Ōtūwharekai Potential Actions

Task 4 – Cost Effectiveness Analysis

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Executive summary

The Ministry for the Environment (MfE) and Environment Canterbury (ECan) engaged NIWA and AgResearch to assist with development of an "Ōtuwharekai Action Plan". The work to prepare the Action Plan is made up of four parts. Part one was carried out by Environment Canterbury and Agresearch (farm systems), and parts two (in-lake mitigations), three (catchment interventions) and four (cost and impact) and final compilation of the Action Plan, are to be delivered by NIWA.

The brief for this report, the fourth component of the Ō Tū Wharekai (Ashburton Lakes) Action Plan, is to address the cost-effectiveness of intervention options. Annualised 50-year Lifecyle costs were estimated for mitigation options identified in parts 1,2 and 3.

Cost Effectiveness Analysis (CEA) is used to compare the cost of different options to achieve a minimum targeted outcome. while lifecycle costing was used to estimate costs over a fifty-year period, which also enables comparison of various mitigation bundles. Cost-effectiveness is presented in terms of reduction in catchment tonnes of Nitrogen, Phosphorus, and a weighted combination of the two.

Table 8 summarises economic metrics for mitigation of nitrogen. Across all lakes, removal of cattle and intensive winter grazing (IWG) have the largest impacts on N. Removal of cattle in Autumn or Winter has the lowest estimated cost at $3,000-$4,000/t removed. The denitrification walls are also relatively cost effective ($1,000-$6,000/t), although the assumptions about impact need to be refined through further investigation. IWG has the next lowest cost at $5,000-$7,000/t.

Stock exclusion is the most expensive option for most lakes (average $76,000/t). The use of wetlands to reduce nitrogen loads varies from $4,000 - $65,000/t. The full report notes that assuming that fencing an existing wetland has the same benefit as constructing a new wetland may be optimistic, so the apparently cheaper wetland actions may not be as cost-effective as they appear. The abatement costs appear similar to other estimates reported in New Zealand.

Table 1-1: Summary of interventions with N reductions.

Lake	Mitigation	Catchment N reduction (t)	Catchment N reduction %	Annual LCC	Cost / reduction t
Constructed wetland	Back Māori Lake	2.26	30%	$81 000	$36 000
	Front Māori Lake	10.90	19%	$44 000	$4 000
	Lake Camp	1.45	28%	$11 000	$8 000
	Lake Clearwater	0.83	7%	$22 000	$27 000
	Lake Denny	1.06	15%	$25 000	$23 000
	Lake Emily	0.78	10%	$51 000	$65 000
	Lake Heron	3.47	12%	$81 000	$23 000
	Lake Roundabout/Emma	1.94	30%	$12 000	$6 000
	Lake Trinity	3.26	60%	$9 000	$3 000
Denitrification wall	Lake Clearwater	5.70	48%	$6 000	$1 000
	Lake Heron	4.85	16%	$30 000	$6 000
Improve N use	Front Māori Lake	2.84	5%	$129 000	$45 000
	Lake Heron	5.50	18%	$251 000	$46 000
Italian rye grass	Back Māori Lake	0.32	4%	$15 000	$46 000
	Front Māori Lake	1.07	2%	$115 000	$107 000
	Lake Clearwater	0.83	7%	$8 000	$9 000

Lake	Mitigation	Catchment N reduction (t)	Catchment N reduction %	Annual LCC	Cost / reduction t
	Lake Denny	0.27	4%	$25 000	$92 000
	Lake Heron	0.46	2%	$93 000	$199 000
	Lake Trinity	0.14	3%	$10 000	$74 000
IWG removal	Back Māori Lake	7.09	94%	$41 000	$6 000
	Front Māori Lake	19.87	34%	$96 000	$5 000
	Lake Denny	5.80	83%	$29 000	$5 000
	Lake Emily	5.60	73%	$32 000	$6 000
	Lake Heron	7.86	26%	$53 000	$7 000
No wintering of cattle	Front Māori Lake	19.62	34%	$77 000	$4 000
	Lake Denny	3.63	52%	$15 000	$4 000
	Lake Emily	1.19	16%	$3 000	$3 000
	Lake Heron	11.21	37%	$48 000	$4 000
Plantain	Front Māori Lake	0.64	1%	$24 000	$38 000
	Lake Heron	0.61	2%	$16 000	$26 000
Remove cattle in Autumn	Back Māori Lake	1.19	16%	$5 000	$4 000
	Front Māori Lake	1.19	2%	$5 000	$4 000
	Lake Heron	3.57	12%	$15 000	$4 000
	Lake Roundabout/Emma	3.63	56%	$10 000	$3 000
Stock exclusion	Back Māori Lake	0.38	5%	$55 000	$146 000
	Front Māori Lake	1.82	3%	$155 000	$85 000
	Lake Camp	0.24	5%	$11 000	$46 000
	Lake Denny	0.18	3%	$8 000	$45 000
	Lake Emily	0.11	1%	$7 000	$60 000
	Lake Heron	0.58	2%	$115 000	$199 000
	Lake Roundabout/Emma	0.32	5%	$8 000	$24 000
	Lake Trinity	0.27	5%	$1 000	$3 000

Table 9 summarises economic metrics for mitigation of phosphorus. The cost-per-tonne of phosphorus reduced is higher for P than N. Removal of cattle is again the most cost-effective option with costs ranging from $15,000-$24,000/t Stock exclusion is again relatively expensive (average $433,000 / t) and wetlands average $280,000/t. P-inactivation ranges from $165,000-$1.9 million per equivalent catchment reduction tonne. The cost of removal of phosphorus via a sediment trap in Lake Clearwater is estimated to cost $75,000/t. The abatement costs are in a similar range to those reported in other analyses in New Zealand.

Table 1-2: Summary of interventions with P reductions.

Intervention	Lake	Catchment P reduction (t)	Catchment P reduction %	Annual LCC	Cost / reduction t
Constructed wetland	Back Māori Lake	0.11	1.4%	$81 000	$772 000
	Front Māori Lake	0.61	1.0%	$44 000	$72 000
	Lake Camp	0.08	1.6%	$11 000	$135 000
	Lake Clearwater	0.21	1.8%	$22 000	$106 000
	Lake Denny	0.10	1.4%	$25 000	$256 000

Intervention	Lake	Catchment P reduction (t)	Catchment P reduction %	Annual LCC	Cost / reduction t
	Lake Emily	0.06	0.7%	$51 000	$919 000
	Lake Heron	1.79	5.9%	$81 000	$45 000
	Lake Roundabout/Emma	0.13	2.0%	$12 000	$93 000
	Lake Trinity	0.07	1.3%	$9 000	$122 000
IWG removal	Back Māori Lake	0.01	0.1%	$41 000	$4 261 000
	Front Māori Lake	0.04	0.1%	$96 000	$2 389 000
	Lake Denny	0.01	0.1%	$29 000	$3 076 000
	Lake Emily	0.01	0.1%	$32 000	$4 583 000
	Lake Heron	0.02	0.1%	$53 000	$2 369 000
No wintering of cattle	Front Māori Lake	3.76	6.4%	$77 000	$20 000
	Lake Denny	0.63	9.0%	$15 000	$24 000
	Lake Emily	0.21	2.7%	$3 000	$15 000
	Lake Heron	2.00	6.7%	$48 000	$24 000
P-inactivation	Lake Clearwater	1.52	12.6%	$250 000	$165 000
	Lake Denny	0.32	4.5%	$60 000	$190 000
	Lake Emily	0.16	2.1%	$40 000	$245 000
	Lake Roundabout/Emma	0.19	3.0%	$370 000	$1938 000
Remove cattle in autumn	Back Māori Lake	0.21	2.8%	$5 000	$23 000
	Front Māori Lake	0.21	0.4%	$5 000	$23 000
	Lake Heron	0.62	2.1%	$15 000	$24 000
	Lake Roundabout/Emma	0.63	9.8%	$10 000	$16 000
Sediment trap	Lake Clearwater	1.44	12.0%	$109 000	$75 000
Stock exclusion	Back Māori Lake	0.03	0.4%	$55 000	$1 747 000
	Front Māori Lake	3.36	5.8%	$155 000	$46 000
	Lake Camp	0.03	0.5%	$11 000	$441 000
	Lake Denny	0.03	0.4%	$8 000	$274 000
	Lake Emily	0.01	0.2%	$7 000	$467 000
	Lake Heron	0.54	1.8%	$115 000	$215 000
	Lake Roundabout/Emma	0.03	0.5%	$8 000	$229 000
	Lake Trinity	0.02	0.3%	$1 000	$47 000

1 Introduction

The Ministry for the Environment (MFE) have engaged NIWA to develop an Ō Tū Wharekai Action Plan. The Ō Tū Wharekai Action Plan is intended to be a structured and well-evidenced programme of work to stop further degradation of the lakes and wetlands of the Ō Tū Wharekai catchment and deliver a pathway to restoration.

The work to prepare the Action Plan is made up of four parts. Part one was carried out by Environment Canterbury and AgResearch (farm systems), and parts two (in-lake mitigations), three (catchment interventions) and four (cost and impact) and final compilation of the Action Plan, are to be delivered by NIWA.

The brief for this report, the fourth component of the Ō Tū Wharekai (Ashburton Lakes) Action Plan, is to address the cost-effectiveness of intervention options. This report evaluates the intervention options described in the other three parts of the action plan. Part 1 comprises four farm environmental reduction reports written by environment Canterbury and reviewed by Agresearch (Gilmer, 2022 a, 2022 b, 2022 c, 2022 d). Part 2 comprises a report reviewing available and feasible in-lake mitigations (Hofstra et al., 2022). Part 3 comprises a report that examines the catchment load characteristics and makes recommendations for appropriate on-land interventions.

2 Method

2.1 Cost-effectiveness analysis

When mitigation benefits can be converted to a common unit of impact (e.g., load reductions), the most efficient option can be determined without monetisation of benefits. Cost effectiveness analysis (CEA) compares the cost of different options to achieve a minimum targeted outcome. CEA is used when the monetary value of benefits of an investment are not easily monetizable and is a highly suitable assessment method when water quality objectives are well defined (Balana et al., 2015). Cost-effectiveness is estimated by dividing the costs of an investment by a non-monetised benefit to estimate the average cost per unit of the benefit created from a project (e.g., cost per kilogramme (kg) of nitrogen reduced). This ratio is called an incremental cost-effectiveness ratio (ICER), and is used to rank options in terms of cost per unit of benefit. The option with the smallest ICER is the most cost-effective.

2.2 Lifecyle costing

This analysis converts all costs of mitigation to a 50-year Life Cycle Cost (LCC). The LCC approach is consistent with approach used for Auckland's Freshwater Management Tool (Muller et al., 2020). The conversion to a common period of time enables comparison of different mitigation bundles. Cost components of each mitigation bundle include outlay, capital, maintenance and opportunity costs. The discount rate used is 5%, consistent with the default rate recommended by the New Zealand Treasury[1].

2.3 Benefit metric

Necessary load reductions of nitrogen (N) and phosphorus (P) across the Ō Tū Wharekai (Ashburton Lakes) catchment area were converted to a single metric using a weighting dependent on the reductions required for each lake. The catchment loads and required reductions are sourced from Kelly et al. (2021) who distinguished between small (<33%), moderate (34-66%), and large (>66%) reductions. For the purposes of this economic analysis, it was assumed that a reduction target that falls in the middle of each range will achieve the policy objective. That is, a "small" reduction is 17%, moderate is 50%, and large is 83%.

The relative importance (weighting) of N and P reductions, W_{TN} and W_{TP}, was calculated as follows:

$$W_{TN} = \frac{R_{TN}}{R_{TN} + R_{TP}} \quad \text{and} \quad W_{TP} = \frac{R_{TP}}{R_{TN} + R_{TP}}$$

where R_{TN} and R_{TP} are the percentage catchment load reductions required for total N (TN) and total P (TP). In other words, if both TN and TP required the same percentage reduction both W_{TN} and W_{TP} would be 0.5. Four of the lakes only require TN reductions so these have $W_{TN} = 1$ and $W_{TP} = 0$ (Table 2-1). No lakes require only TP reductions. This weighting assumes that meeting policy objectives is **equally important in all lakes.**

[1] https://www.treasury.govt.nz/information-and-services/state-sector-leadership/guidance/financial-reporting-policies-and-guidance/discount-rates

Table 2-1: TN and TP annual reductions and weightings by lake catchment.

Lake catchment	Catchment load (t)		Catchment reduction required		Reduction %		Weighting	
	TN	TP	TN	TP	R_{TN}	R_{TP}	W_{TN}	W_{TP}
Lake Camp	5.18	0.18	Large		83%	0%	1.00	0.00
Lake Clearwater	12.01	1.83	Moderate	Large	50%	83%	0.38	0.62
Lake Denny	6.99	0.38	Moderate	Large	50%	83%	0.38	0.62
Lake Emily	7.66	0.33	Large	Moderate	83%	50%	0.62	0.38
Lake Emma	6.46	0.23	Moderate	Large	50%	83%	0.38	0.62
Lake Heron	30.04	9.28	Small		17%	0%	1.00	0.00
Lake Māori - front	58.26	1.96	Large		83%	0%	1.00	0.00
Lake Māori - back	7.52	0.21	Large		83%	0%	1.00	0.00

The required catchment reductions in terms of tonnes of N and P vary significantly between lakes. Lake Emma, for example, only requires a reduction of 3.23 t N, while front Māori lake requires 58.26 t N. The benefit metric therefore converts the weights to a per-tonne basis as follows:

$$B_{TN} = \begin{cases} W_{TN} RT_{TN}, & W_{TN} > 0 \\ 0, & W_{TN} = 0 \end{cases}$$

$$B_{TP} = \begin{cases} W_{TP} RT_{TP}, & W_{TP} > 0 \\ 0, & W_{TP} = 0 \end{cases}$$

where RT_{TN} and RT_{TP} are the catchment reductions required in tonnes. Table 2-2 shows that the benefit per tonne is higher for P than N, and highest at Lake Emma where only a few tonnes are needed. This metric assumes that there is no benefit to reductions where the policy objective is assumed to have already been met.

Table 2-2: Relative benefit per tonne of nutrient reduction in each lake catchment.

Lake catchment	Relative benefit per tonne reduced	
	B_{TN}	B_{TP}
Lake Camp	0.23	0.00
Lake Clearwater	0.06	0.41
Lake Denny	0.11	1.98
Lake Emily	0.10	2.30
Lake Emma	0.12	3.27
Lake Heron	0.20	0.00
Lake Māori - front	0.02	0.00
Lake Māori - back	0.16	0.00
Lake Trinity	0.12	0.99

Lake Trinity was not included in the modelling by Kelly et al. (2021) but was included in load reduction calculations by Canterbury Regional Council so has been included in this economic analysis.

Without having any estimates of catchment loads, it has been assumed that the values for B_{TN} and B_{TP} are the average of the other 8 lakes.

The total benefit of a mitigation in a lake catchment is therefore calculated as $B_{TN}N + B_{TP}P$ where N and P are the loads of nutrients (expressed in tonnes) removed by a mitigation.

2.4 Mitigation load reduction estimates

The load reductions for catchment mitigations are sourced from four farm environmental load reduction reports written by Environment Canterbury (Gilmer, 2022 a, 2022 b, 2022 c, 2022 d).

2.5 Limitations of the analysis

Cost-effectiveness can only be estimated for options where the impact on water quality can be estimated. For the in-lake mitigations, only indicative values are provided in this analysis because there is currently not enough information about impact and limited information about costs. Revisions to cost assessments for mitigations are presently underway through Environment Canterbury but these are not yet available to include in the analysis. When these are complete, the lifecycle costings may be adjusted accordingly. Cost-effectiveness can be recalculated by dividing the new cost by estimated reductions.

The benefit metric assumes policy objectives in all lakes are equally important, and that the estimated reductions will achieve the objectives with certainty. Reality is considerably less certain and the assessment would ideally be updated over time as actions are completed and new monitoring data becomes available.

3 Assessed mitigations and costs

3.1 P-inactivation

The assessment of P-inactivation is based on figures from the Lake Okaro case study (Gibbs, 2010). Lake Okaro has an area of 31 ha and a 5-year median TP of 35 mg/ m_3 according to LAWA[2]. The case study used a product called Aqual-P[3] to deactivate P. The lake had two applications of Aqual-P, 60 t granular in 2009 and 120 t fine powder in 2010 (Gibbs, 2010). The second application was more effective, so the second dose rate was applied in the following calculations.

Mitigation costs are estimated only for the four lakes that required TP reductions. Accurate dose estimation requires taking sediment cores so, for this desktop analysis, it was assumed that required dose is approximately proportional to lake area and median TP concentration from Kelly et al. (2021). Estimated dose (D_i) for lake *i* is therefore:

$$D_i = 120(TP_i/TP_{Okaro})(area_i/area_{Okaro})$$

Aqual-P currently sells for $2,500/ t (Blue Pacific Minerals, personal communication, October 13, 2022). Delivery and application costs (e.g., barge hire) add an estimated 20 per cent to costs, making the final cost $3,000 /t.

P-inactivation may be effective for internal load control for 10 years (Welch & Cooke, 1999). However, dosing will need to be more regular if P inputs from the catchment continue. A frequency of **5-yearly dosing** has been assumed for this assessment.

It is assumed that the estimated Aqual-P doses will achieve policy objectives for TP, so the benefit is simply W_{TP} for each lake.

The incremental cost effectiveness ratio (ICER) ranges from $100,000 for Lake Denny to $590,000 for lake Emma (Table 3-1). It should be noted that these estimates are highly sensitive to estimated dose and size and duration of impact.

Table 3-1: P-inactivation cost and cost-effectiveness

Lake	Area (ha)	Mean TP (mg/m³)	Dose (t)	Cost per application	Annualised cost (5-yearly dose)	Benefit (W_{TP})	ICER
Lake Clearwater	208	16.6	382	$1 150 000	$250 000	0.62	$400 000
Lake Emma	189	27	566	$1 700 000	$370 000	0.62	$590 000
Lake Denny	6	125.5	87	$260 000	$60 000	0.62	$100 000
Lake Emily	21	29.2	67	$200 000	$40 000	0.38	$110 000

3.2 Perch removal

Part 2 of the Ō Tū Wharekai Potential Actions recommends investigating the willingness of stakeholders to remove the pest fish perch. Although perch increase turbidity and negatively affect biodiversity, there is no direct link with the reduction in fish population and the policy objectives for N and P, so an ICER cannot be estimated. However, costs can be estimated based on the perch fish-

[2] You appear to be missing your footnote
[3] https://www.bpmnz.co.nz/environmental/

down programme for Lake Wainamu, described in Surrey and Neale (2015) as well as through a possible fish poisoning programme.

3.2.1 Perch fish down

Lake Wainamu has an area of 14.5 ha with a maximum depth of 12 m. The control programme used an average of 48 littoral nets each night for 62 nights over 11 years. Total catch was 17 722 perch. The catch per unit effort (CPUE) was therefore 285 (Surrey & Neale, 2015).

The 30 m long monofilament nets used in the control programme cost $179 each[4] and it was assumed that each net can be used an average of 10 times before it is damaged. Small boat hire was approximately $400 per day[5]. Labour requirements for Lake Wainamu were 3 people per boat plus a shore crew of 10 people. Labour cost was approximately $30/ hour based on an average full-time salary of $55,000. Travel costs were also significant since the nearest town to the Ashburton lakes (Ashburton) was 75 km away. Total costs were expected to be around $4,400 per trip (Table 3-2). Perch are said to be good eating[6], but may not be sold or traded, so the catch has no market value.

Table 3-2: Perch fishing costs

Unit	Cost per unit	Units per trip	Total cost per trip	Explanation
30 m gill net	$18	48	$860	$179 divided by 10 uses
Boat hire/day	$400	2	$800	2 boats
Crew labour/ hour	$30	24	$720	6 people x 4 hours
Shore labour/ hour	$30	20	$600	10 people x 2 hours
Travel labour/ hour	$30	32	$960	16 people x 2 hours
Travel mileage/ km	$0.83	600	$500	4 vehicles x 150 km
Total cost per trip			$4 400	

The two lakes with perch are Lake Denny (6.2 ha) and Lake Camp (44.3 ha). If it is assumed that average depth is half the maximum depth (2 m and 13 m respectively), the lake volumes are approximately 62,000 m^3 and 2.9 million m^3 . Lake Wainamu volume is 870 000 m^3.

Fish-down costs are adjusted for the different volumes of the lakes relative to Lake Wainamu. A perch fish-down programme would therefore cost at least $7,200 per year for Lake Denny and $332,000 for Lake Camp. Since Surrey and Neale (2015) found that the fishing effort was insufficient to noticeably reduce the perch population in Lake Wainamu, the fish down presented here is likely to be insufficient to achieve the reduction needed in perch. As a result, costs estimated here are probably significantly lower than the true cost required to have a significant impact.

Fish poison such as Rotenone is a relatively expensive approach to remove perch in large water bodies. However, this would require a once-off cost, rather than being an ongoing requirement. Rotenone costs around $80 USD per gallon, and requires 1 gallon per acre-foot of water volume[7].

Converting to metric units and NZD suggests a cost of $0.11 / m^3. Rotenone would cost $331,000 for Lake Camp and $7,200 for Lake Denny. However, this cost is only for the poison and does not include

[4] https://www.marine-deals.co.nz/fishing-nets/bait-net-40md-0-35mm-mono-30m
[5] https://www.getmyboat.com/boat-rental/New-Zealand/?currency=NZD
[6] https://fishingmag.co.nz/coarse-fishing/perch
[7] https://www.bassresource.com/fish_biology/rotenone.html

delivery or application costs so again represents a minimum. The Rotenone may also need to be pumped to the bottom of the lakes to be effective. Since the Rotenone is a once-off cost (unless perch are re-introduced or somehow evade the poison), the estimated annualised LCC over 50 years is significantly lower than that for fish-down LCC (Table 3-3).

Table 3-3: Minimum expected cost for perch control or removal.

Lake	Area (ha)	Fish-down LCC	Rotenone LCC
Lake Camp	44.3	$332 300	$18 126
Lake Denny	6.2	$7 200	$392

This assessment does not include the cost of harm to other species that might result from net fishing or poisoning.

3.3 Other in-lake mitigations

The cost-effectiveness of erecting or growing wind breaks, in-lake wave baffles, and increasing macrophyte coverage were not estimated because these mitigations were not recommended Ō Tū Wharekai Potential Actions - Task 2, and it is unclear whether they would be helpful (section 6.3).

3.4 Wetlands

The costs of establishing effective wetlands are based on use of the sites identified in the farm environmental load reduction reports by Environment Canterbury. These reports identified 32 wetland sites, over a total area of 174 hectares. Some are existing wetlands that only require fencing (5 sites), some involve expanding existing wetlands with fencing and planting (12 sites), while the remaining sites will require wetland construction (Table 3-4).

Table 3-4: Wetland recommendations activities required by area

Activity required	Count of sites	Hectares
Construct, fence and plant	15	30.1
Fence and plant	12	43.1
Fence only	5	101.2
Total	32	174.4

3.4.1 Wetland costs

Wetland construction costs were sourced from Tanner (2013), which indicates that per-hectare costs decrease with increasing size (Figure 3-1). It was assumed that the wetlands would be the partially excavated type. Costs were adjusted for inflation using the consumers price index (CPI). Inflation-adjusted costs including construction, planting and fencing range from $236 ,000 / ha for a site less than 1 ha, to $120,000/ ha for a site greater than 5 ha. This cost range is similar to that used in Muller et al. (2020), which describes a capital cost of $164,000/ ha for a small wetland and $126,000/ ha for a large wetland.

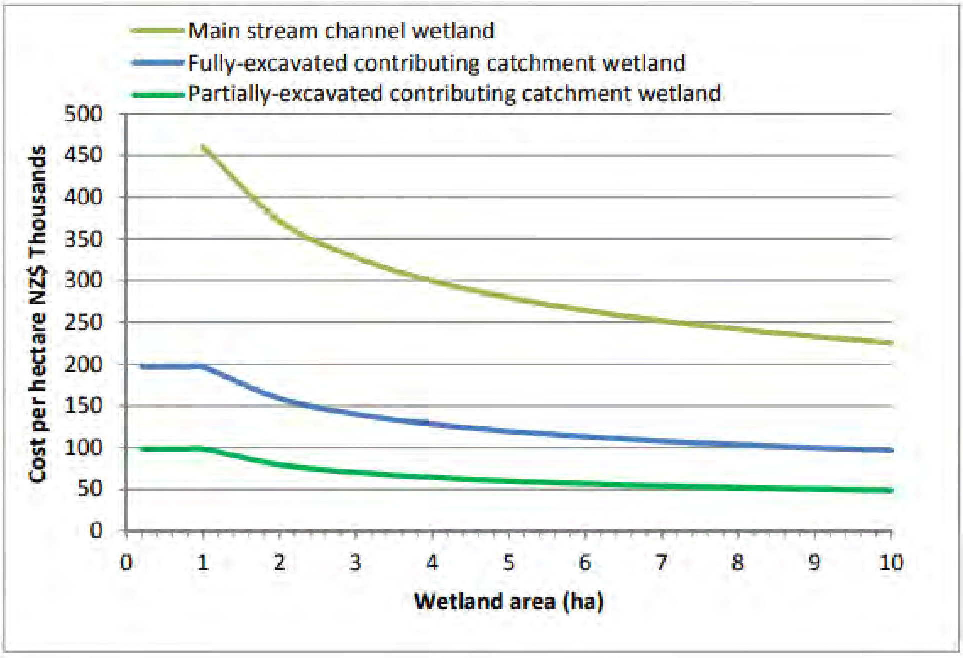

Figure 3-1 - Wetland construction costs per hectare (Tanner et al. 2013)

The breakdown of costs for a constructed wetland are approximately 55% for headworks and excavation, 38% for planting, and 7% for fencing (Praat et al. 2015). For sites that do not require excavation or planting, the cost is reduced accordingly.

Annual wetland maintenance (weed and pest control) costs were sourced from Tanner et al. (2013, p. 25) and estimated at $360/ ha after adjusting for inflation. If the activity is fencing only (no planting), then maintenance was assumed to be 75% less ($90/ ha) because it was assumed that existing vegetation is mature.

Wetland creation or supplementation implies the area is retired from grazing. Retirement opportunity costs are assumed to be $378/ ha/ year as per Muller et al. (2020).

3.4.2 Wetland impacts

The farm environmental reports by Environment Canterbury specified a percentage reduction in N and P for each wetland site. It was assumed that fencing an existing wetland would provide the same benefit as constructing or planting an entirely new wetland. The percentage reductions were multiplied by the total catchment load and the proportion of catchment occupied by each farm to estimate reductions in tonnes.

The benefit metrics for each lake were calculated as per section 2.3, and ranged from 0.14 for Lake Clearwater to 0.68 for Lake Heron. A benefit of 1 would mean the mitigation achieved the full reduction required to meet policy objectives. The annualised LCC ranged from $14,700 for Lake Trinity to $140,000 for Back Māori Lake (Table 3-5). The ICER, or cost divided by benefit, ranges from $30,650 for Lake Roundabout/ Emma to $250,000 for Lake Emily.

Table 3-5: Wetlands cost effectiveness

Lake	Wetland Hectares	N reduction (t)	P reduction (t)	Benefit metric	Annualised LCC	Cost / benefit
Back Māori Lake	14.9	2.26	0.11	0.36	$81 300	$224 400
Front Māori Lake	41.57	10.90	0.61	0.23	$43 800	$194 400
Lake Camp	1.92	1.45	0.08	0.34	$11 300	$33 500
Lake Clearwater	4.24	0.83	0.21	0.14	$22 200	$160 600
Lake Denny	4.22	1.06	0.10	0.30	$24 600	$80 800
Lake Emily	72.5	0.78	0.06	0.20	$50 900	$250 000
Lake Heron	26.5	3.47	1.79	0.68	$80 500	$118 400
Lake Roundabout/ Emma	5.3	1.94	0.13	0.64	$19 589	$30 650
Lake Trinity	3.2	3.26	0.07	0.48	$8 500	$17 900

The farm environmental reports assumed wetlands would reduce the catchment loads by 30%-60% for N and 50-80% for P. However, Tanner and Sukias (2022) suggest that performance might be at the bottom of the range for a cool climate, which would mean 22-28% for N. The cost effectiveness of the wetlands will reduce significantly if this is the case. In addition, fencing existing wetlands may only increase their performance above the existing level by 20%, so would not be as effective as assumed (Tanner, Principal Scientist - Aquatic Pollution, NIWA, personal communication, October 2022).

3.5 Denitrification trenches

A denitrification wall or trench is a subsurface trench filled with an organic carbon source (e.g., wood chips or sawdust) that intercepts shallow groundwater flows (Hudson et al. 2018). An appropriately installed wall is assumed to remove more than 95% of nitrates (Tanner and Sukias, 2022). The bioavailable carbon is expected to last 23 years (Schmidt & Clark, 2012), and would be replaced in years 24 and 48.

For the purposes of costing, a generic denitrification trench is assumed to be 1.5 m wide and 2 m deep. The trench is filled with wood chip up to the last metre, upon which soil may be replaced (Tanner, personal communication, October 2022). The most significant costs are for excavation and fill. An 8-tonne excavator cost $180 / hour including a driver in 2019 (Muller, 2019), so total excavation cost is assumed to be around $200/ hour in 2022. With a bucket capacity of 0.38 m^3, one excavator is expected to move around 190 m^3 of earth in a 6-hour workday, assuming access is not too difficult. The excavation cost per m^3 is therefore $6.32.

