south east segment AA’ located in Figure 3-1, and based on three bore logs, is presented in Figure 3-2.

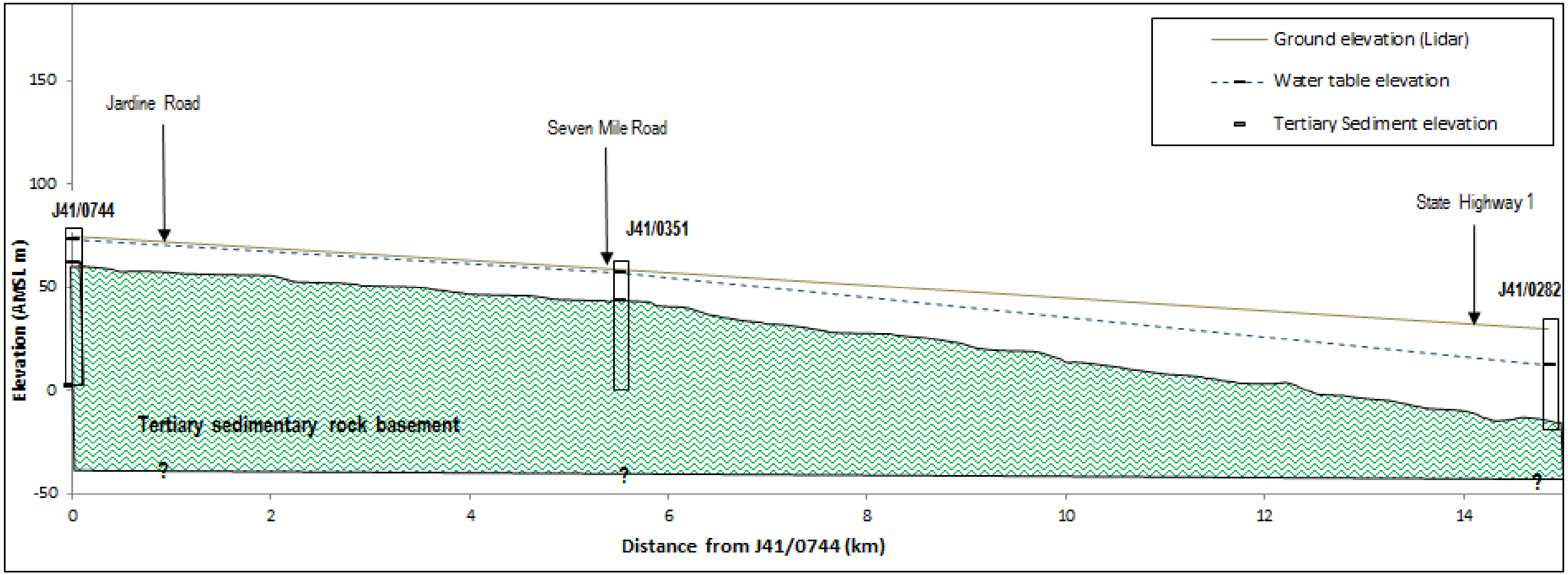

Figure 3-2 Approximate cross-section through the Lower Waitaki Plains from the west of Jardine Rd to the east of the State Highway Rd

3.2. Aquifer properties

There is only a small number of aquifer test results available for the Lower Waitaki Alluvium Aquifer, and their quality is relatively limited. The tests carried out are usually short duration tests, without observation piezometers. The inferred aquifer transmissivities and hydraulic conductivities are presented in Table 3.1.

Table 3.1 Summary of the aquifer tests carried out for bores located in Lower Waitaki Plains

Well Number	Date	Type	Transmissivity (m^2/d)	Saturated thickness (m)	Hydraulic conductivity(m/d)
J41/0584	31-May-06	Constant Rate	1,800	4.5	400
J41/0710	9-Dec-05	Constant Rate	> 1,000	1.3	> 770
J41/0721	3-Jun-08	Constant Rate	1,000 - 2,500	1.7	590 – 1,470
J41/0730	12-Nov-10	Constant Rate	1,200	25.3	47

The assessed transmissivity values range between 1,000 and 2,500 m^2/d, and the estimated hydraulic conductivity values are between 50 and 1,500 m/d. This is consistent with typical published values for permeability in gravel, which range from 100 to 1,000 m/d (Bouwer, 1978). A previous study (ORC, 2000) estimated higher transmissivity values up to 9,000 m^2/d along the river margins on the basis of saturated thickness values and hydraulic conductivities assigned from the bore logs and literature values.

The bore logs indicate that the alluvium consists of alternative lenses of poorly sorted gravel, sands and clay, hence the relative variability which can be observed over the area of the study. Throughout the plains, the gravels are “cleaner” in the recent deposits and the silt and clay fraction is more important in areas in the south and west near the hill margins.

The only aquifer test carried out with observation piezometers was on bore J41/0730 (Waitaki District Water Supply Bore). A specific yield of 0.02 was derived from this test, which is lower than general literature values, but seems reasonable when considering the potential vertical anisotropy linked to the type of deposits.

3.3. Aquifer hydrograph

ORC has been monitoring the groundwater levels in the Lower Waitaki Plains within the Dennisons Bore (J41/0377) since April 1997. This 30 m deep borehole is located in the south-eastern part of the aquifer (see Figure 3-1), approximately 1km from the coastline. Figure 3-3 presents its groundwater hydrograph over this period, with the daily rainfall recorded at the Oamaru airport rain gauge (NIWA AWS Station).

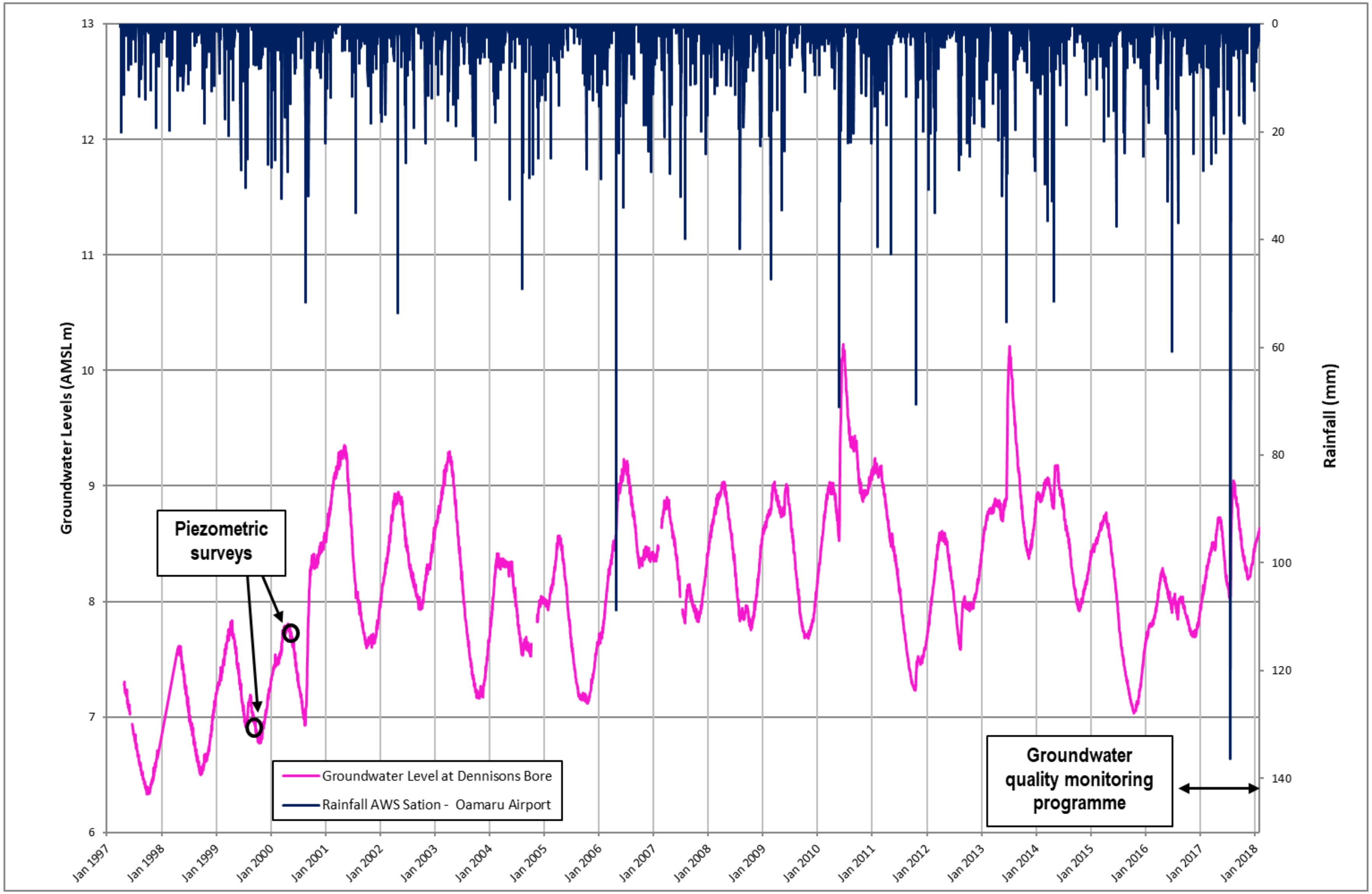

Figure 3-3 Groundwater levels at Dennisons Bore (Lower Waitaki Plains) and rainfall (Oamaru Airport) between April 1997 and January 2018

The groundwater hydrograph shows the following seasonal cycles:

- A gradual recovery and the highest levels generally observed during the summer months, when irrigation is significantly recharging the aquifer. The peak months are usually March or April. Rare spikes have been occurring over the winter (July 2013 and 2017) and might be linked to high recharge events;
- A gradual recession and the lowest levels usually observed during the winter months, with the greatest depressurisations occurring near the end of the winter in September and October.

A global upward trend of the groundwater levels is noticeable from 1997 to 2002, as a result of the aquifer re-equilibration to the new recharge dynamics imposed by the irrigation scheme. A more stable trend has been observed since; however, the hydrograph cycles seem less regular since 2010. The “usual” seasonal variation between summer and winter is approximately 1.5 m but can reach almost 2 m some years, for example in 2011.

Regarding the absence of stage monitoring sites within the lower part of the Waitaki River and the location of the current groundwater level monitoring bore (approximately 9km from the Waitaki River and on the lowest margin of the aquifer along the coast), the interactions between the aquifer and the river are not clearly captured.

3.4. Water table elevation

The latest groundwater level surveys in the Lower Waitaki Plains were carried out over 20 bores between September 1999 and May 2000.

The plots of depth to the water table (Figure 3-4 and Figure 3-5) indicate that the groundwater is located at depth between 1 m and 15 m below ground level (BGL), with the shallowest water table measurements along the river margin and the deepest ones near the coast.

These shallow levels extend further towards the central area of the plains during the summer, as a result of the irrigation recharge.

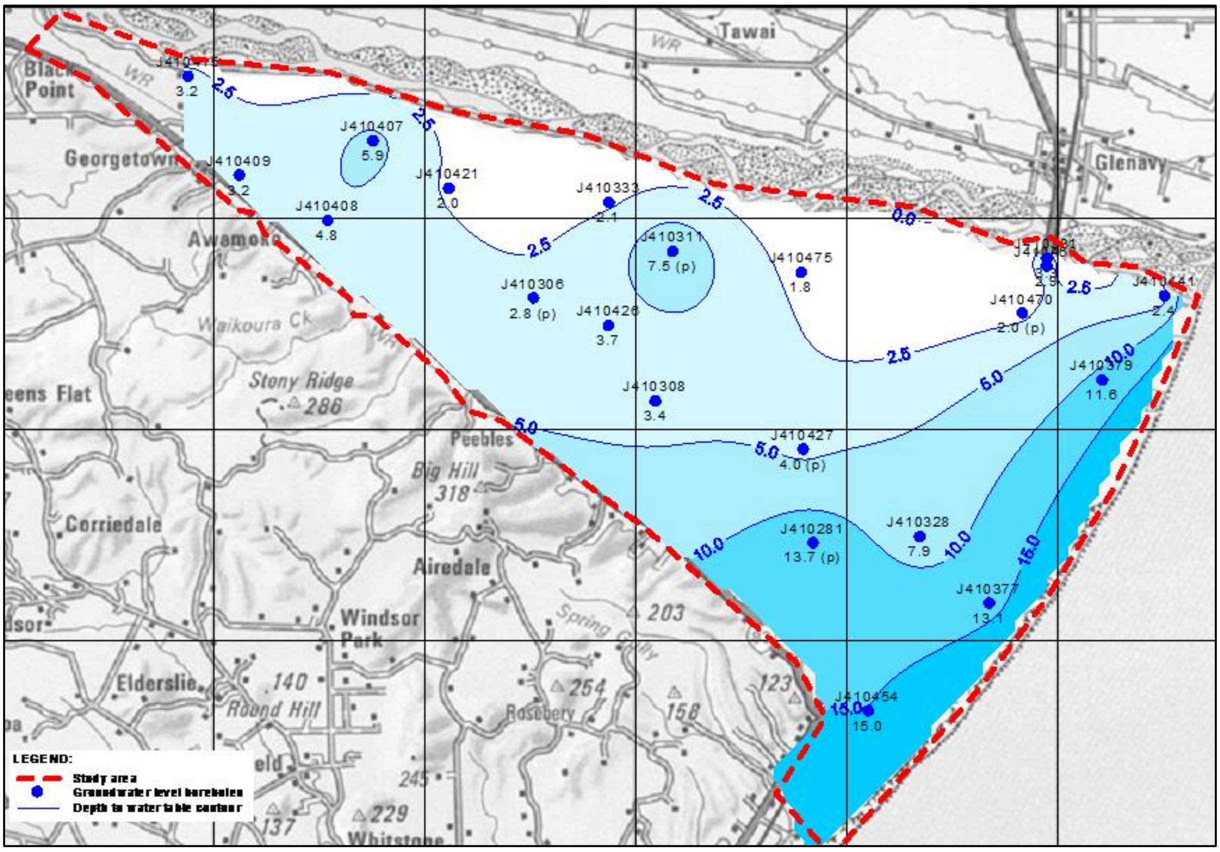

Figure 3-4 Depth to water table 14-15 September 1999 (ORC, 2000)

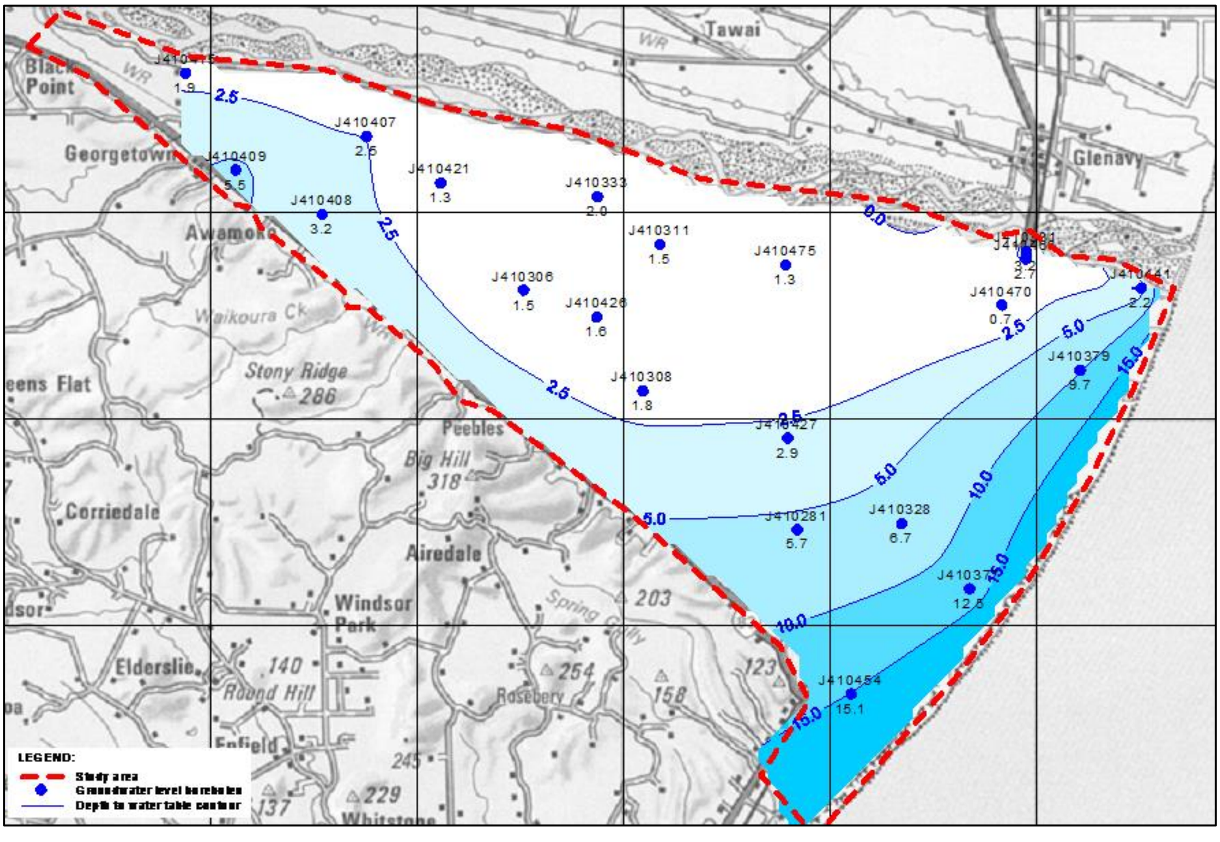

Figure 3-5 Depth to water table 22-23 February 2000 (ORC, 2000)

Between June 2016 and December 2017, monthly measurements of depths to the water table have been collected on three boreholes (J41/0442, J41/0571 and J41/0576, location provided in Figure 4-4). The depths to groundwater were between 2.2 m BGL near the river margin and 6.7 m BGL near Hilderthorpe, with fluctuations ranging between 0.6 and 1.4 m.

3.5. Groundwater flow patterns

On the basis of the groundwater level measurement campaigns carried out between September 1999 and May 2000 and mentioned previously, piezometric (potentiometric) maps have been prepared and are presented in Figure 3-6 and Figure 3-7. The groundwater level results were reduced to Otago datum (mRL Otago), which is 100 m above mean sea level (AMSL).

These maps display the water table heights collected, the piezometric contours and flow patterns inferred.

These contours are globally following the topography, with the highest levels near Black Point and the lowest ones along the coast line. The inferred flow directions are:

- From the southwest towards the northeast /and the Waitaki River in the upper part of the aquifer; and
- From the west towards the east - south-east /and the ocean in the lower part of the aquifer.

The hydraulic gradient (spacing between the groundwater level contours) is lower along the western part of the foot hills (recharge area) and higher in the southeast near the coast (discharge area). The average gradient is approximately 1:270 m, which is very similar to the land surface gradient.

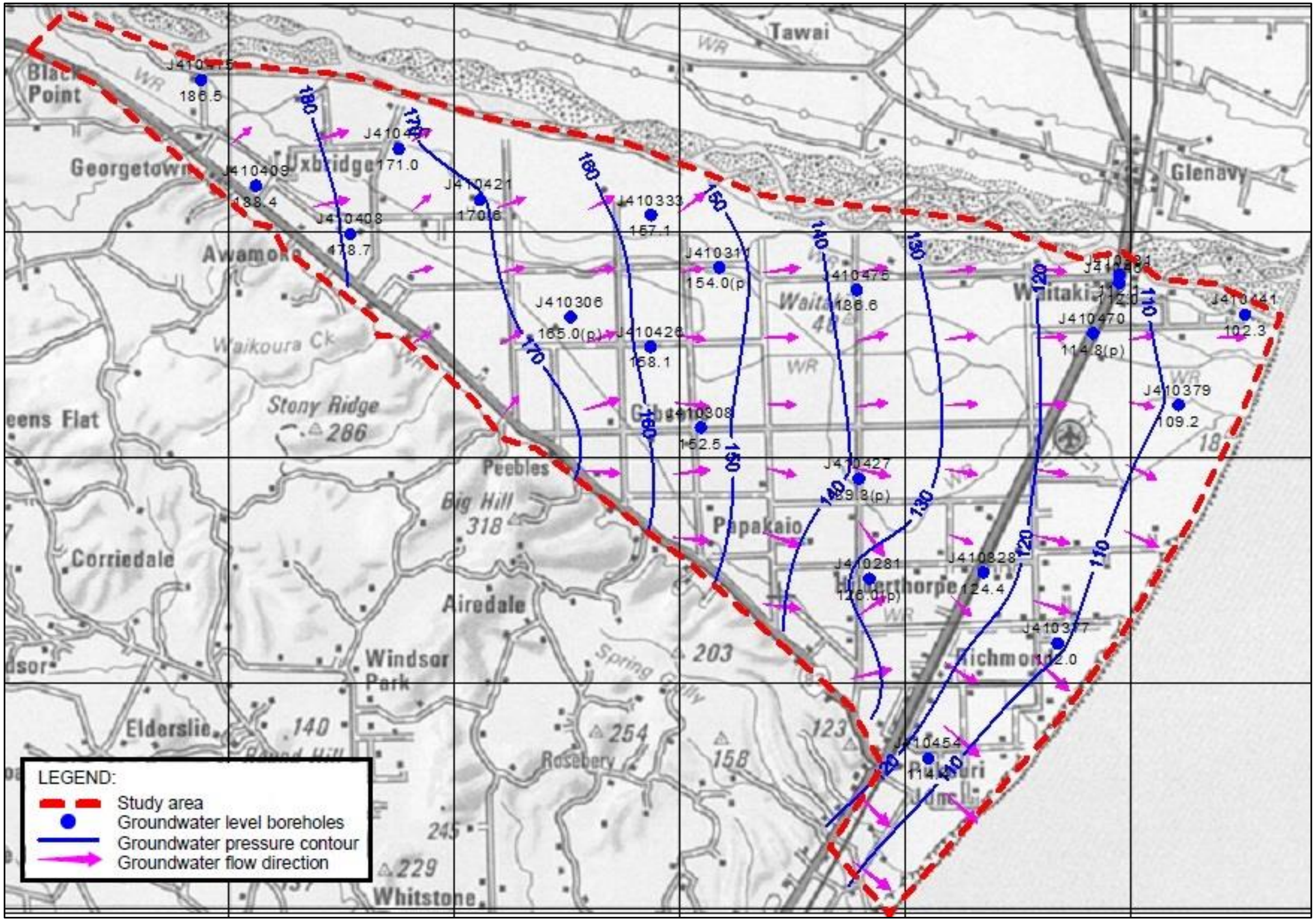

Figure 3-6 Piezometric Surface 14-15 September 1999 (ORC, 2000)

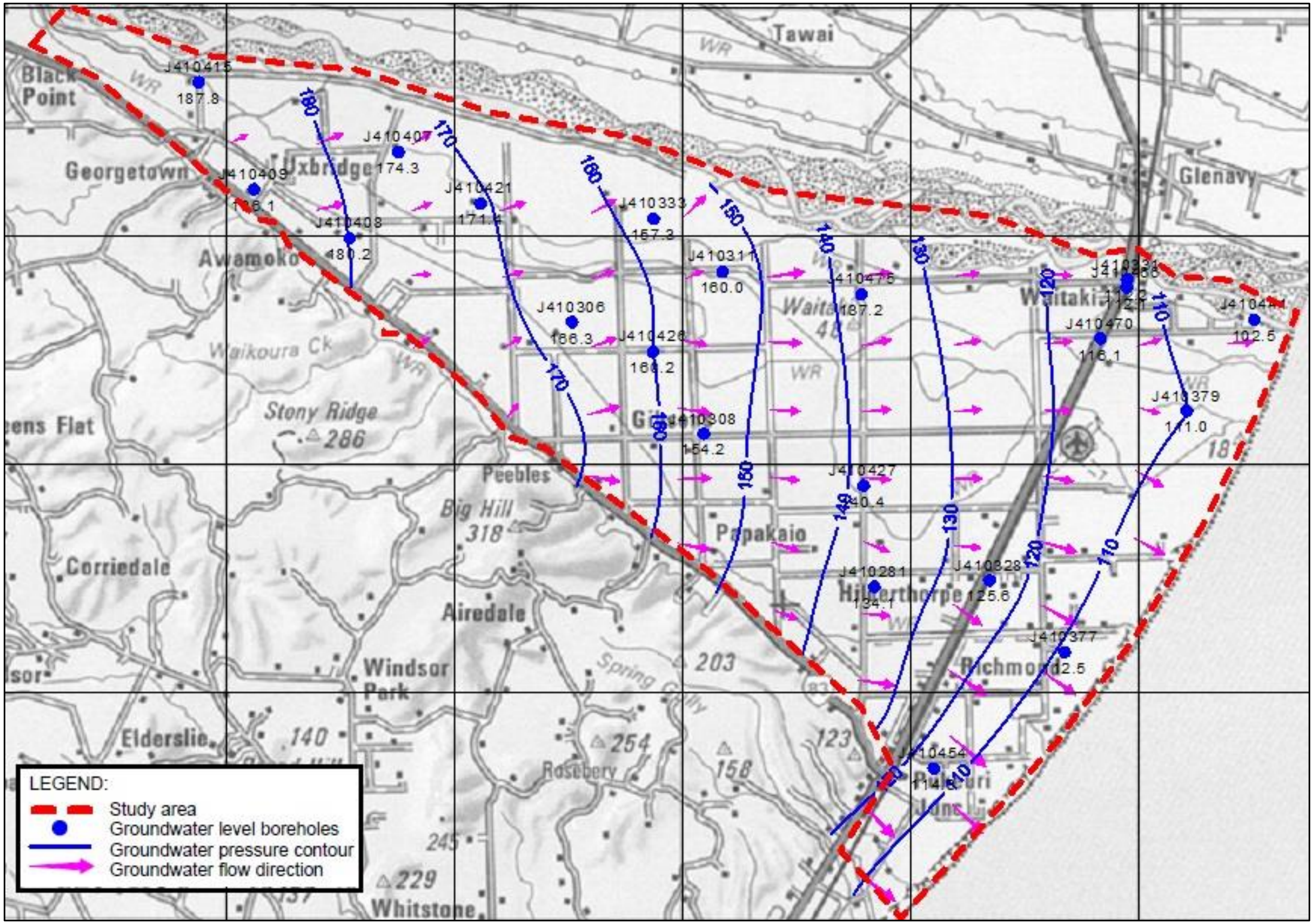

Figure 3-7 Piezometric Surface 2-3 May 2000 (ORC, 2000)

3.6. Aquifer water balance

Previous assessments of the aquifer water balance include the following components (e.g., ORC, 2000):

- As recharge sources: rainfall infiltration; infiltration from border dyke and spray irrigation waters; seepage from the water races, and streams traversing the plains, localised infiltration of creeks and streams at the edge of the Maerewhenua Hills, and flow from the Waitaki River (when the aquifer level is lower than the river stage);
- As discharge sources: offshore flow to the Pacific Ocean; baseflow to the Waitaki River and to Welcome Creek, and groundwater abstractions.

In 2011, a case study and some preliminary modelling works were carried out by the ORC on the Lower Waitaki alluvial aquifer. The mean annual water balance calculated from this study is presented in Table 3.2. A brief description of the hydraulic model is provided in Appendix B.

Table 3.2 Estimate of the aquifer recharge and discharge fluxes (ORC, 2011)

Waitaki Plain	Calc Fluxes (Mm^3/y)	Calc Fluxes (m^3/d)	Calc Fluxes (l/s)	Estimated Flux (l/s)
River Recharge	98.0	268,372	3,106	
Water Race Leakage	59.7	163,600	1,894	1,800
Land Surface Recharge (rainfall)	67.0	183,480	2,124	
Stream Recharge	7.0	19,210	222	
Total Recharge	**231.7**	**634,662**	**7,346**	
River Baseflow	75.7	207,380	2,400	2,500
Welcome Creek	61.2	167,746	1,942	1,220
Abstraction	2.3	6,407	74	
Drains	17.5	47,850	554	500
Offshore Flow	74.9	205,279	2,376	
Total Discharge	**231.7**	**634,662**	**7,346**	

The water balance proposed by the first study (ORC, 2000) was based on measured and estimated components for winter and summer conditions of 1999 to 2000, and somewhat differs from the water balance established during the modelling works (ORC, 2011).

Indeed, the latest study identified the main recharge source to be the Waitaki River, whereas the 2000 study identified the irrigation returns as the major input to the aquifer.

Therefore, the piezometric maps presented previously (Figure 3-6 and Figure 3-7 from ORC, 2000) do not show the Waitaki River as a significant recharge source. The comparison of the Waitaki River and groundwater levels using the Lidar data for a few points indicates that the water table level appears to be higher than the river level, and that the groundwater is discharging into the river during normal events (providing river baseflow).

It is therefore likely that the aquifer is recharged episodically by the Waitaki River during large flow events, especially in winter when the water table is at its lowest height and the river level elevated.

Another difference between the two approaches, is that the 2011 modelling study considered that the groundwater could seep through the drains present over the plains, which was excluded from the previous study (ORC, 2000).

3.7. Aquifer Allocation

The current allocation limit for the Lower Waitaki alluvial aquifer has been assessed as 50% of the mean annual recharge, on the basis of the recharge figures presented in Table 3.2. The allocation limit for the aquifer is therefore115.85 million m^3 per year (Mm^3/y).

Currently, there are 15 consented groundwater takes operating within the aquifer, and 3.1 Mm^3/y of groundwater is allocated. The irrigation scheme supplies the bulk amount of water through the area, sourced from surface water.

4. Summary of the bores investigations

4.1. Preliminary Investigations

Based on the ORC 2000 report, ten bores were selected for groundwater quality monitoring. On 28-29/04/2016, a site visit was carried out for each site and a water quality sample was collected from nine of these bores. The bore locations and the nitrate-N concentrations are provided in Figure 4-1. The field investigations have shown that most of the bores were difficult to sample (no direct access to sample water close to the bore head and especially before the pressure or storage tanks) and that many bores were not properly maintained and sealed (see Figure 4-2 and Figure 4-3).

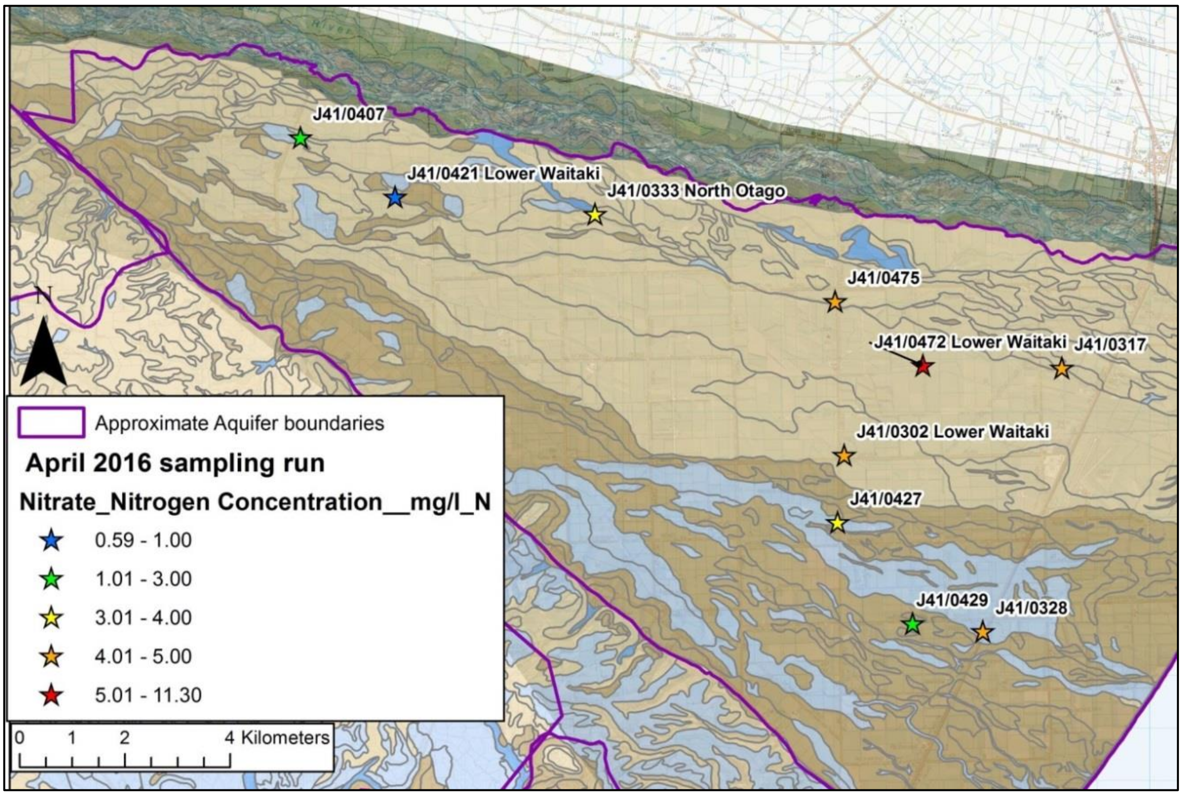

Figure 4-1 Location and Nitrate-N concentrations of the boreholes sampled in April 2016

Figure 4-2 Example of poorly maintained and unproperly sealed bores (April 2016)

Figure 4-3 Example of unsealed bore in dairy farm (April 2016)

4.2. Refined investigations

On 7 June 2016, a second site visit and sampling run was carried out on five bores. Bore logs and well construction details were available for the boreholes and they appeared to be more suitable for monitoring purposes. Installation of taps on the outlet pipe to ease the sampling process was required for a couple of sites.

These bores are mainly located within the lower part of the aquifer (Figure 4-4), where cumulative effects are more likely to occur, and where the highest nitrate concentrations have been measured in the past. The selection process has also considered the aquifer vulnerability in relation to the soil types and depth of the water table.

A final investigation was carried out on 22 June 2016, to identify an additional bore (J41/0442) to monitor the upper part of the aquifer (Figure 4-4).

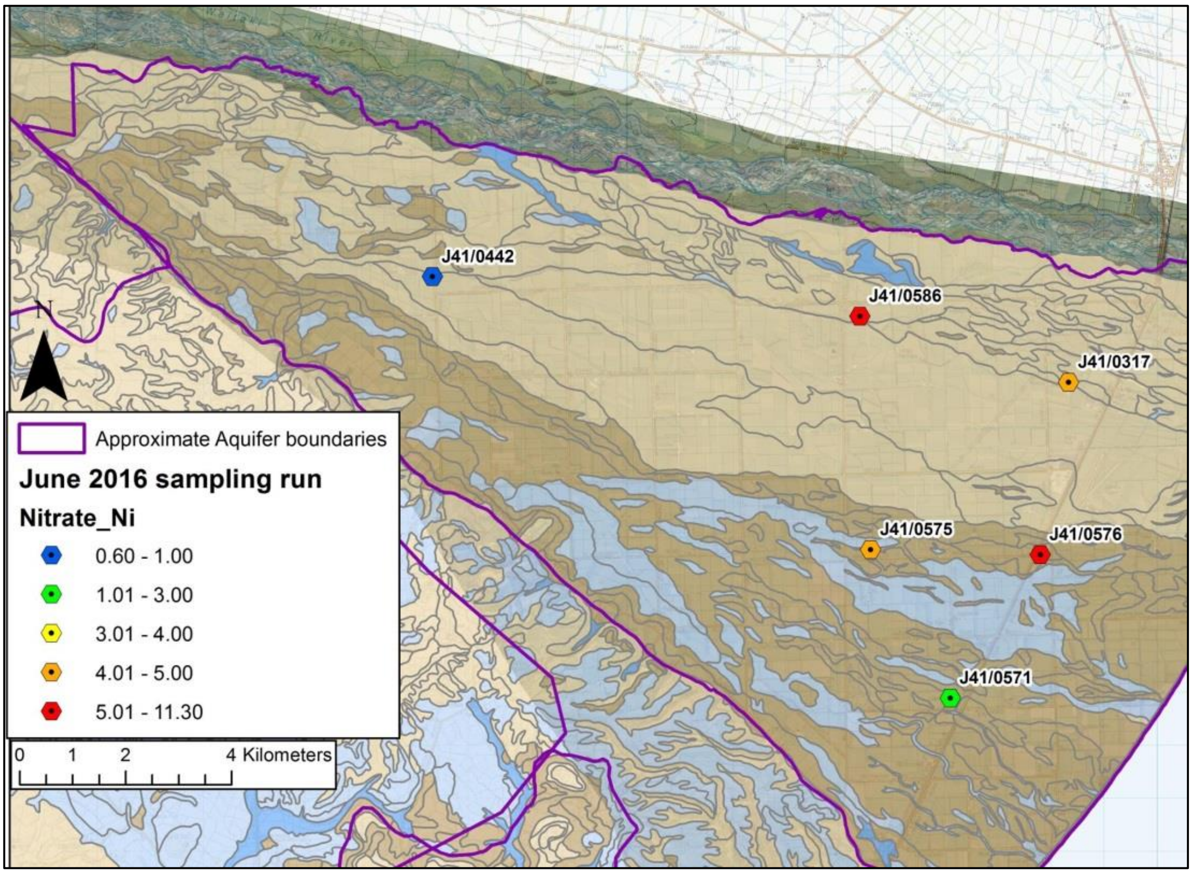

Figure 4-4 Location and Nitrate-N concentrations of the boreholes sampled in June 2016

4.3. Final selection for quality monitoring

After finalizing the bores selection, ORC acquired the landowners' authorisations to use the bores for continuous monitoring purposes and arranged a specialist to add sampling taps for the monitoring bores. Waterforce performed this task on 3 August 2016 on bores J41/0571, J41/0576 and J41/0586. This work was not necessary for J41/0442, where it was possible to sample by adding a valve close to the bore head (Figure 4-5).

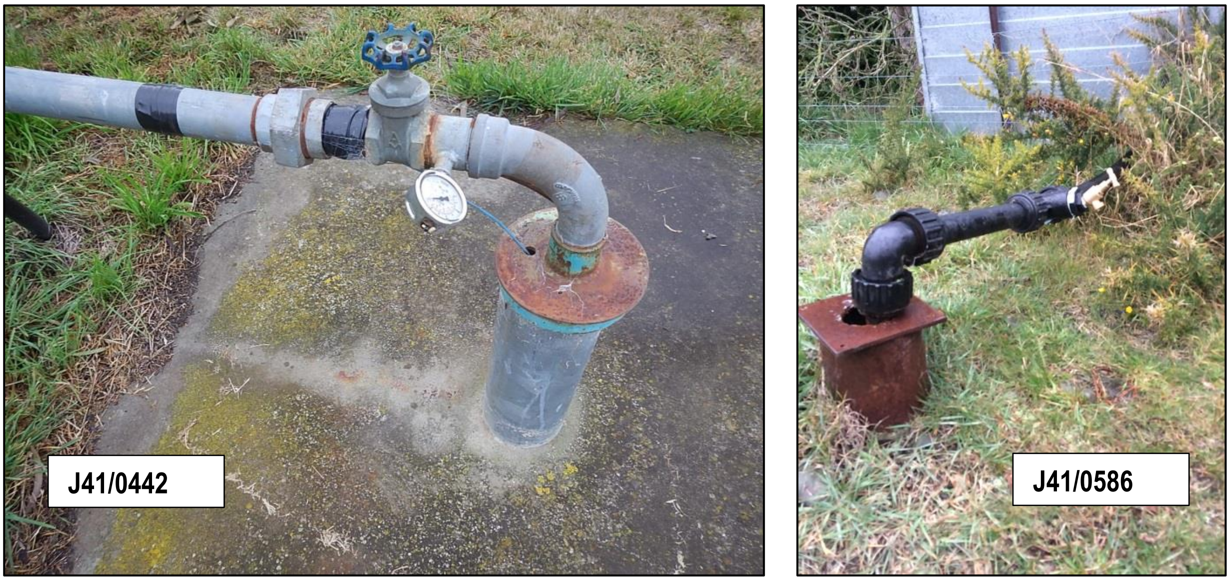

Figure 4-5 Example of bores selected for sampling purposes

A monthly sampling programme was launched in August 2016 on the bores listed in Table 4.1

Table 4.1 Characteristics of selected quality monitoring bores and long term SOE Bore

Well Number	Depth (m)	Diameter (m)	Groundwater Depth (m)	Drill Date	Location
J41/0442	11.4	0.15	4.1	3/06/1997	Jardine Rd
J41/0571	24.0	0.15	7.0	12/08/2002	RD 1H
J41/0576	23.0	0.20	13.0	5/12/2003	RD 5H
J41/0586	10.9	0.13	3.0	25/07/2003	Ferry Rd
SOE Site J41/0317	16.5	0.15	6.0	01/01/1983	Stewart Rd

The long-term state of the environment (SOE) quality monitoring bore J41/0317, sampled quarterly, adds complementary data to this selection and gives a broader context with its long term dataset.

5. Groundwater quality

The groundwater quality in the Lower Waitaki Plains has been monitored since June 1993 by five monitoring bores, however the sampling stopped for most of them apparently after the 2007 Groundwater Monitoring Review, and currently only J41/0317 is still operational.

Other samplings rounds have been carried out in September and December 1999, February and May 2000 (ORC 2000), and more recently in January and February 2013 on 15 bores.

This report does not intend to summarise all the historical water quality data collected so far for the Lower Waitaki alluvial aquifer, but will focus on:

- The April 2016 - January 2018 data set collected on the new bores selection and;
- The long-term data set collected on the SOE bore J41/0317.

A summary of the median values and number of analyses is provided in Table 5.1 and the complete results of these analyses are given in Appendix C.

Table 5.1 Median values measured between June 2012 - January 2018 for J41/0317 and June 2016 - January 2018 for the other sites

	Parameter	T	pH	EC	O_2	SiO_2	Ca	Mg	Na	K	HCO3	CO2	F	Cl	SO_4	NO_3	NH_4	DRP	Fe	Mn
	Unit	°C		µS/cm	mg/l															
J41/0317 TD=16.5 m SWL*=5.5 m	Median	13.2	6.8	153	5.66		15.0	3.3	7.2	2.0	44.0	10.0	(0.09)	4.7	13.2	4.3	0.011	0.029	0.014	(0.0007)
	Number of samples	19	19	19	10	-	19	19	19	19	19	19	2	19	19	19	5	19	13	3
J41/0442 TD=11.4 m SWL=3.4 m	Median	13.0	6.7	85	4.54	(7.2)	9.9	1.7	3.5	1.7	32.0	14.0	(0.08)	2.8	6.9	0.99	(0.008)	0.005	0.0215	(0.0008)
	Number of samples	13	13	13	12	1	13	13	13	13	13	13	1	13	13	13	2	13	12	3
J41/0571 TD=23.5 m SWL=6.0 m	Median	14.6	6.6	246	1.00	(22)	20.0	5.8	20.0	2.2	81.0	35.0	(0.18)	11.0	27.0	3.8	0.045	0.006	0.079	0.0170
	Number of samples	13	13	13	12	1	13	13	13	13	13	13	1	13	13	13	9	12	11	9
J41/0576 TD=23 m SWL**=13.0 m	Median	12.5	6.3	217	1.12	(18)	20.0	5.7	12.0	1.6	53.0	55.0	(0.1)	10.0	22.0	6.6	0.008	0.012	0.28	0.0045
	Number of samples	13	13	13	12	1	13	13	13	13	13	13	1	13	13	13	8	12	11	12
J41/0586 TD=10.9 m SWL=2.6 m	Median	12.5	6.5	187	3.35	(19.1)	17.0	4.4	11.0	1.6	56.0	32.0	(0.13)	5.5	17.0	5.7	(0.007)	0.008	0.0087	0.0009
	Number of samples	13	13	13	12	1	13	13	13	13	13	13	1	13	13	13	3	13	11	8

For each monitoring bore the total depth (TD) and mean static water level (SWL) in meter below top of the casing are provided over the monitoring period.
** The SWL value for J41/0317 is the mean value between Feb. 2002 - March 2007, **and value recorded after drilling for J41/0576*

5.1. Major ion composition

The major ions form the main constituents of the water. The concentration of the various constituents of the groundwater is controlled by the availability of elements in the soils and rocks through which the water has passed and various geochemical constraints and complex processes. An easy way to classify the groundwater types is to represent its composition on a Piper Diagram, which plots the percentage constituent of the major ions onto two trilinear diagrams: one for the cations and one for the anions. The samples are then projected from these trilinear diagrams onto a diamond-shaped field for classification.

Figure 5-1 provides a classification of the five groundwater sites and of one river site.

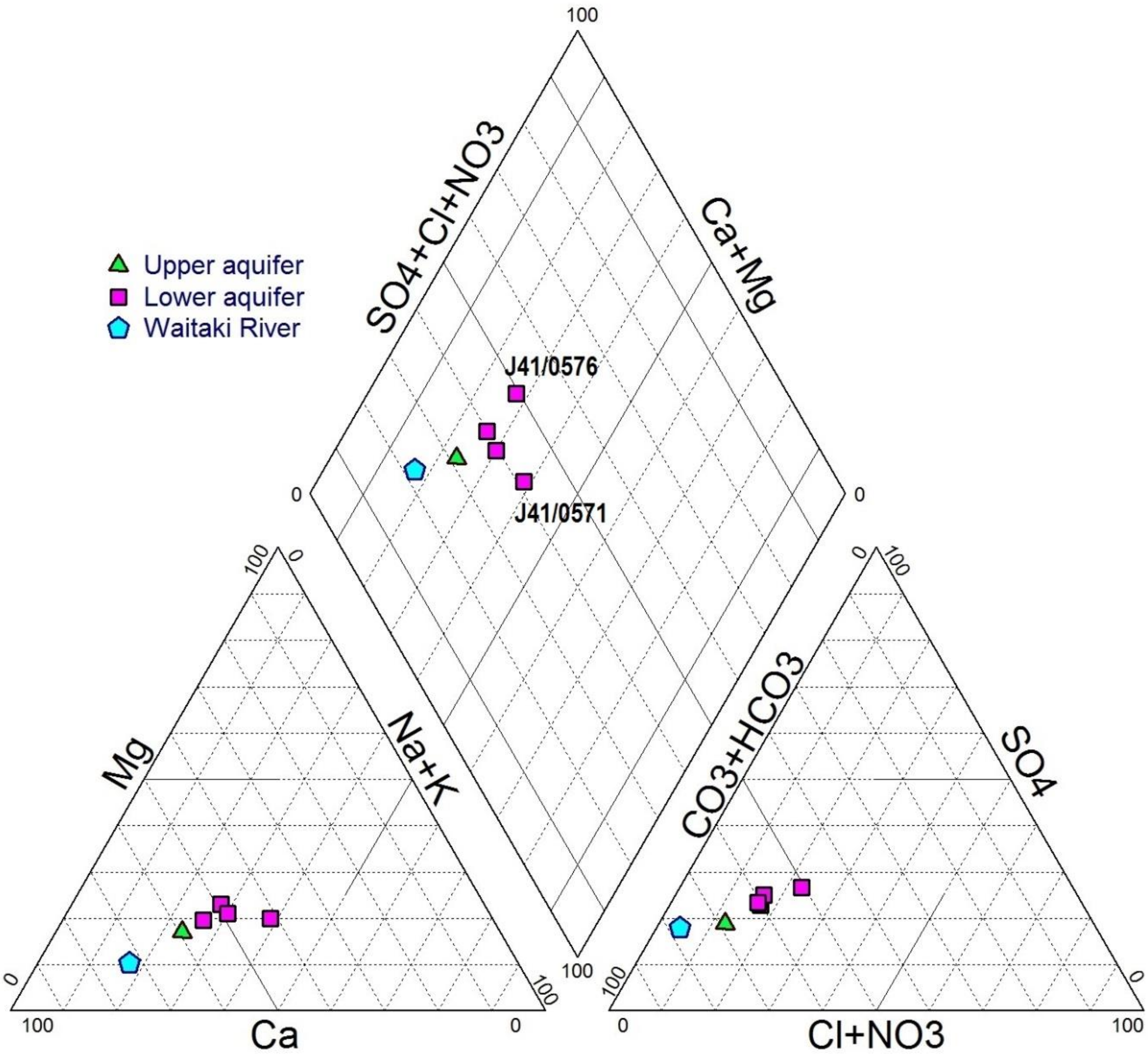

Figure 5-1 Piper Diagram of the median values of the major ions for of the samples collected between June 2016 and December 2017

According to the location of the samples (on the left-hand part of the diamond shaped field) the water is calcium-magnesium-bicarbonate type, which is characteristic of immature, relatively freshly recharged groundwater. The samples range between two main poles, the low mineralised Waitaki River water (sampled near the end of Ferry Road) and the most mineralised groundwater signatures for bore J41/0576 and J41/0571. The upper gradient bore J41/0442 signature is also quite distinctive and closer to the river water.

Another interesting way to compare the major ion composition of different samples is to plot the results on a Stiff diagram, as presented in Figure 5-2.

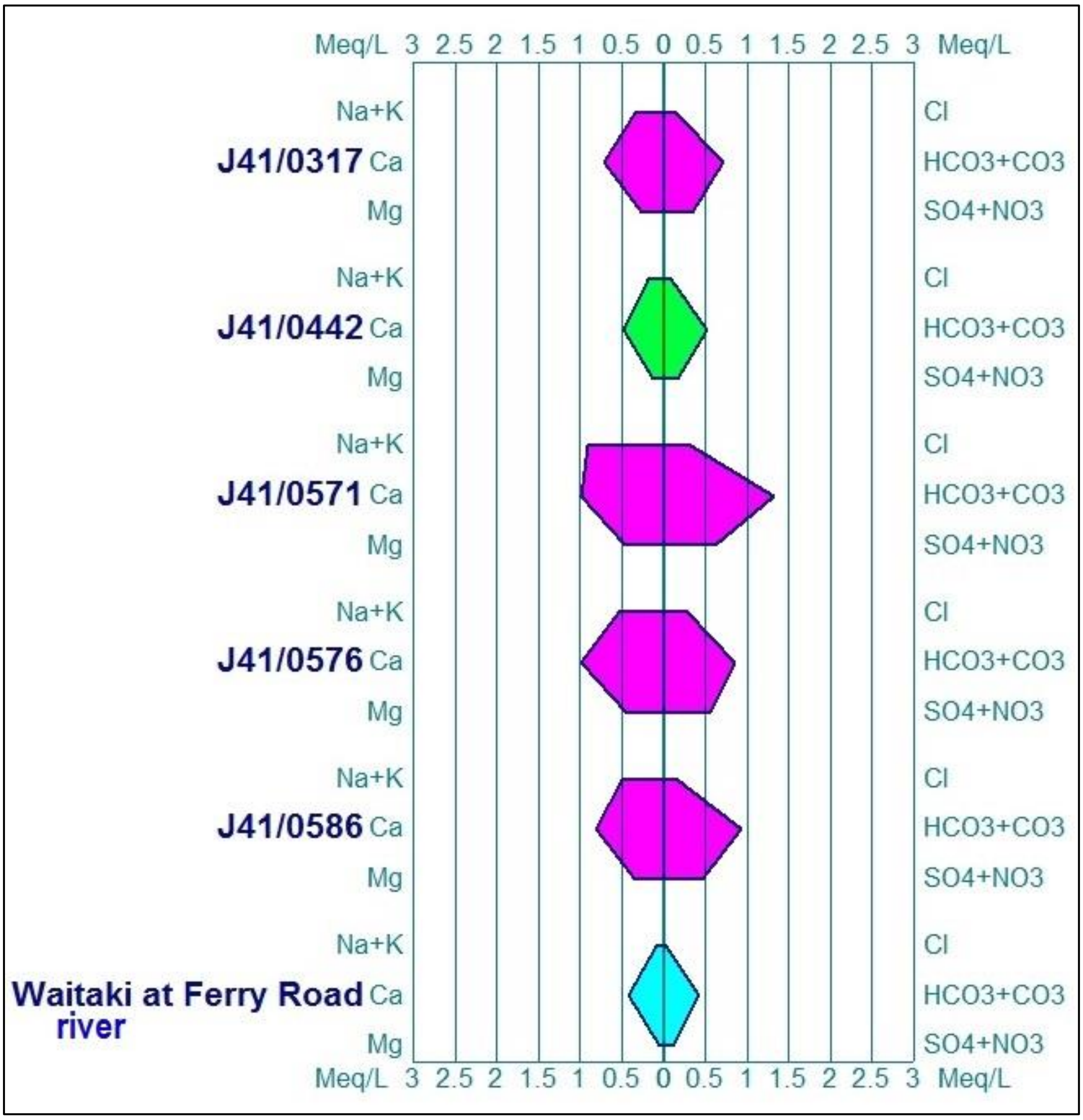

Figure 5-2 Stiff Diagram of the median values of the major ions for of the samples collected between June 2016 and December 2017

The cations are now represented on the left side and the anions on the right side according to their concentrations. In this manner, each sample has a “polygonal” signature and it is visually easy to determine the similarities and differences.

In addition, the size of the polygons is proportional to their mineralisation.

According to Figure 5-2 there is an obvious similarity between the upper aquifer bore (J41/0442) and the Waitaki River signatures, which have a similar water composition (shape of the polygon) and close mineralisation (size of the polygon). This is not surprising as there is an irrigation race in the vicinity of borehole J41/0442, and this could suggest some recharge in the area.

Bore J41/0571 has a quite different signature/shape with a non-negligible sodium and potassium component. The groundwater is more mineralised too, which could potentially indicate older groundwater (especially regarding the bicarbonate and reactive silica concentrations) or a more important contamination (for example by sodium).

The signatures of bores J41/0317, J41/0586 and J41/0576 are relatively similar in shape/composition. The sizes/mineralisation show that J41/0317 is less mineralised than these two other bores.

5.2. Nitrate

5.2.1. Long term trend for SOE Bore

The nitrate nitrogen concentrations have been monitored in the SOE Bore J41/0317 since June 1993. The results are presented in Figure 5-3 along with the groundwater levels in the Dennisons Bore.

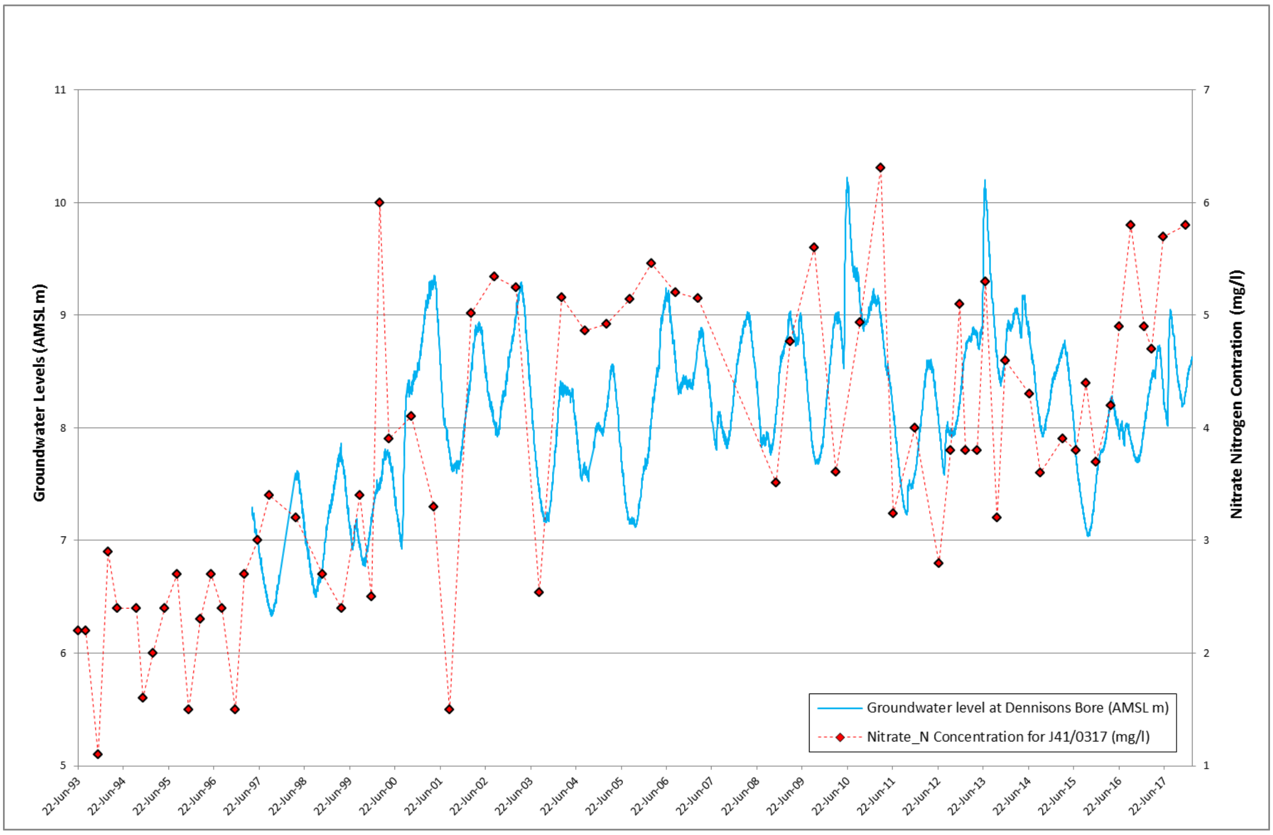

Figure 5-3 Evolution of the Nitrate-N concentrations from June 1993 to January 2018 for SOE Bore J41/0317

According to Figure 5-3, there has been an increasing trend in the nitrate-N concentrations from approximatively 2 mg/l up to around 5 mg/l between 1993 and approximately 2000. This rise in the nitrate-N concentrations is also correlated with the elevation of the water table. This could be caused by an increased drainage linked to the development of the irrigation scheme (increase in irrigation), providing less opportunity for pasture uptake, and consequently leading to a more important leaching of nitrates into the aquifer. Since 2001, the aquifer state appears more stable. In particular, the Nitrate-N concentrations in the aquifer storage seem to fluctuate around 5 mg/l and to have reached a relative equilibrium with the nitrogen inputs. The more recent trend is less clear, with slightly declining concentrations between 2011 and 2015, and another increasing trend between 2016 and 2018. Additional monitoring data over subsequent years will be required to determine a more certain trend.

5.2.2. Results for the extended monitoring network

The location of the sampling sites and the evolution of nitrate-N concentrations since 2011 for the long term SOE bore, and since June 2016 for other bores are given in Figure 5-4 and Figure 5-5.

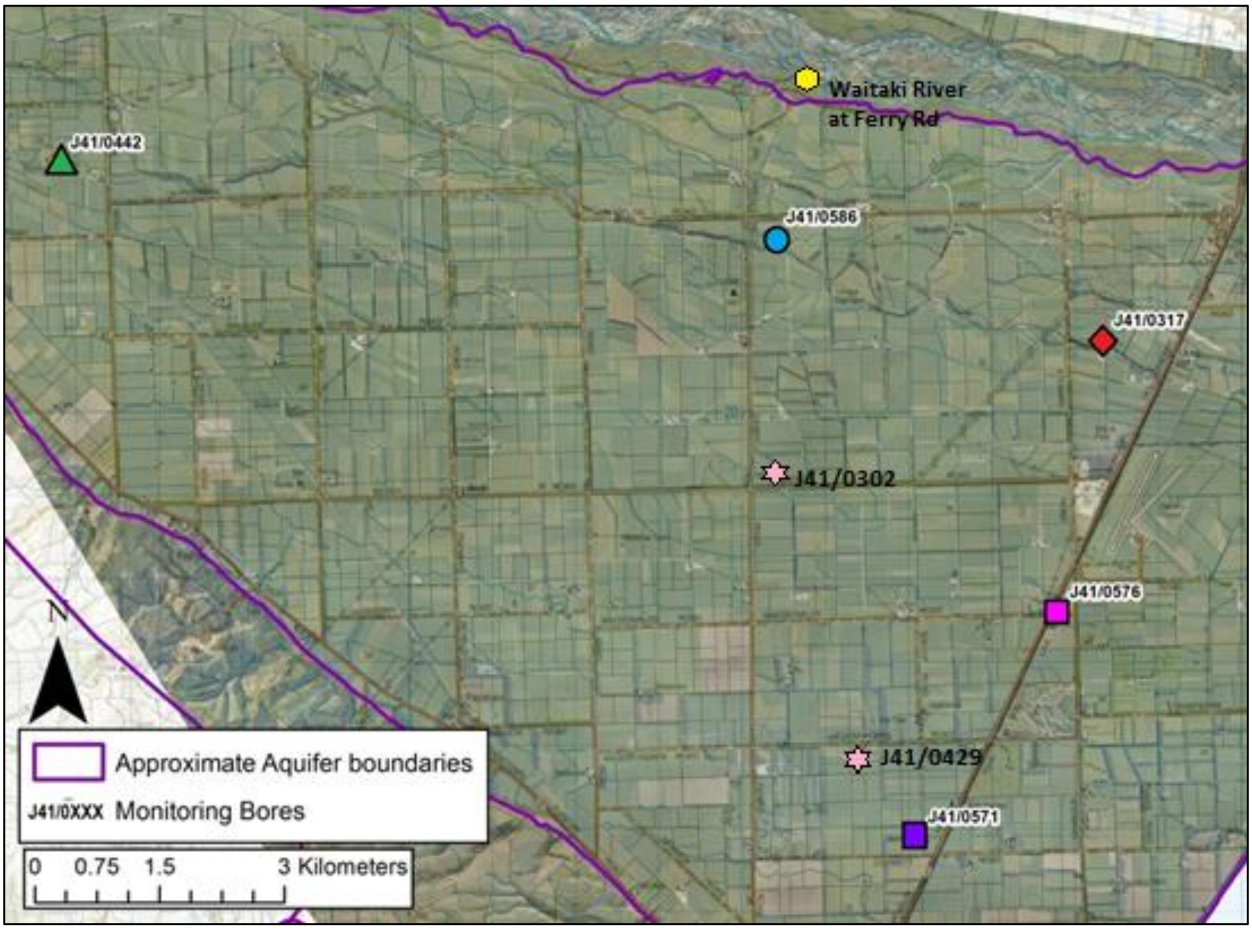

Figure 5-4 Location of the quality and age dating sampling sites

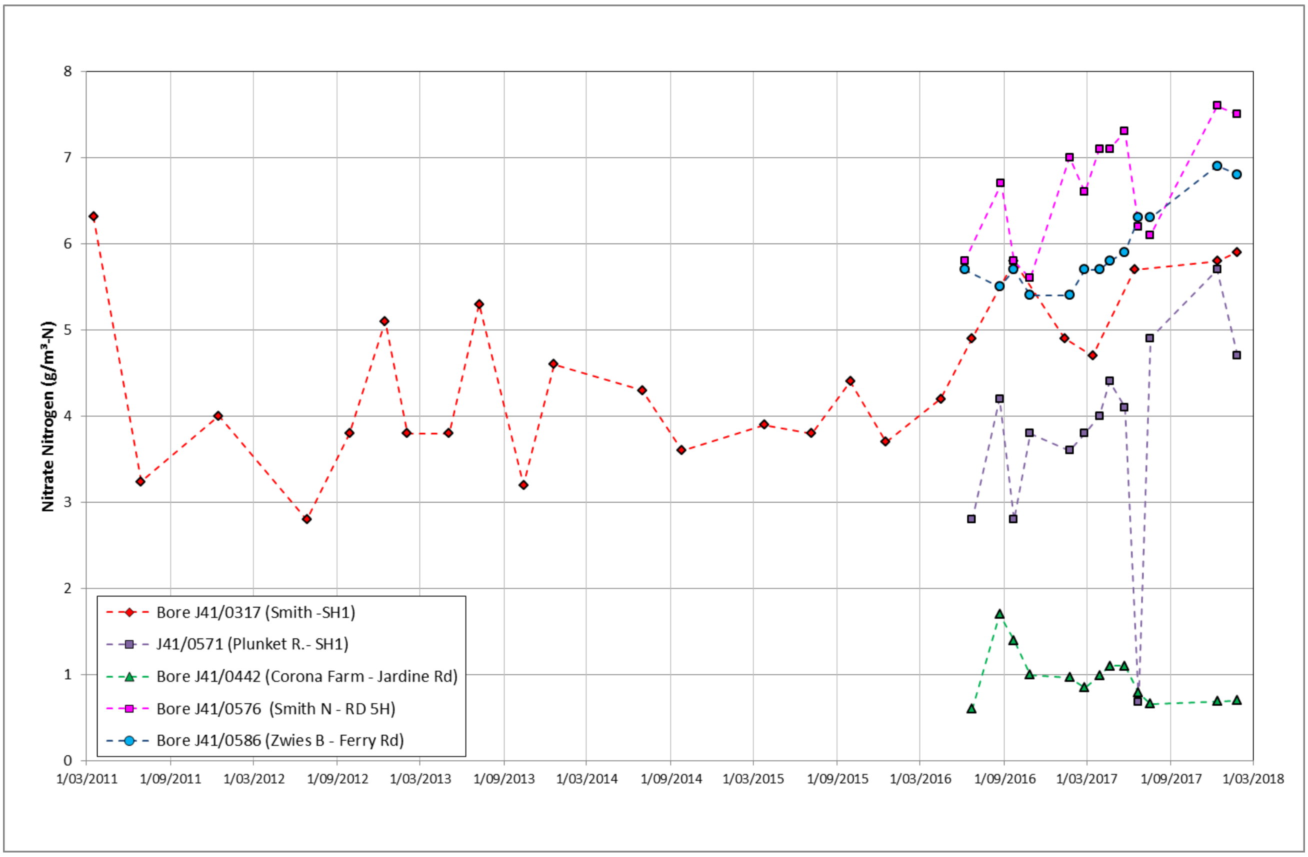

Figure 5-5 Nitrate-N concentrations for the Lower Waitaki Plains (August 2011 – January 2018)

Statistics based on the monitoring results are provided in Table 5.2.

Table 5.2 Statistics for the nitrate-N concentrations monitored on the sampling sites (June 2016-January 2018)

	J41/0442	J41/0317 since June 2012	J41/0317 since Nov 2008	J41/0571	J41/0576	J41/0586	Waitaki River at SH1 bridge	Welcome Creek at Stewart Rd
Median	**0.99**	5.7	4.30	3.8	**6.6**	5.7	0.034	1.32
Average	**0.99**	5.39	4.42	3.71	**6.56**	5.82	0.045	1.57
95% Percentile	**1.52**	5.87	5.80	5.22	**7.42**	6.54	0.121	2.96
Number of samples	13	7	29	13	13	13	79	39

The nitrate-N median concentrations range between 0.99 and 6.6 mg/l within the aquifer between the most up-gradient bore J41/0442 and the most down-gradient one J41/0576.

This trend is also noticeable with the 95% percentile with the lowest nitrate-N concentration of 1.52 mg/l for J41/0442 in the western area, and the highest one of 7.42 mg/l east of the state highway for J41/0576.

For comparison, the concentrations for the Waitaki River at State Highway Bridge and for Welcome Creek at Stewart Road are also provided in Table 5.2.

The data collected are therefore showing that the land use is impacting on the plains aquifer quality, with a cumulative effect as we travel along the groundwater flow paths towards the ocean. At this stage, the highest nitrate-N concentrations are still below the NZ Drinking Water Standards Maximum Acceptable Value (MAV) of 11.3 mg/l of nitrate-N.

The water quality of Welcome Creek, which is receiving irrigation run-off and a large groundwater base flow contribution, can therefore also be impacted by the local land use and groundwater quality degradation. However, the nitrate-N concentrations remain significantly below the lower aquifer concentration level, which might suggest nutrient intake processes occurring within the creek and/or dilution by less impacted water (irrigation race water).

5.2.3. Indications about the nitrogen source

During the investigations carried out in 1999-2000 (ORC 2000), 10 samples were analysed for ^{15}N isotope signature to characterise the source of the nitrate in terms of animal sewage, soil organic matter and fertiliser. The results are provided in Figure 5-6 and the location of the bores in Appendix D.

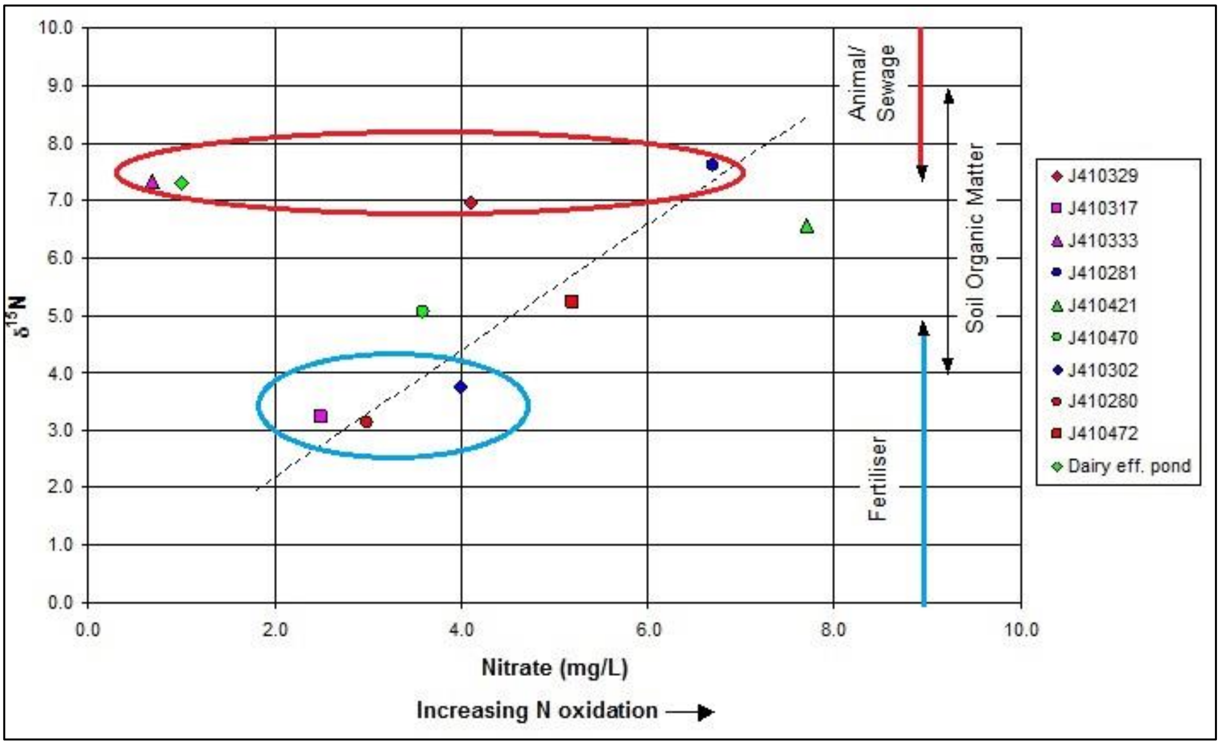

Figure 5-6 Nitrate versus Nitrogen 15 Isotope (ORC, 2000)

This plot indicates that two distinctive clusters of samples can be identified regarding their ^{15}N signature:

- Samples with $\delta\ ^{15}N$ values ranging between + 2.5 and + 4.5 ‰ and characteristic of nitrate sourced primary from fertilisers;
- Samples with more elevated $\delta\ ^{15}N$ values (+ 7 and + 8‰), showing nitrate mainly sourced from animal sewage.

A few $\delta\ ^{15}N$ values range between + 5 and 6.5 ‰ and likely result from mixing between these two sources.

However, enrichment in ^{15}N can also be linked to denitrification processes and volatilization of an ammonia based fertilizer within specific soil conditions (Townsend MA, 2003). Therefore, it would be prudent to carry out further analysis on both ^{18}O and ^{15}N of the nitrate to update and refine the understanding of the nitrate sources in the Lower Waitaki Plains.

5.3. Groundwater redox status

The amount of dissolved oxygen present in groundwater is a very important feature, as it is one of the potential drivers to reduction-oxidation reactions, which can significantly influence on the water quality. These redox reactions, which are defined by the transfer of electrons from one chemical species (electron donor) to another (electron acceptor), are catalysed by bacteria in groundwater. These micro-organisms are gaining energy in the process and tend to favour the acceptor supplying the greatest amount of energy (Figure 5-7).

Usually dissolved and particulate organic carbon is the most common electron donor found in groundwater, but minerals such as pyrite and glauconite can also play this role occasionally.

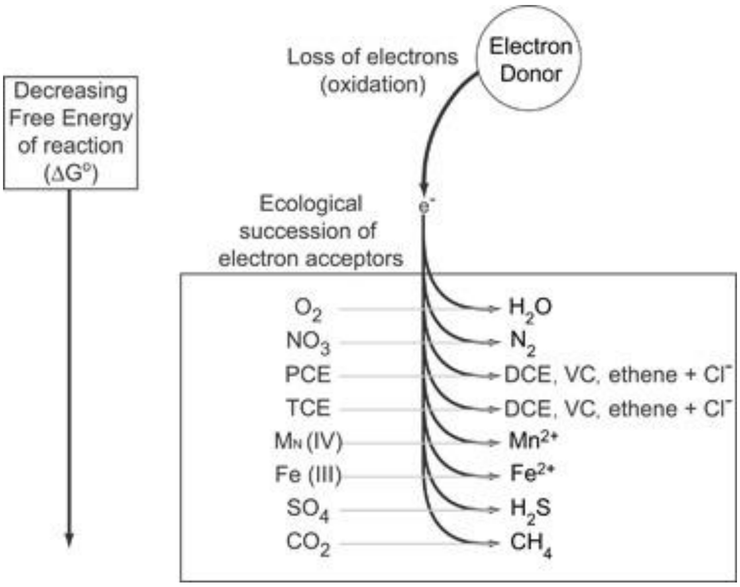

Figure 5-7 Ecological succession of electron-accepting processes, figure from Mc Mahon and Chapelle (2008)

According to Figure 5-7 oxygen is consumed first, followed by nitrate, manganese, ferric iron, sulphate and carbon dioxide.

Hence the interest to classify groundwater samples regarding their oxygen status or “redox status”. A classification using the method of McMahon and Chapelle and the USGS Excel Workbook (McMahon and Chapelle, 2008) for identifying Redox Processes in Groundwater is provided in Table 5.3 for the median values of the monitoring sites, the redox assignment criteria used for the purpose are provided in Appendix E.

Table 5.3 Redox Assignement for the monitoring sites (median concentrations)

Sampling Site	Dissolved O_2 (mg/l)	NO_3 as N (mg/l)	Mn^{2+} (mg/l)	Fe^{2+} (mg/l)	SO_4^{2-} (mg/l)	General Redox Category	Redox Process
J41/0317	5.66	4.4	0.001	0.017	13.2	Oxic	O_2
J41/0442	4.52	0.99	0.001	0.023	6.9	Oxic	O_2
J41/0571	1.01	3.8	0.012	0.079	27	Oxic	O_2
J41/0576	1.12	6.6	0.004	0.28	22	Mixed (oxic-anoxic)	O_2 Fe(III)/SO_4
J41/0586	3.35	5.7	0.001	0.009	17	Oxic	O_2

The redox assignments reflect the dominant redox state (oxic, suboxic, mixed (oxic-anoxic)) and the relative concentration of the terminal electron acceptor.

According to Table 5.3, the samples collected on the monitoring sites between June 2016 (June 2012 for J41/0317) and January 2018 are showing that the groundwater is mainly oxic within the Lower Waitaki Plains aquifer.

Figure 5-8 presents the nitrate-N concentrations versus the dissolved oxygen concentrations measured between June 2016 (December 2011 for J41/0317) and January 2018 for the same monitoring sites.

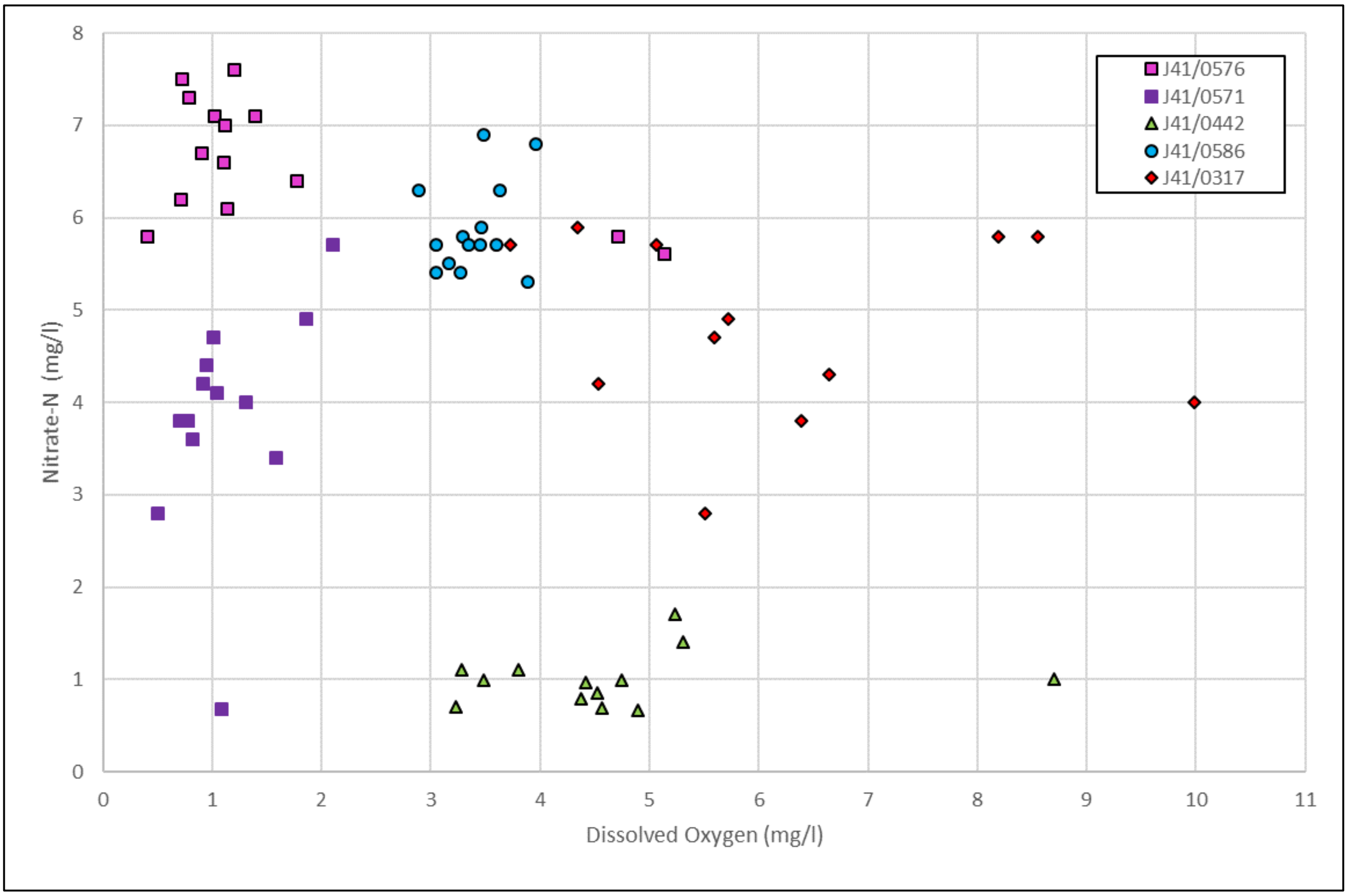

Figure 5-8 Nitrate–N concentrations versus dissolved oxygen concentrations for the monitoring sites (December 2011 – January 2018)

A quite wide range of dissolved oxygen values has been measured, with nearly fully oxidised (10 mg/l) to almost entirely oxygen depleted (0.41 mg/l) groundwater samples.

Boreholes J41/0571 et J41/0576 have usually the lowest dissolved oxygen contents, the details of the redox status according Mc Mahon and Chapelle (2008) for the samples collected on these two boreholes is provided in Appendix F.

Reduced conditions are definitely observed for several samples collected on J41/0571 and J41/0576 but are unexpectedly accompanied with consistent high nitrate concentrations and low ammonia concentrations. This may indicate that the denitrification capacity is exceeded.

The only low nitrate-N concentration (0.68 mg/l) was observed on bore J41/0576 on 22/06/2017, while the dissolved oxygen was 1.09 mg/l and the dissolved manganese concentration was elevated (0.057 mg/l), attesting of denitrification involving manganese reduction.

5.4. Pathogens

The Lower Waitaki alluvial aquifer is used for drinking purposes by the community (for example by the Waitaki District Council) and private landowners; the water is occasionally consumed without any treatment. The aquifer is identified as having a value of human use without treatment in the Schedule 3A of the Regional Plan: Water.

Testing for pathogenic bacteria has therefore been carried out as part of the environmental monitoring within the aquifer. The results since September 1999 for the long term SOE bore J41/0317 are provided in Figure 5-9, along with nitrate-N concentrations.

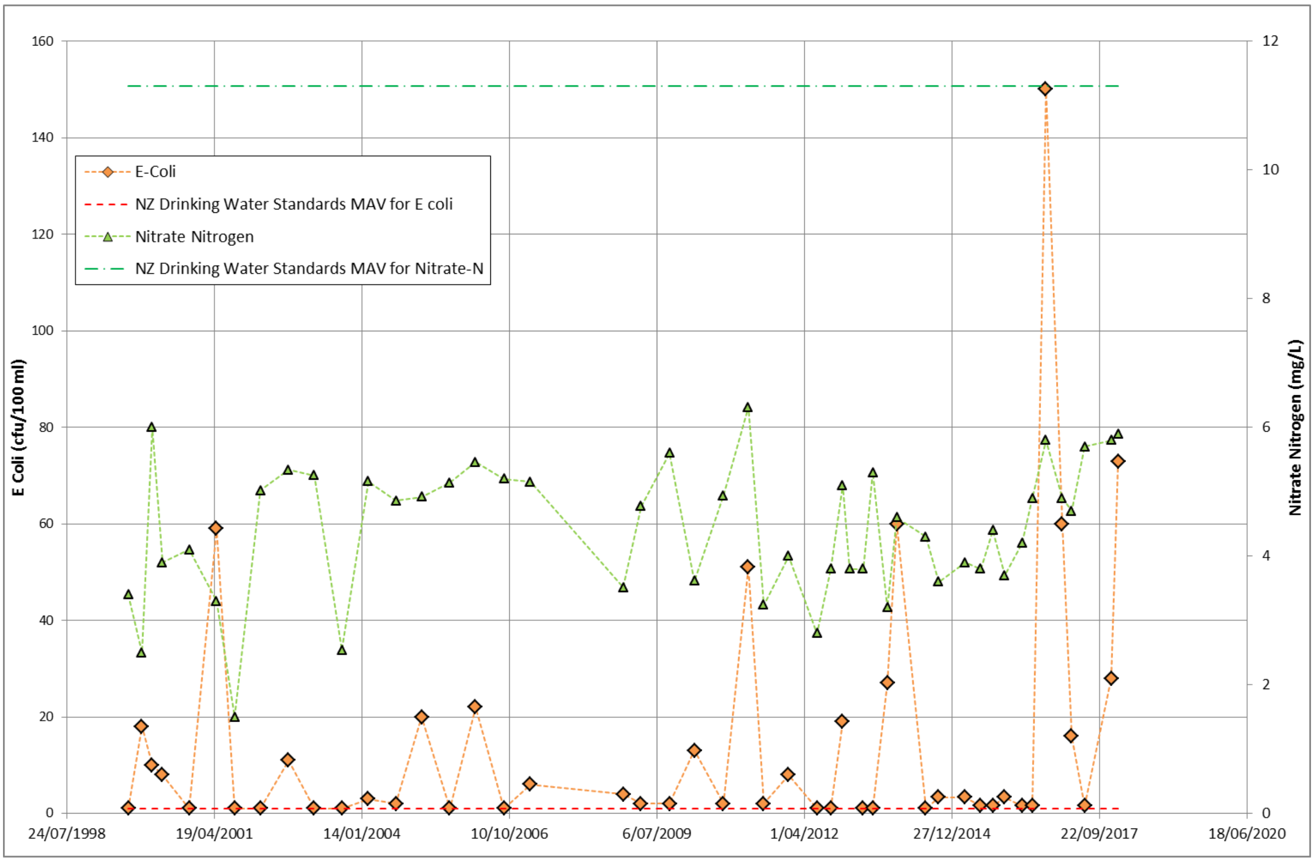

Figure 5-9 ***E. coli*** **and Nitrate-N concentrations for J41/0317 (September 1999 - January 2018)**

Escherichia coli (*E. coli*) is a subset of faecal coliforms, which are found within the bowels of warm blooded animals. These bacteria do not generally cause problems themselves but indicate the likelihood that pathogenic bacteria from the same source will be present.

Therefore, any water that contains faecal coliforms (*E. coli*) is considered unsuitable to drink without treatment (chlorination, ozonation, boiling, adequate filtration), and the NZ Drinking Water Standards Maximum Acceptable Value (MAV) for *E. coli* is less than 1 cfu/100 ml.

Figure 5-9 outlines that recurrent faecal contamination events ranging between 1 and 150 cfu/100 ml have been detected on J41/0317 over the last 20 years.

Figure 5-10 focusses on the results since September 2012 and includes those from the new monitoring bores. *E. coli* concentrations above the NZ Drinking Water Standards MAV (ranging between 2 – 73 cfu/100 ml) have also been recorded on the new monitoring bores especially during summer 2017, where irrigation was occurring, and the water table was relatively high.

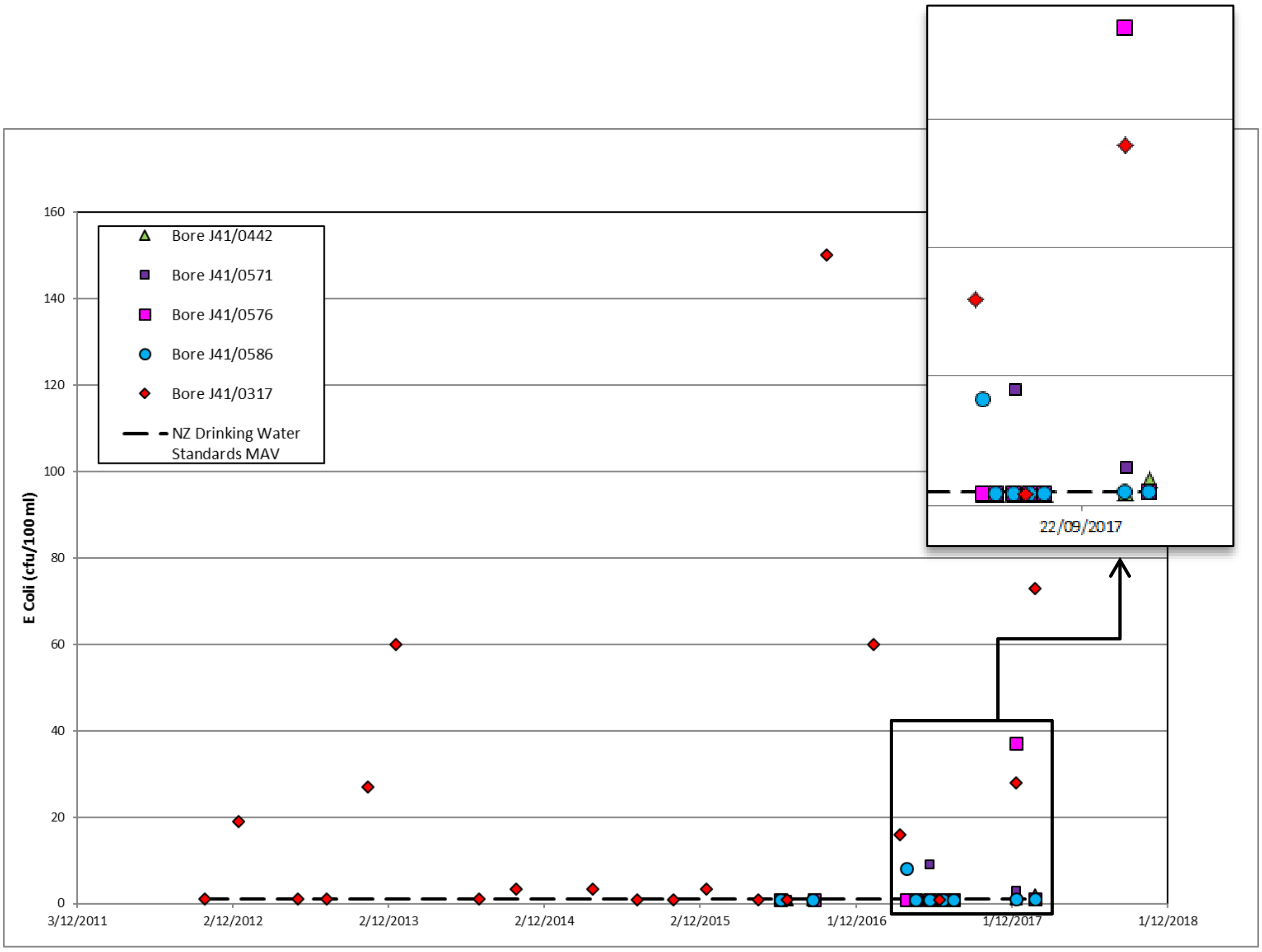

Figure 5-10 *E. coli* concentrations for the monitoring bores (September 2012 – January 2018)

The contamination has been signalled to the landowners and to the Lower Waitaki Irrigation Company. Discussions have been carried out along with the Waitaki District Council and the Southern District Health Board although the source of the contamination has not been confirmed yet.

It is anticipated that these contaminations may be linked to sub-standard bore head protections or/and poor effluent management practices. A bore inspection visit has already highlighted that the sealing of some bores should be improved.

In addition, *E. coli* might enter the aquifer via border dyke irrigation practices, combined with macropore flow in light soils and/or cracking of heavier soils from drying after saturation. As shown in Figure 2-7 and Figure 5-4, most of the bores exhibiting pathogen detections are located in the vicinity of irrigation races and areas where border dyke irrigation is carried out.

Further analysis such as PCR markers, faecal sterol analysis, and fluorescent whitening agent analysis should also be carried out to identify the source of contamination.

An outreach programme has also been initiated with the edition of a bore head protection brochure.

5.5. Age dating

To better understand the hydrodynamics of the aquifer and the time it takes for the water to transfer from the surface (when it infiltrates through the soils), until its abstraction in a bore, age dating samples have been collected on three bores as shown in Figure 5-11.

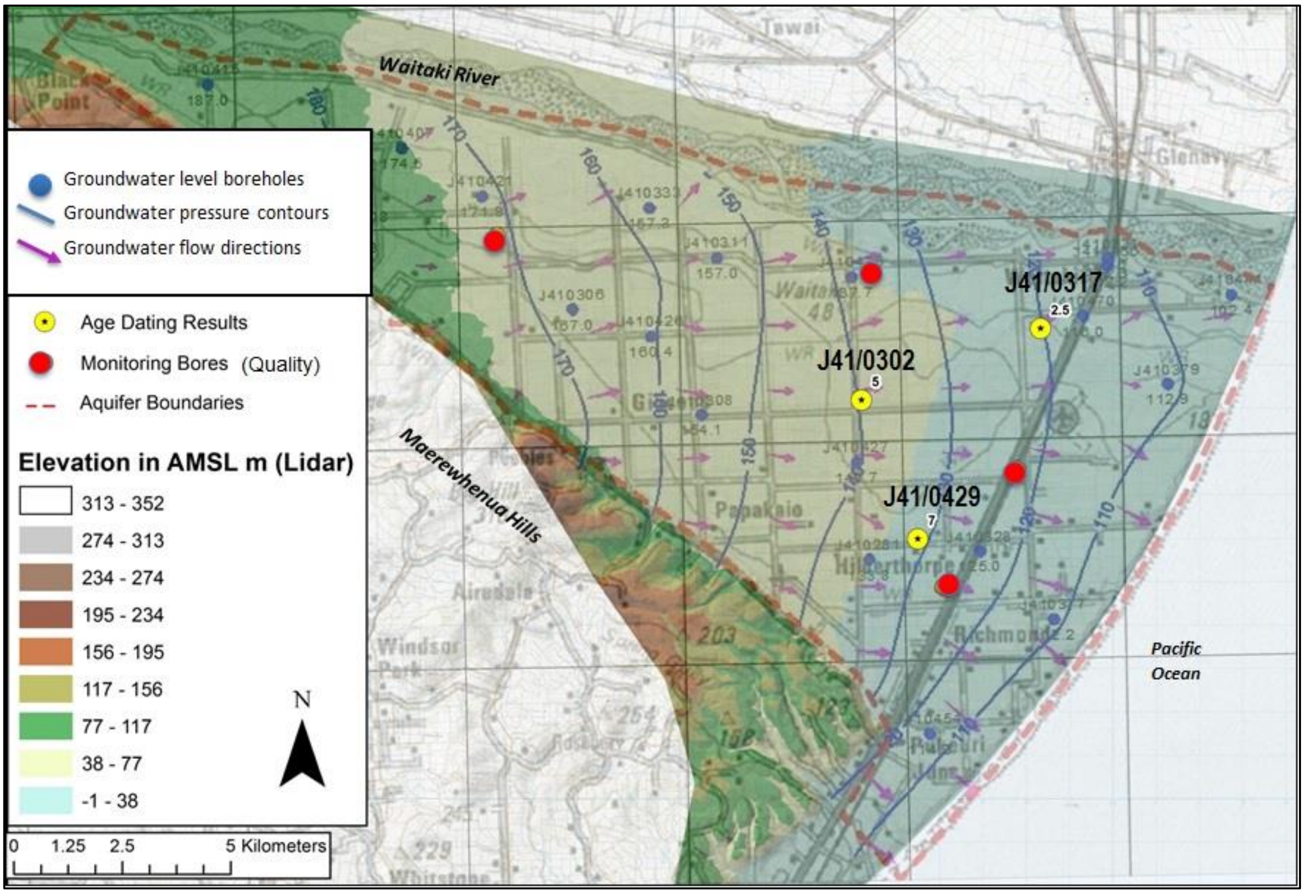

Figure 5-11 Age dating, elevation, groundwater contours and flows

These samples have been analysed for tritium and one sample also for chlorofluorocarbons (CFC) and sulfur hexafluoride (SF6). A summary of the results is given in Table 5.4 and the detail of the analyses is provided in Appendix G.

Table 5.4 Mean Residence Time inferred from age dating sampling

Well Id	Depth (m)	Water Table Depth (m)*	Sampling Date	Tritium	Mean Residence Time (years)
J41/0302	17	Not available	29/04/2016	1.903 ± 0.031	5
J41/0429	18	2.4	29/04/2016	1.746 ± 0.035	7
J41/0317	16.5	6.0	14/03/2012	2.141 ± 0.038	2-3

** value from the ORC Bore database, depth to water table measured after bore completion*

According to these results, the groundwater is young (between 2 and 7 years old), which is consistent with most of the hydrochemical signatures mentioned in section 5.1.

If we consider the results of the bores located in the middle of the plain (J41/0302 and J41/0429), the age increases from the west to the east, which is in accordance with the groundwater flow pattern suggested by the piezometric maps.

However, the mean residence time for J41/0317 located along the river margin is shorter than expected while considering the flow patterns of the piezometric maps (Figure 3-6 and Figure 3-7). Along the river margin the soils are lighter and more freely draining, and the alluvium more transmissive too due to more reworking. These characteristics, combined to the proximity of a discharge boundary (Waitaki River and/or Welcome Creek) provide a more efficient discharge mechanism for the aquifer, resulting a shorter mean residence time.

6. Recommendations

The investigations carried out around the Lower Waitaki Plains bores and the analysis of the existing and collected data have highlighted the need for further research and monitoring including:

- The installation of another groundwater level monitoring bore, with a more central location over the Plain (for example south of Gibson Road and Mc Pherson Road Crossing) and of a stage monitoring site within the Waitaki River (for example near the end of Gibson Road) to better understand the interactions between the aquifer and the river. This monitoring bore may also enable better understanding of the interactions with the foot hill streams recharge;
- The organisation of at least two additional groundwater level measurement campaigns over summer and winter, with updated piezometric maps to assess potential flow pattern changes linked to the development of the irrigation scheme through the plains since 1999 - 2000;
- Further investigations around the faecal contamination issues to enable identification of the source in order that appropriate mitigation measures can be adopted;
- Investigations to better understand the sources of nitrogen, with nitrate isotopes analysis (^{15}N, ^{18}O);
- Science studies and monitoring of the local water bodies interacting with the aquifer, the coastal environment and the stygofauna communities/groundwater ecosystems to ensure their values are preserved;
- Study of the potential / benefit for performing managed aquifer recharge through the plains for the purpose of improving the groundwater quality.

Ideally, the development of an integrated flow and transport model (soil/groundwater/surface water) would help to assess/test different scenarios and optimise the water resource management. Some work has already been initiated in that direction (ORC, 2011), and it would be very beneficial to complete it with the additional local knowledge gained, the extended data sets available, as well as the outputs of recent research works. The experience gained during the development of the Kakanui-Kauru Alluvium Aquifer integrated flow and transport model would be of particular interest (Aqualink, Lincoln Agritech, NIWA, Landcare Research, 2018).

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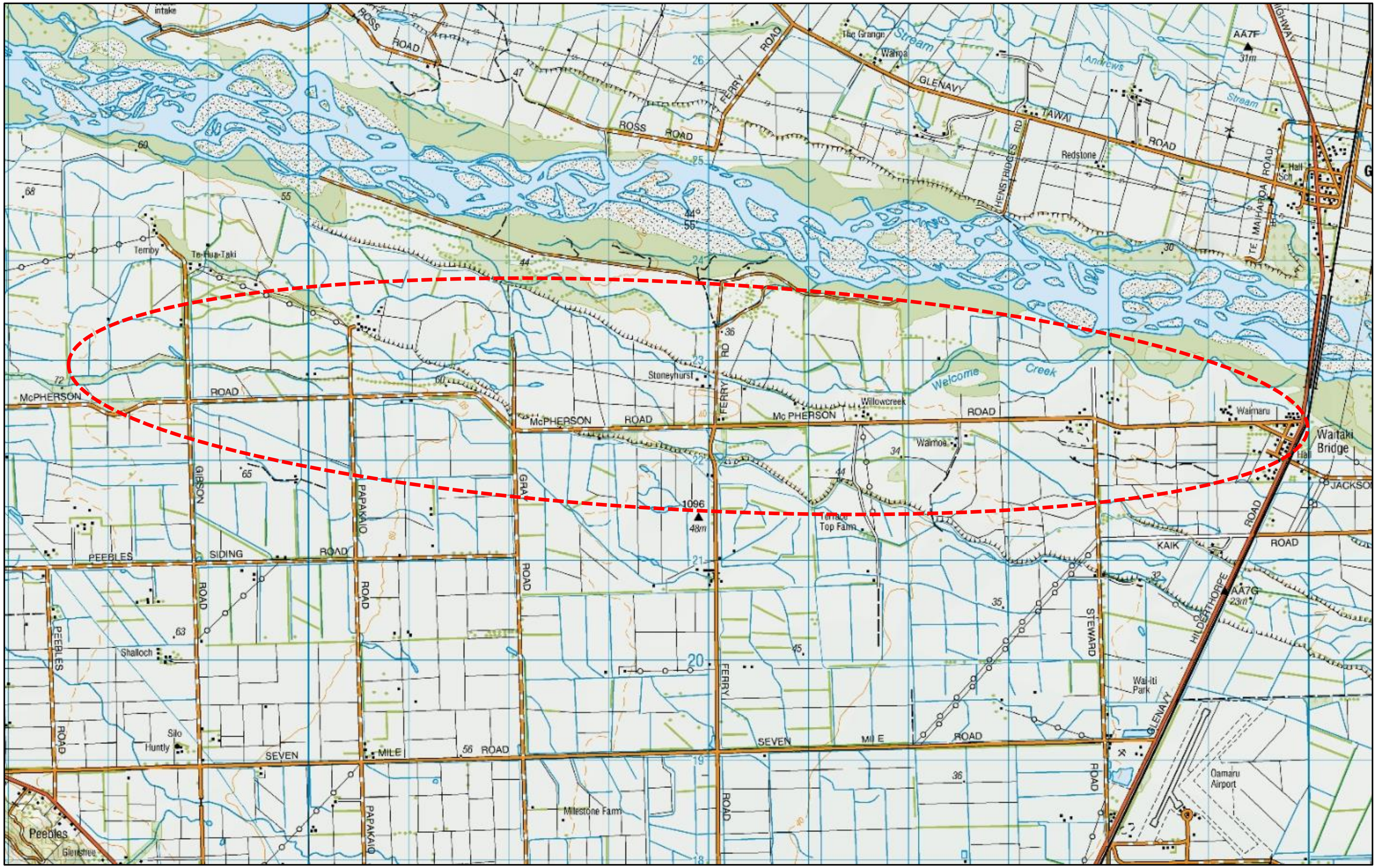
Appendix A: Welcome Creek Location



Appendix B: Brief description of the Hydraulic Model

(extract of "*Assessing the impact of nitrate-N leaching limits on Otago groundwater*" Memorandum, ORC 2011)

Model Domain

The focus of this model is the Lower Waitaki Plain. The model extends from Black Point, where the valley begins to widen to form the plain, to the sea. The north bank and Oamaru coastal areas have been included solely to create a suitable flow field around the model periphery.

Dimensions: 22 km x 32 km

Total area: 70,400 Ha; Aquifer area 25,000 Ha

Minimum layer thickness: 15m

Cell dimensions: 500m x 500m

Total active cells: 1,799

Boundary Conditions

Rivers	Bed Conductivity (m/d)	Bed Conductance (m^2/d)
Upper Waitaki	28	1,414,362
Lower Waitaki	8	398,595
Welcome Creek	14	9,928

Hydraulic Properties

Unit	Zone	Conductivity (m/d)
Q1 Alluvium	kx1	1,735
Q2 Alluvium	kx2	143
Q4 Steward Soil	kx3	381
Q4 Pukeuri Soil	kx4	123
Q4 Coastal Oamaru	kx5	91

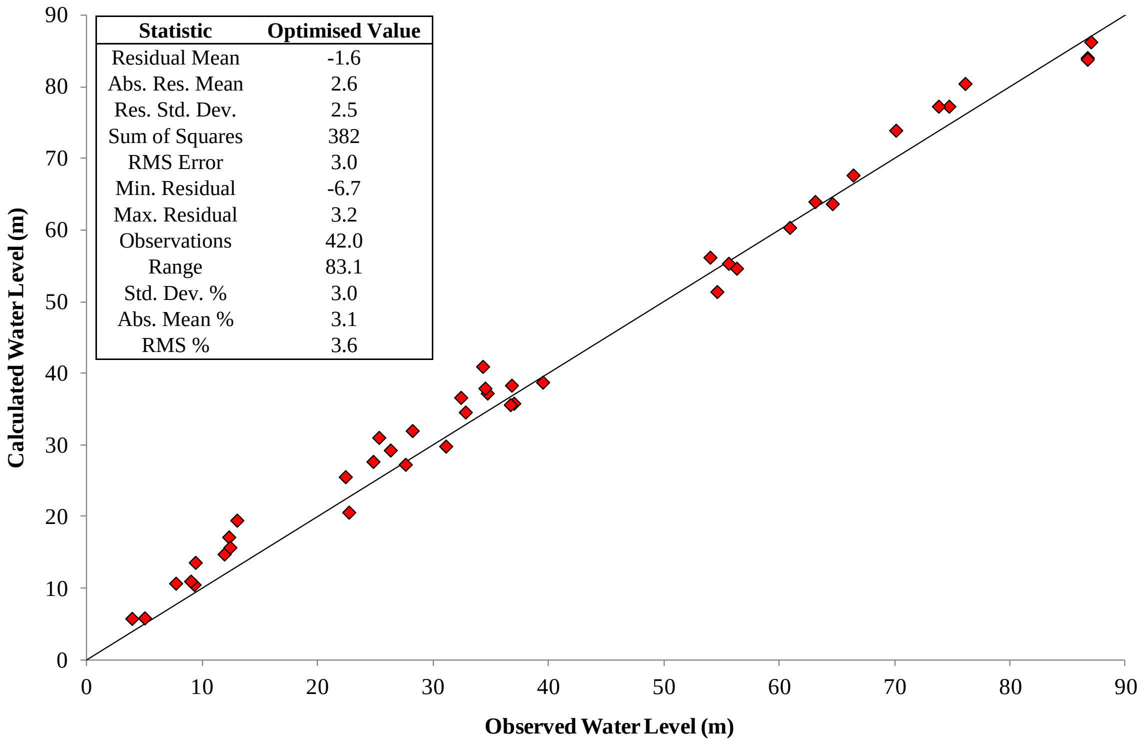
Land Surface Recharge

Zone	Soil Description	PAW	RCH (mm/y)	Recharge Type	Nitrate-N (Kg N/Ha)	Nitrate-N (mg/l)
1	Stony sand	30	161	Rainfall	16	3.62
2	Shallow stony sand	80	103	Rainfall	13	4.60
3	Silt loam	150	42	Rainfall	8	6.91
4	Stony sand	30	644	Flood	104	5.90
5	Stony sand	30	243	Spray	39	5.85
6	Shallow stony sand	80	468	Flood	64	4.99
7	Shallow stony sand	80	204	Spray	29	5.19
8	Silt loam	150	315	Flood	60	6.96
9	Silt loam	150	170	Spray	15	3.22
10	Stony sand	30	644	Dairy Flood	133	7.54
11	Stony sand	30	243	Dairy Spray	50	7.50
12	Shallow stony sand	80	468	Dairy Flood	81	6.31
13	Shallow stony sand	80	204	Dairy Spray	36	6.45
14	Silt loam	150	315	Dairy Flood	76	8.81
15	Silt loam	150	170	Dairy Spray	19	4.08
16	Silt loam	150	300	Nursery	50	6.08

Optimisation

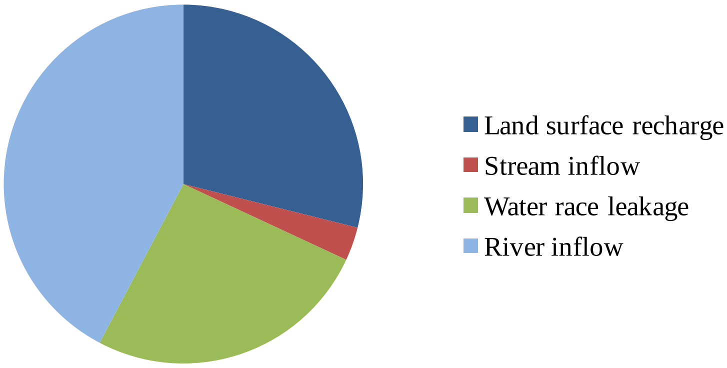
Lower Waitaki Plain was optimised for median annual conditions. Water level targets were taken from drillers well logs. The objective was to obtain a representative flow field rather than accurate individual heads. Hydraulic conductivity values for the aquifer and river bed conductance have 95% confidence limits within 50% of the optimised value. Exceptions are the narrow band of Q2 gravels and the upper reach of the river which have no observation data to constrain them.

Heads



Fluxes

The mean annual water balance for Waitaki Plain is shown graphically below. Land surface recharge and water race leakage constitute a large proportion of the mass balance.



Appendix C: Groundwater Quality Raw Data for the sampling carried out between April 2016 and January 2018

J410317

Date	Alkalinity Hydroxide	Alkalinity (HCO3)	Alkalinity Phenol-pthalein	Alkalinity Total	Alkalinity to pH 8.3 (as CO3)	Ammoniacal Nitrogen	Arsenic Dissolved	Biochemical Oxygen Demand 5d	Boron Total	Calcium Dissolved	Carbonate Alkalinity	Chloride	Conductivity (Field)	Conductivity (Lab)
	(mg/L)	(g/m³)	(g/m³ CaCO3)	(g/m³)	(g/m³)	(g/m³-N)	(g/m³)	(g/m³-O)	(g/m³)	(g/m³)		(g/m³)	(mS/cm)	(mS/m)
22/06/1993		44		0.001	0.006	0.014		<1.000		14.4			0.15	
23/08/1993		35		35		0.012		<1.000		10.5		2.2	0.11	
29/11/1993		24		24		<0.01000		<1.000		8.9		2.4	0.1	
18/01/1994														
21/02/1994		40		40		0.013		<1.000		14.1		4.4	0.15	
3/05/1994		46		46		<0.01000		<1.000		17.6		4.8	0.16	
5/10/1994														
30/11/1994								<1.000						
16/02/1995				40		0.009		<1.000		10		3.2		
24/05/1995						0.013								
30/08/1995														
29/11/1995														
6/03/1996				36.9		0.011		<1.000		12		3.6		
29/05/1996														
28/08/1996														
11/12/1996														
26/02/1997		50				0.023		<1.000		13		6.2		
11/06/1997						0.013								
10/09/1997														
15/04/1998						<0.005000						5.1		0.15
18/11/1998		52			0	0.021						4.4		0.15
21/04/1999		41			0	0.012						3.3		0.12
16/09/1999		49			0	<0.005000			<0.0300	13		4		0.15
15/12/1999		39			0	0.005				11		3.2	0.113	
23/02/2000		47			0	0.013				16		8.4	0.24	
2/05/2000		52			0	<0.005000				14		5.9	153	
1/11/2000			25		<2.000	<0.01000						5.8		12.3
3/05/2001			35		<2.000	<0.01000						5.8	0.118	
6/09/2001			18		<2.000	<0.01000						2.9	0.066	
25/02/2002			50		<2.000	<0.01000						5.5	0.199	
3/09/2002		55			<1.000	<0.01000						7	0.235	
24/02/2003		54			<1.000	<0.01000						8.1	0.181	
1/09/2003		58			<1.000	<0.01000						7.9	0.151	
25/02/2004		56			<1.000	<0.01000						7.8	0.19	
1/09/2004		58			<1.000	0.01						7.5	0.173	
23/02/2005		51			<1.000	0.01						6.8	0.172	
31/08/2005		39			<2.000	<0.01000						5.14	0.142	
21/02/2006		43			<2.000	<0.01000						7.61	0.175	
5/09/2006		46		46	<2.000	<0.01000				18.8		8.29	0.119	
1/03/2007		47		47	<2.000	<0.01000				19.2		7.92	0.161	
20/11/2008		35	<1.000	35		<0.01000	<0.005000		<0.0050	16.7		4.59		0.14
18/03/2009		37		37	<1.000	<0.01000	<0.005000		0.006	16.7		6.39	0.161	0.15
30/09/2009		33		33	<1.000	<0.01000	<0.005000		0.014	17.1		4.36		0.152
17/03/2010		34		34	<1.000	<0.01000	<0.005000		0.016	14.2		3.7	0.118	0.13
29/09/2010		39		39		<0.01000	<0.001000		0.014	15.6	<1.000	5.98	0.126	0.15
16/03/2011		29		29		<0.01000	<0.001000		0.017	15.9	<1.000	5.34	0.164	0.13
28/06/2011		39		39		<0.01000			<0.0300	14.7	<1.000	4.13	0.14	0.14
15/12/2011		39		32		<0.01000	<0.001000		0.014	15.3	<1.000	4.8	0.136	14.1
26/06/2012		44		36		<0.01000	<0.001000		0.011	12.4	<1.000	3.7	0.119	15.8
27/09/2012		43		35		<0.01000	<0.001000		0.012	14.1	<1.000	4.7	0.142	14.1
14/12/2012		42		34		<0.01000	<0.001000		0.016	15.6	<1.000	5.4	0.153	14.6
1/02/2013		41		34		<0.01000	<0.001000		0.015	12.9	<1.000	3.8		12.3
3/05/2013		46		38		<0.01000	<0.001000		0.015	14.3	<1.000	4.2	0.147	14.3
9/07/2013		43		35		<0.01000	<0.001000		0.017	17	<1.000	6.1	0.165	16.6
14/10/2013		40		33		<0.01000	<0.001000		0.012	12.2	<1.000	3.1	0.123	12.3
19/12/2013		42		35		<0.01000	<0.001000		0.017	16.1	<1.000	5	0.154	15.2
1/07/2014		47		39		<0.01000	<0.001000		0.018	15.1	<1.000	4.2	0.082	15.3
26/09/2014	<1.000	38		31		<0.005000	0.00025		0.011	13	<0.600	4.2	0.135	13.5
25/03/2015	<1.000	44		36		<0.005000	0.00034		0.011	15	<0.600	5	0.158	15.9
7/07/2015	<1.000	46		38		<0.005000	0.00026		0.012	14	<0.600	4.6	0.151	15.4
30/09/2015	<1.000	46		38		0.007	0.00025		<0.0050	14	<0.600	4.6	0.161	15.3
17/12/2015	<1.000	41		34		<0.005000	0.00023		<0.0050	15	<0.600	4.5	0.153	14.7
15/04/2016	<1.000	46		38		0.011	0.00031		<0.0050	15	<0.600	5.1	0.157	16.4
22/06/2016	<1.000	49		40		<0.005000	0.00027		<0.0050	16	<0.600	5.7	0.168	17.3
22/09/2016	<1.000	47		38		<0.005000	0.00032		0.014	18	<0.600	5.7	0.166	17.3
11/01/2017	<1.000	44		36		0.011	0.00022		0.012	16	<0.600	5	0.158	16.4
14/03/2017	<1.000	41		34		<0.005000	0.00022		0.0078	16	<0.600	5.8	0.154	16.1
14/06/2017	<1.000	49		41		0.008	0.00027		<0.0050	19	<0.600	6.4	0.183	19
12/12/2017	<1.000	33		33		0.025	<0.001000			17	<1.000	4.7	0.147	16.5
25/01/2018	<1.000	39		39		<0.005000	<0.001000			17.4	<1.000	4.1	0.167	17
13/03/2018	<1.000	39		39		<0.005000	<0.001000			18.5	<1.000	6.3	0.177	17.8

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Date	Copper Total	Dissolved Oxygen	Dissolved Oxygen Saturation (Field)	Dissolved Reactive Phosphorus	E-Coli MPN	Escherichia coli Type 1	Faecal Coliforms-membrane filt	Free carbon dioxide	Ion Balance	Iron Acid Soluble	Iron Dissolved	Iron Total	Magnesium Total
	(g/m³)	(mg/L)	(%)	(g/m³-P)	(cfu/100mL)	(cfu/100mL)	(cfu/100mL)	(g/m³)		(g/m³)	(g/m³)	(g/m³)	(g/m³)
22/06/1993	<0.02000			<0.00500			0	61				<0.1000	3.4
23/08/1993	<0.02000			<0.00500			0	20				<0.1000	2.7
29/11/1993	<0.02000			0.008				51				<0.1000	2.1
18/01/1994							4.5						
21/02/1994	0.22			0.007			5	47				<0.1000	3.4
3/05/1994	0.03			0.012			5	31				<0.1000	3.6
5/10/1994							80						
30/11/1994							16						
16/02/1995	<0.02000			0.015			3					<0.1000	2.2
24/05/1995							<1.000						
30/08/1995							3						
29/11/1995							19						
6/03/1996	<0.02000			0.018			90					<0.1000	2.8
29/05/1996							<1.000						
28/08/1996							<1.000						
11/12/1996							9						
26/02/1997	<0.002000			0.026			210					0.024	3.3
11/06/1997							<1.000						
10/09/1997							<1.000						
15/04/1998				0.025			60						
18/11/1998													
21/04/1999				0.035									
16/09/1999				0.039		<1.000	<1.000					<0.05000	3
15/12/1999				0.012		18	18						2.8
23/02/2000				0.06		<10.00	<10.00						3.5
2/05/2000				0.032		8	8						3.9
1/11/2000				0.013		1	1						
3/05/2001				0.011		59	86						3.4
6/09/2001				0.015		<1.000	<1.000						
25/02/2002				0.006		<1.000	<1.000						
3/09/2002				<0.00500		11	11						
24/02/2003				0.009		1	80						
1/09/2003				<0.00500		<1.000	<1.000						
25/02/2004				0.005		3	10						
1/09/2004				0.016		2	2						
23/02/2005				0.025		20	38						
31/08/2005				0.016		<1.000	<1.000						
21/02/2006				0.026		22	23						
5/09/2006				0.016		<1.000	<1.000			0.022			5.16
1/03/2007				0.038		6	6						5
20/11/2008				0.031		4		28	12.1		0.074		3.73
18/03/2009				0.032		<2.000	<2.000	15	3.69		<0.005000		3.47
30/09/2009				0.025		<2.000	<2.000	10	3.29		0.027		3.85
17/03/2010				0.033		13	13	14	0.17		<0.005000		3.03
29/09/2010				0.034		2	2	20	1.7		0.005		3.54
16/03/2011				0.039		51	66	9	1.25		0.012		3.3
28/06/2011				0.033		<2.000	<2.000	5	0.17		0.02		3.16
15/12/2011		9.99		0.023		8	8	6	0.71		<0.02000		3.3
26/06/2012		5.51	51	0.029		<1.000	<1.000	2	2.5		<0.02000		2.9
27/09/2012		6.39		0.025	<1.00		<1.000	6.1	2.9		<0.02000		2.9
14/12/2012				0.025	19		19	9.7	0.34		<0.02000		3.6
1/02/2013				0.027				5.1	2.6		<0.02000		2.9
3/05/2013				0.029	<1.00		<1.000	14.3	1.62		<0.02000		3.4
9/07/2013				0.028	<1.00		<1.000	6.2	0.86		<0.02000		3.7
14/10/2013				0.029	27		36	4.8	0.39		<0.02000		2.8
19/12/2013				0.026	60		70	2.3	<0.1000		<0.02000		3.5
1/07/2014		6.64	63.5	0.029	<1.00		<1.000	3.7	2.8		0.03		3.3
26/09/2014				0.033	3.3			11	1.3		0.0095		3
25/03/2015				0.039	3.3			15	2.5		0.0068		3.3
7/07/2015				0.028	<1.60			9.7	3.6		0.0099		3
30/09/2015				0.027	1.6			9.9	6.6		0.034		3
17/12/2015				0.031	3.3			6.5	1.3		0.053		3
15/04/2016		4.53	44.8	0.063	1.6			19	4		0.034		3
22/06/2016				0.033	<1.60			14	3.6		0.025		3.6
22/09/2016		8.19	77.7	0.029	150			19	1		0.014		3.8
11/01/2017		5.72	49.6	0.027	60			29	0.37		0.014		3.5
14/03/2017		5.59	53.8	0.029	16			14	0.85		0.0094		3.4
14/06/2017		5.06	49.2	0.027	<1.60			10	0.85		0.0085		4.2
12/12/2017		8.55	80.2	0.023	28			11	1.69		0.03		4
25/01/2018		4.34	41.4	0.026	73			9.8	1.89		<0.02000		3.6
13/03/2018		3.73	36.4	0.029	1			7.9	0.31		<0.02000		3.9

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Date	Manganese Acid Soluble	Manganese Dissolved	Manganese Total	Nitrate Nitrogen	Nitrite/ Nitrate Nitrogen	Potassium	Quality Codes - Water Temperature	Salinity (Field)	Sodium Total	Sulphate	Sulphide Total	Sum of Anions + Cations	Suspended Solids	Temp.	Total Anions	Total Cations
	(g/m³)	(g/m³)	(g/m³)	(g/m³-N)	(g/m³-N)	(g/m³)			(g/m³)	(mg/L)	(g/m³)	(meq/L)	(g/m³)			
22/06/1993			<0.02000		2.2	1.5			8.9	9.6			<1.000			
23/08/1993			<0.02000		2.2	1.7			6.5	7.8			<1.000			
29/11/1993			<0.02000		1.1	1.5			5.9	6.8			3			
18/01/1994																
21/02/1994			<0.02000		2.9	1.5			11	11			4			
3/05/1994			<0.02000		2.4	1.5			7	11			<1.000			
5/10/1994					2.4											
30/11/1994					1.6											
16/02/1995			<0.2000		2	1.6	0		6	8.1			1	17		
24/05/1995					2.4		0							15		
30/08/1995					2.7		0									
29/11/1995					1.5		0									
6/03/1996			<0.02000		2.3	1.7	0		6.5		8.7		<1.000	14		
29/05/1996					2.7		0									
28/08/1996					2.4		0									
11/12/1996					1.5		0							14		
26/02/1997					2.7	2.9	0		6.2	9			<3.000			
11/06/1997					3		0									
10/09/1997					3.4		0							14		
15/04/1998					3.2	2	0		7.1	10				17.5		
18/11/1998					2.7	2	0		6.7	10						
21/04/1999					2.4	1.8	0		6.1	11				16		
16/09/1999					3.4	1.9	0		8	9.5				13.2		
15/12/1999					2.5	1.6	0		6	8.3				12.9		
23/02/2000					6	7.8	0		5.9	11				14		
2/05/2000					3.9	2	0		8.9	11				14.6		
1/11/2000					4.1	1.67	0		5.22	10.7				11		
3/05/2001					3.3	2.1	0			11.5				12.6		
6/09/2001					1.5	1.3	0		3.8	9.6				8.19		
25/02/2002					5.02	1.63	0		12.7	10.4				17.13		
3/09/2002					5.34	0.77	0		11	14.2				8.9		
24/02/2003					5.25	0.82	0		11	15				15.02		
1/09/2003					2.54	1.5	0		10	15.3				12.83		
25/02/2004					5.16	1.6	0		11	16				15.11		
1/09/2004					4.86	1.7	0		12	15				6.52		
23/02/2005					4.92	1.9	0		8.7	13				16.53		
31/08/2005					5.14	1.68	0		10.6	12.6				8.34		
21/02/2006					5.46	2.09	0		11.5	16.8				20.18		
5/09/2006	<0.005000				5.2	1.8	0		12.9	18.1				9.41		
1/03/2007	<0.005000				5.15	2.19	0		11.3	18.6				15.85		
20/11/2008		0.006		3.51		2.06	0		8.12	12.6					1.22	1.55
18/03/2009		<0.005000		4.77		2.1	0		8.57	14.7				14	1.43	1.54
30/09/2009		<0.005000		5.6		2.12	0		7.22	12.4				14.03	1.44	1.54
17/03/2010		<0.005000		3.61		2.17	0		6.23	11.4				13.9	1.28	1.29
29/09/2010		<0.005000		4.94		2.17			9.36	13.6					1.59	1.53
16/03/2011		<0.005000		6.31		3.99			7.14	12.4					1.44	1.48
28/06/2011		<0.0005000		3.24		1.94			7.47	11.4					1.36	1.37
15/12/2011		<0.0005000		4		1.84			6.9	14.6					1.37	1.39
26/06/2012		<0.0005000		2.8		1.95		0.06	7.4	12.4					1.3	1.23
27/09/2012		<0.0005000		3.8		1.95			6.7	12.3					1.36	1.29
14/12/2012		0.0017		5.1		1.94			7.1	11.6					1.44	1.44
1/02/2013		<0.0005000		3.8		1.85			6.6	10.8					1.28	1.22
3/05/2013		<0.0005000		3.8		2			7.5	13.2					1.42	1.37
9/07/2013		<0.0005000		5.3		2.1			8.3	14					1.55	1.57
14/10/2013		<0.0005000		3.2		1.94			6.9	10.1					1.18	1.19
19/12/2013		<0.0005000		4.6		1.99			7.2	14.3					1.46	1.46
1/07/2014		<0.0005000		4.3		1.93			7.5	13.4					1.48	1.4
26/09/2014		<0.0005000		3.6		2.2			6.7	11		2.5			1.2	1.3
25/03/2015		<0.0005000		3.9		1.7			7.6	14		2.8			1.4	1.4
7/07/2015		<0.0005000		3.8		1.8			7.4	13		2.7			1.4	1.3
30/09/2015		0.00066		4.4		1.8			6.4	12		2.7			1.5	1.3
17/12/2015		<0.0005000		3.7		1.9			6.6	13		2.7			1.3	1.3
15/04/2016		0.00054		4.2		2.8			6.9	15		2.9			1.5	1.4
22/06/2016		<0.0005000		4.9		2.1			8.2	16		3.2			1.6	1.5
22/09/2016		<0.0005000		5.8		2.2			7.2	12		3.2			1.6	1.6
11/01/2017		<0.0005000		4.9		3.1			7.7	14		3			1.5	1.5
14/03/2017		<0.0005000		4.7		2.2			7.2	14		2.9			1.5	1.4
14/06/2017		<0.0005000		5.7		2.3			9.8	17		3.5			1.8	1.8
12/12/2017		<0.0005000		5.8	5.8	2.2			7.2	14.2					1.5	1.55
25/01/2018		<0.0005000		5.9	5.9	2.1			7.6	13.5					1.61	1.55
13/03/2018		<0.0005000		5.7	5.7	2.3			9.2	15.6					1.69	1.7

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Date	Total Dissolved Phosphorus	Total Dissolved Solids	Total Hardness	Total Nitrogen	Total Organic Carbon	Total Phosphorus	Total carbon dioxide	Turbidity	Water Level	Water Temp. (Field)	Water Temp.	Zinc Total	meq/L Diff.	pH	pH (Field)
	(g/m³-P)			(g/m³)		(g/m³-P)	(g/m³)	(NTU)	(m)	(°C)	(°C)	(g/m³)	(meq/L)		(pH)
22/06/1993			50	2.2		<0.005000	100	0.2				<0.02000		6.3	
23/08/1993			37	2.2		0.006	51	0.4				<0.02000		6.5	
29/11/1993			31	1.2		0.009	72	0.1				<0.02000		6.5	
18/01/1994															
21/02/1994			49			0.007	82	0.4				0.33		6.2	
3/05/1994			59			0.02	71	0.2				0.03		6	
5/10/1994															
30/11/1994															
16/02/1995			34.03			0.016		0.25			0.017	<0.02000		6.4	
24/05/1995				2.5							0.015				
30/08/1995											0.015				
29/11/1995											0.014				
6/03/1996	0.018					0.018		0.25			0.014	<0.02000			
29/05/1996											0.014				
28/08/1996											0.014				
11/12/1996											0.014			6.3	
26/02/1997						0.033		0.6			0.014	<0.005000			
11/06/1997											0.014			6.6	
10/09/1997											0.014				
15/04/1998				3.2	1.4						0.018			5.9	
18/11/1998											0.017			6.7	
21/04/1999				2.7	0.9						0.016				6.1
16/09/1999		100		4.5	1	0.039					0.013	0.016		5.8	
15/12/1999				2.8	1						0.013				5.84
23/02/2000				5.5	2.6						0.014				5.88
2/05/2000				4.2	0.9						0.015				5.8
1/11/2000					<3.000						0.011			6.1	
3/05/2001				5	2.3						0.013				6.3
6/09/2001				2	1.4						0.008				6.55
25/02/2002				5.08	1.3				4.665		0.017				6.17
3/09/2002				5.34	0.8				6.9		0.009				6.38
24/02/2003				5.71	1.3				4.06		0.015				6.65
1/09/2003				4.77	0.6				5.15		0.013				6.57
25/02/2004				4.66	0.8				6.32		0.015				6.2
1/09/2004				4.87	0.6				5.97		0.007				7.16
23/02/2005				5.16	2.2				4.49		0.017			6.68	
31/08/2005				5.16	0.8				8.18		0.008				6.77
21/02/2006				5.52	1.6				4.83		0.02				6.29
5/09/2006				5.48		0.016					0.009				6.57
1/03/2007	0.037			5.27	1	0.037			4.68		0.016				6.95
20/11/2008		78	54								0.014			6.4	
18/03/2009		84	56								0.014			6.7	6.49
30/09/2009		81	59								0.014			6.8	6.81
17/03/2010		69	48								0.014			6.7	7.09
29/09/2010			54							12.96				6.6	8.19
16/03/2011			53							14.96				6.8	6.33
28/06/2011			50							13.82				7.2	7.67
15/12/2011			52							13.34				7	6.39
26/06/2012			43							13.16				7.6	6.34
27/09/2012			47							12.99				7.1	6.45
14/12/2012			54							12.96				6.8	6.76
1/02/2013			44											7.1	
3/05/2013			50							14.02				6.7	7.11
9/07/2013			58							13.33				7.1	8.16
14/10/2013			42							13.4				7.1	8.61
19/12/2013			55							12.75				7.5	7.24
1/07/2014			51							13.65				7.3	7.6
26/09/2014			45							12.78			0.031	6.8	6.46
25/03/2015			50							14.37			0.072	6.7	6.92
7/07/2015			48							13.14			0.1	6.9	7.77
30/09/2015			48							13.76			0.18	6.9	7.18
17/12/2015			49							12.78			0.033	7	7.83
15/04/2016			51							14.6			0.12	6.6	6.43
22/06/2016			56							14.19			0.11	6.8	7.09
22/09/2016			60							12.97			0.033	6.6	6.51
11/01/2017			54							12.74			0.011	6.4	6.25
14/03/2017			53							13.81			0.024	6.7	6.47
14/06/2017			65							14.01			0.03	6.9	6.81
12/12/2017			59	5.7		0.023				12.46				6.8	6.04
25/01/2018			58	5.5		0.026				13.15				6.9	5.67
13/03/2018			62	5.2		0.032				14.25				7	5.76

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Date	Alkalinity Hydroxide	Alkalinity (HCO3)	Alkalinity Total	Ammoniacal Nitrogen	Arsenic Dissolved	Boron Total	Calcium Dissolved	Calcium Hardness	Calcium Total	Carbonate Alkalinity	Chloride	Conductivity (Field)	Conductivity (Lab)	Dissolved Inorganic Carbon	Dissolved Organic Carbon	Dissolved Oxygen	Dissolved Oxygen Saturation (Field)	Dissolved Reactive Phosphorus
	(mg/L)	(g/m^3)	(g/m^3)	(g/m^3-N)	(g/m^3)	(g/m^3)	(g/m^3)		(g/m^3)		(g/m^3)	(mS/cm)	(mS/m)	(mg/L)	(g/m^3)	(mg/L)	(%)	(g/m^3-P)
22/06/2016	<1.000	32	26	<0.005000	<0.0001000	<0.00500	8.4			<0.600	1.8	0.075	8					0.005
22/08/2016	<1.000	32	26	<0.005000	<0.0001000	<0.00500	11	26	10	<1.000	4.2	0.095	10.3			5.23	49.3	0.005
22/09/2016	<1.000	31	25	<0.005000	<0.0001000	0.007	10	24	9.7	<1.000	3.7	0.092	9.8			5.31	49.4	0.005
28/10/2016	<1.000	31	25	<0.005000	<0.0001000	0.0065	9	23	9	<1.000	2.9	0.083	8.8			8.7	80.8	0.006
22/12/2016	<1.000	32	26	<0.005000	<0.0001000	0.0051	11	23	9.4	<1.000	3.1	0.085	9.1			4.75	44.5	0.005
23/01/2017	<1.000	36	29	<0.005000	<0.0001000	0.0071	10	26	10	<1.000	2.8	0.087	9.4			4.42	41.8	0.005
24/02/2017	<1.000	34	28	0.0086	<0.0001000	0.0063	9	23	9.4	<1.000	2.4	0.085	8.9			4.52	45.2	0.005
29/03/2017	<1.000	34	28	<0.005000	<0.0001000	<0.00500	9.4	25	10	<1.000	2.6	0.088	9.4			3.48	33.3	0.007
20/04/2017	<1.000	35	28	0.0077	<0.0001000	<0.00500	10	23	9.2	<1.000	2.8	0.087	9.5			3.28	31.4	0.005
23/05/2017	<1.000	34	27	<0.005000	<0.0001000	<0.00500	9.9	25	9.8	<1.000	3.2	0.091	9.6			3.8	36.5	0.005
22/06/2017	<1.000	34	28	<0.005000	<0.0001000	<0.00500	9.4	24	9.5	<1.000	2.4	0.085	9	7	0.8	4.37	41.5	0.005
18/07/2017	<1.000	31	26	<0.005000	<0.0001000	<0.00500	8.6	22	9.1	<1.000	2.1	0.079	8.3			4.89	46.4	0.006
12/12/2017	<1.000	30	30	<0.005000	<0.001000		10.1			<1.000	2.1	0.077	8.4			4.56	42.1	0.0021
25/01/2018	<1.000	30	30	<0.005000	<0.001000		9.4			<1.000	1.8	0.085	8.5			3.23	35.6	0.0023

Date	E-Coli MPN	Free carbon dioxide	Ion Balance	Iron Dissolved	Magnesium Hardness	Magnesium Total	Manganese Dissolved	Nitrate Nitrogen	Potassium	Silicon as Silica Total	Sodium Total	Static Water Level	Sulphate	Sum of Anions + Cations	Total Anions	Total Cations	Total Dissolved Carbon	Total Hardness	Water Temperature [Water Temperature (Field)]	meq/L Difference	pH	pH (Field)
	(cfu/100 mL)	(g/m^3)		(g/m^3)		(g/m^3)	(g/m^3)	(g/m^3-N)	(g/m^3)	(mg/L)	(g/m^3)	(m)	(mg/L)	(meq/L)					(°C)	(meq/L)		(pH)
22/06/2016	<1.60	14	0.012	0.023		1.6	0.00079	0.6	1.7		3.50	4.079	6.4	1.5	0.75	0.75		28	13.68	0	6.6	6.45
22/08/2016	<1.60	11	1.6	0.041	8.1	2	<0.0005000	1.7	1.9		3.60	4.051	8.3	1.8	0.93	0.91		34	12.81	0.029	6.8	6.27
22/09/2016		16	1.1	0.036	7.5	1.8	<0.0005000	1.4	1.8		3.5	4.097	8	1.7	0.88	0.86		32	12.12	0.02	6.6	6.43
28/10/2016		13	1.4	0.01	6.9	1.7	<0.0005000	1	1.8		3.3	4.054	7.4	1.6	0.81	0.79		29	11.98	0.022	6.7	6.38
22/12/2016		14	3.7	0.04	6.9	1.7	<0.0005000	0.99	1.7		3.2	3.975	6.9	1.7	0.82	0.89		30	12.4	0.064	6.7	6.02
23/01/2017		16	0.76	0.023	8.8	2.1	<0.0005000	0.97	1.7		4	3.518	7	1.7	0.88	0.87		37	12.82	0.013	6.7	5.96
24/02/2017		15	3.1	0.01	7.1	1.7	0.00058	0.85	1.5		3.5	3.467	6.8	1.6	0.83	0.78		30	13.71	0.05	6.6	6.4
29/03/2017	<1.60	15	2.8	0.038	7.3	1.8	0.001	0.99	1.7		3.6	3.742	6.9	1.7	0.85	0.81		32	13.19	0.046	6.6	5.99
20/04/2017	<1.60	17	1.1	0.018	7.4	1.8	<0.0005000	1.1	1.7		3.6	3.764	7.3	1.7	0.87	0.85		30	13.3	0.02	6.5	6.26
23/05/2017	<1.60	18	1.5	0.017	7.2	1.7	<0.0005000	1.1	1.8		3.6	4.026	7.3	1.7	0.87	0.85		32	13.47	0.025	6.5	5.5
22/06/2017	<1.60	8.5	0.69	0.019	7.1	1.7	<0.0005000	0.79	1.8	8.6	3.6	4.069	6.9	1.6	0.83	0.82	7.8	31	13	0.011	6.9	4.8
18/07/2017	<1.60	14	1.1	0.02	6.3	1.6	<0.0005000	0.66	1.7		3.3	4.082	6.7	1.5	0.76	0.74		29	13.06	0.017	6.7	5.98
12/12/2017	<1.00	4.8	0.75	<0.02000		1.77	<0.0005000	0.69	1.85		3.5	3.712	6		0.84	0.85		32	11.78		7.1	6.36
25/01/2018	2	8.7	0.4	0.03		1.75	<0.0005000	0.7	1.82		3.5	3.728	5.5		0.81	0.81		31	13.18		6.8	5.95