Excavation volume is doubled to account for filling the trench as well, and it was assumed that excess fill will be used somewhere on site (that is, there are no disposal costs). The cheapest woodchip (green and unprocessed) costs around $400 for a 10 m^3 truckload delivered (Groundzone Tree Care, personal communication, 15/09/2022). Sawdust is an alternative carbon source but appears to have a similar cost of $40/ m^3 excluding delivery (https://www.cypress-sawmill.co.nz/garden-sawdust-auckland).

A 100 m denitrification trench therefore costs, at a minimum, $3,800 for the excavation and $6,000 for the carbon source (totalling $9,800).

Tanner and Sukias (2022) recommend a 4 km denitrification wall for Lake Heron and an 0.8 km wall for Lake Clearwater. The N reductions cannot be accurately predicted without further work, so this analysis optimistically assumes the denitrification walls will achieve 95% of the required N reductions in both catchments. Since Lake Clearwater requires larger P reductions than N reductions, the benefit metric is only 0.36 for Clearwater catchment. The resulting ICER is $31 700 for Lake Heron and $16,700 for Lake Clearwater (Table 3-6).

Table 3-6: Denitrification trench cost-effectiveness.

Catchment	Length (km)	Benefit	Annual LCC	ICER
Lake Heron	4	0.95	$30 200	$31 700
Lake Clearwater	0.8	0.36	$6 030	$16 700

3.6 Sedimentation traps

Sedimentation traps are depressions constructed to slow sediment-laden water. Major capital costs include excavation, bank battering, and overheads for design and project management. The size of the pond required depends on catchment size and flood flows. As a rough estimate, a pond area of 16 m^2 is required per hectare of catchment (Tanner, personal communication, November 2022). The average depth should be 3 m with battered sides.

The excavation cost is assumed to be $6.32/ m^3, the same as for denitrification trenches. Muller (2019) recommends a battering cost of $6.20/ m, which is $6.75 in 2022 dollars. Overheads for pond design, consent, and project management are assumed to add 10% to the capital cost of the pond. The edge of the pond may be planted with flax or *carex* but planting is not critical to the function so is not included in the cost.

Sedimentation traps require regular dredging to remove sediment. It was assumed that dredging will take place every 2 years on average, removing a third of the volume of the pond in sediment (Tanner, personal communication, November 2022). Depending on the size of the pond, it may be more cost effective to use a suction dredge rather than draining and excavating. Small-scale dredging costs were estimated to be $300 per hour ($446 in 2022 dollars) with a daily volume of 500 m^3 for a treatment pond in the Bay of Plenty (Analytical & Environmental Consultants, 2007, p. 43).

Tanner and Sukias (2022) recommend using sedimentation traps for the Western entrance to Lake Clearwater. The total catchment area coming off the Western hills is 1 100 ha. The total pond area would therefore need to be 17.6 ha. If it was a single pond and roughly circular the battered perimeter would be 470 m.

The capital costs would total $370,000 including $333,000 for excavation, $3,150 for battering, and $33,600 for overheads. Dredging for maintenance would take 35 days and cost $82,000 every two years. The annualised cost would be $108,900. Assuming the intervention completely achieves the P reduction target, the benefit metric would be 0.62 and the ICER $175,600.

3.7 Stock exclusion

The environmental load reduction reports (Gilmer, 2022 a, 2022 b, 2022 c, 2022 d) provided tables of stock exclusion recommendations with fence lengths, buffer widths, and percentage N and P reductions for each farm in each lake catchment. This economic analysis excluded stock exclusion actions marked as already completed. As for the wetland calculations, the percentage reductions are multiplied by lake catchment load and the proportion of the farm within the lake catchment to convert to load of nutrient reduced (tonnes).

Fencing costs were $16.80/ m for 8-wire non-electric on rolling land in January 2017 (Ministry for Primary Industries, 2017). After adjusting for inflation, the cost was assumed to be $19.90/ m in October 2022. The annual cost of fence maintenance is $0.16/ m (Muller et al., 2020). Fencing is expected to last 25 years, so the present value over 50 years is $28.96/ m.

It was assumed that the buffer strips would be grass rather than the more expensive planted option. Buffer maintenance costs (e.g., weed control) and lost grazing opportunity costs are $0.25/ m and $0.10/m respectively for a 5 m buffer (Muller et al., 2020). The annual cost is therefore $700/ ha, with a present value of $12,800/ ha. In comparison, a planted buffer would incur additional capital costs of at least $25,000/ ha for planting (Ministry for Primary Industries, 2017).

The annualised LCC ranged from $700 for the smallest lake (Trinity) to $155,100 for Front Māori Lake (Table 3-7). The cost divided by relative benefit ranged from $14,400 for Lake Trinity to $4,127,200 for Front Māori Lake.

Table 3-7: Stock exclusion cost effectiveness

Lake	Length (km)	Buffer area (ha)	Benefit metric	Annualised LCC	Cost / benefit
Back Māori Lake	23.36	25.86	0.06	$55 200	$913 300
Front Māori Lake	65.50	73.10	0.04	$155 100	$4 127 200
Lake Camp	4.20	6.30	0.06	$11 100	$197 500
Lake Clearwater	1.00	1.50	0.03	$2 600	$76 300
Lake Denny	5.80	5.80	0.08	$7 900	$103 900
Lake Emily	2.50	3.75	0.04	$6 600	$152 300
Lake Heron	48.99	53.51	0.11	$115 200	$1 016 700
Lake Roundabout/ Emma	3.00	4.50	0.15	$7 900	$52 600
Lake Trinity	0.27	0.40	0.05	$700	$14 400

If it turns out that stock exclusion can be done more cheaply than the literature suggests, the LCC could be adjusted accordingly. For example, a 4-wire electric fence costs half as much as an 8-wire non-electric fence and would reduce the total cost by 35%. Using the buffer for cut-and-carry, rather than retiring the land, could decrease the net cost by up to 33%.

3.8 Removal of intensive winter grazing blocks

The environmental load reductions reports (Gilmer, 2022 a, 2022 b, 2022 c, 2022 d) specify areas where intensive winter grazing (IWG) will cease, and the associated reductions in catchment loads. In Lake Clearwater and Lake Camp there is no IWG so the estimated costs and reductions for those lakes are not included.

The cost of ceasing IWG depends on the value of the feed produced, minus costs, minus the opportunity cost of not growing pasture. The cost estimates are based on the economics of fodder beets, a commonly grown crop for non-lactating cows in cooler areas such as Canterbury. Average yield is expected to be 17 t DM/ ha[8]. The energy value of fodder beet is 12 MJ/ kg DM[9]. The value of fodder is $21 / t DM / MJ. Costs are around $2,500 / ha[8]. The net benefit from the crop is therefore $4,284 - $2500 = $1,784 / ha.

Over the 150 days required for a crop of fodder beet, 3-11 tDM/ ha pasture could be grown instead (DairyNZ, 2020). The energy value of summer/ autumn pasture ranges from 9 – 11.5 MJ/ kg DM[9]. Assuming a relatively low pasture yield of 6 tDM / ha (due to no irrigation and low N), the value of the pasture would be $1,200/ ha. The net cost of removing IWG is therefore $1,784 - $1,200 = $584/ ha. This cost estimate is sensitive to yield and fodder values. Fodder value is significantly higher during a drought; however, fodder beet yield would also suffer from drought stress.

The annual LCC ranges from $8,800 for Lake Trinity to $95,900 for Front Māori Lake (Table 3-8). The load reductions estimated by Environment Canterbury exceed what is required to achieve lake targets for Back Māori Lake and Lake Heron. The ICER ranges from $34,200 for Lake Heron to $233,500 for Front Māori Lake.

Table 3-8: Total impact and cost effectiveness of removal of IWG.

Lake	Ha	N reduction (t)	P reduction (t)	Benefit metric	Annual LCC	ICER
Back Māori Lake	70.00	7.09	0.01	1.14	$40 900	$36 000
Front Māori Lake	164.19	19.87	0.04	0.41	$95 900	$233 500
Lake Denny	50.00	5.80	0.01	0.64	$29 200	$45 500
Lake Emily	54.90	5.60	0.01	0.57	$32 100	$56 700
Lake Heron	90.00	7.86	0.02	1.54	$52 600	$34 200
Lake trinity	15.00	1.74	0.00	0.22	$8 800	$40 000

3.9 No wintering of cattle

The environmental reductions reports (Gilmer, 2022 a, 2022 b, 2022 c, 2022 d) specify reductions in cattle numbers during winter, and the associated reductions in catchment loads.

Liveweight prices are currently around $2.40-2.90/ kg depending on the grade of cattle [10]. The midpoint $2.70/ kg is used for this analysis. Cattle gain approximately 0.87 kg/ day in winter[11], so the revenue for 91 days is $214/ head. The cost of various inputs (e.g., healthcare) is assumed to be $60 based on bi-monthly figures in Addis et al. (2021).

It was not clear from the environmental reduction reports whether the sheep stocking rate would increase to make up for the removal of cattle. The calculations do not include a load increase from sheep so it was assumed that there will be no increase in sheep numbers.

One steer would require around 500-600 kgDM of fodder for this weight gain (Beef+Lamb New Zealand, 2017). This fodder may be worth $128 ($0.21/ Mj/ kg DM) but if it cannot be grazed by

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[8] https://www.dairynz.co.nz/feed/crops/fodder-beet/growing-fodder-beet/
[9] https://www.dairynz.co.nz/feed/supplements/feed-values/
[10] https://nzfarmlife.co.nz/trading-cattle-margins/
[11] https://beeflambnz.com/sites/default/files/factsheets/pdfs/fact-sheet-119-growing-cattle-fast-on-pasture.pdf

sheep, it is lost and has no economic value. The opportunity cost of not wintering cattle is therefore assumed to be $214 - $61 = $153 / head.

The ICER for removing cattle in winter ranges from $5,100 for Lake Emily to $188,600 for Front Māori Lake (Table 3-9).

Table 3-9: Total impact and cost effectiveness of removal of cattle wintering. .

Lake	Cattle heads	N reduction (t)	P reduction (t)	Benefit metric	Annual LCC	ICER
Front Māori Lake	500	19.62	3.76	0.41	$76 500	$188 600
Lake Denny	100	3.63	0.63	1.64	$15 300	$9 300
Lake Emily	20	1.19	0.21	0.60	$3 100	$5 100
Lake Heron	313	11.21	2.00	2.20	$47 900	$21 800
Lake Trinity	20	0.63	0.13	0.21	$3 100	$14 800

3.10 Removal of cattle in autumn

The environmental reduction reports (Gilmer, 2022 a, 2022 b, 2022 c, 2022 d) also recommend removing cattle from several sensitive blocks where they currently graze for 59 days in Autumn. It was assumed that these are larger cattle that are sold before winter (i.e., 500 kg starting weight). To gain 0.875 kg / day these cattle would need 744 kgDM over 59 days. Similar to the wintering calculations, it was assumed that this grazing would be lost. Costs are assumed to be $40/ head (Addis et al., 2021) so the opportunity cost of not grazing cattle in autumn is $98/ head.

The ICER for removing cattle in autumn ranges from $4 000 for Lake Roundabout/ Emma to $199,300 for Front Māori Lake (Table 3-10).

Table 3-10: Total impact and cost effectiveness of removal of cattle in Autumn .

Lake	Cattle heads	N reduction (t)	P reduction (t)	Benefit metric	Annual LCC	ICER
Back Māori Lake	50	1.19	0.21	0.19	$4 900	$25 700
Front Māori Lake	50	1.19	0.21	0.02	$4 900	$199 300
Lake Heron	150	3.57	0.62	0.70	$14 700	$21 100
Lake Roundabout/ Emma	100	3.63	0.63	2.48	$9 800	$4 000

3.11 Use of Italian annual ryegrass

The recommendations by Environment Canterbury (Gilmer, 2022 a, 2022 b, 2022 c, 2022 d) include replacing perennial pasture in some blocks with Italian annual ryegrass. Annual pasture renewal costs $1,000/ ha (Journeaux, 2021), but the cost should be compensated by increased productivity. The literature shows that increasing pasture renewal rates from 2% to 8% can increase dry stock farm profits by 18% (Sanderson & Webster, 2009), and a renewal rate of 20% (5-yearly) could increase profit by $409 / ha.

Although annual ryegrass has higher yield than perennial grass and can provide higher quality feed during autumn and winter[12], there is insufficient information to determine if annual pasture renewal will increase or decrease net profit/ha on a dry stock farm. For the purpose of this analysis, it was

[12] https://pasture.io/ryegrass/annual

assumed that the productivity increase from annual pasture will reduce the cost of the option to $500 / ha.

Conversion to annual ryegrass only reduces N leaching, and the ICER ranges from $144,000 for Lake Clearwater to $5.2 million for Front Māori lake (Table 3-11). The cost for Lake Clearwater is relatively low because the farm report (Gilmer, 2022 c) estimates relatively high reductions from only 15 ha re-grassed.

Table 3-11: Total impact and cost effectiveness of Italian annual ryegrass.

Lake	Ha	N reduction (t)	Benefit	Annual LCC	ICER
Back Māori Lake	30	0.32	0.05	$15 000	$288 200
Front Māori Lake	230	1.07	0.02	$115 000	$5196 000
Lake Clearwater	15	0.83	0.05	$7 500	$144 600
Lake Denny	50	0.27	0.03	$25 000	$852 600
Lake Heron	185	0.46	0.09	$92 500	$1018 100
Lake Trinity	20	0.14	0.02	$10 000	$591 700

3.12 Incorporate plantain

Plantain is a herb that may reduce N leaching when incorporated in the pasture mix[13]. Plantain may offer pasture production benefits (particularly in the summer) in addition to reducing leaching, but those benefits were not quantified. The cost of including and maintaining plantain in the sward is assumed to be $120/ ha to maintain a steady-state 15% plantain, from the Dairy NZ Plantain Establishment Cost Calculator[14]. Plantain was recommended only for Front Māori Lake and Lake Heron, and the ICERs are $1.8 million and $135 200 respectively.

Table 3-12: Total impact and cost effectiveness of Plantain.

Lake	Ha	N reduction (t)	Benefit	Annual LCC	ICER
Front Māori Lake	200	0.64	0.01	$24 000	$1 816 200
Lake Heron	135	0.61	0.12	$16 200	$135 200

3.13 Improve N use efficiency

Soil sampling and testing was recommended by Environment Canterbury for Front Māori Lake and Lake Heron catchments to minimise the risk of applying excess N. The N mass balance budget involves the use of regular testing with quick test strips (10-20 cores per paddock) and 5 samples/ ha /year for anaerobically mineralizable N laboratory tests (Foundation for Arable Research, 2020).

Quick test strips cost $63 for 50[15] and it was assumed that 10 samples / ha will be required twice per year. Labour cost is $2.92 per test based on 5 minutes per test and an hourly rate of $30. The annual lab tests cost $18 each[16] so $100 / ha for 5 samples plus postage. The total cost is therefore $173/ ha. The financial benefits of improved N efficiency were not quantified due to lack of information.

[13] https://www.dairynz.co.nz/feed/crops/plantain/
[14] https://www.dairynz.co.nz/media/5791871/plantain-establishment-costs-v1.xlsx
[15] https://www.ecplabchem.co.nz/nitrate-test-strips-50s-1415914
[16] https://portal.hill-laboratories.com/lab-tests

The ICER is $2.2 million for Front Māori Lake and $233,000 for Lake Heron, although it should be noted that both the reductions and net costs are very uncertain.

Table 3-13: Total impact and cost effectiveness of improved N use.

Lake	Ha	N reduction (t)	Benefit	Annual LCC	ICER
Front Māori Lake	744	2.84	0.06	$129 100	$2 198 300
Lake Heron	1446	5.50	1.08	$250 900	$233 000

4 Summary of results

For each lake catchment there are multiple possible interventions identified in parts 1, 2 and 3 of the action plan. An estimated life cycle cost has been calculated for each intervention. The most appropriate cost-effectiveness metric to consider depends on whether decision makers are interested in the N target, P target, or both. The choice of metric also depends upon whether decision makers are most interested in cost per reduction tonne or cost per percentage of target achieved. This summary presents cost-effectiveness results for N and P separately, and the combined benefit metric.

4.1.1 Interventions with N reductions

Across all lakes, removal of cattle and IWG have the largest impacts on N (Table 4-1). Removal of cattle in Autumn or Winter has the lowest estimated cost at $3,000-$4,000/ t. The denitrification walls are also relatively cost effective ($1,000-$6,000/ t), although the assumptions about impact should be refined through further investigation. IWG has the next lowest cost at $5,000-$7,000. Stock exclusion is the most expensive option for most lakes (average $76,000 / t). The cost of wetlands varies from $4,000 - $65,000/t. As noted in the wetlands section, the assumption that fencing an existing wetland has the same benefit as constructing a new wetland may be optimistic, so the cheaper wetland actions may not be as cost-effective as they appear.

Some of these interventions score well on cost effectiveness compared with the N abatement costs of $10,000-$40,000/ t suggested by Daigneault et al (2018). Their national-scale analysis of multiple agri-environmental policies included dairy grazing land, which has higher economic costs for de-intensification.

Table 4-1: Summary of interventions with N reductions.

Lake	Mitigation	Catchment N reduction (t)	Catchment N reduction %	Annual LCC	Cost / reduction t
Constructed wetland	Back Māori Lake	2.26	30%	$81 000	$36 000
	Front Māori Lake	10.90	19%	$44 000	$4 000
	Lake Camp	1.45	28%	$11 000	$8 000
	Lake Clearwater	0.83	7%	$22 000	$27 000
	Lake Denny	1.06	15%	$25 000	$23 000
	Lake Emily	0.78	10%	$51 000	$65 000
	Lake Heron	3.47	12%	$81 000	$23 000
	Lake Roundabout/Emma	1.94	30%	$12 000	$6 000
	Lake Trinity	3.26	60%	$9 000	$3 000
Denitrification wall	Lake Clearwater	5.70	48%	$6 000	$1 000
	Lake Heron	4.85	16%	$30 000	$6 000
Improve N use	Front Māori Lake	2.84	5%	$129 000	$45 000
	Lake Heron	5.50	18%	$251 000	$46 000
Italian rye grass	Back Māori Lake	0.32	4%	$15 000	$46 000
	Front Māori Lake	1.07	2%	$115 000	$107 000
	Lake Clearwater	0.83	7%	$8 000	$9 000
	Lake Denny	0.27	4%	$25 000	$92 000
	Lake Heron	0.46	2%	$93 000	$199 000
	Lake Trinity	0.14	3%	$10 000	$74 000

Lake	Mitigation	Catchment N reduction (t)	Catchment N reduction %	Annual LCC	Cost / reduction t
IWG removal	Back Māori Lake	7.09	94%	$41 000	$6 000
	Front Māori Lake	19.87	34%	$96 000	$5 000
	Lake Denny	5.80	83%	$29 000	$5 000
	Lake Emily	5.60	73%	$32 000	$6 000
	Lake Heron	7.86	26%	$53 000	$7 000
No wintering of cattle	Front Māori Lake	19.62	34%	$77 000	$4 000
	Lake Denny	3.63	52%	$15 000	$4 000
	Lake Emily	1.19	16%	$3 000	$3 000
	Lake Heron	11.21	37%	$48 000	$4 000
Plantain	Front Māori Lake	0.64	1%	$24 000	$38 000
	Lake Heron	0.61	2%	$16 000	$26 000
Remove cattle in Autumn	Back Māori Lake	1.19	16%	$5 000	$4 000
	Front Māori Lake	1.19	2%	$5 000	$4 000
	Lake Heron	3.57	12%	$15 000	$4 000
	Lake Roundabout/Emma	3.63	56%	$10 000	$3 000
Stock exclusion	Back Māori Lake	0.38	5%	$55 000	$146 000
	Front Māori Lake	1.82	3%	$155 000	$85 000
	Lake Camp	0.24	5%	$11 000	$46 000
	Lake Denny	0.18	3%	$8 000	$45 000
	Lake Emily	0.11	1%	$7 000	$60 000
	Lake Heron	0.58	2%	$115 000	$199 000
	Lake Roundabout/Emma	0.32	5%	$8 000	$24 000
	Lake Trinity	0.27	5%	$1 000	$3 000

4.1.2 Interventions with P reductions

The cost-per-tonne of reductions is higher for P than N. Removal of cattle is again the most cost-effective option with costs ranging from $15 000-$24 000 / t (Table 4-2). Stock exclusion is again relatively expensive (average $433 000/ t) and wetlands average $280 000/ t. P-inactivation ranges from $165,000-$1.9 million per equivalent catchment reduction tonne. The sediment trap in Lake Clearwater is estimated to cost $75 000/ t.

Daigneault et al (2018) suggested P abatement costs of $50 000-$200 000/ t, so cattle removal and sediment ponds appear cost-effective by comparison.

Table 4-2: Summary of interventions with P reductions.

Intervention	Lake	Catchment P reduction (t)	Catchment P reduction %	Annual LCC	Cost / reduction t
Constructed wetland	Back Māori Lake	0.11	1.4%	$81 000	$772 000
	Front Māori Lake	0.61	1.0%	$44 000	$72 000
	Lake Camp	0.08	1.6%	$11 000	$135 000
	Lake Clearwater	0.21	1.8%	$22 000	$106 000
	Lake Denny	0.10	1.4%	$25 000	$256 000
	Lake Emily	0.06	0.7%	$51 000	$919 000

Intervention	Lake	Catchment P reduction (t)	Catchment P reduction %	Annual LCC	Cost / reduction t
	Lake Heron	1.79	5.9%	$81 000	$45 000
	Lake Roundabout/Emma	0.13	2.0%	$12 000	$93 000
	Lake Trinity	0.07	1.3%	$9 000	$122 000
IWG removal	Back Māori Lake	0.01	0.1%	$41 000	$4 261 000
	Front Māori Lake	0.04	0.1%	$96 000	$2 389 000
	Lake Denny	0.01	0.1%	$29 000	$3 076 000
	Lake Emily	0.01	0.1%	$32 000	$4 583 000
	Lake Heron	0.02	0.1%	$53 000	$2 369 000
No wintering of cattle	Front Māori Lake	3.76	6.4%	$77 000	$20 000
	Lake Denny	0.63	9.0%	$15 000	$24 000
	Lake Emily	0.21	2.7%	$3 000	$15 000
	Lake Heron	2.00	6.7%	$48 000	$24 000
P-inactivation	Lake Clearwater	1.52	12.6%	$250 000	$165 000
	Lake Denny	0.32	4.5%	$60 000	$190 000
	Lake Emily	0.16	2.1%	$40 000	$245 000
	Lake Roundabout/Emma	0.19	3.0%	$370 000	$1 938 000
Remove cattle in Autumn	Back Māori Lake	0.21	2.8%	$5 000	$23 000
	Front Māori Lake	0.21	0.4%	$5 000	$23 000
	Lake Heron	0.62	2.1%	$15 000	$24 000
	Lake Roundabout/Emma	0.63	9.8%	$10 000	$16 000
Sediment trap	Lake Clearwater	1.44	12.0%	$109 000	$75 000
Stock exclusion	Back Māori Lake	0.03	0.4%	$55 000	$1 747 000
	Front Māori Lake	3.36	5.8%	$155 000	$46 000
	Lake Camp	0.03	0.5%	$11 000	$441 000
	Lake Denny	0.03	0.4%	$8 000	$274 000
	Lake Emily	0.01	0.2%	$7 000	$467 000
	Lake Heron	0.54	1.8%	$115 000	$215 000
	Lake Roundabout/Emma	0.03	0.5%	$8 000	$229 000
	Lake Trinity	0.02	0.3%	$1 000	$47 000

4.1.3 All interventions and combined benefits

Table 4-3 summarizes the benefit of each intervention in terms of both N and P reduction targets. As described in section 2.3, an intervention that achieved both N and P reduction targets for a lake would have a benefit score of 100%. Because the benefit metric depends on a percentage reduction, the relative cost is higher for large lakes where many tonnes need to be removed to achieve the target.

For almost all the lakes, the most cost-effective option is removal of cattle in winter or autumn. For Lake Camp, the most cost-effective option is a constructed wetland. For Lake Clearwater, the denitrification wall looks to most cost-effective. For Lake Trinity, the most cost-effective option is stock exclusion, an anomaly because the length of fencing required is short and the estimated benefits are high (Gilmer, 2022 c).

Most catchments will require a combination of interventions to achieve the targets. However, the combined effectiveness of interventions is uncertain and simply summing the reductions is probably inappropriate.

Table 4-3: Summary of all interventions and combined benefit.

Lake	Mitigation	Benefit	Annual LCC	Cost / benefit
Back Māori Lake	Remove cattle in Autumn	0.19	\$4 900	\$25 700
	IWG removal	1.14	\$40 900	\$36 000
	Italian rye grass	0.05	\$15 000	\$288 200
	Constructed wetland	0.36	\$140 000	\$386 200
	Stock exclusion	0.06	\$55 200	\$913 300
Front Māori Lake	No wintering of cattle	0.41	\$76 500	\$188 600
	Remove cattle in Autumn	0.02	\$4 900	\$199 300
	IWG removal	0.41	\$95 900	\$233 500
	Constructed wetland	0.23	\$77 200	\$342 300
	Plantain	0.01	\$24 000	\$1 816 200
	Improve N use	0.06	\$129 100	\$2 198 300
	Stock exclusion	0.04	\$155 100	\$4 127 200
	Italian rye grass	0.02	\$115 000	\$5 196 000
Lake Camp	Constructed wetland	0.34	\$22 100	\$65 800
	Stock exclusion	0.06	\$11 100	\$197 500
	Perch fish-down	N/A	\$332 300	N/A
	Perch poisoning	N/A	\$18 100	N/A
Lake Clearwater	Italian rye grass	0.05	\$7 500	\$144 600
	Constructed wetland	0.14	\$42 600	\$307 700
	P-inactivation	0.62	\$250 000	\$403 200
	Denitrification wall	0.36	\$6 000	\$16 700
	Sediment trap	0.62	\$108 900	\$175 600
Lake Denny	No wintering of cattle	1.64	\$15 300	\$9 300
	IWG removal	0.64	\$29 200	\$45 500
	P-inactivation	0.62	\$60 000	\$96 800
	Stock exclusion	0.08	\$7 900	\$103 900
	Constructed wetland	0.30	\$47 800	\$157 000
	Italian rye grass	0.03	\$25 000	\$852 600
	Perch fish-down	N/A	\$7 200	N/A
	Perch poisoning	N/A	\$400	N/A

Lake	Mitigation	Benefit	Annual LCC	Cost / benefit
Lake Emily	No wintering of cattle	0.60	$3 100	$5 100
	IWG removal	0.57	$32 100	$56 700
	P-inactivation	0.38	$40 000	$105 300
	Stock exclusion	0.04	$6 600	$152 300
	Constructed wetland	0.20	$67 900	$333 400
Lake Heron	P-inactivation	0.62	$370 000	$596 800
	Remove cattle in Autumn	0.70	$14 700	$21 100
	No wintering of cattle	2.20	$47 900	$21 800
	IWG removal	1.54	$52 600	$34 200
	Plantain	0.12	$16 200	$135 200
	Constructed wetland	0.68	$142 100	$209 000
	Improve N use	1.08	$250 900	$233 000
	Stock exclusion	0.11	$115 200	$1 016 700
	Denitrification wall	0.95	$30 200	$31 800
Lake Roundabout/Emma	Italian rye grass	0.09	$92 500	$1 018 100
	Remove cattle in Autumn	2.48	$9 800	$4 000
	Constructed wetland	0.64	$19 600	$30 700
	Italian rye grass	0.02	$10 000	$591 700
	Stock exclusion	0.15	$7 900	$52 600
Lake Trinity	Stock exclusion	0.05	$700	$14 400
	Italian rye grass	0.02	$10 000	$591 700
	No wintering of cattle	0.21	$3 100	$14 800
	Constructed wetland	0.48	$14 700	$30 900
	IWG removal	0.22	$8 800	$40 000

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6 Glossary of abbreviations and terms

Aqual-P17	
B_{TN}	Relative benefit per tonne nitrogen reduced
B_{TP}	Relative benefit per tonne phosphorus reduced
CEA	Cost-effectiveness analysis
CLUES	
D	
Di	estimated dosage
DM	dry matter
Denitrification trenches	a subsurface trench filled with an organic carbon source that intercepts shallow groundwater flows
IWG	
MFE	Ministry for the Environment
LCC	Life Cycle Cost
Ō Tū Wharekai Action Plan	
ICER	incremental cost-effectiveness ratio
Rotenone	
R_{TN}	percentage catchment load reductions required for nitrogen
R_{TP}	percentage catchment load reductions required for phosphorus
S	
Sedimentation traps	on-land depressions constructed to slow sediment-laden water
TN	total nitrogen level
TP	total phosphorus level
U_{TN}	
U_{TP}	
W_{TN}	relative importance (weighting) of nitrogen reductions
W_{TP}	relative importance (weighting) of phosphorus reductions

[17] https://www.bpmnz.co.nz/environmental/

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Ōtūwharekai Potential Actions

Part 2 - In lake mitigations

Prepared for Ministry for the Environment and Environment Canterbury

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Executive summary

The Ōtūwharekai lakes (Ashburton Lakes) are situated in the South Canterbury high country, within a basin at the head of the Southern Branch of the Hakatere/Ashburton River. These lakes and the surrounding wetlands are nationally significant, highly vulnerable and have immense cultural value. However, there has been a steady decline in water quality (TLI, trophic lake index) since annual monitoring began in 2005. Targets have been set for TLI and LakeSPI (submerged plant indicators) for the lakes, and all of them currently fail to meet those targets based on the most recent data, with the exception of Lake Camp, which meets its LakeSPI target.