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Date	Alkalinity Hydroxide	Alkalinity (HCO3)	Alkalinity Total	Ammoniacal Nitrogen	Arsenic Dissolved	Boron Total	Calcium Dissolved	Calcium Hardness	Calcium Total	Carbonate Alkalinity	Chloride	Conductivity (Field)	Conductivity (Lab)	Dissolved Inorganic Carbon	Dissolved Organic Carbon	Dissolved Oxygen	Dissolved Oxygen Saturation (Field)	Dissolved Reactive Phosphorus	E-Coli MPN
	(mg/L)	(g/m³)	(g/m³)	(g/m³-N)	(g/m³)	(g/m³)	(g/m³)		(g/m³)		(g/m³)	(mS/cm)	(mS/m)	(mg/L)	(g/m³)	(mg/L)	(%)	(g/m³-P)	(cfu/100 mL)
22/06/2016	<1.000	88	72	0.073	<0.0001000	<0.005000	18			<0.600	9.9	0.238	24.3					<0.00200	<1.60
22/08/2016	<1.000	83	68	0.041	<0.0001000	<0.005000	20	51	21	<1.000	12	0.253	27.1			0.92	9	0.008	<1.60
22/09/2016	<1.000	88	72	0.51	<0.0001000	0.026	21	48	19	<1.000	11	0.262	26.9			0.5	5.1	0.004	
28/10/2016	<1.000	81	66	<0.005000	<0.0001000	0.026	19	48	19	<1.000	10	0.247	25.9			0.7812	7.6	0.01	
22/12/2016	<1.000	92	75	0.11	<0.0001000	0.024	21	48	19	<1.000	12	0.256	26			1.58	16.6	0.004	
23/01/2017	<1.000	81	66	0.013	<0.0001000	0.027	20	51	20	<1.000	10	0.242	25.1			0.82	8.8	0.01	
24/02/2017	<1.000	85	70	0.045	<0.0001000	0.026	18	47	19	<1.000	9.8	0.237	24			0.7	7.6	0.005	
29/03/2017	<1.000	76	62	<0.005000	<0.0001000	0.022	19	48	19	<1.000	10	0.24	24.9			1.31	12.3	0.009	<1.60
20/04/2017	<1.000	78	64	0.018	<0.0001000	0.021	21	48	19	<1.000	12	0.246	26.7			0.95	9.6	0.005	<1.60
23/05/2017	<1.000	77	63	<0.005000	<0.0001000	0.012	19	49	20	<1.000	11	0.246	26			1.04	10.1	0.007	<9.00
22/06/2017	<1.000	99	81	0.12	<0.0001000	0.013	19	49	20	<1.000	10	0.239	9	16	1.1	1.09	11.1	0.005	<1.60
18/07/2017	<1.000	80	66	<0.005000	<0.0001000	0.016	21	52	22	<1.000	12	0.265	27.2			1.86	17.6	0.008	<1.60
12/12/2017	<1.000	70	70	<0.005000	<0.001000		23			<1.000	12.8	0.27	29.7			2.1	19.5	0.0018	3
25/01/2018	<1.000	65	65	0.008	<0.001000		19.8			<1.000	11.2	0.251	25.4			1.01	10.3	0.0021	<1.00

Date	Free carbon dioxide	Ion Balance	Iron Dissolved	Magnesium Hardness	Magnesium Total	Manganese Dissolved	Nitrate Nitrogen	Potassium	Silicon as Silica Total	Sodium	Static Water Level	Sulphate	Sum of Anions + Cations	Total Anions	Total Cations	Total Dissolved Carbon	Total Hardness	Water Temp. (Field)	meq/L Difference	pH	pH (Field)
	(g/m³)		(g/m³)		(g/m³)	(g/m³)	(g/m³-N)	(g/m³)	(mg/L)	(g/m³)	(m)	(mg/L)	(meq/L)					(°C)	(meq/L)		(pH)
22/06/2016	33	3.5	1.7		5.4	0.12	2.8	2.1		19	6.866	24	4.7	2.4	2.3		67	11.76	0.16	6.6	6.94
22/08/2016	27	3.1	0.085	26	6.2	0.0025	4.2	2.3		20	5.974	29	5	2.6	2.4		77	14.55	0.16	6.8	6.3
22/09/2016	36	0.83	1.2	25	6	0.026	2.8	2.2		20	6.206	29	5.2	2.6	2.6		72	16.84	0.043	6.7	6.63
28/10/2016	50	0.35	0.0046	23	5.7	<0.000500	3.8	2.3		20	6.311	27	4.9	2.5	2.4		71	12.91	0.017	6.5	6.4
22/12/2016	30	2.4	0.44	24	5.9	0.017	3.4	2.2		20	6.743	28	5.2	2.7	2.5		72	18.37	0.13	6.8	6.45
23/01/2017	40	0.23	0.087	27	6.6	0.0043	3.6	2.1		22	6.493	26	4.8	2.4	2.4		82	19.11	0.011	6.6	6.47
24/02/2017	30	6	<0.002000	23	5.5	0.026	3.8	2		19	6.279	25	4.7	2.5	2.2		69	18.58	0.28	6.7	6.38
29/03/2017	36	2.9	0.0038	23	5.6	<0.000500	4	2		19	5.934	25	4.6	2.4	2.2		71	12.62	0.13	6.5	6.27
20/04/2017	35	2	0.063	24	5.8	0.0074	4.4	2.3		19	5.444	28	5	2.5	2.4		72	15.69	0.1	6.6	6.47
23/05/2017	35	1.8	0.011	23	5.7	0.001	4.1	2.1		19	5.564	26	4.7	2.4	2.3		72	13.97	0.087	6.6	6.13
22/06/2017	20	1.7	0.079	24	5.8	0.057	0.68	2.2	16	19	5.983	23	4.8	2.4	2.3	17	73	16	0.079	7	5.36
18/07/2017	40	2	0.005	25	6.6	<0.000500	4.9	2.3		20	6.293	29	5.1	2.6	2.5		82	12.62	0.1	6.6	6.35
12/12/2017	10.5	1.24	<0.02000		7.5	<0.000500	5.7	2.6		24	5.987	29		2.8	2.9		87	12		7.1	6.13
25/01/2018	17.8	1.96	<0.02000		6	0.0021	4.7	2.2		19.4	5.876	26		2.5	2.4		74	15.85		6.9	6.02

J410576

Date	Alkalinity Hydroxide	Alkalinity (HCO3)	Alkalinity Total	Ammoniacal Nitrogen	Arsenic Dissolved	Boron Total	Calcium Dissolved	Calcium Hardness	Calcium Total	Carbonate Alkalinity	Chloride	Conductivity (Field)	Conductivity (Lab)	Dissolved Inorganic Carbon	Dissolved Organic Carbon	Dissolved Oxygen	Dissolved Oxygen Saturation (Field)	Dissolved Reactive Phosphorus
	(mg/L)	(g/m³)	(g/m³)	(g/m³-N)	(g/m³)	(g/m³)	(g/m³)		(g/m³)		(g/m³)	(mS/cm)	(mS/m)	(mg/L)	(g/m³)	(mg/L)	(%)	(g/m³-P)
7/06/2016	<1.000	79	65	0.007	<0.001000	<0.05000	23			<0.600	7.7	0.234	24.2			4.71		0.013
25/08/2016	<1.000	52	43	0.009	0.00011	<0.05000	20	51	20	<1.000	11	0.217	22.9			0.91	8.5	0.011
22/09/2016	<1.000	57	47	0.013	<0.0001000	0.016	19	46	19	<1.000	8.5	0.211	22			0.41	3.8	0.009
28/10/2016	<1.000	83	48	0.008	<0.0001000	0.017	19	47	20	<1.000	8.4	0.209	21.9			5.14	47.3	0.012
22/12/2016	<1.000	53	43	0.005	<0.0001000	0.015	20	49	20	<1.000	10	0.217	22.7			1.78	16.7	0.013
23/01/2017	<1.000	53	43	0.028	<0.0001000	0.019	21	54	22	<1.000	11	0.221	23.4			1.12	10.6	0.016
24/02/2017	<1.000	52	42	<0.005000	<0.0001000	0.017	20	50	20	<1.000	8.9	0.22	22.3			1.11	10.4	0.012
29/03/2017	<1.000	29	24	<0.005000	0.00011	0.015	20	52	21	<1.000	12	0.277	23.7			1.39	13	0.014
20/04/2017	<1.000	52	43	0.0085	<0.0001000	0.014	21	50	20	<1.000	13	0.223	24.3			1.02	9.6	0.01
23/05/2017	<1.000	52	42	<0.005000	<0.0001000	0.0062	21	53	21	<1.000	13	0.135	24.9			0.79	9	0.012
22/06/2017	<1.000	60	49	<0.005000	<0.0001000	<0.005000	20	51	21	<1.000	8.1	0.226	25.4	9.3	1.3	0.71	6.6	0.007
18/07/2017	<1.000	59	49	0.006	<0.0001000	<0.005000	20	49	21	<1.000	8.1	0.217	21.8			1.14	10.6	0.007
12/12/2017	<1.000	43	43	<0.005000	<0.001000		22			<1.000	11.7	0.212	23.4			1.2	11.2	0.01
25/01/2018	<1.000	47	47	<0.005000	<0.001000		21			<1.000	9.3	0.224	22.9			0.73	6.9	0.0073

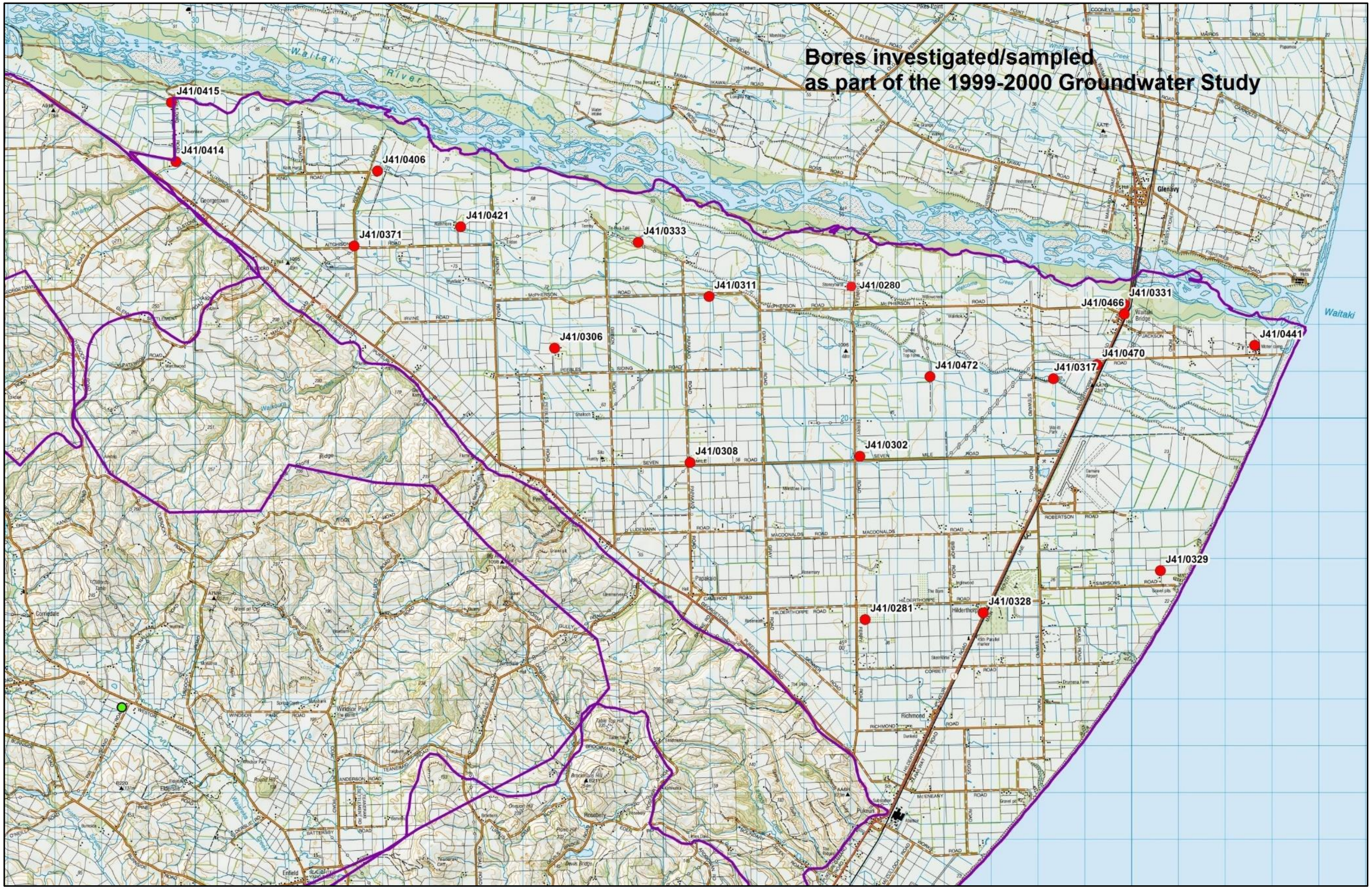
Date	E-Coli MPN	Free carbon dioxide	Ion Balance	Iron Dissolved	Magnesium Hardness	Magnesium Total	Manganese Dissolved	Nitrate Nitrogen	Potassium	Silicon as Silica Total	Sodium Total	Sulphate	Sum of Anions + Cations	Total Anions	Total Cations	Total Dissolved Carbon	Total Hardness	Water Temperature (Field)	meq/L Difference	pH	pH (Field)
	(cfu/100 mL)	(g/m³)		(g/m³)		(g/m³)	(g/m³)	(g/m³-N)	(g/m³)	(mg/L)	(g/m³)	(mg/L)	(meq/L)					(°C)	(meq/L)		(pH)
7/06/2016	<1.60	22	1.7	<0.02000		6	<0.005000	5.8	1.3		13	19	4.6	2.3	2.3		82	5.95	0.08	6.8	5.21
25/08/2016	<1.60	67	1	0.32	24	5.8	0.0063	6.7	1.8		12	22	4.1	2.1	2.1		75	12.53	0.043	6.2	6.12
22/09/2016		58	1.6	0.28	22	5.3	0.004	5.8	1.4		11	21	4	2	2		68	12.48	0.063	6.3	6.3
28/10/2016		52	1.4	0.16	23	5.4	0.006	5.6	1.6		12	21	4	2	2		70	12.6	0.057	6.5	6.32
22/12/2016		55	0.99	0.17	23	5.6	0.0036	6.4	2.2		11	22	4.1	2.1	2		72	12.47	0.04	6.3	5.92
23/01/2017		65	0.16	0.23	27	6.6	0.0029	7	1.6		14	23	4.3	2.1	2.1		86	12.52	0.007	6.2	5.97
24/02/2017		56	2.3	0.15	23	5.5	0.0034	6.6	1.5		12	21	3.9	2	1.9		73	12.78	0.089	6.2	5.9
29/03/2017	<1.60	34	5.5	0.23	23	5.6	0.0036	7.1	1.7		13	23	3.8	1.8	2		76	12.6	0.21	6.2	5.9
20/04/2017	<1.60	56	1.5	0.28	23	5.7	0.005	7.1	1.7		12	24	4.4	2.2	2.2		73	12.5	0.065	6.2	6.24
23/05/2017	<1.60	64	2.2	0.32	24	5.7	0.0059	7.3	1.7		13	24	4.4	2.2	2.1		76	12.79	0.096	6.1	5.89
22/06/2017	<1.60	38	1.4	0.35	23	5.6	0.0073	6.2	1.5	18	12	21	4.1	2.1	2	11	75	12.39	0.059	6.5	5.13
18/07/2017	<1.60	46	3	0.34	22	5.8	0.0074	6.1	1.5		11	22	4.1	2.1	2		75	12.51	0.12	6.4	6.07
12/12/2017	37	9.6	0.24	<0.02000		6.6	0.0013	7.6	1.84		12.9	23		2.2	2.2		82	12.15		7	5.79
25/01/2018	<1.00	25	2.9	<0.02000		6	0.0014	7.5	1.71		12	23		2.2	2.1		76	12.81		6.6	5.7

J410586

Date	Alkalinity Hydroxide	Alkalinity (HCO3)	Alkalinity Total	Ammoniacal Nitrogen	Arsenic Dissolved	Boron Total	Calcium Dissolved	Calcium Hardness	Calcium Total	Carbonate Alkalinity	Chloride	Conductivity (Field)	Conductivity (Lab)	Dissolved Inorganic Carbon	Dissolved Organic Carbon	Dissolved Oxygen	Dissolved Oxygen Saturation (Field)	Dissolved Reactive Phosphorus
	(mg/L)	(g/m³)	(g/m³)	(g/m³-N)	(g/m³)	(g/m³)	(g/m³)		(g/m³)		(g/m³)	(mS/cm)	(mS/m)	(mg/L)	(g/m³)	(mg/L)	(%)	(g/m³-P)
7/06/2016	<1.000	55	45	0.005	<0.001000	<0.0500	17			<0.600	5.3	0.186	19.4			3.6		0.008
22/08/2016	<1.000	58	47	<0.005000	<0.0001000	<0.0050	17	43	17	<1.000	5.4	0.186	19.8			3.17	30.1	0.01
22/09/2016	<1.000	55	45	<0.005000	<0.0001000	0.015	18	40	16	<1.000	5.4	0.187	19.6			3.45	32.2	0.01
28/10/2016	<1.000	56	46	<0.005000	<0.0001000	0.015	17	41	16	<1.000	5.6	0.186	19.6			3.05	29.3	0.009
22/12/2016	<1.000	54	45	<0.005000	<0.0001000	0.014	17	42	17	<1.000	5.5	0.186	19.3			3.89	37.7	0.007
23/01/2017	<1.000	57	47	<0.005000	<0.0001000	0.017	18	48	19	<1.000	5.5	0.188	19.7			3.27	31	0.009
24/02/2017	<1.000	56	46	<0.005000	<0.0001000	0.015	17	43	17	<1.000	5.4	0.188	19.2			3.05	29.6	0.008
29/03/2017	<1.000	56	46	<0.005000	<0.0001000	0.013	17	43	17	<1.000	5.4	0.189	19.8			3.35	31.8	0.01
20/04/2017	<1.000	56	46	0.0067	<0.0001000	0.011	18	41	16	<1.000	5.6	0.185	20			3.29	31.9	0.007
23/05/2017	<1.000	54	44	<0.005000	<0.0001000	<0.0050	17	43	17	<1.000	5.6	0.193	19.9			3.46	32.7	0.007
22/06/2017	<1.000	57	47	<0.005000	<0.0001000	<0.0050	18	44	18	<1.000	5.4	0.192	22.8	8.3	0.9	3.63	34.1	0.007
18/07/2017	<1.000	56	46	0.013	<0.0001000	<0.0050	17	43	18	<1.000	5.7	0.194	20			2.89	25.4	0.006
12/12/2017	<1.000	45	46	<0.005000	<0.001000		18.9			<1.000	5.5	0.177	19.6			3.48	32.4	0.003
25/01/2018	<1.000	48	48	<0.005000	<0.001000		17.7			<1.000	5.6	0.195	19.7			3.96	37.8	0.0031

Date	E-Coli MPN	Free carbon dioxide	Ion Balance	Iron Dissolved	Magnesium Hardness	Magnesium Total	Manganese Dissolved	Nitrate Nitrogen	Potassium	Silicon as Silica Total	Sodium Total	Static Water Level	Sulphate	Sum of Anions + Cations	Total Anions	Total Cations	Total Dissolved Carbon	Total Hardness	Water Temp. (Field)	meq/L Difference	pH	pH (Field)
	(cfu/100mL)	(g/m³)		(g/m³)		(g/m³)	(g/m³)	(g/m³-N)	(g/m³)	(mg/L)	(g/m³)	(m)	(mg/L)	(meq/L)					(°C)	(meq/L)		(pH)
7/06/2016	<1.60	24	2.2	0.028		4.4	<0.005000	5.7	1.6		12.00		18	3.6	1.8	1.7		60	11.23	0.079	6.6	5.51
22/08/2016	<1.60	24	3.3	0.027	19	4.5	0.0027	5.5	1.6		12	2.947	18	3.6	1.9	1.7		61	13.02	0.12	6.7	6.18
22/09/2016		37	0.89	0.0039	18	4.4	<0.0005000	5.7	1.5		11	3.064	18	3.6	1.8	1.8		59	12.26	0.032	6.5	6.47
28/10/2016		36	1.3	0.0098	18	4.4	0.0038	5.4	1.6		11	3.017	18	3.6	1.8	1.8		59	12.36	0.047	6.5	6.31
22/12/2016		28	1.4	0.0035	18	4.4	<0.0005000	5.3	1.6		11	2.435	17	3.5	1.8	1.7		59	13.98	0.05	6.6	6.18
23/01/2017		36	0.18	0.0068	22	5.2	<0.0005000	5.4	1.6		13	2.371	17	3.7	1.8	1.8		70	12.29	0.007	6.5	6.15
24/02/2017		28	3.8	<0.002000	18	4.4	0.00062	5.7	1.5		11	2.477	17	3.5	1.8	1.7		61	14.03	0.13	6.5	6.32
29/03/2017	8.2	33	4.1	0.008	18	4.4	0.0011	5.7	1.5		12	2.634	17	3.5	1.8	1.7		62	13.11	0.15	6.5	6.13
20/04/2017	<1.60	32	1.6	0.0059	18	4.3	0.00053	5.8	1.6		11	2.449	18	3.7	1.9	1.8		59	12.6	0.059	6.5	6.45
23/05/2017	<1.60	35	2.3	0.0087	18	4.5	0.00072	5.9	1.5		11	2.895	18	3.6	1.8	1.8		62	12.84	0.082	6.4	5.74
22/06/2017	<1.60	23	2	0.011	20	4.7	0.00072	6.3	1.6	18	12	3.054	17	3.7	1.9	1.8	9.2	63	12.46	0.074	6.7	4.97
18/07/2017	<1.60	36	2.6	0.078	19	4.8	0.0078	6.3	1.8		11	3.104	17	3.7	1.9	1.8		66	9.6	0.097	6.5	6.2
12/12/2017	<1.00	10.4	1.39	<0.02000		5.4	<0.0005000	6.9	1.73		12.2	2.716	16.6		1.91	1.96		69	12.17		6.9	6.08
25/01/2018	<1.00	14.5	3	<0.02000		4.9	<0.0005000	6.8	1.67		11.6	2.543	16.6		1.94	1.83		64	13.3		6.8	5.88

Appendix D: Location of the bores sampled during the 1999-2000 investigations (ORC, 2000)



Appendix E: Criteria and threshold concentrations for identifying redox processes in groundwater (McMahon and Chapelle, 2008)

Table 1. Criteria and threshold concentrations for identifying redox processes in ground water.

[Table was modified from McMahon and Chapelle, 2008. Redox process: O2, oxygen reduction; NO3, nitrate reduction; Mn(IV), manganese reduction; Fe(III), iron reduction; SO4, sulfate reduction; CH4gen, methanogenesis. Chemical species: O2, dissolved oxygen; NO3-, dissolved nitrate; MnO2(s), manganese oxide with managanese in 4+ oxidation state; Fe(OH)3(s), iron hydroxide with iron in 3+ oxidation state; FeOOH(s), iron oxyhydroxide with iron in 3+ oxidation state; SO42–, dissolved sulfate; CO2(g), carbon dioxide gas; CH4(g), methane gas. Abbreviations: mg/L, milligram per liter; —, criteria do not apply because the species concentration is not affected by the redox process; ≤, less than or equal to; ≥, greater than or equal to; <, less than; >, greater than]

			Criteria for inferring process from water-quality data					
Redox category	**Redox process**	**Electron acceptor (reduction) half-reaction**	**Dissolved oxygen (mg/L)**	**Nitrate, as Nitrogen (mg/L)**	**Manganese (mg/L)**	**Iron (mg/L)**	**Sulfate (mg/L)**	**Iron/sulfide (mass ratio)**
Oxic	O2	$O_2 + 4H^+ + 4e^- \rightarrow 2H_2O$	≥0.5	—	<0.05	<0.1	—	
Suboxic	Suboxic	Low O2; additional data needed to define redox process	<0.5	<0.5	<0.05	<0.1	—	
Anoxic	NO3	$2NO_3^- + 12H^+ + 10e^- \rightarrow N_{2(g)} + 6\ H_2O$; $NO_3^- + 10H^+ + 8e^- \rightarrow NH_4^+ + 3H_2O$	<0.5	≥0.5	<0.05	<0.1	—	
Anoxic	Mn(IV)	$MnO_{2(s)} + 4H^+ + 2e^- \rightarrow Mn^{2+} + 2H_2O$	<0.5	<0.5	≥0.05	<0.1	—	
Anoxic	Fe(III)/SO4	Fe(III) and (or) $SO4^{2-}$ reactions as described in individual element half reactions	<0.5	<0.5	—	≥0.1	≥0.5	no data
Anoxic	Fe(III)	$Fe(OH)_{3(s)} + H^+ + e^- \rightarrow Fe^{2+} + H_2O$; $FeOOH_{(s)} + 3H^+ + e^- \rightarrow Fe^{2+} + 2H_2O$	<0.5	<0.5	—	≥0.1	≥0.5	>10
Mixed(anoxic)	Fe(III)-SO4	Fe(III) and $SO4^{2-}$ reactions as described in individual element half reactions	<0.5	<0.5	—	≥0.1	≥0.5	≥0.3, ≤10
Anoxic	SO4	$SO_4^{2-} + 9H^+ + 8e^- \rightarrow HS^- + 4H_2O$	<0.5	<0.5	—	≥0.1	≥0.5	<0.3
Anoxic	CH4gen	$CO_{2(g)} + 8H^+ + 8e^- \rightarrow CH_{4(g)} + 2H_2O$	<0.5	<0.5	—	≥0.1	<0.5	

Appendix F: Redox assignments for J41/0571 and J41/0576 according to McMahon and Chapelle, 2008 thresholds

J41/0571

Sample ID	Redox Variables	Dissolved O_2	NO_3^- (as Nitrogen)	Mn^{2+}	Fe^{2+}	SO_4^{2-}	Sulfide (sum of H_2S, HS^-, S^{2-})	Redox Assignment		
	Units	millig/L	millig/L	millig/L	millig/L	millig/L	millig/L			
	Threshold values	0.5	0.5	0.05	0.1	0.5	none	Num of Params	General Redox Category	Redox Process
22/08/2016	Clear Redox Assignments	0.92	4.2	0.0025	0.085	29		5	Oxic	O2
22/09/2016		0.5	2.8	0.026	1.2	29		5	Mixed(oxic-anoxic)	O2-Fe(III)/SO4
28/10/2016		0.7812	3.8	0.00025	0.0046	27		5	Oxic	O2
22/12/2016		1.58	3.4	0.017	0.44	28		5	Mixed(oxic-anoxic)	O2-Fe(III)/SO4
23/01/2017		0.82	3.6	0.0043	0.087	26		5	Oxic	O2
24/02/2017	Assign Redox Categories and Processes	0.7	3.8	0.026	0.001	25		5	Oxic	O2
29/03/2017		1.31	4	0.00025	0.0038	25		5	Oxic	O2
20/04/2017		0.95	4.4	0.0074	0.063	28		5	Oxic	O2
23/05/2017		1.04	4.1	0.001	0.011	26		5	Oxic	O2
22/06/2017		1.09	0.68	0.057	0.079	23		5	Mixed(oxic-anoxic)	O2-Mn(IV)
18/07/2017		1.86	4.9	0.00025	0.005	29		5	Oxic	O2
12/12/2017		2.1	5.7	0.00025	0.01	29		5	Oxic	O2
25/01/2018		1.01	4.7	0.0021	0.01	26		5	Oxic	O2

J41/0576

Sample ID	Redox Variables	Dissolved O_2	NO_3^- (as Nitrogen)	Mn^{2+}	Fe^{2+}	SO_4^{2-}	Sulfide (sum of H_2S, HS^-, S^{2-})	Redox Assignment		
	Units	millig/L	millig/L	millig/L	millig/L	millig/L	millig/L			
	Threshold values	**0.5**	**0.5**	**0.05**	**0.1**	**0.5**	**none**	Num of Params	General Redox Category	Redox Process
7/06/2016	Clear Redox Assignments	4.71	5.8	0.0025	0.01	19		5	Oxic	O2
25/08/2016		0.91	6.7	0.0063	0.32	22		5	Mixed(oxic-anoxic)	O2-Fe(III)/SO4
22/09/2016		0.41	5.8	0.004	0.28	21		5	Mixed(anoxic)	NO3-Fe(III)/SO4
28/10/2016		5.14	5.6	0.006	0.16	21		5	Mixed(oxic-anoxic)	O2-Fe(III)/SO4
22/12/2016		1.78	6.4	0.0036	0.17	22		5	Mixed(oxic-anoxic)	O2-Fe(III)/SO4
23/01/2017	Assign Redox Categories and Processes	1.12	7	0.0029	0.23	23		5	Mixed(oxic-anoxic)	O2-Fe(III)/SO4
24/02/2017		1.11	6.6	0.0034	0.15	21		5	Mixed(oxic-anoxic)	O2-Fe(III)/SO4
29/03/2017		1.39	7.1	0.0036	0.23	23		5	Mixed(oxic-anoxic)	O2-Fe(III)/SO4
20/04/2017		1.02	7.1	0.005	0.28	24		5	Mixed(oxic-anoxic)	O2-Fe(III)/SO4
23/05/2017		0.79	7.3	0.0059	0.32	24		5	Mixed(oxic-anoxic)	O2-Fe(III)/SO4
22/06/2017		0.71	6.2	0.0073	0.35	21		5	Mixed(oxic-anoxic)	O2-Fe(III)/SO4
18/07/2017		1.14	6.1	0.0074	0.34	22		5	Mixed(oxic-anoxic)	O2-Fe(III)/SO4
12/12/2017		1.2	7.6	0.0013	0.01	23		5	Oxic	O2
25/01/2018		0.73	7.5	0.0014	0.01	23		5	Oxic	O2

Appendix G: Details of the Age Dating Analyses

Site Id: J41/0317

Sampling date: 14/03/2012

Tritium concentration		Atmospheric Partial Pressure in pptv (calculated)							
TU	±	SF6	±	CFC-11	±	CFC-12	±	CFC-113	±
2.141	0.038	6.51	0.48	379	27	899	59	67.9	6.1

measured concentration in solution by purge&trap/GC-ECD							
$fmolkg^{-1}$		$pmolkg^{-1}$		$pmolkg^{-1}$		$pmolkg^{-1}$	
SF_6	±	CFC-11	±	CFC-12	±	CFC-113	±
3.14	0.09	7.16	0.11	4.53	0.11	0.4	0.02

Stable isotopes		measured concentration in solution by purge&trap/GC-TCD					calculated variables (from Ar-N2)			
		$mL(STP).kg^{-1}$				$\mu mol.kg^{-1}$	rech temp		excess air	
$\delta^{18}O$	δ^2H	Ar	±	N_2	±	CH4	°C	±	$mL(STP).kg^{-1}$	±
-9.56	-70.42	0.391	0.006	15.88	0.19	<1	11.8	1.3	2.5	0.6

Letter Report No: CR 2017/50LR
Project No: 630W0567
Client reference: PO No. 48596

GNS SCIENCE TE PŪ AO

21 March 2017

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Otago Regional Council
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Dunedin 9054

Attention: Frederika Mourot

Dear Frederika

Groundwater residence time determination for the lower Waitaki wells J41/0302 and J41/0429

1.0 INTRODUCTION

This report provides an interpretation of groundwater residence times based on the results of tritium analyses for the wells J41/0302 and J41/0429, located in the Lower Waitaki River Catchment. Well details are presented in Table 1, and tritium concentrations and interpreted mean residence times in Table 2.

Table 1 Well details.[1]

Well ID	E [2]	N [2]	Well depth (m below ground level)
J41/0302	1444192	5019188	17
J41/0429	1445494	5015989	18

1. Well details from Otago Regional Council (Otago Regional Council 2016)
2. Coordinates are NZTM

Groundwater mean residence time (MRT) has been calculated using an exponential piston flow model (EPM) matched to the tritium concentrations (Table 2). The age distribution parameters for the wells have been estimated at 70% exponential mixed flow. For the tritium concentrations observed at these wells, use of other proportions of exponential mixed flow would alter the calculated MRT by less than one year.

The interpreted residence times are dependent on several assumptions. Firstly, the tritium input to the system is based the historical rainfall tritium record from Kaitoke, New Zealand, scaled to infer a local input function for the aquifer system. For this work a scaling factor of 1.3 has been used. Altering the scaling factor by ±0.1 leads to a change in the calculated MRTs of ± 2 years.

Secondly, he tritium input (and scaling of such) is also dependent on the nature of the recharge mechanism to the groundwater. For example, recharge from local rain will have a different input signal to recharge derived from the Waitaki River. The Waitaki River catchment is large and consists of a mix of high and low altitude sub-catchments, resulting in the river water itself having a residence time which would be in the order of several years. Thus, if the river is the main source of recharge to the wells, the groundwater MRT may actually be several years younger than that presented.

Table 2 Groundwater mean residence time.

Well ID	Sampling Date	Tritium [TR] [1]	Exponential mixed flow %	MRT (years)
J41/0302	29/04/2016	1.903 ± 0.031	70	5
J41/0429	29/04/2016	1.746 ± 0.035	70	7

1. Tritium concentrations are expressed as 3H:1H ratios where 1 tritium unit (TR) signifies a ratio of 1:1×1018.

2.0 SUMMARY

Tritium concentrations in water from both of the wells is high and indicate relatively young groundwaters with MRTs of 5 years for well J41/0302 and 7 years for well J41/0429. If the groundwater from these wells is recharged from the Waitaki River the actual groundwater MRT may be several years younger.

Yours sincerely

Rob van der Raaij
Scientist, Isotope Hydrology

Heather Martindale
(Reviewer)

APPENDIX 1: DETERMINATION OF GROUNDWATER RESIDENCE TIME USING TRITIUM, CFCS AND SF_6

Chlorofluorocarbons (CFCs) are entirely synthetic compounds. Significant production of CFCs began in the 1930s. Sulphur hexafluoride (SF_6) is predominantly anthropogenic with industrial production beginning in the 1950s. However, a small amount of SF_6 is also produced in certain volcanic minerals and fluids. Groundwater age-dating using CFCs and SF_6 is possible due to the steady increase in atmospheric concentrations of these gases since production began (Figure A2.1). These gases are dissolved in recharge waters and are isolated from the atmosphere when this recharge enters the groundwater zone. Thus the gases hold a record in the groundwater of past atmospheric concentrations. CFCs have been measured continuously in the atmosphere at various sites worldwide since the late 1970s but their concentrations have begun to decline since use of them was phased out following the Montreal Protocol in 1987 thus losing effectiveness for age-dating over this period (IAEA 2006).

After measured CFC and SF_6 concentrations in groundwater are corrected for excess air, they are used to calculate relative atmospheric concentrations using Henry's Law and an estimated recharge temperature. Excess air is air in excess of the equilibrium soluble amount at the given recharge temperature and is thought to originate by processes such as bubble entrapment occurring during recharge. The excess air correction and recharge temperature are calculated from the ratio of dissolved nitrogen and argon concentrations (Heaton and Vogel, 1981). These nitrogen and argon concentrations are measured simultaneously with the CFC concentrations. The calculated atmospheric concentrations are then used to calculate the CFC and SF_6 model residence times of the groundwater (Plummer and Busenberg, 2000).

Under certain circumstances, CFCs and SF_6 can undergo diffusive exchange processes in the unsaturated zone, increasing their concentrations in groundwater. In these cases the model ages derived from the CFC and SF_6 concentrations should be regarded as minimum ages for groundwater. CFCs are also susceptible to degradation processes underground, particularly in anoxic environments, and to contamination. SF_6 is less susceptible to these but is affected more by excess air and diffusion.

Tritium (^3H) is a component of the water molecule and thus forms an ideal tracer for groundwater studies. Age-dating using tritium is based on radioactive decay of tritium after rainwater penetrates the ground during recharge. The half-life of tritium is 12.32 years. Tritium is produced naturally by cosmic radiation in the upper atmosphere but was also released into the atmosphere by nuclear weapons testing. Figure A1.1 shows the history of the tritium concentration in rainfall; the peak in tritium concentration in the 1960s and early 1970s is a result of this testing (Stewart and Morgenstern, 2001). Tritium data may give ambiguous residence times, because of this irregularly shaped peak. Often this will be resolved by measuring the change in tritium concentration in groundwater over a time interval of a few years or by comparison to CFC and SF_6 data.

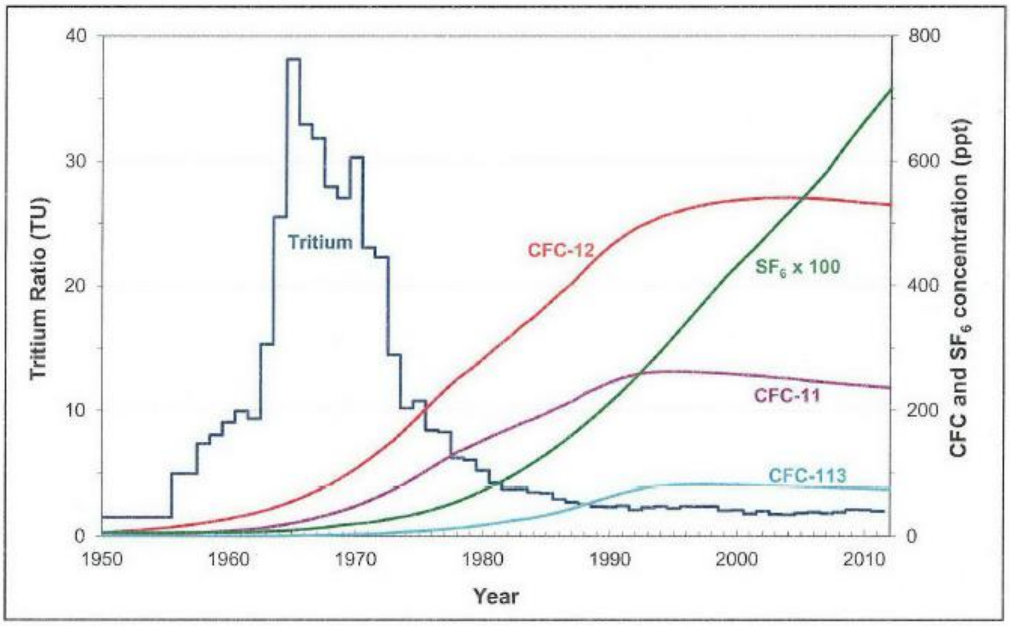

Figure A1.1 Tracer input curves. The tritium concentrations are from rainfall at Kaitoke, New Zealand, and have been measured monthly since the 1960s. CFC and SF_6 concentrations are for southern hemispheric air (IAEA 2006).

Groundwater extracted from a bore or other discharge point is a mixture of water with different ages due to the convergence of different flow lines within the aquifer at the discharge point (Figure A1.2). Groundwater age-dating therefore yields an average age of the water. To calculate the average age the distribution of groundwater age must also be determined. This distribution can be described using lumped-parameter mixing models. Lumped parameter models are a commonly used method of interpreting groundwater ages in scientific studies (Turnadge and Smerdon 2014) and are well-suited for characterisation of data-poor groundwater systems. Piston flow is a simplified approximation of no mixing of flow lines and is suitable for aquifers in which the recharge zone is narrow with respect to the overall distance from recharge zone to sampling point, while the exponential model describes complete mixing of the flow lines within a system. The mixing of different flow lines occurs at the sampling point. For more realistic scenarios which are intermediate between piston flow and exponential mixing, the exponential piston flow model (EPM) may be applied (Maloszewski and Zuber, 1982). The EPM has been applied successfully to groundwaters from many areas of New Zealand (Daughney et al., 2010; Morgenstern and Daughney 2012).

The EPM is described by two parameters - the MRT and the fraction of exponential mixed flow. The fraction of exponential mixed flow is a measure of the degree of mixing and reflects the distribution of travel-times of different components of groundwater around the MRT (Figure A1.3). The fraction of exponential mixed flow observed at the bore depends on the characteristics of the sampling point as well as the hydrogeologic attributes of the aquifer concerned (which affect the variety of possible flow paths that may be intersected by the bore). This fraction is best estimated by matching to the tritium data using a series of measurements separated in time by several years. If such a time series is not available, comparison of the tritium data to CFC and SF_6 data can sometimes be used for less precise estimates of the mixing fraction, but should be confirmed by future sampling.

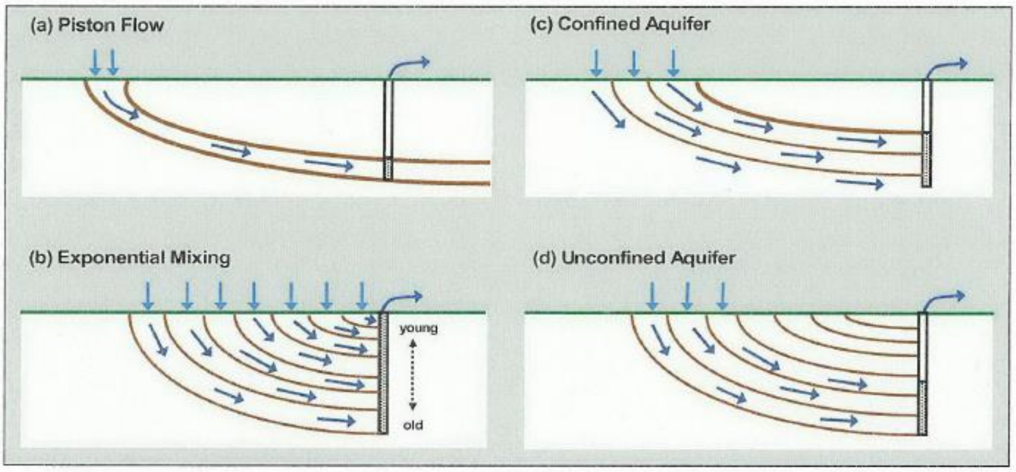

Figure A1.2 Conceptual groundwater flow situations which can be described by lumped parameter mixing models.

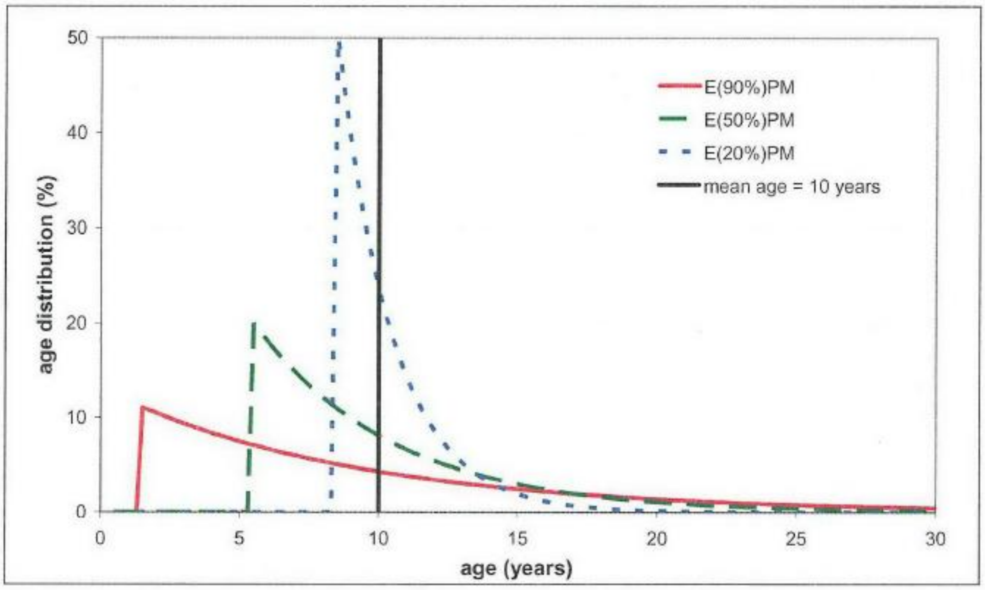

Figure A1.3 Age frequency distributions for the exponential piston flow model for MRT = 10 years, with typical parameter values (20%, 50% and 90% of the flow is exponential mixed flow).

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APPENDIX A1 - LAKE HAYES REMEDIATION OPTIONS

MAY | JUNE | JULY | AUGUST | SEPTEMBER | OCTOBER | NOVEMBER | DECEMBER

NIWA options assesment
completed in March

LTP starts 1 July

Economic assessment (from March)
Draft findings presented early June
Final report end June

Lake modelling study (Univ. Waikato)
~3 months from late May

Paper to technical Committee
Workshop with Council on options
1-Aug

Public Consultation on options and funding
4 weeks in this period

Analysis of Consultation

Preferred option identified
By end of 2018

Catchment study (ORC)
Planning and implementation
Catchment study sampling commences (from September)

Monitoring Buoy (Univ. Waikato)
Design and comissioning
Deployment

Arrow offtake pipe and discharge structure
Design, procurement and installation
[Must be installed end September]

APPENDIX A2 - OPTION IMPLEMENTATION TIMEFRAMES

2018		2019				2020			
July	October	January	April	July	October	January	April	July	October

Preferred option identified (Dec)

Arrow water augmentation

Offtake Installed

Complete offtake structure, obtiain discharge consents, legal agreements.

Augmentation operational | Reduced summer flow | Reduced su

Destratification

Identify site, building permission , procure compressor and pipes ?? ??

Summer operational period (if ready) | Summer operational p

Sediment Capping

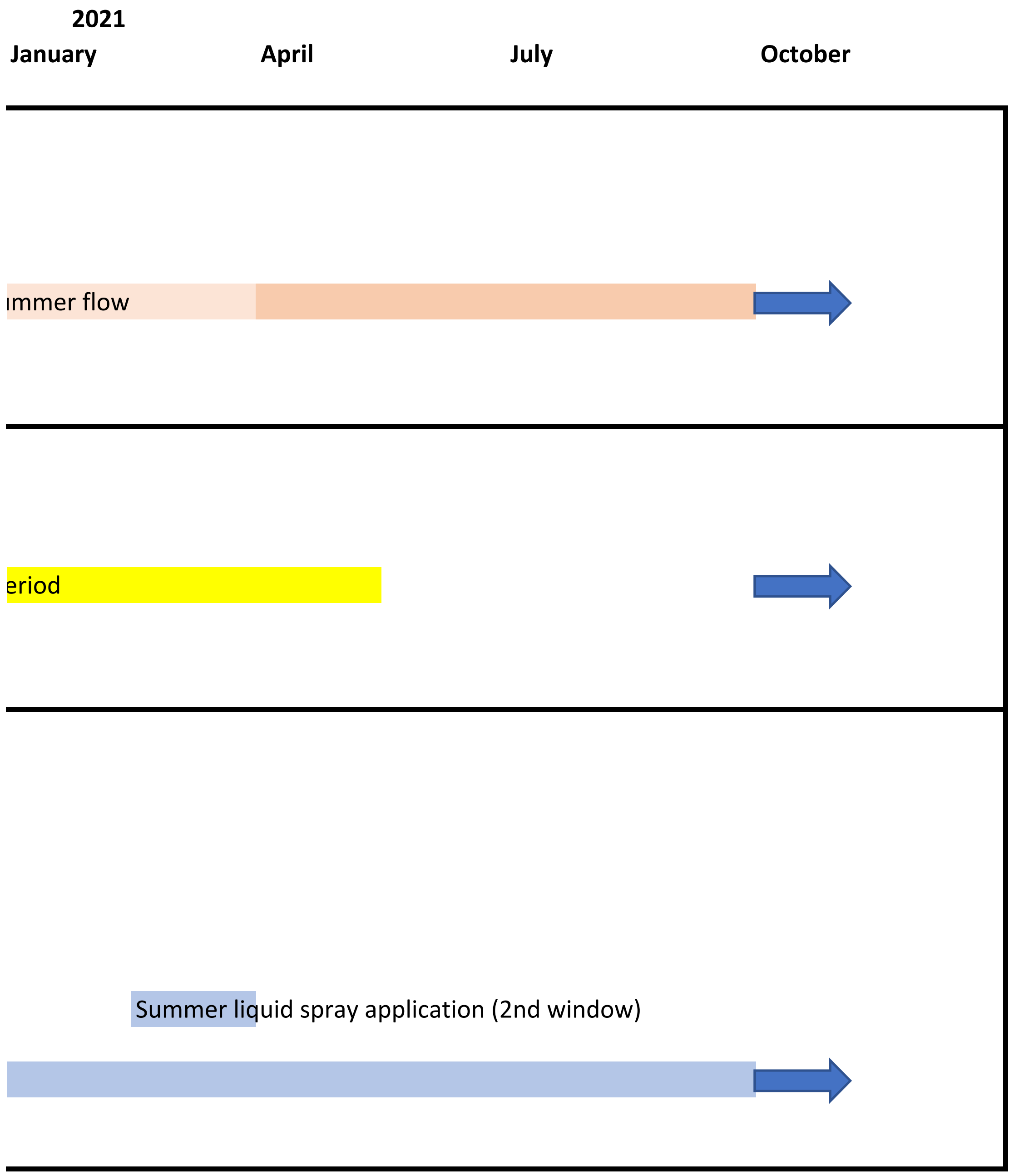
Determine application method

Obtain discharge consents ??

EITHER **Spread granules (one off)** Winter granule application (1st window) | Winter granule application (2nd window)

OR **liquid surface spray (one off)** Summer liquid spray application (1st window)

OR **Mill Ck injection (continuous)** design/build applicator structue | Injuection rate fluctuates seasonally


2021
January
April
July
October
mmer flow
eriod
Summer liquid spray application (2nd window)

Lake Hayes Water Quality Remediation Options

Prepared for Otago Regional Council

March 2018

Prepared by:
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Cover Photo: Reflections of the southern hills from Lake Hayes [Photo montage by Max Gibbs].

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Executive summary

To progress the development of a Long-Term Plan (LTP) for the management and restoration of Lake Hayes, Otago Regional Council (ORC) commissioned the National Institute of Water and Atmospheric Research Ltd (NIWA) to provide expert advice for detailed scoping on the options for remediation of Lake Hayes water quality. NIWA was asked to review a recent report, prepared for the Friend of Lake Hayes Society Inc (Schallenberg and Schallenberg 2017), which provides a restoration and monitoring plan for Lake Hayes and covers a range of options for improving the water quality of Lake Hayes. In addition, NIWA was asked to expand the detail on three of these options in order that they could be costed. Within the scope of the report, NIWA was requested to present information on other options that might be useful in the restoration of Lake Hayes and to include information and comments on a new nanobubble technique. Comments on catchment management were to be minimal as this is being covered in a separate report.

Report review

The Schallenberg and Schallenberg (2017) report is well written and provides a wealth of information on the history of the lake and the probable causes of the lake changing from a presumably pristine alpine lake to a highly degraded, supertrophic lake. A key message in the report is that the lake appears to be at a 'tipping point' for recovery. However, although the report was written in 2017, it contains few data beyond 2015, without presenting more recent evidence to support this contention. Neither is there an indication of what level of improvement might be expected. The report explains much of the variability in the data by conventional limnological processes and also includes a novel '*Ceratium* pump' theory to explain recent increases in total phosphorus (TP) and total nitrogen (TN) in the upper water column during the period of thermal stratification.

The sections on monitoring (at different time scales) and the need for a nutrient budget are comprehensive and the report reinforces the need for long-term datasets for trend analysis and understanding how the lake might be restored. The report presents a restoration strategy for Lake Hayes and includes a feasible restoration plan and timeline. Some of the actions in this restoration plan are explained in more detail in the appendices. Specifically: 1) the use of food web biomanipulation to reduce the magnitude of the summer bloom of *Ceratium hirundinella*; 2) the potential for augmentation of the inflows of Lake Hayes with Arrow River irrigation water; 3) alum dosing for phosphorus immobilisation – together with a rough estimate of the cost of alum dosing, and 4) catchment management to restore and protect Lake Hayes.

While these actions are well presented, there is an error in the section on augmentation of Mill Creek with Arrow River irrigation water that needs to be corrected to give a better estimate of the likely efficacy of this option.

Three defined options for water quality improvement

To aid interpretation of some of the data and provide information on the hydrodynamics of Lake Hayes, additional NIWA and University of Otago unpublished data was included in this report. Additional information on the growth habit of *Ceratium hirundinella* was compiled from the literature and included in this report.

Defined option 1: *Flow augmentation from the Arrow River irrigation scheme*. Although this option was considered to of minor importance in the Schallenberg and Schallenberg (2017) report, when the entrainment factor of surface lake water into the density current from Mill Creek is included, there is the potential for this option to prevent the bottom water (hypolimnion) becoming anoxic during the stratified period, thereby eliminating the internal phosphorus (P) load which is required by *Ceratium* for growth. There are issues with this option in that it is 'fragile' and relies on the ambient temperature to cool the Mill Creek water sufficiently to cause the density current to plunge to the hypolimnion. In a warm year this might not happen and the internal P load could return along with a substantial algal bloom.

Defined option 2: *Destratification*. The use of a bubble plume air curtain across the middle of the lake would generate circulation currents that would prevent the lake from becoming thermally stratified and mix the lake water column, keeping the lake well oxygenated. These conditions would eliminate the internal P load and reduce the proliferation of algal blooms. The mixing regime would not eliminate algal growth in the lake, rather it would change the algal assemblage from harmful cyanobacteria and *Ceratium* species to more benign species. Eventually these would diminish dramatically and the lake water quality would improve rapidly.

Defined option 3: *Sediment capping*. The range of available sediment capping and P elimination agents has been discussed. It was concluded that the 'best' option would be to treat the lake with alum using a low dose drip feed into the Mill Creek inflow. This approach would deliver the alum to the hypolimnion in the density current without impact on the lake surface waters. Using real-time monitoring data, adaptive management strategies could control the application to when it was required and was being delivered into the hypolimnion for the most efficient and cost-effective management of the internal P load.

Other options 1: *Nanobubble technology*. This was determined to be un-proven technology and the information available was entirely based on company brochures with no peer-reviewed scientific papers to back up the claims made. While this situation frequently arises when new techniques are developed, it is common to put out a methods paper for the scientific community to review. This has yet to be done. The technique appears to work but has the limitation that it requires the nanobubbles to provide all of the oxygen to aerate the hypolimnion. In Lake Hayes, this represents about 1.6 t O_2 d^{-1}. Putting this into perspective, the underflow from Mill Creek with augmentation can provide up to 1.8 t O_2 d^{-1}. The other main drawback for nanobubble technology is the cost at about $4.7 million for the 7 units required for Lake Hayes.

Other options 2: *Hypolimnetic withdrawal*. A switchable outflow control structure is described that allows the current outflow regime to operate from autumn to spring and then change to draw hypolimnetic water in summer. During summer the hypolimnetic siphon would reduce the DRP in the lake and discharge it into the Kawarau River. This approach has been used successfully overseas and on one lake in New Zealand, and requires further investigation by an engineer to assess feasibility for Lake Hayes, if it is considered as a practical mitigation measure.

Other options 3: *Biomanipulation*. Biomanipulation is a technique which relies on a set of conditions prevailing to enable a specified end result. This approach is covered in detail in the Schallenberg and Schallenberg (2017) report. However, while biomanipulation has been successfully used in many shallow lakes in Europe and the USA as well as in New Zealand, the success stories have all been in shallow lakes and ponds and there is no compelling literature base that suggests the technique would work in a lake as deep as Lake Hayes.

Other options 4: *Catchment management*. This strategy is assessed as being fundamental to the long-term restoration of Lake Hayes. A key element of catchment management would be the reduction of fine sediment loads to Lake Hayes, as this is the primary vector for transporting P into the lake. Catchment management will take time to become apparent but, in the long-term, it will improve water quality in Lake Hayes.

It is the considered opinion of the author of this report that the most effective short-term mitigation method in Lake Hayes would be destratification.

1 Introduction

Otago Regional Council (ORC) are developing a Long-Term Plan (LTP) for the management and restoration of Lake Hayes. To help facilitate this, ORC commissioned the National Institute of Water and Atmospheric Research Ltd (NIWA) to provide expert advice for detailed scoping on the options for remediation of Lake Hayes water quality. A report prepared for the Friend of Lake Hayes Society Inc (Schallenberg and Schallenberg 2017) provides a restoration and monitoring plan for Lake Hayes that covers a range of options for improving the water quality of Lake Hayes that require further consideration.

The specific tasks requested and covered in this report are:

1. Review and comment on the Schallenberg and Schallenberg (2017) report advising on mitigation options for improving Lake Hayes water quality.

2. Look at, but not be limited to, three defined options for water quality improvement, being:
   a. Flow augmentation from the Arrow River irrigation scheme - The best strategic timing of water releases, the potential efficacy based on flow rates and volumes available. Seasonality of releases.
   b. Destratification – Provide advice on what the challenges are; if it is possible given the shape and stratification dynamics of the lake; what the best options would be; Provide sufficient advice to allow for design and costings to be carried out. Required duration of de-stratification?
   c. Sediment capping – what the best medium is to use, amounts required and how best to apply it. How long will sediment capping be effective for?

3. Comment on the likely effect of each option on water quality in Lake Hayes, including algal growth.

4. Assess the appropriateness and infrastructure/cost (if available) requirements to employ nanobubble technology to aerate the Lake Hayes hypolimnion.

5. Provide comment on the importance of long-term catchment management given current nutrient and sediment loads entering Lake Hayes from Mill Creek.

6. Provide sufficient detail to quantify what is required to install and manage each of the options so that detailed cost estimates can be made to support our Long-Term Plan (LTP) development. How much, where, when etc., and if required, discuss and advise service providers on technical details of the options. Describe the secondary or amenity effects of each option (noise, water clarity, odour, visual impact, etc.).

To facilitate this work, ORC will provide:

- All available relevant data on water quality and stratification dynamics on Lake Hayes.

- Technical details and constraints in relation to the flow augmentation option, including maximum flow rates, seasonality of water availability and augmentation water quality.

- Comments on the completed draft within 5 working days of receipt of the draft report.

The output from this work will be a comprehensive report, peer reviewed to NIWA standards, providing expert advice for detailed scoping on the options for remediation of Lake Hayes water quality.

1.1 Background information

This section presents the current state of knowledge about Lake Hayes in terms of its biogeochemistry in order to understand the factors that are driving the lake water quality and whether/how the water quality is changing over the period of the available monitoring data. This section also includes aspects of the physics, hydrodynamics and biology in order to adequately review the Schallenberg and Schallenberg (2017) report. It may also help understand how this lake works and to identify any critical points in the various biogeochemical cycles where an intervention at a specific time may have a major effect on the water quality.

The water quality of Lake Hayes has been classified as supertrophic with a Trophic Level Index (TLI) >5. This classification is addressed in the Schallenberg and Schallenberg (2017) report. Conversely, the National Policy Statement on Freshwater (Freshwater NPS), National Objective Framework (NOF) attributes for phytoplankton abundance (measured as chlorophyll *a*), total nitrogen (TN) and total phosphorus (TP) (MfE 2014), which are the legislated management requirements, is not addressed and are outside the scope of this report and are not discussed.

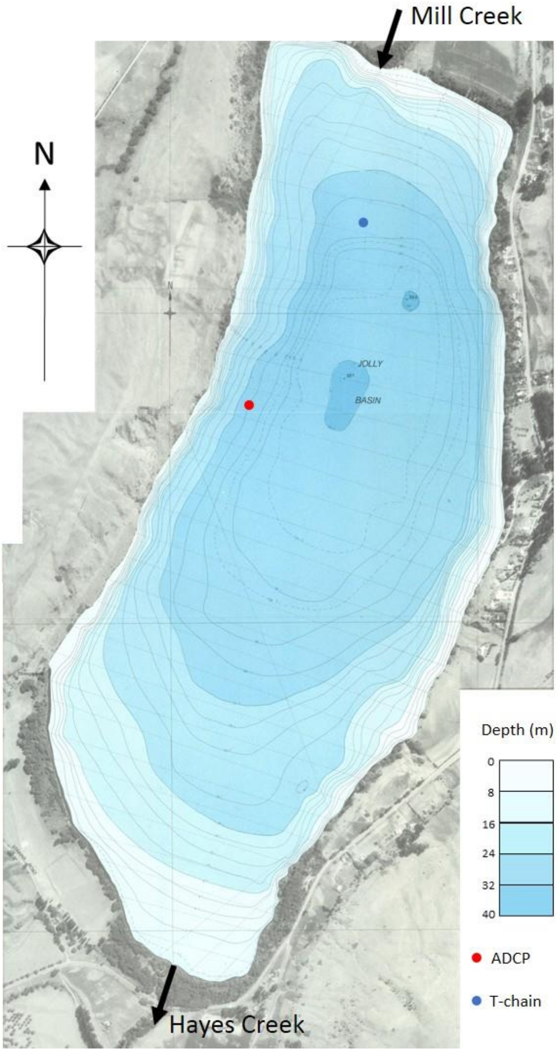
Lake Hayes is a small glacial lake formed about 10,000 y BP (Lowe and Green 1987). Prior to 1740, the landscape was likely to have been Kahikatea forest with large wetlands in the western Mill Creek area. Deforestation of the catchment began around 1740 when the Kahikatea forest was largely destroyed by fire and probably continued through the 1800's as miners and settlers harvested trees for shelter and firewood (Robertson 1988). The catchment became native tussock grassland. At present, it is largely agricultural land, which was developed by drainage of wetlands and the application of superphosphate to grow grass for the dairy industry (Schallenberg and Schallenberg 2017).

Accelerated eutrophication of the lake began when superphosphate was introduced in 1950, spread by aerial top-dressing across the Mill Creek catchment. A top-dressing plane crashed into Lake Hayes in August 1952. A cheese factory north of the lake discharged effluent whey, with a phosphorus (P) load of about 1000 kg y^{-1}, into Mill Creek from 1912 to 1955 (Robertson 1988). Septic tank effluent from lake shore residences enter the lake through the groundwater inflow (Selvarajah 2015).

Investigation of possible nutrient limitation in Lake Hayes in 2006 (Bayer et al. 2008), when *Ceratium* was the dominant phytoplankton species, indicated that algal growth was stimulated by additions of N and the trace element boron and zinc. P additions had no effect. Bayer et al. also comment that P was often in surplus in the lake in relation to nutrient demands of phytoplankton i.e., low N:P ratios[1]. Schallenberg and Schallenberg (2017) comment that reducing P levels in the lake to the point where they can restrict phytoplankton blooms is important. They also comment that "the re-establishment of P-limitation of phytoplankton growth would have the added benefit of removing the competitive advantage of N-fixation, which historically dominant bloom-forming phytoplankters such as *Anabaena sp*. are capable of."

[1] N:P ratios <10 are considered to indicate N-limitation and alga growth is likely to be stimulated by the addition of N. N:P ratios >17 are considered to indicate P-limitation and algal growth is likely to be stimulated by the addition of P. Between these values, addition of either N or P, or both N and P, is likely to stimulate algal growth.

Fine sediment, washed into Mill Creek from the catchment (Figure 1-1), is accumulating in the bottom of Lake Hayes. This sediment typically has high P concentrations bound to the iron and manganese oxides in the soil particles. When the lake water is well oxygenated, the P remains attached to the metal oxides and is in a particulate form, which is not available for plant growth i.e., it cannot be taken up by phytoplankton (free floating algae). Conversely, when the lake has low dissolved oxygen (DO) concentrations, the iron and manganese dissolve and the P is released into the water column as phosphate, commonly called dissolved reactive P (DRP). This transformation is reversible with DRP being bound to the iron and manganese oxides when the lake becomes oxygenated again (Schallenberg and Schallenberg 2017). DRP is readily available to sustain phytoplankton growth. When DRP is released from the sediment stored in the bottom of the lake it is referred to as internal cycling.



Lake Hayes morphological characteristics:

Area	2.76 km^2
Length	3.1 km
Mean depth	18.0 m
Maximum depth	33.0 m
Volume	55.1 x 10^6 m^3
Volume of hypolimnion	28.9 x 10^6 m^3
Catchment land area	44.0 km^2
Surface elevation	315 m
Theoretical mean residence time[2]	3.82 y
Residence time (added inflow)[3]	2.98 y.
Inflow via Mill Creek (mean 1984-97)	0.43 m^3 s^{-1}

Figure 1-1: Lake Hayes bathymetry. Red dot indicates position of an acoustic doppler current meter (ADCP) deployment. Blue dot is position of thermistor chain deployment. (Chart redrawn from Hurley 1981).

[2] Lake volume divided by total annual inflow (data from Caruso 2000a, b)
[3] See section 3.2 calculations

Apart from monitoring data from Otago Regional Council and University of Otago, there have been several short-term investigations by other institutes, including NIWA, that provide additional information on in-lake processes in Lake Hayes. The data may aide interpretation of the monitoring data.

For example, when the lake is thermally stratified in summer, wind blowing across the lake establishes a circulation pattern in the lake with surface water moving with the wind flow but the upper and lower water columns moving in the opposite direction (Figure 1-2). Water movement around the thermocline was in the same direction as the wind with evidence of turbulence (Figure 1-2, C). These data were recorded at the ADCP site (Figure 1-1) on a bottom-mounted ADCP in 24 m water depth recording in 1-minute bursts every 5 minutes with 1-m depth intervals.

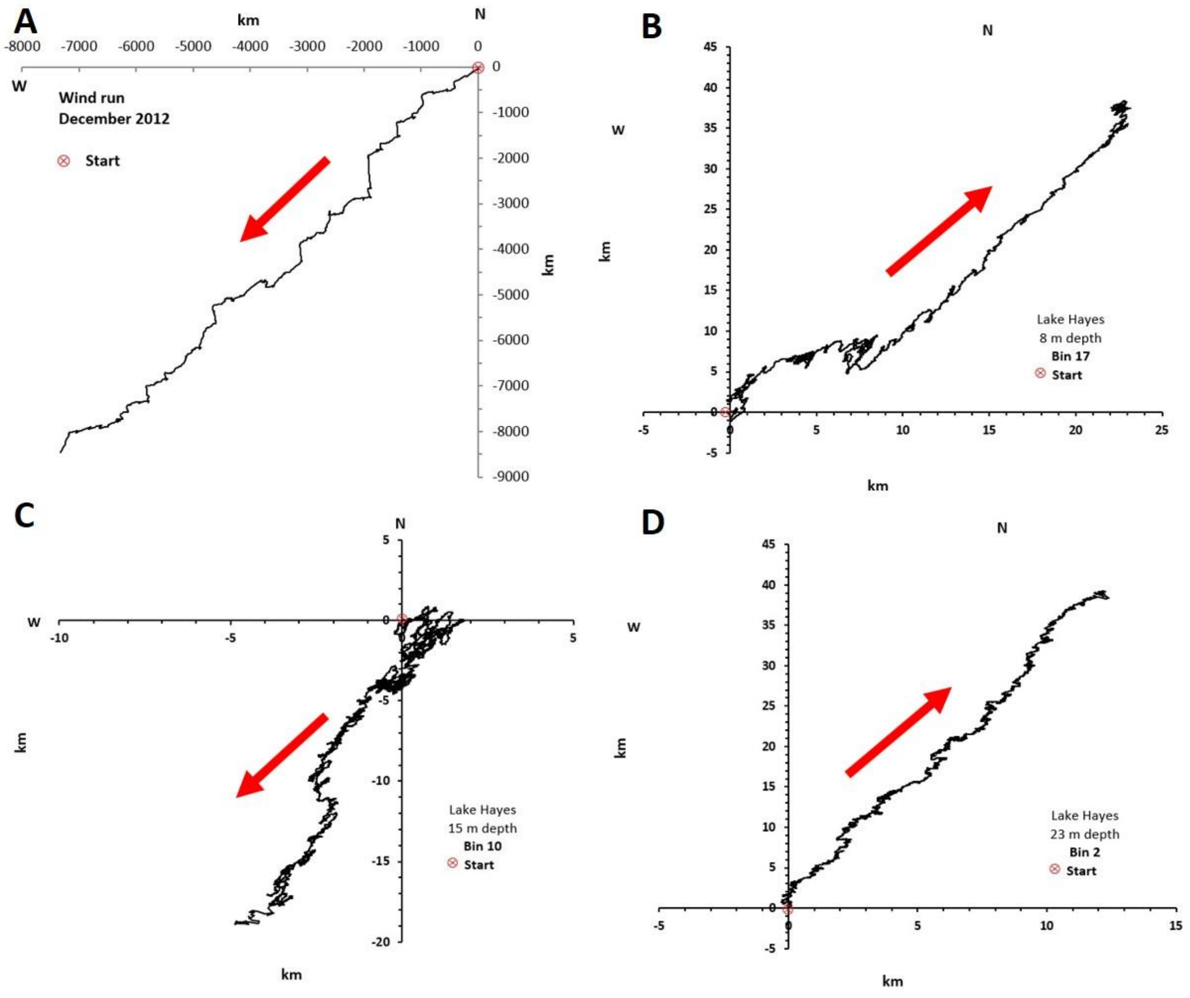


Figure 1-2: Wind run and acoustic doppler current meter (ADCP) progressive vector plots at selected depths. Flow directions indicated by red arrows: A) Persistent wind run from the north; B) Epilimnion current flow at 8 m depth was to the north; C) Thermocline current flow at 15 m depth was to the south; D) Hypolimnion current flow at 23 m depth was to the north. (NIWA unpublished data).