The Ministry for the Environment (MFE) engaged NIWA to develop an Ōtūwharekai Action Plan to address a recommendation from a Ministerial briefing (BRF 487), to "address nutrient reductions in the lakes". The work to prepare the Action Plan is made up of four parts. Part 1 was carried out by AgResearch (Farm System), and Part 2 (In-lake mitigations), Part 3 (Catchment interventions beyond reductions), Part 4 (Cost and Impact), and a final compilation of the Action Plan, were carried out by NIWA.

This report addresses the in-lake mitigations component (part 2) by:

- focussing on lake condition and restoration goals (i.e., Canterbury Land and Water Regional Plan objectives), and identifying which lakes may require in-lake mitigation,
- assessing feasible mitigation options on a lake-by-lake basis, and
  - this includes identifying lake priorities for mitigation (ecological values and pressures), and the causes of degradation, and
  - review of available options (e.g., flocculating agents, sediment capping agents, aeration to break up stratification, pest control), and expected lake health outcomes.
- recommending a course of action to halt degradation and deliver a pathway to restoration of these lakes.

Most of the Ōtūwharekai lakes have scored high, or high to moderate, for their ecological values. The exception was Lake Denny, which scored moderate to low. Pressures and threats to all lakes scored in the high to moderate category. These scores indicate that although most of the lakes still have high ecological values, they are also at risk. This finding supports the requirement for urgent mitigation action to restore and protect all of the lakes.

The combined scores for ecological value and risk, provide a means to prioritise the lakes for mitigation actions (from highest to lowest) as follows:

Heron > Māori East > Emily = Clearwater > Māori West > Camp = Emma > Denny

Improving lake water quality, while native plants are still present, provides the best chance for improvement of lake ecological condition. Once the macrophytes are lost (an inevitable consequence of continued decline in water quality), the cost, time and difficulty to restore the lakes will increase, and the probability of success will diminish significantly.

Recommended actions are:

1. Turn off the tap - Reduce catchment nutrient loads (see Part 3, Tanner and Sukias 2022). The only sustainable "lever" currently available to improve lake condition is reduction of the nutrient and sediment loads delivered from the respective lake catchments.

2. Manage in-lake nutrient issues where they exist with sediment capping/P inactivation agents. This is a "band-aid" approach. Reapplication of these materials will be necessary over time, leading to ongoing expense. There are caveats to the use of P inactivation agents that must be considered (and in some cases resolved), before they can be used.

3. Prevent any new incursions of alien invasive species or non-native species. Continue the existing surveillance programme and design a targeted community awareness campaign for the Ōtūwharekai lakes, focusing on prevention of new incursions of invasive species and reducing the risk of spreading non-native fish between lakes. Investigate the appetite/will amongst mana whenua, mandated agencies with management responsibility, and other stakeholders, to support a programme for the removal of perch.

4. Continue to monitor the lakes (as recommended in Bayer and Meredith 2020), including additional targeted monitoring to improve understanding of the processes contributing to internal nutrient cycling in each of the lakes.

1 The Ōtūwharekai Lakes - introduction

The Ōtūwharekai lakes (also known as the Ashburton Lakes) are situated in the South Canterbury high country, within a basin at the head of the Southern Branch of the Hakatere/Ashburton River. The area is characterised by an inter-montane wetland system and several high-country lakes. The lakes mostly drain to the Ashburton River, although Lake Heron drains to the Rakaia River, and Lake Denny drains to the Rangitata River. With the exceptions of Lakes Heron, Camp and Clearwater (which have deeper basins), the Ōtūwharekai lakes are small and relatively shallow. The lakes are located at a similar altitude (600 - 700 m asl) and are of glacial origin.

Intensive forest clearance occurred in the catchment between c. AD 1200 and 1600, followed by catchment erosion and increased sedimentation. These changes were most obvious in the sediment cores from Lake Clearwater and the Māori Lakes (Woodward et al. 2014). Large areas of the basin are now farmed, with higher altitude areas classified as Public Conservation Land. The lake catchments were predominantly tussock grassland under sheep and beef grazing in 2013, with wetland areas associated with some of the lakes (de Winton et al. 2013b), but more recently cropping has been observed in the catchment (e.g., de Winton and Burton 2017).

The area is home to a number of native plant, bird and fish species including species that are rare or threatened (Hoosen 2015, Canterbury Land and Water Regional Plan (CLWRP) 2019). Its ecological values make the basin nationally significant, and in 2007 was incorporated in the national Arawai Kākāriki wetland restoration programme managed by the Department of Conservation (DOC) (Te Rūnanga o Arowhenua et al. 2010, Drinan and Robertson undated, Bayer and Meredith 2020). The area is of “immense cultural significant to Ngāi Tahu Whanui”, both as an important seasonal mahinga kai area and historically as destinations along the route between settlements on the east and west coast of the South Island (Te Rūnanga o Arowhenua et al. 2010). The area is also important for recreation, with bach settlements and campsites, supporting fishing, swimming and boating (Bayer and Meredith 2020).

1.1 Project background

Monitoring of the Ōtūwharekai lakes since 2005, has shown a decline in water quality over that period and the need for action to prevent irreversible state-change in the lakes was recognised. Minister Parker requested a briefing (BRF-487) on Ōtūwharekai and the wider Canterbury region’s freshwater bodies after seeing a press article regarding the degraded state of the Ōtūwharekai lakes. In this briefing were recommendations which were endorsed by Minister Parker. The first recommendation was “To work with Environment Canterbury and Ngāi Tahu to address nutrient reductions in the lakes.”

The Ministry for the Environment (MFE) engaged NIWA, AgResearch and private consultants to develop an Ōtūwharekai Action Plan to address this recommendation. The work to prepare the Action Plan is made up of four parts. Part 1 was carried out by AgResearch (Farm System), and Parts 2 (In-lake mitigations), 3 (Catchment interventions beyond reductions) and 4 (Cost and Impact) and final compilation of the Action Plan, were carried out by NIWA.

1.2 Scope of Part Two – in-lake mitigations

This present report addresses Part 2. The scope of this task was to "Identify which lakes may need/require in-lake mitigation"……… "Most of the lakes are increasingly tending towards a eutrophic state but are yet to exhibit structural changes such as loss of the macrophyte beds or sustained increase in turbidity/loss of clarity. It may therefore be premature to require in-lake mitigations for most of the lakes. For those lakes that may require in-lake mitigations feasible options will need to be assessed on a lake-by-lake basis. This includes reviewing the available range of mitigations (flocculating agents, sediment capping, aeration, for example), and quantifying expected impacts/projections on lake health outcomes" (excerpt from the contract).

2 Methodology/Approach

2.1 Lake condition

As part of the process leading to identifying appropriate in-lake mitigations, we first describe existing lake conditions, and then identify target "restored lake condition" by reviewing existing aims, targets or outcomes for the lakes of the Central Canterbury alpine rivers subregion. For example, the Canterbury Land and Water Regional Plan states objectives (outcomes) and limits for:

- eutrophication (e.g., Trophic Level Index (TLI) maximum of 2 to 4),
- ecological health (e.g., > 70% hypolimnion Dissolved Oxygen (DO), <10°C, high to excellent condition using LakeSPI), and
- other important values such as "high naturalness".

While our review focused on biophysical lake condition, we also considered cultural and social expectations where these appeared strongly linked to certain biophysical conditions. Consideration of the appropriateness of plan targets and limits was beyond the scope of this project.

2.2 Lake prioritisation for mitigation

In this section we assessed the lakes and prioritised them for mitigation actions. The prioritisation process for lake management considered values, current or future pressures and threats, and their magnitude (Champion and de Winton 2012, Champion 2014). This approach was based on expert opinion with agreed weighting of multiple aspects. This prioritisation process was limited by the information that was available for the lakes, and we have highlighted critical knowledge gaps. We acknowledge that this methodology provides one way to prioritise the lakes, but is not the only method. For example, cultural and community values could also be assessed and integrated, however that was outside of the scope of this project.

The prioritisation process included lakes that are still in relatively good condition, because mitigations that protect or maintain current good condition can be more cost effective and likely to succeed than measures aimed at restoring substantially degraded lakes. To obtain the information required to guide lake prioritisation, we accessed data from online databases/portals (e.g., Freshwater Ecosystems of New Zealand, New Zealand Freshwater Fish Database (NZFFD), LakeSPI, Lakes 380), publicly available reports from ECan, as well as from other sources of biophysical information relevant to lakes of the Ashburton basin.

2.3 Identification of causal factors

The causes of lake degradation influence the selection of possible mitigation measures. The factors responsible for degradation of lake water quality are unknown for several of the lakes. Until these factors are identified, key data and information required to guide mitigation actions also remain unknown, which creates the potential for information gaps. Therefore, the identification of the sources of sediment/nutrients or other contributory factors driving water quality may be subject to substantial information gaps. Climate change may also influence lake conditions and potentially influence the choice and effectiveness of mitigation measures. Information gaps were identified as the Ōtūwharekai Action Plan was developed, and the likely effectiveness of mitigation measures was described wherever possible. Recommendations have been made for targeted investigations to fill critical information gaps.

2.4 In-lake mitigation actions

The potential mitigation options were reviewed and combinations of measures that could be used at each lake were identified. Mitigation measures such as the application of flocculants and P-inactivation agents, flushing, aeration, biomanipulation and freshwater pest control were considered. Our assessment considered the efficacy, immediacy, and longevity of mitigation measures where possible. We considered how far mitigation measures would go towards meeting the target "restored lake condition". Where possible, information on the optimal timing of mitigation measures was provided to guide their use, noting that the extent to which we could progress this component of the work was dependent on the adequacy of data and information available.

3 Ōtūwharekai lake condition

3.1 Past and current lake condition

In this section information and data for eight Ōtūwharekai lakes (Bayer and Meredith 2020) is summarised to describe past condition for the lakes in general, and current condition of each lake separately.

Past condition

Long-term monitoring data for the lakes is limited, and prior to 2005 there is little consistent information on the state of the lakes against which changes can be measured (Bayer and Meredith 2020). For example, only Lakes Camp, Clearwater, Emma, Emily, and Heron were included in the 1986 lake inventory (Livingstone et al. 1986), with data available for few water quality characteristics.

Since the establishment of ECan's helicopter-based sampling programme in 2005 there has been a more consistent approach to water quality monitoring. Data derived from this programme are used by ECan to report on water quality and ecological state and trends in the high-country lakes (e.g., Bayer and Meredith 2020). However, it should be noted that the summer and autumn only (December and May) collection of data, most likely covers only the period of the year when the lake productivity is highest (Bayer and Meredith 2020).

Historical information, on aquatic macrophytes (prior to the early 2000s) for the Ōtūwharekai lakes was restricted to a number of species presence records (Wood and Mason 1977, Orchard 1979), and abundance data for three lakes (Clearwater, Camp and Heron) from a survey in the 1980's (Tanner et al. 1985). More recently LakeSPI (lake submerged plant indicators) surveys have been undertaken. The LakeSPI method was developed in 2002, as a management tool that uses submerged plants for assessing the ecological condition of lakes and for monitoring changes in lakes (Edwards and Clayton 2002). Key assumptions of the LakeSPI method are that native plant species and high plant diversity represents healthier lakes or better lake condition, while invasive plants are ranked for undesirability based on their displacement potential and degree of measured ecological impact (Clayton and Edwards 2006a,b, www.lakespi.niwa.co.nz). Fundamentally, submerged plants are integrators of changing conditions, so their presence or absence and depth distribution provides an indication of how, for example, water clarity may have reduced or improved over time, and reflects overall lake ecological condition.

LakeSPI has been used in the Ōtūwharekai lakes on three occasions (de Winton 2008, de Winton et al. 2013, de Winton and Burton 2017). While lake-specific details are provided in each lake section, in general the 2017 report noted that at that time most lakes were stable with little or no change (de Winton and Burton 2017).

More recent reports on lake condition –based on physico-chemical water quality (Bayer and Meredith 2020), kākahi (freshwater mussels, *Echyridella menziesii*) density and population structure (Burton et al. 2022), and observations of macrophytes (NIWA unpublished data 2021) – together indicate that the condition of some lakes is in decline, and not meeting ECAN's target conditions (CLWRP 2019) (Bayer and Meredith 2020). For example, while the kākahi survey in 2021 showed that the littoral distribution of aggregations, density, and population size structure had increased or remained similar for most of the lakes (since 2012), decreases were observed in two lakes and poor shell condition was noted in another lake (Burton et al. 2022). Furthermore, many of the lakes have high concentrations of total nitrogen and phytoplankton biomass compared with set limits and all but one of the lakes failed to reach their CLWRP TLI objectives between 2015 and 2019 (based on the 5-year average (Bayer and Meredith 2020)). Concern about the vulnerability of the lakes to further degradation and possible macrophyte collapse in some lakes has been raised (e.g., Kelly et. al 2014, refer to specific lake sections).

Description and current condition

The sections below provide specific information on current condition in each of the lakes. Additional detail is also presented in the form of lake ecological values and pressure tables used for lake prioritisation (Appendix E, and Appendix F) that is not necessarily duplicated in the narrative below. Sections of text in italics in the Water Quality summaries (of section 3.1) come directly from Bayer and Meredith 2020, unless otherwise indicated.

3.1.1 Clearwater, Te Puna a Taka

Lake Clearwater is 1.97 km^2 in area and has a maximum depth of 19 m within a deep central basin (Caruso et al. 2013), but much of the lake's area is shallow (ca. 1-4 m for ca 80% of lake), with a deep hole near the middle of the lake (i.e., 4-6 m for ca 10% and 6-19 m for 10%) (Figure 3-1, Figure 3-2).

Lake Clearwater was a permanent settlement and remains a significant site for Ngāi Tahu (Te Rūnanga o Arowhenua et al. 2010). However, evidence of long-term modification and pressure on this lake has been noted during its cultural health assessment, including high *E.coli* counts in the outlet, noticeable vegetation damage from browsing, and the prevalence of exotic vegetation (Te Rūnanga o Arowhenua et al. 2010).

The lake catchment is estimated at 46 to 56.5 km^2 (Wadsworth-Watts et al. 2013, Woodward et al. 2014). Wetlands in the Lake Clearwater catchment are largely pristine examples of native intermontane wetland systems, consisting of ephemeral turfs, streams, swamps and bogs. The wetland vegetation is predominately red tussock (*Chionochloa rubra*), purei (*Carex secta* and *C. diandra*), and bog rush (*Schoenus pauciflorus*) growing in a peat organic-rich soil (Wadsworth-Watts et al. 2013, Burge et al. 2020). Pre-2007 (prior to DOC ownership), the majority of the catchment was lightly grazed as leasehold farmland (Wadsworth-Watts et al. 2013). Since 2009, roughly 60% of the farmed land in the Lake Clearwater catchment has been ploughed and over-sown with rough pasture or brassica (Wadsworth-Watts et al. 2013). In 2013, the Lake Clearwater catchment was described as 90% unfarmed tussock grassland, while 9% of the catchment is farmed (Wadsworth-Watts et al. 2013). The remaining ~1% of the catchment consists of a small residential holiday home community (ca. 180 dwellings, Ashburton District Council 2022) located between Lake Camp and Lake Clearwater. Land to the north of Lake Clearwater is currently managed by the Department of Conservation (DOC) and is largely natural. Lake Clearwater receives water from Lake Camp via a small outflow (Bayer and Meredith 2020). There are three other streams entering the lake of which the largest is Craddock Stream (Wadsworth-Watts 2013).

The lake itself is a Wildlife Refuge, with no motorboats permitted, however it is popular for wind surfing (Bayer and Meredith 2020).

Figure 3-1: **Lake Clearwater.** Photo by D. Sutherland (2012).

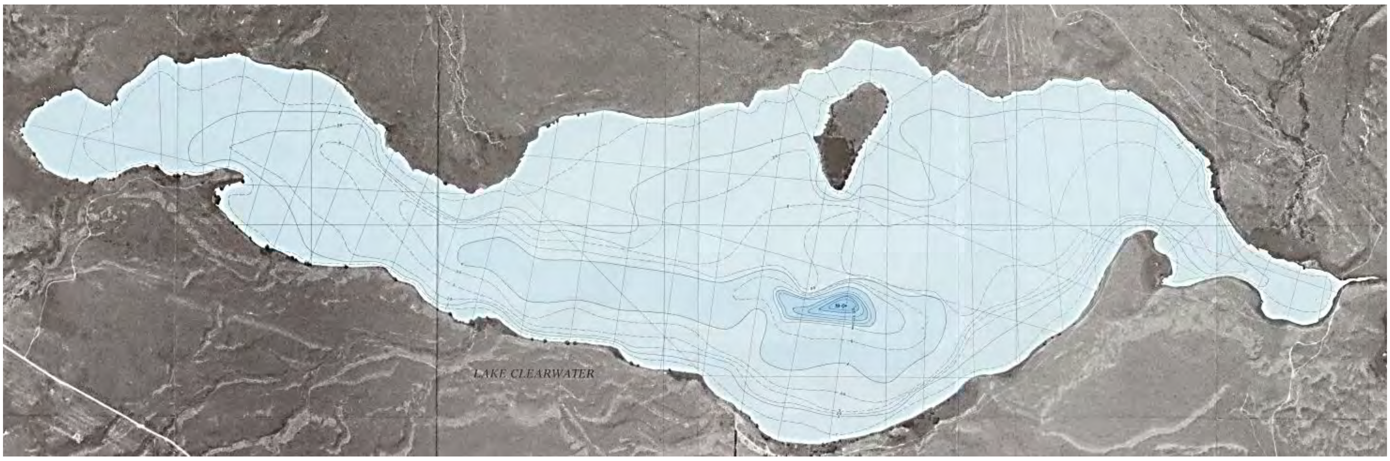

Figure 3-2: **Bathymetric map of Lake Clearwater.** Source: Irwin 1985, Scale 1: 5,000. Contour lines are from 1, 2, 2.5, 3, 3.5, 4, 5, 6, 8, 10, 12, 14, 16, 18 to 19 m in the centre.

Water quality summary (from Bayer and Meredith 2020)

Lake Clearwater is enriched with nutrients, having been either mesotrophic or eutrophic since 2006 (Table 3-1). It appears that the lake has undergone a period of sustained nutrient enrichment since 2005 and has subsequently failed to achieve its CLWRP objectives.

Lake water quality was potentially affected by:

- *Land intensification upstream.*
- *Possible septic tank leakage from Clearwater Huts village.*
- *Water quality of Lake Camp.*
- *Groundwater inputs to the lake.*

A hydrological and nutrient load balance for the Lake Clearwater catchment indicated that nitrogen export from the lake exceeded estimated inputs, suggesting an additional unaccounted source of nitrogen into Lake Clearwater. This source was identified as being possibly via shallow groundwater. The study also suggested that nutrients were elevated downstream of farmland, and that nitrate in farmland subsurface runoff contributed >50% of total nitrogen yield from farmland (Wadworth-Watts, 2013).

The relative contributions from Lake Camp are likely to be low because nutrient concentrations in Lake Camp are significantly lower than in Lake Clearwater. Modelling of potential nutrient *contributions by water birds were shown to be less than 1% of total TN* (total nitrogen) *load and less than 2% of total TP* (total phosphorus) *load (Kelly et al. 2021). Losses of nutrients from the Clearwater huts septic fields are unknown. Both Lakes Camp and Clearwater tended to have very high TN:TP ratios, indicating that phosphorus limitation of phytoplankton growth likely prevails in these lakes. There is very close tracking of chl-a and TP (but also of chl-a and TN) concentrations over the monitoring record. As such, management of P inputs to Lake Clearwater is of highest priority, but to meet CLWRP objectives TN inputs also need to be reduced (*Bayer and Meredith 2020).

Table 3-1: Trophic level index and attribute states for Lake Clearwater. Sourced from Bayer and Meredith (2020) and ECan data supplied to NIWA.

Year	Trophic Level Index			Numeric Attribute State (in µg/L)			
	TLI	Grade	CLWRP met	TN	TP	Chl-a - MED	Chl-a - MAX
2005	2.89	OLIGO	Yes	240	5	1.0	1.5
2006	3.38	MESO	No	320	5	3.5	6.6
2007	3.60	MESO	No	440	8.5	3.2	3.9
2008	4.16	EUTRO	No	620	14	3.9	9.4
2009	3.83	MESO	No	545	12	3.4	5.1
2010	4.27	EUTRO	No	670	19	4.6	7.5
2011	4.08	EUTRO	No	645	17.5	3.9	4.7
2012	3.89	MESO	No	580	14	2.3	3.9
2013	3.83	MESO	No	440	14	0.9	9.0
2014	3.10	MESO	No	380	8	0.9	1.8
2015	3.84	MESO	No	480	13	2.7	5.2
2016	3.90	MESO	No	510	13	3.0	6.0
2017	4.12	EUTRO	No	580	15	4.1	6.0
2018	3.62	MESO	No	500	11	2.1	3.9
2019	3.85	MESO	No	490	12	4.2	6.4
2015 to 2019	3.86	MESO	No	512	12.8	3.2	6.4
2017 to 2022	4.51	EUTRO	No	592	20	6.6	40

MED = seasonal median concentration (Dec-May) and MAX = seasonal maximum concentration (Dec-May); second to last line is 5-year average from 2015-2019 (Bayer and Meredith 2020). The last line is the average for TLI, TN and TP, the median for Chl-a MED, and maximum value for Chl-a Max from 2017-2022 calculated from ECan data supplied to NIWA.

Aquatic life

In 2017 Lake Clearwater was described as 'stable' and having 'moderate' condition according to a LakeSPI survey, with no changes detected from the earlier surveys in 2007 and 2012 (de Winton and Burton 2017). The submerged vegetation extended to an average of 7.7 m depth, with only the relatively deep basin unvegetated (de Winton and Burton 2017). Vegetation depths were slightly greater than those recorded in 2012 (5.8 m) and 2007 (6.5 m). In 2017 high cover 'meadows' of *Chara australis* grew to an average depth of 7.5 m, leaving a relatively small area in the central deep (19 m) sub-basin of the lake un-vegetated. Other charophyte species contributed to a diverse vegetation community on mid-depth slopes.

Native pondweeds (*Potamogeton ochreatus, P. cheesemanii*) and milfoils (*Myriophyllum triphyllum*) formed an open canopy above the charophyte meadows to 3.5 m depth. In the shallow zone (< 2 m depth), *Isoëtes alpina* formed swards to depths slightly greater than the turf plants (de Winton and Burton 2017).

Elodea canadensis (elodea) was the only invasive plant recorded from Lake Clearwater. It was found at water depths < 5m, and commonly formed bands of partially-or fully-closed canopy, but did not exceed 25% of the vegetated area (de Winton and Burton 2017) (Figure 3-3).

A kākahi survey in 2012 reported animals in shallow depths in the western arm of the lake, averaging 35.4 animals per m^2 (de Winton et al. 2013b). Although the deep-basin area in the lake was described as having vegetation-bare areas of silty sediment that appeared suitable for kākahi, they were absent (or <1 m^2) (de Winton et al. 2013). The apparent absence of kākahi from deeper water in Lake Clearwater was considered as an indication of periodic low oxygen levels (de Winton et al. 2013b). However, measurement taken at later dates (February 2017, February 2021, and September 2021) did not show DO levels ≤5 mg/L below 5.5 m depth (T. Bayer, pers. comm., ECan, 17/09/2021) (Burton et al. 2022).

In 2021, kākahi were again only detected from one site (Site 3) in the western arm of Lake Clearwater amongst very soft flocculent sediments. The average kākahi density was 85 per m^2. The absence of kākahi within the deeper lake basin (5.5 – 6 m) could not be reconfirmed in 2021 due to unfavourable conditions for diving (poor water clarity, with zero visibility). However, a grapnel sample taken subsequently retrieved a live kākahi from the deep basin (T. Bayer, pers. comm., ECan). Kākahi mean length remained similar at 59 mm in 2012 to 58 mm in 2021, and most individuals fell within the 55 – 60 size range (Burton et al. 2022). All kākahi collected in 2021 from Lake Clearwater were highly deformed, irregular, and rounded in shape, with obvious flaking and thickening evident on shells (Burton et al. 2022).

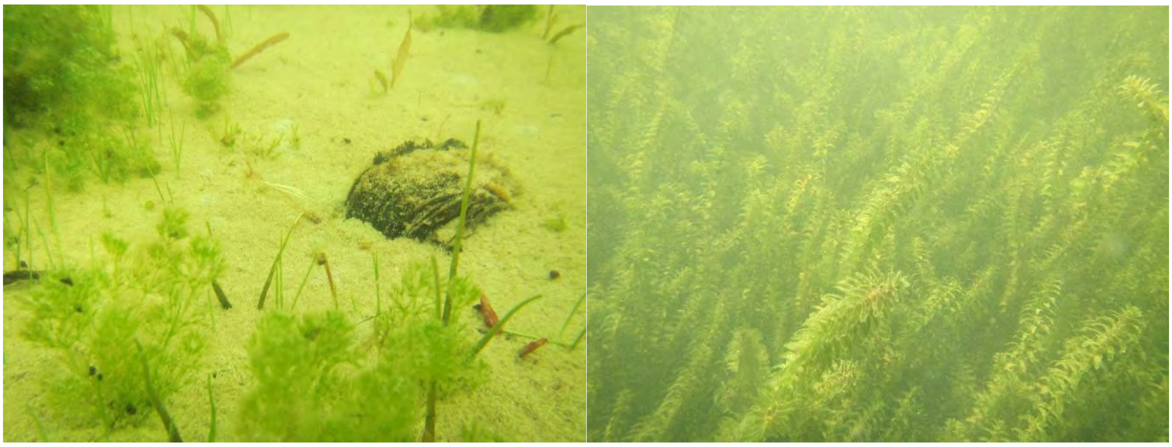

Figure 3-3: Macrophytes and kākahi in Lake Clearwater. Native plants (left) and elodea (right). Photos by T. Burton (2017).

3.1.2 Lake Camp, Ōtautari

Lake Camp is a small (ca 0.44 – 0.49 km^2) lake of intermediate depth (18.9 m maximum depth) (Figure 3-4, Figure 3-5). The lake is popular for swimming, boating and water skiing. Its swimming beach is monitored as part of the recreational lake quality monitoring programme.

The lake is a significant site for Ngāi Tahu. However, evidence of long-term modification and pressure on this lake was noted during cultural health assessments, including the prevalence of exotic vegetation and fish, including perch[1] (*Perca fluviatilis*) (Te Rūnanga o Arowhenua et al. 2010).

Lake Camp has a small outflow into Lake Clearwater (Bayer and Meredith 2020) and there is a suspected underground flow from Lake Camp into Lake Clearwater (referred to in Lake-Camp-and-Lake-Clearwater-Consultation-Document.pdf (ashburtondc.govt.nz)).

Figure 3-4: Lake Camp. Photo by T. Burton (2017).

[1] Perch are designated as a sport fish by Fish and Game new Zealand

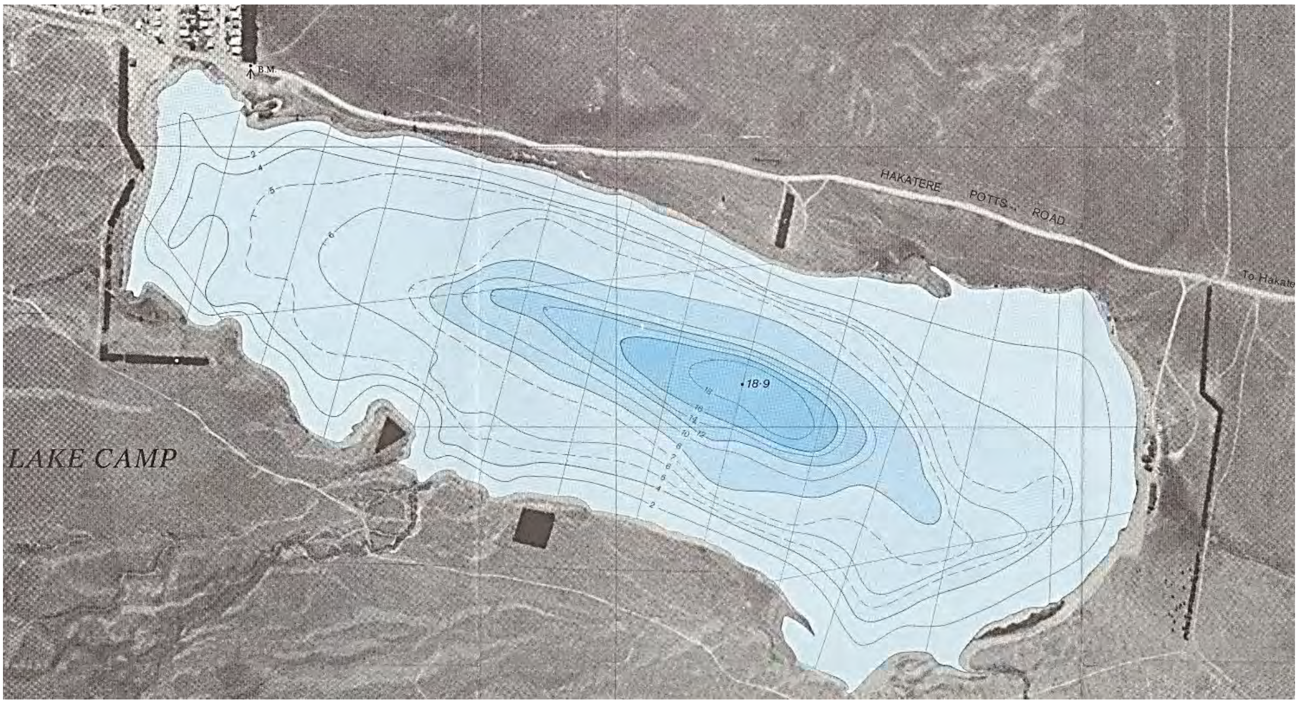

Figure 3-5: Bathymetric map of Lake Camp. Source: Irwin 1985, Scale 1: 5,000. Contour lines are from 2, 4, 5, 6, 7, 8, 10, 12, 14, 16, 18 to 18.9 m in the centre.

Water quality summary (from Bayer and Meredith 2020)

*Lake Camp was mostly mesotrophic due to elevated total nitrogen concentrations, failing to meet its CLWRP objectives (*Table 3-2*). Intermittent high TP concentrations also contributed to higher than normal TLI values in some years. TLI fluctuations may have been partly driven by stratification events, and an anoxia driven mortality event of kākahi populations occurred in* 2013 (see section below on Aquatic life)*. In years in which the lake stratified thermally, bottom waters may have gone anoxic, which may have resulted in a release of phosphorus from sediments. This release in turn could have fuelled the increases in TP and phytoplankton biomass observed in some years. To better understand and manage the lake ecosystem the continuous monitoring of dissolved oxygen is recommended in summer. Due to the small volume of the lake and its hypolimnion, Lake Camp may be more susceptible to increases in organic loads from its catchment or phytoplankton production which drive anoxia in the lake bottom waters. Therefore, maintaining low phytoplankton biomass, and minimising sediment loading from the catchment are priorities for the lake to protect it from further anoxia events.*

Table 3-2: Trophic level index and attribute state for Lake Camp. Sourced from Bayer and Meredith (2020) and ECan data supplied to NIWA.

Year	Trophic Level Index			Numeric Attribute State (in µg/L)			
	TLI	Grade	CLWRP met	TN	TP	Chl-a - MED	Chl-a - MAX
2005	2.57	OLIGO	Yes	210	4	0.9	1.3
2006	2.77	OLIGO	Yes	235	3.5	1.4	2.4
2007	3.18	MESO	No	290	7	2.1	3.9
2008	3.09	MESO	No	350	6	1.5	2.5
2009	3.41	MESO	No	345	12.5	1.7	2.8
2010	2.86	OLIGO	Yes	310	2.5	1.0	1.6
2011	3.19	MESO	No	320	9.5	1.5	2.3
2012	3.18	MESO	No	320	8	2.0	2.5
2013	3.38	MESO	No	320	10	1.0	1.2
2014	3.01	MESO	No	340	8	1.1	1.7
2015	3.23	MESO	No	380	9	1.4	2.1
2016	3.33	MESO	No	330	8	1.9	2.7
2017	3.22	MESO	No	430	7	2.1	2.3
2018	3.19	MESO	No	300	5	2.3	3.4
2019	3.22	MESO	No	330	2	4.2	4.4
2015 to 2019	3.24	MESO	No	354	6.2	2.4	4.4
2017 to 2022	3.5	MESO	No	334	8.2	3.4	8

MED = seasonal median concentration (Dec-May) and MAX = seasonal maximum concentration (Dec-May); second to last line is 5 year average from 2015-2019 (Bayer and Meredith 2020). The last line is the average for TLI, TN and TP, the median for Chl-a MED, and maximum value for Chl-a Max from 2017-2022 calculated from ECan data supplied to NIWA.

Aquatic life

In 2017, Lake Camp was categorised as being in high ecological condition and stable according to a LakeSPI Index of 65% (de Winton and Burton 2017). From 2007 to 2012 there was an increase in the Native Condition Index, followed by a small decline between 2012 and 2017. The high LakeSPI score in 2017 was driven by vegetation extending to an average depth of 11.5 m, charophyte meadows to an average of 11 m depth, the limited impact of exotic weeds (ranging from "occasionally present", to "present with an open plant canopy"), and a diverse native vegetation commonly comprising four community types. Submerged vegetation (Figure 3-6) has always covered most of the bed of Lake Camp. Turf plants dominated by *Lilaeopsis ruthiana* contributed to vegetation diversity in the shallow zone (≤3 m depth), although boulders also exclude macrophytes and kākahi from some shallow areas (<2m). Native pondweeds and milfoils were common to ≤5 m depth (Figure 3-6). Introduced weeds, mostly elodea and occasionally water buttercup (*Ranunculus trichophyllus*) have been minor components of the vegetation (≤25% vegetated area). Three native charophyte species commonly contributed to a zonation pattern of *Chara fibrosa, C. globularis* and *C. australis* with increasing depth (de Winton and Burton 2017) (Figure 3-6).

A mass die-off of kākahi in Lake Camp in summer 2013 was linked to strong thermal stratification and almost no measurable dissolved oxygen below 12 m depth (Beech 2013, Sutherland 2013). Re-survey of deeper kākahi populations in the lake in May 2013 indicated a 50% and 30% reduction in live animals at 16.2 m and 14.5 m respectively (Sutherland 2013, Burton et al. 2022). Sediments at 14.5 m to 15.5 m in Lake Camp were 'soft clay-like' (NIWA unpubl. diver obs.).

During both the 2012 and 2021 surveys, aggregations of kākahi were located at depths beyond the limit of submerged vegetation at the two sampled sites. There was a 53% increase in kākahi densities measured at the survey sites between the 2012 and 2021 surveys (eight-year period), with the average density of kākahi increasing from 195 per m^2 to 298.5 per m^2 (Burton et al. 2022). In 2021 a total of 995 individual kākahi were collected from sites 1 and 2 for density assessment, and 160 were processed for size data. Kākahi shell length ranged from 45 – 71 mm with most kākahi falling into the 55 – 60 mm size range (Burton et al. 2022).

A large school of perch (*Perca fluviatilis,* not native) were observed in Lake Camp in 2017 (de Winton and Burton 2017) (Figure 3-7).

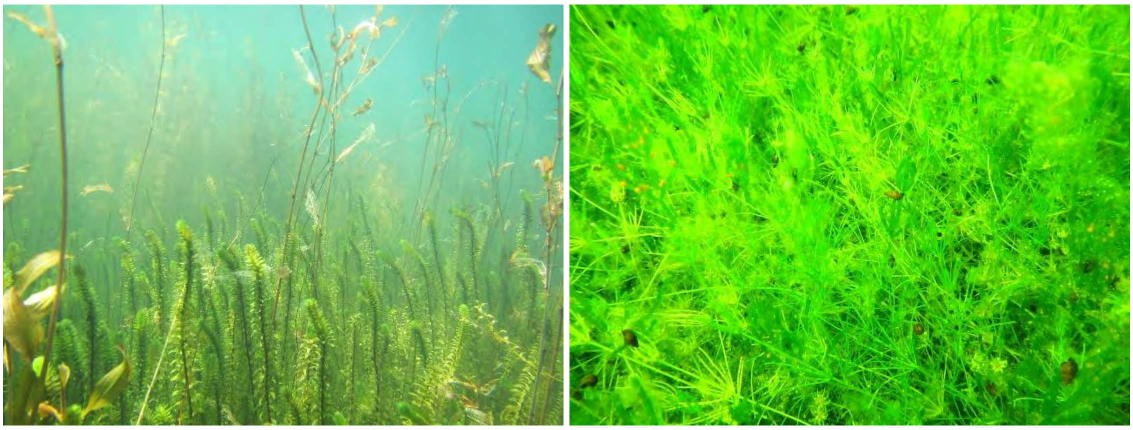

Figure 3-6: Macrophytes in Lake Camp. Native pondweed and milfoil, with dense elodea (non-native) (left image) and native charophytes (right image). Photo by T. Burton (2017).

Figure 3-7: Large school of perch in Lake Camp. Photo by T. Burton (2017).

3.1.3 Lake Emma, Kirihonuhonu

Lake Emma is a shallow (2.7 m) (de Winton and Burton 2017), medium sized (1.6 km^2) lake (Figure 3-8, Figure 3-9) with a catchment of ca 28.19 km^2 (Woodward et al. 2014). The lake is a significant site for Ngāi Tahu, having been a permanent settlement in the past (Te Rūnanga o Arowhenua et al. 2010). It has been described as having diverse and intact wetland fringes on the western side (Te Rūnanga o Arowhenua et al. 2010). Prior to inclusion of the surrounding area in the conservation estate, there were reports of poor management practices, including livestock accessing lake marginal areas (Bayer and Meredith 2020). Evidence of long-term modification (such as occurrence of herbaceous weeds) and pressure (human induced) on this lake was noted during a cultural health assessment (Te Rūnanga o Arowhenua et al. 2010).

The wetlands are now part of the Lake Emma Government Purpose Reserve (Bayer and Meredith 2020).

Figure 3-8: Lake Emma. Photo by T. Burton (2017).

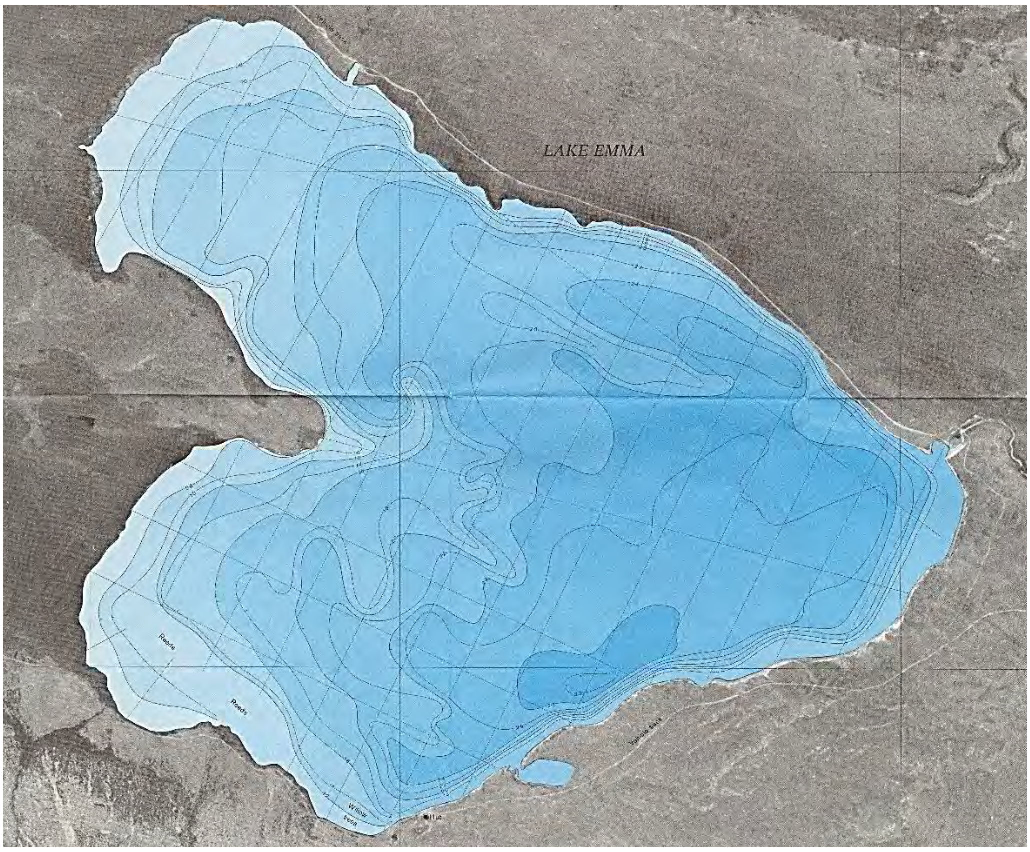

Figure 3-9: Bathymetric map of Lake Emma. Source: Irwin 1985, Scale 1: 5,000. Contour lines are from 1.2, at 0.2 m intervals, to 3 m at the deepest point in the lake.

Water quality summary (from Bayer and Meredith 2020)

Lake Emma has undergone alternate periods of mesotrophic conditions with extensive macrophytes, and highly enriched (eutrophic or more) conditions with little or no macrophytes. This pattern of 'flipping' between a clear macrophyte dominated state (e.g., 2012-2014) and a turbid, phytoplankton dominated state (e.g., 2008-2009, 2016-2017) is prevalent amongst some shallow lakes in New Zealand (Schallenberg & Sorell 2009).

The reporting of a macrophyte collapse in 2007 (referenced in Bayer and Meredith 2020 to Kelly et al., 2014, and Schallenberg and Sorell 2009) is however at odds with the November 2007 LakeSPI survey, which in 2008 reported a LakeSPI Index of 37%, Native Condition Index (NCI) of 45%, and an Invasive Impact Index (III) of 69%. There has been little change in these LakeSPI Index values across the three monitoring events of November 2007 (37%), November 2012 (32%), and February 2017 (35%) (see section below on Aquatic Life).

Further, Schallenberg and Sorrell (2009) describe that they "collected information on New Zealand lakes known to have shifted between … the presence of beds of submerged macrophytes and … a lack of (or distinct reduction in) macrophyte biomass. This information was collected from peer reviewed scientific literature, unpublished theses, technical reports, and from canvasing members of the New Zealand Freshwater Sciences Society, including freshwater researchers, managers, employees of the Department of Conservation and environmental consultants. Information requested included the name and location of the lake, dates of regime shifts, the name of the species of the common macrophytes that collapsed, and references to any relevant published or unpublished data on the lakes." Nowhere do Schallenberg and Sorrell (2009) describe the source of the evidence for a macrophyte collapse in Lake Emma.

*Water level in Lake Emma was possibly impacted by a diversion of some flow of the inflowing stream to Lake Camp in recent years (DOC Geraldine, pers. comm.). Impacts of low water level on lake health include warmer summer temperatures, the 'concentration' of nutrients in the lake (*Table 3-3*), effects on macrophytes due to low water level and elevated temperatures, and increased sediment resuspension.* Lake sediments have been observed to be soft and flocculent to soft and 'jelly like' in the deeper areas of the lake (> 2 m), and kākahi have sunk up to 8 cm deep into the sediment (NIWA unpubl. diver obs.).

Bayer and Meredith (2020) recommended the installation of a continuous water level recorder at Lake Emma and suggested that nutrient management or weed control measures could be required to stabilise the lake in a mesotrophic, clearwater state.

Table 3-3: Trophic level and attribute states for Lake Emma. Sourced from Bayer and Meredith (2020) and ECan data supplied to NIWA.

	Trophic Level Index			Numeric Attribute State (in µg/L)			
Year	**TLI**	**Grade**	**CLWRP met**	**TN**	**TP**	**Chl-a - MED**	**Chl-a - MAX**
2005	3.88	MESO	No	320	12	5.4	10.0
2006	3.93	MESO	No	360	12	5.2	6.6
2007	4.64	EUTRO	No	575	29	8.1	15.1
2008	6.20	HYPER	No	1400	110	36.0	52.0
2009	5.74	SUPER	No	1300	80	22.3	37.0
2010	4.31	EUTRO	No	720	25	4.2	5.3
2011	4.78	EUTRO	No	625	35.5	6.4	20.0
2012	3.45	MESO	No	460	18	0.8	1.3
2013	3.76	MESO	No	430	21	1.4	2.0
2014	3.75	MESO	No	370	20	1.1	4.3
2015	3.46	MESO	No	400	13	0.8	3.8
2016	4.53	EUTRO	No	620	29	4.4	16.0
2017	4.94	EUTRO	No	880	31	13.0	18.0
2018	4.28	EUTRO	No	530	19	7.0	10.0
2019	5.12	SUPER	No	730	40	21.0	29.0
2015 to 2019	4.46	EUTRO	No	632	26.4	9.2	29.0
2017 to 2022	4.84	EUTRO	No	634	30.1	13	48

MED = seasonal median concentration (Dec-May) and MAX = seasonal maximum concentration (Dec-May); second to last line is 5-year average from 2015-2019 (Bayer and Meredith 2020). The last line is the average for TLI, TN and TP, the median for Chl-a MED, and maximum value for Chl-a Max from 2017-2022 calculated from ECan data supplied to NIWA.

Aquatic life

In 2017, Lake Emma remained categorised as in moderate ecological condition according to a LakeSPI Index of 35%, which was not significantly different to results from 2007 and 2012. Submerged vegetation extended across most of the lakebed to a maximum depth of 2.1 m in 2017, but there were also bare patches in the central area of the lake. The "moderate" status reflects continuing dominance by elodea, which generally formed a closed canopy but was not tall growing (<1.5 m in height) (Figure 3-10). The invasive water buttercup *Ranunculus trichophyllus* was also common but restricted to shallow margins. Up to five native plant community groups were still present. Milfoil (*Myriophyllum triphyllum*) was locally abundant and surface-reaching in patches across the lake. In contrast, pondweed (*Potamogeton ochreatus*) and charophytes (three species) were observed at low covers only. Turf plants *Lilaeopsis ruthiana* and *Ranunculus limosella* were common at low covers at the lake margins (<1.5 m). Quillwort (*Isoëtes alpina*) also formed swards limited to the same shallow margins. Another native pondweed *Stuckenia pectinata* was only seen at one of the boat-launch sites (de Winton and Burton 2017).

Low kākahi densities (<1 per m^2) were recorded in Lake Emma in 2012 (de Winton et al. 2013b). It has been suggested (de Winton 2013) that low kākahi numbers could be related to significant blooms of cyanobacteria (*Dolichospermum*) recorded in the years prior to the 2012 survey (Sullivan et al. 2012), despite the presence of apparently suitable habitat. Increased mortality of juvenile kākahi is known to occur under toxin concentrations typical of a severe cyanobacteria (*Microcystis*) bloom (Clearwater et al. 2012, Burton et al. 2022).

In 2021, kākahi were present at low densities (≤ 10.2 per m^2) at all sites, and were generally in shallow water (0.3 – 1.5 m depth) at the interface between the mostly rocky lake margin and dense beds of elodea that extended over much of the lake bottom. Kākahi were not observed in depths of 2.3 to 2.4 m within this c. 2.7 m deep lake. Sediments were very soft and jelly-like at one site, and kākahi were only detected by feeling c. 8 cm below the sediment surface (Burton et al. 2022). Individual shell lengths ranged from 57 – 96 mm, with most falling into the 80 – 85 mm size class (Burton et al. 2022).

Lake Emma had very low eel abundance and diversity but was assigned an overall Takiwā rating of 'good' (Te Rūnanga o Arowhenua et al. 2010). The presence of a large, intact remnant wetland was noted (Te Rūnanga o Arowhenua et al. 2010, Bayer and Meredith 2020).

The non-native fish, perch have been reported from Lake Emma (Appendix F).

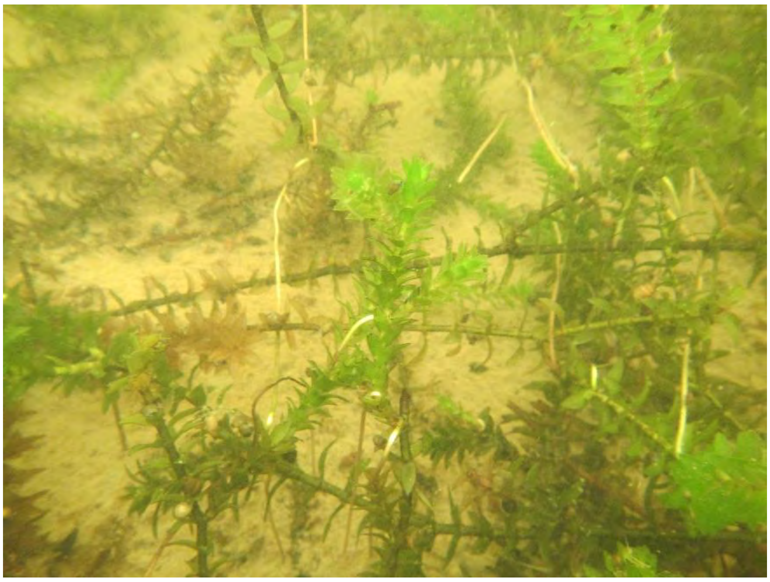

Figure 3-10: Short elodea plants in Lake Emma. Photo by T. Burton (2017).

3.1.4 Lake Denny

Lake Denny (Figure 3-11) is a small (0.05 km^2), shallow lake (2.1 m maximum depth), situated in a catchment of redominantly productive exotic grassland. The lake drains to the Rangitata River (Bayer and Meredith 2020).

Figure 3-11: Lake Denny. Photo by T. Burton (2017).

Water quality summary (*Bayer and Meredith 2020*)

Lake Denny is an extremely nutrient enriched lake in a farmed catchment. The lake frequently does not meet national bottom lines for water quality and failed to achieve CLWRP objectives in any of the monitoring years. The lake recently "flipped" and has undergone a total loss of macrophyte community.

Presumably, this statement on "flipping" stems from the 2017 LakeSPI assessment that describes "an apparent loss of vegetation at sites on the south-eastern side of the lake as well as a significantly lower Native Condition Index at the remaining north-western sites" (de Winton and Burton, 2017).

However, it should be noted that macrophytes were still present (but in very low abundance), as indicated by the LakeSPI method (<10% cover) (see below, Aquatic life).

This change in state implies that 'clear water years' such as 2014 may become unlikely to occur because of its shallow depth and susceptibility to sediment resuspension unless macrophytes can re-establish. Fortunately, macrophytes re-established by February 2021 (see below, Aquatic Life).

The lake has experienced frequent high turbidity events, often associated with high nutrients and chl-a. In March 2018, Environment Canterbury staff observed in-stream works in the inflow stream that sent large sediment plumes into the lake, resulting in a very brown lake with extremely high turbidity and TP in the lake.

A lack of fencing of the inflowing stream has been reported, along with instances of cattle observed in the stream, a loss of adjacent wetlands and erosion at the lake edge.

Improvements in nearby land management may facilitate improvements in water quality, and it is encouraging that the lake returned to a mesotrophic state in 2019. However, in-lake measures may be required to return the lake to a clear state.

Deterioration in Lake Denny is likely linked to significant land use intensification (de Winton and Burton 2017). For example, in 2017 the terraces above the lake had been planted with a forage crop, and images (Google Earth) also indicate an increase in cultivation and bare ground.

Table 3-4: Trophic level and attribute states for Lake Denny. Sourced from Bayer and Meredith (2020) and ECan data supplied to NIWA.

	Trophic Level Index			Numeric Attribute State (in µg/L)			
Year	**TLI**	**Grade**	**CLWRP met**	**TN**	**TP**	**Chl-a - MED**	**Chl-a - MAX**
2013	6.21	HYPER	No	1900	106	9.0	144.0
2014	3.80	MESO	No	400	30	1.4	2.7
2015	5.69	SUPER	No	730	86	16.0	45.0
2016	6.51	HYPER	No	2200	146	55.0	140.0
2017	5.26	SUPER	No	770	73	12.0	24.0
2018	6.22	HYPER	No	780	88	3.8	56.0
2019	3.88	MESO	No	270	26	3.9	6.2
2015 to 2019	5.51	SUPER	No	950	83.8	18.1	140.0
2017 to 2022	5.17	SUPER	No	582	93.5	5.7	56

MED = seasonal median concentration (Dec-May) and MAX = seasonal maximum concentration (Dec-May); second to last line is 5 year average from 2015-2019 (Bayer and Meredith 2020). The last line is the average for TLI, TN and TP, the median for Chl-a MED, and maximum value for Chl-a Max from 2017-2022 calculated from ECan data supplied to NIWA.

Aquatic life

Following a 2012 LakeSPI survey, Lake Denny was described as "stable" and in "moderate condition" (de Winton et al. 2013). The lake was dominated by elodea, but native milfoil formed surface-reaching, flowering patches amongst the elodea beds, while the exotic water buttercup *Ranunculus trichophyllus* was widespread at low covers. There were scattered plants of a charophyte (*Nitella tricellularis*), a turf plant (*Ranunculus limosella*), and raupo (*Typha orientalis*). By 2017, lake condition was regarded as "deteriorating" and "poor" (de Winton and Burton 2017). This deterioration was due to a loss of vegetation at sites on the south-eastern side of the lake as well as a significantly lower Native Condition Index score at the remaining north-western sites. In 2017, water clarity was also poor and vegetation was restricted to the lake edges to a maximum of c. 1 m depth in this 2 m deep lake. Only two submerged species, representing two plant community types, were recorded in 2017. The turf plant *Ranunculus limosella* was present at the margins of the north-western shore, while elodea dominated the vegetation (de Winton and Burton 2017). Divers' observations recorded during the 2021 kākahi survey indicated that macrophytes were more abundant than in 2021

In 2012, kākahi densities (derived from four sites) average approximately 70 per m^2. In 2021, densities exceeded approximately 97 per m^2 (noting the average density was potentially skewed by a large increase at one site). Most kākahi were noted as having 'thickened' shells (usually associated with deformity or infection (Phillips 2007)), with some erosion (Burton et al. 2022).

The non-native fish, perch have been reported from Lake Emma (Appendix F).

3.1.5 Lake Heron, Ō Tū Roto

Lake Heron is the largest and deepest lake in the Ashburton Lakes Basin, with a surface area of 6.95 km^2 and a maximum depth of 36.2 m (Figure 3-12, Figure 3-13). It receives water from small streams and shingle fan seepages. Underwater springs have been noted in the southwest arm of the lake by divers (NIWA unpublished records). The lake has three basins but only the largest basin to the South is sampled.

Catchment land cover is largely native vegetation or gravel/rock, plus fringing wetlands with 30% exotic grassland (9% highly producing and 21% low producing). Substantial areas near the lake were converted from low- to high producing grassland between 1996 and 2001, and between 2008 and 2012. Parts of the catchment are a Nature Reserve and Lake Heron has been given Wildlife Refuge status and Nature Reserve status, banning motorised crafts and protecting native species. The lake is popular for trout fishing (Bayer and Meredith 2020).

Lake Heron was a permanent settlement and is a significant site for Ngāi Tahu (Te Rūnanga o Arowhenua et al. 2010). A cultural health assessment provided evidence of long-term modification (such as the prevalence of exotic vegetation) and other pressures on this lake (Te Rūnanga o Arowhenua et al. 2010).

Figure 3-12: Lake Heron. Photo supplied by T. Burton (2017).

Figure 3-13: Bathymetric map of Lake Heron. Source, Irwin and Main (1984). Scale 1: 8,000. Contour lines in the southern basin are marked at 2 m intervals, with the maximum depth recorded at 36.2 m.

Water quality summary (*Bayer and Meredith 2020*)

Lake Heron was mostly oligotrophic between 2005 and 2016 but has become mesotrophic. *Turbidity and phytoplankton biomass have increased significantly between 2007 and 2019, with particularly large increases in chl-a* since 2017. *Annual average chl-a increased from less than 2.5 µg/L prior to 2015 to more than 7.5 µg/L in 2018.* The maximum chl-a value in 2019 was >14 times greater than in 2015, and similarly there was a large increase in the five-yearly average up to 2022 (Table 3-5).

Between 2007 and 2019 total nitrogen in the lake also increased. The recent deterioration in water quality, particularly the increase of phytoplankton biomass, is worrying and indicates that current protection provisions for Lake Heron may be insufficient and the catchment nutrient load may need to be reduced to meet plan outcomes and maintain the current ecological condition. Catchment nutrient limit setting should also consider the difference in impact of immediately bioavailable dissolved nutrients vs. less bioavailable particulate loads, as e.g., limiting total nitrogen without limiting its dissolved forms (nitrate and ammonia) may allow for a much larger phytoplankton response than could be expected based on the total load limit alone.

Table 3-5: Trophic level index and attribute states for Lake Heron. Sourced from Bayer and Meredith (2020) and ECan data supplied to NIWA.

Year	Trophic Level Index			Numeric Attribute State (in µg/L)			
	TLI	Grade	CLWRP met	TN	TP	Chl-a - MED	Chl-a - MAX
2005	2.22	OLIGO	Yes	93	3	0.7	1.0
2006	2.30	OLIGO	Yes	104	2.5	1.5	2.7
2007	2.66	OLIGO	Yes	120	4.5	1.1	4.5
2008	2.76	OLIGO	Yes	160	6	1.3	2.0
2009	3.03	MESO	No	140	9	2.0	2.2
2010	2.89	OLIGO	Yes	140	6	1.4	4.8
2011	3.11	MESO	No	140	8.5	1.7	2.5
2012	2.92	OLIGO	Yes	160	8	1.2	4.3
2013	2.78	OLIGO	Yes	161	7	0.8	2.6
2014	2.92	OLIGO	Yes	161	6	1.5	2.8
2015	2.64	OLIGO	Yes	147	6	1.3	2.7
2016	2.91	OLIGO	Yes	147	6	3.8	7.5
2017	3.19	MESO	No	144	6	3.9	7.7
2018	3.45	MESO	No	200	6	7.7	14.0
2019	3.54	MESO	No	140	4	5.2	38.0
2015 to 2019	3.15	MESO	No	156	5.6	4.4	38.0
2017 to 2022	3.83	MESO	No	186	10	9.3	60

MED = seasonal median concentration (Dec-May) and MAX = seasonal maximum concentration (Dec-May); second to last line is 5 year average from 2015-2019 (Bayer and Meredith 2020). The last line is the average for TLI, TN and TP, the median for Chl-a MED, and maximum value for Chl-a Max from 2017-2022 calculated from ECan data supplied to NIWA.

Aquatic life

At the time of the most recent LakeSPI survey (2017), Lake Heron was in moderate ecological condition and stable (de Winton and Burton 2017) – the LakeSPI indices did not vary significantly between any of the survey years (2007, 2012 or 2017), and vegetation composition remained similar. Up to five native submerged plant community types were also recorded, along with the non-native elodea. The vegetation was notable for its diverse assemblage of nine charophyte species, with native charophyte meadows (≥75% cover) found deeper than the main elodea bed. Native milfoil (*Myriophyllum triphyllum*) and pondweeds (*Potamogeton ochreatus, P. cheesemanii*) were common at generally low covers and <5 m depth. Turf plants were recorded at the shallow margins of all sites, and high cover swards of quillwort (*Isoëtes alpina*) occurred at most sites. Didymo (*Didymosphenia geminata*) was observed on shallow rocks at the south-western shoreline, but not on deeper vegetation. Greater vegetation depths were (as in past years) reported in the north-east arm (1.5 to 3.3 m deeper), where spring inflows have been observed. It was noted that this natural variability in vegetation depths is unusual in one waterbody and make detection of significant future change difficult (de Winton and Burton 2017).