The full suite of ADCP profile data show the currents reverse at different depth ranges (Figure 1-3). This limited flow data indicates that persistent wind flow across the surface of the lake in one direction can induce a subsurface return flow which has resulted in a pair of vertical circulation cells, one in the epilimnion and the other in the hypolimnion. The southwards return flow of the epilimnion cell coincides with the thermocline, which is a zone of turbulence and can result in upwards mixing of nutrient enriched bottom water into the epilimnion and downwards mixing of DO from the epilimnion into the hypolimnion.

A similar current feature has been recorded in Lake Rotorua (Gibbs et al. 2016). There is insufficient ADCP data from Lake Hayes to define the horizontal currents as found in the Gibbs et al. (2016) study. These are highly likely and would account for spatial distribution of a *Ceratium hirundinella* bloom reported in Schallenberg and Schallenberg (2017).

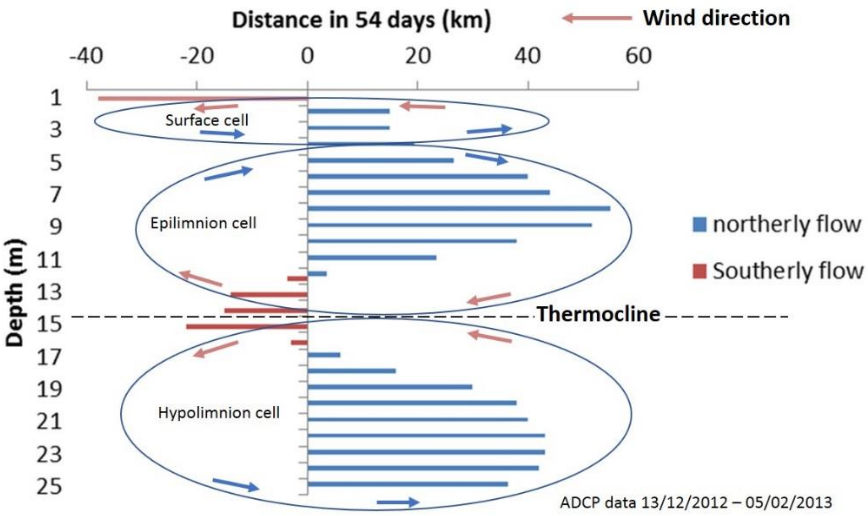


Figure 1-3: Current flow direction in Lake Hayes in response to wind forcing from the north. Stylised circulation cells are consistent with data shown. There is insufficient data available to show lateral circulation patterns. Current data recorded at the ADCP location (Figure 1-1) between 13/12/2012 and 05/02/2013. (NIWA unpublished data).

Temperature and dissolved oxygen data, from a thermistor chain moored in 25 m water depth (Figure 1-1)and recording at 15-minute intervals, provide insights into possible mixing processes in Lake Hayes (Figure 1-4).

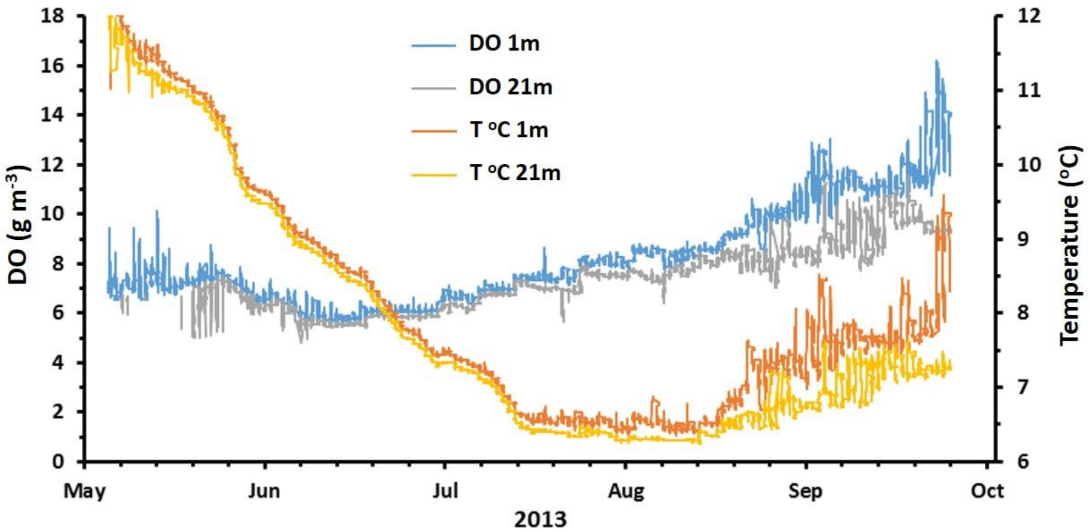


Figure 1-4: Surface and bottom water temperature and dissolved oxygen data from Lake Hayes. Data from a thermistor chain deployed in 25m water depth (Figure 1-1) between May and September 2013 showing the beginning of thermal stratification. (NIWA unpublished data).

These data, recorded during the winter mixed period, show the expected diurnal temperature cycle of heating and cooling in the surface waters as well as diurnal changes in the DO concentrations apparently associated with phytoplankton photosynthesis during the daylight hours in spring (Figure 1-4). These diurnal cycles can be detected down to 8 m depth, synchronous with the surface temperature and DO. The unexpected diurnal cycles in the 21 m deep data (Figure 1-4) are lagging about 3 hours behind the daylight cycles (Figure 1-5) and can only have been caused by temperature and DO changes in the Mill Creek inflow water, which would have been plunging as a density current at that time and intruding through the lake at a depth of equal density or along the lake bed. The lag will be due to the travel time from the Mill Creek mouth to the thermistor chain, a distance of about 500 m and represent an average flow velocity of about 5 cm s^{-1} for the intrusion current.

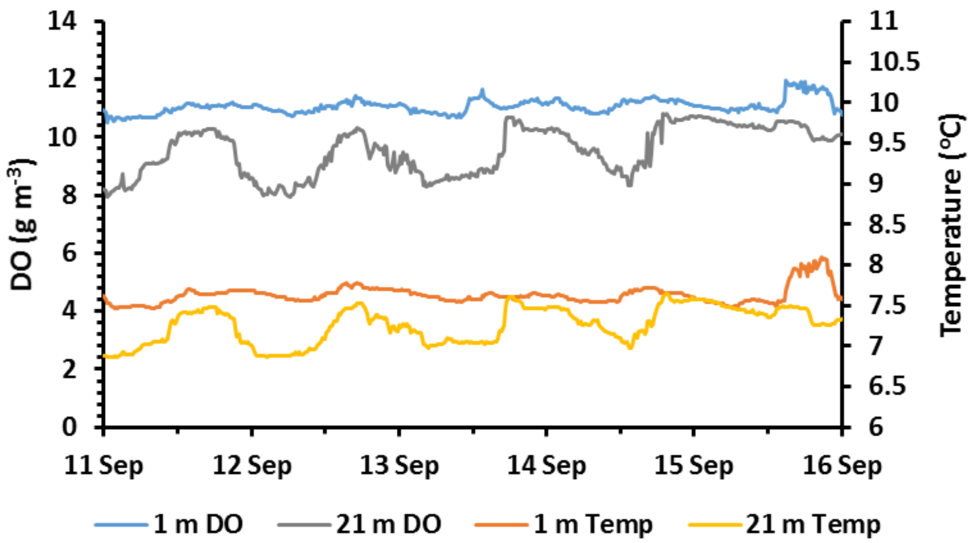


Figure 1-5: Surface and 21 m depth temperature and dissolved oxygen from a thermistor chain in Lake Hayes in winter 2013. While the surface lake water data show little variation over the diurnal cycle, the temperature and DO in the 21-m data show substantial variation over the diurnal cycle, which appears lag the midday high by about 3 hours. This is evidence that the Mill Creek inflow forms a density current. (NIWA unpublished data).

The mean annual flow in Mill Creek is estimated to be 0.43 m^3 s^{-1} (Caruso 2000a, b). If this inflow is >0.5°C cooler than the lake surface water temperature, the inflowing water will plunge as a density current and sink to a depth of equal density before penetrating into the lake as an intrusion layer (Gibbs and Hickey 2012). In the plunge process the inflow water entrains surface lake water into the density current and this can increase the volume of the intrusion layer by about 5-fold, depending on the temperature difference between the inflow and the lake. It also provides a mechanism by which nutrients and phytoplankton cells in the lake surface water can be rapidly dispersed throughout the lake (Vincent et al. 1991).

This dispersal process is potentially very important during storm events when Mill Creek can transport relatively large amounts of fine sediment from the catchment into the lake. Fine sediment typically has high concentrations of P bound to the iron and manganese particles in the sediment and is the main vector for P transport into lakes. This is recognised in the Schallenberg and Schallenberg (2017) report as a major source of the legacy[4] P accumulating in the lake sediments.

An example of a flood event hydrograph relative to the total P (TP) transported in Mill Creek (Caruso 2000b) suggests that the majority of the TP may enter the lake in just a few hours during a storm event (Figure 1-6). This is consistent with a study on the Ngongotaha Stream at Rotorua, which found that 42% of the annual sediment load was carried on just three days and that 25% of the annual load was carried in a period of 16 hours at the peak of the flood event (Hoare 1982).

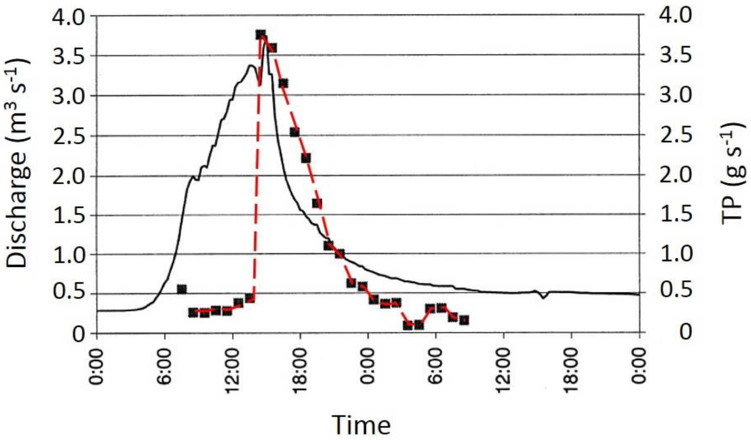


Figure 1-6: Correlation between a flood hydrograph and Total P (TP) load in Mill Creek. Red dashed line indicates the TP load. (Redrawn from Caruso 2000b).

These additional data and information augment the data and information presented in the Schallenberg and Schallenberg (2017) report.

[4] Legacy P: P transported into a lake in suspended sediment settles to the bottom and accumulates rather than being flushed out. Accumulation occurs over many years and remains as a legacy from previous historical activities in the catchment.

2 Review of the Schallenberg and Schallenberg (2017) report

2.1 Overview

The Schallenberg and Schallenberg (2017) report provides a comprehensive general review and analysis of the information on Lake Hayes, and includes recommendations on restoration options for this lake. It brings together the history of Lake Hayes and the changes and events that have transformed a pristine, and presumably oligotrophic, alpine lake into a highly degraded, supertrophic lake, which has, in recent times, experience substantial algal blooms. The report summarises the physical, chemical and biological aspects of the Lake Hayes ecosystem and gives a good background into some of the factors that have resulted in the accelerated eutrophication of this lake. It also offers some interesting ideas, such as the "*Ceratium* pump theory", to explain some of the temporal changes in in-lake nutrient concentrations. The recommendations are aligned with the evidence available on how the lake is operating and contemporary understanding of lake restoration science.

However, although the report was written in 2017, there is little contemporary data beyond 2015 and the appendices are essentially reproductions of earlier work without major updates. There is a serious flaw in the discussion of the use of irrigation water from the Arrow River to augment the flow in Mill Creek for enhanced flushing as a restoration strategy. Parts of that section of the report should be rewritten to provide correct information.

Most of the terminology is explained but not coherently in one place. For example, an important concept is the trophic level index (TLI). This is introduced on Page 5 without an explanation of what it is and how it is determined, and no reference as to where that information can be obtained. The trophic level classification table from Burns et al. (2000), which contains the TLI scale and parameter ranges appears in an appendix on Page 45, should be cross-referenced from the text.

2.2 Detailed comments

Since the TLI was developed in the late 1990's (Burns et al. 2000), the TLI assessment of water quality has been widely used in New Zealand to compare the water quality of all lakes. Although intended as an index for comparison of water quality between different lakes, the TLI has also been used as a tool for management of lake water quality. How this is achieved using the linkage between the four TLI components – water clarity, chlorophyll *a*, total nitrogen (TN) and total phosphorus (TP) – is not mentioned but should be to provide improved clarity.

The report identifies that Lake Hayes has poor water quality and is classified as supertrophic based on the TLI of >5, but suggests that the lake may be approaching a tipping point of recovery. This conclusion is based on changes in water quality and biological indicators. For example, the report authors suggest that there has been an apparent decrease in TP concentrations in the hypolimnion since about 2000 indicating a reduction in internal P cycling in the lake, and the report claims there have been several recent years with very low phytoplankton biomass and high water clarity in summer: 2009/10, 2012/13 and 2016/17. A clear water phase also occurred in 2017/18. Notwithstanding this, the ORC time-series chlorophyll profile data do not support that 2012/13 was a clear water year. The ORC data show that, in 2012/13, mean maximum chlorophyll concentrations averaged 72.2 mg m^{-3} and the peak concentration was 137 mg m^{-3}. In 2009/10, the mean maximum chlorophyll concentrations averaged 6.1 mg m^{-3} and the peak concentration only reached 13.5 mg m^{-3}.

There are other indicators of water quality improvement and lake recovery which have not been discussed. These are found in the more recent time-series data that were previously used to show the progressive water quality degradation over time.

Evidence of lake water quality degradation is shown in the report are a progressive increase in TLI from about 3.6 (mesotrophic) in 2004 to about 5.2 (supertrophic) in 2015 and a reducing mean water clarity, measured as Secchi disk depth, from about 6 m in 1950-53 to <2.5 m in 2015. The loss of bottom water (hypolimnion) dissolved oxygen is mentioned but not analysed. A consequence of hypolimnetic anoxia is the release and recycling of P, which has been brought into the lake with the elevated sediment loads during storm events (Figure 1-6). This is discussed in the report.

Assessment of oxygen loss from the hypolimnion can be standardised and expressed as a hypolimnetic oxygen depletion (HOD) rate for interannually comparison. This has not been done in the Schallenberg and Schallenberg (2017) report. As part of this review a simple evaluation of the changes in HOD was carried out using available data from the summer stratified period (Nov-April inclusive). This information indicates a long-term degradation of lake water quality, which was already degraded in the 1950s due to land drainage, farming and the discharge of effluent whey from a cheese factory into the lake (Schallenberg and Schallenberg 2017). Literature data show a progressive loss of bottom water oxygen which was at a minimum of 13% saturation in 1953-54 (Jolly 1968) to complete anoxia (0% saturation) for a period of about 4 months each year in 1969-71 (Burns and Mitchell 1974). This period of anoxia gradually increased to about 5 months in 2008 (ORC data). The rate of change in the HOD rate, calculated from September to February each summer, has been relatively slow with a mean HOD rate of 97.5 mg m^{-3} d^{-1} from 1992 to 2008 (minimum 81 mg m^{-3} d^{-1}, maximum of 115 mg m^{-3} d^{-1}) (Figure 2-1).

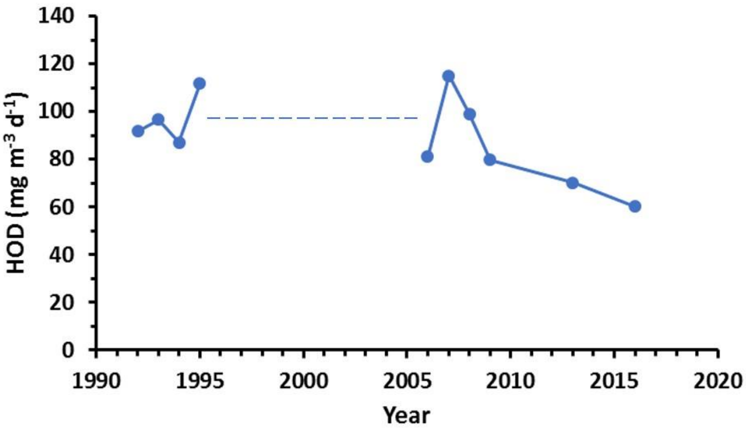


Figure 2-1: Time-series hypolimnetic oxygen depletion (HOD) rates for Lake Hayes. Mean HOD rate from 1992 to 2008 is 97.5 mg m^{-3} d^{-1}. From 2008 to 2016 the HOD rate appears to have been decreasing indicating a reduction in sediment oxygen demand, implying an improvement in water quality (Otago Regional Council data).

Recent data from two of these measures of degradation also support the suggestion in the report that Lake Hayes might be at or approaching a recovery tipping point. Firstly, the HOD rate has decreased from 99 mg m^{-3} d^{-1} in 2008 to 60 mg m^{-3} d^{-1} in 2016 (Figure 2-1). This change can be seen in the time-series DO profile data (Figure 2-2) with the slower reduction in DO concentration in spring 2016 and anoxia reached in March 2017 instead of January as in the 2007-8 plot.

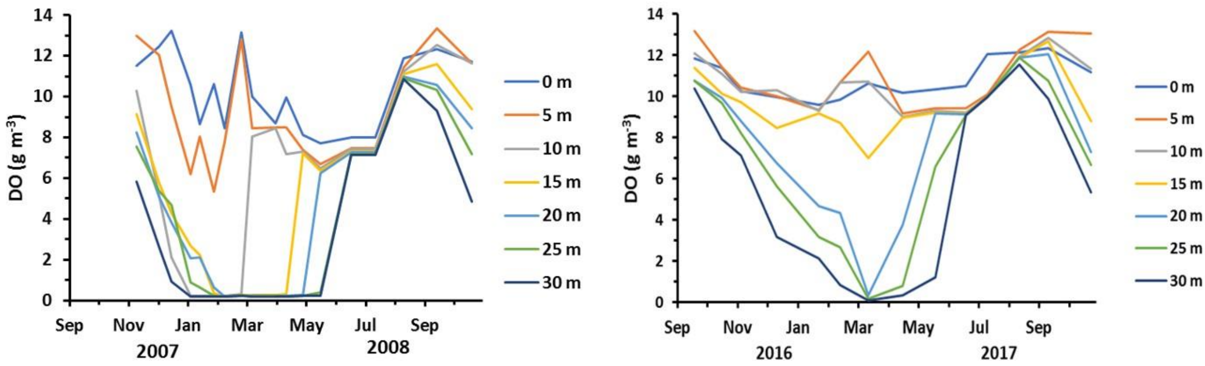


Figure 2-2: Time-series dissolved oxygen profile data from 2007-8 and 2016-17. In 2007-8 the period of bottom water anoxia was 5.5 months but in 2016-17 it was about one month.

Secondly, in 2007-8, the period of bottom water anoxia was 5.5 months but in 2016-17 it was only about one month. The major differences between these two time-series plots are that 1) in 2007-8 the anoxic zone extended up to 8 m below the surface while in 2016-17 the anoxic zone was below 20 m, and 2) re-oxygenation of the bottom water began in March 2017 while it began in May in 2008. These recent DO concentrations indicate that the period when P is recycled from the sediments is greatly reduced, consistent with the author's suggestion of a reduction in internal P cycling in the lake. The DO time-series plots suggest this change has been more recent i.e., since 2008 and not 2000.

The correlation between the three clear water years with low *Ceratium hirundinella* biomass and the super abundance of the water flea, *Daphnia pulex*, is important as it indicates that there is a food web mechanism that might be used to reduce or manage the recent *Ceratium* blooms. Two possible scenarios are that either the large *Daphnia* are grazing on the *Ceratium* cells directly or they are grazing on the bacteria used by *Ceratium* as a food source to augment their nutrient requirements.

Coupled with the other indicators of water quality improvement (Schallenberg and Schallenberg 2017 report section 2.3), the report correctly interprets the marked fluctuations in water clarity as indications that the lake is close to a tipping point for recovery from eutrophication but to what degree is not specified. An expectation of the degree of recovery should be given. The final key point raised in the report (section 2.3) summaries the current state of knowledge as well as a way forward:

> "*The current situation suggests that appropriate restoration measures could result in stable improvements in summer water clarity, reductions in* Ceratium *summer biomass, and the re-oxygenation of the bottom waters of the lake. These factors appear to be facilitated by maintaining a low nutrient availability and a high summer* Daphnia *density.*"

2.3 Restoration strategies

Restoration strategies suggested in section 4 of the Schallenberg and Schallenberg (2017) report fall into five main categories: 1. catchment rehabilitation to reduce external nutrient loads to the lake, 2. reduction of internal nutrient loads/recycling, 3. food web manipulation, 4. flushing of water through the lake, and 5. other in-lake actions (listed in the report, Table 1). The discussion of these options in this section of the report provides a comprehensive basis for the development of a restoration plan for Lake Hayes.

One factual issue with this section of the report is the statement that *"Lake Hayes has a water residence time of around 1.8 years (Caruso 2000)"*, is incorrect. The value of 1.8 years in the Caruso (2000b) paper is attributed to a paper by Jolly (1968) but that residence time value is not apparent in the Jolly (1968) paper. As part of this review the residence time[5] for Lake Hayes was calculated to be 3.8 years, based on the volume of the lake and the average total annual inflow volume from 1984 to 1997 via Mill Creek, as published in the Caruso (2000b) paper. This is more in keeping with expectations for a mean annual inflow of 0.43 $m^3 s^{-1}$.

A second issue with this section of the report is that, while the formation of a temperature driven density current from Mill Creek inflow is recognised, the calculations used do not take into account the entrainment[6] of surface lake water into that inflow when considering oxygen and nutrient transport.

The entrainment effect can be understood by looking at the situation in other lakes. For example, the Tongariro River inflow to Lake Taupo, entrains surface water into the plunging inflow and has increased the volume of the inflow by a factor of up to 4. The Ohau Channel inflow to Lake Rotoiti was measured at 3.5 (Vincent et al. 1991) before the diversion wall was installed. A more detailed evaluation of the oxygen transport into Lake Rotoiti via underflowing density current used an entrainment factor of 5 (Gibbs 1986). The amount of water entrained into the inflow is a function of the Froude number, the temperature difference between the inflow water and the surface water where the inflow enters the lake, the velocity of the inflow and the depth and width of the inflow channel (Spigel et al. 2005). As the temperature difference and/or the velocity increases, the entrainment factor decreases to around 2, but as the temperature difference decreases the entrainment factor increases to the point where a density current is not formed and the inflow water disperses across the lake surface as a buoyant plume (Gibbs and Hickey 2012).

In the heat of summer, the effect of entraining surface water is to raise the temperature of the resultant density current so that it is less likely to plunge below the thermocline and is more likely to insert as an intrusion layer in the epilimnion or around the depth of the thermocline in summer.

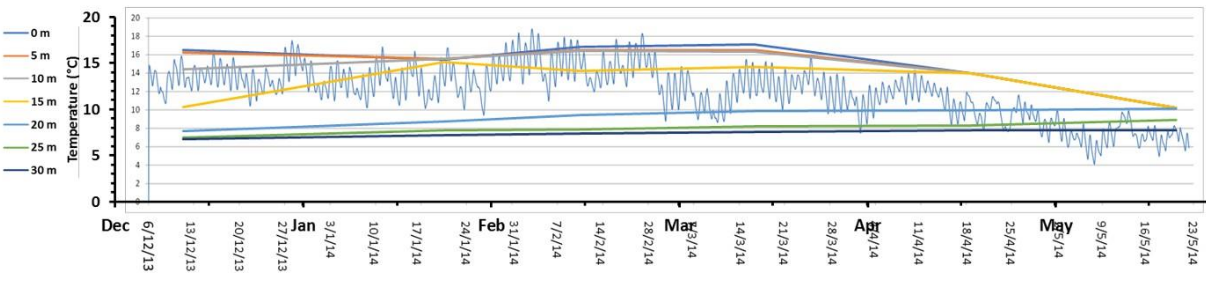


Figure 2-3: Example of time-series temperature data from Mill Creek (blue wavy line) overlaid with 2016/17 Lake Hayes temperature profile data (colour depth scale on left axis). Mill Creek data (from 2012 to 2013) show a strong diurnal cycle that was mostly warmer than the thermocline at 15 m depth in spring and summer before cooling to metalimnetic temperatures in autumn (March). Entrainment of warmer surface lake water into the Mill Creek inflow would result in the inflow water inserting at a depth higher in the water column than predicted by the absolute temperatures shown. Temperature data from ORC.

[5] Residence time is calculated as lake volume divided by the total inflow per year.
[6] An inflow that plunges as it enters a lake draws surface lake water into the inflow and forms a temperature-induced density current with a temperature somewhere between the inflow temperature and the temperature of the lake surface water. The density current will plunge to a depth of neutral density and then flow as an intrusion layer into the lake at that depth. The amount of water entrained is the entrainment factor. An entrainment factor of 4 means the density current has a volume 4 times larger than the inflow volume.

In Lake Hayes, this could result in the transport of fine sediment and nutrients from the catchment via Mill Creek and particulate matter, including phytoplankton, from the littoral zone of the lake, into the epilimnion or metalimnion (thermocline) in summer. As climate warming progresses, Mill Creek may become warmer in summer with the diurnal temperature cycle favouring more inflow into the epilimnion than the hypolimnion (Figure 2-3). The result of this process is likely to be seen as an increase in TP and TN in the epilimnion and a decrease in these nutrient components in the hypolimnion in summer. This effect could explain the recent increase in TP and TN concentrations in the epilimnion, as shown in Figure 6 of the Schallenberg and Schallenberg (2017) report, and which are the basis for the *Ceratium* nutrient pump theory.

In contrast with hot summers, as the lake cools in autumn and develops cold edge water at night because of frosts, the low night time temperatures in the Mill Creek water will be cooled further by entraining the cold edge water, causing the density current to plunge as an underflow to a greater depth than predicted by the absolute temperatures shown (Figure 2-3). The cold underflow is likely to flow along the lake bed and, as the water in that flow would be 100% saturated with oxygen, it would begin to re-oxygenate the hypolimnion. This is consistent with re-oxygenation of the hypolimnion from March onward, as observed in the 2016-17 DO time-series data (Figure 2-2). This hydrodynamic regime may account for the gradual reduction in the HOD rate since 2008 (Figure 2-1).

With this hydrodynamic regime distributing inflow water and associated nutrients to continuously differing depths in the lake over the diurnal temperature cycle, the restoration strategy of reducing the external loads to the lake will be fundamental to the long-term restoration of the lake.

Because catchment rehabilitation actions are typically slow to show improvement in lake water quality, the implementation of additional restoration strategies using food web manipulations, augmented hydraulic flushing and other in-lake actions, such as P sequestration, are likely to accelerate the restoration of the lake. Discussion of these additional restoration strategies is well considered and forms a coherent overall strategy for the restoration of Lake Hayes. The report includes tables showing the proposed restoration plan for the lake, a suggested time table and the goals/targets that should achieve a successful restoration of Lake Hayes.

2.4 Lake water quality and health monitoring

Although Lake Hayes is one of the most thoroughly monitored and studied lakes in New Zealand, there are gaps in the cumulative data that can lead to uncertainty in identifying trends. This section of the report (section 5) is excellent and reinforces the need for a monitoring program that has been designed for the lake, and includes monitoring factors beyond simple water quality variables. To this end the report includes a well thought out monitoring program for the lake and has co-ordinated this into a table that prioritises the type of monitoring, its frequency and the technology required.

2.5 Appendices

Additional details to the brief discussion of the four main restoration proposals in the body of the report are contained in a set of four appendices – 1) Food web biomanipulation, 2) Augmentation of inflows, 3) In-lake actions (alum dosing) and 4) Catchment management.

Appendix 1: **Food web biomanipulation as a restoration tool for Lake Hayes** is well set out and the practical considerations for this option are discussed. Because it is not practical to breed enough *Daphnia* to stock the lake or to remove the juvenile perch, which are zooplankters (zooplankton grazers), by netting, the key to the success of this option may be the ability to stock the lake with piscivorous trout and eels or large piscivorous sterile perch to consume the juvenile perch. There are other possibilities discussed.

Overall, the biomanipulation option as a concept appears feasible. Inclusion of examples where this approach has succeeded would be useful.

Appendix 2: A preliminary assessment of the potential for augmentation of the inflows of Lake Hayes with Arrow River irrigation water to speed the recovery of the lake involves some complicated calculations around the addition of Arrow River water to Mill Creek in order to increase the flushing rate and reduce the residence time. The Arrow River water has lower DRP concentrations than Mill Creek and would not add substantial amounts of P to the lake. This section sets out the relationship between dissolved oxygen and the release of P from the legacy in the sediments and revisits an earlier report on this option (Schallenberg 2015). It sets out to answer four questions that could affect lake water quality:

1. *Could the augmented inflow flush displace substantial amounts of water from the lake?* Although the incorrect residence time of 1.8 y instead of 3.8 y is given, this does not affect the calculation of the additional proportion of water displaced from the lake. At just 7% increase it would take a long time (probably 10s of years) for the lake water quality to improve to eutrophic or mesotrophic classification.

2. *Would the augmentation flow displace bottom water?* The calculations for this question do not include the effects of entrainment of surface lake water into the density current generated by the Mill Creek water inflow. Consequently, the conclusions derived from the analyses done are incomplete and do not consider the changes in the depth of intrusion for the density current over the diurnal temperature cycle and over the summer – autumn temperature cycle.

3. *Could the augmented inflow supply enough dissolved oxygen to the bottom water to prevent its deoxygenation and, thereby, prevent P release from the sediments?* Again, the calculations for this question do not include the effects of entrainment of surface lake water into the density current generated by the Mill Creek water inflow. As the entrainment factor can substantially increase the volume of the density current, the mass of oxygen transported will be increased by a similar ratio over the amount estimated for the Mill Creek inflow by itself. While the question asks about the <u>augmented flow</u>, the answer given focuses only on the <u>augmentation water</u> rather than the combined flow. While the conclusion that "injecting the Arrow River augmentation flow directly into the bottom waters would not overcome deoxygenation in this lake" is correct, the option of mixing the Arrow River water into Mill Creek before it enters the lake, and thereby increasing the amount of surface water entrained in the underflowing density current, may result in a very different conclusion.

4. *How much P and chlorophyll a could the augmented flow flush from the lake and what effect would this have on trophic state*? These calculations are essentially correct except that the question has been answered on the basis of the augmentation water rather than the augmented flow i.e., Mill Creek plus Arrow River irrigation water. The augmentation water would from the Arrow River would displace about 13% of the epilimnion above 12 m depth while the augmented flow would displace about 64%. In

terms of the displacement of chlorophyll *a* (algal biomass) the augmented flow could effectively reduce the average chlorophyll *a* concentration from 30 mg m^{-3} to around 12 mg m^{-3}, improving the water quality from supertrophic (TLI >5) to eutrophic (TLI between 4 and 5). However, algal growth would continue to increase the concentration of chlorophyll *a* and there would most likely be no change in the lake trophic status. The report comments around a gradual improvement in lake water quality over time with enhanced flushing probably are correct.

The caveats included in this section are valid and should be considered before employing augmentation as a tool for the restoration of Lake Hayes.

Because of the various errors and omissions associated with the entrainment of surface water into the inflow and the consideration of the augmentation water rather than the combined augmented inflow, the information and conclusions drawn in this appendix need to be revised after new calculations.

Appendix 3: A rough Lake Hayes alum dosing estimate. This appendix is a 'cut and paste' of calculations done for Lake Hayes by John Quinn, Max Gibbs and Chris Hickey (NIWA) in 2012.

Appendix 4: Catchment management to restore and protect Lake Hayes should be the primary focus of any restoration package on Lake Hayes. Catchment management strategies will have long-term benefits and can be augmented by short-term strategies that address specific issues designed to improve the water quality of the lake.

The identification of which part of the catchment the phosphorus is coming from (e.g., Caruso 2000b) should allow targeting of management strategies for best use of limited resources.

2.6 Summary

Within the context of the original request by the Friend of Lake Hayes Society Ltd, the Schallenberg and Schallenberg (2017) report presents a comprehensive description of the current state of the Lake Hayes and summarises the major issues affecting lake water quality. It presents recommendations and discusses realistic restoration options together with a restoration strategy with timelines. The report also recommends monitoring strategies for assessing lake status and recovery to a stable water quality state.

Data used within the report is mostly pre-2015 and the more recent data may have changed or reinforced some of the conclusions. The omission of the entrainment factor requires a re-assessment of the conclusions about the efficacy of the use of irrigation water to augment the Mill Creek inflow as a strategy for accelerating the restoration of Lake Hayes water quality.

2.7 Recommendations

- The report needs to be revised to correct or qualify the identified errors and to revisit the effects of entrainment on the oxygen transport via the Mill Creek density current into the hypolimnion.

- The report needs to manage the reader's expectation of what is meant by 'recovery' and 'restoration' in terms of the likely level of improvement and the timeframes for that improvement.

3 A look at three defined options for water quality improvement

The three defined options for water quality improvements – flow augmentation from the Arrow River irrigation scheme, destratification and sediment capping – are in-lake remedial actions that could potentially have immediate impacts on lake water quality. Flow augmentation from the Arrow River irrigation scheme has largely been covered in the Schallenberg and Schallenberg (2017) report review but will also be addressed in this section in more detail with an inclusion of the entrainment factor associated with the Mill Creek inflow.

To assess the three defined options for water quality improvement, all available and relevant data on Lake Hayes were compiled to provide time series databases of nutrients and chlorophyll-*a* (as an indicator of algal biomass), as well as temperature, DO, pH, suspended solids and alkalinity. This information includes depth-correlated temperature, DO, chlorophyll-*a* and specific conductivity profiles through the full depth of the lake water column, and data from occasional research studies.

These occasional research studies include periods of high frequency temperature and DO data from a thermistor chain with multiple sensors at fixed depths and fitted with near surface and near bottom DO loggers, recording at 15-minute intervals in summer (Figure 1-4, Figure 1-5). Lake circulation patterns were inferred from an acoustic Doppler current profiler (ADCP) mounted on the bottom of the lake on the western side and recording in burst mode for 1 minute at 5-minute intervals (Figure 1-2, Figure 1-3). Key findings from these ad hoc research projects have been presented in Section 1.1 of this report.

An outcome from the Schallenberg and Schallenberg (2017) report review is the inverse relationship between water clarity and biomass of *Ceratium hirundinella*. If the water clarity in Lake Hayes is to improve, *Ceratium* biomass needs to be reduced and therefore it is important to understand the factors that favour *Ceratium* growth and dominance in the lake phytoplankton community. Consequently, information on the physiology of this nuisance bloom-forming dinoflagellate has been compiled from literature, including information from the Schallenberg and Schallenberg (2017) report. A summary of key factors are presented here.

3.1 *Ceratium hirundinella*

Ceratium hirundinella is a motile, horned dinoflagellate, with a cell length of 150 µm to 200 µm. It does not have a silica sheath as found on diatoms. Rather it has an armoured thecae comprising cellulose plates and giving the cells a high carbon to volume ratio. Because it is motile, it has the ability to regulate its depth for optimal light for photosynthesis during the day, and can move down to the thermocline to find nutrient N and P from the hypolimnion at night. It usually confines itself to a well-defined depth layer during the day (Figure 3-1) but may disperse more widely at night. It has essentially the same physical requirements for growth as for cyanobacteria (i.e., calm conditions and thermal stratification) and its growth is favoured by low N:P ratios. (Hart and Wragg 2009). They found that *Ceratium* became dominant when N:P ratios were around 5 but disappeared when the N:P ratio was >10. Nakano et al. (1999) also found that *Ceratium* blooms in Lake Biwa were related to increases in P in the water column but found that growth may be nitrogen limited. i.e., increases in both N and P with an N:P ratio <10 are conditions that favour the development of *Ceratium* blooms.

Unlike cyanobacteria, *Ceratium* is mixotrophic, i.e., it can gain energy both through photosynthesis as plants do, and by feeding on bacteria as some protozoans and zooplankton do (Gerdeaux and Perga 2006). Consequently, while *Daphnia* may be physically large enough to consume *Ceratium* cells, their ability to outcompete *Ceratium* for available bacteria may be the main cause of the reduction in *Ceratium* biomass in some years., i.e., the *Ceratium* may become carbon limited.

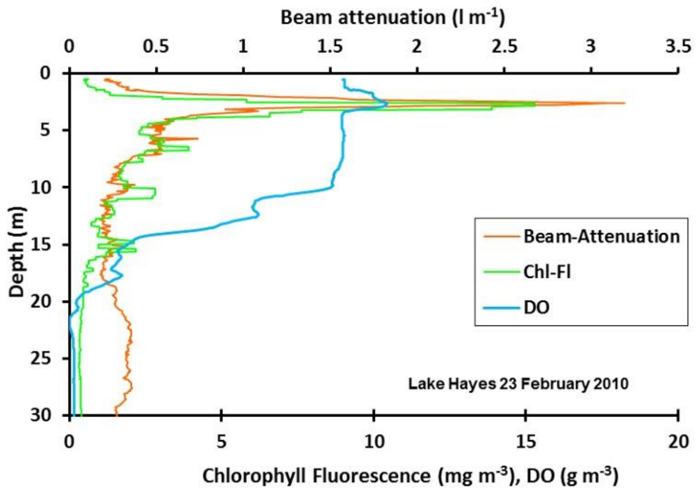


Figure 3-1: Particulates (beam attenuation), chlorophyll fluorescence and dissolved oxygen profiles in Lake Hayes on 23/02/2010, about 10 am. The beam attenuation shows that most particles were concentrated in a thin layer around 3 m depth. The chlorophyll fluorescence peak at the depth of the beam attenuation peak shows that the particles were phytoplankton (identified as *Ceratium hirundinella*). The DO increase at the depth of the chlorophyll peak shows that the *Ceratium* were photosynthesising and therefore they were alive. The thermocline depth was 10 m. (NIWA unpublished data).

Ceratium can tolerate a broad range of temperature with a development range between 4°C and 23°C (Heaney et al. 1983) and an optimal growth range between 12°C and 23°C (Hutchinson 1967, Heaney et al. 1988). *Ceratium* over-winters by producing cysts that fall to the sediment in autumn and germinate in spring (Heaney et al. 1983).

During the summer growth period, *Ceratium* cells divide at night (between 9 pm and 9 am with the majority of divisions occurring at about 3 am in the morning (Heller 1977, Heaney and Talling 1980). At this time, the *Ceratium* cells would be on the thermocline with access to high P concentration from the hypolimnion. It is possible that they rest on the thermocline to facilitate cell division.

Low light and temperature are unfavourable conditions for *Ceratium* in temperate zone lakes (Heaney et al. 1988) whereas high temperatures are unfavourable conditions in subtropical zone lakes in summer (Pollingher and Hickel 1991).

Growth is potentially regulated by ecological factors such as the presence of an anoxic hypolimnion and water column stability (Pérez-Martínez and Sânchez-Castillo 2002). In this case the anoxic hypolimnion provides the excess P required for growth.

Based on the above information, managing *Ceratium* blooms in Lake Hayes will require the reduction of both external and internal P inputs to the lake water column. Destratification of the lake could remove the thermocline, which appears to be a resting zone for *Ceratium* at night as well a place for cell division and a 'refuelling' site for gathering P.

Destratification would also oxygenate the bottom water and reduce or prevent the release of the DRP flux from the sediment – a key part of the internal cycling of P. Augmenting the Mill Creek inflow with excess irrigation water from the Arrow River would increase the transport of oxygen into the lake. This could reduce the DRP flux from the sediment by precipitating it with the ambient iron and manganese oxides. Sediment capping with alum would also sequester any DRP from the water column and would cause the precipitation of the particulate P and *Ceratium* cells in the resultant floc.

3.2 Flow augmentation from the Arrow River irrigation scheme

While it would be a relatively simple procedure to introduce irrigation scheme water from the Arrow River into Mill Creek, estimating the effect of that augmentation on the restoration prospects for Lake Hayes is more complex. If it is assumed that Arrow River and the Mill Creek waters are 100% saturated with oxygen, say 10 g m^{-3}, the mass of oxygen in the combined flow can be calculated as the total flow times the oxygen concentration. Using the irrigation water volumes provided in the Schallenberg and Schallenberg (2017) report and the mean monthly flow data from the Caruso (2000b) report, the estimated mean daily mass transport of oxygen from the Mill Creek inflow in January would be about 0.45 t d^{-1}. This is much less than the oxygen demand estimated in the HOD rate of about 1.57 t d^{-1} for January 2017. However, if an entrainment factor of 4 is included in the calculation, the mass of oxygen in the density current formed by the Mill Creek inflow would be about 1.8 t d^{-1}, more than enough to compensate for the oxygen demand in the hypolimnion.

The complexity of this process is apparent when considering that Mill Creek is already flowing into the lake carrying about 1.6 t d^{-1} during January and the augmentation water would only add another 0.2 t d^{-1}. Despite this, the hypolimnion is still losing oxygen so that it became anoxic in March 2017. This implies that other factors are influencing the mass transport of oxygen into the hypolimnion of Lake Hayes and that the HOD rate is the net loss of oxygen from the hypolimnion.

The main factor affecting oxygen transport into the hypolimnion will be the temperature difference between inflow water from Mill Creek and the surface water in Lake Hayes (Figure 3-2).

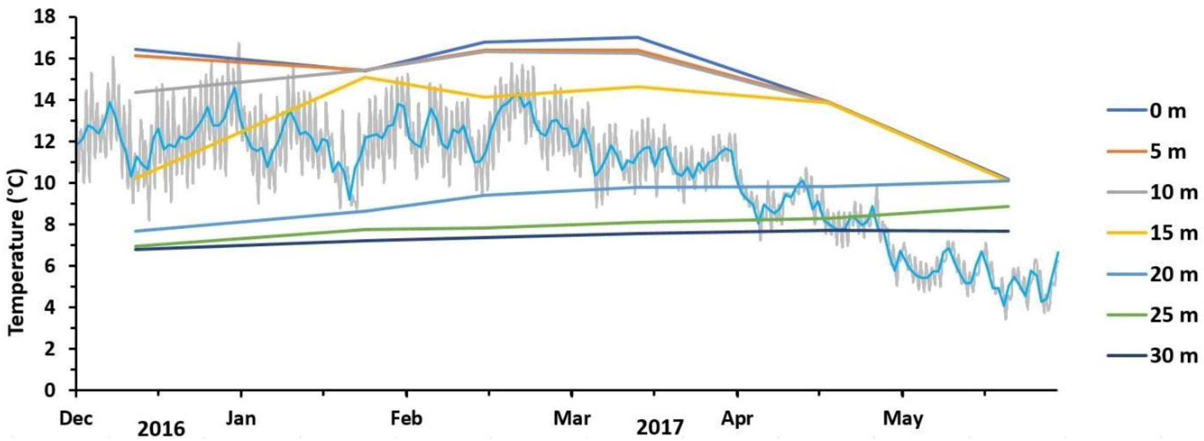


Figure 3-2: Comparison of Mill Creek water temperatures (wavy blue [daily] and grey [hourly] line) with lake temperatures at different depth (coloured lines - depth scale right hand axis). Mill Creek temperature data from 2016 to2017 show the diurnal cycle of heating and cooling (grey hourly line) and have been manually aligned with periodic Lake Hayes temperature profile data. (Temperature data from ORC).

Mill Creek data show a strong diurnal temperature cycle which has a range of about 4°C between midday and midnight. In Lake Rotoiti, that level of temperature difference was enough to direct water from the Ohau Channel from Lake Rotorua into a surface buoyant plume during the day or cause it to plunge as an underflowing density current along the bed of Lake Rotoiti at night (Vincent et al. 1991). There is no question that a similar effect will be happening in Lake Hayes, although the magnitude will be smaller. The intrusion layer from Mill Creek inflow is apparent in the 21-m depth temperature and DO data from the thermistor chain (Figure 1-5). The other difference will be how long the underflowing density current feature is operating each day and whether the density current flows into the hypolimnion (condition A, Figure 3-3) or finds a depth of neutral density higher in the lake and lifts off the lake bed to become an intrusion flow (condition B, Figure 3-3).

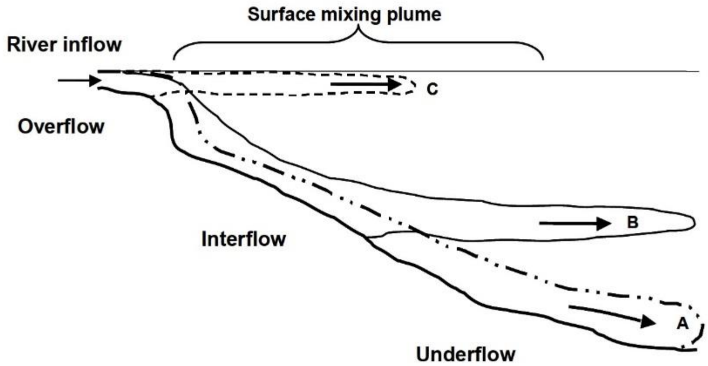


Figure 3-3: Density currents. **A**) Very cold water will become an underflow along the lake bed, **B**) medium temperature water will plunge to a depth of equal density where it becomes an interflow, **C**) warmer water enters the lake surface as a buoyant overflow which may form a visible plume on the surface. Both interflow and underflow entrain surface water with the flow. (Figure 3-7 from Gibbs and Hickey 2012).

Physical factors that influences the formation of a density current is the shape (width and depth) of the stream mouth and the bed shape in the lake at the mouth. In Lake Rotoiti, the Ohau Channel flowed over a sill with a 5-m drop-off giving a sharp transition from inflow to lake and a geometry that encourages entrainment of the lake surface water into the flow (entrainment factor of about 4-5). In Lake Hayes, Mill Creek flows into 8 m deep water and this is likely to result in a similar inflow/entrainment geometry. The entrainment factor can be calculated but requires more data than is currently available. It would be reasonable to expect an entrainment factor of around 4.

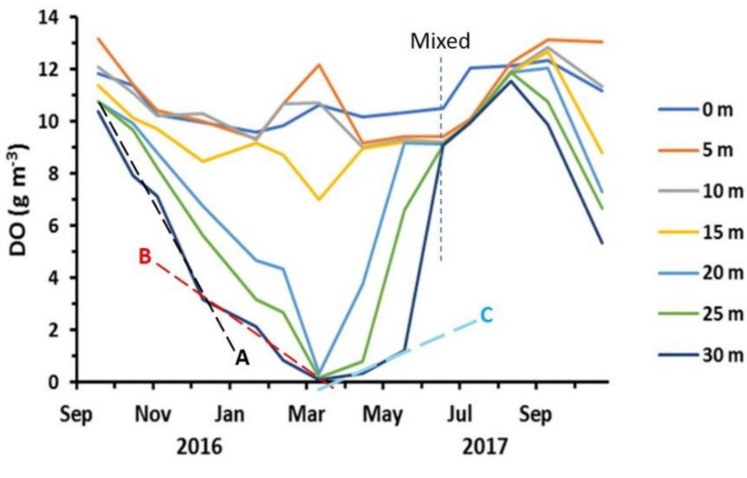


Figure 3-4: Time-series dissolved oxygen (DO) profiles in Lake Hayes 2016/17. Broken lines are regression lines which indicate the rate of hypolimnetic DO depletion at different times across the stratified period: A) spring (November-December: [HOD rate 85 mg m^{-3} d^{-1}], B) summer (January-March): [HOD rate 35 mg m^{-3} d^{-1}], C) Autumn (March-May): [HOD rate -16 mg m^{-3} d^{-1}]. The lake was fully mixed in June.

Comparing the Mill Creek temperature data with the 15-m depth temperature (Figure 3-2), it is apparent that through December 2016 the inflow would most likely form an interflow into the epilimnion or metalimnion (thermocline). Without a large proportion of underflow, the oxygen depletion rate is relatively rapid at 85 mg m^{-3} d^{-1} (line A, Figure 3-4). In January 2017, the inflow is likely to be switching between interflow and underflow with an increasing proportion of underflow contributing oxygen the hypolimnion resulting in a reduction in the oxygen depletion rate to around 35 mg m^{-3} d^{-1} (line B, Figure 3-4).

Around March, the Mill Creek water temperature decreased markedly (Figure 3-2) and it is likely that the density current would be mostly as an underflow. If the transport of oxygen in that underflow was greater than the oxygen demand in the hypolimnion, the hypolimnion would begin to re-oxygenate. The apparent reoxygenation starting in March 2017 at a rate of around 16 mg m^{-3} d^{-1} (line C, Figure 3-4) is consistent with an oxygen input load greater than the oxygen demand in the hypolimnion. The oxygen recharge between the May and June sampling dates shows an average reoxygenation rate of around 300 mg m^{-3} d^{-1}. However, this reoxygenation phase is normally associated with lake turn over that occurs over a period of about a day, which would represent a rate of around 8000 mg m^{-3} d^{-1} (Figure 3-5).

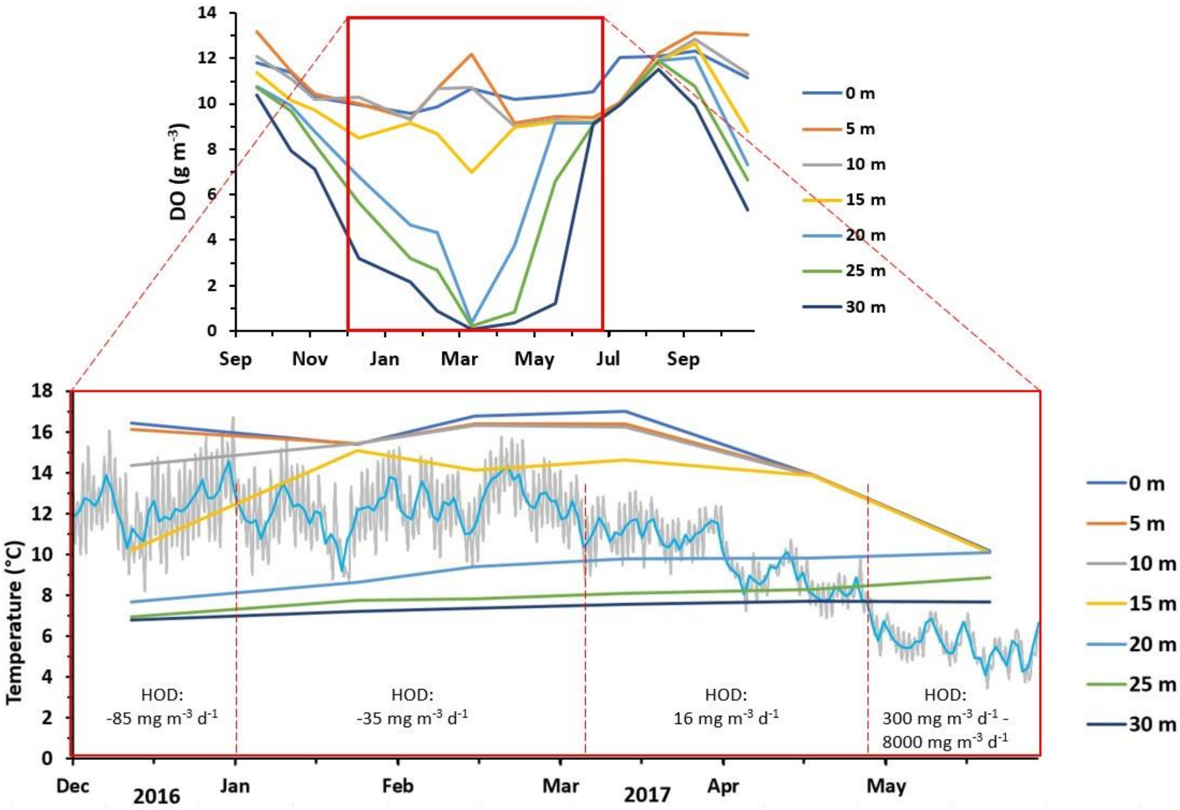


Figure 3-5: Alignment of dissolved oxygen data with the temperature data from Figure 3-4. Negative HOD values indicate oxygen loss.

These data indicate that, assuming the water temperatures in Mill Creek follow a similar pattern each year, the present inflow from Mill Creek with entrainment is sufficient to slow the oxygen depletion in summer and reoxygenate the hypolimnion in autumn. Augmentation of the Mill Creek inflow with irrigation water from the Arrow River would increase the mass of oxygen transported by the density current. If the temperature of the water from the Arrow River is colder than Mill Creek (large rivers are often colder than small streams) then the augmented flow would be likely to enhance underflow instead of interflow through November to January. This would greatly reduce the degree of oxygen depletion in the hypolimnion.

The brief period of anoxia in March 2017 suggests that the lake is close to a tipping point of no anoxia and therefore a greatly reduced internal loading of DRP, which is likely to be driving the growth of the *Ceratium* blooms. Enhanced underflow in spring and early summer could be sufficient to prevent that anoxia occurring.

Caution: These scenarios are based on the temperature regimes in the lake and in Mill Creek in 2016-17. There is no guarantee these conditions are the new norm and will not change to a different regime that reduces the underflow as climate variability causes larger swings in the temperature cycle.

For example, temperature data from Mill Creek in 2018 reached day time maxima of 23.4°C (29 January 2018), which was 5°C warmer than the same period in 2017. There is no lake profile water quality data available for early 2018 at the time of writing this report to determine what affect this had on hypolimnetic oxygen depletion. There was, however, an increase in algal biomass suggesting a possible release of DRP from the sediment. A chlorophyll fluorescence profile on 20 January 2018 indicates thermal stratification and the dominant algal species was the diatom. cf. *Cyclotella* sp.

When the Mill Creek inflow temperatures are aligned with the depth of the thermocline i.e., about 15 m depth, from January to March (Figure 3-2), there is another factor that that may determine the depth of insertion of the density current and its ultimate mixing in Lake Hayes, i.e., wind speed and direction.

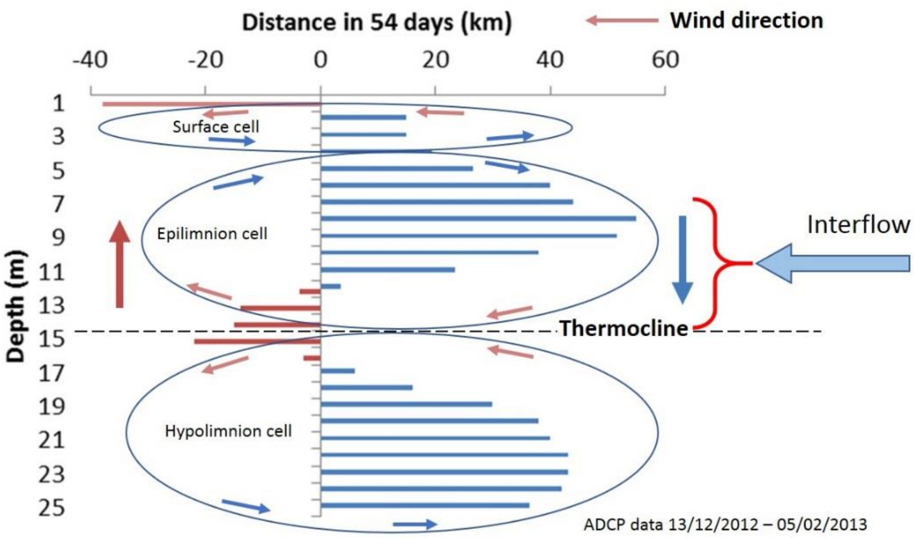


Figure 3-6: Lake circulation currents are likely to affect the depth of insertion of the interflow. Figure shows the effect of a persistent northerly wind flow from December 2012 to February 2013. (NIWA unpublished data).

The short-term deployment of an ADCP in Lake Hayes in December 2012 through to February 2013 found that the persistent northerly wind flow along the axis of the lake set up internal lake currents that formed contra-rotating vertical cells in the water column. While this is not unusual in many lakes, in Lake Hayes, the cells meet at the thermocline depth with downwelling at the upwind end and upwelling at the downwind end. This means that water from Mill Creek will be drawn down to the thermocline if it forms an interflow that inserts below about 10 m depth (best guess) (Figure 3-6). Consequently, the oxygenated water in the density current is likely to be entering the hypolimnion for a longer period each day than indicated by temperature difference alone. When the inflow forms an underflowing density, this pattern of lake currents will not affect the depth of insertion below the thermocline but will rapidly disperse the oxygenated water throughout the hypolimnion. When the wind direction changes to a persistent southerly, the downwind end is against the Mill Creek inflow and this would most likely prevent all but an underflowing density current from reaching the hypolimnion.

Further evidence of reducing oxygen depletion is shown in DRP concentration data from the hypolimnion (Figure 3-7) and the maximum chlorophyll data in the epilimnion (Figure 3-8).

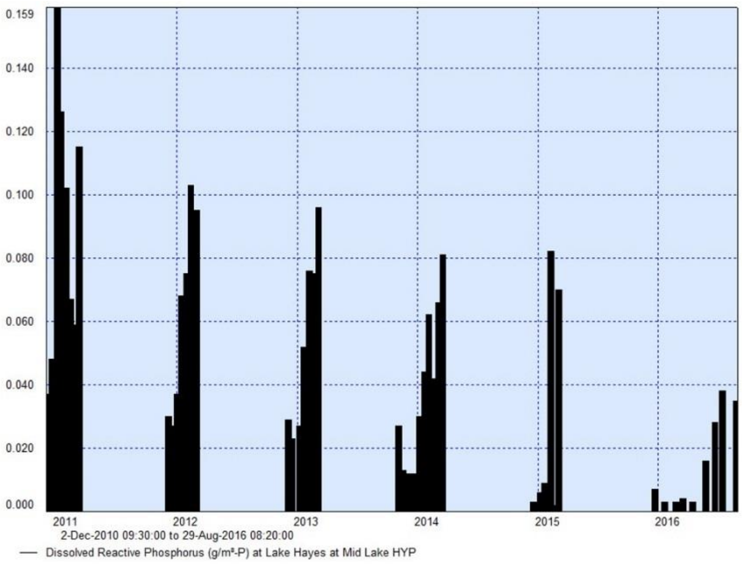


Figure 3-7: Time-series dissolved reactive phosphorus (DRP) concentrations in the hypolimnion from 2011 to 2016. Between 2011 and 2015 (inclusive) data was only collected between December and March (inclusive). (Data from ORC).

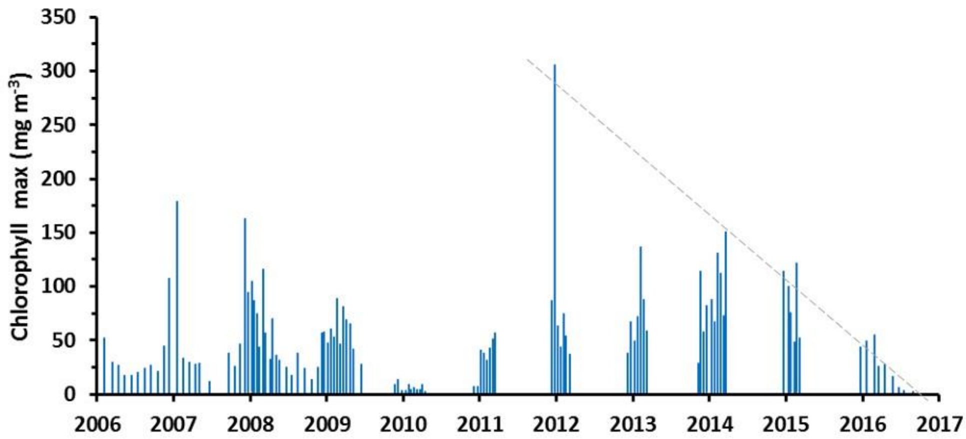


Figure 3-8: Maximum chlorophyll concentrations in summer from time-series profiles in Lake Hayes. Profiles were collected at the mid lake site as part of the ORC monitoring of the lake. From 2011 to 2015, the data were only collected between December and March (inclusive) and do not show the winter concentrations. Dashed line provides a visual indication of trends based on the maximum chlorophyll concentrations.

Phosphorus is released from the sediment under anoxic conditions and, between 2011 and 2016, the DRP concentrations have gradually reduced with the release time moving towards March, consistent with a reduction in the intensity of anoxia. The very low DRP release in 2016-17 is consistent with the very brief period of anoxia that summer (Figure 3-4) and may be the cause of the low algal biomass that summer (Figure 3-8) rather than just grazing pressure from zooplankton.

The apparent decrease in maximum chlorophyll concentrations from the extreme high in 2012 (Figure 3-8) appears to be consistent with the reduction in DRP in the hypolimnion (Figure 3-7), although there may be other plausible explanations for this change in the time-series data. Note that summer 2009-10 was a clear water year (Schallenberg and Schallenberg 2017) with the appearance of *Daphnia pulex* in high abundance. This occurrence disrupted NIWA and University of Otago experiments looking at flocculation and sediment capping (personal observations by the author).

The 2016-17 chlorophyll maximum concentrations were not as low as in 2009-10 but this was also classified as a clear water year, with the occurrence of high numbers of *Daphnia pulex* (Schallenberg and Schallenberg 2017) although there are zooplankton data to support this statement.

A prediction in December 2017 that summer 2017-18 was also set to be a clear water year (Adam Uytendaal, ORC, pers. comm.) did not allow for climate variability and the warmest summer in many years. This presumably resulted in release of DRP from the sediment, which a proliferation of algae and a reduction in water clarity.

3.2.1 Conclusions

There are indications that Lake Hayes is moving from a supertrophic condition towards a eutrophic condition with slowly improving water quality. Whether this is a tipping point for recovery, as suggested in the Schallenberg and Schallenberg (2017) report, is uncertain. Indications of improving water quality include:

- Hypolimnetic oxygen depletion rates are decreasing and are presently less than half the maximum rate recorded in 2007.

- The period of bottom hypolimnion anoxia has reduced from >5 months in 2007-8 to <1 month in 2016-17.
- Reoxygenation of the hypolimnion from the Mill Creek inflow is implied in the data from January in the summer of 2016-17. There was no indication of reoxygenation in summer 2007-8 until March when the water column began to reoxygenate from the 10-m isobath downwards rather than from the bottom (Figure 2-2).
- The release of DRP from the sediment has been slowly decreasing since 2011.
- Algal biomass, as indicated by chlorophyll concentrations, appears to have been decreasing since 2011.

These changes are all consistent with a change in the amount and duration of the underflow from the Mill Creek inflow. Other factors could also explain these changes. For example, a gradual reduction in maximum algal biomass in the epilimnion would result in a lower organic carbon load in the sediment and therefore a reduced sediment oxygen demand. This would be consistent with the reducing HOD. This mechanism appears to be occurring in Lake Rotorua where the water quality has improved from eutrophic to mesotrophic after reducing the DRP input to the lake.

Acoustic doppler current profiler data indicates that wind flow along the axis of the lake may be enhancing the mixing of DO into the lake when there is a northerly wind and could be suppressing that mixing during a southerly wind. A schematic diagram (Figure 3-9) shows stylized flow paths of oxygen transport from several different sources using the different mechanisms. This is based on the minimal amount of data available and, although consistent with other studies, is speculative for Lake Hayes and requires further investigation.

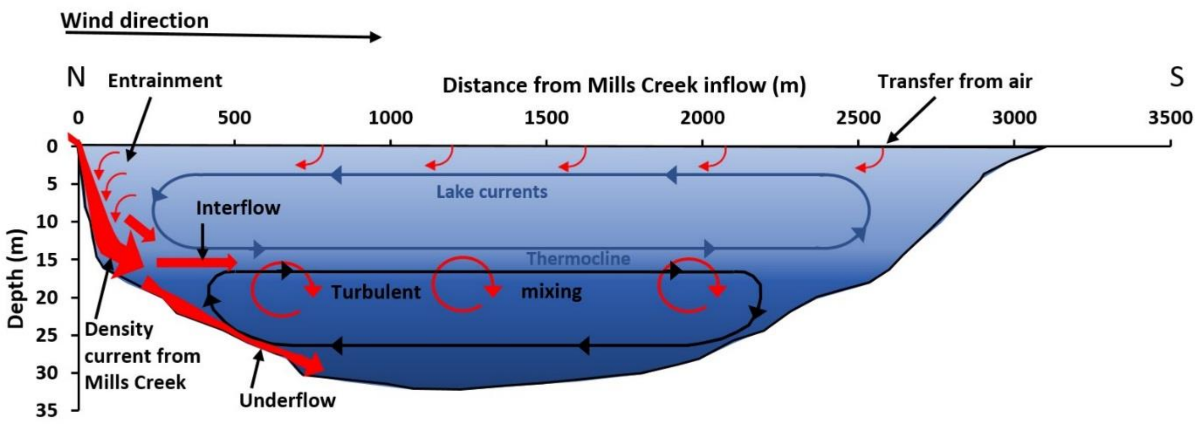


Figure 3-9: Schematic diagram showing a vertical cross-section along the north-south axis of Lake Hayes overlaid with oxygen flow pathways and transport mechanisms. Stylised oxygen input pathways (red) show where entrainment occurs. Internal lake currents – blue (epilimnion), Black (hypolimnion) – interact with the oxygen pathways and mix oxygen into the hypolimnion due to turbulent mixing at the thermocline (Schematic by Max Gibbs).

If the addition of irrigation water from the Arrow River causes the temperature to fall, the period of underflow will increase and the hypolimnion is likely to remain aerobic throughout summer and the recovery of the lake will be accelerated.

However, if the addition of the irrigation water causes the temperature in Mill Creek to rise, the period of underflow most likely will be reduced and the apparent natural recovery of the lake could be reversed.

Caution is required. This is a very fragile situation that is likely to change if climate variability causes Mill Creek to warm as is indicated in the 2018 data, i.e., the reoxygenation of the hypolimnion by the oxygen transfer from the Mill Creek inflow cannot be relied on as the long-term solution for restoration of Lake Hayes.

3.2.2 Recommendations

A lake monitoring buoy should be installed on Lake Hayes to obtain high frequency (15-minute interval) temperature and dissolved oxygen concentrations at selected depths in the lake water column in order to understand how this lake works.

As temperature during the stratified period appears to be critical to the recovery of Lake Hayes using the augmentation of Mill Creek, it is critical to measure the temperatures of Mill Creek and the irrigation water before any augmentation is implemented. Temperature monitoring should continue for a full year to enable comparison with temperature measurements from the lake monitoring buoy and determine when it is appropriate to add irrigation water and when it is not.

In the interim, the long-temperature record from Mill Creek and the periodic temperature and DO profiles from Lake Hayes may provide sufficient data to give an indication of what temperature regime causes underflow and reoxygenation of the lake hypolimnion, and how often those conditions occurred over the data record. These data could be modelled to determine when and where in Lake Hayes the Mill Creek water inserts and mixes each year.

In-lake currents induced by different wind directions should be measured in order to understand how these currents affect the dispersion of DO into Lake Hayes. A pair of ADCPs should be used, one on each side of the lake, in order to determine the lateral circulation patterns as well as the vertical currents. These data should be correlated with wind speed and direction and rainfall records for the Lake Hayes area to further develop the mechanism for reoxygenation suggested.

3.2.3 Secondary or amenity effects

Because the addition of augmentation water to Mill Creek will only increase the volume by a small amount (14%, Schallenberg and Schallenberg 2017) there should be no visual impact on the stream and any change in water clarity would most likely be imperceptible to the eye. Provided the irrigation water added to Mill Creek is odourless, there should be no perceptible change in odour and only a small change in the sound of the stream as flows over the gravel beds.