Daphnia pulex (an introduced daphnia) was found for the first time in Lake Heron in 2011, having not been detected in samples from 1963 -1990 (Wood 2011). In 2011 it was the dominant species, having almost completely displaced *D. carinata* and *Bosmina* in Lake Heron (and in Lake Roundabout) (Wood 2011). Non-indigenous daphnia in the South Island was later considered to be *Daphnia pulicaria* based on genomic sequencing (Ye et al. 2021). *Daphnia pulicaria* was present in far greater numbers than *Daphnia carinata* in samples collected in November 2020 and February 2021 (Ludgate 2021). However, it should be noted that *D. pulex* is part of a species complex that is not well defined (e.g., Ye et al. 2021).

In Lake Heron the sediments were observed to be soft and clay-like in 2021, with kākahi on the sediment surface (NIWA unpubl. diver obs.). Kākahi populations were surveyed in Lake Heron in 2012 and 2021. The average density of kākahi in 2021 (51.9 per m^2) was very similar to the value of 52.4 per m^2 recorded in 2012. Kākahi aggregations in 2021 were re-recorded at the same sites as in 2012. Shell lengths were slightly higher on average for the 2021 survey, with a mean length of 56 mm recorded in 2012 and 57 mm in 2021. Most kākahi were in good condition, with varying degrees of erosion noted on the shells (Burton et al. 2022).

Outside of the zones of aggregation, kākahi were frequently observed at densities <1 per m^2 across the remainder of the vegetated dive transects, although they were absent where vegetation was densest (elodea or turfs of *Isoëtes alpina*), or in the shallows where the shore was covered by large boulders (Burton et al. 2022).

Longfin eels are present in Lake Heron (Te Rūnanga o Arowhenua et al. 2010).

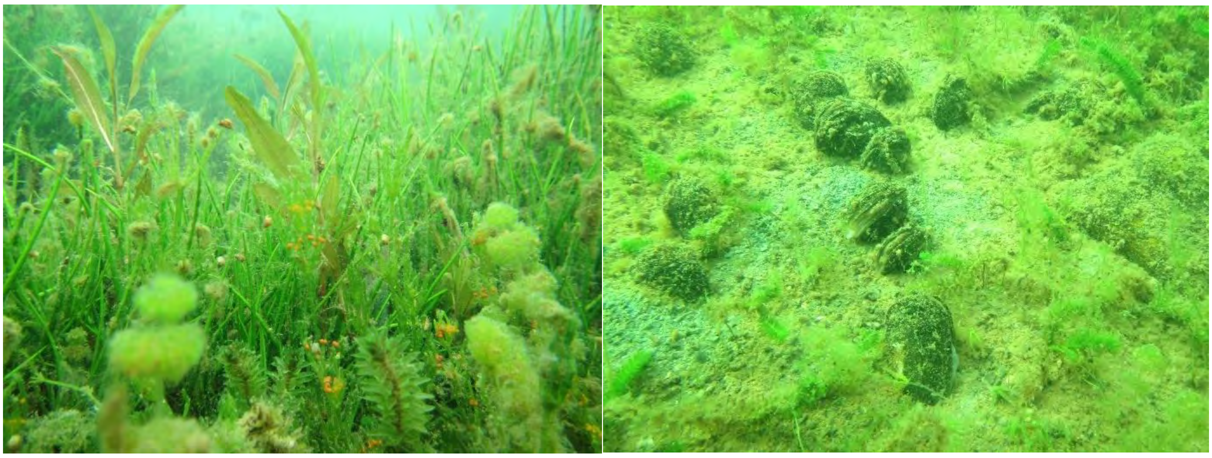

Figure 3-14: Diverse assemblage of macrophytes (left) and kākahi (right) in Lake Heron in 2017 Photo T Burton (2017).

3.1.6 Lake Emily

Lake Emily is small (0.19 km^2), shallow and 'bowl' shaped (scuba diving observation by de Winton, no bathymetric map available). The lake is recorded in Bayer and Meredith (2020) as having a maximum depth of 2.3 m and is polymictic. There is a large wetland margin, and the catchment is predominantly public conservation land. The 50 ha swamp to the west and northwest collects water from numerous streams and seeps. Drainage of the wetland occurs to the Māori Lakes via Jacobs Stream. The lake has high angler usage (Bayer and Meredith 2020).

Figure 3-15: Lake Emily. Photo by D Sutherland (2012).

Water quality summary (*Bayer and Meredith 2020*)

Lake Emily went through a period of eutrophic conditions between 2012 and 2016 but returned to a mesotrophic status in 2017 and 2018. However, average values from 2012 to 2022 indicate that the lake was eutrophic (Table 3-6).

High numbers of waterfowl reside in the lake and surrounding wetlands. Because of the high number of waterfowl and the relatively small catchment area, it was modelled that bird contributions to total nutrient loads could be relatively high, with 6% of total TN and 26% of total TP load being from bird sources (Kelly et al., 2014). The percent of total TP attributed to waterfowl was increased to 32.5% in 2021 based on updated calculations, taking into account bird population trends between 2010 and 2019 (Kelly et al. 2021).

The TLI objective for Lake Emily is mesotrophic recognising that the lake was likely naturally more productive and utilised extensively by water birds. However, Lake Emily exceeded its TLI target in a number of years being periodically eutrophic, mostly due to high TN (Bayer and Meredith 2020) (Table 3-6).

Table 3-6: Trophic level and attribute state for Lake Emily. Source Bayer and Meredith (2020) and ECan data supplied to NIWA.

Year	Trophic Level Index			Numeric Attribute State (in µg/L)			
	TLI	Grade	CLWRP met	TN	TP	Chl-a - MED	Chl-a - MAX
2008	3.77	MESO	Yes	450	15	2.5	2.8
2009	3.98	MESO	Yes	455	16.5	4.1	5.4
2012	4.15	EUTRO	No	410	17	2.4	10.8
2013	4.56	EUTRO	No	450	42	5.9	20.0
2014	4.17	EUTRO	No	330	20	6.0	17.0
2015	4.19	EUTRO	No	410	20	2.0	21.0
2016	4.20	EUTRO	No	440	20	2.0	19.0
2017	3.94	MESO	Yes	480	16	1.4	7.0
2018	3.99	MESO	Yes	390	23	3.4	6.0
2019	4.83	EUTRO	No	430	26	5.5	50.0
2015 to 2019	4.23	EUTRO	No	430	21	2.9	50.0
2017 to 2022	4.40	EUTRO	No	444	26	4.3	50.0

MED = seasonal median concentration (Dec-May) and MAX = seasonal maximum concentration (Dec-May); second to last line is 5 year average from 2015-2019 (Bayer and Meredith 2020). The last line is the average for TLI, TN and TP, the median for Chl-a MED, and maximum value for Chl-a Max from 2017-2022 calculated from ECan data supplied to NIWA.

Aquatic biota

Most of the lake basin is occupied by elodea to the maximum depth of 2.3 m. In places, native pondweed contributes to the overall cover of macrophytes, in particular at the edge of the main elodea bed. Turf plants comprised five species and included the aquatic fern *Pilularia novae-hollandiae* that was commonly found at the lake margin (≤1.0 m). *Isoetes alpina* also contributed to the turf community locally. LakeSPI indices for Lake Emily between 2007 and 2012 were relatively unchanged. The only major change in vegetation composition recorded in 2017 was the absence of charophytes, which had been recorded in the shallows at low covers previously (de Winton and Burton 2017). In 2017 Lake Emily was categorised as stable and in moderate ecological condition according to a LakeSPI Index of 28% (de Winton and Burton 2017).

Kākahi in Lake Emily were surveyed in 2012 at four sites, and resurveyed at two of the four sites in 2021 (de Winton et al. 2013b, Burton et al. 2022). Kākahi were present at both sites in gravel/stone substrates in the shallow margins (≤ 0.5 m depth). Between the two years, the average density of kākahi decreased c. 60% from 78 per m^2 in 2012 to 31.8 per m^2 in 2021. Kākahi lengths were very similar between survey periods, with a mean length of 56 to 57 mm in both years. Most kākahi were described as having thickened shells with some level of mild deformity and/or erosion. No kākahi were observed amongst the dense elodea (Burton et al. 2022).

3.1.7 Māori Lake East

Māori Lake East is also referred to as Front Lake or Lake A in other documents. Māori Lake East is a small (0.09 km^2), shallow lake with maximum depth of approximately 1.3 m (de Winton and Burton 2017). It has a residence time of about 4 days. The lake sits within a lake-wetland complex (ca. 100 ha swamp and 30 ha open water), with wetland margins (Bayer and Meredith 2020). The catchment is large (estimated area of 80.28km^2 (Woodward et al. 2014)), dominated by high producing exotic grassland (45% or more of the catchment area).

Both Māori Lakes – Ōtūwharekai – are significant sites for Ngāi Tahu (Te Rūnanga o Arowhenua et al. 2010). A cultural health assessment provided evidence of long-term modification and pressure on this lake. For example, extensive lake bed siltation, high *E. coli* counts in the outlet, and noticeable vegetation damage from browsing was noted (Te Rūnanga o Arowhenua et al. 2010).

Waterfowl and fish (including trout and eels) are abundant in Māori Lake East. The Māori Lakes have been given Wildlife Refuge status (this prohibits shooting of indigenous species and the use of motorboats) as well as Nature Reserve status (this protects the lakebed and a narrow marginal area but does not control land use in the catchments).

Figure 3-16: Māori Lake East. Photo supplied by M. de Winton (2017).

Figure 3-17: Bathymetric map of Māori lake east. Source: Irwin 1985, Scale 1:5,000. Contour lines indicate depths of 0.6, 0.8, 0.9, 1.0 and 1.2 m.

Water quality summary (*Bayer and Meredith 2020*)

At times the lake has been dominated by macroalgae attached to the lake sediment. When macroalgae are abundant and dominate primary production, current assessment methods may not accurately reflect primary production in the lake as macroalgae are not captured by either LakeSPI surveys or open water-column sampling of planktonic chl-a. The dominance of macroalgae in Māori Lake East is potentially linked to the very short hydraulic residence time of ca. 4 days (David Kelly, pers. Comm.). However, stream diversions may have altered sediment and water influx and residence times. In addition to the macroalgae which cover the lakebed, the lake also seemed to be experiencing blooms of planktonic algae. These phytoplankton blooms may be linked to reduced flows and prolonged residence times in dry periods. Peaks of turbidity and total phosphorus concentrations coincided with high chl-a, suggesting phytoplankton blooms or possibly high sediment re-suspension during low flows as the cause of high turbidity. The median TN concentration (2015-2019) in Māori Lake East were the highest among the high-country lakes sampled, suggesting a large external source of nitrogen to the lake. The wetlands around the lake are likely to be more effective in retaining phosphorus than nitrogen inputs from surrounding pastoral farming (Bayer and Meredith 2020).

Over the last five years (on average), the lake has been eutrophic (Table 3-7). However, Māori Lake East has low chlorophyll-a levels for a eutrophic lake – the high TLI score is largely due to a high TN to TP ratio.

Table 3-7: Trophic level index and attribute states for Māori Lake East. Source Bayer and Meredith (2020) and ECan data supplied to NIWA.

Year	Trophic Level Index			Numeric Attribute State (in µg/L)			
	TLI	Grade	CLWRP met	TN	TP	Chl-a - MED	Chl-a - MAX
2008	4.47	EUTRO	No	480	33	6.7	12.3
2009	2.94	OLIGO	Yes	295	7.5	0.9	1.4
2012	4.28	EUTRO	No	370	10	1.7	50.0
2013	3.24	MESO	Yes	530	11	0.2	1.7
2014	5.45	SUPER	No	740	9	1.0	61.0
2015	2.69	OLIGO	Yes	410	8	0.3	0.5
2016	3.02	MESO	Yes	500	4	1.0	2.2
2017	4.74	EUTRO	No	510	11	2.9	34.0
2018	4.13	EUTRO	No	770	12	0.5	21.0
2019	4.70	EUTRO	No	630	6	1.8	137.0
2015 to 2019	3.86	MESO	Yes	564	8.2	1.3	137.0
2017 to 2022	4.39	EUTRO	No	665	14.4	2.1	137

MED = seasonal median concentration (Dec-May) and MAX = seasonal maximum concentration (Dec-May); second to last line is 5 year average from 2015-2019 (Bayer and Meredith 2020). The last line is the average for TLI, TN and TP, the median for Chl-a MED, and maximum value for Chl-a Max from 2017-2022 calculated from ECan data supplied to NIWA.

Aquatic biota

In 2017, Māori Lake East was categorised in a moderate ecological condition with a LakeSPI Index of 47%. This result was significantly higher than the 2012 or 2007 LakeSPI Index, when extremely sparse submerged vegetation fell below the 10% cover threshold, generating a default LakeSPI Index of 0%. During the recent 2017 survey, a limited community of turf plants comprising three species was present in the shallow margin at the vehicle access site. Taller vascular plants recorded across the shallow bed of the lake (to 0.7 m depth), included the invasive elodea, and native plants, *Ruppia polycarpa* and *Potamogeton ochreatus*. As noted in 2012, the low stature of these plants (≤0.2 m) might suggest browsing and uprooting by waterfowl or wave action in this shallow lake, which has loose flocculent sediments (de Winton et al. 2013). Didymo was previously reported in Māori East Lake, but was not observed during the 2017 or 2012 surveys (de Winton and Burton 2017).

Kākahi were surveyed in 2012 and 2021 in Māori Lake East (de Winton et al. 2013b, Burton et al. 2022). Two sites were surveyed, but kākahi were only found at a single site (both years) in shallow water (0.6 m depth) and gravel substrate. No kākahi were observed across the mostly un-vegetated lake basin (c. 1.2 m depth), or where the shoreline was dominated by raupō (*Typha orientalis).* Average kākahi densities increased approximately 50%, from 13.2 per m^2 in 2012 to 19.8 per m^2 in 2021 (Burton et al. 2022). Kākahi sizes decreased slightly (mean decrease in length of 4 mm) between the surveys with a mean length of 86 mm in 2012 and 82 mm in 2021. Kākahi were observed to be in good condition (no shell deformities) (Burton et al. 2022).

Long- and short fin eels have been recorded from the Māori Lakes (Te Rūnanga o Arowhenua et al. 2010). Despite this being the most abundant tuna (eel) fishery of the Ashburton Lakes, the cultural health assessment generated a “moderate” score (Te Rūnanga o Arowhenua 2010). The assessment noted extensive siltation of the lakebed, high *E. coli* results and the presence of agricultural anti-biotic resistant *E. coli* at the outlet.

Willow control, and measures to control siltation and bacterial contamination (e.g., by developing better buffers along incoming water ways), were recommended to improve the health of the lake. Also recommended were the “complete and ongoing removal of exotic fish from the Māori Lakes and work towards making the lake complex a native fish only area” (Te Rūnanga o Arowhenua et al. 2010) (Bayer and Meredith 2020). The only non-native fish identified in Māori Lake East for this present project (Part 2) were brown trout (Appendix F) – no records were found for other non-native species.

3.1.8 Māori West Lake

Māori Lake West is also known as Back Lake and Lake B. Māori Lake West is a small (10 ha), shallow lake (maximum depth of 1.8 m (LAWA)) within a lake-wetland complex (ca. 100 ha swamp and 30 ha open water) (Figure 3-18). Previous estimates of maximum lake depth have ranged from 2.2 to 2.6 m (de Winton and Burton 2017). The catchment is dominated by highly producing exotic grassland (45% or more of the catchment area) (Bayer and Meredith 2020).

The Māori Lakes have been given Wildlife Refuge status (this prohibits shooting of indigenous species and the use of motorboats) as well as Nature Reserve status (this protects the lakebed and a narrow marginal area, but does not control land use in the catchments) (Cromarty and Scott 1995, Bayer and Meredith 2020).

The lakes are significant for Ngāi Tahu (Te Rūnanga o Arowhenua et al. 2010).

Figure 3-18: Māori Lake West. Photo supplied by T. Burton (2017).

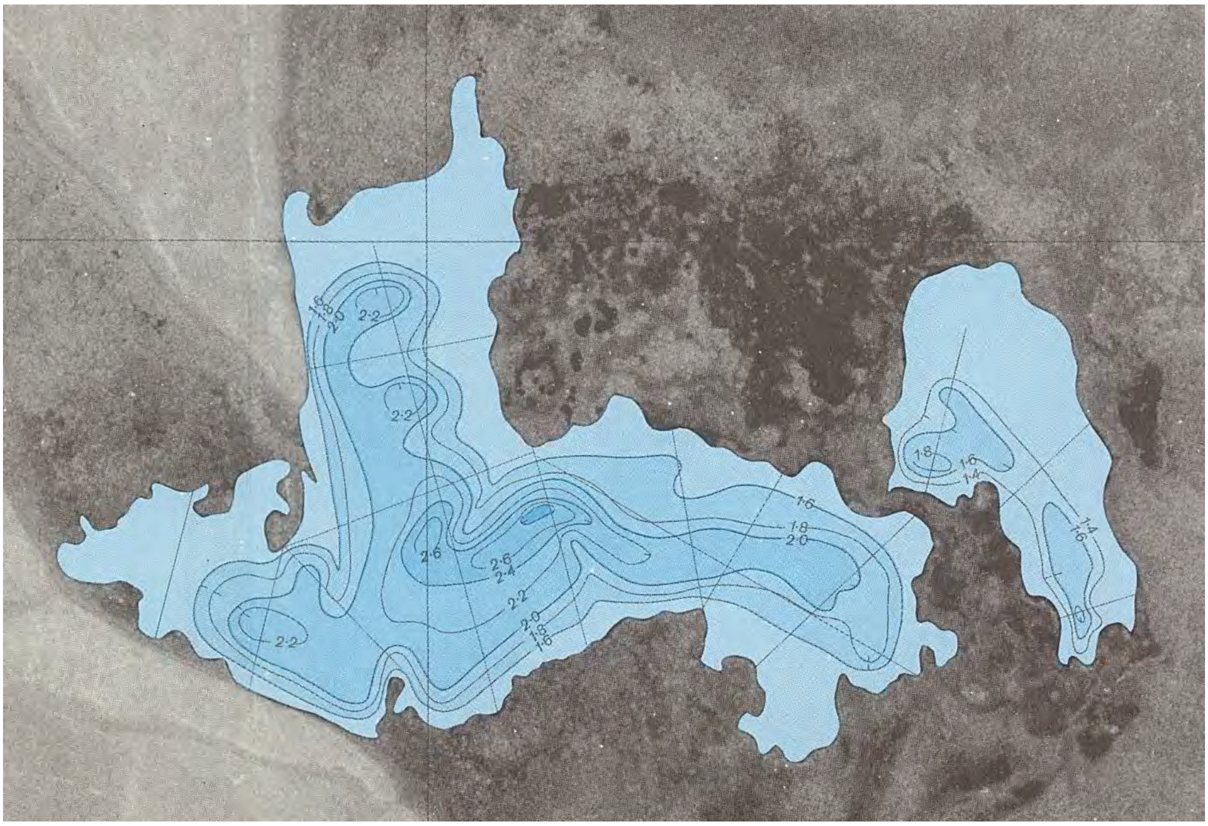

Figure 3-19: Bathymetric map of Māori Lake West. Source: Irwin 1985, Scale 1:5,000. Contour lines indicate depths of 1.6, 1.8, 2.0, 2.2, 2.4, 2.6 m.

Water quality summary (*Bayer and Meredith 2020*)

Between 2008 and 2016 the lake declined from being mesotrophic to eutrophic, but returned to a mesotrophic state in 2018 and 2019 (Table 3-8). *In 2017, concentrations of phytoplankton biomass and nutrients were exceptionally high. Nutrient sources include land development for intensive pastoral farming and the large number of waterfowl observed on the lake. However, the very large catchment for the lake means that waterfowl contributions are relatively small compared with predicted catchment sources (Kelly et al., 2014). If the high concentrations of phytoplankton and* *nutrients observed in 2017 were to reoccur, Māori Lake West could become at risk of 'flipping' into a turbid, phytoplankton dominated state and losing its macrophyte community.*

Although lake cultural health was assessed as 'good' by Te Rūnanga o Arowhenua, the lake was reported to have very low eel abundance and diversity (Te Rūnanga o Arowhenua et al. 2010).

Table 3-8: Trophic level index and attribute states for Māori Lake West. Source Bayer and Meredith (2020) and ECan data supplied to NIWA.

	Trophic Level Index			Numeric Attribute State (in µg/L)			
Year	**TLI**	**Grade**	**CLWRP met**	**TN**	**TP**	**Chl-a - MED**	**Chl-a - MAX**
2008	3.76	MESO	Yes	460	15	2.9	3.2
2009	3.78	MESO	Yes	420	13	2.7	3.3
2012	3.56	MESO	Yes	350	15	0.8	3.1
2013	3.96	MESO	Yes	330	17	1.7	7.3
2014	4.07	EUTRO	No	330	14	2.4	19.8
2015	3.87	MESO	Yes	400	13	2.1	6.0
2016	4.02	EUTRO	No	400	19	4.3	12.0
2017	5.20	SUPER	No	760	22	9.0	80.0
2018	3.94	MESO	Yes	410	13	2.1	9.0
2019	3.77	MESO	Yes	360	10	3.1	10.0
2015 to 2019	4.16	EUTRO	No	466	15.4	4.1	80.0
2017 to 2022	4.62	EUTRO	No	532	12.0	4.25	80.0

MED = seasonal median concentration (Dec-May) and MAX = seasonal maximum concentration (Dec-May); second to last line is 5 year average from 2015-2019 (Bayer and Meredith 2020). The last line is the average for TLI, TN and TP, the median for Chl-a MED, and maximum value for Chl-a Max from 2017-2022 calculated from ECan data supplied to NIWA.

Aquatic biota

A LakeSPI score of 28% for Māori Lake West indicated a moderate ecological condition in 2012 (de Winton et al. 2013). The aquatic weed elodea dominated the vegetation, but patches of native charophytes and scattered milfoil and pondweed plants occurred. The LakeSPI score had dropped by 10% between 2007 and 2012 due to reduced diversity and occupation of the lakebed by native plants, however variability between the sites meant this change was not statistically significant.

In 2017, Māori Lake West remained categorised in a moderate ecological condition with a LakeSPI Index of 37%, which was similar to the result in 2007. In 2017, submerged vegetation was recorded across the entire bottom of the lake to a maximum depth of 1.9 m. The aquatic weed elodea dominated, but as in 2012, patches of native charophytes and pondweeds were common (de Winton and Burton 2017).

A report by Clucas (2010) described high densities of kākahi at Māori Lakes Outlet, where a single quadrat (0.33 m^2) returned 52 mussels (ca. 156 mussels per m^2), while the mean density was 25 mussels per m^2. However, kākahi were only dense in a short section (167 m long) of the 800 m that was surveyed (Clucas 2010). In the lake itself, kākahi surveys indicated a 17% decline in kākahi densities at the one site they were found, with the average density of kākahi decreasing from 78.6 per m^2 (2012) to 65.4 per m^2 (in 2021). Kākahi were concentrated in a narrow band along the shoreline of this site, but elsewhere were absent from the shallow areas, where raupō dominated the silty shoreline or formed a floating raft over shallow water to 1.2 m depth. Dense submerged vegetation extended over the deeper basin to 2.1 m depth without any kākahi being observed in 2021 (Burton et al. 2022).

Waterfowl and fish (including brown trout and eels) occur in Māori Lake West (Bayer and Meredith 2020).

3.2 Potential for restored lake conditions

To identify target restored lake condition, this section is separated into two parts. The first describes desirable conditions for lakes of the type in the Ashburton basin, with a focus on the biophysical characteristics of a theoretical reference lake. The second part reviews the existing goals (aims, targets, objectives and outcomes) for the Ōtūwharekai lakes that have already been documented or published (e.g., in management plans) and provides a summary of existing goals relative to reference lake condition.

3.2.1 Theoretical reference lake

In general terms lakes, and particularly shallow lakes (i.e., most of the Ōtūwharekai lakes) are considered to be in good ecological condition when the water is clear and the dominant primary producers are submerged macrophytes. Macrophytes generally occur along the littoral zone of lakes, and potentially across the entire lakebed, where water depths are shallow and light penetration is sufficient to allow plants to grow. By anchoring themselves in the sediment, macrophytes bind the sediment and reduce the potential for wind-driven sediment resuspension, and through photosynthesis they oxygenate the water column and the sediment layers at the root zone. These oxygenated waters and sediment have life-supporting capacity for a range of fauna. Habitat is provided for aquatic insects and small fish, including refugia amongst plant stems from larger organisms seeking prey. When a diverse native macrophyte community is present, the mixture of architecture (stems, leaf shapes and sizes) amongst species, benefits a greater variety of aquatic organisms (Sloey et al. 1996, Celewicz-Gołdyn and Kuczyńska-Kippen 2017).

In contrast, degraded lakes become algal-dominated, turbid and lose their macrophytes. In a simplified summary, nutrient enrichment leads to increased algal production, which along with unnaturally high sediment loads, results in loss of water clarity and eventual loss of macrophytes. Cycles of algal productivity and senescence, fuelled primarily by internal P loads (in stratified or polymictic systems) (Cooke et al. 2005, Moss et al. 1996a, Hilt et al. 2013), results in increasingly flocculent substrates, where native macrophyte seed banks become buried (reducing germination opportunities (de Winton and Clayton 1996, de Winton et al. 2000)). These substrates become increasingly poor habitat (e.g., low density sediment, Barko et al. 1986) to support macrophyte re-colonisation (e.g., poor anchorage, anaerobic root zone). In the absence of established macrophytes, substrates are more readily resuspended by wind and wave action (Ozkundacki and Allan 2019), providing a feedback loop of poor water clarity that limits opportunity for native seedling recruitment. These periodic switches in stable state cause loss of macrophytes, and may be exacerbated by weed removal such as deliberate weed control, overgrazing by birds, or pest fish species, changes in water level or salinity (Moss et al. 1996a).

Alien invasive species (AIS) act in concert with above stressors – they can destabilise ecosystem processes and lead to degradation (or a lake regime shift) on their own (see section 6.5). Invasive fish and plants are drivers of native biodiversity loss (de Winton et al. 2001, Closs et al. 2004), through mechanisms which include trophic cascades. These occur when invasive fish species exacerbate eutrophication through top-down processes, as each species contains a planktivorous life stage that has the potential to overgraze zooplankton, allowing unbalanced phytoplankton growth. For example, perch and gambusia have been associated with reductions in water quality through 'top-down' predatory effects on zooplankton (Rowe and Smith 2001, Meiro et al. 2001, Nagdali and Gupta 2002, Rowe 2007, Smith and Lester 2007) (see section 6.5). For perch, impacts on zooplankton are most pronounced when the population structure is dominated by juvenile fish (<120 mm; Smith and Lester 2007). Adult life stages of invasive fish may either graze on macrophytes (e.g., rudd, *Scardinius erythrophthalmus*) or disturb sediments (eg., koi, *Cyprinus rubrofuscus*), exacerbating poor water clarity. Invasive plants have also been responsible for lake regime shifts, following root die-off in anoxic sediments with subsequent lake-scale vegetation death (Champion et al. 1993).

Within a range of nutrient conditions, the clear water state and the degraded algal dominated state are considered to be alternative stable states, i.e., both states are possible, and both are resistant to change (Scheffer 1989, Scheffer et al. 1993). The simplified foodwebs associated with the degraded state are resistant to change, as are the more complex food webs associated with the clear water macrophyte dominated state (Janse 2005). The resilience of stable states has significant implications for restoration actions and trajectories (Moss et al. 1996b, Sondergaard et al. 2007, Scheffer et al. 1993). On a global scale, few successful lake restoration examples exist for systems where macrophytes have been lost, because the challenges to restoration beyond that tipping point are significant. Hence there is a very compelling reason to protect native macrophytes while they are still present in the Ōtūwharekai lakes.

3.2.2 Restoration goals for lake condition

To identify target restored lake condition, the existing aims, targets, and desired outcomes for the Ōtūwharekai lakes were reviewed. The terms "aims, targets, and outcomes" are deliberately broad. These include lake conditions recommended as priorities for the future (e.g., Te Rūnanga o Arowhenua et al. 2010), those identified as narrative objectives or outcomes in the CLWRP (2019), or associated with numeric targets or limits (CLWRP 2019). Where possible the terminology in source documents is used – collectively these are referred to as the goals for lake condition.

The Canterbury Land and Water Regional Plan (CLWRP 2019) describes objectives (outcomes) for the region, and policies to be implemented to achieve those objectives. Lakes are listed as waterbodies of High Naturalness (Appendix C) when characteristics such as:

- outstanding natural features and landscapes,
- regionally significant wetland complex,
- habitat of threatened/endangered indigenous birds and freshwater species including eels and mussels, and
- high visual amenity value, exist.

Indicators of lake condition have been established (e.g., LakeSPI, TLI) against which the measured lake state can be compared (Table 3-9, Table 3-10). For each lake there are specific targets, which vary according to lake type such as large or small to medium sized high-country lakes. In addition, for Natural State waterbodies within land administered by DOC, the desired freshwater outcome is that the "lakes are maintained in a natural state" (Table 3-9). Importantly, all of the Ōtūwharekai lakes with the exception of Lake Camp, currently fail to meet their TLI targets, (Table 3-10, Table 3-11).