3.3 Destratification

3.3.1 Why destratify the lake?

Destratification removes the thermocline, which is the single greatest barrier to the transfer of dissolved oxygen down to the bottom of the lake. The loss of DO due to sediment decomposition processes is the main cause of bottom water anoxia. This causes P release from the sediment as an internal load that stimulates the development of summer algal blooms, most recently of the dinoflagellate, *Ceratium hirundinella*. As discussed in section 3.1, *Ceratium* moves up through the water column to the well-lit photic zone during the day and then settles down to the thermocline at night. It is this night phase where the thermocline is important as the *Ceratium* cells use it as a resting platform so they can take up DRP and ammonium from the hypolimnion. They may also use it as a resting place during cell division, which also occurs at night. Removing the thermocline would interrupt feeding and cell division, thereby attacking the key growth steps in the *Ceratium* life-cycle. Studies also indicate that surged operation of the aeration system may be used manage some algal species (Lilndenschmidt 1999).

A paper by Kirke (2000) provides an overview of destratification and mixing using aeration and mechanical mixers. There are several options for lake mixing (e.g., Singleton and Little 2006a, b) and destratification. Most of these either use air-lift techniques or propellers to induce vertical mixing currents to the surface, or water jets to carry surface water down into the lake. The simplest aerator is the bubble-plume system where compressed air is blown through pipes to one or more diffusers on the lake bed. This diffuser system is referred to as a sparge pipe. The design of the sparge pipe produces a rising column of air bubbles, which entrains bottom lake water into the plume and induces a vertical current in the water column (Schladow 1993) (Figure 3-10). The depth of the air outlet is important - the greater the depth, the more efficient the mixing (Cooke et al. 1993). This is because bubbles produced from the sparge pipe expand as they rise to the surface. When they leave the sparge pipe jets they double in size and their volume increases by a factor of 4 for every 10 m they rise (Figure 3-10 A). This means that, in Lake Hayes at 30 m maximum depth, the bubbles will expand by a factor of 64 as they reach the surface.

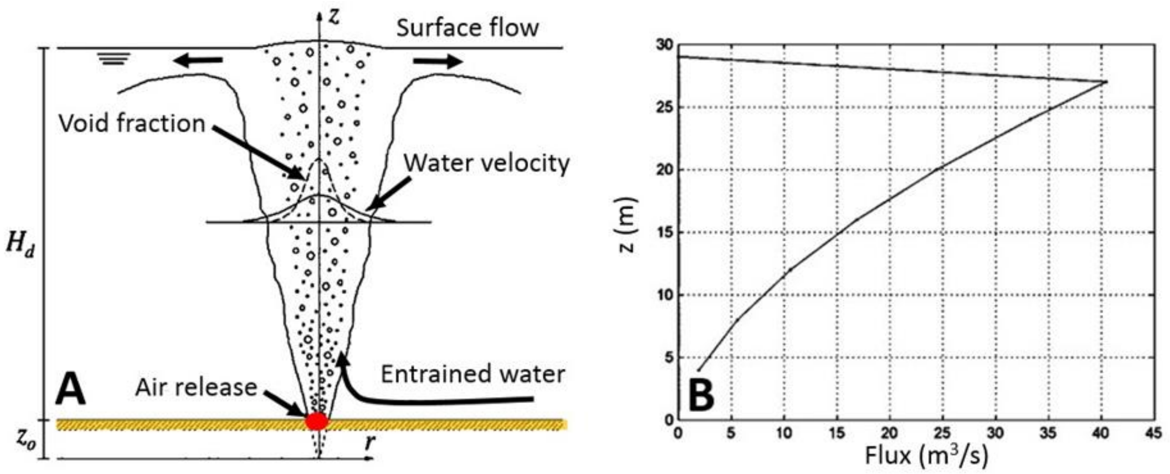


Figure 3-10: Bubble plume representation **A**) Near isothermal water column, **B**) estimation of vertical water movement (flux) in $m^3\ s^{-1}$ (Stylised images redrawn from internet images. Flux values would depend on the volume of air and the length of the diffuser(s)).

Because the air bubbles occupy a finite volume in the water column but have almost no mass, the water in the bubble plume is less dense (more buoyant) than the adjacent lake water and the water inside the bubble plume rises to the surface as a buoyant plume. As the bubble plume rises, it entrains bottom water and ambient water from outside the plume so that the total flux of water moving upwards in the plume increases (Figure 3-10,B). The amount of air pumped into the sparge line will influence the efficiency of the rising plume. Too little and the plume may not form. Too much and the bubbles may disrupt the integrity of the plume reducing its efficiency.

The air flow required to de-stratify a lake is a function of the shape of the lake and the surface area. USEPA estimate that a compressed air flow rate of 9.2 $m^3\ min^{-1}\ km^{-2}$ lake area should be sufficient to achieve adequate surface re-aeration in most lakes (Lorenzen and Fast 1977). This is equivalent to 9.2 x 35.31 = 325 cubic feet per minute (cfm).

At the surface, the entrained water is dispersed laterally away from the bubble field as a surface current (Figure 3-11), which will propagate across the lake until it reaches an obstruction or the shore. There the hydraulic inertia of the water in the surface current causes it to plunge to the lake bed where it will replace the bottom water being entrained into the bubble plume. Note that, although the air bubbles in the plume make the water less dense, the density difference is small and insufficient to pose a hazard to boats or swimmers.

Figure 3-11: View of the surface bubble field along the axis from a bubble plume aeration system. This bubble plume aerator was installed in Lake Waikapiro, Hawke's Bay. This photo shows the current lines moving away to the left and right from the line of the bubble plume. [Photo by Andy Hicks, Hawke's Bay Regional Council].

The bubble plume aeration/mixing mechanism is more efficient than any other method of de-stratifying a lake, or preventing it from becoming thermally stratified. The air used to generate the bubble plume has very little effect on the oxygenation of the bottom water entrained to the surface in the plume. Oxygen adsorption from the atmosphere re-oxygenates the oxygen depleted water as it flows across the surface of the lake.

3.3.2 Bubble plume aerator design and positioning

Because Lake Hayes was formed from in a glacial valley, the lake morphometry is simple with no ridges or rock features on the bottom (i.e., there is nothing to obstruct the circulation currents that will be generated by the destratification mechanism). The lake is elongated – 3.1 km long by about 800 m wide and is essentially bath-shaped. The main inflow, Mill Creek, is at the northern end and the outflow, Hayes Creek, is at the southern end. The deepest part of the lake is near the middle at about 1.2 km from the northern end. This would be the ideal position for the installation of a bubble plume aeration system.

The design of the bubble plume aerator is very simple. It comprises a sparge pipe diffuser attached by tethers to a heavy cable (ballast rope) that is laid across the lake bed (or reservoir) and is anchored at each end with heavy concrete blocks (Figure 3-12). The sparge pipe diffuser has a series of small (1–1.5 mm) holes drilled through the upper side at 20–30 cm intervals along its length. Compressed air pumped into the tube from a shore mounted compressor, escapes through these holes as tiny bubbles, which rise to the surface as a bubble plume. The heavy ballast rope keeps the sparge pipe diffuser at the bottom of the lake when that pipe is full of air. A buoyancy tube is also attached to the ballast rope (Figure 3-12). During normal operation, the buoyancy tube is full of water and is held at the bottom of the lake by the weight of the ballast rope. When the water is blown out of the buoyancy tube and it is full of air, its positive buoyancy exceeds the negative buoyancy of the ballast rope and the aerator system floats to the surface. This enables the whole system to be installed or raised for cleaning without requiring the use of divers.

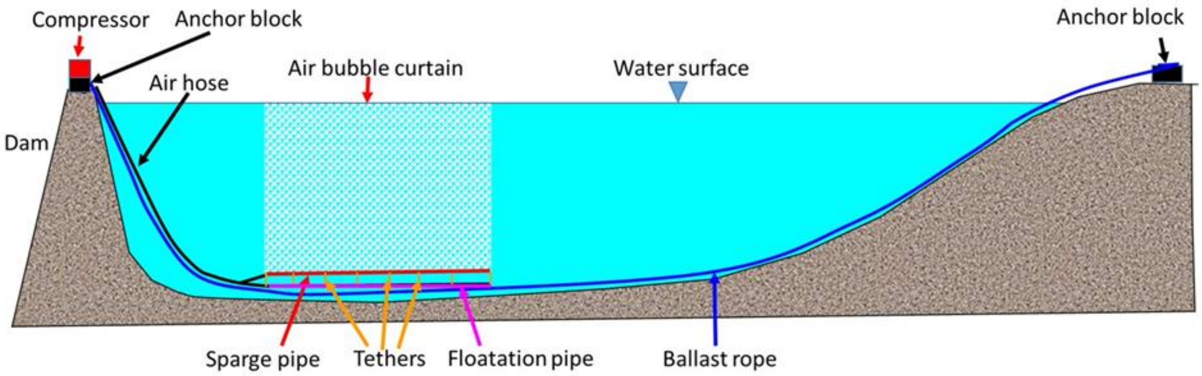


Figure 3-12: Schematic diagram of a bottom-mounted bubble curtain aerator system in a reservoir. The sparge pipe for small reservoirs is typically 50–100 m but can be much longer for larger reservoirs and lakes. The ballast rope is anchored at both ends and orientated along the deep axis of the reservoir or across the deepest part of a lake, so that it lies on top of the bed. The ballast rope can be 32 mm trawler rope or a similar sized plastic sheathed multi-core steel cable.

Orientation of the sparge pipe and its length depends on how the system is to be used. In operation, the bubble plume forms a curtain along the length of the sparge pipe. This results in a continuous strip of rising water that generates a laminar flow current on each side of the bubble curtain. In Lake Hayes, the sparge pipe should be placed across the deepest and narrowest part of the lake so that the laminar current flow is focused along the length of the lake (Figure 3-13).

The circulation currents will flow to each end of the lake at the surface adsorbing oxygen from the atmosphere before plunging to the bottom and flowing along the lake bed to the aerator sparge line. This orientation generates a circulation flow path (Figure 3-14) that carries oxygen to the bottom of the lake from the atmosphere and enhances the underflow from the Mill Creek inflow.

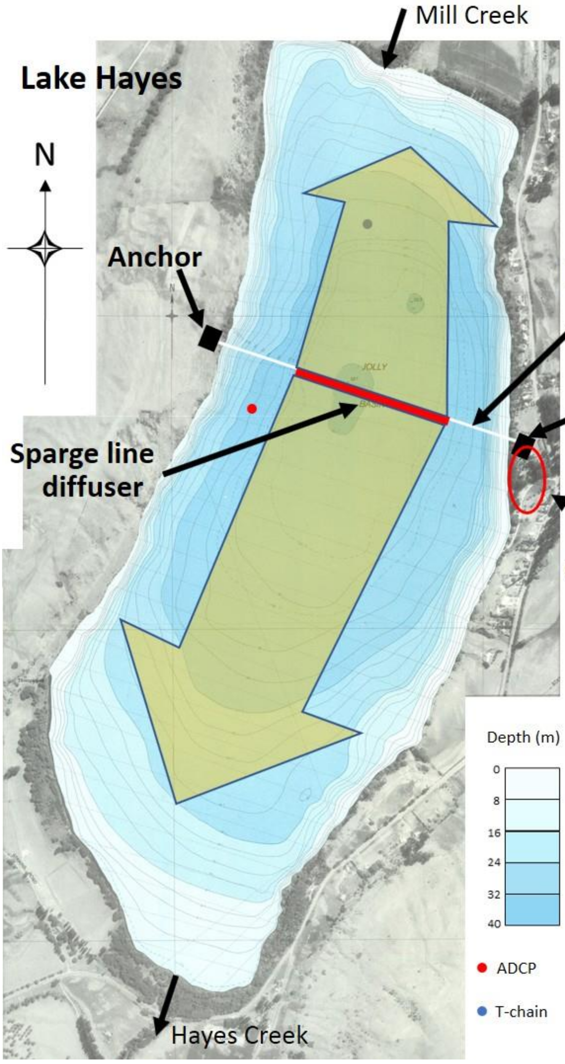


Figure 3-13: Optimal positioning of bubble curtain aeration sparge line diffuser in Lake Hayes. The expected surface flow patterns are indicated by the orange shaded arrows. The return paths are along the lake bed beneath.

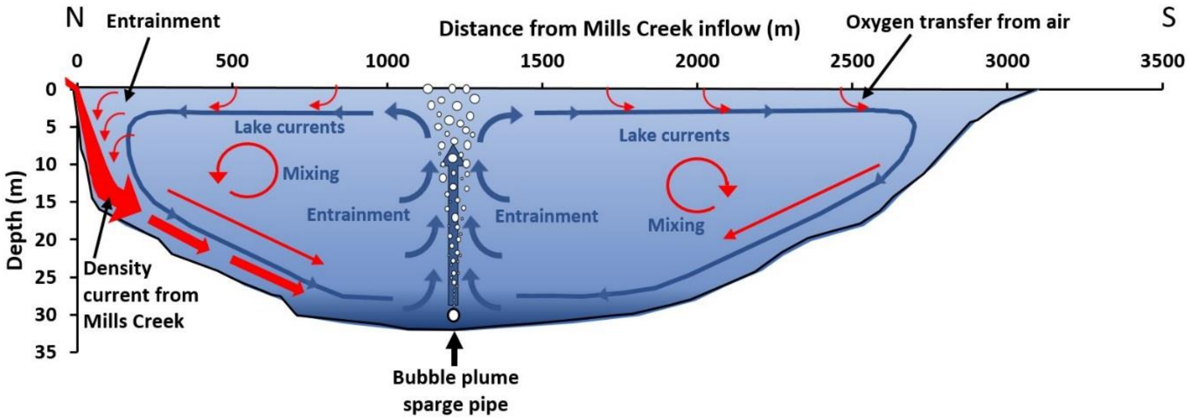


Figure 3-14: Schematic diagram of the likely circulation flow patterns in Lake Hayes with the bubble plume aerator operating. Red flow paths indicate oxygenated water. Blue lines are expected lake current flow paths. [Schematic drawn by Max Gibbs].

3.3.3 Compressor Cost estimate

Destratification of Lake Hayes is feasible and has the potential to prevent the annual cycle of bottom water anoxia and P release from the sediment as an internal load thereby suppressing the development of summer algal blooms. With an area 2.76 km^2 and a depth of 33 m, Lake Hayes is well within the capability of a single stage aeration system using a bubble plume aerator. An example of a large reservoir (72 m deep, surface area 6.22 km^2 and capacity 139 million m^3) that has successfully been mixed using aeration is El Capitan, a water supply reservoir for San Diego city in California (Fast 1968).

The bubble plume aeration for Lake Hayes (33 m deep = 3.3 atmospheres at 14 psi per at atmosphere) would require an air compressor with a minimum pressure of 3.3 x 14 psi = 46 psi or 315 kpa). With a surface lake surface area of 2.8 km^2, the compressor would need a minimum air flow of 2.8 x 9.2 = 25.8 m^3 min^{-1} or 910 cfm. For continuous operation, these specifications can be met using an 11kW screw compressor. Servicing would be once per year in winter when the aerator would be off. Estimated capital cost of the compressor is about $12K plus running costs.

The cost of the sparge line diffuser system would depend on the length of the sparge pipe (about 500 m) and diameter of the various pipes used (sparge pipe at 32 mm ID and floatation tube at 40 mm ID plus the compressed air line ~200 m), the heavy rope ballast weight (800 m) and the size of the anchor blocks required to secure the system in the lake. The system also requires a manifold to connect the compressor to the air line and connect the other pipes together.

As an example of likely costs, a similar size system installed in Lake Opuha near Timaru is estimated to have cost about $250 ± 50k installed. The compressor used is 75 kW in size, delivering 400 cfm at 80 psi. Costs associated with electricity supply are extra.

A more accurate price estimate of capital equipment, as well as running and maintenance costs for a system for Lake Hayes could be made once the design has been finalised.

3.3.4 Operational considerations

The **timing for starting the aeration system is critical**:

- It takes less energy to mix the lake when the temperature gradient across the thermocline is small, i.e., commencement of aeration should not be delayed until the thermocline is well-established.
- If it is delayed until thermal stratification has become established and the hypolimnion has become anoxic, the rising plume will entrain nutrient rich bottom water into the epilimnion and stimulate a phytoplankton bloom.

The turn on time must be determined for each year from in-lake measurements rather than by calendar date. Based on data from two periods in the temperature – DO records from Lake Hayes (Figure 3-15) an aeration system, if installed, should have been turned on in September and turned off in May the following year. Ideally, the turn on should be earlier than later and the aerator should always be turned on if the DO concentration falls below 7 g m^{-3}. Turn off should be at the end of April or early May but is not as critical.

The caveat to this turn on regime is that, if the optimum start time is missed, a late turn on may still be possible if the water column nutrient concentrations remain low and there is no algal proliferation in the lake. The risk is greatly increased that any nutrients accumulating in the hypolimnion will be mixed up into the water column and will stimulate an algal bloom. In that situation, it may be prudent not to use the aeration system for that year.

For example, a late turn on resulted in a major algal bloom in Lake Waikopiro, Hawke's Bay in summer 2017-18. When the bloom suddenly collapsed, the oxygen demand from the decomposing algal cells caused the entire lake to become hypoxic (<2 g m^{-3} DO) and resulted in an extensive fish kill.

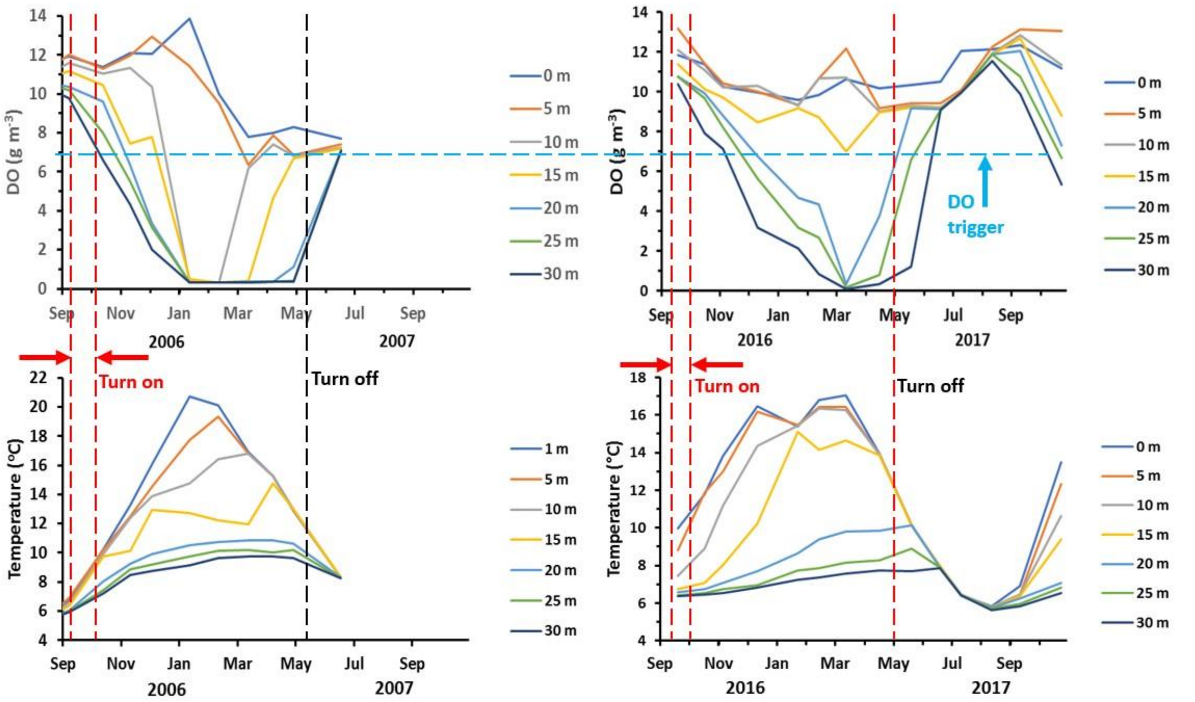


Figure 3-15: Examples from 2006-07 and 2016-17 of how to determine the time for turning the aeration system on and off. The two red vertical dashed lines between horizontal red arrows indicate the range tolerance for turn on based on the degree of thermal stratification and the level of oxygen depletion. The single red vertical dashed line indicates when it is safe to turn the system off. The horizontal blue dashed line is the trigger level for dissolved oxygen. If the DO concentration falls below this line the aeration system must be turned on as soon as possible.

3.3.5 Conclusions

Destratification will mix the lake through its full depth, providing several lake water quality benefits:

- An aeration system using a sparge diffuser across the narrowest part of Lake Hayes will cause the lake to become de-stratified in summer and will improve the water quality of the lake.
- Development of bottom water anoxia will be eliminated, minimising release of DRP from the sediments.
- Ammonium released from the sediments will be nitrified into nitrate, some of which may be denitrified into nitrogen gas and lost to the atmosphere.
- The reduction in DRP concentration without a substantial reduction in DIN produces a high N:P ratio which does not favour the growth of cyanobacteria or *Ceratium*

hirundinella, so these species are likely to become a minor component of the phytoplankton assemblage in the lake.

- The supply of oxygen to the sediment-water interface will facilitate decomposition of organic carbon in the sediment and water column, converting it to carbon dioxide.
- Organic carbon removed as carbon dioxide will reduce the sediment oxygen demand, allowing the lake to begin the recovery process.
- The release of carbon dioxide will generally reduce the lake pH, minimising the possibility of toxic ammonia formation.
- The full depth mixing and associated turbulence will not favour the development of cyanobacteria or *Ceratium hirundinella*, which require calm conditions, and they will be carried below their critical depth where they will die due to light limitation. However, other phytoplankton species that require turbulence to suspend them in the water column, such as diatoms and chlorophytes, are likely to become dominant and the lake algal assemblage will change from toxic blue greens and nuisance dinoflagellate species to non-toxic greens and diatoms. These species have critical depth limitations associated with light limitation so their abundance will also decrease due to full lake depth mixing. Consequently, phytoplankton biomass will decrease generally, leading to other lake water quality improvements:
  - The water column will clear (have lower mass of suspended particulate material), and the critical depth will increase because light can penetrate deeper into the lake.
  - The depth of the euphotic zone will also increase and the native aquatic macrophytes will increase in their depth range, providing safe refugia for small fish (bullies) and zooplankton from predatory trout and perch.
  - Higher oxygen levels at increasing depths should also benefit filter-feeding freshwater benthic organisms (e.g., freshwater mussels) as well as fish, which will have an increased habitat range encompassing the full depth of the lake.

The main disadvantage of destratifying the lake is that the lake temperature will rise. In Lake Hayes the temperature of the bottom water will increase, although it is unlikely to reach the mean temperature of 17-18°C at 2 m depth in summer, and the surface temperature is likely to fall as solar heating is dispersed through a greater volume of water and mixes with the cooler bottom water. This degree of lake warming is well below temperature that would be harmful to fish (maximum for trout is ~25°C) and therefore should not cause a problem.

The aeration could be run in conjunction with the augmentation of Mill Creek inflow providing a more efficient use of the aeration system in cooler years.

Aeration is not normally a long-term solution when restoring lakes and the destratification system would need to be run each year to manage the internal P load and the proliferation of algae. However, over a period of 5 to 10 years, with reduced inputs of organic matter and particulate P from the catchment, it is likely that sediment oxygen demand in the lake will reduce to the point where it is insufficient to cause hypolimnetic anoxia if the destratification system was turned off. Once this level of recovery has been reached, it would be a negative feed-back loop where the algal biomass in the lake reduces each year causing a lower sediment oxygen demand, which does not cause hypolimnetic anoxia and therefore does not release P to stimulate algal growth.

3.3.6 Recommendations

If an aeration system is to be considered for Lake Hayes, it should be a bubble curtain type using a sparge line diffuser across the bottom of the lake at the location and orientation indicated in Figure 3-13.

Dissolved oxygen and temperature profiles should be measured in the lake at weekly intervals from aeration turn on to March and thereafter at two weekly intervals until turn off, either manually of from a monitoring buoy. This will enable adaptive management of the amount of aeration required as the lake mixes and to maintain that mixing.

3.3.7 Secondary or amenity effects

Because the air compressor will be running continuously (24/7), there will be a potential for noise that will need to be blocked by the design of the compressor shed and the mounting block for the compressor. The bubble curtain produced by the aeration system will disturb the lake surface immediately above the sparge line. This will be very local to the aeration system and is not a health and safety issue for recreational users of the lake. However, it may disrupt the perfect reflections from the lake surface in the morning and evening under calm conditions. To accommodate tourists and photographers in general, the bubble curtain could be switched off for an hour at those times on calm days. Operated as the designed and with correct timing of turn on each year, the aeration system should have no adverse effect on water clarity and is likely to improve the clarity by preventing the development of algal proliferations.

3.4 Sediment capping

P removal using sediment capping, attacks the problem of the internal P load from the sediment driving the algal blooms, by sequestering the P into a non-bioavailable form. A description of the internal P load cycle is provided in the literature (James 2016; Spears et al. 2007) along with management strategies, which are discussed in detail in the lake sediment phosphorus release management—decision support and risk assessment framework (Hickey and Gibbs 2009).

The internal P load cycle is regulated by the dissolved oxygen concentration in the water. As the oxygen concentration reduces, the reduction-oxidation (REDOX) potentials of iron (Fe) and manganese (Mn) minerals in the sediment fall and, when the REDOX potential reaches a value around zero, these minerals transform from an insoluble oxidised ferric (Fe^{3+})/manganic (Mn^{3+}) form to a soluble reduced ferrous (Fe^{2+})/manganous (Mn^{2+}) form. Under well oxygenated conditions, ferric and manganic oxides sequester DRP from the water column and bind it as non-bioavailable P. Under anoxic conditions, the ferrous and manganous ions are in solution and there is no solid mineral matrix to hold the DRP, which is freely available for plant growth in the water column. The solubility of the metals changes with the REDOX potential and the sequestration/release process is reversible.

Another factor affecting the P binding to the iron oxides is pH. Under high pH (>9.2) the iron oxides form oxy-hydroxides which are soluble and the DRP bound to the insoluble iron oxides is released into the water column, even in well oxygenated water. High pH values are produced during photosynthesis by bicarbonate-adapted plants such as exotic aquatic macrophytes (pond weeds) and cyanobacteria. The ability of cyanobacteria to raise the pH and release DRP allows it to 'mine' P from the inshore sediments where it can grow to bloom proportions (Seitzinger 1991; Gao et al. 2012) when the epilimnion is otherwise depleted in P. It is not known whether *Ceratium hirundinella* can do the same.

Because the iron and manganese processes that bind DRP are reversible, these two metals are unsuitable as sediment capping agents. In contrast, aluminium (Al) compounds can irreversibly bind DRP from aerobic waters at normal pH ranges, i.e., the bound P is not released when the water column becomes anoxic and, if applied to anoxic water, it can sequester all of the DRP from that water. The rare earth metal, lanthanum (La), has similar P-binding characteristics as aluminium. Calcium can also sequester DRP in some conditions (high pH), but generally has a low affinity for P. All of these metals form the basis of commercially available sediment capping agents but behave differently in waters of different pH (Figure 3-16).

The P-binding efficacy of Al, La and calcium are strongly influenced by pH with the operating pH range for each metal increasing from Al < La < Ca (Figure 3-16).

3.4.1 Alum

The most common sediment capping agent is aluminium sulphate, which is supplied as the octadecahydrate ($Al_2(SO_4)_3.18H_2O$) in a 47% solution that has a pH of 3. The pH curve (Figure 3-16, Al) shows that alum is most efficient at binding DRP in the pH range of 4 to 6.5. Above that the P binding efficacy decreases, becoming <15% at pH 9. The major concern with using alum is that, below pH 5, it is in the trivalent ionic form Al^{3+}, which is highly toxic to aquatic biota. If there is insufficient DRP to bind to the aluminium applied, there will be free Al^{3+} ions in the water. To overcome this problem in soft water lakes, alum is applied with a buffer.

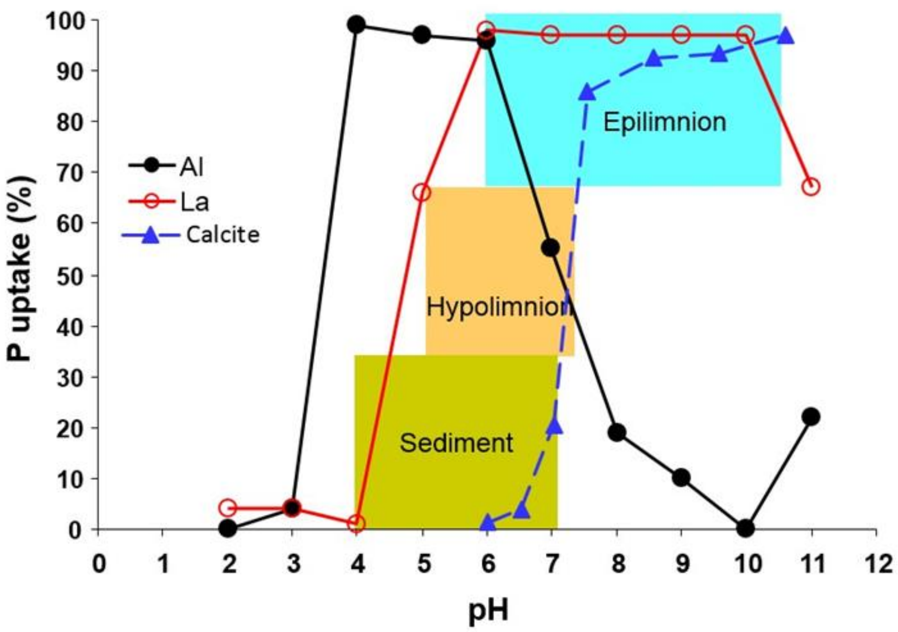


Figure 3-16: pH effects on DRP precipitation with Al and La salts, and adsorption by calcite. (Al and La data redrawn from Peterson et al. 1976, calcite data extracted from Olila & Reddy 1995). Coloured blocks indicate the typical pH ranges found in different parts of a lake. (Redrawn from Gibbs and Hickey 2012, Gibbs and Hickey 2017)).

Because it is a solution, alum can be sprayed on the lake surface or injected at the required depth into the lake water column. It can also be added to an inflow stream which could carry the alum floc in the density current formed when the stream enters the lake.

While the highest P-binding efficiency for alum is around pH 4 (Figure 3-16), the alum floc forms best at around pH 6.5 to 7. At lower pH the floc may not form leaving toxic trivalent Al^{3+} ions in the water if there is insufficient DRP to bind with. In soft water lakes the alum solution may require buffering to around pH 6.5 with sodium aluminate or sodium bicarbonate, to allow the alum floc to form.

Soft water lakes generally have low alkalinity of around 20 gEq (gram equivalents measured as g $CaCO_3$ m^{-3}). The nominal alkalinity value above which little or no additional buffer is required is 80 gEq. Lake Hayes has an alkalinity of around 150 gEq (Reid et al. 1999) and an average pH of around 7—7.5, and therefore should not require additional buffering over the natural buffering capacity of the lake water. When tested in Lake Hayes in 2010, the floc formed rapidly (Figure 3-17) and slowly settled though the water column over a period of about a day. The alum floc adsorbs DRP and aggregates particulate material, including zooplankton and algal cells, as it settles to the lake bed.

Figure 3-17: Example of alum floc formation in Lake Hayes in a trial mesocosm. [Photo: Ciska Overbeek, 22/02/2010].

3.4.2 Lanthanum

The use of La in the sediment capping material works differently to alum. This formulation, marketed as Phoslock®, requires an inert carrier for the La salt, Lanthanum chloride. To achieve this, during manufacture, it is physically blended with a fine bentonite clay, which is subsequently dried into fine granules. This granular material is typically applied as a surface treatment (Spears et al. 2013) that disperses through the water leaving a suspension of clay particles in the water column for up to 20 days, as the fine clay particles are slow to sink.

The highest P-binding efficiency for lanthanum is between pH 6 and pH 10 (Figure 3-15). Below and above this pH range, the P-binding efficiency is greatly reduced. Lanthanum binds rapidly with DRP in the water to form rhabdophane, a hydrous phosphate of La ($La(PO_4).H_2O$), which is insoluble and non-toxic. However, if there is insufficient DRP in the water column to bind all of the La, the residual La can form toxic trivalent ions (La^{3+}) ions in the water column below pH 6. This is comparable with the trivalent ions formed by aluminium at pH below 5.

3.4.3 Calcite

Calcium carbonate (calcite, $CaCO_3$) or calcium hydroxide (lime, $Ca(OH)_2$) can be added to lakes as phosphorus precipitants (Dittrich et al. 1997; - 2011). Calcite sorbs DRP, especially when the pH exceeds 9.0, and results in significant P removal from the water column. Phosphate adsorbs at the calcite surface, or binds inside a crystal during the formation of $CaCO_3$ precipitate when calcium hydroxide is applied (Kleiner, 1988; House, 1990). Various calcite forms have also been reported for their potential use as active barriers for sediment capping to reduce phosphorus release from sediments (Hart et al. 2003).

Unlike alum and lanthanum, where P-binding occurs at circum-neutral to moderately high pH, calcium binding of P only occurs at high pH. At high pH, and concentrations of Ca^{2+} and soluble P, hydroxyapatite is formed.

$$10\ CaCO_3 + 6\ HPO_4^{2-} + 2\ H_2O = Ca_{10}(PO_4)_6(OH)_2 + 10\ HCO_3^-$$

Hydroxyapatite has its lowest solubility at pH >9.5 and binds phosphorus strongly at high pH (Cooke et al. 2005). The major drawback for using calcite is the amount required. As an active barrier, it needs to be applied in a layer about 5 cm thick whereas alum and lanthanum capping layers are typically <1-2 mm thick.

3.4.4 Other capping agents

Other products that could be used for P-sequestration if applied as a sediment cap include:

- **Aqual P™**, which is a modified zeolite used to carry alum or poly aluminium chloride (PAC) to the bed of a lake. Because the active ingredient is aluminium, the properties are similar to alum except that it doesn't release DRP under anoxic conditions (Gibbs et al. 2011). The main advantages are that it is in a granular form for lake treatment and the zeolite used in its manufacture adsorbs ammonium from the water column so the product reduces both P and N in the one treatment. The disadvantages are that it is more expensive than alum and it has little capacity as a flocculent so does not clear the water column to the same degree as alum, which is a flocculent.

  However, because it settles rapidly, it can be targeted to specific areas such as the hypolimnion with little drift into the epilimnion or littoral zones. It is designed to block the release of DRP from the sediment under anoxic conditions. For this reason, Aqual-P™ is applied at the end of the mixed period when all DRP has been sequestered into the sediment and before thermal stratification and reduced DO concentrations begin to release DRP into the water column.

  Where cyanobacteria may be a nuisance inshore, an application of Aqual-P™ in the littoral zone may prevent the bloom developing by blocking the high pH release from the cyanobacteria. [To start the pH driven release the cyanobacteria must grow using the natural diffusion of DRP across the sediment-water interface. As this increases it becomes a positive feedback loop with the pH rising thereby releasing more DRP and allowing more cyanobacteria growth (Seitzinger 1991)].

- **Allophane**, is a natural volcanic rock, rich in both iron and aluminium. It has similar P-binding characteristics as alum and doesn't release DRP under anoxic conditions (Gibbs et al. 2011). The major problem for the use of allophane is finding a source which is not already contaminated with P and has a useful amount of P binding capacity left.

Both of these capping agents work well when applied as a thin layer 2 mm to 5 mm thick of < 1 mm grainsize material. The disadvantage is the amount required to put a 5-mm layer across the whole lake. For Lake Hayes, with a surface area of 2.76 km^2, the required alum application would be equivalent to a 0.4 mm layer across the lake bed.

3.4.5 What is the best medium to use?

Alum is the most cost effective permitted sediment capping agent available in New Zealand. This is because it is: 1) easy to transport (in 220 litre barrels or tanker trucks); 2) readily available around New Zealand for drinking water supply treatment; and 3) easy to apply, either by surface spray (Figure 3-18), subsurface injection, or drip feed into streams. The other capping agents cost more by up to a factor of 5 and, as solid material, application costs are more than for the liquid.

The amount of alum required to treat Lake Hayes as a one-off dose was estimated in a 2012 report prepared by John Quinn, Max Gibbs and Chris Hickey (NIWA) and reproduced in the Schallenberg and Schallenberg (2017) report. In 2012, the DRP concentration in Lake Hayes hypolimnion was around 300 mg m^{-3} (M. Schallenberg, unpublished data) and it was estimated that it would require 535,343 kg of alum to bind all the P released from the lake sediments. Alum costs about $1000 per tonne, giving a capping agent cost of NZ$535,343.

Since 2012, the mean DRP concentration in the hypolimnion of Lake Hayes during the stratified period has reduced from 300 mg m^{-3} to about 50 mg m^{-3} in 2016 (ORC monitoring data). Without more detailed data, a best estimate of the amount of alum required in 2016 is from a pro rata scaling from the 2012 estimate which gives the total amount as about 90,000 kg at a cost of NZ$90,000. These numbers need to be checked using the latest nutrient data from Lake Hayes and the current price of alum from Ixom (formerly Orica Chemicals).

3.4.6 Application method

Alum has been applied to lakes overseas using sun-surface injectors. While this technique has been used to apply a slurry of Aqual-P™, it has not been used for alum. Alum has been applied by surface spraying from an air boat (Figure 3-18) (Paul et al. 2008).

Figure 3-18: An air boat was used to apply alum to Lake Okaro near Lake Rotorua.. [Photo by Max Gibbs].

Lake Okaro with a surface area of 0.31 km^2 is much smaller than Lake Hayes with a surface area of 2.76 km^2 and took two days to complete the alum application. On a pro rata basis, it is likely to take about 18 days to spray Lake Hayes. Because the alum is applied at the surface and must pass through the complex circulation currents in the lake water column, it is not certain where the alum would settle in the lake over the 18 day period. Using the estimated 5 cm s^{-1} flow rate (section 1.1), the alum would travel about 15.5 km from where it was applied, i.e., about 3 times around the lake.

In lake Rotorua, alum has been drip fed at a rate of about 1 g m^{-3} into two inflow streams, which have naturally high DRP concentrations, to reduce the external P-load on that lake. A similar approach could be used in Lake Hayes by drip feeding the alum into the Mill Creek inflow and allowing the density current and lake circulation flows (Figure 3-9) to disperse the alum into the hypolimnion. This would require a storage tank and a metering pump to be installed beside Mill Creek but eliminates the dosing of the lake surface where the alum could adversely affect zooplankton populations.

The advantage of this technique is that the dose used can be adaptively managed by adjusting the drip rate up or down for optimum DRP sequestration as the DRP concentration changes in response to DO concentration changes in the hypolimnion. The adaptive management process could be automated by linking it with the temperature data from the lake and Mill Creek, so that alum dosing only occurred when the density current was plunging to the lower water column. This application process would minimise the amount of alum applied to the lake and would allow the dosing cost to be spread over time rather than have a single lump-sum cost.

This method of application will still cap the sediment and the capping effect will last as long as it takes for fresh sediment inputs from the catchment to bury the alum layer in the lake sediments. This may range from 5 to 10 years, given that the DRP concentrations are naturally declining, implying a reduction in sediment from the catchment.

3.4.7 Summary

Sediment capping is a method of preventing the release of P from the lake sediments. The most cost-effective product is alum either as a liquid from Ixom (previously Orica Chemicals) or bound to a zeolite carrier as a fine granular product available as 'Aqual-P™' from Blue Pacific Minerals, Tokoroa.

- Applied as single dose of the granular product in winter, it forms a pre-emptive sediment cap that will intercept the DRP diffusing out of the sediments as the DO concentrations reduce during the onset of thermal stratification in spring.
- Applied as a single dose of the liquid product via a surface spray or subsurface injection at the end of summer when the lake is strongly thermally stratified and all of the DRP has been released into the hypolimnion but before the thermocline begins to sink, it will sequester all of the DRP and irreversibly bind the P, which will settle onto the sediment surface.
- Applied as a liquid via a drip feed into the Mill Creek inflow, with automated adaptive management, it will be transported to the hypolimnion in the natural density current only when it will be able to sequester the DRP, thereby reducing the amount of product applied and focusing the product where it is needed. This should reduce wastage and therefore cost.
- Alum can be applied to Lake Hayes water without the need for an additional buffer.
- Once the DRP is bound to the alum and has settled to the sediment surface, it cannot be released by the hypolimnion becoming anoxic or the sediment being anoxic when it is subsequently buried.
- The longevity of the treatment is determined by the amount of new P-enriched sediment carried into the lake from the catchment or sedimenting algal biomass that grew in the lake. This will bury the original alum-bound P, which will remain in the sediment as a thin layer of non-bioavailable P. This layer will not be affected by diagenesis but may continue to sequester DRP released by diagenetic processes deeper in the sediment.
- Whichever method dosing is used, alum treatment is compatible with the augmentation of Mill Creek with irrigation water from the Arrow River, which has low DRP concentrations.

3.4.8 Recommendations

An Otago Regional Council representative should visit Bay of Plenty Regional Council to talk about the dosing systems they have put in place and see the dosing stations.

If cyanobacteria are beginning to grow in the edge water from accumulation as wind-drifts, a targeted dose of either Aqual-P™ or Phoslock® granules in the littoral zone is likely to prevent these wind-drift accumulations becoming substantial blooms by blocking their pH driving mechanism for the release of bioavailable P and causing local P-limitation.

3.4.9 Secondary or amenity effects

If alum is used for P removal and sediment capping and is applied to Mill Creek by drip feed, there will need to be a stream-bank installation similar to those used by Bay of Plenty Regional Council on the Utahina and Puarenga streams at Rotorua or on Soda Spring stream at Lake Rotoehu. The latter installation was housed in a small shed, behind a wooden fence and has minimal visual impact. The periodic delivery of bulk alum to the storage tank would be required.

3.5 Other mitigation techniques

There are several other mitigation strategies that could improve Lake Hayes water quality.

3.5.1 Nanobubble technology

This very recent technology which is said to take the hypolimnetic oxygenation process using Speece cones (Speece et al. 1973) to the next level. At present there have been a number of product documents and videos presentations but no peer reviewed publications, which give the information required to assess the efficacy of this product. The product brochures say the nanobubbles are produced through the walls of a special ceramic cone over which water flows. This water carries the nanobubbles into the lake where they sink the bottom. This concept may work in rivers and shallow lakes but may not work where the internal lake currents can be substantial, as in Lake Hayes. If it did work as described and the nanobubbles were entrained into the density current, this could be a useful tool.

The main issue with the nanobubble technology is that it relies on providing all of the oxygen to sequester the DRP in the hypolimnion. Based on the calculations of HOD in December 2016 (average 60 mg m^{-3} d^{-1}) it would need to provide >1.57 t O_2 d^{-1} (Section 3.2) just to hold the DO concentration at the level it was when the nanobubble technology was switched on. That oxygen input would need to be much larger if the nanobubble technology was going to re-oxygenate the lake. The other issue is that the special ceramic cone used to generate the nanobubbles is produced in only one facility in Japan. I believe that no one else has been able to reproduce this cone, without which the nanobubble machine does not work. This is a high risk situation if this vital part fails.

The supply agents advise that, if the power fails, water can slowly seep into the ceramic cone causing a blockage and the nanobubble machine will not work when the power is restored. This situation can be recovered by removing the cone from the water and allowing it dry for 24-48 hours.

A quotation for using the nanobubble technology in Lake Hayes was obtained (Appendix A). The design supplied included seven nanobubble 'engines' spaced around the lake at an installation cost of $4.7 million.

3.5.2 Hypolimnetic withdrawal

Hypolimnetic withdrawal is an in-lake restoration technique based on the selective discharge of bottom water to enhance the removal of nutrients and dissolved metals that accumulate when the hypolimnion becomes anoxic. Comparison of water quality variables before and during treatment in about 40 European and 8 North American lakes indicates that hypolimnetic withdrawal is an efficient restoration technique in stratified lakes (Nürnberg 2007). This technique was successfully used in New Zealand in the Upper Huia Reservoir for Watercare Services Ltd before aeration systems were installed (Spigel and Ogilvie 1985; Gibbs and Howard-Williams 2017).

Nürnberg (2007) reported that water quality improvement was apparent in decreased summer average epilimnetic P and chlorophyll-*a* concentrations, increased water clarity (Secchi disk depth), decreased hypolimnetic P concentration and anoxia. In particular, summer average P decreases were significantly correlated with annual water volumes and P masses withdrawn per lake area, indicating the importance of hydrology and timing of the treatment. Lake size is normally not a problem and hypolimnetic withdrawal has been used on lakes from 1.5 to. 1500 ha, spanning a 1000-fold area and a 2300-fold volume.

While almost all hypolimnetic withdrawal applications are conducted in lakes or reservoirs with outlets, the technique has also been applied to a relatively large seepage lake with no outlet by using a siphon (Lathrop et al. 2005). The hypolimnetic siphon approach is passive and can be designed for almost any natural lake where there is no bottom water offtake valve as found in reservoirs. In such lakes, the hypolimnetic siphon would draw the outflow water from the bottom of the lake rather than the surface. The disadvantage of this is that the bottom water would be rich in DRP, dissolved metals (primarily iron and manganese) and potentially ecotoxic high ammonia and sulphide concentrations. To overcome this problem, the hypolimnetic water would need to pass through an oxygenation process, such as a gravel cascade to agitate the water, followed by a settling pond/wetland to remove the particulates.

The design of the hypolimnetic siphon is conceptually simple, comprising a weir through which the draw pipe from the bottom of lake passes and a downstream wet land or aeration station through which the water flows before reaching the Kawarau River.

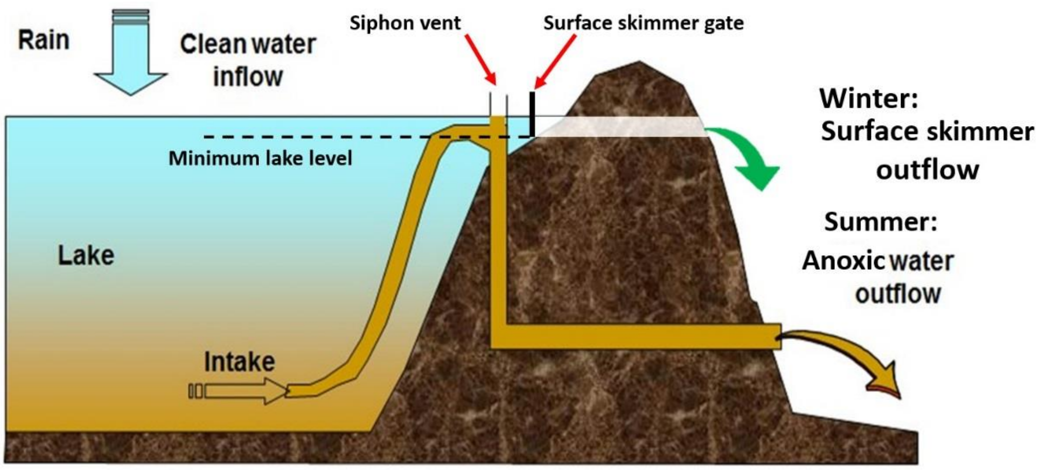


Figure 3-19: Stylized hypolimnetic siphon design through a weir, with a change-over gate to allow surface skimming if a surface algal bloom develops. (Schematic diagram drawn by Max Gibbs).

In summer, anoxic water drawn from the bottom of the lake flows out under the hydraulic pressure of the overlying lake water. The vent tube could be set at the height of the maximum required lake level so that the lake is always within a specified water level range. Note that in summer, the outflow is less than in winter. Because the lake naturally drains into the Kawarau River, the anoxic water would rapidly oxygenate and any DRP would be sequestered by the iron in that river sediment.

In winter when the lake is mixed, or during an algal bloom, the bottom water draw tube to the siphon would be closed and the skimmer gate opened to allow algal proliferations to be skimmed off the lake surface. This is the natural flow path and the siphon would be left in this configuration to accommodate flood events in winter. The surface skimmed material input to the Kawarau River would be unchanged from what it currently is as that is the natural connection from the lake.

Apart from the initial construction cost, this mitigation measure would have very low operational costs and would gradually mine the lake of DRP. If considered practical, a feasibility study would be required.

3.5.3 Biomanipulation

Biomanipulation is a technique which relies on a set of conditions prevailing to enable a specified end result. This approach is covered in detail in the Schallenberg and Schallenberg (2017) report.

Biomanipulation has been tried over many years (Shapiro and Wright 1984; Shapiro 1990) and successfully used in many shallow lakes in Europe and the USA (e.g., Carpenter et al. 1987; Dawidowicz 1990; Irvine et al. 1990; Jeppesen et al. 1990; Moss 1990; Timms and Moss 1984). In New Zealand, Howard-Williams (1981) found that the native aquatic macrophyte, *Potamogeton pectinatus*, was able to remove dissolved nitrogen and phosphorus compounds from lake water.

Biomanipulation can be a successful alternative to physical and chemical treatments to accelerate lake recovery often delayed because of persistently high rates of internal loading. Therefore, a combination of load reduction and in-lake restoration measures including biomanipulation is likely to improve water quality greatly (Kasprzak et al. 2002).

However, while there are success stories, there are also occasions when this technique does not work and cladocera cannot control algal blooms (Gliwicz 1990). Biomanipulation tends to work well in shallow lakes since organisms such as zooplankton are not spatially separated by depth. In deeper lakes, anoxia in the hypolimnion reduces the zooplankton population and the algal cells can fall out of the habitable water column resulting in further reductions in their population due to food limitation.

No compelling examples could be found of successful restoration of lakes as deep as Lake Hayes but there are numerous successful restorations of small shallow lakes and ponds.

3.5.4 Catchment management

Catchment management is undoubtedly the ultimate solution to the long-term restoration of Lake Hayes. With the development of appropriate farm management plans and restrictions placed on the application of superphosphate fertiliser in the catchment, the external nutrient loads, both N and P, on the lake will reduce to the point where the lake no longer develops an anoxic hypolimnion and the internal P load is reduced to near zero.

Fine soil has high concentrations of P and therefore, surface runoff will carry the P from diffuse sources in different sub catchments (Caruso 2000b). This information can be used to target the sub catchments with the highest P output. Mitigation measures could include the installation of detention bunds and wetlands to trap the fine sediment before it gets into the surface waters of a stream.

Catchment management strategies will take time (many years) to become apparent but the long-term effects will be a stable recovery of the lake not reliant on a cold year for enhanced underflow (augmentation), continuous aeration each summer or the addition of a sediment capping agent to reduce the DRP from the internal load. The internal load will no longer be replenished from the catchment and will gradually become buried or mined from the lake by natural in-lake processes.

Notwithstanding the importance of long-term catchment management given current nutrient and sediment loads entering Lake Hayes from Mill Creek, it is appropriate to implement short-term strategies that will reduce the magnitude of the water quality problems while the catchment management strategies become established. These can be phased out when the catchment management strategies are established.

4 Summary

This report has reviewed the Schallenberg and Schallenberg (2017) report and found a section that requires further work in the form of calculating the likely effects of using the excess irrigation water from the Arrow River to augment the flow in the Mill Creek inflow to Lake Hayes. Otherwise the Schallenberg and Schallenberg (2017) report provides a comprehensive summary of the issues and develops a workable set of management strategies with timelines for implementation and expected responses, that should be considered further.

In this report three defined options for water quality improvement have been elaborated in detail making use of the most recent data from ORC and the previously unpublished data from NIWA and University of Otago one-off studies to aide interpretation. Other potential mitigation strategies have also been considered and together with the importance of long-term catchment management given current nutrient and sediment loads entering Lake Hayes from Mill Creek.

The following table lists some of the overview assessments of these mitigation measures:

Table 4-1: Assessment of mitigation strategies considered for Lake Hayes in this report. Based on assessments in Section 3 in that order.

Strategy	Pros/Cons	Issues	Risk	Compatibility	Cost
Augmentation (Section 3.2). (Medium to long-term strategy).	**Pro**: Can potentially transfer sufficient oxygen to hypolimnion to prevent deoxygenation when it works but fragile. **Con**: Dependent on climate patterns.	Relies on cold temperatures in spring (Mill Creek and Arrow River irrigation water) to enhance underflow.	A warm dry spring could result in negligible oxygen transfer and strong bottom water anoxia (with subsequent P-release from sediments).	Augmentation of a natural process. Will complement other remediation processes.	Unknown.
Destratification (Section 3.3). (Short-term strategy – run on annual basis).	**Pro**: Will eliminate the internal P load. Disrupts *Ceratium* growth pattern. International precedents as effective remediation approach. **Con**: Will raise the temperature of the bottom water. Need for regular or continuous monitoring for operational management. Will probably need to be maintained for 5–10 years	Timing of turn-on is critical – must be before thermal stratification develops in spring.	Turn on while there is stratification and hypolimnetic deoxygenation and algal proliferation in the lake could trigger a major algal bloom; Failure of the compressor or power supply.	Will enhance augmentation density currents; Not compatible with sediment capping.	\$250K to \$350K capital cost plus running cost.

Strategy	Pros/Cons	Issues	Risk	Compatibility	Cost
Sediment capping (Section 3.4). (Medium to long-term strategy as a single dose. Short-term strategy run each year, if run as a drip feed.).	**Pro**: Alum or Aqual-P can lock P in the sediments and reset the internal P load to zero. International precedence as effective remediation approach for P-limited lakes. **Con**: Can release toxic trivalent Al^{3+} if the pH is <5. Other products were not recommended **Lanthanum**; **Calcite.**	Addition of a non-natural product to a lake may be seen as being not culturally acceptable. This could prevent the use of this technology to restore the lake.	Little risk in Lake Hayes. Could be applied as a drip feed into Mill Creek to keep the alum additions to a minimum.	Compatible with augmentation; Not compatible with destratification.	$90K to $550K as single dose. longevity 5 to 10 years. Drip feed option costs lower but ongoing.
Nanobubbles (Section 3.5.1) (Short-term strategy – run on annual basis).	**Pro:** Theoretically should work well to oxygenate hypolimnion without de-stratifying lake. **Con**: No documentation to support claims of efficacy. No international precedence. Very high capital cost.	Relies on several proprietary devices to oxygenate the hypolimnion of the lake.	Only one supplier of key ceramic cone element.	Could be used with sediment capping Not compatible with destratification.	$4.7 million capital plus running costs.
Hypolimnetic siphon (Section 3.5.2). (Medium to long-term strategy).	**Pro:** Can be used to reduce internal P load and skim surface algae. **Con:** Need for oxygenation and settling of discharge. Possible downstream effects in Kawarau River. Not readily automated.	Anoxic bottom water will be discharged into Kawarau River.	Design must include human exclusion from the siphon vent.	Not compatible with destratification.	Unknown probably <$100K.
Biomanipulation (Section 3.5.3). (Medium to long-term strategy).	**Pro:** Attacks the *Ceratium* bloom. Many successful studies in shallow lakes. **Con:** No international precedents for use in deep lakes. Likely high risk and low	No effect on internal P load to stop the bloom continuing.	Requires removal of planktievorous fish by enhancing piscivorous fish populations.	Not compatible with sediment capping as a single dose.	Unknown.

Strategy	Pros/Cons	Issues	Risk	Compatibility	Cost
	effectiveness as single management strategy.				
Catchment management (Section 3.5.4). (Long-term management strategy).	**Pro:** Will work given time (>10 years) **Con**: May have high costs for farmers. Probably requires other remediation measures to manage internal lake loads in the short-term.	Slow to see results but when they are achieved it will be a permanent fix.	Cyclonic storms that could destroy the structure put in place to reduce sediment loads etc.	Compatible with all short-term measures.	Unknown – but will cost lots for some options, and less for others.

5 Recommendations

To achieve rapid improvement in the water quality of Lake Hayes requires both short-term and long-term mitigation strategies.

A critical evaluation and multi-criteria scoring evaluation (including weighting of criteria) should be undertaken on the range of potential remediation technologies/approaches which are suitable for Lake Hayes. This will assist in ranking of the suitability of the management options relative to the short- and long-term goals for remediation (e.g., oligotrophic or mesotrophic state) – together with other community asperations. It should be noted that the implementation of multiple approaches is likely to achieve the best management outcomes.

Additional monitoring information may be beneficial in contributing to the technical information to undertake this assessment. The following list is in the order of least intervention in the lake and includes:

- **Catchment management:** Develop and implement catchment strategies. These will take time to become effective, but they will succeed.
- **Augmentation:** Measure the temperatures in the irrigation water and Mill Creek at 15-minute intervals, continuously.
  - Install a lake monitoring buoy in the lake with telemetered output of temperature and DO concentrations at multiple depths for use with automated management systems.
  - Deploy a pair of ADCP current meters on the lake bed, on opposite sides of the lake, to assess both vertical and horizontal in-lake currents. These are important for determining how the lake mixes and where the Mill Creek water inserts in the lake when it forms a density current.
  - After checking the irrigation water and Mill Creek temperature differentials, implement the irrigation water augmentation of Mill Creek. This connection should be managed to only augment when the irrigation water is colder than Mill Creek.
- **Sediment capping** (using P-binding)**:** If permitted, implement drip feed dosing of Mill Creek with alum, using automated feedback from the DO and temperature sensors on the lake buoy and the Mill Creek temperature data for adaptive management of the metering system. Dosing should only be used when the DO data indicates the P is being released from the sediment.
- **Destratification:** If alum dosing is not permitted, install a destratification system across the middle of the lake using the recommended turn-on protocols.
- **General monitoring:** Maintain the current SOE and inflow stream monitoring program with full profiles at two-weekly intervals (summer- January, February, March and April, inclusive) and at monthly intervals (winter- May to December, inclusive) until a telemetered lake buoy can be installed to provide these data at higher frequency.

The monitoring data should be used to drive the development of a DYRESM-CAEDYM model of the lake, which can be used to test these and other restoration measures.

6 Considered opinion

In my opinion, the most effective short-term mitigation method in Lake Hayes would be destratification. This is a proven technique that can be designed to suit the lake and can be adaptively managed as the lake condition improves. It adds nothing to the lake except air to keep the lake mixed. Mixing will prevent cyanobacteria and possibly *Ceratium* blooms developing.

Destratification will not necessarily eliminate algal blooms immediately but it can over time. The initial response will be a shift in the algal species assemblage to non-toxic species and species which will be more susceptible to light limitation from deep mixing below critical depth[7]. At 30 m deep, Lake Hayes has sufficient depth for the critical depth to be effective. Critical depth would likely be around 15 m. If the average algal assemblage spends more time below the critical depth than above, the algal biomass in the lake will decrease and the load of organic carbon reaching the lake bed will decrease. This can become a negative feed-back loop for algal growth and, over time, the biomass produced each year will diminish as the amount of organic matter driving bottom water oxygen depletion reduces annually.

The other effect of destratification is the supply of oxygen to the bottom of the lake and the sediments. The entrained oxygen from the surface in the return flow water will reduce the sediment oxygen demand (SOD) and the decomposing carbon will be released as CO_2 gas. As the SOD decreases, the HOD rate will decrease and there should become a time when, without the destratifier operating, thermal stratification will not result in the development of an anoxic hypolimnion with the concomitant release of DRP from the sediment.

This latter condition is reliant on the catchment management strategies reducing the external carbon, nutrient and suspended solids loads to the lake.

7 Acknowledgements

I wish to thank Otago Regional Council for supplying the extensive data set for Lake Hayes, and Marc Schallenberg (University of Otago) and Chris Hickey (NIWA) for valuable discussions, both historic and recent, about Lake Hayes.

[7] 'Critical Depth' is defined as a hypothesized surface mixing depth at which phytoplankton growth is precisely matched by losses of phytoplankton biomass within this depth interval. If phytoplankton spend more time below the critical depth than above, the biomass will decrease.

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Appendix A Nanobubble installation quotation for Lake Hayes

PACIFIC DRIVELINE LTD
MARINE & INDUSTRIAL DRIVELINE SPECIALISTS

Quotation No: 241

PACIFIC DRIVELINE LIMITED
PO BOX 97869
MANUKAU CITY
MANUKAU 2241, AUCKLAND
Tel: (09) 262-3241
Fax: (09) 262-3240
Email: pacdrive@henleygroup.co.nz
GST: 79-834-009

Attention: Max Gibbs **Company #** P66 **Quote Date** 22.01.2018

To Account:
NIWA
LAKE HAYES

Delivery:
NIWA
LAKE HAYES

Description	Quote Qty	Unit Price	Line Total
NBG-150A2GR NANOBUBBLE GENERATOR SYSTEM: GRID SUPPLY *PDL Scope of supply: Turn key operation. Grid Power Supply.* *1 x Custom Camo 40ft container system* *2 x 150A nanobubble generators,* *1 x Centrifugal Pump System,* *1 x Oxygen Generator System,* *80m suction and delivery pipes, floats, weights, screens, solar light.*	5.00	$537,842.00	$2,961,760.00
NBG-150A2GE NANOBUBBLE GENERATOR SYSTEM: GEN SUPPLY *PDL Scope of supply: Turn key operation. Generator Power Supply.* *1 x Custom Camo 40ft container system* *2 x 150A nanobubble generators,* *1 x Centrifugal Pump System,* *1 x Oxygen Generator System,* *80m suction and delivery pipes, floats, weights, screens, solar light.*	2.00	$592,352.00	$1,184,704.00

WARRANTY / NOTES

Nanobubble Generator Systems come as a complete wired and plumbed integrated package, including; security alarms & cameras, timers, monitored 24/7, installation of container and associated piping on site.

Customer Scope of supply: Road access to agreed system location. Grid power to container as required. Fuel Supply for generator powered systems (2).

Delivery: 24 weeks from receipt of 50% deposit, balance due 30 days following install of each unit model.

Warranty: 18 months from install sign-off and system operation.

Periodic Servicing / Maintenance of system and associated equipment: Price TBA.

Sub Total	$4,146,464.00
G.S.T	$621,969.60
Total	$4,768,433.60

Terms and Conditions: Page 1 of 1
Bank - ANZ Albany, 010277 0047338 00
INDENTED ITEMS ARE NOT RETURNABLE
Customer is liable for collection costs for outstanding accounts.
All above subject to our terms and conditions, a copy of which is available on our website.

NAME:________ **SIGNATURE:**________ **DATE:**________

22.01.2018 10:13:36 AM www.pacificdriveline.co.nz

Lake Hayes Restoration and Monitoring Plan

Photo: David Hamilton

Prepared for the Friend of Lake Hayes Society Inc.

by Marc Schallenberg (PhD) and Lena Schallenberg (BSc)

Hydrosphere Research Ltd

58 Gladstone Rd., Dunedin

17 May 2017

Executive summary

Lake Hayes is a highly-valued lake that has suffered from algal blooms for many decades. Since 2006, the blooms have worsened and lake health and fishing has deteriorated markedly. This report provides relevant historical background to the Lake Hayes water quality story, analyses water quality and ecological data and information, and proposes a restoration strategy for lake recovery.

The worsening of algal blooms since 2006 has occurred when both external and internal nutrient loads to the lake had been either declining or stabilising. So, the reason for the blooms was not related to increased nutrient inputs to the water column. Rather, a change in the dominant algae species occurred and the new nuisance alga, a dinoflagellate called *Ceratium hirundinella*, possesses some particular adaptations that allow it to supplement its nutrition in unusual ways. So, the development of *Ceratium* blooms appears to have been facilitated by a decline in nutrient loads, which gave it a competitive advantage over other algae. The scientific literature contains reports of *Ceratium* and other dinoflagellates sometimes becoming dominant during a recovery from high nutrient loads.

While *Ceratium* seemed to have had a hold on Lake Hayes, two recent summers (2009/10 and 2016/17) have seen the lake exhibit extremely clear waters with very little algae biomass. This could indicate that the lake is approaching a recovery tipping point. How long it will take the lake to achieve consistently high water clarity is unknown. However, observations indicate that high densities of zooplankton (grazers of algae) in summer have been associated with high summer water clarity, suggesting that dynamics of the pelagic food web may play an important role in the lake's recovery.

This report evaluates the potential for many various restoration activities to accelerate the recovery of the lake. Four of these strategies have been selected to be the most promising and cost-effective. These are: (1) food web biomanipulation, (2) enhanced flushing by using surplus irrigation water from the Arrow River, (3) alum dosing to flocculate and bind phosphorus in the lake bed, and (4) a focus on land use activities in the catchment to further reduce nutrient and sediment losses from land to water. These strategies were scrutinised using the available data and some costing were determined. This allowed the development of a restoration strategy proposing the most promising strategies to use, potential timelines to achieve implementation, and suggesting a range of restoration targets by which to measure success. This report also discusses lake monitoring options to help track recovery of the lake and demonstrate effectiveness and cost-effectiveness of the strategies.

Contents

Scope

Lake Hayes is a treasured asset to Tangata Whenua, locals and tourists alike. Located within the Arrow Basin between Queenstown and Arrowtown, the lake is highly visible by road and frequented by lake users and campers year-round. Since the late 1960's the lake has been subject to severe algal blooms, initially as a result of increased nutrients entering through Mill Creek and from springs at the northern end of the lake, which are high in nitrate concentration (Bayer & Schallenberg 2009). From the 1960's through to 2010, lake water quality had steadily decreased to a eutrophic state with blooms of blue-green algae/cyanobacteria, green algae and dinoflagellates occurring in stages throughout that time. Currently the lake suffers from severe blooms of the dinoflagellate alga *Ceratium hirundinella* blooms almost yearly, prompting the Friends of Lake Hayes Society to investigate and instigate restoration measures with the aim of returning the lake to a healthier state.

Multiple reports have described the main issues affecting Lake Hayes as well as a range of potential lake restoration options for consideration. However, with the changeable nature of the lake's algal blooms and the fast-increasing trend towards further eutrophication, this restoration plan was commissioned by the Friends of Lake Hayes Society. The restoration plan aims to describe the current state of the lake, summarise the major issues affecting water quality, recommend and discuss realistic restoration options, provide a restoration strategy with timelines, and recommend useful monitoring strategies for monitoring lake status and recovery to a stable water quality state which aligns with community, stakeholder and tourism values.

1 Background

1.1 Community values, uses and importance

Lake Hayes has been described as one of the most scenically attractive landscapes of its type in New Zealand and it holds significant importance for recreation and tourism (Cromarty & Scott 1995). Surrounding the lake is a vegetated margin with patches of wetland areas supporting a high diversity of endemic, rare or threatened fauna including the koaro, longfin eel and breeding birds such as paradise ducks, New Zealand shovellers, marsh grebes, Australian coots and great crested grebes (Cromarty & Scott 1995). A popular shared-use trail navigates these margins. Parts of the lake surroundings have been granted recreational and wildlife management reserve status as well as belonging to a wider wildlife refuge area covering 354 ha, including the lakebed. The lake and its immediate surrounds are used by locals and tourists alike for a range of recreational activities including rowing, boating, fishing, swimming, running, biking, walking and picnicking.

The Lake is culturally important for its food gathering which has led to the lakes recognition as a treasured resource (Waahitaoka) (ORC 2009) and the Lake Hayes Management Strategy states that "the conservation of the Lake Hayes resource is of regional and national importance both economically, recreationally and for its intrinsic and scenic values" (ORC 1995).

1.2 Historical catchment development

Extending far to the north-west of the lake along Mill Creek, the Lake Hayes catchment (Fig. 1) was likely forested with kahikatea prior to 1740 and a large wetland extended through the western reaches of Mill Creek in the mid-catchment. A number of smaller wetland swamps also existed in the catchment to the west and north of the lake as well as extensive riverine marshes on the banks of Mill Creek and smaller streams (Robertson 1988).