For each lake, Bayer et al. (2020) describe in detail which component metrics have contributed to TLI exceeding set targets since routine seasonal monitoring began. In addition, modelling studies have been undertaken for DOC and ECan to better understand the relationships between nutrient loading, nutrient status of lakes, and associated ecological values of the waterbodies (Kelly et al. 2014, 2021). The Environment Canterbury Land and Water Plan (2019) objective for all lakes in the nutrient sensitive zones is for TLI to not exceed 3 on average, with the exception of the Māori Lakes and Lake Emily where the plan objective is that TLI remains below 4 on average (Table 3-9). Therefore, TN and TP loads and necessary concentration reductions were calculated; these TN and TP concentrations correspond to the maxima for a TLI of 3 (160 mg TN/m^3 and 9 mg TP/m^3) and for a TLI of 4 (340 mg TN/m^3 and 20 mg TP/m^3), respectively (Kelly et al. 2021). Substantial (>66%) TN load reductions are required for all lakes, with the exception of Lake Heron (Table 3-12). Inputs of TP to Lakes Emma, Denny, Clearwater and Emily also need to be reduced considerably (Kelly et al. 2021, Bayer et al. 2021) (Table 3-12).

Management of nutrient loads to maintain aquatic macrophyte communities was described as the key priority for the Ōtūwharekai lakes by Kelly et al. (2014); nutrient load reduction was likely to achieve both macrophyte conservation and water quality management objectives.

A cultural health assessment for the Ōtūwharekai lakes (Te Rūnanga o Arowhenua et al. 2010), identified that “long-term modification of the area, particularly in relation to the historical loss of native flora and fauna and subsequent grazing and stock pressure was the biggest issue facing the Ōtūwharekai / Ashburton Lakes area”. Their recommendations for future management (see **Error! Reference source not found.**) identify priorities likely to be achieved if the TLI and LakeSPI targets for the lakes were met. Recommendations include: enhancing food gathering opportunities, protecting waterways, eradication of pest species, and prioritisation of the Māori lakes, Lake Heron, Clearwater and Emma for restoration.

Common themes identified from stakeholder targets, priorities, objectives, and aims, are described below as overarching goals for the lakes:

- maintain waterbodies and their margins in a healthy state, and improve those that are degraded,
- protect indigenous biodiversity values and natural character, and the life-supporting capacity of freshwaters,
- provide for contact recreation, and
- manage the land and water as integrated natural resources to recognise and enable Ngāi Tahu culture, traditions, customary uses and relationships with land and water.

Achieving the in-lake component of these goals will be largely dependent on:

- meeting the existing TLI and LakeSPI targets (CLWRP 2019), as well as
- developing and implementing an effective biosecurity plan for the lakes (section 6.5.3).

Table 3-9: Freshwater outcomes for Ōtūwharekai lakes. Modified from Table 1b in the CLWRP (p. 61, 2020) to include categories for the Ashburton lakes.

Management unit	Ecological health indicators				Eutrophic indicator	Microbial indicator
	Dissolved oxygen (min, %)		Temp. (max, °C)	LakeSPI (min grade)	Trophic level index (TLI) (max. score)	Suitable for contact recreation [SFRG]
	Hypolimnion	Epilimnion				
Large high-country lakes	70	90	19	Excellent	2	Good
Small to medium sized high-country lakes	70	90	19	High	4 L. Emily, Māori lakes 3 Other lakes	Good
Natural state waterbodies	Lakes are maintained in a natural state					

Table 3-10: Lake current condition and CLWRP limits and targets. From Bayer and Meredith (2020).

Lake	TLI, average (2015 - 2019)	TLI, average (2017-2022)	TLI limit	LakeSPI condition (2017)*	LakeSPI condition – target**	SFRG
Clearwater	3.86	4.51	3	Moderate	High	Very good
Camp	3.24	3.51	3	**High**	High	Very good
Emma	4.46	4.82	3	Moderate	High	No data
Denny	5.51	5.17	3	Poor	High	No data
Heron	3.15	3.83	3	Moderate	High	No data
Emily	4.23	4.4	4	Moderate	High	No data
Māori East	3.86	4.39	4	Moderate	High	No data
Māori West	4.16	4.62	4	Moderate	High	No data

Note: "TLI Limits are TLI objectives in CLWRP. These objectives may be based on 'annual averages', whereas the 5 year average and current TLIs are calculated from seasonal data" (Table 5.1, Bayer and Meredith 2020). Green = objective met. Red = objective not met. *LakeSPI condition from de Winton and Burton (2017). **LakeSPI targets (objective) from Table 5.4 in Bayer and Meredith (2020). SFRG = suitable for recreation grade Table 5.3 in Bayer and Meredith (2020).

Table 3-11: Lake characteristics and statistics for selected water quality variables derived from SOE data for the period 2015-2019. (Data from Bayer and Meredith 2020).

Lake	Area (km^2)	Depth (maximum) (m)	TN (µg/lL)	TP (µg/lL)	TN:TP	Chl a (mean) (µg/L)	Chl a (maximum) (µg/L)	TLI
Clearwater	1.97	18.0	512	12.8	40	3.2	6.4	3.86
Camp	0.44	13.0	354	6.2	57	2.4	4.4	3.24
Emma	1.67	2.7	632	26.4	24	9.2	29.0	4.46
Denny	0.06	2.1	950	83.8	11	18.1	140.0	5.51
Heron	6.95	36.2	156	5.6	28	4.4	38.0	3.15
Emily	0.19	2.3	430	21.0	20	2.9	50.0	4.23
Māori East	0.10	1.8	564	8.2	69	1.3	137.0	3.86
Māori West	0.09	1.3	466	15.4	30	4.1	80.0	4.16

Table 3-12: Reductions of in-lake concentrations and catchment loads needed to meet CLWRP objectives. From Bayer et al. 2021.

Lake	TN in-lake reduction	TP in-lake reduction	Chl-a in-lake reduction	Estimated TN load reduction*	Estimated TP load reduction*
Clearwater	74%	55%	80%	ND* likely>66%**	>66%*
Camp	52%	NV	37%	>66%*	NV
Emma	76%	70%	87%	ND*	ND
Denny	76%	91%	83%	ND* likely>66%**	ND* likely>66%**
Heron	9%	NV	81%	0-33%*	NV
Emily	25%	29%	40%	>66%*	33-66%*
Māori East	45%	NV	54%	>66%*	NV
Māori West	34%	13%	63%	>66%*	NV

"*Kelly et al. (2021), ND = not determined (outside of model). ** Estimated based on 2017-2021 in-lake data only. Lakes Clearwater, Emma and Denny fall outside the regression model. Lake Emma is likely to be affected by internal loading processes. Lake Clearwater and Denny likely have additional sources of nutrients in their catchments" (Bayer et al. 2021) Font colours are used to highlight reductions over 33%. NV = no values reported in the source document (Table 2, Bayer et al. 2021)

4 Lake prioritisation for mitigation

Lake were assessed, scored and ranked through a process that considered ecological values (section 4.1), current or future pressures and threats (section 4.2), and their magnitude of those threats and pressures (Champion and de Winton 2012, Champion 2014). The approach was based on expert opinion with agreed weighting of multiple aspects. The process was limited to the information available.

4.1 Ecological values

The lake ecological values were assessed across eight categories (habitat size, buffering, water quality, aquatic vegetation diversity, vegetation integrity, endangered species, key species presence, and connectivity). These categories are defined below, and assessment scores against these were assigned to each of the lakes (based on the method of Champion and de Winton 2012). In all cases the maximum score reflects the highest value (also see Appendix E).

4.1.1 Habitat size

The largest and deepest lakes are likely to be the most stable in terms of water quality and resilience, and support the greatest diversity of habitat and biota.

Median values were used when there were a range of values across different information sources. For example, depth values were sourced from LakeSPI (de Winton and Burton 2007, Burton et al. 2022), and bathymetric maps. The scores assigned for area were: >100 ha is 3; 10-100 ha is 2; <10 ha is 1; and 1 ha or less scores 0; and for the depth: >25 m is 3, 10-25 m is 2, 10-2 m is 1. and less than 2 m is 0. The two scores (one for area and one for depth) were averaged to produce an overall habitat size score (maximum score 3 – minimum score 0).

4.1.2 Buffering

Lakes are likely to be most stable when their catchments are predominantly in indigenous vegetation, connected to large wetland systems and are surrounded by extensive beds of emergent vegetation.

Land cover maps from Kelly et al. (2021) based on LCBD5 (land cover database version 5, based on 2018 assessment) and Google Earth Pro images formed the basis for discussion with P. Champion (NIWA).[2] Proportion of native vegetation cover (%) in the catchment, Wetland extent as a % of lake area, and Extent of emergent vegetation cover as a % of lake perimeter (emergent bands >20 m width), were estimated and scores assigned (Table 4-1). The three scores were averaged to produce an overall buffering score (maximum score 3 – minimum score 0).

[2] Also see section 3.1 and the references therein.

Table 4-1: Scoring based on percent native vegetation, extent of wetland and extent of emergent vegetation. Scores from the three metrics were averaged to produce a single value.

Native vegetation % cover in catchment	Score	Wetland extent (% of lake area)	Score	Emergent extent (% of lake perimeter)	Score
>50	3	>100	3	100	3
25-50	2	10-100	2	<100 >50	2
10-24	1	>0 - 10	1	25-50	1
<10	0	0	0	<25	0

4.1.3 Water quality

Unimpacted lakes are likely to have good water quality with a TLI (Trophic Level Index) of 3 or less (oligotrophic). Data supplied by ECan were used to calculate TLI for each of the lakes as an average from 2017 to 2021/22. The TLI scores listed for each lake (see section 3.1) were scored as indicated in Table 4-2.

Table 4-2: Water quality ranking.

TLI Score	Trophic level	Score
<3	Oligotrophic, microtrophic or ultra-microtrophic	3
3-4	Mesotrophic	2
4-5	Eutrophic	1
>5	Supertrophic or hypereutrophic	0

4.1.4 Aquatic vegetation diversity

Lakes are likely to be in the best ecological condition when diverse aquatic vegetation is present.

Data on vegetation composition from the most recent ecological lake surveys were analysed (i.e., the species list Appendix A in de Winton and Burton 2017). The total number of indigenous emergent, free-floating, and submerged species was scored as follows: >20 species received a score of 3; 15-20 species was a score of 2; 5-14 species was a score of 1; <5 species was scored 0.

4.1.5 Aquatic vegetation integrity

A large proportion of cover of littoral habitat by native aquatic plants, and greater depth to which native aquatic plants grow (relative to lake depth) both indicate better lake ecological condition.

The native condition index (NCI) component of LakeSPI scores the integrity of submerged vegetation as a percentage of the predicted pre-European (unimpacted reference) state (Clayton and Edwards 2006a; Clayton and Edwards 2006b, de Winton et al. 2012). Data from the most recent ecological lake survey NCI scores (de Winton and Burton 2017) were analysed and scored as follows: NCI >75% – Score 3; >50-75% – Score 2; >20-50% or 1-20%– Score 1; 0%– Score 0.

4.1.6 Endangered species

New Zealand's endangered biota have been assessed and reported in several publications. Species are assessed using the protocols of Townsend et al. (2008) as Nationally Threatened (Nationally Critical, Nationally Endangered & Nationally Vulnerable), At-Risk (Declining, Relictual and Naturally Uncommon), also recognising new species that have naturally colonised New Zealand (Vagrant or Coloniser). Where there is insufficient information, the taxon was recorded as Data Deficient.

Data from the most recent ecological lake surveys were analysed (Te Rūnanga o Arowhenua et al. 2010, Ure 2016, de Winton and Burton 2017, Burton et al. 2022) (Table 4-3). Each Nationally Threatened taxon was given a score of 5, declining species a score of 2 and other At-Risk and new to New Zealand species a score of 1. These were summed and lakes with an endangered species score >15 was given an endangered species score of 3; 5-15 was scored 2; 0-<5 was scored 1, and 0 was scored when no endangered taxa were recorded. Regionally uncommon species were not assessed. Endangered species were not scored when they were formerly recorded, but not found during the latest assessment. Only plants and fish were used in this scoring, as most endangered birds are mobile and may utilise degraded lakes, as well as those with higher ecological integrity (see section 4.1.8).

Table 4-3: Endangered species records.

Species recorded	References	Status
Longfin eel	Te Rūnanga o Arowhenua et al. 2010, Ure 2016, Freshwater Fish Database (FFDB)	"At Risk - declining", Dunn et al. 2018
Galaxias brevipinnis	FFDB, Ure 2016	"At Risk - Declining", Dunn et al. 2018
Galaxias vulgaris, Gobiomorphus hubbsi	FFDB	"At Risk - Declining", Dunn et al. 2018
Kakahi (*Echyridella menzeisii*)	Burton et al. 2022	"At Risk – declining", Grainger et al. 2018
Carex cirrhosa, Ranunculus brevis, Crassula mulitcaulis, Gratiola concinna	Moss et al. 2021	"Nationally endangered", Champion et al. 2021
Triglochin palustre	Moss et al. 2021	"Critical", Champion et al. 2021
Carex decurata, Ranunculus macropus	Moss et al. 2021	"Data deficient", Champion et al. 2021
Isolepis brasilaris	Moss et al. 2021	"At Risk - declining", de Lange et al. 2017

4.1.7 Presence of key species

Kākahi are very important species in shallow water bodies because they are able to filter feed and remove planktonic algae (Champion 2022). Presence of living mussels in a lake adds an additional point to the scoring. The data source was Burton et al. (2022) or see Section 3.1.

4.1.8 Connectivity

Conning and Holland (2003) noted that the abundance of dune lakes and associated wetlands, although discontinuous, collectively provide important habitat for several species of threatened and regionally significant birds. An additional point was added to the scoring when several lakes occur in close proximity (i.e., the collection of lakes provides greater extent of potential habitat) (e.g., the Māori Lakes).

4.1.9 Total Ecological Value Score

Based on the criteria identified earlier (section 4.1), a maximum total Ecological Value Score of 20 could be attained. Lakes were rated as shown in Table 4-4.

Table 4-4: Lake ecological score and rating.

Ecological Value Score	Rating	Lake
13-20	Very high	Heron
11-12	High	Clearwater, Camp
9-10	High-Moderate	Emily, Māori East and West, Emma
7-8	Moderate	
4-6	Moderate-Low	Denny
<4	Low	

4.2 Lake pressures/threats

Many of the pressures on the Ōtūwharekai lakes have previously been recognised, as has the overall deteriorating trend in lake condition (Bayer and Meredith 2020). Lakes are dynamic systems and are naturally subject to change. Although some alien species have arrived in New Zealand naturally (e.g., mostly through prevailing winds and migratory birds) predominantly from Australia, the introduction of species by humans has occurred at a much greater rate; humans have also introduced species that could not have arrived naturally (e.g., freshwater fish and asexually reproducing aquatic weeds) (Champion and de Winton 2012).

In a natural state, the nutrient status of lakes tends to increase over time in response to input from the catchment; for shallow lakes, this may mean they eventually become wetlands. However, human activities such as deforestation, intensifying agriculture and the use of fertilisers have all contributed to and greatly accelerated the natural rate of eutrophication (Champion and de Winton 2012, Champion 2014).

This section (4.2) seeks to assess and numerically score anthropogenic pressures and threats for each of the Ōtūwharekai lakes with respect to biosecurity (pest plants and fish, non-native fish, spread and incursion risk), and eutrophication (pasture nutrient pressure and in-lake enrichment). Numeric values were assigned so that maximum pressure/threat would result in a score of zero (see Appendix F).

4.2.1 Biosecurity- Pest plants

Invasive submerged weeds have impacted the majority of New Zealand lakes, with most problem species solely spread by human activity (contaminated boats and trailers, fishing nets, diggers and deliberate introduction). In addition to the NCI (section 3.1), the second component of LakeSPI is the Invasive Impact Index (III). The latter index captures the degree of impact from invasive weed species, including assessment of species 'weediness', the proportion of available habitat occupied by invasive vegetation, and invasive depth impact (Clayton and Edwards 2006a, Clayton and Edwards 2006b, de Winton et al. 2012). Invasive submerged weed pressure was scored as follows: III<10% - 3;

III>10 <50% - 2; III >50% -1; no vegetation – 0. Data were sourced from de Winton and Burton (2017), and are also available on LAWA (Land, Air and Water Aotearoa[3]).

4.2.2 Biosecurity - Pest and non-native fish

Invasive pest fish have detrimental impacts on lake ecology, such as predation of native fauna (e.g., the extinction of dune lakes galaxias (*Galaxias* sp.) attributed to *Gambusia affinis* (Rowe (1998)), loss of submerged vegetation and reduction in water clarity. Rowe and Wilding (2012) developed a Fish Risk Assessment model (FRAM) to quantify the potential impact of invasive freshwater fish.

Invasive fish pressure was scored as follows, based on the presence of the highest impact species: no pest fish – 3; FRAM <20 – 2; FRAM 20-25 – 1; FRAM >25 – 0. Fish presence data were derived from Ure (2016), Champion et al. (2006), LakeSPI surveys (de Winton and Burton 2017), the Freshwater Fish Database (FFDB), and local knowledge (i.e., confirmation of perch in Lake Clearwater was provided at the Methven Hui, for this project 21 November 2022).

4.2.3 Biosecurity - Risk of spread

The presence of environmentally damaging pest plants and fish (as scored above) does not predict the likelihood of future incursions of other pests. A range of variables, including proximity to population centres and the roading network, and lake access for the public were considered to estimate the risk of new introduction to each. Information sources were Champion et al. 2006, Champion 2014, de Winton and Burton 2017, and NIWA divers' observations (unpublished).

Risk of spread was scored as follows: not accessible and no motorboats permitted -3, accessible off-roading -2, accessible and no motorboats - 1, proximity to main road and concrete boat ramp – 0.

4.2.4 Eutrophication - Pasture nutrient pressure

Verburg et al. (2010) noted that eutrophication was positively correlated to the proportion of pastoral land use in the catchments of New Zealand lakes. Total percentage pasture cover in the catchment was obtained from the FENZ database[4].

Pasture nutrient pressure was scored as follows: <1% - 3; 1-25% - 2; >25-50% -1; >50% - 0.

4.2.5 Eutrophication - In-lake enrichment

Frequent planktonic algal blooms are indicative of nutrient enrichment. Chlorophyll-a data (chl-a, a proxy of the biomass of planktonic algae) were obtained from the data set provided by ECan.

In-lake enrichment pressure was scored as follows: median Chl a < 2 mg/m^3 – 3; Chl a 2-5 mg/m^3 – 2; Chl a >5-12 mg/m^3 – 1; Chl a >12 mg/m^3 – 0.

4.2.6 Total Pressure/Threat Score

Using the criteria above, a maximum total pressure/threat score of 15 could be attained. Lakes were rated as shown in Table 4-5. The combined ecological value and pressure/threat ratings for each of the 8 lakes are shown in Table 4-6.

[3] www.lawa.org.nz
[4] https://www.doc.govt.nz/our-work/freshwater-ecosystems-of-new-zealand/

Table 4-5: Lake pressure / threat score and rating.

Pressure / Threat score	Pressure Rating	Lakes
13-15	Low	
9-<12	Moderate	Emily, Māori East
<9	High	Clearwater, Camp, Emma, Denny, Heron, Māori West

4.3 Lake prioritisation – summary

By combining the assessment scores, it is possible to rank the lakes in terms of their ecological values and the risks to those values from pressures and threats. For example, a theoretical lake with high ecological values (e.g., score 20) and low pressure (e.g., high score of 13-15) will achieve the highest overall potential score. This is a lake in high ecological condition with few threats or pressures on that condition, and would be prioritised highly for protection. A lake with high ecological values (e.g., score 20) and a high pressure (e.g., <9) will have a lower overall score (than the first theoretical example), indicating greater impediment to maintaining the ecological condition of the lake, suggesting lower priority for in-lake interventions. Lakes in poor condition, with high pressures would be given lowest priority for in-lake interventions or mitigations.

Most of the Ōtūwharekai lakes have scored highly or high to moderate for their ecological values, the exception being Lake Denny (moderate to low), with pressures and threats in the high to moderate category for all lakes (Table 4-6).

The combined scores for ecological value and risk, provide a means to rank the lakes from highest to lowest as follows:

Heron > Māori East > Emily = Clearwater > Māori West > Camp = Emma > Denny

The ranked order of the lakes can then be used to prioritise interventions to protect and improve lake condition, with a priority focus on protection. Protection of existing high condition is prioritised because this is the most cost-effective intervention point, before there are further water quality declines or incursions of non-native species (e.g., Muller et al. 2021). As noted in the methods however, this is one approach to prioritisation – others may be developed to achieve different objectives.

With the exception of Lake Denny there is little difference between the lakes, supporting the urgency for mitigation actions to restore and protect all of the lakes (section 3.2). The in-lake mitigation actions are described in section 6, and catchment options are described in the Part 3 report (Tanner and Sukias 2022); all will take time to implement, and further time will pass before beneficial change is observed. By comparison, a biosecurity awareness campaign could be implemented readily with almost immediate protection benefits for the lakes (section 6.5.3). This could build on the existing weed surveillance schedule that Environment Canterbury already has in place.

Table 4-6: Ecological Value and Pressure/Threat Ratings. VH – Very high; H - High; H-M - High to Moderate; M - Moderate; M-L – Moderate to Low.

Lake Name	Ecological value		Pressure / Threat	
	Rating	Score	Rating	Score
Clearwater	H	12.5	H	6
Camp	H	11	H	4
Emma	H-M	9	H	6
Denny	M-L	6	H	5
Heron	VH	14	H	6
Emily	H-M	9.5	M	9
Māori East	H-M	10	M	9
Māori West	H-M	10	H	6

5 Degradation in lakes: general causes and processes

The ecological health of a lake is influenced by a range of processes that occur within the lake and in the lake catchment. The combination of degrading processes to which an individual lake is exposed is unique, as will be the lake's response to these processes. For example, each lake differs in morphology (area, depth, volume, shape and biology) and these characteristics will affect a lake's response to seasonal and extreme weather events, as well as external and internal factors and processes that control nutrient concentrations and phytoplankton biomass.

The following section provides a general discussion on lake processes that influence lake condition. The role of external processes is discussed in Part 3 (Tanner and Sukias 2022). It is important to realise that restoration of lake health is complex, must consider processes and activities that occur in the catchment and within the lake, and may require substantial time to become noticeable or measurable.

5.1 Nutrients: sources, forms and storage

Growth of algae is largely determined by the availability of two key nutrients – N and P, and the relative concentrations in which these two nutrients occur. Abell et al. (2010) have reported that in other New Zealand lakes, TN:TP ratios <10 indicate that algal growth is likely to be N-limited, whereas when TN:TP ratios > 17 occur, algal growth is likely to be P-limited.

Nutrient input to lakes include surface and groundwater inflows, rainfall, and recycling of legacy nutrients stored in lake sediment. Catchment runoff carries dissolved nutrients, as well as nutrients bound to sediment, suspended sediment and microbial contaminants. Dissolved nutrients are bioavailable, and may be used to support algal growth. Nutrients bound to sediments tend to settle to the lakebed where, depending on environmental conditions, biogeochemical processes can convert insoluble and non-bioavailable nutrients into soluble, bioavailable forms such as ammoniacal-N (NH_4-N) and dissolved reactive phosphorus (DRP).

Recent SOE monitoring data (Table 3-11) shows that in almost all Ōtūwharekai Lakes, the TN:TP ratios are very high. This suggests that phytoplankton growth in these lakes is P-limited. Under these conditions, biologically available P (DRP) will be rapidly assimilated by phytoplankton, causing low DRP concentrations in the water column. Table 3-11 shows that the exception is Lake Denny; the TN:TP ratio indicates that typical lake water quality is in the zone of co-limitation, where addition of either N or P would likely stimulate phytoplankton growth.

Despite the P limitation of most of the Ōtūwharekai lakes, inputs and in-lake concentrations of both N and P should be managed (reduced) wherever possible (Lewis and Wurtsbaugh 2008; Abell et al. 2010; Paerl et al. 2011). However, the most cost-effective way to decrease chlorophyll-a concentrations, and therefore improve lake health, will be to target phosphorus, because evidence shows it is the limiting nutrient. We note that target lake concentrations have been identified for both nitrogen and phosphorus in the CLWP (2019).

The largest store of nutrients in most lakes occurs in sediments – these are often referred to as legacy nutrients (they are the results of historical activities). If these nutrients remain in the sediment, they are likely to have little impact on lake water quality. Several biogeochemical processes that occur within a lake and lake sediments allow these nutrients to move into the water column in bioavailable forms that can be utilised by phytoplankton, negatively affecting lake health. These processes are discussed below.

5.2 Diffusion from the sediments

Sediments and associated organic matter eroded from the catchment are transported to the lake where they are deposited. These sediments may contain soluble nitrogen and phosphorus, which can diffusion out of the sediments and into the hypolimnion.

Particulate bound (i.e., non-soluble) N and P stored in the sediments can be converted into soluble forms, such as dissolved reactive P (DRP) and dissolved inorganic nitrogen forms, primarily ammoniacal-N and nitrate-N. Under aerobic conditions, microbially-mediated processes convert ammoniacal-N into nitrate-N, which may be denitrified and lost to the atmosphere as either N_2O or N_2 gas. If the overlying water is anoxic, the N is released from the sediment as ammoniacal-N and remains available for algal growth.

P is stored in the sediment in several forms (Psenner et al. 1988). Some of these forms are mobile, which means that the P may be mobilised in the porewater and subsequently released into the overlying water column (Meis et al. 2012). This is particularly prevalent when lake sediments are anoxic, because the fraction of P bound to Fe^{3+}oxides will be mobilised as these oxides are microbially reduced to the soluble ferrous form (Fe^{2+}). This releases the P into the porewater as DRP. Vopel et al. (2008) found that sediment is anoxic approximately 5 mm below the surface even when the overlying water column is oxygenated, creating a substantial store of bioavailable phosphorus approximately 5 mm below the sediment surface. When oxygenated water overlies lake sediments, Fe^{3+} oxides react with DRP diffusing from the deeper anoxic sediments, thereby reducing the flux of DRP entering the overlying water column. When the overlying water column has low or no dissolved oxygen, surficial sediments will also have low dissolved oxygen content and DRP is more likely to diffuse through the sediments into the overlying water column. The store of bioavailable nutrients in the sediments can also be mobilised into the water column during sediment resuspension events, such as may occur in storms due to strong winds and wave action, or flood water inflow.

5.2.1 Thermal stratification

Thermal stratification occurs when lake surface waters are warmed by the sun and the overlying atmosphere, causing a temperature difference between surface and bottom waters. Temperature differences cause density differences between these water layers, resulting in the formation of two "water bodies" within the lake: the epilimnion (top) and the hypolimnion (bottom). This density difference causes thermal stratification, which limits mixing of the epilimnion and hypolimnion. Alone, thermal stratification does not have a negative effect on lake health. However, as the hypolimnion is more isolated from sunlight (causing low rates of productivity), is not in contact with the atmosphere (and cannot be oxygenated by gas exchange), and is in contact with nutrient rich sediments, it can become anoxic during periods of stratification. Anaerobic conditions favour release of nutrients to lake bottom waters. When thermal stratification breaks down (by wind events, rain or cooling temperatures), surface waters are mixed down into the hypolimnion. When the lake is partially or fully mixed, the nutrient rich bottom waters enter the photic zone, and the bioavailable nutrients fuel phytoplanktonic production.

The amount of bioavailable nutrient that accumulates in the hypolimnion during stratification is influenced by the duration of stratification, the amount of nutrient stored in lake sediments and the oxygen status of the hypolimnion.

The likelihood of anoxia developing during a period of thermal stratification is largely determined by the rate of oxygen consumption in the hypolimnion which is also known as the hypolimnetic oxygen demand (HOD) or sediment oxygen demand (SOD). This oxygen demand is caused by the bacterially mediated breakdown of organic matter that has accumulated in the lake sediments. Generally, larger the stores of organic matter cause higher rates of oxygen consumption. The source of this organic matter is largely from the catchment and decaying algal biomass .

5.3 Sediment re-suspension

Sediment generally stores more nutrients ("legacy nutrients") than the overlying water column. Most of these nutrients are bound to sediment particles or held in the pore water between sediment particles. When sediments are re-suspended, nutrients stored in the pore water will be mobilised into the overlying water column, where they can stimulate phytoplankton growth and impact adversely on lake health and water quality. There are several in-lake processes that contribute to sediment re-suspension, but here we focus on two: wind-driven wave action, and seiches.

5.3.1 Wave action

Wave action is generated by the wind blowing across the lake surface. Waves can resuspend sediments and in shallow water can mobilise nutrients from the sediments and porewater. The magnitude of currents derived from waves are dependent on the amplitude of the waves (Kumagai 1988, Limnology 2009), and can penetrate to a depth of up to nine times the surface wave height (Figure 5-1). So, a 0.5 m wind wave could penetrate to a depth of 4.5 m with sufficient energy to suspend sediment (Gibbs et al. 2016).