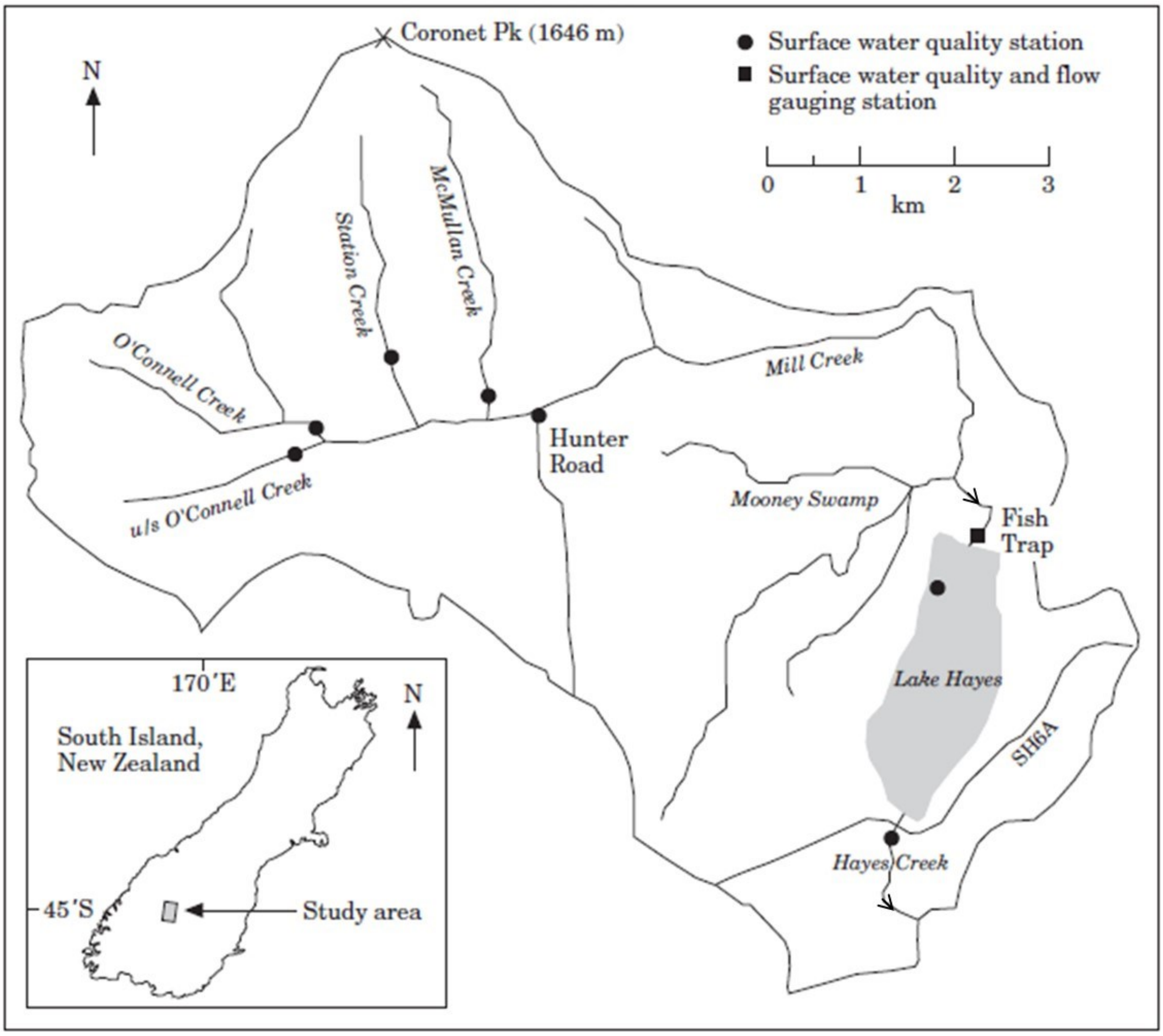

Figure 1. The Lake Hayes catchment (adapted from Caruso 2001)

Deforestation in the catchment began around 1740 when the Kahikatea forest was largely destroyed by fire, and further deforestation likely occurred through the late 1800's as miners and settlers harvested trees for shelter and firewood (Robertson 1988). After the deforestation of the Kahikatea forest, the Lake Hayes catchment comprised mostly native tussock grassland in the high country with swamps and wetlands dominating the lowland areas including the Arrow Basin (Robertson 1988). Mill Creek is the major tributary in the catchment, fed by a number of high country streams including O'Connell Creek, Station Creek and McMullan Creek, sediment and nutrients from which were once immobilised in the wetland before continuing on down Mill Creek towards Lake Hayes. Smaller wetland areas also existed adjacent to the mid-reaches of Mill Creek including Mooney Swamp, which acted as wildlife habitat, flood mitigation and a sediment and nutrient sink. Relatively low concentrations of nutrients are expected to have been transported by Mill Creek and its tributaries through the early 1900's.

The early-to-mid 1900's saw land converted to sheep pasture and further conversions from sheep to cattle and dairy. Superphosphate fertilizer was introduced in the 1950's allowing cattle and dairy to intensify in the catchment and aerial topdressing was common on farms, particularly around the lake, within which a topdressing plane was lost in 1953 (Robertson 1988). From approximately 1912-1955 a local cheese factory operated to the north of the lake where it released whey effluent with a phosphorus (P) load of approximately 1000kg/yr (roughly equivalent to the annual effluent of 2000 cows) directly into Mill Creek (Robertson 1988). Remaining whey was fed to pigs which also contributed further effluent to the creek.

The Otago Catchment Board began major drainage and channeling works in 1961-62, which saw wetlands drained and artificial channelization through what was soon to be high producing exotic grasslands. The initial channel and drainage works in 1961 cut through 80-120ha of wetland in the upper catchment, bringing a significant amount of sediment through Mill Creek and into Lake Hayes (Robertson 1988). Locals recorded the first sighting of brown water flowing into the lake in 1961 which continued sporadically throughout the remainder of the drainage and channelisation works over the next few decades (Robertson 1988). This significant land conversion and sediment immobilization has been touted as a major turning point for lake ecosystem health.

With the conversion of wetlands into pastoral grasslands, the water quality buffering capacity of the catchment decreased. It is estimated that 80% of the P load in Mill Creek came from the tributaries above the large wetland (Robertson 1988) and when in its natural wetland state, sediment and sediment-bound nutrients such as P were trapped and nitrate was denitrified. Nutrient loads from the catchment to the lake via Mill Creek and the springs at the northern end of the lake are likely to have been very low. Robertson (1988) also notes the operation of the Arrow River irrigation scheme which at the time of writing in 1988, was taking 1.75m^3/s of water from the Arrow River and irrigating 1100ha in the middle of the Lake Hayes catchment. Through the 70's, 80's and the early 90's, multiple catchment stressors continued to affect lake water quality including the loss of wetland buffering capacity, the application of superphosphate fertilizers on new pastoral land, and continued catchment cutting and drainage works which delivered further pulses of sediments and nutrients to the lake.

1.3 Fisheries

1.3.1 Trout fishery

Brown Trout were introduced to Lake Hayes in 1870 and the fishery flourished from the late 1800's through the 1930's with fish up to 25lb caught (Fig. 2). In the 1940's, the Wildlife Service set up fisheries operations including the collection of brown trout ova (Robertson 1988; H. Trotter, Otago Fish & Game, pers. comm.) and the construction of a fish trap on Mill Creek near the inflow to Lake Hayes. From 1940-1960, 1000-4000 adult trout passed through the fish trap annually with up to 2 million ova collected annually for national and international fish stocking (Otago Fish & Game, unpublished data). Fish trapping operations slowed through the 1960's and 70's before ceasing in the late 1970's as demand for ova stock decreased and the water quality in Mill Creek declined (Robertson 1988; H. Trotter, Otago Fish & Game, pers. comm.).

Figure 2. Lake Hayes trout fishery, 147 brown trout, 1 October 1897.

1.3.2 Perch fishery

Perch were introduced shortly after trout, in the late 1870's, and quickly established, being the most commonly caught fish species by 1900. The perch population grew and in 1988 the Percy Perch Classic fishing competition was established, running until 1990. The event attracted over 1000 anglers to the lake and more than 13,000 perch were landed over two days in 1989; however around 97% of those adults caught were less than 20cm long, indicating a highly-stunted population (H. Trotter, Otago Fish & Game, pers. comm.). Such stunting can be a result of unrestrained population growth controlled only by competition within the species for food resources as opposed to predatory 'top down' population controls. The lack of predation on Perch continues to result in a high proportion of stunted adults confirmed by a survey in 2016 where 70 Perch were caught in one hour and 97% of adults were stunted (<20cm). (Otago Fish & Game, unpublished data).

The Lake Hayes trout fishery has long been recognized as a regionally important fishery and is highly regarded by anglers for both its recreational and amenity values (H. Trotter, Otago Fish & Game, pers. comm.), however its popularity among anglers has decreased rapidly since the mid 2000's. Annual angler days (a measure of angler effort) were at around 1500 days in the mid 1990's and early 2000's (Fig. 3) but dropped dramatically in 2006/2007 (Otago Fish & Game, unpublished data). Fish & Game received numerous complaints from anglers regarding the "muddy, brown colour" of the lake water and the poor condition and scarcity of trout during the 2006/2007 season, followed by two fish kills observed in March and April 2007 (Otago Fish & Game, unpublished data). In March, Mill Creek was running high and carrying brown sediment into the lake where few trout were seen, which were all in very poor condition and 6 dead trout were found around the creek mouth. In April, emaciated trout were observed in the lower reaches of Mill Creek and there were reports of around 30 dead trout floating near the mouth of Mill Creek. Lake Hayes itself had a bloom of the dinoflagellate *Ceratium hirundinella* during this time and all remaining trout observed were described as emaciated and in very poor condition.

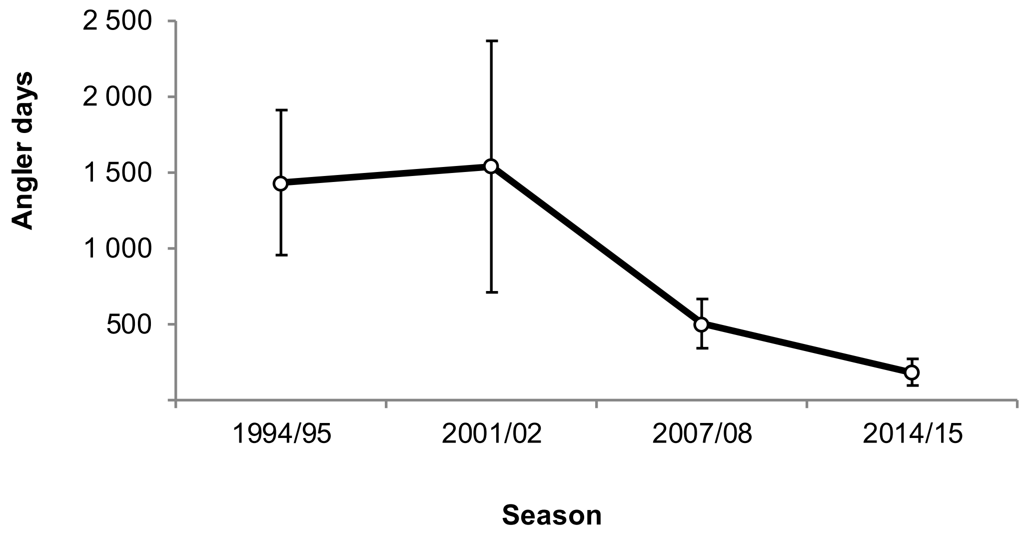

Figure 3. Annual total angler effort estimates for Lake Hayes taken from the National Angling Survey (Otago Fish & Game, unpublished data).

Due to increasingly poor fishing conditions, angler days dropped to around 500 in the 2007/2008 season (Fig. 3) and although no further fish kills have been reported, angler effort continued to decline to a record low of 180 angler days in the 2014/15 season. Over this time, public concerns arose again regarding the low numbers of poor quality trout spawning in Mill Creek, however a Fish & Game monitoring programme set up in response found the condition of trout in Mill Creek had improved over 2013-2015 compared with those found in 2007. While there have been reports of trout in good condition being caught in recent years (H. Trotter, Otago Fish & Game, pers. comm.), anglers remain concerned about the lower numbers of fish caught and the degraded water quality of the lake.

2 Water Quality

2.1 Background

The information presented above describes the situation, whereby Lake Hayes has become a eutrophic lake, with generally relatively low water clarity, poor water quality and frequent algal blooms. Since 2006, the trophic level index (indicating nutrient enrichment) has deteriorated markedly, and in 2015 the water quality of the lake was very poor (supertrophic) (Fig. 4).

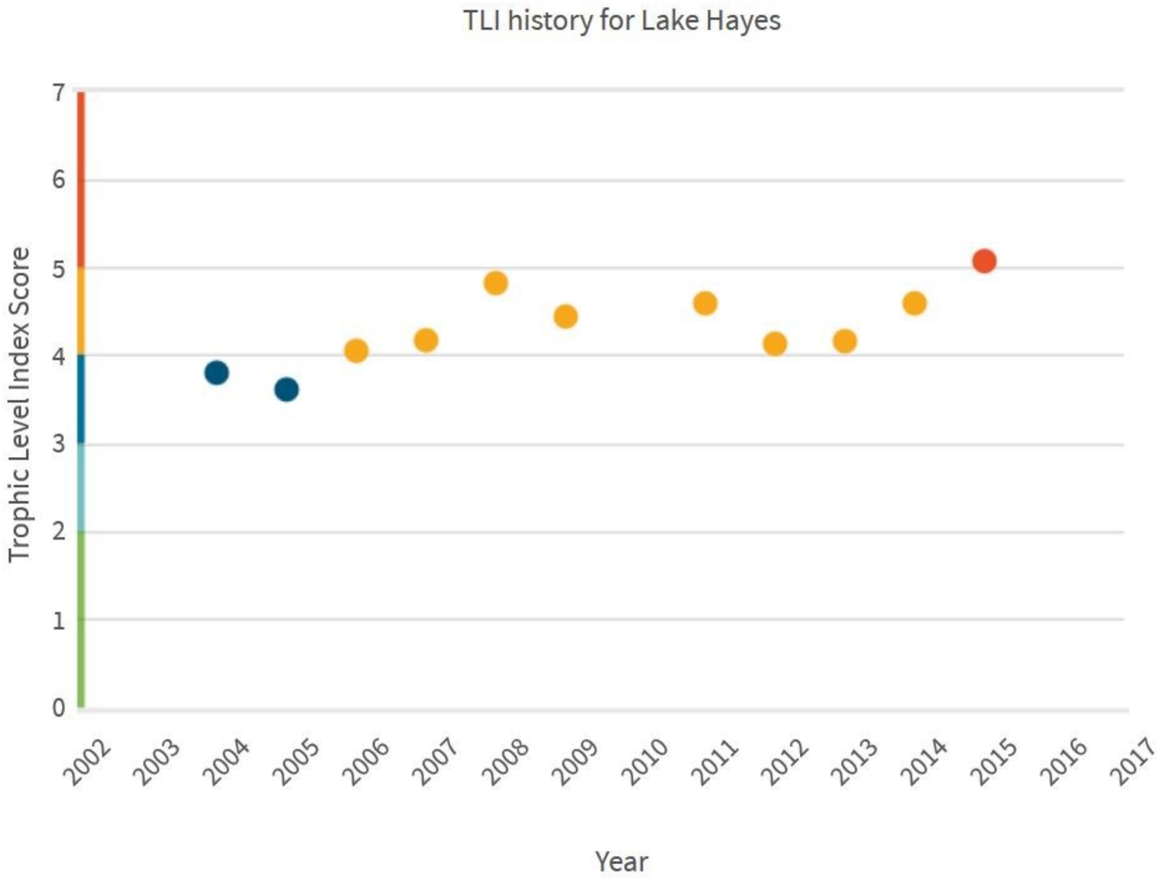

Figure 4. The trophic level index (TLI) score for Lake Hayes from 2004 to 2015. The TLI aggregates total phosphorus, total nitrogen, chlorophyll *a* (an indicator of phytoplankton biomass) and Secchi disk depth (a measure of water clarity) data. TLI between 3 and 4 is mesotrophic (good water quality). TLI between 4 and 5 is eutrophic (poor water quality). TLI between 5 and 6 is supertrophic (very poor water quality). Data and graph are from the LAWA website.

Publicly available data from the LAWA website (Land Air Water Aotearoa; https://www.lawa.org.nz/explore-data/otago-region/lakes/lake-hayes/) only go back to 2004. However, Lake Hayes has been studied since the late 1940s, beginning with the work of Jolly (Jolly 1959). Comparing lake data back as far as Jolly's time provides a useful context for our analysis of the historical and current condition of Lake Hayes.

Figure 5 presents the Lake Hayes Secchi disk depth data, showing how water clarity in the lake has changed over time. Since the 1950s, when the lake's bottom waters were oxygenated in summer (Jolly 1959), water clarity has been variable, but has often been quite poor (e.g., eutrophic or water clarity below 3.6 m) due to algal blooms.

From 1970 onward, nitrogen-fixing cyanobacteria (e.g., *Anabaena* sp.) have often been part of the phytoplankton community, sometimes occurring as the dominant species of phytoplankton (Burns & Mitchell 1974; ORC 1995). Nitrogen-fixing cyanobacteria may outcompete other phytoplankton when excess phosphorus is available because the cyanobacteria are able to harvest nitrogen from the atmosphere (from air dissolved in the lake water). The bottom waters of Lake Hayes have become anaerobic (with a complete loss of dissolved oxygen) since at least 1970 (Burns & Mitchell 1974) and summer deoxygenation of the bottom waters has been recorded, whenever it has been measured, since that time (Robertson 1988; ORC 1995; Bayer et al. 2008; Bayer & Schallenberg 2009; M. Schallenberg, unpublished data; ORC, unpublished data). The loss of dissolved oxygen from the bottom waters not only excludes trout, zooplankton and many invertebrates from the cooler bottom waters of the lake, but it also causes biogeochemical changes in the lake sediments, releasing sediment-bound phosphorus into the water column (Bayer et al. 2008). When the surface of the sediment is oxygenated, the oxygenated minerals (e.g., iron and manganese oxyhydroxides) in the sediments bind a large proportion of sediment phosphorus, preventing its release back into the water column. Deoxygenation of the water and sediment converts the sediments from P sink to a P source. Since at least 1970, the summer bottom waters have been releasing significant amounts of phosphorus into the lake water (Mitchell & Burns 1981; Robertson 1988; Bayer et al. 2008; Bayer & Schallenberg 2009; M. Schallenberg, unpublished data; ORC, unpublished data), recycling historically accumulated and immobilised P back into the lake ecosystem and further fuelling algal and cyanobacterial blooms. Severe blooms eventually settle to the lake bed delivering more P to the sediments as dead phytoplankton cells, where they decompose and consume oxygen, contributing to the next year's summer deoxygenation. This internal anoxia-phosphorus-algae feedback cycle has contributed to maintaining Lake Hayes in a eutrophic state since at least 1970 despite the fact that external nitrogen and phosphorus loading from Mill Creek and the springs at the northern end of the lake decreased into the 1990s and early 2000s (Caruso 2000; Bayer et al. 2008; Bayer & Schallenberg 2009).

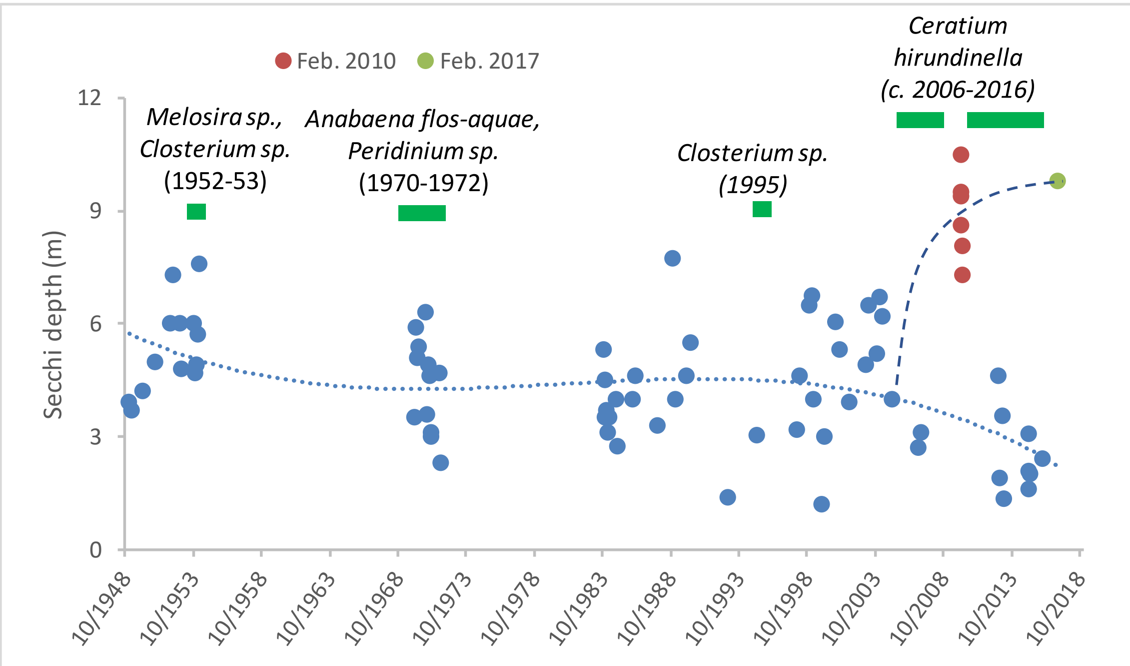

Figure 5. Historical summer (November to April) water clarity (Secchi disk depth) measurements in the open waters of Lake Hayes. The dominant phytoplankton species causing summer blooms are shown, if reported. Phytoplankton information and data from 1952/53 are from H. Jolly (1959) and Burns & Mitchell (1974), from 1970-72 are from Burns & Mitchell (1974), and from 1995 are from C.W. Burns, unpublished data. Blue dot Secchi data from 1984-2015 are from unpublished Otago Regional Council data, M. Schallenberg unpublished data, and Caruso (2001). Red dots are from M. Schallenberg, unpublished data. Green dot is a datum from the Otago Regional Council.

The average water clarity of the lake was rather stable from 1970 to 2006 (Fig. 5), until the brown-coloured dinoflagellate alga, *Ceratium hirundinella*, began to form dense blooms in the lake (Bayer et al. 2008). These blooms were more severe than most previous blooms, further reducing summer water clarity (Fig. 5). *Ceratium* blooms were also associated with fish kills and decreased angler interest in the lake, as discussed in Section 1.3, and began to cause skin and mucous-membrane irritation in at least one long-term local resident who regularly swam in the lake (M. Schallenberg, pers. comm.). Curiously, the severe *Ceratium* blooms were not associated with increased external nutrient loading from Mill Creek or the springs (Bayer & Schallenberg 2009; LAWA website) or with increases in internal nutrient recycling during the summer anoxic period. In fact, phosphorus concentrations in the anoxic deep (25m) summer waters appeared to have been decreasing from around the year 2000 and ammoniacal N concentrations were also low during the *Ceratium* bloom period compared to in the 1980s and 1990s (Fig. 6).

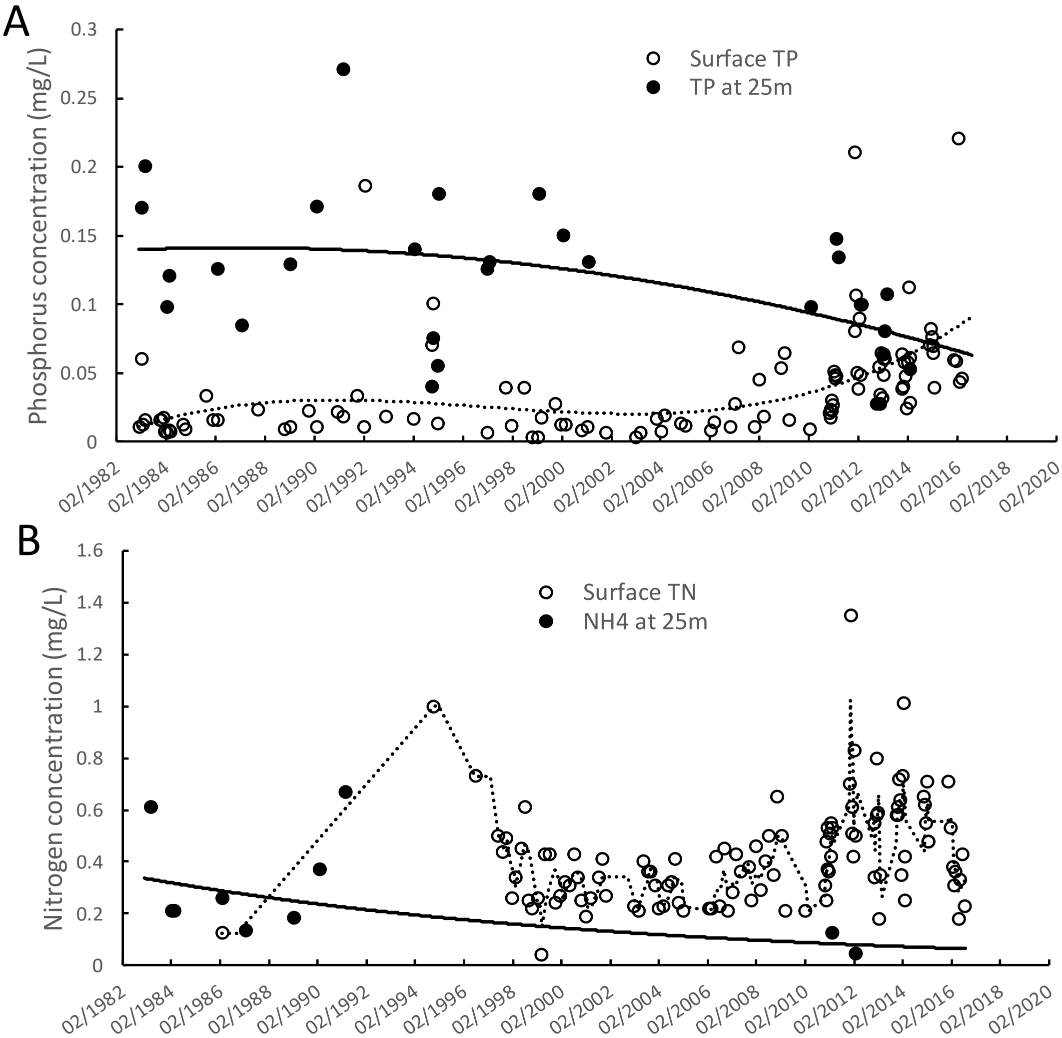

Figure 6. Trends in the concentrations of phosphorus and nitrogen concentrations in Lake Hayes during the stratified period (Nov-April inclusive) from 1983 to 2016. A. Total phosphorus at the lake surface and at 25 m depth (anoxic bottom waters). B. Total nitrogen at the lake surface and ammoniacal nitrogen at 25 m depth (anoxic bottom waters). Data are from the Otago Regional Council.

This intriguing situation of worsening algal blooms while internal and external nutrient loads had not measurably increased was frustrating for locals and recreational users of the lake. However, this apparent enigma can be explained by ecological peculiarities of the *Ceratium* alga, shown in Figure 7. This dinoflagellate is a large, spikey, motile (swimming) alga that is mixotrophic, meaning that it can gain energy both via photosynthesis (as plants do) and also by feeding on bacteria (as some protozoans and zooplankers do).

Figure 7. *Ceratium hirundinella* showing one of the two flagella used for locomotion/swimming. Total length of the cell is typically 150 µm to 200 µm. Photo: https://why.gr.

Mixotrophic dinoflagellates including *Ceratium* have been reported to become abundant and dominant in the phytoplankton communities of lakes during periods of lake recovery from eutrophication (Jeppesen et al. 2003; Gerdeaux & Perga 2006, Mehner et al 2008), partly due to their ability to supplement their nutrient requirements by feeding on bacteria (Gerdeaux & Perga 2006). *Ceratium* is also able to migrate vertically in the water column on a daily basis, enabling it to access recycled nutrients in the deep waters of lakes at night while also enabling high rates of photosynthesis in the upper water column during the day (James et al. 1992). In Lake Hayes, as in other lakes, these strategies probably enabled *Ceratium* to become highly competitive by nocturnally migrating and accessing nutrient-rich bottom waters at night and by feeding on bacteria when nitrate, ammonium and phosphate concentrations in the surface waters of the lake are scarce (i.e. during summer).

2.2 A *Ceratium* nutrient pump hypothesis

If *Ceratium* in Lake Hayes undertakes a day-night migration to harvest recycled N and P from the bottom waters during summer, then it is expected that *Ceratium* would transfer significant amounts of phosphorus from the bottom waters into the mixed layer during daytime, when it migrates to the surface layer to photosynthesise. The accumulation of recycled phosphorus in the bottom waters of Lake Hayes has been a major component of the P budget of the lake; however, Figure 6A shows that the recycled P contribution from summer bottom waters has decreased while the summer surface water phosphorus concentration has increased since the time that *Ceratium* began to bloom, in 2006. This raises the possibility that *Ceratium* may translocate P from the bottom waters to the surface waters of the lake during summer (Fig. 6A). We see a similar pattern over time for N (Fig. 6B), but, unlike for P, recycled ammoniacal N is only a small part of the N budget of the surface waters of the lake.

Because *Ceratium* blooms have been associated with increases in surface water P concentrations in summer (Fig. 6A), the apparent *Ceratium*-mediated transfer of P from the bottom waters to the surface waters in summer is expected to enhance the flushing of P out of the lake via Hayes Creek. This water flowing out of the lake is surface water and, therefore, the transfer of P from bottom waters to surface waters increases the flushing of P out of the lake. This hypothesis is consistent with the increasing concentrations of total phosphorus at the Hayes Creek outflow between 1993 and 2008 reported by Bayer & Schallenberg (2009). This enhanced P flushing should accelerate the recovery of Lake Hayes by eventually breaking the summer anoxia-phosphorus-algae feedback cycle that had been delaying recovery of the lake.

As the flushing of P from the lake progresses, it is expected that *Ceratium* blooms will eventually become self-limiting due to this apparent translocation and enhanced flushing of P from the lake. Currently, the declining levels of P available in the summer deep waters of the lake may already be reducing the competitive advantage that *Ceratium* has over other algae in the lake. A further reduction in *Ceratium's* competitive advantage could occur if the density of bacteria in the lake, which may supplement *Ceratium's* energy requirements, were also to decline.

2.3 Is Lake Hayes approaching a recovery tipping point?

We have shown that the bottom water nutrient concentrations which reflect internal recycling of legacy nutrients, have been decreasing in recent years. The apparent *Ceratium*-mediated transfer of P to the surface layers has probably increased the flushing of P out of the lake via Hayes Creek by increasing surface water nutrient concentrations in summer. Since 2006, *Ceratium* blooms have plagued the lake, where *Ceratium* has outcompeted other algae and cyanobacteria probably by harvesting phosphorus from deeper waters in summer and by grazing on bacteria to supplement its nutrition. While the lake has suffered severe *Ceratium* blooms in most summers since 2006, the developments described above suggest that the lake is on a trajectory toward recovery from historically high nutrient loads.

Further evidence of this is the fact that in the summer of 2009/10 and 2016/17, the lake experienced unprecedented water clarity (Fig. 5) and very low *Ceratium* biomass. The *Ceratium* hiatus in 2009/10 lasted only one summer, but the reduction in algal biomass in the surface waters (Fig. 8) and the increase in water clarity (Fig. 5) was striking. While the reduced internal P recycling probably contributed to these clear water summers, another interesting feature of these summers was the persistence of the water flea, *Daphnia pulex*, in the lake over the summer period. *D. pulex* is an intense grazer of algae (Burns 2013) and our sampling of zooplankton in the summers of 2009/10, 2012/13 and 2015/16 indicate that during summers when *Ceratium* bloomed, *D. pulex* was absent from the lake. Thus, we believe that food web interactions related to summer *Daphnia* presence in the lake also contributed to the sudden shift of Lake Hayes from a eutrophic condition with severe summer *Ceratium* blooms to summers with very low *Ceratium* biomass (Figure 9).

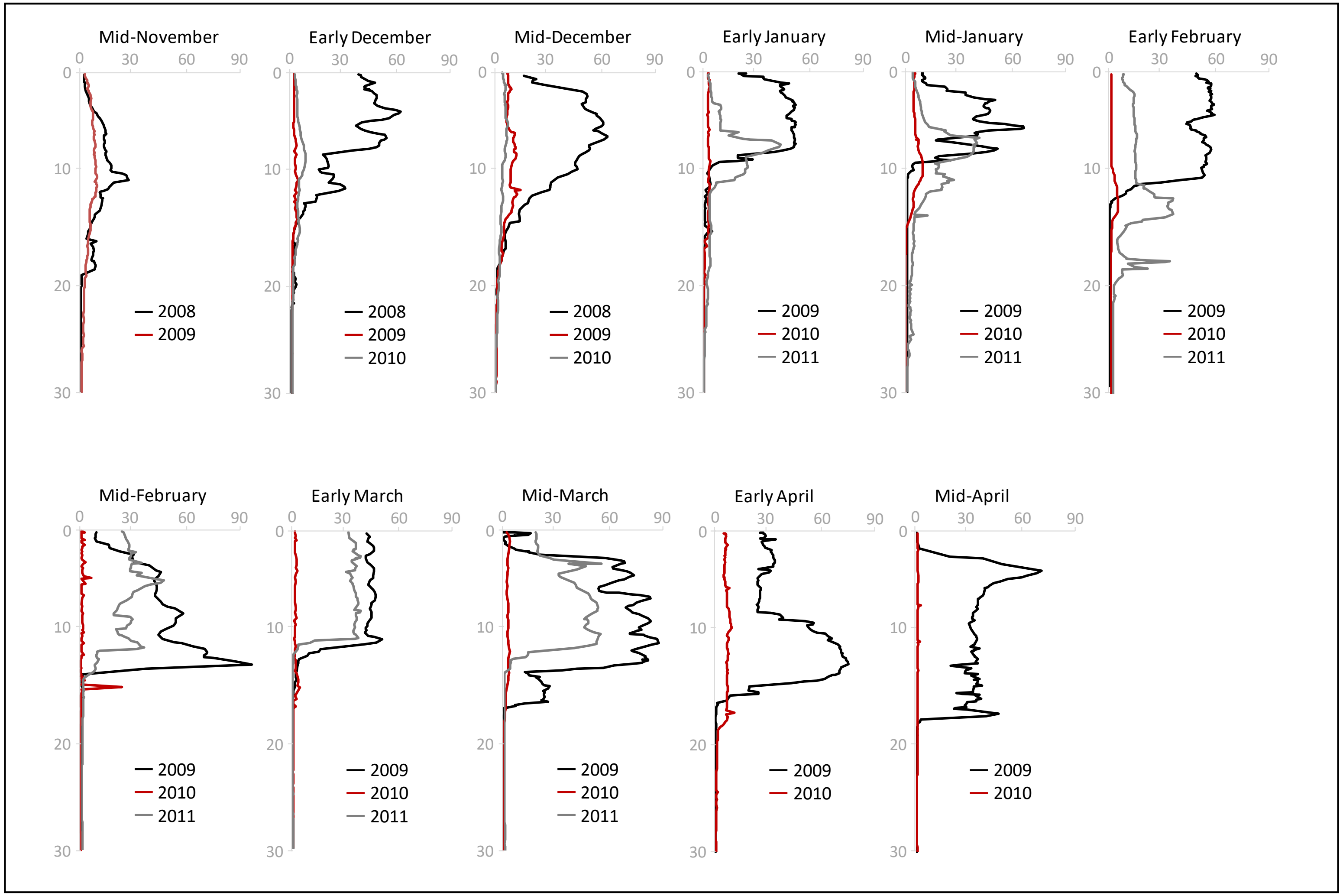

Figure 8. Vertical profiles of chlorophyll *a* in the summers of 2008/09, 2009/10 and 2010/11. Profiles were measured by the Otago Regional Council. Chlorophyll *a* data are *in vivo* fluorescence measurements from an uncalibrated datasonde and are, therefore, approximate concentrations in µg/L.

Rapid changes in trophic state are common in shallow lakes, which can fluctuate markedly in water clarity from year-to-year (Mitchell 1988; Scheffer 2004; Schallenberg & Sorrell 2010), when nutrient loading approaches a tipping point. However, such behaviour is not as common in deep, seasonally stratifying lakes, but has been reported in relation to species invasions (e.g., Lakes Erie and Ontario due to zebra mussel invasion) and to the dynamics of algal pathogens (e.g., pathogenic fungi controlling *Ceratium* spp.; Heaney et al. 1988). Circumstantial evidence described here suggests that Lake Hayes has entered a phase of recovery whereby nutrient availability is approaching a recovery tipping point and that food web interactions in some years may have tipped the lake into a temporary recovery from eutrophication (Fig. 9). These food web interactions are discussed in more detail in Appendix 1.

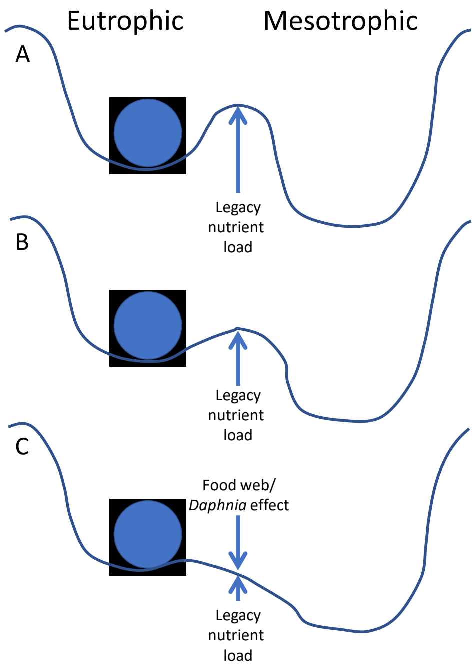

Figure 9. A conceptual model showing alternative stable states for Lake Hayes. The lake is represented by the ball, which is held in the eutrophic state by intensely recycled phosphorus (A). Recent reductions in P recycling have weakened the resistance to recovery (B). In the summers of 2009/10 and possibly 2016/17 (to be confirmed), the combination of reduced P recycling and food web effects reduced resistance to recovery further, allowing a temporary shift to a clear water state (C). In summer 2010/11, the *Ceratium* bloom returned and the lake shifted back to a eutrophic state (Fig. 8).

Experiments done on the Lake Hayes phytoplankton community in 2006 (Bayer et al. 2008) indicated that the phytoplankton community in the lake (dominated by *Ceratium* at the time) was stimulated by additions of N and the trace elements boron and zinc. In the four experiments conducted, phosphorus additions did not stimulate phytoplankton production. This result supported an analysis of N:P ratios in the lake, which also suggested that P was often in surplus in the lake water relative to N (in relation to the nutrient demands of phytoplankton) (Bayer et al. 2008). It would be interesting to now re-examine the nutrient supplies in the lake to test whether a decade of *Ceratium* dominance in the system has reduced phosphorus levels in the lake to the point where they can again begin to restrict phytoplankton blooms. The re-establishment of P-limitation of phytoplankton growth would have the added benefit of removing the competitive advantage of N-fixation, which historically dominant bloom-forming phytoplankters such as *Anabaena sp.* are capable of.

Our analysis of water quality data has yielded some insights into the drivers of phytoplankton blooms in Lake Hayes and it highlights the importance of having a detailed understanding of the nutrient budgets of lakes affected by nutrient enrichment. The combination of Otago Regional Council State of the Environment monitoring data and the University of Otago's occasional research projects on the lake provides a useful perspective on the factors driving large changes in water quality of the lake over time. Although the available data are patchy and many knowledge gaps would need to be filled to confirm the hypothesis presented here, the combined use of lake data, information on overseas lakes and expert experience and deduction provide a compelling hypothesis concerning the recent condition of the lake and what could be done to speed its recovery.

2.3 Key points on water quality analysis

The key points from our analysis of water quality data are as follows:

- The internal recycling of phosphorus during summer, which has been a large source of P to the lake at winter turnover, has been decreasing in recent years such that it is now so reduced so as to have little effect on the surface water concentrations of P.
- *Ceratium* blooms began in response to reduced internal nutrient recycling and external nutrient loading and its proliferation has probably been due to the fact that it can harness nutrient resources unavailable to most other algae.
- The *Ceratium* blooms have probably transferred recycled P from bottom waters to the surface waters, enhancing flushing of legacy P from the lake.
- Since *Ceratium* can't add P or N to the lake, it's apparent translocation of P (and maybe N) to the surface waters probably increases the flushing of these nutrients out of the lake and will eventually reduce nutrient availability to the point where Ceratium may become limited by low nutrient availability.
- Lake Hayes appears to be approaching a recovery tipping point, where nutrient availability and food web factors prevent the development of significant algal biomass in occasional summers.
- At this point, the food web factors assisting the temporary recovery seem to involve *Daphnia* persistence over the summer months.
- The current situation suggests that appropriate restoration measures could result stable in improvements in summer water clarity, reductions in *Ceratium* summer biomass, and the re-oxygenation of the bottom waters of the lake. These factors appear to be facilitated by maintaining a low nutrient availability and a high summer *Daphnia* density.

3 Lake Hayes historical timeline of events

The information presented in Sections 1 and 2 can be summarised in a timeline describing the trajectory of Lake Hayes and its catchment, as related to the health of the lake. The significant historical events in Figure 10 show how complex the Lake Hayes system is and how a combination of historical factors (e.g., fish introductions, fertiliser overuse, dairying, wetland drainage, etc.) and current factors (e.g., *Ceratium* blooms, *Daphnia* dynamics, decreasing nutrient loads, etc.) affect the current conditions and trajectory of the lake. The complexity of the Lake Hayes system highlights that management of the lake must consider a comprehensive range of factors that operate on a range of time scales and that often interact with each other to affect lake health.

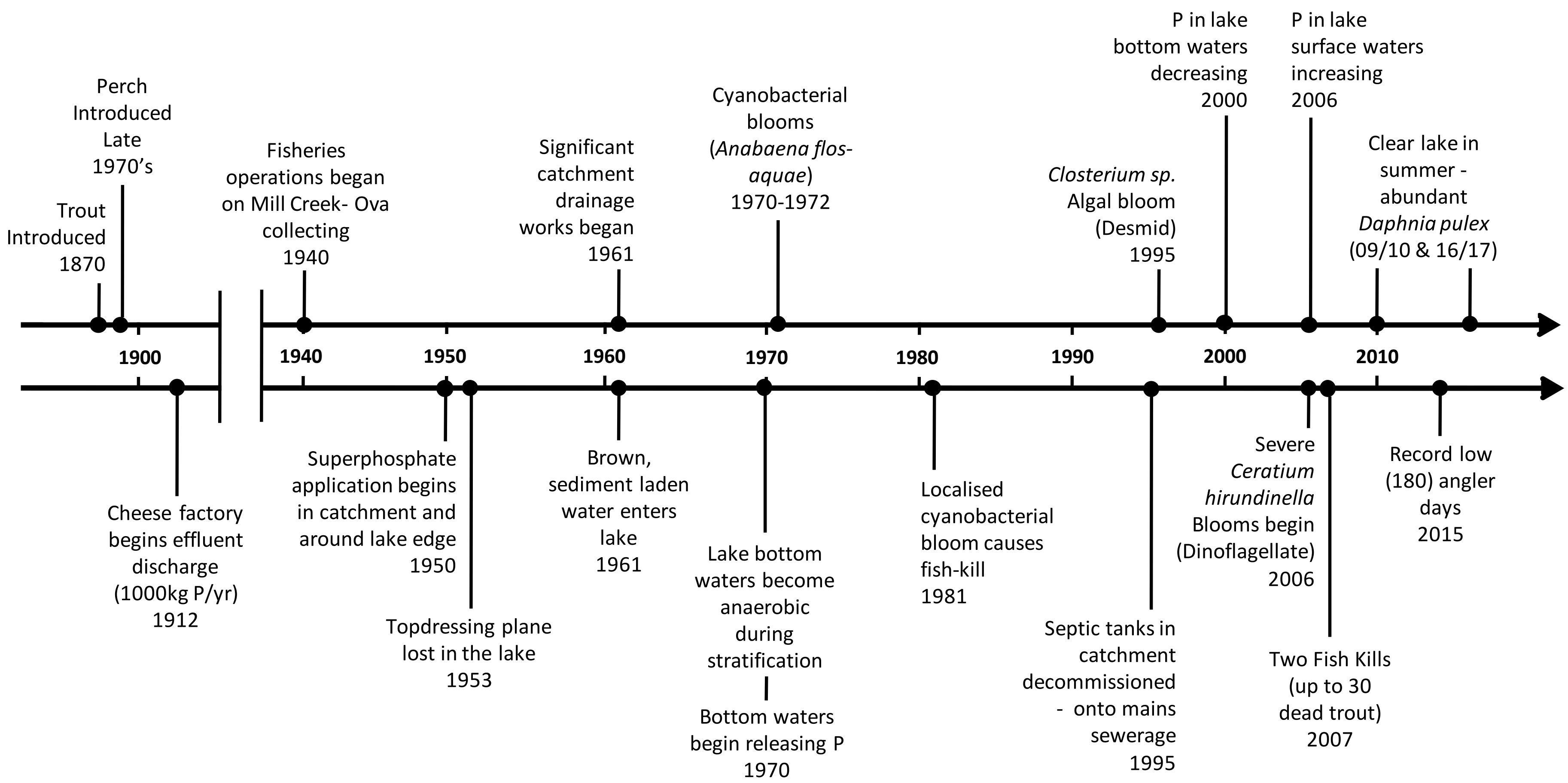

Figure 10. Historical timeline of major events impacting Lake Hayes and its catchment.

4 A restoration strategy for Lake Hayes

Restoration options for Lake Hayes have been discussed since the early 1970s, soon after the lake experienced its first severe algal blooms (Mitchell & Burns 1972) and have been revisited numerous times since then (Robertson 1988; Bayer & Schallenberg 2009; Ozanne 2014). Numerous strategies have been considered in terms of effectiveness and cost-effectiveness. The strategies fall into five main types: 1. catchment rehabilitation to reduce external nutrient loads to the lake, 2. reduction of internal nutrient loads/recycling, 3. food web manipulation, 4. flushing of water through the lake, and 5. other in-lake actions (Table 1).

Mitchell & Burns (1972) discussed and costed options including diversion of inflows away from the lake, flushing the lake with irrigation water and oxygenation of the bottom waters. Robertson (1988) placed much emphasis on catchment mitigation strategies, while subsequent reports have focused more on controlling in-lake P loading/recycling from the hypolimnion. The focus on internal P loading was sensible when the lake was releasing large legacy amounts of P into the water from the anoxic bottom waters on an annual basis and when the growth of the dominant phytoplankers was limited by P availability. However, our analysis of water quality data show that nowadays, the P concentration in the deep bottom waters (25 m) is similar to the P concentration in the surface waters during summer. In addition, the concentration of P in the Hayes Creek outflow had been rising at least up to 2009 (Bayer & Schallenberg 2009), indicating that while external P loads had reduced (Bayer & Schallenberg 2009) more P was being flushed from the lake. Our analysis suggests that *Ceratium* blooms that have plagued the lake since the mid-2000s have transferred bottom water P to the surface of the lake, enhancing the flushing of legacy P out of the lake.

Lake Hayes has a water residence time of around 1.8 years (Caruso 2000). Flushing of the lake could potentially be enhanced by augmenting Mill Creek with cleaner water from the Arrow River Irrigation Scheme, as was suggested by Mitchell & Burns (1972). When surplus water from the scheme is available (November to June – see Appendix 2), it could be used to augment the flushing of the lake. Augmented flushing of the lake could enhance the flushing of algae and P out of the surface waters of the lake. It could also add dissolved oxygen if the augmented flow were to plunge into the bottom waters during the stratified period. Total P concentrations are currently almost uniform in the surface and bottom waters in summer (Fig. 6A) and algae are concentrated in the surface layer in summer (Fig. 8). According to the surplus irrigation water that would be available, the augmentation flow available from the Arrow River would flush an additional 7% of the lake volume if the full surplus were used from September to June (Appendix 2). The relative temperatures (and therefore densities) of Mill Creek and lake water during this period suggests that Mill Creek (including any augmentation flow), is not likely to plunge into the bottom waters of the lake (Appendix 2). Therefore, while flow augmentation would enhance flushing of the lake by 7% per annum, it is unlikely to significantly alter the oxygen concentration or dynamics in the bottom waters of the lake. Thus, according to the analysis, the main benefits of flow augmentation to the recovery of the lake would not occur immediately, but would accrue over time, as long as the nutrient concentrations in Mill Creek don't increase.

We have no data showing that *Ceratium* in Lake Hayes migrates vertically in the water column, drawing recycled nutrients from the bottom waters to the surface waters. However, this nutritional strategy, along with the ability to feed on bacteria, potentially allows Ceratium to persist and outcompete other algae when dissolved, inorganic plant nutrients (i.e., nitrate, phosphate, ammonium) are scares. Thus, studies have shown that dinoflagellates such a *Ceratium* may be indicators of reducing inorganic nutrient availability in lakes (Jeppesen et al. 2003; Gerdeaux & Perga 2006; Mehner et al. 2008).

If our hypothesis is correct and internal nutrient recycling is playing a diminishing role in fuelling algal blooms in the lake, then restoration actions aimed at reducing internal P loading/cycling will provide diminishing benefits into the future. Previous reports have recommended strategies to reduce internal P loading/recycling in the lake. However, in light of the information provided here, the restoration benefits of such actions into the future should be carefully scrutinised in terms of their costs and potential benefits. An analysis of the cost of alum treatment for Lake Hayes was carried out by John Quinn and Max Gibbs of NIWA in 2015 (Appendix 3), based on a maximum accumulation of dissolved reactive phosphorus in the bottom waters of 300 mg/m^3 of P, which was estimated based on previous measurements of P made in 1994/95 and 2012/13 (M. Schallenberg, unpublished data). However, more recent data and the analysis presented in Section 2 suggest that this may be an overestimate of the current maximum accumulated P by a factor of two. So, their estimated cost for an alum treatment of $535,000 (Appendix 3) may also be an overestimate by a factor of around two. Otago Regional Council samples being collected this summer will confirm whether or not the cost estimate in Appendix 3 can be substantially reduced.

While gains have been made in reducing nutrient loads to the lake from septic tanks, and nutrients in Mill Creek and the springs (Bayer & Schallenberg 2009), the condition of Mill Creek has stabilised since around 2005 (LAWA website; Appendix 4) and there are indications that summer nitrogen concentrations in Mill Creek may be increasing, although not yet statistically significant (LAWA website; Appendix 4). In comparison to other upland streams, Mill Creek is higher than average in nitrogen and *E. coli* (faecal bacteria) concentrations and in turbidity, while it is lower than average in phosphorus concentrations (LAWA website). Because Lake Hayes is an important and sensitive receiving environment, we suggest that the concentrations of contaminants flowing into Lake Hayes from all sources should be better than average. We, therefore, recommend that attention be refocused on land practices and nutrient, *E. coli* and sediment losses from the Lake Hayes catchment in order to further improve water quality in the lake (Appendix 4). The rapid rate of land development and the diversity of land uses in the catchment both highlight the importance of the use of best management practices and suggest that land/nutrient management mechanisms might be appropriate to protect the lake from future degradation. These should focus on phosphorus and nitrogen and should also account for fluxes during flood flows.

In the summers of 2009/10 and 2016/17 the lake did not experience algal blooms. This suggests that the lake's condition is destabilising and has begun to flip between algal blooms and clear water summer conditions. This kind of behaviour is a characteristic of complex systems as they approach pressure-response tipping points or thresholds. We hypothesise that the recovery from external and internal nutrient loading is creating instability in the lake and that the clear water summers are evidence that a stable recovery is attainable.

Table 1. Lake Hayes restoration actions discussed and recommended in four reports. Actions recommended in each report are shaded green.

Mitchell & Burns (1972)	Robertson (1988)	Bayer & Schallenberg (2009)	Ozanne (2014)	This report
Reducing external nutrient loading				
	• Wetland re-establishment • Reduce fertilizer application and runoff • Establish streambank buffer • Control channel clearance and drainage operations in catchment • Manage future development P load			**Collaborative catchment management plan** • Wetland re-establishment • Reduce fertilizer application and runoff • Establish streambank buffer • Control channel clearance and drainage operations in catchment • Manage future development P load
Change in catchment land use	Change in catchment land use			
Divert high P water out of the lake	Divert high P water out of lake		Divert high P water out of lake	
	Reduce runoff from animal stocking			
Reducing internal nutrient loading				
	Chemical P precipitation/inactivation	Chemical P precipitation/inactivation	Chemical P precipitation/inactivation	Chemical P precipitation/inactivation
	Hypolimnetic withdrawal	Hypolimnetic withdrawal	Hypolimnetic withdrawal	
Hypolimnetic aeration	Hypolimnetic aeration	Hypolimnetic aeration	Hypolimnetic aeration	
		Dredging	Dredging	
			Sediment oxidation	
			De-stratification	
Food web biomanipulation				
		Daphnia enhancement	*Daphnia* enhancement	*Daphnia* enhancement
Flushing				
Enhance flushing	Enhance flushing, remove flushing restriction at the outlet	Enhance flushing	Enhance flushing	Enhance flushing
Other in-lake strategies				
			Use of floating wetlands, algicides, pathogenic bacteria and ultrasound	

Why did the water clarity of the lake suddenly improve in these summers of 2009/10 and 2016/17? Data and observations suggest that the summer persistence of *Daphnia* in the lake was associated with the clear water phases. Current studies by the University of Otago Zoology Department and collaborators are examining the potential of perch recruitment to regulate *Daphnia* density via predation by juvenile perch on *Daphnia.* Preliminary evidence suggests that the perch life cycle and spawning time in the lake may be associated with crashes in *Daphnia* densities during the summers when algal blooms were severe (Appendix 1). The data in Appendix 1 outline compelling evidence that food web biomanipulation to reduce juvenile perch numbers and increase summer *Daphnia* densities could push the lake into a more stable clearwater phase. While research is ongoing, a key component of the research will be completed by the end of 2017 and we have begun to consider various approaches that could be used to reduce the numbers of young perch in the lake (see Appendix 1 for examples).

4.1 Restoration plan and timeline

Table 2 and Figure 11 outline a proposed restoration plan and timeline for accelerating the recovery of Lake Hayes to a stable desirable state and for controlling catchment development to prevent potential future increases in nutrient and sediment loading from the catchment to the lake. Table 3 sets out potential restoration targets for the recovery. These are suggested targets only because interested members of the community, iwi and stakeholders together with the Otago Regional Council would need to vet any final targets. We present the draft targets in Table 3 to help initiate a collaborative community project to set final restoration targets.

Table 2. Proposed restoration plan for Lake Hayes.

Action - in order of priority	**Reasons/benefits**	**Time**	**Likely effectiveness**	**Cost**	**Cost effectiveness**
Catchment management plan and actions	• High population growth rate • Diverse land uses • Valuable and sensitive receiving environment • Community education, collaboration and buy-in • Holistic improvement	• Start the process immediately • In 2017/18 undertake a management plan feasibility study • Continuing development and refinement over time	• Large number of diffuse and long term benefits including education, stakeholder involvement, and improvement in water quality • Effectiveness in improving water quality will depend on feasibility to be determined in catchment management plan	• Unknown, but some costs can be covered under normal Regional and District Council business	• Likely to be highly cost-effective in the long term
Biomanipulation	• Potentially restore eels to the lake • Potentially increase native bully and koaro densities in the lake • Reduce the density of non-native perch	• Immediately begin looking into options • In early 2018, undertake a feasibility study in light of information in Helen Trotter's MSc thesis (end of year) • Carry out biomanipulations in 2018	• Likely to be effective, but to what extent depends on feasibility study • Some techniques may require follow up or ongoing biomanipulations	• Unknown. Depends on techniques used • Unlikely to be expensive	• Potentially highly cost-effective • Some techniques may require ongoing biomanipulations to keep perch recruitment down

Table 2. Proposed restoration plan for Lake Hayes, continued.

Action - in order of priority	**Reasons/benefits**	**Time**	**Likely effectiveness**	**Cost**	**Cost effectiveness**
Flushing	• Increase flushing rate but requires upfront connection cost and continual water purchase	• Can be carried out immediately if cost-benefit is favourable	• Potentially helpful in the long-term	• $22,000 hook-up cost plus $35,000 p.a. for 200 L/s capacity	• Moderate long-term cost-effectiveness
Alum dosing	• Expensive, but likely to reduce internal P loading further	• Reassess costs in light of declining P concentrations in bottom water • If sufficiently cost-effective, could be carried out in summer 2017/18	• Effective immediately upon use • May require follow up treatments after 5 years	• Between $250,000 and $500,000, depending on reassessment using 2017 lake phosphorus data	• Low to moderate immediate cost-effectiveness

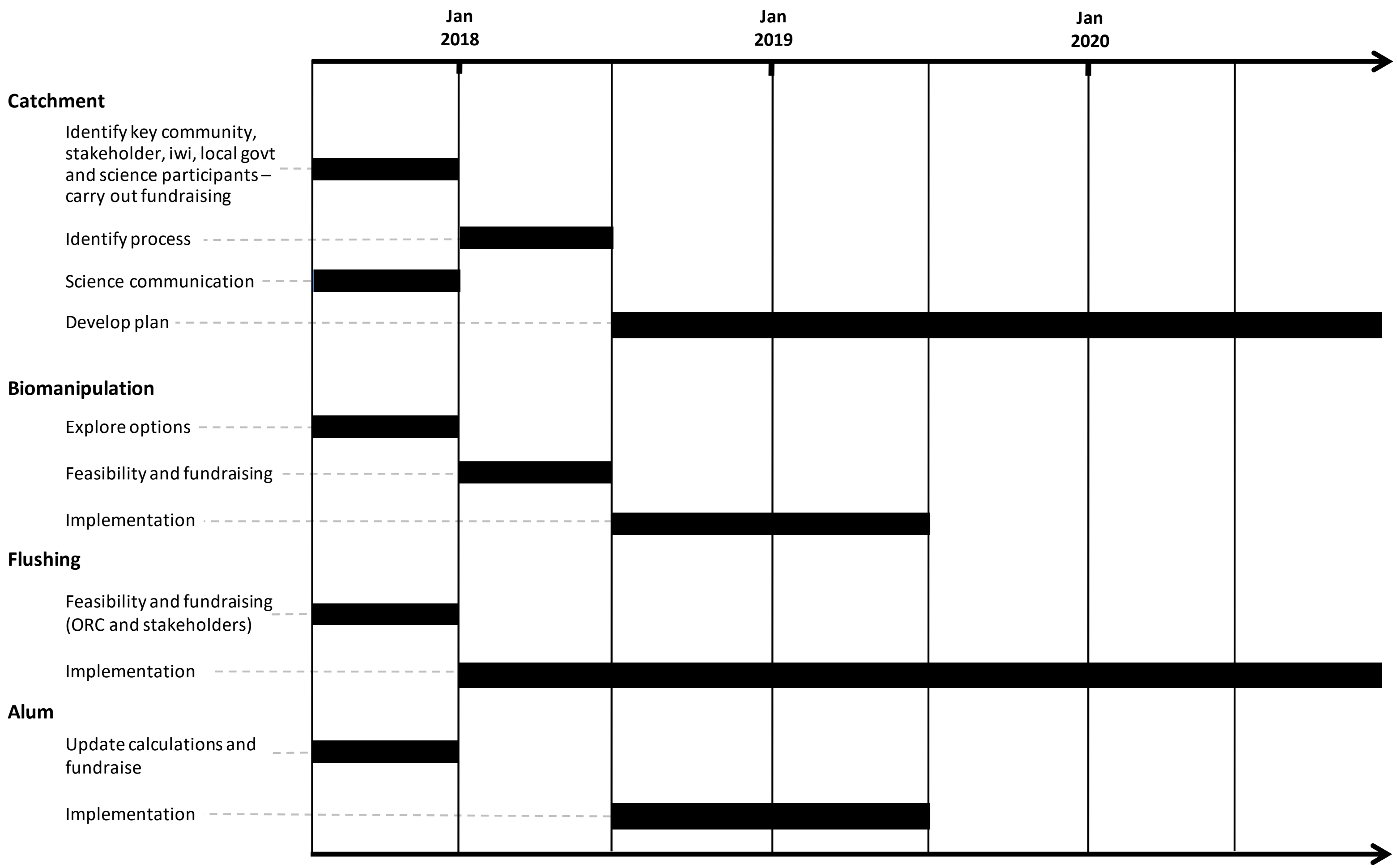

Figure 11. Proposed timeline for planning and implementation of lake restoration strategies.

Table 3. Suggested restoration targets for Lake Hayes. A successful restoration should achieve these targets consistently, from year to year.

Lake condition desired	Measurable targets to meet condition	Parties to help develop final targets
Lack of summer algal blooms and clear water which encourages recreational activities including swimming/bathing	• Trophic lake index during summer months < 4 • Secchi disk depth in summer months > 4 m	Otago Regional Council, locals, Fish & Game, rowing clubs, etc.
Inhibition of internal nutrient loading	• Improving oxygen content of the summer bottom waters to eventually achieve a condition where oxygen is never fully depleted • Nutrient budget shows a net loss of P from the lake on an annual basis, eventually stabilising at an ecologically sustainable level of P accumulation (permanent burial)	Otago Regional Council, University of Otago
Improved trout fishery	• Increase angler hours to 1995 levels • Reduce turbidity and temperature in Mill Creek • Maintain or increase Mill Creek flow rates	Fish & Game
Reduce nutrient loading to the lake from Mill Creek and the springs	• Reduce nitrate concentrations in Mill Creek • Reduce suspended sediment (turbidity) and total phosphorus concentrations in Mill Creek during high flow events • Develop a catchment management plan to ensure improved land management practices are adopted throughout the catchment	Otago Regional Council, Fish & Game and catchment land owners
Restore longfin eels to the lake	Achieve an eel trap and transfer target	Iwi, Contact Energy, DoC, Fish and Game
Maintain or enhance native biodiversity	Develop a strategy to prevent the incursion of invasive non-native species	Ministry of Primary Industries, DoC, Otago Regional Council

5 Water quality and lake health monitoring for Lake Hayes

Lake Hayes is one of the most thoroughly monitored and studied lakes in New Zealand. A combination of interest from university academics since the 1940s coupled with the Otago Catchment Board/Otago Regional Council monitoring of the lake which began in the early 1980s, provides one of the best datasets from which to interpret and understand lake conditions and trends in the country. Despite the scientific interest it has attracted and the relatively good lake monitoring dataset that exists, the cumulative data assembled on the lake are barely enough to allow for a good understanding of the lake’s changes over time.

5.1 The importance of regular and consistent monitoring

Lake Hayes and its catchment constitute a complex, linked terrestrial and aquatic ecosystem, which is highly valued by the local community. The water quality monitoring of the lake by the Otago Catchment Board/Otago Regional Council has been sporadic. Therefore, the dataset for the lake has many gaps and changes in sites and lake depths sampled, making it difficult to extract robust long-term data and to derive clear interpretations of how the lake has responded. The pitfalls of intermittent sampling can be seen in the dataset, where Secchi disk and trophic level indicators were not sampled in the summer of 2009/10, when the lake experienced a highly unusual and dramatic shift to what was likely an oligotrophic condition. This illustrates the potential for the lake to undergo rapid changes from year-to-year. Such dynamics are important to understanding the condition of the lake and its trajectory over time. Variability over time is an important indicator of lake condition, and trend and variability can best be assessed based on regular, long-term sampling of the lake's state.

5.2 The importance of nutrient budgets

Lake Hayes has undergone major changes in water quality as a result of major changes in nutrient loads over time. It's resistance to improvement from the 1980s to 2000s was in part due to the legacy of phosphorus loading that occurred in the 1960s and 1970s, which had been held in the lake bed sediments and recycled on an annual basis for decades. This illustrates the effect of legacies and time lags, which can affect lake condition. Consequently, the lake has probably rarely been in a nutrient equilibrium or steady state, where nutrient loads are balanced with nutrient concentrations and nutrient losses (via sedimentation and outflow).

Nutrient budgets involve the simultaneous intensive sampling of nutrient inputs (surface water and groundwater), nutrient concentrations (vertically resolved in a stratified lake) and nutrient losses. In the absence of such a nutrient budget (calculated at least on an annual basis), understanding of whether the lake is becoming cleaner or more polluted is only inferable via careful inference based on incomplete water quality data.

Nutrient budgets are a standard approach to understanding the nutrient sink/source dynamics of a lake in relation to nutrient loads from the catchment. To fully understand the condition and trajectory of a lake, it helps to understand if the lake is absorbing or shedding nutrients in relation to its nutrient inputs. Robertson (1988) calculated a P budget for the lake over a two-year period, which showed that the lake was retaining P. The current condition of *Ceratium* blooms is probably helping speed the shedding of legacy phosphorus from the lake bed to the outflow of the lake. Therefore, although the lake appears to be degrading, it is probably improving (flushing P), although only a current or recent nutrient budget can confirm this.

5.3 The importance of long term datasets

As described above, Lake Hayes and its catchment have probably rarely been in nutrient equilibrium over the past 50 years. The Secchi disk depth data, reaching back to the late 1940s show how water clarity has degraded, has been highly variable with a stable average for over 30 years, has further declined and has recently begun to swing sharply and intermittently from very low to extremely high clarity. This long-term information is very helpful in understanding many important characteristics of the lake such as its pre-degradation condition, how far the lake has departed from that condition, the time scales of change (which can indicate resistance to change), where the lake's tipping points might be, how close it might be to recovery, and so on.

Furthermore, long term data provide the best chance of identifying small and slow but important changes in the lake such as climate change impacts or depletion of the recycled phosphorus pool. The longer the dataset, the easier it is to distinguish ecological signal from stochastic noise in the dataset.

5.4 The importance of monitoring factors beyond simple water quality variables

Water quality (e.g., nutrients, water clarity, algal biomass) relates directly to the appeal of the lake for recreational activities like boating, fishing and swimming. So, it is understandable that statutory obligations for monitoring lakes focus mainly on water quality. However, to understand how and why changes to water quality occur in lakes, it is important to have information on supporting factors of the lake, which either drive or help explain changes in water quality. These could be related to climate change (temperature, mixing, etc.), which councils routinely monitor. However, key secondary factors often relate to components of the food web, such as the dominant phytoplankton species, zooplankton density, aquatic plant distributions, the presence of invasive species, etc. Early warning indicators of change in lakes are most likely to be changes in community composition of biological communities such as algae or zooplankton (Schindler 1987). For example, our interpretation of the data from Lake Hayes suggests that the development of *Ceratium* blooms in the mid-2000s both indicated an improvement in nutrient conditions and facilitated further improvement in nutrient status. One could not come to this conclusion based on water quality information alone.

5.5 The importance of monitoring change on different time scales

Regular monthly sampling of a lake allows for the analysis of lake changes on three important time scales: monthly, annual, and inter-annual. However, sometimes monthly sampling is too coarse a time scale to understand key drivers and processes. For example, climate warming should increase the period of time that a lake is thermally stratified. Changes in the timing and period of stratification can affect the ecology of the lake ecosystem and water quality (e.g., Winder & Schindler 2004). Unfortunately, monthly sampling will not reveal the exact timing of lake turnover or when the first day of seasonal thermal stratification occurs. So, monthly temperature profiles don't help identify some key climate change-related effects that could affect the state of the lake. Other important factors, such as: i) when the lake's temperature reaches a threshold for perch spawning or ii) when temperature becomes stressful to brown trout or iii) how rapidly dissolved oxygen in the bottom waters is depleted, can best be determined from high frequency measurements using *in situ* lake monitoring sensors.

5.6 The importance of monitoring at different places in the lake

In a lake with a simple basin shape and bathymetry, like that of Lake Hayes, it is tempting to think that monitoring only at the deepest site will be adequate. While one site can provide very useful data, water quality factors can vary substantially across Lake Hayes and with depth in the lake. For example, *Ceratium*, a motile alga, is often observed in distinct brown patches from the surface of the lake (Fig. 12). If the distribution of the dominant alga is patchy in the lake, sampling at a single site won't give an accurate estimate of the biomass of algae in the lake.

Figure 12. Brown-coloured, patchy *Ceratium* bloom in Lake Hayes, summer 2006/07. Photo: Otago Fish & Game.