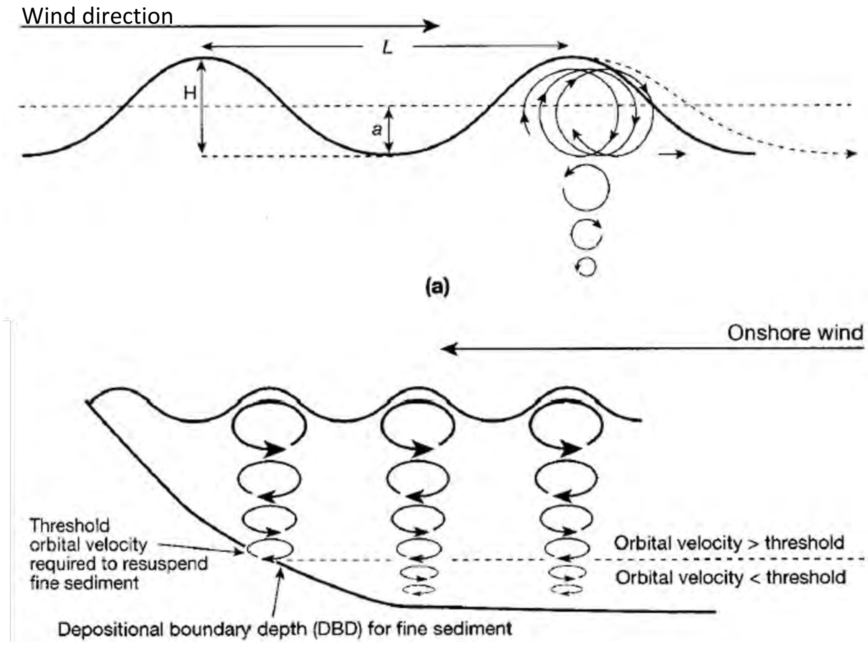

Figure 5-1: Schematic of the orbital motions and lake currents generated by wind stress on the lake surface. Schematic showing orbital motions and lake currents generated by wind stress on the lake surface. (a) The water parcels oscillate elliptically but exhibit only minor forward motion while the vertical movement decreases exponentially with depth. L= wavelength, H = wave height, a = wave amplitude (1/2H). Motion of the water is not to scale. (b) Diagram of the orbital velocity of the elliptical parcels and their ability to resuspend and transport fine sediments along shallow slopes above the DBD. Coarse sediments are found above this orbital velocity threshold where slopes are steep and transport to less turbulent water is possible (from Limnology 2009).

Resuspension of sediment by wind is dependent on the wind velocity at the lake surface and the duration of that velocity of wind (Gibbs et al. 2016; Gibbs et al. 2022). A wind velocity threshold must be exceeded for a specific period of time, before the orbital velocities become established. A recent study on Lake Horowhenua, a shallow (maximum depth 2 m) dune lake, examined wind induced resuspension of sediment relative to changes in lake water quality. Turbidity pulses seen in the water column coincided with wind events that could resuspend the sediment (Figure 5-2a). The relationship between turbidity and wind run (Figure 5-2b) showed that a minimum wind run of about 300 km/d was required to suspend the lake sediments.

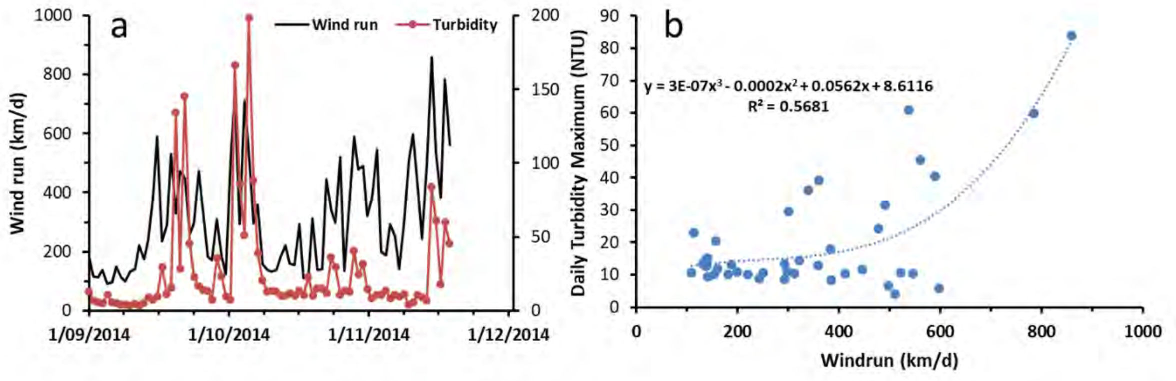

Figure 5-2: a) Correlation between daily wind run and daily mean turbidity and b) Relationship between daily maximum turbidity and daily wind run as the daily wind run increased. Data from Lake Horowhenua, North Island.(Wind data from Metservice: Agent No. 3275, Network No. E05620) (From Gibbs et al. 2022, Supplementary information – with permission).

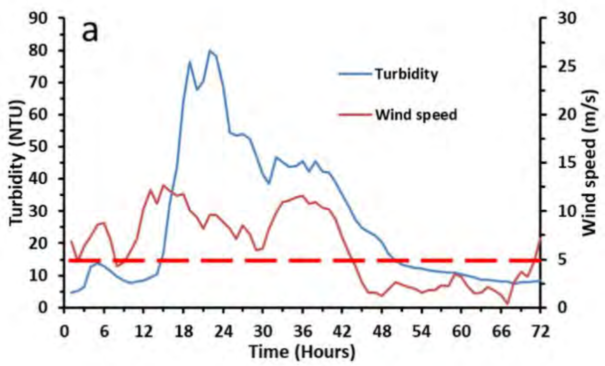

Figure 5-3: a) Time delay for turbidity increase after an increase in wind speed. The red dotted line represents the threshold windspeed for sediment resuspension. Data from Lake Horowhenua, North Island. (Wind data from Met service: Agent No. 3275, Network No. E05620). (From Gibbs et al. 2022, Supplementary information – with permission).

A more detailed analysis of hourly wind speed data showed that sediment suspension occurred when the wind speed exceeded 5 m/s (Figure 5-3a), and that the sediment remained suspended while the wind speed was greater than 5 m/s. The threshold wind speed of 5 m/s and associated lags for resuspension and settling were similar to those found in Lake Rotorua (Gibbs et al. 2016), suggesting that similar wind speed thresholds and lags could apply to the Ōtūwharekai lakes.

5.3.2 Seiches

Another process that can re-suspend sediments and disperse legacy nutrients are seiches (Ostrovsky et al. 1996, Limnology 2009, Kirillin et al. 2015, Cossu et al. 2017). Seiches are standing waves with a periodic oscillation that occur on the lake surface (barotropic) or internally along areas of differing density (baroclinic). A seiche can be created by a several mechanisms – the most common is a prolonged period of steady, strong wind followed by a cessation of this wind.

With barotropic waves, the steady wind forces water to the downwind end of the lake where the water level can be setup by several tens of centimetres, flooding the landscape while the orbital velocities of the nearshore waves can resuspend sediment. When the wind stops, that water flows as a strong full depth current to the other end of the lake and, in shallow lakes (< c. 3 m deep), can resuspend sediment along the whole length of the lake. The surge on shore can also up-root macrophytes (e.g., Lake Waihora).

If the lake is deep enough to thermally stratify and have a stable thermocline (c. 10 m deep), the setup will cause erosion in the wave wash zones between the thermocline depth and the lake surface (Figure 5-4a, green zone), this zone can also be eroded by the bottom water return current during a barotropic wave setup in lakes deeper than 3 m.

With baroclinic waves, the steady wind forces water to the downwind end where the water can cause the thermocline to move downwards (Figure 5-4b). At the upwind end, the thermocline is elevated towards the lake surface. This can cause upwelling of the nutrient rich hypolimnetic water into the epilimnion at the upwind end of the lake, creating the potential for stimulating phytoplankton growth. When the wind stops, the water that has been pushed to the downwind end flows as a strong current to the other end of the lake above and below the thermocline. This results in two circulation currents which move in opposite directions. The vertical movement of the seiche current on the lakebed at each end causes erosion and resuspension of the fine sediments as well as releasing nutrients stored in the sediment pore water. The potential importance of these mixing processes in some of the Ōtūwharekai lakes is demonstrated in section 7.1.1.

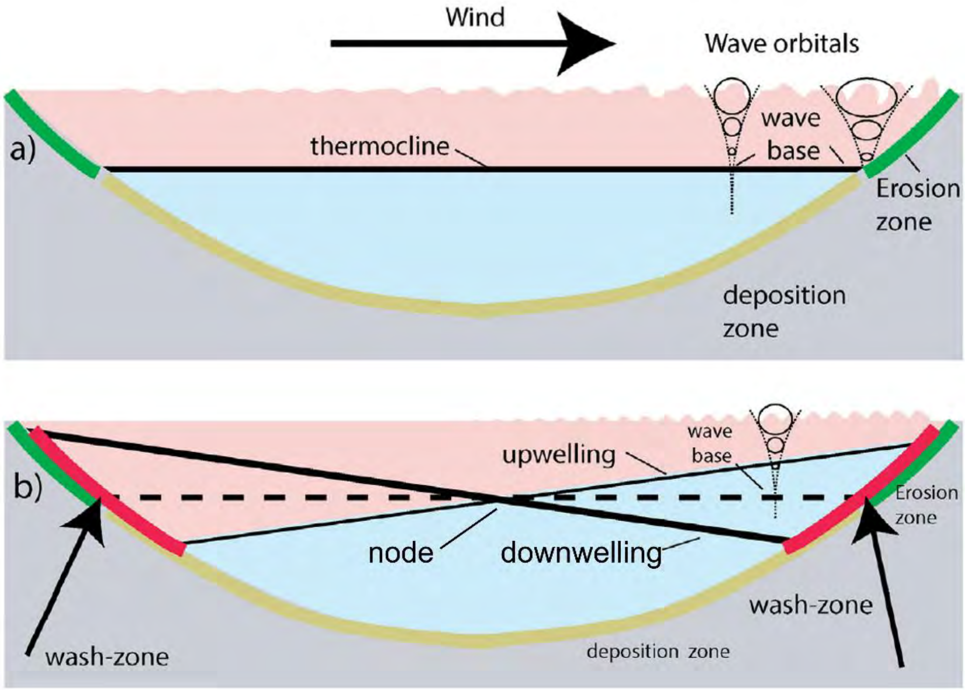

Figure 5-4: Diagrams of erosion zones of a) barotropic waves and b) baroclinic waves. (Adapted from Cossu et al. 2017). The water level at the node on the thermocline or lake surface does not change but the horizontal water flow velocity is greatest at the node.

A study by Tammeorg et al. (2015) examined the relative importance of diffusion and resuspension for phosphorus cycling during the growing season in a large, shallow lake. During spring to early summer, they found the release of P by diffusion was similar in magnitude to release by resuspension. However, in late summer and autumn, resuspension was of greatest importance for P cycling, with the release of P by resuspension at that time being about 40-fold higher than that of P released by diffusion. Assuming similar relationships exist in the Ōtūwharekai lakes, the shift in relative importance over the summer to autumn suggests that the timing of wind disturbance events may be very important for the health of these lakes. It also suggests that these two mechanisms, diffusion and resuspension, could provide a continuous supply of P to the water column during the growing season.

Coarse fish species can also cause sediment re-suspension. This issue is dealt with in the biosecurity section (6.5).

5.4 Localised anoxia

At any time, there are two sources of dissolved oxygen in a lake. The first comes from diffusion from the atmosphere into the lake water, and the second is from photosynthesis by macrophytes and phytoplankton. The internal movement of water distributes the dissolved oxygen within the lake, with the fate and extent of dissolved oxygen movement influenced by multiple factors (e.g., thermal stratification).

Some macrophyte species create dense weed beds, within which the movement of water and associated oxygen may be restricted or slowed. Localised anoxia can develop in these dense beds, potentially increasing the rates of DRP release from sediments. Dense weed beds may increase the likelihood of anoxia by reducing reaeration of lake water from the atmosphere, and increasing the rate of oxygen consumption through respiration (Miranda et al. 2000, Caraco et al. 2006, Vilas et al. 2017).

In addition, the amount of organic matter in various states of decay within a weed bed may increase the demand for oxygen within the bed, contributing to localised anoxia (Vilas et al. 2017). This may increase rates of DRP diffusion from underlying sediment, which may subsequently be transported into more open areas where it could be utilised for phytoplankton growth.

Macrophytes with dense growth forms in Aotearoa New Zealand tend to be invasive alien weeds – native plants tend to be shorter, with more open foliage and growth forms, allowing for more water exchange (Hofstra et al. 2018).

5.5 pH

High pH can also increase the release of DRP from lake sediments (Christophoridis and Fytianos 2006, Jin et al. 2006, Wu et al. 2014). pH levels high enough to facilitate DRP release from sediments can be achieved during periods of high phytoplankton productivity (Gao et al. 2012) and from dense beds of bicarbonate-adapted invasive macrophytes such as *Lagarosiphon major, Egeria densa* and *Ceratophyllum demersum* (Cavalli et al. 2012). These invasive species are not currently present in the Ōtūwharekai lakes (also see section 7.2.3).

5.6 Alien invasive species

Alien invasive species (AIS) may exert stressor effects in concert with stressors mentioned above (see section 3.2 and 4.2).

5.7 Cumulative stressors

Climate change scenarios that include increased atmospheric and water temperatures, and increased frequency and severity of storm events, are well documented (IPCC 2021). These changes will also impact on lake ecosystems, particularly on shallow lakes, where climate change will add to the existing stresses and may result in regime shifts (e.g., Graham et al. 2020, Polst et al. 2022).

6 In-lake mitigation options

Section 3.2 describes the overarching goals for the Ōtūwharekai lakes – halting the decline of lake ecological condition and improving lake ecological condition (section 3.2). From section 5, we have knowledge of the causes of degradation. We now discuss what and where several in-lake mitigation actions can be taken to i) improve water clarity and quality by reducing suspended sediments and algal blooms, and (ii) reduce the impacts of non-native species.

The review of potential mitigation options below describes in general terms when they are most suitable (i.e., the conditions and limitation of their use), before discussing the feasibility of these interventions relative to the desired goals/outcomes for the different Ōtūwharekai lakes (section 7).

6.1 Thermal stratification and hypolimnion anoxia

Stratification disruption, aeration and dosing of P inactivation agents can be used to mitigate the effects of thermal stratification and the potential for anoxia-driven increases in the diffusion of DRP from the sediments.

6.1.1 Stratification disruption

This method breaks up stratification, prevents establishment of anoxic conditions in lake bottom waters, and the associated increased rate of DRP diffusion from the sediments. Stratification disruption can be achieved by creating circulation currents. Generation of a bubble curtain breaks up stratification (Gibbs and Howard-Williams 2018). Bubble curtain aeration systems produce a plume of bubbles that rise from a bottom-mounted aeration device below the thermocline, consisting of a perforated sparge line attached to an air compressor on shore. The design of the sparge line is specific for each lake, but usually consists of a polyethylene tube with 1 mm diameter holes along the top at 30 – 60 cm spacings. Compressed air escaping from these holes forms uniform size bubbles which expand as they rise to the surface. The rising bubble plume entrains hypolimnetic water to the surface where it can adsorb oxygen from the atmosphere. This oxygenated water returns to the bottom (replacing water rising to the surface) with a conveyer belt type current generated by the rising bubble plume. This approach is generally used in lakes deeper than 10 m.

Only three of the Ōtūwharekai lakes would be deep enough to justify use of a bubble curtain aeration/mixing system i.e., Lakes Heron, Clearwater and Camp.

6.1.2 Hypolimnetic aeration

Unlike stratification disruption, hypolimnetic aeration does not break up stratification. Rather it aims to supply oxygen to the hypolimnion to prevent anoxic conditions forming (Liboriussen et al. 2009). The rate of oxygen supply needed to achieve this must therefore be equal to, or greater than, the rate of oxygen consumption in the hypolimnion during stratification. Various methods may be used to achieve the necessary oxygen supply, including pumping water from the hypolimnion to the lake surface to be exposed to, and oxygenated by, the atmosphere before returning it to the hypolimnion, or pumping oxygen into the hypolimnion (Gibbs and Howard-Williams 2018). The efficiency of these aeration systems can be increased five-fold by using pure oxygen in place of air.

6.2 P inactivation

6.2.1 Sediment capping with a P inactivation agent

The purpose of applying a P-inactivation agent is to sequester DRP onto an insoluble carrier that renders this P non-biologically available. Removal of the DRP in this way reduces the impact of thermal stratification and hypolimnion anoxia on water quality by limiting the release of DRP from the sediment; it may also reduce the impact of sediment re-suspension by reducing the load of bioavailable P. This method aims to keep as much P in non-soluble form as possible, rather than preventing the physio-chemical conditions that increase rates of DRP diffusion from the sediment. The most commonly used P inactivation agents are alum, Phoslock® and Aqual-P. Each of these agents have strengths and weaknesses, making each appropriate under different environmental conditions.

Alum is the cheapest and the most readily available P-inactivation agent. It comes in liquid form which is mixed with water before spray application to the lake surface or injected into the bottom waters (hypolimnion) where it forms a floc. It is acidic (pH 2.1) and can reduce the pH of the lake if the alkalinity is low e.g., <80 g $CaCO_3$ m^{-3} equivalent (Cooke et al. 2005). The best operating pH range for alum is 5.5 to 6.5 (Hickey and Gibbs 2009), but it will work from pH 5.0 to 8.5. Below pH 5.0 the floc may not form, and if there is insufficient DRP to sequester, then toxic Al^{3+} may be released. Above pH 8.5 the floc does not bind with DRP (Hickey and Gibbs 2009), and any previously bound DRP may be released when the pH in the lake rises. However, because the pH of lake sediment is mostly between 6 and 7.5, once the floc is on the sediment the DRP is retained permanently. Furthermore, because it is in the surface layer of the sediment, any unused P-binding capacity will be able to sequester DRP being released from the sediment porewater. Alum is also a flocculant and as it sinks to the sediment, will removed suspended particles and DRP from the water column. This attribute allows more flexible in the timing of application is alum will remove P already in the water column. Ideally, application will occur before the onset of stratification and associated hypolimnion anoxia while a greater amount of P is retained in the sediments.

Aqual-P is a pelletised form of alum, which has been used on Lake Okaro (Bay of Plenty). This product was invented in New Zealand and is produced by Blue Pacific Minerals in Tokoroa. In this product zeolite granules are used as a carrier for alum. While more expensive than alum, it has properties that allow it to be used where the lightness of the alum floc may be more problematic. Aqual-P can be applied as a slurry of fine granules at the lake surface from a boat or a helicopter via monsoon bucket. Particle size is important – large granules will sink into the sediment, reducing their sequestration efficacy (Woodward and Hofstra 2018). However, large particles (~0.5 mm Ø) will rapidly settle to the lakebed, forming a uniform cohesive sediment cap (Hickey and Gibbs 2009; Gibbs 2010). This behaviour allows for very targeted application and it is best applied prior to the onset of stratification.

Phoslock® uses Lanthanum, a rare earth metal, to sequester DRP into an insoluble matrix. It has been used widely in Europe for lake restoration. The commercial product is granular; lanthanum chloride is attached to a bentonite substrate which acts as a carrier. The bentonite carrier disperses more than Aqual-P does as it settles, forming a more uniform layer over the sediment. The use of bentonite means there can be a turbid cloud in the water column for several days until it settles. Phoslock® works at a higher pH operating range than does alum (pH 7.5 – 11), which means it can be used during a cyanobacteria bloom. However, at pH below 6.5 it can release toxic La_3^+ trivalent ions if there is insufficient DRP to sequester (Hickey and Gibbs 2009). It should also be applied prior to the onset of stratification and hypolimnion anoxia to retain as much P as possible in the lake sediment.

The application dose of any P-inactivation agent needs to be calculated according to the amount of phosphorus stored in the sediments, recognising that the sequestering efficacy is likely to reduce with time. This decrease over time can be due to the binding sites becoming occupied, or the products sinking into the sediment, being re-distributed within the lake or being buried by incoming sediments. Egemose et al. (2010) showed that alum was the best at sequestering DRP during re-suspension events, followed by Phoslock®.

There is no depth restriction for their use, meaning that P-inactivation agents could be used on any or all of the Ōtūwharekai lakes as whole lake treatments or as targeted applications for specific areas of some lakes. However, social aversion to the introduction of chemicals to natural waters needs to be recognised and discussed well in advance of proposed use. The potential for plants to limit the effectiveness of capping agent application (e.g., unequal sediment cover) must be recognised, as well as the potential for off-target impacts (e.g., adverse effects on existing native plants). This information must be understood to inform their use.

6.3 Wind driven sediment re-suspension

Sediment re-suspension can potentially be mitigated using a number of methods, including the reduction of wind speeds through the use of wind breaks in the catchment, slowing current velocities within the lake with baffles or a greater cover of macrophytes, and increasing the resistance of sediment to resuspension by increasing sediment cohesiveness and/or increasing particle size. Reducing the wind and currents that cause sediment re-suspension comes with a risk of increasing the duration of stratification, anoxia in the hypolimnion, and thereby, the rates of sediment DRP release.

6.3.1 Wind breaks

This comprises planting wind breaks across the upwind end of the lakes, in a manner similar to those used on Kiwi fruit farms in the Bay of Plenty for immediate relief and shelter. Given the substantial knowledge gaps that currently exist, such as: evidence of occurrence, frequency and duration of wind speeds that might induce sediment resuspension in each lake; ability to establish windbreaks of appropriate height and proximity to each lake to reduce wind, as opposed to unintended impacts of prolonged stratification (i.e., wind aids in disrupting stratification), and the implications of climate change predictions of more extreme weather events (IPCC 2021), windbreaks are not recommended as restoration options at this time. In addition, the establishment of wind breaks would introduce vegetation types and visual features that are not currently part of the landscape – these factors would need to be carefully considered by stakeholders.

6.3.2 Flocculants

Flocculants, such as anionic polyacrylamides (PAM), can be used to increase sediment cohesiveness and its ability to resist re-suspension. However, flocculants are generally used in-lake to clear the water column (Gibbs and Hickey 2018), not to prevent sediment re-suspension. PAM has been shown to be highly effective at increasing the cohesiveness of bare soils and reducing erosion on building sites, during motorway construction and in agricultural settings (Hayes et al. 2005). It is likely that these materials could also increase the cohesion of lake sediments. However, there are no references to use of anionic PAM to stabilise lake sediment and these agents would need to be trialled before their use for this purpose could be recommended.

6.3.3 Increasing sediment particle size

Coarser sediments such as sands or gravels could be added to cap the existing lake sediment and provide a protective layer that is more resistant to re-suspension. The costs and benefits of this approach would need careful investigation before it could be recommended. Phoslock® was shown by Egemose et al. (2010) to increase sediment cohesion and reduce sediment re-suspension in a laboratory-based trial.

6.4 Biomanipulation

Biomanipulation refers to the addition, removal or enhancement of specific biota to alter food webs, such as reducing or eradicating pest fish and plant populations, re-establishing native plants in de-vegetated systems, or restocking zooplankton, bivalves or native fish (Burns et al. 2014). Some biomanipulations may be one-off events, but most are likely to require multiple interventions, at least initially, but in some cases for a longer term. Furthermore, desired restoration outcomes are usually only achieved when an integrated approach is followed that makes use of multiple restoration tools (e.g., consideration of all options identified in sections 6 and 7).

Appropriate biomanipulation options must be considered alongside the desired outcomes/goals for the lakes (section 3.2). Options mentioned in the literature that could possibly assist with reducing the duration and severity of algal blooms include stocking of silver carp, stocking zooplankton or kākahi (e.g., kākahi rafts).

Silver carp (*Hypophthalmichthys molitrix*) are an introduced planktivorous fish that have been artificially bred in New Zealand specifically for their potential to control phytoplankton (Rowe 2010). Silver carp feed on suspended particles greater than 0.01 mm in size using specialised filtering apparatus on their gills. Considered opportunistic feeders, silver carp can consume a range of phytoplankton as well as zooplankton and detritus, and they will consume cyanobacteria, including problematic taxa such as *Dolichospermum* and *Microcystis* (Xie and Liu 2001; Ke et al. 2009; Ma et al. 2012). Most information presented here comes from a literature review carried out by Rowe (2010), who concluded that the greatest potential for silver carp use in New Zealand is in cyanobacteria control within eutrophic lakes and ponds. However, there is very little information on the use of silver carp in temperate environments compared to warm tropical or sub-tropical areas, and information available in New Zealand to establish the benefits, or the environmental risks, associated with stocking of silver carp is lacking (de Winton et al. 2013). The prevailing cool temperate environmental conditions in the Ōtūwharekai high country lakes, some of which show at least partial ice cover in winter, may constrain the use of silver carp.

Importantly, silver carp are not an option where a high certainty of outcome for cyanobacteria control is required. The fish may selectively graze large-sized phytoplankton, but this could drive the phytoplankton community to be dominated by smaller species, which can include problematic cyanobacteria. In addition, silver carp can consume large numbers of zooplankton and reduce the potential for zooplankton grazing pressure on smaller phytoplankton, resulting in a reduced effect of phytoplankton removal or even in an increase in phytoplankton biomass (Zhao et al. 2013) and reduced water clarity. Impacts on fish fry that feed on zooplankton in surface waters are also likely. The use of silver carp is not recommended for the Ōtūwharekai lakes.

Zooplankton stocking has been used in experimental investigations to reduce algal abundance in nutrient enriched waters (e.g., Li et al. 2014). However, any attempt to stock zooplankton, must be accompanied by a perch removal programme for the affected lakes, otherwise the zooplankton will simply become fish food for young of the year perch (e.g., Timms and Moss 2004).

Bioremediation of freshwater using the natural filter capacity of kākahi in sufficiently high concentrations has recently been investigated (https://niwa.co.nz/freshwater-and-estuaries/research-projects/freshwater-bioremediation-using-native-mussels-focussed-on-shallow-eutrophic-lakes). In the enriched water of shallow (7 m deep) Lake Ohinewai, modelled stocking densities of ca 40 per m^2 were necessary to have a beneficial impact (Allan et al. 2020). While high stocking densities of kākahi show promise for improving water quality, they should only be viewed as a component of an integrated restoration programme. In the first instance, local kākahi sources would need to be assessed for donor mussels for stocking. Available information (section 3) does not indicate sufficient densities for use elsewhere (i.e., in adjacent lakes or on rafts), nor that juvenile kākahi were present in any of the lakes in large numbers. In the absence of high natural recruitment in the lakes, and with some evidence of poor kākahi health (i.e., shell deformities in Lake Clearwater) any approach to utilise kākahi for restoration must be rigorously developed to ensure that existing populations are not being put at risk (Hofstra et al. 2021).

We recommend that external lake mitigations are implemented prior to in-lake biomanipulations, so that external nutrient inputs can be quantified, planned for, and reduction of loads currently input to the lakes can commence. These actions are likely to reduce the extent of mitigation required from biomanipulation, potentially increasing the efficacy of biomanipulation actions.

6.5 Freshwater pest and undesirable species control

This section targets pest species that impact on lake condition. In this context, pests are defined as non-native species that are currently having an impact on the lakes. The two key species for the Ōtūwharekai lakes are elodea and perch (section 3.1). In addition, the importance of a proactive campaign to reduce the risk of new invasive species entering the lakes is also discussed.

6.5.1 Elodea (*Elodea canadensis*)

Elodea is widely naturalised throughout New Zealand, having first been recorded in the Avon River, Christchurch in 1872. Elodea is a submerged, bottom rooted perennial aquatic plant in the oxygen weed family, that grows in both still and flowing waters. Elodea is a dioecious plant, represented by only one sex in New Zealand, hence reproduction is via fragmentation of brittle stems. Elodea can form dense stands of vegetation in still and flowing water, limiting habitat available for native plants and having a significant impact on their abundance (Wells et al. 1997). Potentially elodea will form nuisance weedbeds in even cool temperate waters, as indicated by its recent establishment in Arctic and Subarctic systems including Alaska (Cery et al. 2016). Elodea is a lesser weed compared with other submerged members of the Hydrocharitaceae that are present in New Zealand (*Egeria densa* and *Lagarosiphon major*) which is reflected in its Aquatic Weed Risk Assessment (AWRAM) score of 46 out of a theoretical 100 (Champion and Clayton 2000).

Control options currently available in NZ that are effective at controlling elodea include herbicide application, harvesting (cutting, mowing), benthic barriers and grass carp (Champion et al. 2019).

Grass carp (*Ctenopharyngodon idella* Val) are the only biological control option that could be used for the control of elodea. Grass carp are herbivorous fish, native to Asia (Cudmore and Mandrak 2004), that were brought to New Zealand to assess their potential use for controlling aquatic weeds in 1966 (Chapman and Coffey 1971), and again in 1971 (Edwards and Hine 1974). Initial studies focussed on feeding preferences (Edwards 1973, 1974, Rowe and Schipper 1985). Grass carp were subsequently released for a variety of field studies in small waterbodies in the North Island to assess their potential impacts (Edwards and Moore 1975, Mitchell 1980, Schipper 1983, Rowe 1984).

Grass carp use is subject to approvals from Department of Conservation (DOC) and Ministry for Primary Industries (MPI). One essential requirement is that fish are contained to the target site. Other matters to consider include: stocking rates required to achieve a targeted level of weed reduction are seldom achieved for extended periods of time, and that grass carp will feed on native macrophytes too (Rowe and Schipper 1985). Furthermore, overstocking has been shown to result in elimination of all submerged vegetation (Hofstra and Clayton 2012). Partial or selective vegetation control is unlikely (Cassani 1996; Bonar et al. 2002; Hofstra 2011). Although grass carp are preferential grazers (they will eat some species first), in time they will consume almost all plants that they have access to, hence there is potential for off-target impacts to outweigh the potential benefits that reducing elodea in these lakes might provide.

The herbicide diquat is the only product registered for aquatic use that is efficacious on elodea. Diquat (diquat dibromide) is the active ingredient (20% a.i.) in Reglone®. This herbicide is used in New Zealand for agricultural operations (root crop desiccation pre-harvest) and has been the primary method of large-scale control of aquatic weed beds in New Zealand lakes and reservoirs since 1960. When diquat is in contact with the green parts of nuisance aquatic weeds (leaves and stems) it is rapidly absorbed, producing peroxide that acts like a bleach, desiccating plant tissue and disrupting cell membranes. Diquat is rapidly removed from the water and is deactivated by adsorption onto negatively charged inorganic and organic compounds in the water and sediments (Clayton and Severne 2005). Adsorbed diquat or diquat bound to sediment has no residual toxicity, and over time this inactive bound form of diquat is degraded by microbial organisms. Weed beds can be controlled with diquat at any time of the year, although efficacy is better in the warmer months (Netherland et al. 2000) and plant decay rates are slower in winter. Effective control is best when plants are clean (i.e., little or no epiphytes, detritus or aufwuchs), and water movement is minimal. Water clarity is also an important consideration, since turbid water can significantly reduce diquat efficacy. Important native plant species, such as *Chara* and *Nitella* species, are not affected by diquat (Clayton 2004, Clayton and Severne 2005, Netherland 2014, https://niwa.co.nz/freshwater-and-estuaries/tools/biosecurity/diquat).