Jolly (1952) indicated that there was high spatial variation in the densities of zooplankton due to the effect of winds and currents. We have also observed high spatial variability in zooplankton distributions in the lake.

Ceratium has the ability to migrate vertically in the water column and is often seen to intensify in biomass from early morning to midday while zooplankton species also migrate vertically in the lake from day to night (Jolly 1952; James et al. 1992). So, monitoring of the lake should take into account such vertical movements of algae and zooplankton so that better estimates of biomass of these can be monitored.

5.7 Suggested monitoring for Lake Hayes

While the historical Otago Catchment Board/Otago Regional Council data has been valuable for explaining changes in water quality and nutrient dynamics, lake monitoring could be improved along the lines discussed above. Comprehensive and good quality lake monitoring data are especially important in the context of lake restoration because the effectiveness and cost-effectiveness of restoration actions can only be ascertained by careful monitoring of changes in the lake. Furthermore, year-to-year variation in factors such as temperature, mixing depth, timing of stratification, zooplankton dynamics, fungal pathogen dynamics and the effect of floods can impact on expected recovery and such dynamic behaviour needs to factored into assessments of restoration success.

Below, we present some options for improving the monitoring of the water quality and health of Lake Hayes to support restoration actions.

Table 3. Suggested lake monitoring approaches in order of priority.

Priority	Type of monitoring	Frequency and technology
1a.	Sampling by boat at 2 deep water sites (31m and c. 26m) 1. CTD datasonde casts (Temp, DO, Chl *a*, phycocyanin) 2. Samples at 5m, 10m, 15m, 20m 25m, 30m for: • Total, dissolved inorganic N and P • Chlorophyll *a* and pH (only at 5m) 3. Samples at 5m, 10m and 15m for phytoplankton species 4. Vertical zooplankton hauls for species and density of Daphnia 5. Secchi depth	Monthly; various standard methods
1b.	P budget Measure total P and flow rate (where relevant) in: • Mill Creek (plus flow) • Spring (plus flow) • 6 depths in the lake at 31m site (1a.) • Hayes Creek outflow (plus flow)	Monthly; standard wet chemistry methods
2.	Profiling lake monitoring buoy at 31m site • Temp • DO • Chl *a* • Phycocyanin (cyanobacteria)	Hourly; Limnotrack monitoring buoy
3.	Survey aquatic plants using divers (e.g., LakeSPI) At 4 fixed transects record: • Maximum depth of plants • Native species distributions and % cover • Presence and cover of non-native species • Health of plants	Every 5 years; Scuba divers (e.g. LakeSPI methodology)

Priority 1a. monitoring covers the basic water quality parameters at 2 sites and 6 depths on a monthly basis. This protocol accounts for spatial variation in the lake by sampling 2 sites and for vertical variation by sampling 6 constant depths and obtaining some more detailed information using a profiling CTD datasonde. It recommends sampling the phytoplankton and zooplankton communities, which provides valuable information on subtle changes in biological communities that can be related to subtle environmental changes. Currently the Otago Regional Council is using a monitoring programme similar to this, but at a single site. The Otago Regional Council monitoring effort has not been consistent on an annual basis, instead involving a cyclical monitoring rotation in which the lake is monitored for three years and then not monitored for a number of years. A regular, annual and long-term monitoring commitment for lakes yields better datasets and leads to a better understanding of lake behaviour and management requirements.

Adding priority 1b. measurements of total P in the inflows and outflows of the lake would provide the data required to calculate a phosphorus budget for the lake. This would indicate the internal and external P loads to the water column and the amounts of P lost via the outflow and via permanent burial in the sediments. With this information, it would be possible calculate a P budget (e.g., Robertson 1988) to determine whether the lake is a sink or source of P in a particular year, enabling the tracking of how the lake is responding to historical and current P loads.

Priority 2 monitoring provides high frequency vertical profiles of selected variables. This would allow the timing of various important events such as stratification/destratification to be identified. In addition, the timing and position in the water column where temperature and DO thresholds are exceeded can also be identified and compared across years. This would provide valuable background information for interpreting various stressors and drivers of changes in the lake.

Priority 3 monitoring focuses on the important littoral (submerged plant) zone of the lake, which is the primary habitat for many important aquatic organisms. It would allow for the determination of the health and biodiversity of the aquatic plant community, the distribution of plants in relation to water depth (indicating whether the plant community is shallowing, stable or deepening in the lake), as well as the status of non-native aquatic plants in the lake. The Otago Regional Council has had such aquatic plant surveys carried out in 1992 and 2001, but to our knowledge, these have not been carried out since. As a result, we have little understanding of how *Ceratium* blooms have affected plant distributions and associated fish habitat in the lake.

The comprehensive monitoring programme described here would provide information on lake health at different spatial and time scales, enabling the careful monitoring of changes in algal blooms, dissolved oxygen, trophic state (nutrient status), climate-related effects, phytoplankton, zooplankton, aquatic plant communities and invasive species within these communities. It would also provide data required to calculate nutrient budgets for the lake at monthly, annual and longer time scales. Together these monitoring strategies would be capable of sensitively tracking changes in the lake and would also provide valuable background information for targeted research on lake ecology. Finally, long term data are extremely valuable in understanding lake functioning and trends and. Therefore, there should be a commitment to continue regular monitoring over the long term, without hiatuses.

6 Acknowledgements

We are very grateful to the many people who provided many kinds of support to our work on Lake Hayes. We thank Kerry Dunlop, Richard Bowman, Rob Hay and other members of the Friends of Lake Hayes Society for funding this work, for logistical support, and for sharing their passion for the lake. We thank Professor Carolyn Burns for her deep insights and long-term commitment to understanding Lake Hayes. John Quinn, Max Gibbs and Chris Hickey from NIWA have been supportive collaborators in much of our work on Lake Hayes and we particularly thank them for their work on the alum dosing calculations and costings (Appendix 3). Rachel Ozanne and Adam Uytendaal have been very helpful in providing Otago Regional Council data for analysis in this report. MS thanks his postgraduate students and interns, Tina Bayer, Helen Trotter and Ciska Overbeek, who have developed their scientific skills and insights and helped uncover some of the mysteries of Lake Hayes during their studies on the lake. We thank Otago Fish & Game who have been generous to us both financially and in terms of providing field assistance. We also thank the University of Otago for support. Carolyn Burns, David Hamilton and John Quinn kindly provided feedback on an earlier version of this report.

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Appendix 1: Food web biomanipulation as a restoration tool for Lake Hayes

Food web cascades in lakes

Studies overseas and in New Zealand lakes have found that lakes can undergo rapid changes in water quality if nutrient loading is pushed beyond certain tipping points and when invasions by non-native fish and aquatic plant species cause major changes in lake food webs and nutrient cycles (Scheffer 2004; Schallenberg & Sorrell 2010). Once in a degraded, eutrophic state, lakes can exhibit inertia and resistance to restoration by nutrient reduction alone (Fig. 1).

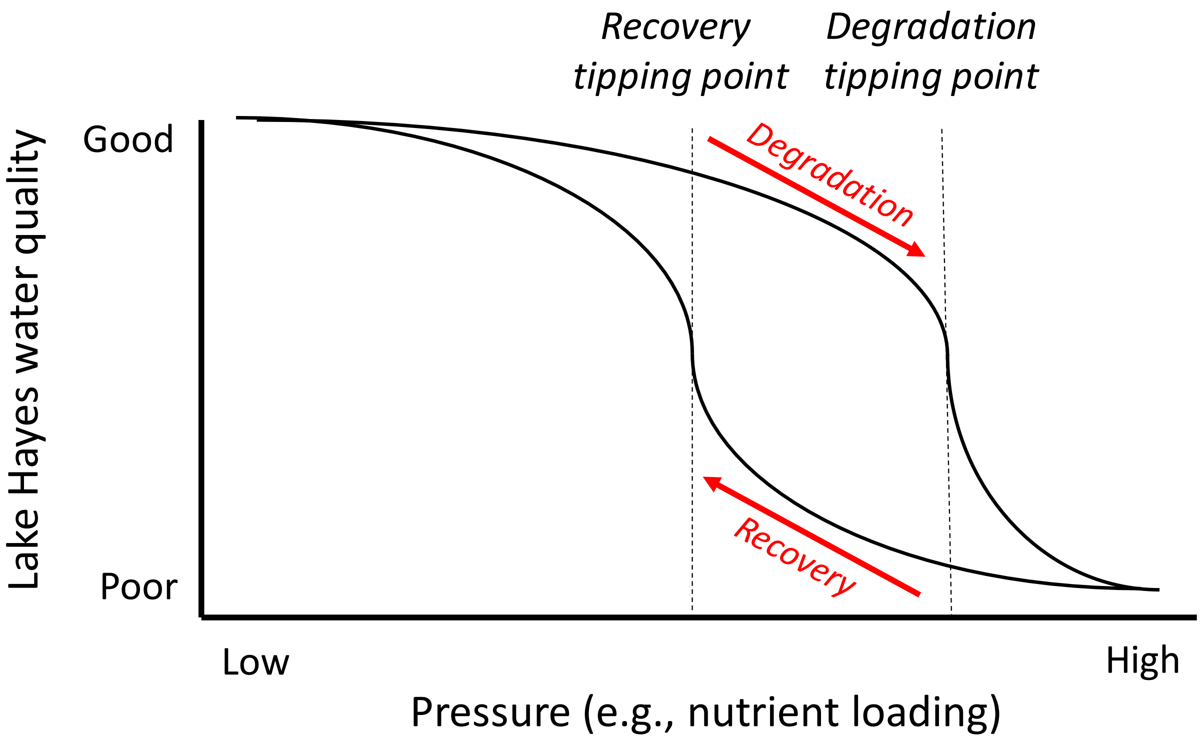

Figure 1. Idealised, non-linear relationship between water quality and environmental pressures in lakes, showing tipping points and inertia to degradation and to recovery that are typical of many shallow lakes.

However, in concert with nutrient load reduction, food web manipulations (fish stocking/removal) have been shown to successfully assist a return to a clear water state, usually by releasing zooplankton (which graze on algae) from predation pressure by fish (Søndergaard et al. 2007). Such food web biomanipulation aims to initiate, or strengthen, a top-down trophic (food chain) cascade by reducing zooplanktivory, resulting in an increase in the abundance and size of zooplankton (Shapiro 1980; Burns et al. 2014). If the zooplankton is dominated by large species (e.g. *Daphnia* sp.), increased grazing pressure on phytoplankton and ultimately increased water clarity result from a successful biomanipulation of the lake food web (Reynolds 1994) (Fig. 2). A cascade to reduce algae biomass may focus on increasing piscivorous fish density and/or decreasing zooplanktivorous fish density.

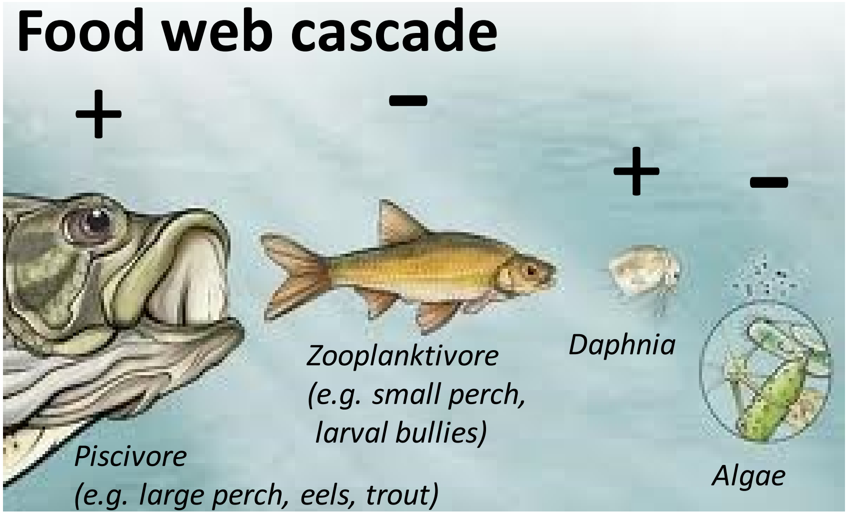

Figure 2. Example of a food web cascade showing how increasing the density of piscivorous fish can cascade (left to right) to reduce densities of algae.

Evidence for a link between *Daphnia* persistence in summer and the inhibition of *Ceratium* blooms in Lake Hayes

In the summers of 2009/10 and 2016/17, Lake Hayes exhibited unexpected, rapid improvements in water clarity and quality. Studies by the University of Otago Zoology Department revealed that the invasive zooplankter, *Daphnia 'pulex'* (Fig. 3), had colonised the lake, probably in the mid-2000s, and was attaining higher densities than had been previously recorded when the native *Daphnia thomsoni* was the sole *Daphnia* species in the lake (Fig. 3). Work by Professor Carolyn Burns of the Zoology Department has shown that *Daphnia 'pulex'* has higher temperature preferences than *Daphnia thomsoni* (called *D. carinata* in her study) and reaches higher densities than the native species (Burns 2013).

Figure 3. *Daphnia 'pulex'* from Lake Hayes, February 2010. Photo: Ciska Overbeek.

High *Daphnia* densities in 2010 occurred at the time when Lake Hayes attained and sustained unusually high water clarity throughout the summer (Fig. 4; Main Report - Fig. 5), suggesting a link between *Daphnia* summer density and summer algal biomass in the lake. In support of this apparent link between high *Daphnia* density and clear water in summer, we have observed that in the summers of 2012/13 and 2015/16, when *Ceratium* blooms occurred, *Daphnia* were absent in the lake during summer. In addition, zooplankton samples collected during the summer of 2016/17 (another summer with unusually high water clarity in the lake, Main Report – Fig 5), *Daphnia* continued (densities not yet determined).

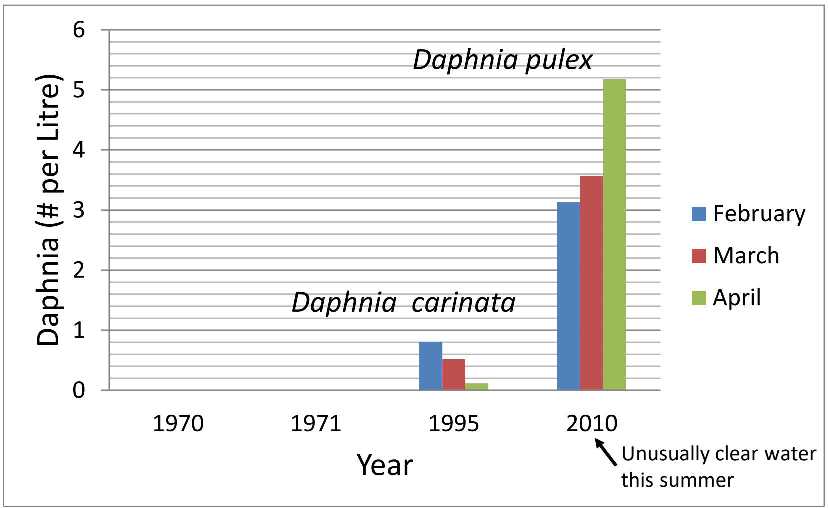

Figure 4. *Daphnia* densities in summer months in Lake Hayes in 1970 and 1971 (no *Daphnia* present in summer), 1995 (low densities present in summer), and 2010 (high densities present in summer).

The potential role of small perch in determining water quality

The clear water summer of 2009/10 did not persist into the summer of 2010/11. Observations of high juvenile perch numbers in 2010/11, suggest that perch recruitment increased in response to the plentiful *Daphnia*, and it is hypothesised that *Daphnia 'pulex'* density (and grazing on phytoplankton) was suppressed by predation on *Daphnia* by juvenile perch. These findings have prompted further investigations into the potential of biomanipulation as an approach to improve the water quality of Lake Hayes.

To test this hypothesis and to confirm that biomanipulation of *Daphnia* densities could facilitate a switch to a clear water state, studies by the University of Otago Zoology Department and collaborators from other research institutes and universities are underway to determine the strength of food web interactions between fish, zooplankton and phytoplankton in Lake Hayes. The relative importance of nutrients versus the food web structure in controlling phytoplankton abundance is being examined in order to evaluate the potential for food web manipulation to facilitate a shift back to a stable clear water state in Lake Hayes.

While declining water quality in New Zealand has attracted many millions of dollars of clean up funds for lake restoration, biomanipulation of the pelagic food web has only been attempted in one other New Zealand lake, the Lower Karori Reservoir, Wellington (Smith & Lester 2007). In this very small reservoir, the removal of perch resulted in enhanced zooplankton densities, reduction in algal biomass, and improved water quality (Burns et al. 2014), suggesting that reducing predation of small perch on *Daphnia* in Lake Hayes may help improve water quality in the lake.

Ideally, the lake food web models produced from this research will provide confidence in applying a biomanipulation approach in the lake. Key information needed to develop a successful biomanipulation approach includes:

1. Which fish species in the lake are responsible for reducing summer *Daphnia* densities, and at what size and life stage does predation on *Daphnia* occur?
2. How might *Daphnia* reduce *Ceratium* biomass? The large size of *Ceratium* in relation to *Daphnia* food size preference suggests that the interaction between *Ceratium* and *Daphnia* is not a direct grazing effect of the zooplankter on the algae. Rather *Daphnia* grazing on bacteria may reduce bacterial prey available for *Ceratium*. Alternatively, strong *Daphnia* grazing on other algae may increase light penetration and oxygen production in the deeper waters of the lake, restricting the upward flux of phosphorus toward the thermocline. There are other possible interactions.
3. What are cost-effective approaches for reducing zooplanktivorous fish densities (i.e., young perch) in spring and early summer?

Helen Trotter is currently doing an MSc thesis study to answer some of these questions. Her preliminary data from 2015/16, show that the recruitment of perch in early summer coincided with a sharp decline and eventual disappearance of *Daphnia* from the lake by January (Fig. 5). When her study is completed (end of 2017), sufficient information will be available to help produce a feasibility study for a biomanipulation intervention to cause a favourable food web cascade in Lake Hayes.

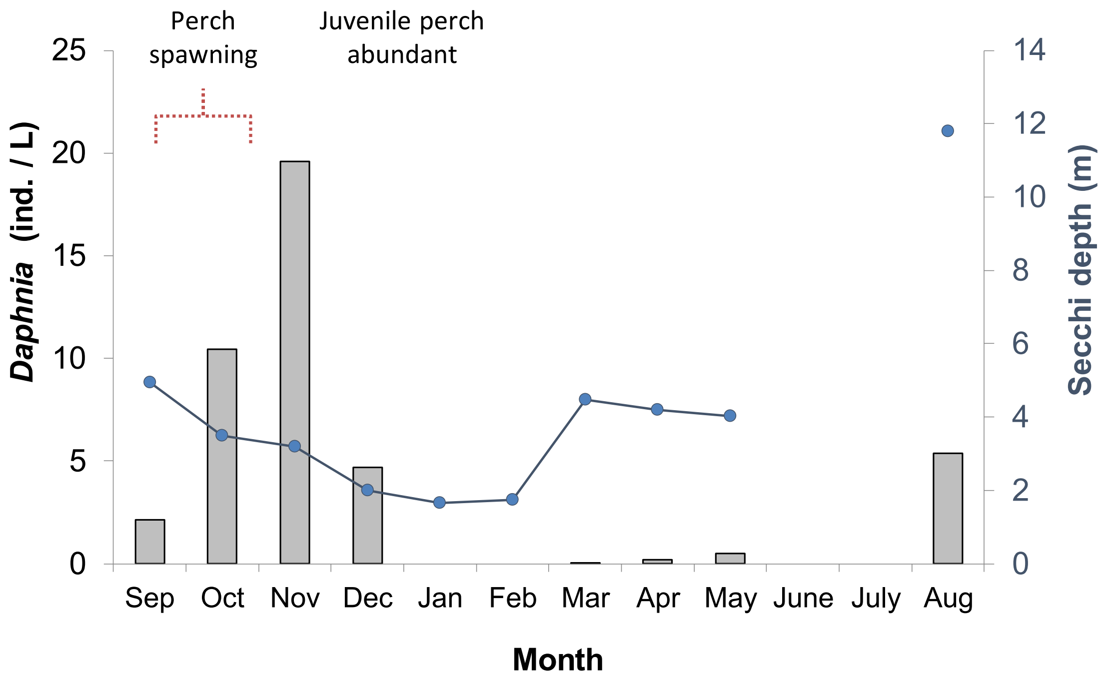

Figure 5. *Daphnia* density (bars) and Secchi disk depth (line) during the period September 2015 to August 2016. Note that when juvenile perch become abundant in December and January, the *Daphnia* decline and disappear from the lake, resulting in a decrease in water clarity to around 2 m. Data from Helen Trotter, University of Otago.

Potential ways to stimulate an effective food web cascade

Below we briefly introduce some potential approaches that could be used to induce a favourable food web cascade in Lake Hayes. These approaches could be further tested with respect to cost-effectiveness once Helen Trotter's study is complete.

1. **Adding *Daphnia* to the lake**. This is not seen as a practical option because of the large numbers of *Daphnia* that would be needed and the size of the facility that would be required to breed up such high numbers of *Daphnia*. *Daphnia* are intense grazers and require large amounts of food, further challenging the ability to produce the numbers that would be needed to affect algae in the lake. Furthermore, unless controls were also placed on planktivorous fish in the lake, the addition of large numbers of *Daphnia* would simply provide more food for planktivorous fish.
2. **Removal of juvenile perch**. This is not seen as practical because of the large size of the lake and because of the presence of aquatic plants in the shallow zones of the lake, which would make large-scale netting ineffective.
3. **Stocking of piscivorous trout and eels**. This may be a cost-effective and practical option. Fish to be stocked should be large enough to no longer feed on *Daphnia*, but instead feed on small fish (zooplanktivores). Fish & Game Otago may be able to raise hatchery brown trout to contribute to such a project. Longfin eels would have recruited into Lake Hayes prior to the construction of the Roxburgh Dam (1957). Subsequent to the dam, recruitment of eels, and eel biomass will have been severely or even completely reduced. Working with Contact Energy to trap and transfer returning eels from the Roxburgh Dam to Lake Hayes may have some merit because eels are effective piscivores capable of preying on young perch.
4. **Rearing and stocking of large, piscivorous sterile perch**. Perch longer than c. 150mm are known to be piscivorous. In some fish species (and possibly also in perch), it is possible to induce sterility by managing temperature of the developing ova (eggs). It may be possible to rear sterile perch to a large enough size so that they shift from zooplanktivory to piscivory. Stocking sterile piscivorous perch into Lake Hayes could reduce juvenile perch numbers through cannibalism, which is common in perch.
5. **Removing perch ova (eggs) from the lake**. Artificial perch spawning substrates could be built and deployed in the lake to attract perch spawn. Once spawning time is over but before egg hatching, the substrates could be removed, reducing perch recruitment. This strategy could complement strategy 4, by providing perch eggs for rearing into piscivorous perch.
6. **Encouraging fishing of perch**. Encouraging the catching of perch could possibly reduce perch numbers in the lake. If spawning perch could be targeted, this could potentially reduce perch recruitment. However, removing large perch from the lake could induce stunting in the population, which could increase predation pressure on *Daphnia* (see section 1.3.2. of the main report).

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Appendix 2: A preliminary assessment of the potential for augmentation of the inflows of Lake Hayes with Arrow River irrigation water to speed the recovery of the lake

This Appendix is updated from: Schallenberg M. (2015) A preliminary assessment of the potential for augmentation of the inflows of Lake Hayes with Arrow River irrigation water to speed the recovery of the lake. University of Otago Limnology Report No. 18, prepared for the Friends of Lake Hayes. Oct. 30, 2015.

Background

Lake Hayes usually undergoes thermal stratification from September to May or June. During this period, the warmer surface water is separated from the denser, colder water at the bottom of the lake. Due to the breakdown of algal material which settles to the bottom, the oxygen content of the bottom water declines during the stratified period, with the lake bed beginning to become anoxic in December to January (Fig. 1).

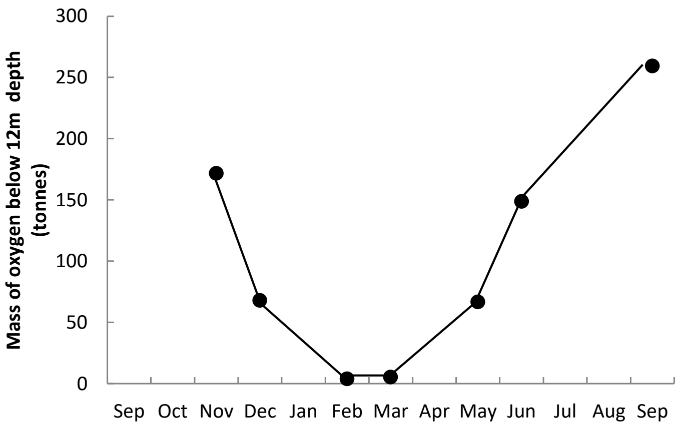

Figure 1. Mass of oxygen in the bottom waters (below 12m) of Lake Hayes, summer 2012/13.

As this occurs, phosphorus, which is bound to the sediments when oxygen is present, becomes liberated from the sediment and diffuses into the bottom waters and accumulating there until the end of the stratified period (Fig. 2).

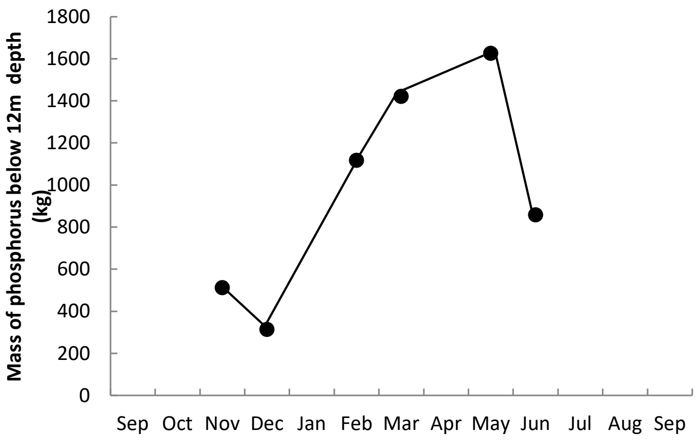

Figure 2. Mass of phosphorus in the bottom waters (below 12m) of Lake Hayes, summer 2012/13.

Stratification usually breaks down in June when the lake again mixes from top to bottom and phosphorus is diluted and also re-bound to particles in the water column.

The Friends of Lake Hayes have been examining potential methods for restoring Lake Hayes. A proposal has been put forward to help speed the recovery of Lake Hayes by augmenting the inflow to the lake at Mill Creek with water from the Arrow River Irrigation Scheme, sourced from the Arrow River near Macetown (Fig. 3).

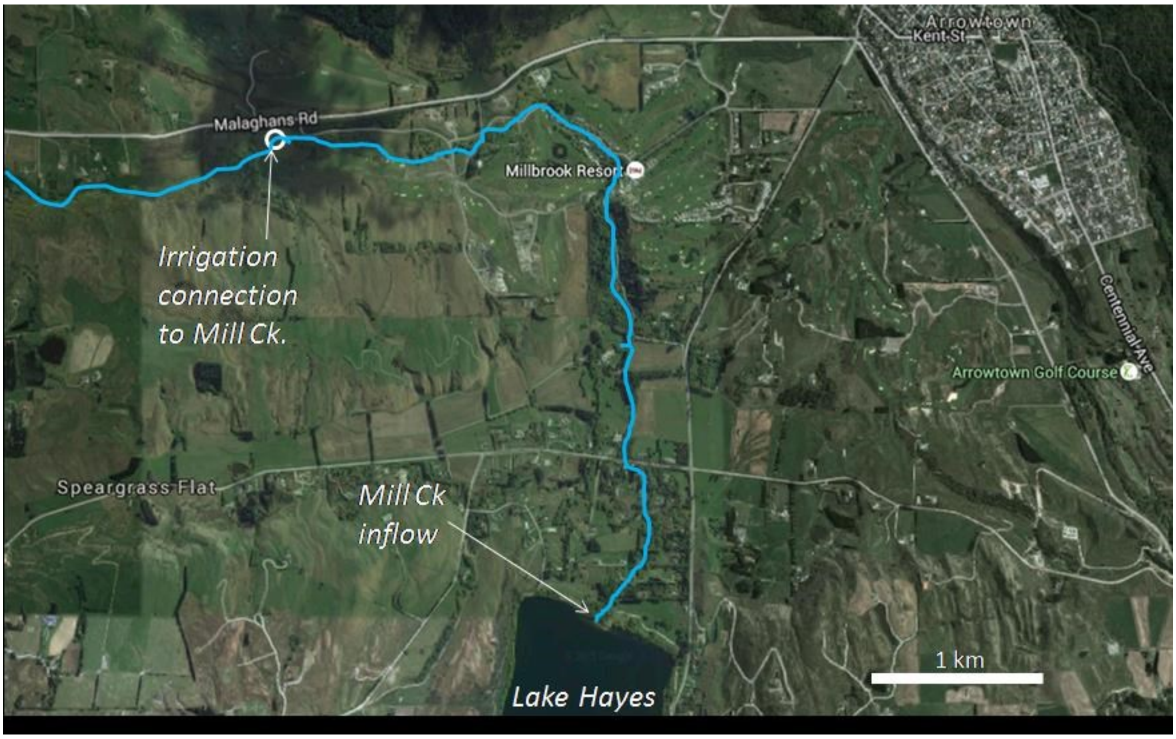

Figure 3. Map of the Lake Hayes area, showing Mill Creek and the potential connection point of the Arrow River Irrigation Scheme.

In this preliminary report, I use available data to try to answer four key questions regarding this potential restoration idea: 1. Could the augmented inflow flush displace substantial amounts of water from the lake? 2. Would the augmentation flow displace bottom water? 3. Could the augmented inflow supply enough dissolved oxygen to the bottom water to prevent its deoxygenation and, thereby, prevent P release from the sediments? 4. How much P and chlorophyll *a* could the augmented flow flush from the lake and what effect would this have on trophic state?

1. Could the augmented inflow displace substantial amounts of water from the lake?

This proposal would increase the flushing of the lake, which currently replaces its water roughly every 1.8 years (Caruso 2000). If the Arrow River water is more dilute than the lake water (with respect to phosphorus), then the flushing effect could remove some of the recycled phosphorus from the lake by displacement. The magnitude of the enhanced flushing effect would be proportional to: 1. the difference in nutrient concentrations between the Arrow River and the lake water that it displaces and 2. the amount of water available for flushing.

Recent ORC data show little difference in TP in surface and bottom waters (see main report, Fig. 8). Below, I have examined how beneficial the augmented flow could be for flushing phosphorus from the lake.

For these calculations, I have used the following information:

1. Available Arrow River flows: 200 litres per second for September, October, April, May and June. 100 litres per second for November to March (inclusive) (Table 1)

2. Arrow river phosphorus concentrations (Otago Regional Council data; Table 2)

3. Lake temperature profiles (University of Otago; Fig. 4)

4. Lake phosphorus concentrations from summer 2012/13 (University of Otago)

Table 1: Available water from the Arrow River Irrigation Scheme (info provided by Rob Hay).

Month	Cubic m per day	Cubic m per month	Cumulative irrigation inflow
Sept	18000	540000	540000
Oct	18000	540000	1080000
Nov	9000	270000	1620000
Dec	9000	270000	1890000
Jan	9000	270000	2160000
Feb	9000	270000	2430000
March	9000	270000	2700000
Apr	18000	540000	2970000
May	18000	540000	3510000
June	18000	540000	4050000

Table 2. Typical phosphorus concentrations of the waters of Lake Hayes (University of Otago) and the Arrow River (Otago Regional Council data from site at Morven Ferry Rd.).

Month	Lake Hayes surface water TP (µg/L)	Arrow River TP (µg/L; ORC data*)
Nov	27	14
Dec	52	9
Feb	47	7
March	116	8
May	43	9
June	69	5

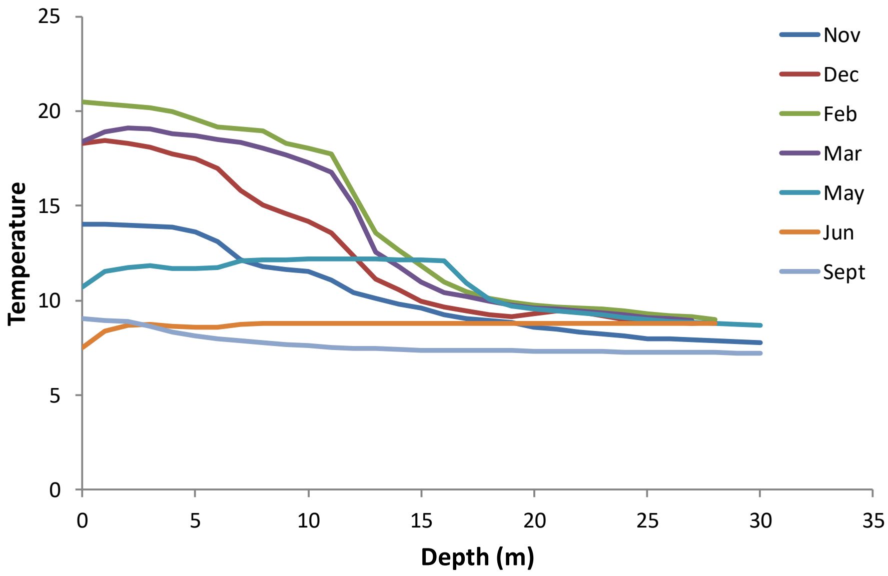

Figure 4. Lake temperature profiles from the summer of 2012/13.

Table 3 shows that the P concentration in the Arrow River is much lower than the lake P concentration of the lake, indicating that the Arrow River water would be suitable for the dilution and displacement of P-rich lake water.

Using the above information, I calculated the cumulative input of Arrow River water from September to June and compared that with the lake volume. I calculated this cumulative flushing volume as a percent of the whole lake volume.

The calculations show that the flushing effect of the Arrow River augmented inflow would displace a small percentage of the lake volume – only approximately 7% of the whole lake volume by the end of the stratified period (Fig. 5).

While these flushing effects are not substantial, they are not insignificant and could, over many years help reduce lake P concentrations and recycling. However, the addition of Arrow River water to Mill Creek could increase the loading rate of dissolved inorganic nutrients to Lake Hayes during the summer months (DSIR 1973), when these are often in very low concentrations in the surface water of the lake. Therefore, even though the concentrations of added nutrients from the proposed augmentation might be small, and the net P and N balance might be negative, it is possible that the addition of small amounts of available N and P could have a somewhat stimulatory effect on the lake's phytoplankton during summer. Therefore, before this restoration method is employed, further thought should be given to this potential stimulatory effect.

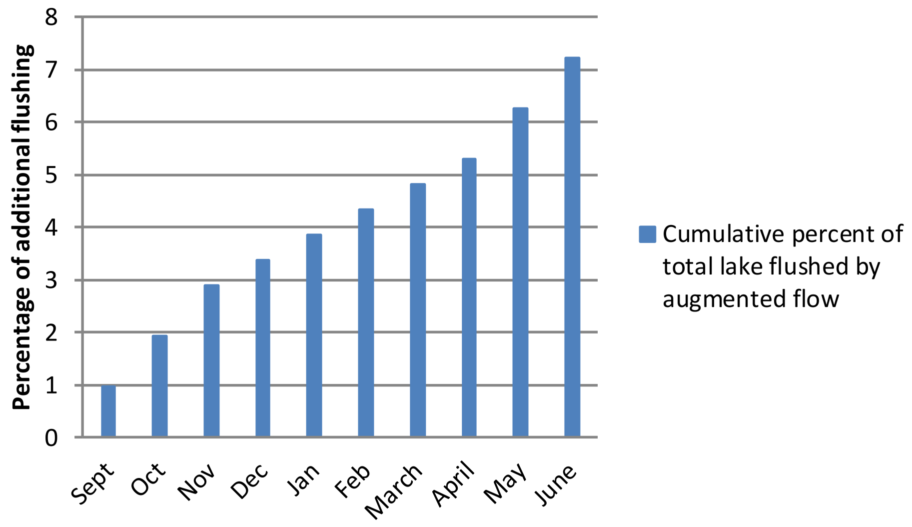

Figure 5. Cumulative proportion of the total lake water that could be flushed by Arrow River water, using the maximum amount of augmentation water available (200 L/s in shoulder seasons and 100 L/s in summer).

2. Would the augmentation water displace bottom water?

The colder the water, the denser it is (this is true down to 4°C). So, to displace the colder bottom water of Lake Hayes, the combined Mill Creek/Arrow River inflow would have to be colder than the surface layer of the lake and, ideally, it should be as cold/dense as the bottom water of the lake.

For these calculations, I have used the following information:

1. Available Arrow River flows: 200 litres per second for September, October, April, May and June. 100 litres per second for November to March (inclusive) (Table 1)

2. Lake temperature profiles (University of Otago; Fig. 4)

3. Mill Creek temperatures (Otago Regional Council data; Fig. 6).

I have assumed the following for these calculations:

1. The combined Mill Creek/Arrow River inflow would be the same temperature as the current Mill Creek inflow.

To test whether the inflow would be likely to plunge to the bottom layer of Lake Hayes, I compared the temperatures of Mill Creek with the temperatures of the lake, over the stratified period (Fig. 6). The data show that only toward the very end of the stratified period (in May), does the temperature of Mill Creek. approach that of the bottom water of the lake. Prior to that time, the inflow would either flow into the warm surface water or would flow between the layers (but not enter the bottom water layer).

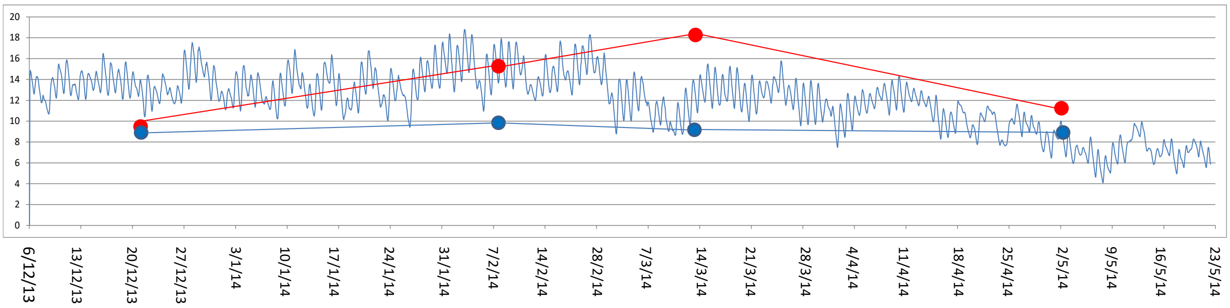

Figure 6. Temperature data for Mill Creek (blue line; 2013/14) and Lake Hayes (blue and red lines with dots; 2012/13). The blue dots show the lake bottom water temperatures and the red dots show the lake surface water temperatures. Mill Creek data were supplied by the Otago Regional Council.

Addressing the above assumption, is it possible that the temperature of the Arrow River augmented flow might lower the temperature of Mill Creek enough to allow both volumes of water to plunge into the bottom of Lake Hayes? Unfortunately, we don't have temperature data for the Arrow River at the offtake site or at the site where the irrigation water would connect to Mill Creek. This connection site is 4 km upstream from where Mill Creek enters Lake Hayes (Fig. 3), so even if the Arrow River water were substantially colder than Mill Creek, by the time it was transported from near Macetown to the Mill Creek connection site, diluted by Mill Creek and then transported 4 km downstream, any temperature benefit from the Arrow River is likely to have been lost. However, I have not been able to confirm this with data or modelling.

3. Could the augmented inflow supply enough dissolved oxygen to the bottom water of Lake Hayes to prevent its deoxygenation?

Another potential benefit of the injection of Arrow River water into the bottom waters of Lake Hayes is that the addition of oxygenated Arrow River water to the bottom waters of the lake might prevent deoxygenation of the bottom waters, maintaining P binding in the sediment of the lake.

For these calculations, I have used the following information:

1. Available Arrow River flows: 200 litres per second for September, October, April, May and June. 100 litres per second for November to March (inclusive) (Table 1)

2. Lake temperature profiles (University of Otago; Fig. 4)

3. Estimates of the volume of 1 m-thick slices of Lake Hayes (calculated from the NZ Oceanographic Institute bathymetric chart)

I have assumed the following for these calculations:

1. The combined Mill Creek/Arrow River inflow would discharge into the bottom waters of Lake Hayes
2. That the combined Mill Creek/Arrow River inflow would have an oxygen content approximating 100% air saturation (i.e., equilibration with the atmosphere).

For these calculations, I cumulatively added the mass of oxygen that would exist in the Arrow River augmented flow over the period for which water would be available. This mass of oxygen was then compared to the mass of oxygen in the bottom waters of Lake Hayes during the same period (the stratified period). Figure 7 shows that the cumulative input of oxygen is only relatively minor compared to the oxygen holding capacity of the bottom waters of Lake Hayes (indicated by the September value, when the bottom waters were mostly oxygenated). The rate of oxygen supply to the bottom waters (the slope of the line = 0.0888 tonnes of oxygen supplied per day) is also small compared with the rate of oxygen loss from the bottom waters in spring and summer (from November-February; 1.93 tonnes of oxygen consumed per day). Thus, the rate of oxygen consumption in the bottom water is 22 times greater than the rate of oxygen supply which could be contributed to the Arrow River augmentation, if it were injected directly into the bottom waters. This indicates that injecting the Arrow River augmentation flow directly into the bottom waters would not overcome deoxygenation in this lake.

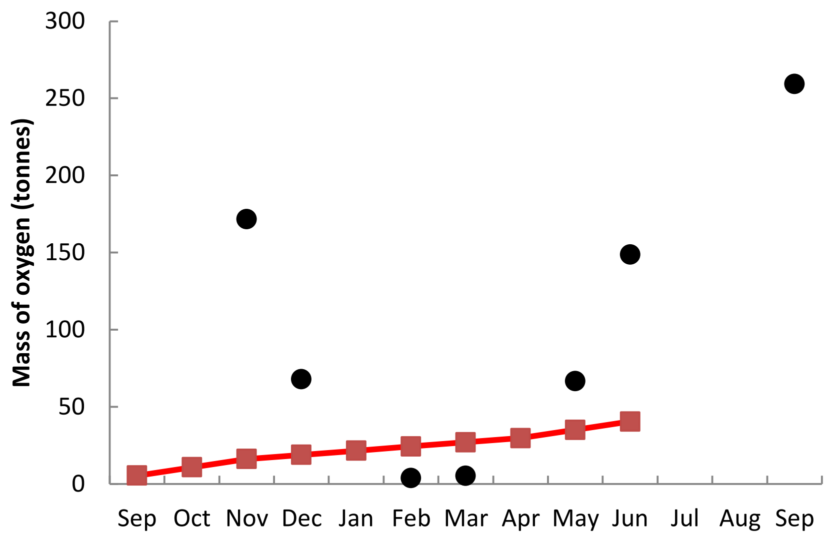

Figure 7. The mass of oxygen in the bottom water of Lake Hayes (2012/13; black dots) and the cumulative mass of oxygen estimated to be in the proposed augmented Arrow River inflow (red squares).

3. How much P and chlorophyll a could the augmented flow flush from the lake and what effect would this have on trophic state?

Displacement of surface water:

It appears from the above analysis in Section 2 that the augmented flow from the Arrow River would largely flow into the upper surface water layer of Lake Hayes. I calculated the amount of lake surface water that would be displaced by the cumulative input of Arrow River water from September to June. The volume of the surface water layer (to 12 m depth) is 31.03 million cubic metres, and the cumulative inflow from the Arrow River is 4.05 million cubic metres by the end of June. Thus, the Arrow River would displace around 13% of the lake's surface water over the stratified period.

Displacement of total phosphorus:

The average total phosphorus concentration in the surface water of Lake Hayes from September to June is 59 mg/m^3, while that in the Arrow River (at Morven Ferry) is 9 mg/m^3 (Table 2). The difference in concentration is 50 mg/m^3. When multiplied by the volume of the lake's surface layer and by the cumulative inflow from the Arrow River, respectively, the phosphorus in the lake displaced by the augmented flow would equal approximately 11% of the phosphorus content of the surface layer of the lake. This would bring the average phosphorus concentration in the surface water down from 59 mg/m^3 to around 52.5 mg/m^3, by the end of the augmentation period in June. The lake's trophic state would remain high as the boundary between mesotrophic (moderately productive) and eutrophic (productive) is 20 mg P/m^3. By these estimates of the average augmented lake phosphorus concentration, the lake would remain in the supertrophic category (48 – 96 mg P/m^3) (see Appendix 2.1). However, persistent flushing of this sort over a number of years could contribute to an improvement of the lake's trophic state.

Displacement of chlorophyll a (algal biomass):

The average chlorophyll *a* content of the surface water of Lake Hayes from September to June is estimated to be around 30 mg/m^3 (Bayer & Schallenberg 2009). We have no chlorophyll *a* data for the Arrow River, but this is expected to be quite low during moderate to low flow periods (probably not more than 2 mg/m^3 of chlorophyll *a* during the augmentation period). Again, multiplying by the volume of the lake's surface layer and by the cumulative inflow from the Arrow River, respectively, the chlorophyll *a* in the lake displaced by the augmented flow would equal approximately 12% of the chlorophyll *a* content of the surface layer of the lake. This would bring the average chlorophyll *a* concentration in the surface water down from 30 mg/m^3 to around 26.7 mg/m^3, by the end of the augmentation period in June. The lake's trophic state would remain high as the boundary between mesotrophic (moderately productive) and eutrophic (productive) is 5 mg $Chla/m^3$. By these estimates of the average augmented lake chlorophyll *a* concentration, the lake would remain in the supertrophic category (12 – 31 mg $Chla/m^3$) (see Appendix 1). However, persistent flushing of this sort over a number of years could contribute to an improvement of the lake's trophic state.

Caveats

There are a number of caveats that should be considered before employing augmentation flow from the Arrow River to help flush and, thereby, restore Lake Hayes. For example, the increased flow discharge and velocity of Mill Creek could increase stream bed erosion and reduce nutrient attenuation by stream periphyton due to the more rapid descent of water downstream to the lake. This would have the effect of increasing sediment and nutrient loads to the lake. In addition, some of the costly augmented flow could be lost to aquifer recharge in the catchment, in effect reducing the desired flushing effect. A groundwater hydrologist could advise on the likelihood of this occurring. Furthermore, as mentioned above, the augmentation flow, although low in nutrient concentrations relative to the lake, would likely add dissolved inorganic N and P to the lake during the summer months, when these nutrients are in short supply. This could stimulate phytoplankton production.

Summary

In Table 3, I summarise the information presented in this report and I show some issues to consider regarding the findings of the report. The above caveats should also be carefully considered before augmentation flow is employed for lake flushing.

Table 3. Summary of findings assessing the potential for Arrow River augmentation to speed the recovery of Lake Hayes.

Augmentation questions	Answer	Things to consider
1. Would it flush a substantial amount of phosphorus from the lake?	• Up to 11% per annum	• If internal load increases again, this could be useful if it could displace bottom water.
2. Would it naturally plunge into the bottom waters or would it flow into the surface waters of the lake?	• Naturally, the inflow is likely to be less dense than the cold bottom water, meaning it will flow over top of the bottom water, displacing and flushing surface water only.	• This conclusion assumes that the combined Mill Creek/Arrow River inflow would not be colder/denser than the current Mill Creek inflow. Temperature data are lacking to test this assumption.
3. If it were injected into the bottom waters, could it supply enough oxygen to prevent the bottom water from losing all of its oxygen during the stratified period?	• No, the oxygen augmentation effect is small compared to the oxygen demand of the bottom waters of the lake.	• In the calculations, I didn't include the oxygen that could also be supplied by the Mill Creek inflow. • Assuming that the Mill Creek discharge is around the same as the Arrow River augmented flow, and assuming that Mill Creek flows could also be harnessed and injected into the bottom waters of the lake, then the oxygen supply rate that I calculated would be doubled. • Injecting both these inflows into the bottom waters would still be insufficient to prevent deoxygenation of the bottom waters because the oxygen demand is around 10 times greater than the combined supply rate would be.
4. Could the augmented flow displace substantial amounts of phosphorus and chlorophyll *a* from the lake?	• The augmented flow would reduce the average surface water phosphorus concentration in the period from September to June by 11% and the chlorophyll *a* concentration by 12%. Neither of these reductions would reduce the trophic status of the lake from its current supertrophic condition.	• Persistent flushing of around 11% of the phosphorus and 12% of the phytoplankton from the lake per year could contribute to a speeding of the lake's recovery if maintained for a number of years.

Acknowledgements

I thank Rob Hay (Friends of Lake Hayes) for providing information on the available flows from the Arrow River irrigation scheme and for other background information about the proposed augmented inflow. I thank Dean Olsen from the Otago Regional Council for providing data on Mill Creek and

Arrow River temperatures and water quality. The New Zealand Ministry of Business, Innovation and Employment have funded this work via a subcontract with NIWA.

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Appendix 2.1

Attribute ranges for different lake trophic levels. From Burns, N, Bryers, G, & Bowman, E (2000).Protocols for monitoring trophic levels of New Zealand lakes and reservoirs.Available from www.mfe.govt.nz.

Lake type	Trophic level	Chla ($mg\ m^{-3}$)	Secchi depth (m)	TP ($mg\ m^{-3}$)	TN ($mg\ m^{-3}$)
Ultra–microtrophic	0.0–1.0	0.13–0.33	31-23.5	0.84–1.8	16–34
Microtrophic	1.0–2.0	0.33–0.82	23.5-14.8	1.8–4.1	34–73
Oligotrophic	2.0–3.0	0.82–2.0	14.8-7.8	4.1–9.0	73–157
Mesotrophic	3.0–4.0	2.0–5.0	7.8-3.6	9.0–20	157–337
Eutrophic	4.0–5.0	5.0–12	3.6-0.7	20–43	337–725
Supertrophic	5.0–6.0	12–31	0.7-0.3	43–96	725–1558
Hypertrophic	6.0–7.0	>31	<0.3	>96	>1558

Appendix 3: A rough Lake Hayes alum dosing estimate

This Appendix is based on a report that was prepared by John Quinn, Max Gibbs and Chris Hickey (NIWA) in 2012.

The most common chemical method for capping phosphorus in lake bed sediment is to distribute alum (aluminium sulphate) solution into the lake, which flocculates and settles to the lake bed where it binds free phosphorus in the sediments, even under anoxic conditions. During the process of flocculation, alum also collects algae and suspended solids, clarifying lake water. Under conditions of pH > 6.5, alum can bind sufficient phosphorus to create conditions where restricted P availability limits algal growth. Alum applications have been successfully used in Lakes Okaro (Paul et al. 2008) and Rotorua (Hamilton et al. 2015) in the Bay of Plenty to reduce phosphorus concentrations in lake water.

Because of alum's flocculating capability, it is best used when the bottom waters are anoxic and phosphorus released from the sediments has accumulated to its maximum level (toward the end of the stratified period in Lake Hayes). As a sediment capping agent, the effect of a single appropriately-dosed treatment can last for 5 years and sometimes up to 20 years (Welch & Cooke 1999). Studies on toxicity of aluminium derived from alum to sediment-dwelling fauna and to fish indicate that as long as the lake pH buffering (i.e. alkalinity) is sufficient to preclude acidification, then toxic effects are minimal (Tempero 2015).

The least expensive way to deliver alum to lakes is to add it to inflowing tributaries. This will be most effective if the tributaries are colder than the lake water and carry the alum directly to the bottom waters where dissolved reactive phosphorus concentrations are highest and where phosphorus cycling is strongest. This is the approach used to deliver alum to Lake Rotorua (Hamilton et al. 2015).

Below is a rough calculation of the estimated cost of an alum treatment for Lake Hayes. The amount of P to be sequestered in this calculation is based on the maximum dissolved reactive phosphorus concentration at end of stratification in recent years (c. 300 ppb; M. Schallenberg, unpublished data) multiplied by the hypolimnetic (pertaining to the deep water layer) volume (28,937,495 m^3). This calculates a total hypolimnetic dissolved reactive phosphorus amount to be 8,681 kg, equating to 4.15 g P/$m^2$of hypolimnetic lake-bed area. When the top of the hypolimnion (deep water layer) is at 10m depth, around 78% of the lake bed is within the hypolimnion.

At a pH of 7 the aluminium-phosphorus binding is < 50%. If we conservatively assume 20% binding efficiency, then we will need an Al:P ratio of 5. Alum comes as 666.42 g/mol (octadecahydrate) containing 2mol of Al, which is 54g/mol, so the amount of alum needed is 535,343 kg. Alum comes in an aqueous solution of 47% alum, so the volume needed for the treatment of Lake Hayes would be 856,412 L- equivalent to ca. 40 standard (22,000L) water tanks full or ca. 20 tanker-trailer loads.

At a cost of alum solution at $1000/tonne, the cost of the alum for a Lake Hayes treatment under the above assumptions would be $535,343, which would provide an alum dose to the hypolimnetic sediments of 200 g alum per m^2 of hypolimnetic lake bed.

Dosing would occur during the stratified period (mid-October to May) by addition to the Mill Creek inflow stream which would ideally carry the alum into the bottom waters. Lake currents would provide further mixing of the alum floc around the lake. Studies carried out by NIWA in 2011/12 confirmed the existence of sufficient hypolimnetic currents to distribute the alum throughout the hypolimnion.

The lake is well buffered, so buffering the alum additions to avoid pH drops below 5 (which would cause concern regarding aluminium toxicity) would not be necessary.

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Appendix 4: Catchment management to restore and protect Lake Hayes

Importance of the Mill Creek catchment

Prior to major catchment drainage operations in 1961, there had been no reports of discoloured waters flowing into Lake Hayes and the bottom waters were aerobic during summer with little internal P release. After the wetland drainage and channelization works in the early 1960's, growing pressures from further catchment alterations and land use conversions continued and lake health further declined into the 70's and 80's. This prompted investigations into catchment nutrient loading and ways to mitigate the nutrient increases observed. The vast majority of nutrients enter the lake through the Mill Creek catchment (ORC 1995), with around 80% of the P load bound to soil particles which were historically mobilised through channel cutting and removal of the catchment wetlands which acted as sediment sinks. The high historical catchment P load had settled in the lake to become the internal P load which greatly contributed to the decline and maintenance of poor lake health seen over recent decades. Previous publications have outlined the importance of reducing nutrient concentrations in Mill Creek and noted this as a requirement for the successful restoration of Lake Hayes (Robertson 1988; ORC 1995).

Bayer & Schallenberg (2009) found nutrient levels in Mill Creek had decreased from the 1980's to 2009 however, more recent data (2005-present) available on the LAWA website, indicates the improving water quality trends in Mill Creek have since stabilised and may be, in some cases reversing. Fore example, summer nitrate and ammonia concentrations in Mill Creek appear to have been increasing since 2005 along with *E. coli* counts (Fig. 1). Compared with other upland rivers in New Zealand, levels of *E. coli*, turbidity, TN and nitrate are worse than the average upland river (LAWA 2017). The sensitivity and importance of this lake, together with the current and projected population growth rate and associated land use changes in the area, should recommend better than average water quality in the Mill Creek catchment.

Previous suggestions for catchment management

Robertson (1988) looked at the Lake Hayes catchment in detail, describing historical land use change and outlining potential mitigation measures that would lead to reduced inputs of P into the lake. Most of these recommendations involved controlling catchment landuse practices such as reducing fertiliser runoff, controlling channelling operations and managing future development in the catchment to ensure a reduced external P load to the lake. In addition, Robertson (1988) suggested many of what we now call on-farm best management practices (BMPs) such as reducing fertiliser use and establishing stream bank buffers. Improvement of land use practices in the Mill Creek catchment was recommended by the author to be highly important for the restoration of Lake Hayes.

In 1995 the Otago Regional Council (ORC) developed a Lake Hayes Management Strategy with the overall goal being to 'to improve the water quality of Lake Hayes, to achieve a standard suitable for contact recreation year round and to prevent further algal blooms' (ORC 1995). The strategy highlighted the major catchment issues affecting water quality and put in place relevant policies as well as outlined ambitious actions the ORC would take to reduce the P load in the catchment. Examples of these actions included negotiating with landowners around Mill Creek to establish riparian zones, advocating and assisting with the protection and re-establishment of wetlands and encouraging sustainable land use in the catchment, among many others. Some successes have since been documented, an example being the decommissioning of septic tanks in the catchment as encouraged in the strategy, however it is unknown how many actions have been carried out and how many policies have been implemented. Section 1.9 of the management strategy stated that reviews of the strategy will be undertaken at 5 year intervals, including an assessment of any changes in the catchment P load, however no notes on these can be found. It would be useful to document which policies and actions outlined in the strategy have been implemented and which are still required, to assess both the success of the management strategy to date and what remains to be done in the future, particularly in light of the rate of development in the catchment.

Wetland restoration/re-establishment

The two major catchment management directions actions recommended for Lake Hayes are wetland restoration/re-establishment, and on-farm BMP's (Robertson 1988; ORC 1995; Bayer 2009; Ozanne 2014). Wetland re-establishment was discussed by Robertson (1988) who noted the draining and channelization of numerous wetlands in the catchment during the 1960's led to a decrease in the sediment retention and buffering capacities of the Mill Creek catchment. The option for wetland restoration in the catchment was looked into to recover some of these lost ecosystem services and the cost was estimated at around $50,000 (Robertson 1988).

However, the Lake Hayes Management Strategy (ORC 1995) mentioned a report commissioned by the ORC looking into the viability of wetland re-establishment in the catchment which found it to be unfeasible. The main reasons given were the long calculated retention time required to settle out sediment bound P and the large areas required in order to reduce the catchment P load by a significant amount (ORC 1995). The study reportedly stated that the largest site available for wetland re-establishment was 93ha of land which was deemed non advantageous due to its position in the upper catchment. A review of the methodology of the report, particularly in identifying P loss hotspots in the catchment, and therefore how much can be removed by targeting different subcatchment areas, would be useful. Caruso (2001) filled some of these knowledge gaps by measuring P loads at multiple points in different subareas of Mill Creek to determine P hotspots, even identifying these down to the individual property level. Interestingly the author found that the O'Connell Creek catchment was a hotspot of P release (Figure 2) and it was suggested that the results of the investigation could inform more targeted mitigation actions in the catchment (Caruso 2001).

Best Management Practices (BMPs)

Best management farming practices include actions such as managing land for erosion and leaching, managing to minimise losses of sediment and nutrient to waterways, and stock exclusion from waterways. Ensuring on-farm BMPs are employed in the catchment is an obvious requirement for successful lake restoration. Using the results from Caruso (2001), a strategy for targeting BMPs, particularly in subareas or even on properties which are contributing high P loads would be highly beneficial. BMPs for lifestyle blocks, golf courses and activities that disturb ground in the catchment should also be communicated to relevant land owners and developers.

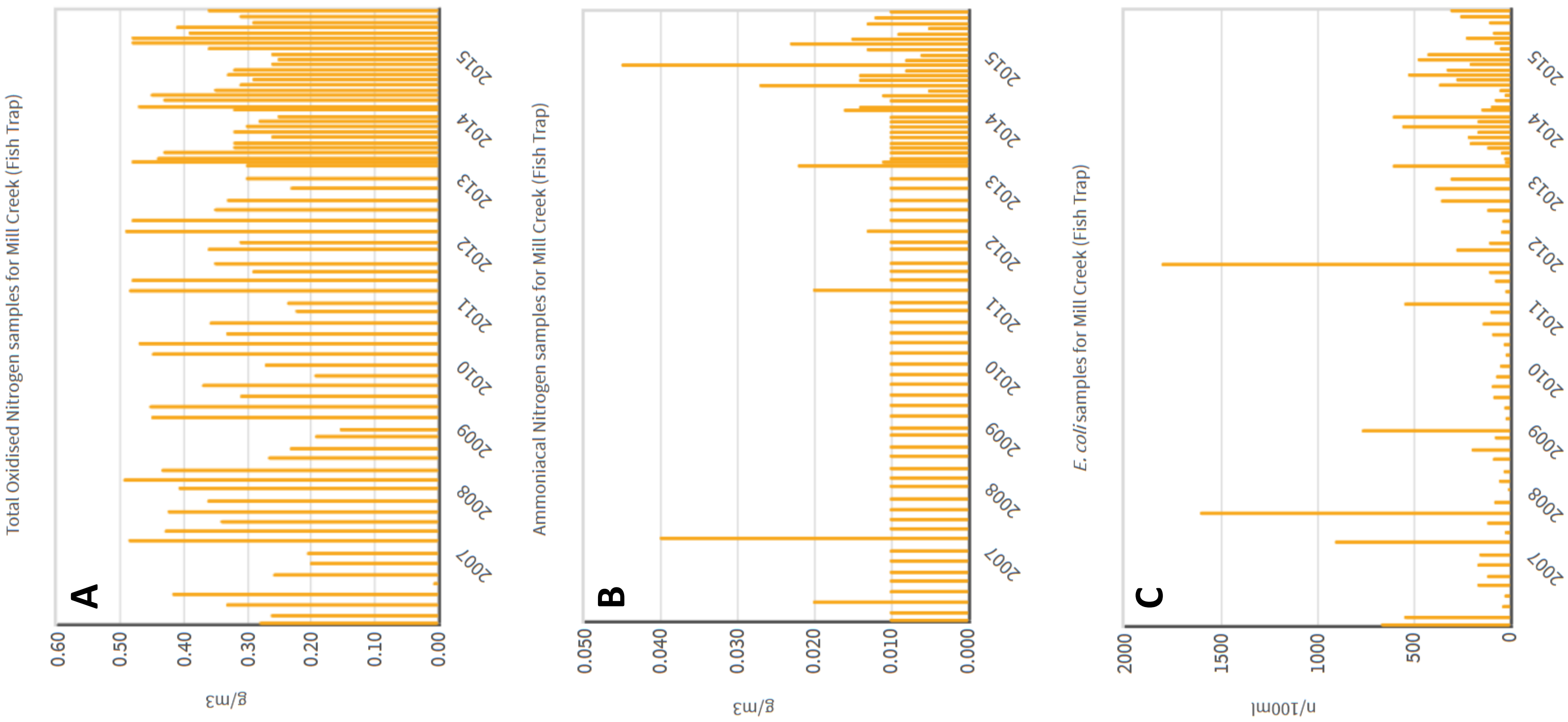

Figure 1. Nitrate (A), ammonium (B) and *E. coli* concentrations (C) in Mill Creek from 2006-present. Summer nitrate concentrations (the regular periods of lowest nitrate concentrations) appear to have been increasing over the sampling period (A). In addition, recent ammoniacal N (B) and *E. coli* concentrations (C) may also be increasing. From LAWA website (https://www.lawa.org.nz/explore-data/otago-region/river-quality/clutha-river/mill-creek-(fish-trap)/)

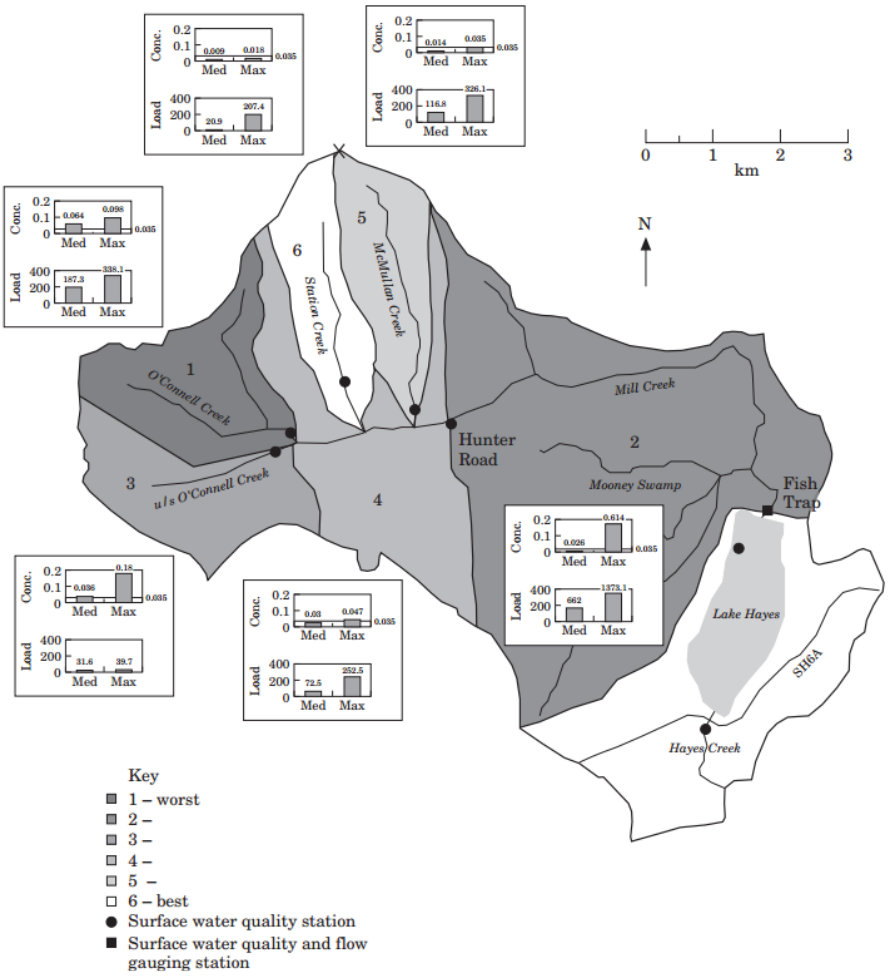

Figure 2. Results from Mill Creek catchment subarea phosphorus targeting in Caruso (2001), showing catchments ranked from highest to lowest P contribution. Taken from Caruso (2001).

A lake catchment management plan to ensure continuing reductions in nutrient and sediment losses from the catchment.

In order to ensure that landowners in the catchment minimise nutrient and sediment concentrations in streams and springs draining into Lake Hayes, a collaborative, community-driven lake and catchment management plan could be developed and implemented by undertaking the following:

i. Identify iwi, stakeholders, industries, scientists and other interested parties.
ii. Review the ORC (1995) Lake Hayes Management Strategy and its implementation.
iii. Undertake a catchment-wide N, P, sediment and *E. coli* survey based on the design of Caruso (2001) to identify current hotspots of contaminant contributions to the lake.
iv. Determine the feasibility of setting nutrient caps on the catchment.
v. Collaboratively develop a lake/catchment management plan with community participation at all stages.

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Economic Assessment of Lake Hayes Remediation

Report to Otago Regional Council

June 2018

Acronyms and Abbreviations

BCR	Benefit Cost Ratio
CBA	Cost Benefit Analyses
FoLH	Friends of Lake Hayes
LAWA	Land Air Water Aotearoa
NPV	Net Present Value
ORC	Otago Regional Council
PV	Present Value
QLDC	Queenstown Lakes District Council
TLI	Trophic Level Index

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Executive Summary

Lake Hayes (the Lake) is a small scenic lake, located in Central Otago, between Arrowtown and Queenstown, with views over the mountains in the Wakatipu Basin. It is a popular destination in the region for recreation, including walking, cycling and swimming.

The Lake has experienced an accumulation of phosphorous in the lake bed, most of which is historical, which can be responsible for feeding algal blooms. Algal blooms can affect the colour of the Lake, turning it a brown or greenish colour, and cause scums on the surface. Under certain circumstances types of algae can produce toxins which can cause rashes, nausea and be potentially deadly for dogs to drink.

Otago Regional Council (ORC) has been investigating remediation options to inhibit algal growth in the Lake, and have identified three potential intervention options. Castalia have been engaged by ORC to conduct an economic assessment of the remediation options.

How did we go about undertaking this economic evaluation of remediation options?

Three key steps were undertaken in this economic evaluation. They were:

- Determine what happens if there is no remediation
- Identify the costs and benefits that may occur if the remediation options are implemented, and identify which are economically material
- Quantify all the material benefits and costs and determine net benefits and benefit cost ratios.

Three options for remediation of the Lake have been identified

NIWA (2018) identified three remediation options that were worth assessing:

- Flushing
- Destratification
- Capping

Flushing involves increasing water inflow to improve the net balance of nutrients. Flushing is a medium-long-term option that creates additional inflow, and nutrient balance, but has a lower certainty of impact. It does not impose any noticeable side effects. Costs are estimated at $150k for implementation, with $30k per annum ongoing costs.

Destratification involves intervening in the Lake to prevent layers forming. The option of destratification uses a bubble plume across the middle of the Lake, powered by a compressor, to prevent stratification and eutrophication of the lower layers. The initial cost estimates for this option are in the range of $300k with ongoing operational costs.

A capping option involves chemicals being added to the Lake to cap the sediment layer for a period. Costs are highly uncertain for this option, ranging from $90k-$550k with longevity of 5-10 years. We used a high and low-cost scenario to understand the range of costs this option could impose.

The state of the Lake in the absence of remediation is uncertain

There is no clear scientific consensus on what will happen to the Lake without remediation. Some scientific opinions supports a natural recovery, as the catchment is in balance in terms of nutrient loads and flushing. Trophic level (water clarity) measurements are, however, worsening. Consultation with local residents and businesspeople supports a counterview that measurement is inadequate, nutrient loads may not be in balance, and not all inflows are understood.

For this analysis we investigate three counterfactual scenarios, to account for the uncertainty regarding what might happen to the Lake:

- **Stable** – the current state of water quality in the Lake continues, the trophic level remains the same. The current costs from poor water quality, e.g. cancelled triathlon, periods of no swimming, will continue on a cyclical basis.
- **Natural Recovery** – The Lake recovers naturally, the trophic levels decline (improve), the Lake becomes mesotrophic. The frequency of poor water quality events declines over time.
- **Deteriorates** – the state of water quality deteriorates further and begins causing further costs, e.g. an increase in the days unable to swim. The trophic level increases and the Lake becomes supertrophic. The water looks sludgy and loses all reflection etc.

Most of the remediation options are economically viable, with positive net present values under most counterfactual scenarios

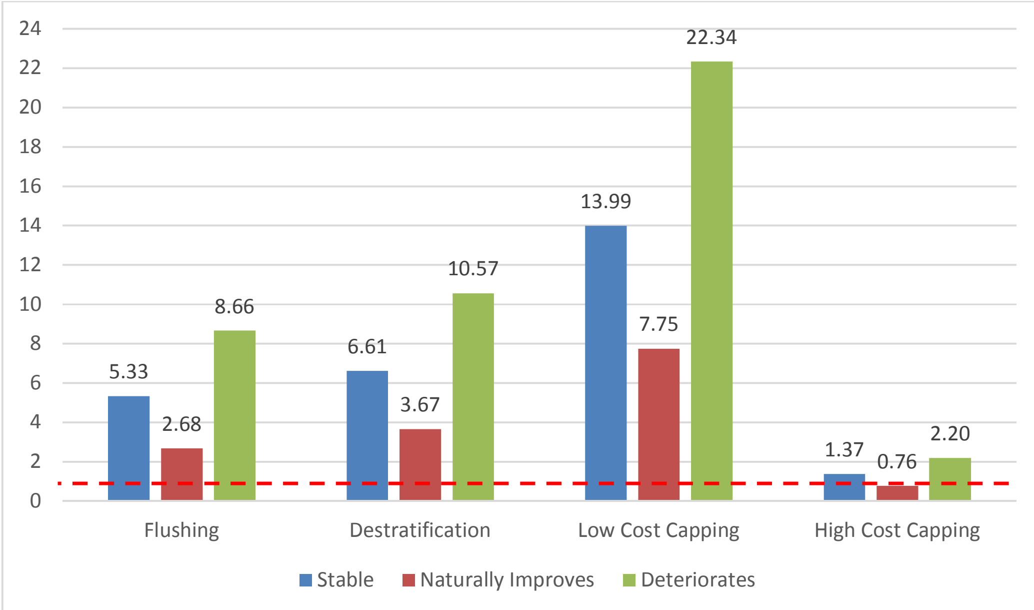
Table E.1 shows the Net Present Value (NPV) of (successful) remediation for the options of flushing, destratification and low and high cost capping, under the three counterfactual scenarios.

Table E.1: NPV of Successful Remediation

	Flushing	Destratification	Low Cost Capping	High Cost Capping
Stable	$1,612,000	$2,105,000	$2,302,000	$681,000
Natural Recovery	$625,000	$1,001,000	$1,197,000	-$423,000
Deteriorates	$2,848,000	$3,585,000	$3,782,000	$2,161,000

A positive NPV implies a benefit cost ratio (BCR) of greater than one, and that the option is economically viable. Figure E.1 shows the BCRs related to the above NPVs. The BCRs show that if all remediation options are successful, and capping is able to be implemented with the lowest estimated costs, it is the most economically viable option, followed by destratification.

Figure E.1: BCR of Successful Remediation



Remediation options have different side-effects and likelihoods of success that need to be considered

Flushing is predicted to have the least side-effects, but also the lowest chance of success. Destratification has been shown to be effective, but may create side-effects of noise, and visual impacts. Capping is potentially the most economically viable, but comes with important considerations about community views on adding non-natural chemical products to a freshwater lake.

There are four broad categories of activity that are impacted by Water Quality

These are:

- **Lake Based Recreation –** Water quality directly influences recreation activities at the Lake, such as swimming and sightseeing. This will include any regular scheduled events that take place at the Lake.
- **Local Business Sales–** Businesses sell products to lake visitors and this will include local food and wine venues, accommodation and tours. There is reputational overlap with local businesses that could impact their sales.
- **Real Estate –** There is reputational overlap with property values and this will include the effect that living near the Lake could have on the value of properties as well as developments using the name.
- **Tourism –** There is reputational overlap in the wider tourism market from any negative reputation of the Lake.

Recreational benefits are the most significantly impacted by remediation

We have identified recreational values as the most significant, with a total annual recreational activity value, impacted by water quality, of $1.34 million.

The incremental changes to these recreational values from remediation will happen through two main effects:

- A reduction in no swim days will increase the volume of recreational activities
- An increase in trophic level will improve the quality of recreational activities

Table E.2: Incremental Benefit (PV) of a Successful Remediation by Category

Factor	Stable	Natural Recovery	Deteriorates
Sightseeing	$619,000	$344,000	$982,000
Swimming	$684,000	$374,000	$1,155,000
Walking / running	$544,000	$302,000	$864,000
Cycling	$99,000	$55,000	$157,000
Kayaking	$205,000	$112,000	$346,000
Club Rowing	$131,000	$74,000	$195,000
Fishing	$311,000	$144,000	$816,000
Triathlon	$79,000	$71,000	$179,000

There are other potential benefits of remediation, but we have not considered them likely to significantly materialise unless the Lake deteriorates much further than what is plausible given the information that we have.

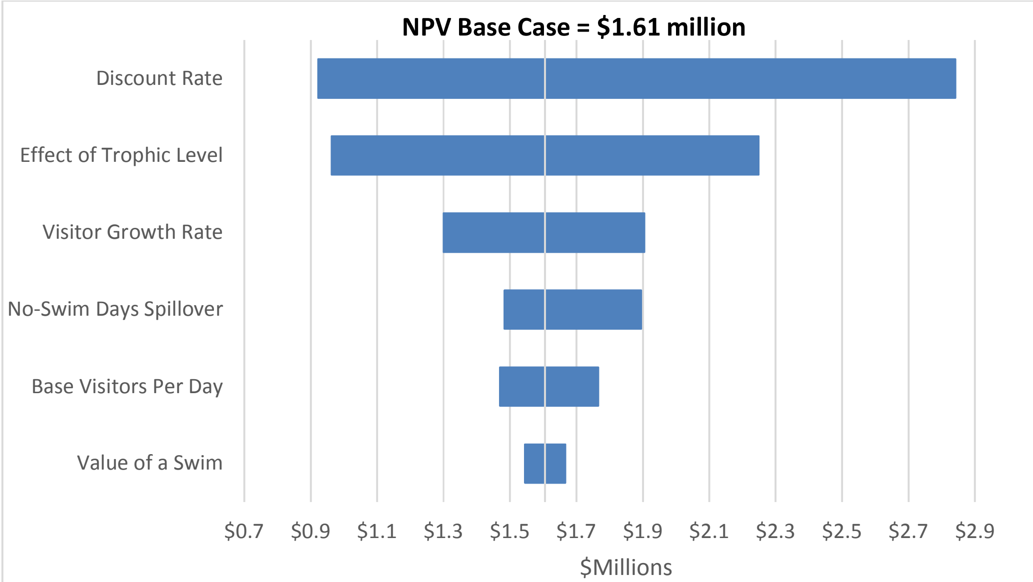
The economic viability of most of remediation options depends on some key assumptions

We tested several assumptions by varying them by +/- 50 percent of what was used in the model. The assumptions with the most significant impact on the outcomes of the analysis are:

- The discount rate
- The effect of trophic level on recreational quality and value
- The underlying visitor growth rate

Figure E.2 below shows the effect of varying key assumptions.

Figure E.2: Impact on NPV Benefits of Different Assumptions (Flushing compared against stable counterfactual)



The benefits from improvements to water quality are concentrated around the Lake and nearby residents

Residents of Lake Hayes and Lake Hayes South will see the most additional benefits of improved water quality at the Lake. However, a significant number of visitors from further afield, including all of Queenstown Lakes District Council, Otago Regional Council, and tourists from outside of the region will also benefit from the remediation.

Table E.3: Geographic Distribution of Benefits

Area	Proportion of Benefits
Lake Hayes	43 percent
Lake Hayes South	31 percent
The District (QLDC residents)	13 percent
Outside of the District (Including ORC residents from outside QLDC, national and foreign tourists)	13 percent

1 Background and Methodology

Lake Hayes (the Lake) is a small scenic lake, located in Central Otago, between Arrowtown and Queenstown, with views over the mountains in the Wakatipu Basin. It is a popular destination in the region for recreation, including walking, cycling and swimming.

The Lake has experienced an accumulation of phosphorous in the lake bed, which can be responsible for feeding algal blooms. Algal blooms can affect the colour of the Lake, turning it a brown or greenish colour and cause scums on the surface. Under certain circumstances certain types of algae can produce toxins which can cause rashes, nausea and be potentially deadly for dogs to drink.

Otago Regional Council (ORC) has been investigating remediation options to inhibit algal growth in the Lake, and has identified three potential intervention options. Castalia has been engaged by ORC to conduct an economic assessment of the remediation options.

The purpose of this paper is to:

- Assess the benefits and economic value of improved water quality and a reduction in algal blooms at Lake Hayes
- Conduct an economic assessment of the three proposed remediation options
- Identify who the beneficiaries of improved water quality are, and how the benefits could be apportioned

1.1 What is the Current and Predicted State of Water Quality at the Lake?

Lake Hayes is 2.7km^2 in area, and lies approximately 10km east of Queenstown and 4km south of Arrowtown. Figure 1.1 shows a map of the region.

Figure 1.1: Map of the Lake Hayes Region

The region has experienced significant growth, both in residency and tourism, over the last ten to twenty years. Queenstown Lakes District Council (QLDC) was the second fastest growing district in all of New Zealand between 2001 and 2013. In addition, Lake Hayes South was the fastest growing area in QLDC due to the development of several new housing estates. The population of the area is predicted to grow by 30 percent over the next ten years.[1]

Tourism underpins Queenstown's economy, and experienced nearly one million passenger arrivals, both domestic (709,000) and international (281,000), in 2017.

The Lake has experienced a large surplus of nutrients in the past which are now stored in the sediment

The detailed history and condition of the Lake has been discussed in two recent papers, one commissioned by the Friends of Lake Hayes (FoLH) group written by Schallenberg & Schallenberg (2017), and the most recent a NIWA report (2018) commissioned by ORC.

The papers describe how the Lake catchment was likely deforested as far back as 1740, and served largely as agricultural land from the mid 1900's. Since this time significant amounts of superphosphates, and phosphorous (P) have flowed into the lake and accelerated eutrophication.

Since the late 1960's the Lake has experienced severe algal blooms, and the state has severely deteriorated.

Lake Hayes is now a eutrophic Lake

Lake Hayes has become a eutrophic lake, often with poor water clarity and regular algal blooms. Eutrophic refers to a lake having an abundant accumulation of nutrients that support high densities of algae, fish and other aquatic organisms. As eutrophic lakes have so much biomass, there is a lot of decomposition occurring at the bottom of the lake which consumes oxygen causing the bottom of the lake to become anoxic (low in oxygen) in the summer. This summer (1 Dec 2017 to 31 March 2018) Lake Hayes was closed due to cyanobacteria (toxic algae) from the 12 February to the 26 February, and another high result closed the Lake from 7 March to the 19 March. High bacteria (E. coli) counts closed the Lake on the 5 March with signs taken down on the 12 March.[2] E. coli closures and occurrence are not the primary focus of the remediation efforts as they are caused by factors other than algae, and therefore will not be the main consideration of this paper.

Predictions of the future quality of water in Lake Hayes are uncertain

Both the Schallenberg & Schallenberg report and the NIWA report suggest that the Lake may be at a potential recovery tipping point.

Schallenberg & Schallenberg suggest that despite the ceratium blooms that have been experienced in most summers since 2006, recent developments, predominantly the increase in the flushing of P out of the Lake via the Lake Hayes creek, suggest that the Lake may be on a trajectory toward recovery from the high nutrient loads and eutrophication.

They suggest that appropriate restoration measures could result in stable improvements in summer water clarity and reductions in ceratium summer biomass and the re-oxygenation

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[1] Queenstown Lakes District Growth Projection 2018-2058, (May 2017)

[2] Otago Regional Council

of the bottom waters of the Lake. However, the degree or timeframe are not given, and in general, the trajectory of the quality of the Lake appears unable to be confidently predicted.

Regarding this economic assessment report, it is important to emphasise that this is not a scientific paper and it is not the role of this report to be making scientific assessments or judge the remediation options on their scientific basis. We can only assess the economic value of scenarios based on the scientific information available.

1.2 Our Approach to Economic Assessment

Economic evaluations, such as the cost benefit analyses (CBA) method being used in this report, are powerful tools to evaluate planning decisions.

Economic evaluations require a complete assessment of all the costs and benefits measured over time. Costs and benefits must be in terms of the impacts on people.

It is not just a financial assessment, but rather an assessment that includes all the non-financial, public and private benefits and costs that could be impacted by water quality at Lake Hayes.

The costs and benefits are forward looking and must be related directly to the decision at hand, e.g. if they would occur anyway then they will not be included. Another important factor of an economic evaluation is to be aware of double counting and additionality. Economic costs and benefits must be net changes relative to the status quo (our counterfactual).

There are three key steps in the assessment of costs and benefits:

- Determine the counterfactual; what happens if there is no remediation?
- Qualitatively assess the costs and benefits that may occur if the remediation options are implemented, and identify which are economically material
- Quantify all the material costs and benefits to determine net benefits and benefit cost ratios

What will happen to Lake Hayes if there is no remediation?

The counterfactual is the state of the world if the intervention does not occur. This requires an estimation and projection of the likely state of the water quality in Lake Hayes if there was no intervention. Account must also be taken of the projected growth rates and trends in the area.

For this analysis we investigate three counterfactual scenarios, to account for the uncertainty regarding what might happen to the Lake:

- **Stable** – the quality of water that the Lake continues, the trophic level remains the same and the Lake stays eutrophic (see Table 2.6 for a full description of trophic levels). The current costs from poor water quality, e.g. cancelled triathlon and periods of no swimming will continue. The frequency of these events will be projected into the future in the cyclical nature that the Lake has been experiencing.
- **Natural Recovery** – The Lake recovers naturally, the trophic levels decline, the Lake becomes mesotrophic. The frequency of poor water quality events are projected to decline over time.
- **Deteriorates** – the state of water quality deteriorates further and begins causing further costs, e.g. an increase in the days unable to swim. The trophic level

increases and the Lake becomes supertrophic. The water looks sludgy and loses all reflection etc.

What are the impacts on people if a remediation occurs?

The qualitative assessment will consider and discuss which of the potential factors related to the Lake are strongly influenced by the water quality.

The assessment will be broken up into a rating of negligible, low, moderate and high. These are defined in Table 1.1. Impacts assessed as moderate or high will go on to the next stage and be quantified.

Table 1.1: Description of Scale of Impact Rating

Rating	Description
Negligible	Water quality has no impact
Low	Water quality may have a minor impact, but only if it was an extreme change or highly unlikely to occur (e.g., turned to a sludge pit)
Moderate	Water quality is related and would have a noticeable impact on some factors with a medium level of change in lake quality. Even infrequent bad water quality events have an impact, in proportion to their frequency
High	Water quality in the Lake is significantly related and will have a large impact. Small changes will be noticeable, and large changes would have significant effects beyond the immediate period of the event

What is the value of the costs and benefits?

Impacts that were assessed as 'moderate' or high' will then be quantified. This involves calculating:

- **Activity volume** – Determines the volume of activity and events related to the Lake (e.g. visitor numbers)
- **Unit value** – Research and determine an appropriate method and value to assign to each of the factors[3]
- **Identify variation –** Identify the level of changes in the volume and values projected over the next thirty years that are likely to occur in each of our counterfactual scenarios and the remediation options
- **Costs** – the projected costs of each of the proposed remediation options will be assessed

The volume, value and variation figures are used to calculate total benefit and costs of remediation. A discount rate is applied to future values to determine their value in today's terms. A Net Present Value (NPV) for each intervention cost and benefit is generated, and this is used to determine a Benefit Cost Ratio (BCR). Any option with a BCR of greater than one shows that the options benefits outweigh its costs.

A sensitivity analysis is conducted to show effects of uncertainty, and to determine the critical assumptions that the analysis is dependent upon.

[3] Value is measured by willingness-to-pay. This can be identified through or willingness-to-accept, travel cost, hedonic analysis, or revealed preferences. This includes applying these methods and values from other comparable research and literature.

2 Who is Impacted by the Water Quality at Lake Hayes?

The standard of water quality at Lake Hayes has important implications both locally and further afield, such as visitors from around New Zealand and even internationally. Recently the Lake has experienced serious issues to do with bad water quality which give an insight into what, how and who this can impact.

There are two main causes of bad water quality at the Lake[4]:

- **Blue/green algae (cyanobacteria) –** The development and proliferation of algal blooms is due to a combination of environmental factors including an increased concentration of certain nutrients, deoxygenation, temperature, and sunlight amongst others. This is the issue that the remediation options are looking to inhibit and prevent.
- **E-Coli** – E-Coli is caused by different sources, such as human or animal faeces. This is not specifically the water quality issue that the remediation options are intended to solve

2.1 What Activities are Impacted by Water Quality at the Lake?

Below we describe the major categories of impacts that we have identified. There are four categories of impact:

- **Lake Based Recreation –** Water quality directly influences recreation activities at the Lake, such as swimming and sightseeing. This will include any regular scheduled events that take place at the Lake.
- **Local Business Sales–** Businesses sell products to lake visitors and this will include local food and wine venues, accommodation and tours. There is reputational overlap with local businesses that will impact their sales.
- **Real Estate –** There is reputational overlap with property values and this will include the effect that living near the Lake could have on the value of properties, as well as developments using the name.
- **Tourism –** There is reputational overlap in the wider tourism market from any negative reputation of the Lake.

2.2 How Material Are the Impacts of Water Quality on These Activities?

In the following section we consider each category, and assess the materiality of the impact that water quality at Lake Hayes may have on it. Factors assessed as having the potential to have a moderate or high impact will be taken to the next level of analysis to assess their volume and values.

Table 2.1 considers the recreational activities and regular events that occur in and around the Lake that could be affected by water quality.

[4] https://www.lawa.org.nz/learn/factsheets/coastal-and-freshwater-recreation-monitoring/

Table 2.1: Lake Based Recreation Materiality Assessment

Activity	Assessment	Reasoning
Sightseeing /photography	High	▪ Significantly affected if the Lake water quality continues to deteriorate. Affects the colour and clarity, making the Lake less picturesque and impacting flora and fauna. ▪ May result in less people wanting to visit to sightsee, take pictures etc.
Swimming	High	▪ Swimming will be the most significantly affected activity of bad water quality. When the Lake has a toxic algal bloom (as in summer of 17/18) no swimming can occur at all as it causes rashes, nausea, etc. ▪ Even if there are no specific toxic algal bloom, the more common blue/green algae and state of eutrophication make the water quality less clear which may make people less inclined to swim. ▪ Although there are other lakes in the region (such as Wakatipu) Lake Hayes is particularly popular with swimmers as it is significantly warmer to swim in.
Walking/ running	Moderate	▪ Walking and running around the Lake are not directly affected by the water quality, but people may be less inclined to visit the Lake when there is bad water quality, resulting in a reduction in the total number of people walking or running around the Lake. ▪ A toxic algal bloom could have serious effects on dog walkers, as dogs can get seriously sick if they drink the water.
Cycling	Moderate	▪ As with walking, the ability to cycle around the Lake is not directly affected by water quality, but may be impacted by a general decline in visits, particularly if cyclists hear they would not be able to subsequently swim to cool off etc.
Kayaking / paddle boarding	High	▪ Recreational activities such as kayaking, and paddle boarding are significantly impacted by water quality, and will be almost 100 percent affected by no-swim-days
Fishing	High	▪ The number of people choosing to fish in the Lake is significantly affected by the water quality. Incidences of stunted and dead fish have been witnessed, and water clarity will influence how appealing the Lake is to fish in.
Rowing	Moderate	▪ It is still possible to participate in club rowing when the water quality in the Lake is bad, (e.g., no illness or rashes were reported during-no-swim-days in 17/18) but the appeal of joining the rowing club may decline, and if it deteriorated significantly the rowing club may consider relocating[5]
Lake Hayes Triathlon	High	▪ There are two Lake Hayes triathlons, Christmas and Easter. ▪ The triathlons would be significantly affected by bad water quality, so much so that the Easter event had to be cancelled

[5] Head Coach, Wakatipu Rowing Club

Activity	Assessment	Reasoning
		in 2018 due to the bad water quality resulting in less than 30 registrations.
Lake Hayes Annual A&P Show	Low	▪ The show is held on the second Saturday in January, and has approximately 5,000 people attend, 75 – 80 percent local. ▪ The show is not directly related to water quality and would still be hosted even when water quality is bad.[6]
Queenstown International Marathon	Low	▪ The Queenstown marathon route has gone past the Lake in the past, but the marathon is not specifically designed around Lake Hayes and there are other routes are available, so Lake Hayes water quality would not be likely to affect the event being held or the number of people participating

Table 2.2 considers which type of businesses and in which locations could be impacted by water quality in the Lake.

Table 2.2: Local Businesses Materiality Assessment

Type	Impact Assessment	Reasoning
Directly on the Lake, e.g. Amisfield, Stoneridge	Low (Note exception)	▪ There are not a significant number of businesses located directly on or overlooking Lake Hayes. The businesses of Amisfield and Stoneridge are currently very popular, and may only experience a very small effect from less people visiting the Lake, or if there was a serious deterioration in lake quality beyond what has been predicted (e.g., there were significant Lake quality issues in 17/18 summer but no reports of negative effects on business) ▪ There is considerable reputational overlap, of Amisfield in particular, with Lake Hayes. Under the stable or improves counterfactual scenarios we do not feel that material damage will be done. However, a significant 'deteriorates' scenario (e.g., the Lake becoming permanently unswimable) could have significant reputational damage for Lake Hayes branding at Amisfield.
Commercial accommodation	Low	▪ There is not a significant amount of local accommodation, and effects are only likely to be experienced once there has been a serious and widespread effect of the reputation of the Lake due to a severely deteriorates scenario.
Local Tours	Low	▪ There are local tourist tours in the region to Arrowtown and nearby vineyards that go past or stop at Lake Hayes. However, the Lake is not the only or ultimate destination for any tours from Queenstown. Therefore, the individual effect of changes in water quality on these tours is likely to be low.

[6] Chairman and secretary, Lake Hayes A&P society

Business & accommodation in the wider area, e.g. cafés, or dairies in Arrowtown	Low	▪ The area surrounding Lake Hayes is popular and busy for many reasons other than the Lake itself, so it is unlikely that they would be noticeably affected by a potential reduction in visitors to the Lake.

Table 2.3 considers how the financial value of real estate in the area could be affected by the water quality in the Lake.

Table 2.3: Real Estate Materiality Assessment

Type	**Impact Assessment**	**Reasoning**
Lake view properties	Moderate (note double counting concern)	▪ The properties directly overlooking Lake Hayes already have significantly greater land value than properties in the vicinity without lake views. ▪ There is various literature investigating the effect that water quality can have on housing value, and a specific study using hedonic pricing methods showed that water quality in the local vicinity had a statistically significant relationship to house price [7] ▪ However, there is a double counting concern with our methodology in that we have counted the recreational value directly and we do not wish to include the indirect effect that this has on house prices again. E.g., house prices are affected by the quality of nearby recreation.
Lake Hayes South subdivision estates: ▪ Lake Hayes Estate ▪ Bridesdale ▪ Hayes Creek Estate	Low	▪ The quality of the Lake will have effects on the value of the recreation of local residents, but the effect on the total value of real estate is likely to be low (or double counted as above). ▪ The subdivisions are already completely sold and in high demand given the relative lack of available housing in the region.
Wider Queenstown and the Wakatipu Region	Low	▪ Similar to above, the water quality of the Lake may affect resident recreation, but is unlikely to have a significant or measurable impact on housing value in the wider Queenstown region ▪ An exception to this is a total deterioration of the Lake which impacts the reputation of the area.

[7] Nicholls & Crompton (2018) A Comprehensive Review of the Evidence of the Impact of Surface Water Quality on Property Values

Table 2.4 considers the possible effect that water quality in Lake Hayes could have on the wider tourism industry of Queenstown and New Zealand.

Table 2.4: Tourism Materiality Assessment

Type	Impact Assessment	Reasoning
Reputation effects for tourism in Queenstown and New Zealand	Low / Unquantifiable (but note the risk)	▪ Tourism is critical to New Zealand and the largest driver of the Queenstown economy, which is experiencing significant growth rates. ▪ Reputation is important to this, and Lake Hayes is a contributor, especially given the fact that it is highly visible on the flight path in to Queenstown. However, there are such a wide variety of other scenic locations and tourist activities in the region, that the water quality in Lake Hayes on its own is unlikely to have a significant or measurable effect on total tourist numbers. ▪ Freshwater quality standards across the whole country are an important factor in the overseas reputation of New Zealand, but Lake Hayes is only a small contributor to this overall image. ▪ The exception to this is if a serious, prolonged deterioration occurs on the Lake and its water quality. This could become a reputational drag on the entire industry in the area.

2.3 What Measures Can be Used to Value Impacts?

In this section we consider the factors that could be used as effective measures to monitor, assess and predict the condition of the Lake. Table 2.5 describes the measures:

Table 2.5: Effectiveness of Measures to Monitor Lake Condition

Measure	Relationship with Lake Water Quality	Reason
No-swim-days	Significant	▪ No-swim-days warnings are posted to warn people of the dangers of swimming ▪ No-swim-days make a good measure of remediation benefits as they are clearly observable and allow specific assumptions to be made. ▪ As well as the official no-swim-days there are the spill-over effects that would happen when people are not up-to-date with the warnings being cancelled etc.
Event Cancellations	Significant	▪ Occasionally events are cancelled due to the risk that there could be a no swim notice or other impacts affecting the event ▪ Event cancellations are a direct effect of water quality and provide a specific valuation to be made.
Angler Days	Significant	▪ There are surveys the count the number of angler days on the Lake

Measure	Relationship with Lake Water Quality	Reason
		▪ Angler days in the Lake have dropped significantly over the last 20 years. Any increase or decrease in this would be an effective measure of an improvement of water quality in the Lake.
Trophic Level	Significant	▪ Water quality level can be measured directly through 'trophic level' measures, which define the overall health of New Zealand Lakes. ▪ Table 2.6 gives a description of the Trophic Level Index used by Land Air Water Aotearoa ▪ Changes in trophic level of the Lake would be an effective measure of how water quality has increased or decreased, but has difficulties associated with using it as a specific measure of the value of the Lake

Table 2.6 gives a summary of the Land Air Water Aotearoa (LAWA) trophic levels and how they are used as a measure and describe water quality.

Table 2.6: LAWA Trophic Level Descriptions

Trophic Level	Status	Condition
0-2	Very Good	Microtrophic: A trophic level of less than 2 meaning the water quality is very good. The Lake is clear and blue with very low levels of nutrients and algae.
2-3	Good	Oligotrophic: A term often used to describe the characteristic of a lake. Oligotrophic lakes are relatively poor in plant nutrients, leading to sparse growth of algae and other organisms, and have a high oxygen content owing to the low organic content (compare with euthrophic). The Lake is clear and blue, with very low levels of nutrients and algae.
3-4	Average	Mesotrophic: A trophic level of 3-4 meaning the water quality is average. The Lake has moderate levels of nutrients and algae.
4-5	Poor Quality	Eutrophic: Term used to characterise a lake. This refers to a lake having an abundant accumulation of nutrients that support a dense growth of algae and other organisms, the decay of which depletes the shallow waters of oxygen in summer which can result in death of animal life. (Compare with Oligotrophic). In LAWA a trophic level of 4-5 meaning the water quality is poor. The Lake is green and murky, with higher amounts of nutrients and algae.
Greater than 5	Very Poor	Supertrophic: A trophic level greater than 5 meaning the water quality is very poor. The Lake is fertile and saturated in phosphorus and nitrogen, often associated with poor water clarity.

Land Air Water Aotearoa - Trophic Level Index

2.4 Who is Impacted?

As well as the categories of impacts, this report seeks to determine the key groups who are affected by the costs and benefits. Benefits will be apportioned in to the categories shown in Table 2.7.

Table 2.7: Categories of Beneficiaries of Water Quality at the Lake

Area	Included	Description
Lake Hayes Residents	▪ Approximately 210 dwellings, 370 population	▪ Residents surrounding the Lake are highly invested in the water quality that they look over every day. They are also impacted by factors such as smell, noise and wildlife. ▪ These residents have the benefit of being able to use the Lake for recreational purposes whenever they wish, and can easily keep track of when there are any significant water quality issues impacting swimability.
Lake Hayes South Residents	▪ Approximately 920 dwellings, 3,000 population.	▪ This includes Lake Hayes Estate and the new subdivisions south of the Lake such as Shotover Country ▪ For residents of Lake Hayes South, the Lake is their primary recreational location. They can easily visit the Lake before or after work or school via a short 5-minute drive.
The District (QLDC)	▪ Approx. 18,000 dwellings, 35,000 population (excluding above)	▪ Other residents of QLDC are involved with the Lake to varying degrees. From downtown Queenstown the Lake is approximately a 20-30-minute drive, and can easily be used for recreation on weekends, however, there are also many other outdoor recreation locations to choose from. ▪ QLDC residents further away, such as in the Wanaka ward, would not regularly use the Lake for recreation, but would still be familiar with the Lake, and likely drive past it relatively regularly, such as when visiting Queenstown.
The Region (ORC)	▪ Approx. 170,000 population (excluding above)	▪ ORC residents outside of QLDC are invested in the Lake to the degree that it impacts on the council reputation, and the fact they are more likely to be regular visitors to the area than other parts of NZ.
Outside of Region	▪ All other visitors	▪ All other visitors from outside of ORC benefit from the Lake when they are

		tourists to the area, which is unlikely to be on a regular or consistent basis.[8]

3 What is the Economic Benefit of Remediation of the Lake?

The economic factors identified in Section Two can be quantified and a value assigned to each of the activities, including the costs, to obtain a current value for remediation.

For economic goods and services that are not traded, market prices cannot be observed. In this situation non-market valuation techniques must be used. For this study we use inferred willingness-to-pay based on non-market studies conducted for similar things as no new studies were undertaken within the scope or budget of this report, except for informal consultation.

This section will begin by examining the current volumes, trends and unit values that can be associated with the Lake, and then calculate an approximate total economic value of activities impacted by water quality.

We will then calculate the benefits that can be attributed specifically to remediation, by estimating the level of activity that would occur if the water quality were to improve. This is measured for 30 years into the future and discounted back to a single Present Value (PV) figure. Finally, the estimated costs of the remediation options will be discussed.

3.1 Total Volume and Value of Recreational Activity at the Lake

Table 3.1 shows a summary of the approximate total current annual value of recreational activities (impacted by the water quality) of the Lake.

Table 3.1: Annual Value of Recreational Activity at Lake Hayes

Activity	Annual Volume	Single Value	Overall Value
Sightseeing / photography	70,080	$5	$350,400
Swimming	17,520	$20	$350,400
Walking/running	30,800	$10	$310,000
Cycling	5,600	$10	$56,000
Kayaking / paddle boarding / windsurfing	3,500	$30	$105,000
Club Rowing	95 (members)	$850 membership	$80,750
Fishing	180	$20	$3,600
Triathlon	Two scheduled triathlons scheduled, with approximately 250 participants	$100	$50,000
Annual Lake Recreation Value			$1.3m

[8] For the purposes of this study we have not distinguished between visitors from New Zealand and overseas

Table 3.2 discusses the assumptions and analysis behind the above values and results

Table 3.2: Willingness to Pay for Recreational Activities

Activity	Discussion
Sightseeing	▪ For the purpose of this study, all daily visitors to the Lake will be considered 'sightseers' (despite the fact that this ignores a significant number of people who drive by and are not recorded as visitors). ▪ QLDC figures estimate the daily average visitors to be 192, (ranging to a peak day of 500), resulting in an annual volume of **70,080**. ▪ Sightseeing is quantified based on it being the least intensive and possibly shortest duration of the recreational activities undertaken. This is half the value given to walking and cycling of **$5.00**.
Swimming	▪ The annual swimming volume is based on research conducted by MPI in 2016, which found that an average of 25 percent of visitors to a lake in New Zealand were likely to go swimming, resulting in an annual figure of **17,520.**[9] ▪ Transport cost methods are one way of assuming a visitor's willingness-to-pay, based on how much they spend to travel to the activity. For example, an estimate could be made based on an average distance of travelling to and from downtown Queenstown, of approximately 30km, at ten litres per 100km, and $2.20 a litre, equating to $6.60. However, this only considers locals in the area, or tourists staying in Queenstown, and does not take into account consumer surplus, or the fact this may not represent the total value of what people would be willing-to-pay. ▪ Another method may be comparing it to people's willingness-to-pay for similar activities. For example, the price of a swimming at the nearby Alpine Aqualand is $8, or lakes in Europe frequently have admittance fees, of up to €10 euro ($16.72 NZD). ▪ Contingent valuation is another method in which people are directly asked how much they would be willing-to-pay for an activity or product. For example, a 2004 study investigated improving water quality freshwater rivers in New Zealand, and found a range of between $17 and $27 for the ability to swim. [10] in clean water. ▪ From a combination of the above rationale, and casual interviews conducted around the Lake, we have based the value of a swim in the Lake as **$20.00** for the central scenario.
Walking/ Running	▪ The annual walking numbers are based on the figures that DOC provided from the Lake Hayes walkway pad counter.[11] Figures ranged from 5,277 in January 2018, to 1,428 in July 2017 with an average of 100 people a day on data going back to 2015, or 36,500 annually. ▪ The counter is also triggered by cyclists, so an estimate of 85 percent will be assigned to walking (or 44 percent of total average visitors); **31,025**

[9] Non-market valuation of improvements in freshwater quality for New Zealand residents, from changes in stock exclusion policy, Ministry for Primary Industries (July 2016)

[10] Instream Water Values: Canterbury's Rakaia and Waimakariri Rivers (2004), Lincoln University.

[11] Department of Conservation Lake Hayes Walkway Pad Counter (2010-2017)

Activity	Discussion
	▪ There are several methods that can be used to calculate the economic value of a walk around the Lake, such as comparing it to willingness-to-pay for similar activities. An example in New Zealand is that hikers regularly pay for a return shuttle bus to walk the Tongariro track which costs $30. However, this has limitations because this is considered one of New Zealand 'great' walks and is considerably longer. ▪ The cycling value is being calculated based on a willingness-to-pay for a hire bike in the vicinity. A walk around the Lake is estimated to have a similar utility value as a cycle around the Lake, and is therefore being assigned the same value of **$10.00**
Cycling	▪ The Lake Hayes Circuit is a central part of the Queenstown Trail one of New Zealand's most popular cycle rides ▪ The cyclist numbers are based on the DOC walkway pad counter of 36,500 annually and an assumption of 15 percent of total traffic, **5,475** ▪ The value of a bike ride is based on a quarter of the average cost to hire a bike for four hours in the region of $40[12] (given the Lake Hayes circuit would only be a portion of the entire distance of the cycle). This value is **$10.00**
Club Rowing	▪ The Wakatipu rowing club has been located on Lake Hayes for decades ▪ The club has 90-100 active members—**95** has been used. ▪ The Wakatipu Rowing Club confirmed that their membership fees per season are $800-900—**$850** has been used.
Kayaking, Paddle boarding	▪ The number of visitors engaging in activities such as kayaking, or paddle boarding is taken from same 2016 MPI report as used for swimming, which gives an estimation of five percent of year-round visitors, equating to an average of 9.6 per day and an annual figure of **3,504** ▪ There are several recreational hire shops in Queenstown that rent kayaks and recommend Lake Hayes as a destination ▪ The willingness-to-pay value is based on an average cost to hire a kayak for two hours from a rental business in Queenstown of **$30.00** [13]
Fishing	▪ Fishing has been an activity associated with Lake Hayes since trout were introduced to the Lake in 1870 ▪ Estimated fishing (angler) days were provided by Otago Fish and Game. 17/18 numbers were not available, but 14/15 numbers were reported to be **180**±90[14] ▪ The value of a one-day fishing licence issued by Otago Fish & Game is **$20.00**
Triathlons	▪ Lake Hayes has two triathlon events scheduled every year at Christmas and Easter with approximately **250** participants.[15] ▪ Participants in events will be counted as additional on top of the average visitor figures. ▪ The registration fees are approximately **$100**[16]

[12] $45 from: https://www.queenstownbiketours.co.nz/trail-rides/lake-hayes-loop/

[13] http://www.whatsupqueenstown.co.nz/prices/ https://www.bookme.co.nz /

[14] Otago Fish and Game

[15] Taken from 2017 Christmas results

[16] Could not get directly in touch with Lake Hayes triathlon coordinator.

3.2 The Benefits of Remediation of Lake Hayes Water Quality

The benefits of remediation must be measured by the projected changes that remediation will bring, compared to the counterfactual scenarios for the next 30 years.[17] There is considerable uncertainty in the trajectory of water quality at Lake Hayes without remediation. This has been simplified and standardised into three possible counterfactual and one successful remediation outcome. We will assess uncertainty in the success rate and timeframe of remediation in sensitivity testing.

Table 3.3 shows the projected assumptions that will be used to calculate the benefits of remediation.

Table 3.3: Possible Outcomes for Lake Hayes Water Quality

Scenario	Events
Stable	▪ 26 days no swim days every eight years with 50 percent spillover days ▪ One triathlon event cancellation every eight years ▪ No increase in angler days from 180 ▪ Trophic level stays between 4-5
Natural Recovery	▪ Reduction in no swim days: halving in number every eight years until less than one ▪ One triathlon event cancellation every ten years ▪ Increase in angler days from 180 of 2.5 percent per annum ▪ Trophic level slowly decreases to between 3-4
Deteriorates	▪ Increase in frequency of no swim days ▪ two event triathlons every 5 years ▪ Decrease in angler days from 180 of 5 percent every year ▪ Trophic level increase to over five
Successful Remediation	▪ Complete reduction in no swim days by year five ▪ No further event cancellations ▪ Increase in angler days from 180 of 5 percent per annum ▪ Trophic level declines to between 2-3

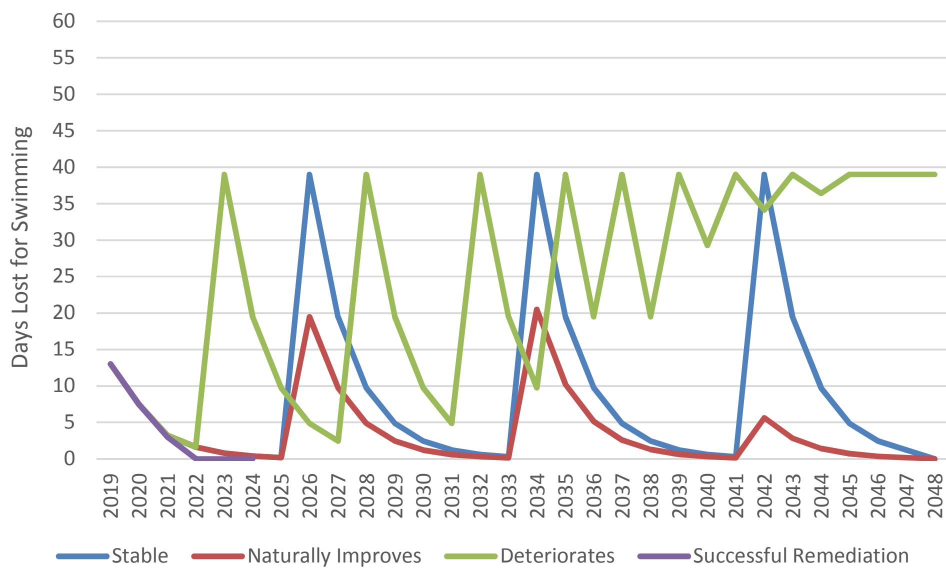
The benefits of reduced no-swim-days has a direct effect and a spillover effect

There were 26 no-swim-days in the summer of 17/18. These no swim-days will also have a spillover effect. When discussing the Lake with people in the area, it showed that even when the Lake was officially open again, many people may not have heard water quality had improved, and continued not to visit for a period. We use an assumption that this could also have a flow on effect to subsequent seasons, e.g., there will be less people visiting the Lake next summer because of hearing about the no-swim-days this summer, but that this would continue to decline in the following seasons if there were no further events.

The frequency of no-swim-days used reflects the cyclical condition of the Lake, which is significantly worse in certain years due to warm weather conditions etc. Figure 3.1 shows the number of no-swim-days that have been used to calculate benefits for each scenario.

[17] 30 years is the recommended CBA time period from NZ treasury

Figure 3.1: Number of No-Swim-Days for Each Scenario



The effect that the no-swim-days have on other Lake recreational activities is another factor that will drive the benefit value of remediation. Swimming and kayaking are calculated as losing 100 percent of their value on no-swim-days, whilst the other recreational activities of sightseeing, walking and cycling assume losing 50 percent of their value or utility.

Table 3.4: Value Generated by Reducing No- Swim- Days

Counterfactual Scenario	Present Value of Benefits
Stable	$305,000
Natural Recovery	$141,000
Deterioration	$808,000

There are benefits from reduced event cancellations

In 2018 there was a cancellation of the Easter triathlon. The assumptions of event cancellations were based on the fact that the condition of the Lake is cyclical, and unlikely to affect every year continuously unless there was a serious decline in condition.

Table 3.5: Value Generated by Reducing Event Cancellations

Counterfactual Scenario	Present Value of Benefits
Stable	$61,000
Natural Recovery	$35,000
Deterioration	$122,000

There are benefits of increased angler Days

The condition of the Lake and the effect it will have on water clarity, and potentially fish population, will be represented by angler days in the Lake. It is assumed that angler days will increase for the naturally improving and remediation scenarios, whilst in the deterioration scenario angler days decline to less than 50 over the next 20 years.

Table 3.6: Value from Increasing Angler Days

Counterfactual Scenario	Present Value of Benefits
Stable	$44,000
Natural Recovery	$27,000
Deterioration	$64,000

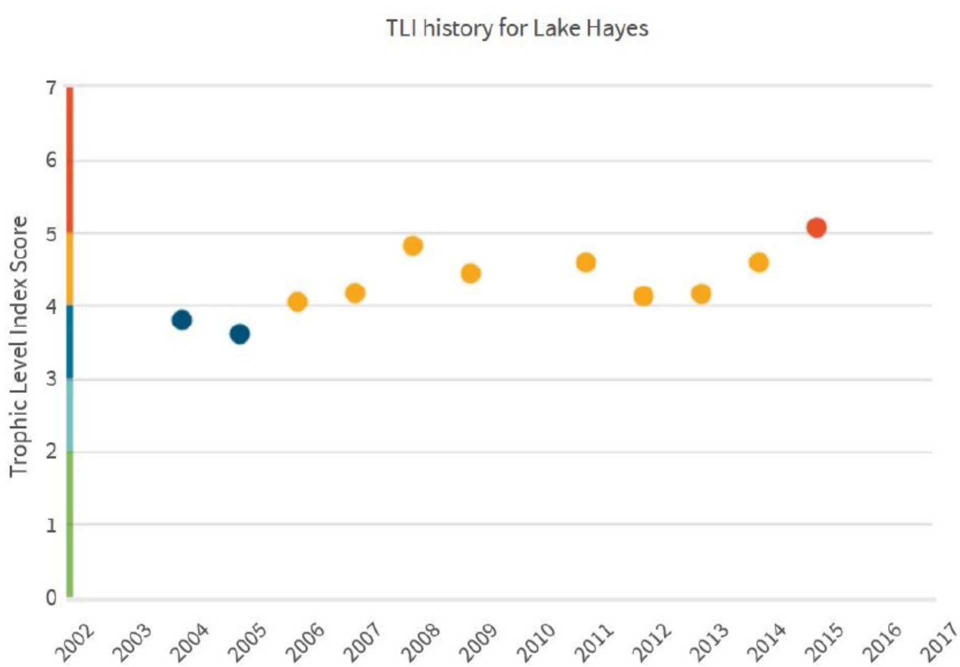
There are benefits of an improved trophic level

Trophic level is an effective measure of the clarity of the Lake (described in detail in Table 2.6). High trophic levels indicate bad water quality, and low trophic levels indicate good water quality. Trophic level impacts the quality and utility value of recreational experience outside of no-swim-day impacts, such as by enhancing the overall recreational experience by engaging with a more attractive lake.

Other lakes in the vicinity of Lake Hayes have relatively low trophic levels, such as Wakatipu (1.6), Hawea (1.6), Wanaka (2.1) and Te Anau (1.5).

Since 2005 Lake Hayes has had a Trophic Level Index of 4-5 putting it in the category of poor water quality. Figure 3.2 shows a history of the Trophic Level Index (TLI) in Lake Hayes

Figure 3.2: Trophic Level Index History for Lake Hayes



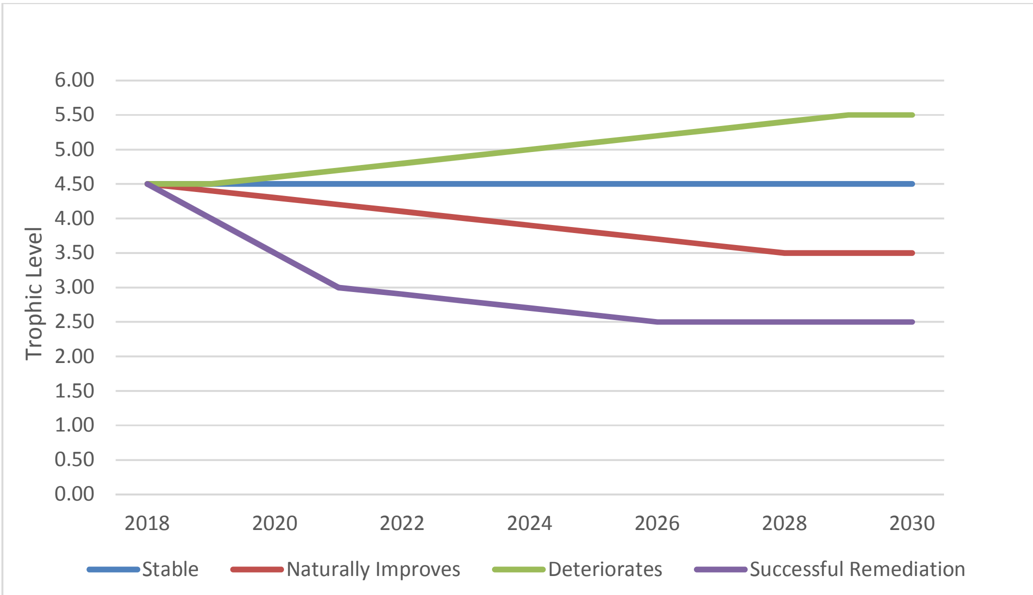
LAWA TLI History

The effect of trophic level on overall lake valuation is uncertain and will be tested in sensitivity. For the proposed remediation scenario our assessment is based on American literature that found that people had a willingness-to-pay of ten percent extra on their expenditure based on a one-meter improvement in water clarity. [18]

For this study we will apply a 0.5% increase (or decrease) to the value of recreational activities at the Lake for every 0.1 change in trophic level. This equates to a ten percent increase if the Lake falls two levels, from 'poor quality' to 'good quality'.

Figure 3.3 shows the assumptions made about Trophic level for each scenario

Figure 3.3: Scenarios for Change of Trophic Level



The resulting changes in value due to trophic level are shown in Table 3.7.

Table 3.7: Value of Remediation Improving Trophic Level

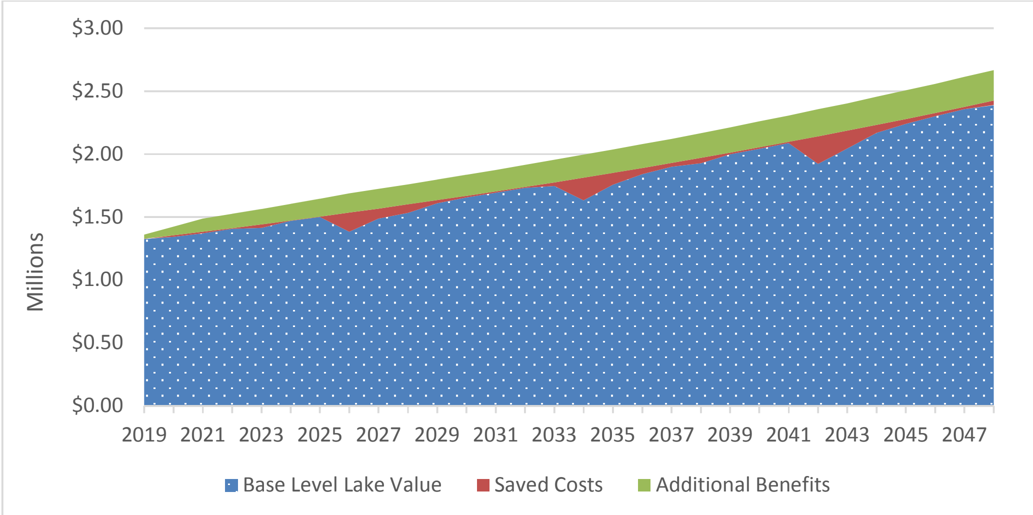
Counterfactual Scenario	Present Value of Benefits
Stable	$2,061,000
Natural Recovery	$1,168,000
Deterioration	$3,016,000

Although the potential benefits of remediation are large, they represent only a small proportion of the entire recreational value of the Lake.

Figure 3.4 demonstrates the value of remediation of the Lake if the water quality stays stable, compared to the total recreational value over the next 30 years.

[18] An Assessment of the Economic Value of Clean Water in Lake Champlain (2015) Prepared for the Lake Champlain Basin Program.

Figure 3.4: Total Recreational Value of Lake Hayes Including Remediation (Stable Counterfactual Scenario)



3.3 How Much Would Remediation Cost?

The costs of remediation are based on the 2018 NIWA report which defined three primary remediation options:

- Flushing
- Destratification
- Capping

From an economic perspective the difference in the options are relevant only in terms of:

- Their relative implementation and ongoing costs
- The relative likelihood of success
- The value of additional side effects that each option may impose

Flushing involves increasing water inflow to improve the net balance of nutrients

Flushing is a medium-long-term option that has the potential to transfer additional inflow, providing sufficient oxygen to prevent deoxygenation. It has the benefit of being a natural process, but has a lower certainty of success as it is highly dependent on climate patterns. It does not impose any noticeable side effects, and could be used tandem with other remediation processes.

The provided costs are an initial implementation of $100-$200k, and an ongoing cost of $30k per annum that would have to be run over a period of years to see a positive impact on water quality. For the purpose of this analysis we will use $150k capex, and $30k opex per year for a period of ten years, with benefits not beginning to be felt until year five.

Destratification involves intervening in the Lake to prevent layers forming

The option of destratification uses a bubble plume across the middle of the Lake, powered by a compressor, to prevent stratification and eutrophication of the lower layers. Capital costs were estimated at $250-$350k plus an annual running, monitoring and maintenance fee that was not provided. For this analysis we will estimate $10k per year for the first ten years.

There are international precedents showing it to be an effective remediation approach, but there are risks associated with the timing of when it is turned on and off, or that the compressor or power supply could fail, and if this occurred at the wrong time it could trigger an algal bloom. The effect of this risk will be shown in sensitivity testing.

Side effects include the potential sound of the power supply / compressor, and visual impacts of the bubble curtain disturbing the Lake's reflective surface.

Chemicals can be applied to cap the sediment layer for a period of time

Capping involves sediment capping and phosphorous binding agents. Costs are highly uncertain for this option, the costs are estimated at $90k-$550k if implemented as a single dose with longevity of 5-10 years, or lower initial cost, but ongoing, if drip fed.

For the purpose of this analysis we have used two scenarios for capping, one with the lowest estimated costs of $90k every ten years, and the other with the highest estimated costs of $550k every five years. This can be used to see the effect that this large range of estimated costs has on the analysis.

This option could be used with flushing, but not destratification. There is a low risk of failure with this option as it is tried and tested with a reliable impact.

Side effects include visual side effects, such as a colouring of the sediment in the Lake. There are also the cultural considerations associated with adding further non-natural products to the Lake and the community opinions around this.

Table 3.8: Cost Assumptions of Remediation Options

Remediation Option	Initial Cost	Ongoing Cost	Present Value of Costs
Flushing	$150,000	$30,000 per year for ten years	$371,000
Destratification	$300,000	$10,000 per year for ten years	$374,000
Low Cost Capping	$90,000	$90,000 every ten years	$177,000
High Cost Capping	$550,000	$550,000 every five years	$1,797,000

4 What is the Net Value of Remediation?

The Present Values of costs and benefits of the remediation options can now be used to calculate the overall Net Present Value (NPV) of remediation options, and the associated Benefit Cost Ratios (BCR).

These values are measured in today's dollars using an NPV, which is a sum of the discounted values 30 years into the future. Benefits minus costs provides a total net benefit, and benefits divided by costs provides a BCR. A BCR ratio of higher than one shows that the option is economically beneficial.

A sensitivity analysis is undertaken to test key assumptions within plausible ranges to see how important they are to the analysis and what a realistic range of possible outcomes might be.

4.1 The Net Present Value and Benefit Cost Ratios of Remediation

Table 4.1 shows the NPV of (successful) remediation for the options of flushing, destratification, low and high cost capping under the three counterfactual scenarios. A positive NPV show that the option is economically viable.

Table 4.1: NPV of Successful Remediation

	Flushing	Destratification	Low Cost Capping	High Cost Capping
Stable	$1,612,000	$2,105,000	$2,302,000	$681,000
Natural Recovery	$625,000	$1,001,000	$1,197,000	-$423,000
Deteriorates	$2,848,000	$3,585,000	$3,782,000	$2,161,000

The results show that a successful low-cost-capping remediation under a counterfactual scenario in which the Lake quality would otherwise continue to decline, produces the largest economic benefit of $3.8m.

If a high-cost-capping option is the chosen remediation, and the Lake would have otherwise recovered naturally without any remediation needed, the economic loss is $423k.

Benefit Cost Ratios are positive for most options and potential outcomes

BCRs are calculated by dividing the total benefits by total costs. Any ratio of greater than one indicates that the option is economically viable. The highest BCR indicates the most cost-effective option.

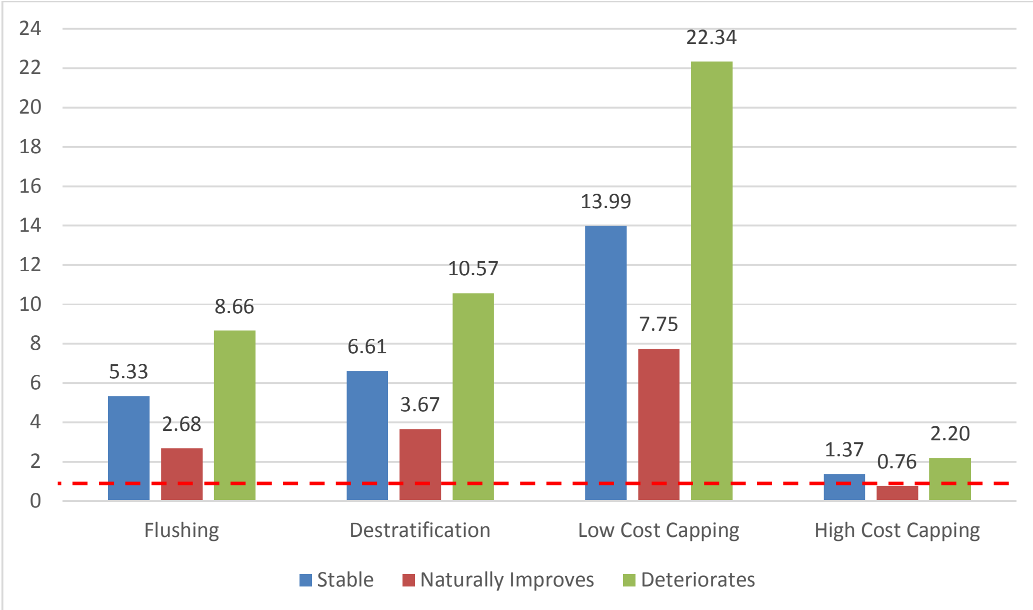
Table 4.2 shows that successful flushing, destratification and low-cost-capping remediation options are all economically viable when compared against all of the counterfactual scenarios. High-cost-capping is viable except when compared against the counterfactual in which the Lake recovers naturally.

Table 4.2: BCR of Successful Remediation

	Flushing	Destratification	Low Cost Capping	High Cost Capping
Stable	5.33	6.61	13.99	1.37
Natural Recovery	2.68	3.67	7.75	0.76
Deteriorates	8.66	10.57	22.34	2.20

Figure 4.1 shows the above BCR ratios visually. The red line represents a BCR of one, under which the option is not economically viable.

Figure 4.1: BCR of Successful Remediation



The likelihood of success and the potential side-effects of each option still need to be taken into consideration

The above results consider the viability of each option based on a 100 percent success rate, and without putting a financial value on any potential side-effects. However, these are important factors to consider.

The flushing option has the benefit of being a natural process with limited side effects, but comes with a lower likelihood of success, since it is highly dependent on weather conditions.

Destratification has proven to be effective in other similar scenarios internationally, but still has a risk of failure related to if the timing is not appropriately managed, or the compressor or power supply fail. Side effect costs could be the sound of the compressor, or the impact that the bubble curtain has on the visual value of the Lake.

Based on the NIWA report, the capping option appears to have the highest likelihood of success. However, there are significant community considerations surrounding the concept of remediating a natural lake through the addition of non-natural products (or "chemicals"). This could result in some members of the local community choosing to no-longer use the Lake for swimming (e.g., families with young children).

Table 4.3 shows the effect that hypothetical risk and side-effect costs could have on the NPV of each option.

Table 4.3: NPV with Potential Risk and Side Effect Costs

	Flushing	Destratification	Low Cost Capping	High Cost Capping
Estimated Success rate	50%	85%	75%	95%
Side Effect Costs	Nil	50% reduction in value of sightseeing	10% reduction in swimming	10% reduction in swimming
Effect on NPV				
Stable	$620,000 (-$992,000)	$1,469,000 (-$636,000)	$1,631,000 (-$671,000)	$492,000 (-$189,000)
Natural Recovery	$127,000 (-$498,000)	$648,000 (-$353,000)	$826,000 (-$371,000)	-$527,000 (-$104,000)
Deteriorates	$1,238,000 (-$1,610,000)	$2,579,000 (-$1,006,000)	$2,707,000 (-$1,075,000)	$1,855,000 (-$306,000)

Including some hypothetical estimates of risk and potential side-effect costs reduces the overall NPV of the options, however, the economic viability does not significantly change, with no further options becoming unviable.

Further analysis shows that even under the most conservative counterfactual scenario of natural recovery, flushing remains economically viable (i.e., a BCR of greater than 1) as long as it has a greater than 38 percent chance of being successful, destratification a greater than 31 percent chance, and low-cost capping a greater than 13 percent chance.

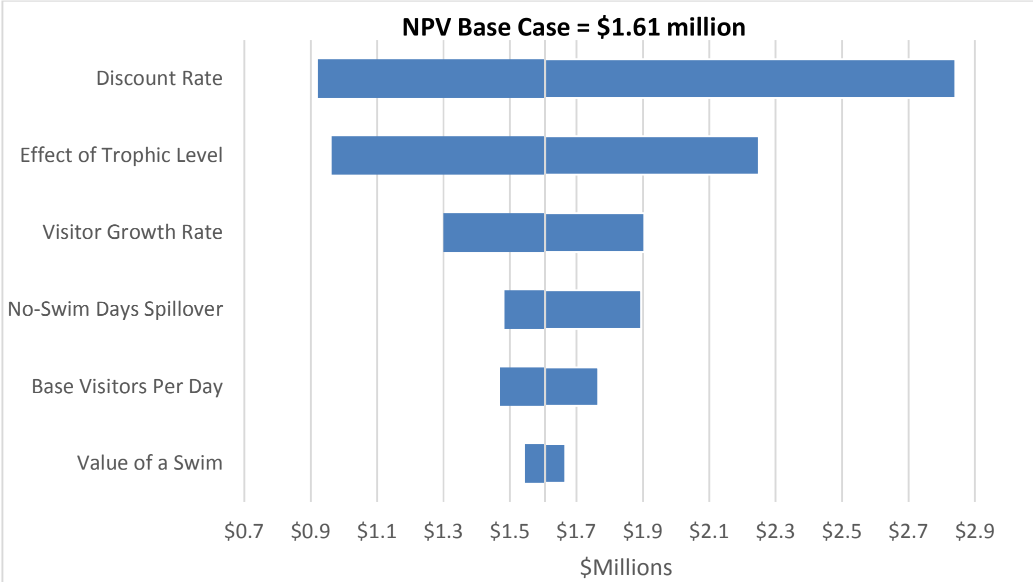
4.2 The Results are Sensitive to some Key Assumptions

This study had uncertainty in many of the assumptions necessary to obtain an NPV for the Lake's remediation. These included:

- Economic model variables including the discount rate
- The values of the willingness-to-pay for recreational variables
- The volumes and growth rate of the recreational activities occurring at the Lake
- The additional value that improvement of water quality would add to recreational activities at the Lake.

Figure 4.2 below shows a visual representation of how some of the results are impacted when assumptions are varied by plus or minus 50 percent. It uses the example of the effect on the NPV of 'flushing option' when compared against the 'stable' counterfactual.

Figure 4.2: Impact on NPV Benefits of Different Assumptions (Flushing compared against stable counterfactual)



The graph shows that discount rate is the variable that has the greatest overall effect on NPV. The model uses a discount rate of 6 percent as recommended by the New Zealand Treasury. When this is reduced to 3 percent, the NPV significantly increases, as benefits in the future are given more weighting.

The effect that trophic level has on the value of recreational activities also has a significant effect. The model assumes that for every 0.1 drop in trophic level, the value of recreational activities at the Lake increases by 0.5%. When this reduces to an increase of 0.25%, the NPV decreases from $1.61m to $0.96m. The reason that this seemingly small variable has such a large effect is that it is working off such a large base, i.e., the total value of recreation at the Lake was determined to be $1.3m per year, and NPV is calculated over a 30-year period.

Other variables that appear to be changing to a greater degree can have a relatively small impact, such as varying the value of a swim from $20 to $10 only decreases the NPV from $1.61m to $1.54.

The other variables represented in the graph show the effect of varying the base number of estimated visitors in 2018 from 192, varying the visitor growth rate from 2.1% and varying the spillover effect of no-swim days from 50%.

The full results of the sensitivity tests across all remediation options are available in Appendix A.

4.3 Geographic Distribution of Beneficiaries

Section 2.4 detailed the range of groups that will experience benefits from improvement of water quality at the Lake. Table 4.4 provides a breakdown of how these benefits can be apportioned geographically.

Table 4.4: Geographic Distribution of Benefits

Area	Proportion of Benefits
Lake Hayes	43 percent
Lake Hayes South	31 percent
The District (QLDC residents)	13 percent
Outside of the District (Including ORC residents from outside QLDC, national and foreign tourists)	13 percent

Percentages are based on the 2018 QLDC population figures that gave a breakdown of the population of Lake Hayes on an average day, including 'usually resident' population and 'visitors'. Usually resident population were estimated as being 43 percent of the total in the area. This estimation was supported by consultation with residents of the local area.

Visitor numbers were broken down by:

- Visitors in commercial accommodation
- Visitors in private residences
- Day visitors

These are not able to be assigned into the geographic categories, so some reasonable assumptions were made.

Visitors in private residences were assigned to the Lake Hayes South category. This group are predominantly within the Lake Hayes South subdivisions, who can easily visit the Lake via a short 5-minute drive or a 20-minute walk (an hour to the main swimming site).

Day visitors were 22 percent of visitors, (13 percent of people in the area). These are the visitors within driving distance, e.g., within the Queenstown Lakes District.

The remainder of visitors were classified as staying in commercial accommodation, i.e. tourists to Queenstown Lakes District. This would equate to approximately 115 people on a peak day. This figure would include residents of ORC, as those who live outside of the Queenstown Lakes District are predominantly not within easy driving distance for a daily visit, so would be staying in accommodation if they were visiting.

Appendix A Sensitivity Testing Results

The following Tables show the results of sensitivity testing of various variables on each remediation option, compared to the stable counterfactual scenario.

Table A.1: Effect on NPV of Discount Rate (used 6 percent)

	9 percent	3 percent	Percentage Change
Flushing	$923,000	$2,841,000	208%
Destratification	$1,344,000	$3,416,000	154%
Low-cost-capping	$1,556,000	$3,588,000	131%
High-cost-capping	$256,000	$1,447,000	466%

Table A.2: Effect on NPV of Base Average Visitors Per Day[19] (used 192)

	96 Visitors	288 Visitors	Percentage Change
Flushing	$1,470,000	$1,765,000	20%
Destratification	$1,953,000	$2,268,000	16%
Low-cost-capping	$2,150,000	$2,464,000	15%
High-cost-capping	$529,000	$844,000	59%

Table A.3: Effect on NPV of Visitor Growth Rate (used 2.1%)

	1.05% per annum	3.15 % per annum	Percentage Change
Flushing	$1,300,000	$1,905,000	47%
Destratification	$1,772,000	$2,413,000	36%
Low-cost-capping	$1,969,000	$2,610,000	33%
High-cost-capping	$349,000	$990,000	184%

Table A.4: Effect on NPV of Value of a Swim (used $20.00)

	$10 per swim	$30 per swim	Percentage Change
Flushing	$1,545,000	$1,667,000	8%
Destratification	$2,032,000	$2,163,000	6%
Low-cost-capping	$2,229,000	$2,360,000	6%
High-cost-capping	$609,000	$739,000	21%

[19] I.e., the number first used as a base 2018 figure, that is then increased by a growth rate over the next 30 years.

Table A.5: Effect on NPV of No-Swim Days Spillover of (used 50%)

	25%	75%	Percentage Change
Flushing	$1,483,000	$1,895,000	28%
Destratification	$1,964,000	$2,433,000	24%
Low-cost-capping	$2,161,000	$2,629,000	22%
High-cost-capping	$540,000	$1,009,000	87%

Table A.6: Effect on NPV of the Effect of Trophic Level on Recreational Value (used 0.5% increase (or decrease) for every 0.1 change in Trophic Level)

	0.25% per 0.1	0.75% per 0.1	Percentage Change
Flushing	$963,00	$2,249,000	134%
Destratification	$1,273,000	$2,922,000	129%
Low-cost-capping	$1,470,000	$3,119,000	112%
High-cost-capping	-$150,000	$1,498,000	N/A[20]

[20] Percentage change not applicable to negative NPV

CASTALIA
Strategic advisors

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