Physical weed control is a broad category involving vegetation or biomass removal (e.g., mechanical or manual harvesting), or habitat manipulation such as benthic barriers to smother plants. Cutting and harvesting is a weed control (reduction) method, not an eradication tool. For cutting and harvesting, a boat-mounted sickle bar cuts the weed below the water surface and the weed is entrained onto a conveyor belt as the harvester moves forward. The method is not selective, so all plant species within reach of the cutter bars are cut. The collected lake weed may then be transported to shore directly for "out-of-lake" disposal. Consideration should be given to the risk of fragmentation and weed spread (Champion et al. 2019). Harvested weed may be shredded using a boat-mounted unit to reduce the bulk of harvested material thereby increasing the amount of weed that can be harvested prior to off-load at the shore, or for in-lake disposal (Sabol 1987, Madsen 2000, Hofstra et al. 2015). Mechanical harvesting will not remove all weed biomass and weed beds can re-establish relatively quickly from remnant stems. To manage the regrowth, harvesting may need to be repeated within a growing season (Howard-Williams et al. 1996). Aquatic weed harvesting for nutrient remediation purposes is only possible under specific situations (Verburg et al. 2018) and is unlikely to be feasible in any of the Ōtūwharekai lakes.

Benthic barriers, also known as bottom lining, can be used to smother submerged aquatic plants initially, and impede access to the substrate for rooting by plant fragments or propagules (Hofstra et al. 2015). The use of benthic barriers for localized control of aquatic weeds is well documented

(Peterson et al. 1974, Perkins et al. 1980, Engel 1983, Nichols and Shaw 1983, Jones and Cooke 1984, Killgore 1987, Gunnison and Barko 1992, Newroth 1993, Payne et al. 1993, Carter et al. 1994, Eakin and Barko 1995, Eichler et al. 1995, Helsel et al. 1996, Madsen 2000), and many products are readily available for small-scale use. Considerations for successful weed control with a benthic barrier include choice of product, permeability of the product, ease of placement and retrieval (if necessary), the nature of the site (gradient, underlying substrate, wave fetch) and duration of barrier placement required. For example, wind or wave exposed sites with steep shores are not suitable for benthic barriers. The ability to secure the benthic barrier at a potential site, in a way that supports the uses of the area, must also be considered (Champion et al. 2019).

Non-permeable products, such as plastic sheeting, inhibit exchange of water and gases between the benthos and the water column; these products may be difficult to install and secure to the bottom of the lake and may billow-up in places due to ebullition of gas from decomposing plants. Non-permeable benthic barriers are generally nonselective, in that all plants are smothered and controlled over time, with some variation between species in their ability to withstand shading for periods of time. Plants may also grow through slits in barriers made for gas release, and with permeable barriers some species may persist and grow through the apertures, as well as on top of the barriers from newly arrived plant fragments, particularly once suspended sediments have accumulated.

Jute or hessian fabric has also been used to smother invasive weeds with the added benefit that desirable native plants are able to regenerate from the seed bank below and grow through the small apertures in the barrier fibre (Caffrey et al. 2011). This type of benthic barrier provides for native plant re-establishment as the barrier degrades (Caffrey et al. 2011, Hofstra and Clayton 2012), and in recent years has been used to control lagarosiphon beds in Lake Wanaka (de Winton 2020a). Another advantage is the natural decomposition of the barrier, with breakdown evident c. 7-10 months before disintegration (Caffrey et al. 2010).

Benthic barriers are appropriate for control of submerged weed species in small areas (such as boat ramps) or systems. Hessian is currently in use in Lake Wanaka to control the weed *Lagarosiphon major* at a large scale (km of shoreline). Weed control can be achieved for a number of years, with little on-going costs, depending on the site, and the choice of barrier product used (de Winton et al. 2013). For example, most effort is required during the installation phase, although regular (annual) checks should be undertaken to ensure the barrier has not shifted, or to identify whether any repairs are required (dependent on product choice and site conditions). Temporary dropping of water levels may be advantageous for the installation of benthic linings. Likewise, the removal of weed biomass by harvester or herbicide might be required prior to laying benthic barriers (de Winton et al. 2013). Biodegradable benthic barriers can be used in combination with other methods (e.g., localised hand weeding), and a monitoring programme, to achieve eradication of weeds at treated sites. However, the potential for re-infestation must be considered and managed.

As with grass carp, these tools will not selectively remove elodea without some off-target impacts, although those impacts could be managed more effectively than with grass carp use in these lakes. In addition, any off-target impacts with these other methods would not be long-lived i.e., only for the operational timeframe of the benthic barrier (for example), as opposed to grass carp that once stocked could be present for decades (Hofstra and Clayton 2012). The current state of the lakes indicates that although elodea has an impact (III scores, section 3.1) it is a lesser weed (Champion and Clayton 2000, AWRAM), with which native plants are able to coexist. Any management interventions to reduce or eradicate the elodea must be weighed against the risks of destabilising the lakes, when there are already significant pressures (section7), and none of the current control methods could selectively remove only the elodea.

Control of elodea (reduction in biomass or eradication) is not recommended for any of the lakes at this time.

6.5.2 Perch (*Perca fluviatilis*)

Perch are a designated sports fish by Fish and Game New Zealand. In the context of the Ōtūwharekai lakes (and specifically in this report) they are recognised as a non-native fish with undesirable impacts on lake ecology.

Perch first established in New Zealand between 1868 and 1877 in Canterbury, the West Coast, Wellington, Whanganui and Taranaki. They were subsequently spread to other parts of the country and are now present in lakes, ponds and reservoirs throughout most of the west coast of the North Island and the east coast of the South Island. Stocking was carried out in the 1880s and early 1900s by early settlers to create sports fisheries. Since the 1970s, new populations have been established illegally in many lakes and ponds to create coarse fishing opportunities. Deliberate illegal introduction as a source of food is a likely, but currently unquantified dispersal pathway (Champion et al. 2020).

Through their life perch undergo several size-related shifts in their diet and habitat use, so can affect different aspects of freshwater ecosystems. In lakes, larval perch are generally pelagic zooplankton feeders that form in shoals in shallow, open water and along littoral zones. Mid-sized perch (30–80 mm long) feed mainly on benthic macroinvertebrates, while larger perch (130–180 mm) become predominantly solitary and piscivorous (Closs et al. 2003), consuming prey up to a third of their length, including smaller perch (Collier and Grainger 2015). Perch will reduce the abundance of common bullies (*Gobiomorphus cotidianus*) and planktivorous fish (i.e., smelt (*Retropinna retropinna*) and galaxiids (*Galaxias* spp.)) and kōura (freshwater crayfish *Paranephrops planifrons*) in lakes. In addition, perch are associated with the development of cyanobacterial blooms in lakes through predation of zooplankton, reducing zooplankton grazing pressure on algae (Rowe 2007). Perch “may contribute to elevated suspended sediment levels through bioturbation when present in high numbers in shallow lakes. As well as affecting turbidity through sediment resuspension, bioturbation can disturb benthic habitats for native biota (e.g., freshwater mussels), mobilise nutrients and recruit resting cyanobacterial colonies into the water column” (Adámek & Maršálek 2013 in Collier and Grainger 2015).

Although the FRAM score for perch is 45 (i.e., 2nd worst pest/non-native fish), perch have no status under the Biosecurity Act, and are regarded as a sports fish (Freshwater Fisheries Regulations 1983) requiring a Fish and Game licence to catch them.

There are examples where perch removal has been/is sought from lakes in New Zealand. However, there are no specific control methods for perch, and past control or eradication programmes have relied on sustained fishing (setting nets) or rotenone. For example, Collier and Grainger (2015) report that Auckland/Waikato Fish & Game staff are actively looking for a means to remove perch from Lake Ototoa to restore the highly-valued dwarf īnanga and rainbow trout populations. Surrey and Neale (2015) describe the control of perch in Lake Wainamu (Auckland), where the netting programme removed nearly 20,000 perch over a decade. But Surrey and Neale (2015) conclude that “there is no compelling evidence that the programme has reduced the size or changed the size structure of the perch population within the lake”. There is recognition that the persistent application of traditional fishing methods can reduce populations of fish to manageable levels in the short term, but continually fishing any commercially undesirable species is economically unsustainable. Complete elimination of any species by fishing is unlikely given the exponential increase in effort required as catch-per-unit-effort declines (Ling 2002). Other than complete and prolonged dewatering, toxicants, such as rotenone, are considered the only method that is likely to completely eliminate undesirable fish in a body of water (Ling 2002).

Rotenone is a natural toxin produced by several tropical plants and has been used for centuries as a selective fish poison and more recently as a commercial insecticide. It is highly toxic to fish and other aquatic life, but has low toxicity to birds and mammals and is not persistent in the environment (Ling 2002). Rotenone is registered for use in New Zealand as a pesticide to control insect pests (insecticide) on ornamental and crop plants. It is sold widely in garden centres and supermarkets as Derris Dust. Non-target fish species in New Zealand are likely to be greatly affected by rotenone treatments designed to eliminate nuisance fish species, and any eradication programme should assess the potential impact on non-target species in order to compare the relative merits of dispersed applications or rotenone baits (Ling 2002).

Rotenone has been applied successfully in New Zealand in small waterbodies (e.g., the 0.7 ha Lake Parkinson near Auckland—Tanner et al. 1990; Rowe & Champion 1994) and routinely by the Department of Conservation to control and eradicate invasive fish (Hicks et al. 2015). It has been used at 58 sites for pest fish control and perch have been eradicated from 6 sites (Grainger 2015).

Recent research initiatives to control or eradicate perch include the potential for spawning disruption to be used in an integrated approach to managing populations, but this investigation is in its early stages (Baker 2022).

We recommended that the appetite/will amongst mana whenua, mandated agencies with management responsibility, and other stakeholders, be investigated for all lakes where potential exists for the removal of perch (Clearwater, Camp, Emma, Denny).

6.5.3 Biosecurity awareness - advocacy to prevent new species introduction

The value of biosecurity to prevent new incursions cannot be overstated. Recent publications highlight the benefits, both economic and ecological, and the feasibility of acting early to prevent the establishment of invasive species (Muller et al. 2021, Ahmed et al. 2022).

Assessment of high-risk species yet to become naturalized within a region (Champion et al. 2020); likelihood of their introduction (identify nearest sources) and potential introduction pathways, can be used for the preparation of an exclusion list (Champion et al. 2018).

Many of the species are likely to be managed by exclusion from sale, propagation and distribution through the National Pest Plant Accord, and regional council staff routinely inspect plant nursery and aquarium outlets for compliance. However, deliberate and accidental spread remains a possibility.

Passive surveillance and reporting of threat species by the public could be achieved through public awareness campaigns. Additionally, increasing awareness among individuals involved with recreational boating and fishing of these species and the mechanisms likely to spread aquatic weeds and other freshwater pests would be cost-effective measures.

MPI Biosecurity New Zealand currently leads the Freshwater Biosecurity Partnership Programme, with two Regional Coordinators that organize education campaigns focused on boat users, in high use sites, throughout New Zealand.

NIWA created an inventory of aquatic plants, and prioritised weed surveillance actions in the high-country lakes of the Canterbury Region in 2005 (Champion et al. 2006). This report included a ranking for lakes, including the Ōtūwharekai lakes, based on the risk of lagarosiphon introduction and predicted impact. Lagarosiphon is a submerged aquatic weed that is already established in large South Island lakes (e.g., Lake Wanaka). Lake Camp had the highest risk rating (moderate) of the Ōtūwharekai lakes, and annual surveillance was recommended.

Since then, a recommended surveillance schedule has been followed, including annual surveillance at Lake Camp boat ramp, three-yearly surveillance at Lake Heron, and five-yearly checks at main access points (and LakeSPI sites during surveys) for Lakes Clearwater, Denny, Spider, Donne, Roundabout, Māori West, Māori East, Emily and Emma (de Winton 2020b).

We recommended that the surveillance programme is continued and that a targeted community awareness campaign is designed for the Ōtūwharekai lakes, with the objectives of preventing new incursions of invasives species and reducing the risk of spread between lakes of non-native fish that are currently present.

7 Lake specific issues, potential mitigations and recommendations

In this section we use the available data to assess the possibility that internal mechanisms are contributing to the declining water quality in the lakes, as opposed to the nutrients and sediments that are entering the lakes from external sources (catchment loads) covered in Part 3 of this project (Tanner and Sukias 2022).

However, the data available were often inadequate to definitively identify whether internal processes (and the mechanisms involved) are contributing to the decline in water quality observed in the Ōtūwharekai lakes. We applied the following logic to assess whether the loading that is driving the decline in water quality was sourced internally or externally:

- Evidence indicates that all of the lakes, apart from Lake Denny, are P limited with respect to algal growth and therefore an increase in the availability of DRP would result in an increase in chlorophyll-*a* concentration.
  - All SOE data are collected in summer when solar radiation is greatest, so light availability is unlikely to be a factor limiting algal growth, provided lake water turbidity is low.
- Externally sourced phosphorus is likely to be attached to particles, and not immediately available to phytoplankton, whereas internally sourced phosphorus (phosphorus released from sediments as DRP) is bioavailable.
  - Internal cycling of P, released as DRP into the water column, should be expected to result in an accompanying increase in chlorophyll-a.

However, what is summarised above is to some extent speculative, and we recommend that data should be gathered to increase the certainty regarding the relative importance of various internal loading processes – this information is important because it will allow the mitigation actions most likely to improve lake ecological condition to be identified and implemented.

7.1 Lake Clearwater

7.1.1 Degradation

Available information suggests that several internal processes may be contributing to the degradation of Lake Clearwater, including sediment re-suspension caused by wind-induced mixing, anoxia-driven P release from sediments, the presence of perch, and input of nutrient loads from the catchment. The role of wind in lake mixing is suggested by the correlation between wind speed and mixing, where mixing occurs (as shown by temperature at different depths becoming more similar), immediately after periods with high wind gusts (Figure 7-1).

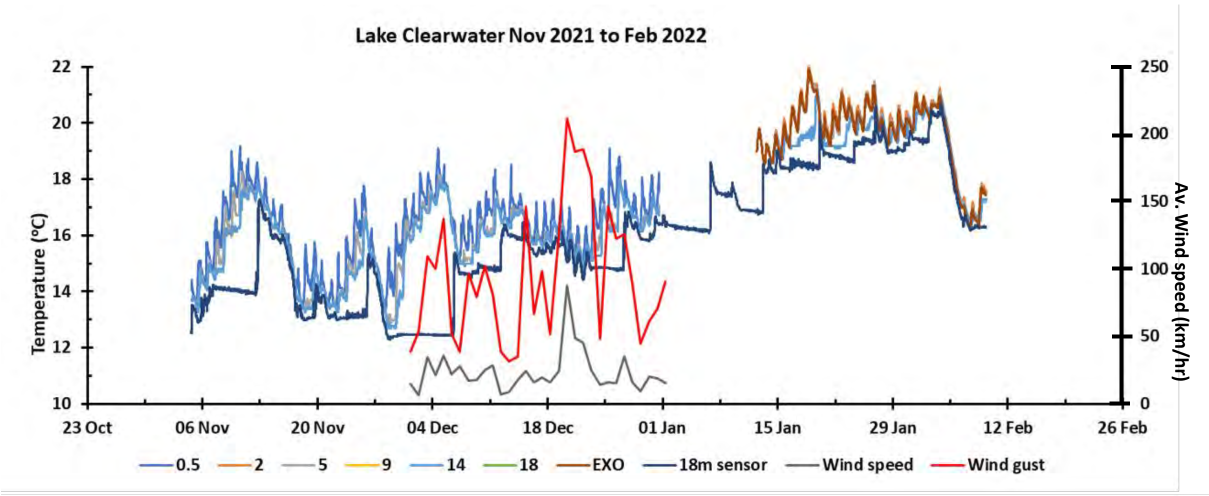

Figure 7-1: Average wind speed (black) and maximum wind speed of gusts (red) and thermal stratification at Lake Clearwater. Wind speed data from the Mount Potts Electronic Weather Station (EWS) and NIWA Virtual Climate Station Network (VCSN). The depths of the temperature sensors are in meters and "EXO" refers to an EXOsonde that was deployed at 2 meters depth. The "18 m sensor" is the temperature on the DO probe deployed at this depth.

This pattern is consistent with a wind setup and relaxation process, where the wind pushes the water to one end of the lake and holds it there until wind speeds decrease and the water then sloshes back and forth within the lake until equilibrium conditions are re-established. The currents associated with this back and forth movement as a barotropic wave mix lake waters and can cause sediment re-suspension. This pattern of mixing once wind speeds dropped occurred on the 6th, 12th and 26th of December 2021. The time series data also showed that the lake became completely mixed early in February 2022 and began cooling (Figure 7-2). However, there was no stratification during the strongest winds on the 19th of December and stratification formed as these winds were dropping on the 23rd and 24th of December. If this mechanism was primarily responsible for deteriorating conditions in Lake Clearwater it seems plausible that Lake Clearwater would not be known as a clear lake, but rather that degradation would have occurred long before recent increase in TLI. In addition, the bottom depth limit of macrophytes was ca 7 m in 2017 (de Winton and Burton 2017) – combined with knowledge that water depths below ca 6 m account for ca. 90% of the area of the lakebed (Figure 3-2), this indicates that lakebed sediments would likely be well protected from mobilisation.

In the summer of 2013, kākahi deaths occurred in the adjacent Lake Camp (Beech, 2013) during an extended calm period which caused thermal stratification and hypolimnetic anoxia. From the time series DO data (Figure 7-2) the average rate of hypolimnetic oxygen depletion (HOD) in Lake Clearwater was estimated to be 1.1 g O_2 $m^{-3}d^{-1}$. This suggests it would take approximately 10 days for the hypolimnion to become anoxic. The bathymetry of the lake spatially limits the extent of the hypolimnion to the deeper hole in the middle of the lake, and hence limits the amount of DRP potentially mobilised. However, at the time of the Lake Camp kākahi die-off event, SOE monitoring in Lake Clearwater showed TP concentrations reached 27 mg m^{-3} – this P could have been released from the sediments due to thermal stratification and anoxia. It is likely that longer periods of stratification have occurred in other years, also resulting in sediment P release.

In addition, perch have been confirmed in Lake Clearwater. Even though perch are likely to be less abundant in Lake Clearwater than in Lake Camp, given that divers observed large schools in Lake

Camp in 2017 but did not observe any perch in Lake Clearwater that same year, their presence may be starting to exert pressure on the water quality of lake Clearwater.

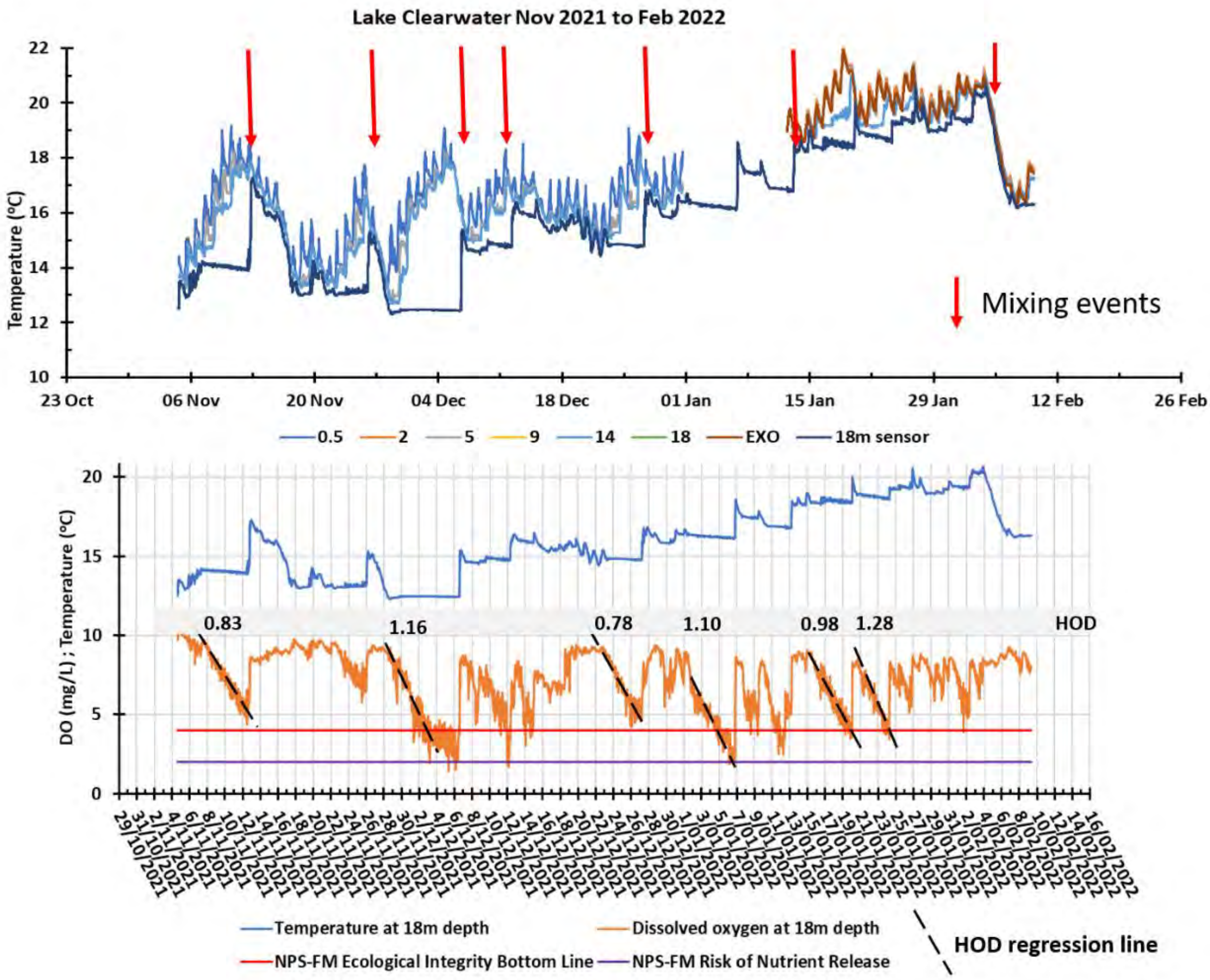

Figure 7-2: Lake Clearwater temperatures (top) and bottom water dissolved oxygen (DO) at 18 m (bottom figure). Full depth lake mixing occurs where the bottom temperature rises vertically to equal the surface temperature (red arrows on upper graph). Regression lines (dashed lines on lower graph) were used to calculate hypolimnetic oxygen depletion (HOD) rates (mg of oxygen L^{-1} d^{-1}); these rates are shown in the centre panel in the lower figure above.

7.1.2 Potential mitigations

Methods to increase sediment cohesion and reduce P mobilisation may be appropriate. A P-inactivation agent could be used in the deep hole in the middle of the lake as this is the most likely location for anoxic conditions to form. A coarser grained P-inactivation agent such as Phoslock or Aqual-P could be applied to the wider lake, as it would provide some protection against sediment resuspension as well as sequestering P mobilised during these events (with caveats, see 6.2.1).

7.1.3 Recommendations

The SOE monitoring data are insufficient to accurately predict the effects of climate variability on the water quality and health of Lake Clearwater. This is because the period between sampling is too large and requirement to sample in calm conditions is unlikely to capture potential nutrient release events due to sediment resuspension from high wind gusts.

Historically, the variables measured were turbidity, temperature, chlorophyll-*a*, TN and TP. Recently DRP, NO_3-N, NH_4-N have been added. Total Dissolved Nitrogen (TDN) and Total Dissolved Phosphorus (TDP) are also being reported, allowing the calculation of particulate N (PN) and particulate P (PP). We recommend continuing with measurement of the extended suite of variables because they will provide insight into the mechanisms of internal nutrient loading within the lake. Important variables that should be added to the monitoring are pH (if it is not already being measured) and turbidity. Turbidity should be continuously recorded to identify if sediment resuspension is occurring during high wind events.

The use of a thermistor chain with DO loggers should be continued through several summers to identify if stratification events of sufficient length occur, creating the anoxic conditions that facilitate DRP releases. If deployment of a thermistor chain is not continued, we recommend that an estimate of HOD is taken by measuring sediment oxygen demand (SOD – see glossary for further details).

Additionally, taking targeted DRP samples from the deepest point in the lake when it has been still for a prolonged period would help assess the role and extent of anoxia-induced sediment DRP release. Similarly, targeted investigations of pH and DRP or DO during prolonged still periods in summer within the macrophyte beds will help assess role of localised anoxia and pH to internal nutrient loading. Data derived from a few instances of both of these measurements would provide an understanding of the potential role that anoxia, and localised anoxia and pH may play in internal nutrient loading.

Measuring the size of the legacy nutrient stores, specifically: total carbon, nitrogen and phosphorus within the lake sediments, and sediment bulk density, would provide insight in the magnitude of legacy nutrient stores within the Lake. This information would also help inform the amount of P inactivation agent that would be needed to treat the lake.

Conversations about perch, their impacts, consequences of their continued presence and management options should be initiated amongst stakeholders.

7.2 Lake Camp

7.2.1 Degradation

Lake Camp is polymictic, experiencing short periods of stratification. Unfortunately, there are no continuous temperature and dissolved oxygen data available to assess the potential impact of stratification on oxygen concentrations in the hypolimnion. However, a kākahi mass death event in February 2013 occurred during an extended period of calm weather which caused thermal stratification and hypolimnetic anoxia (Beech 2013). The calm weather period began on 27th January and ended on 23rd March (Figure 7-3).

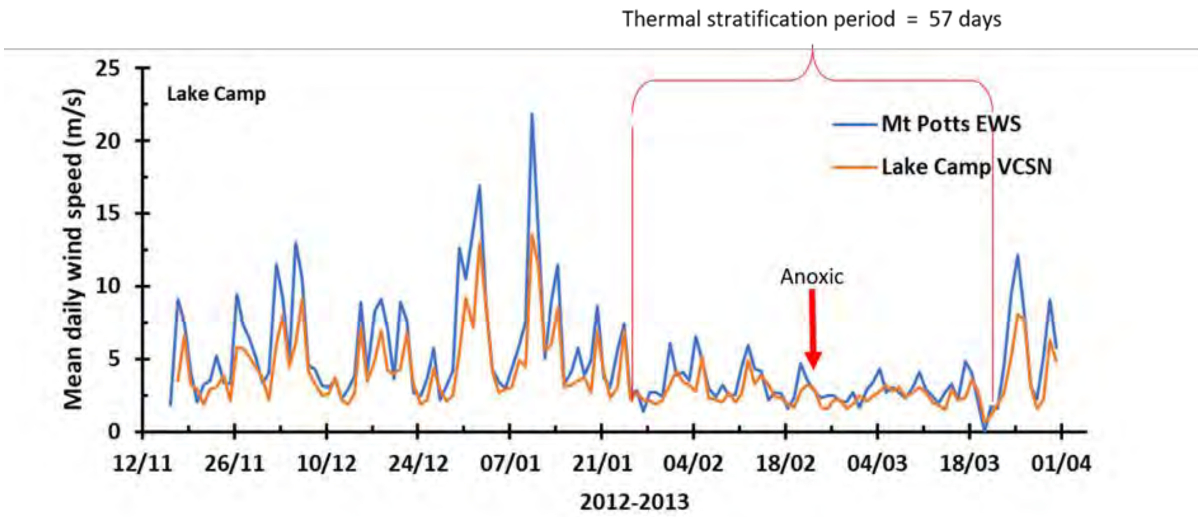

Figure 7-3: Wind stress over Lake Camp showing the period of low wind between 27 January and 23 March 2013. Thermal stratification occurred, allowing hypolimnetic oxygen depletion to develop. The red arrow labelled 'Anoxic' was a field observation on 20th February, confirming hypolimnetic anoxia a few days after dead kākahi were found in the lake. Wind speed data from the Mount Potts Electronic Weather Station (EWS) and NIWA Virtual Climate Station Network (VCSN).

The wind speed records from Mount Potts for Lake Camp during this period showed that daily mean wind speed was below 5 m/s for 57 days (Figure 7-3). The development of thermal stratification during this period is therefore consistent with wind speeds below the 5 m/s threshold that were required to induce lake currents that break up stratification in another shallow lake (Gibbs et al. 2016). The comparison of the windspeed data from Mount Potts meteorological station and the NIWA Virtual Climate Station Network (VCSN) data (Figure 7-3) indicates good agreement. This means that either wind record could be used to assess the occurrence and duration of windspeeds likely to result in stratification during summer. Conversely, the wind records could be used to identify periods when high speed winds could induce wave action or lake currents that could resuspend sediment.

Lake Camp (Figure 7-4a), being a small lake adjacent to Lake Clearwater, it likely experiences the same wind stress as Lake Clearwater. Because of its elongated basin shape, it could in theory develop a barotropic seiche when the lake is not thermally stratified and could develop a baroclinic seiche during periods of thermal stratification that could erode the lakebed at both ends of the lake (Figure 7-4b). However, the high cover of macrophytes and depth limits (e.g., charophyte meadows to 11 m (de Winton and Burton 2017)) do not support this mechanism of seiche driven sediment resuspension.

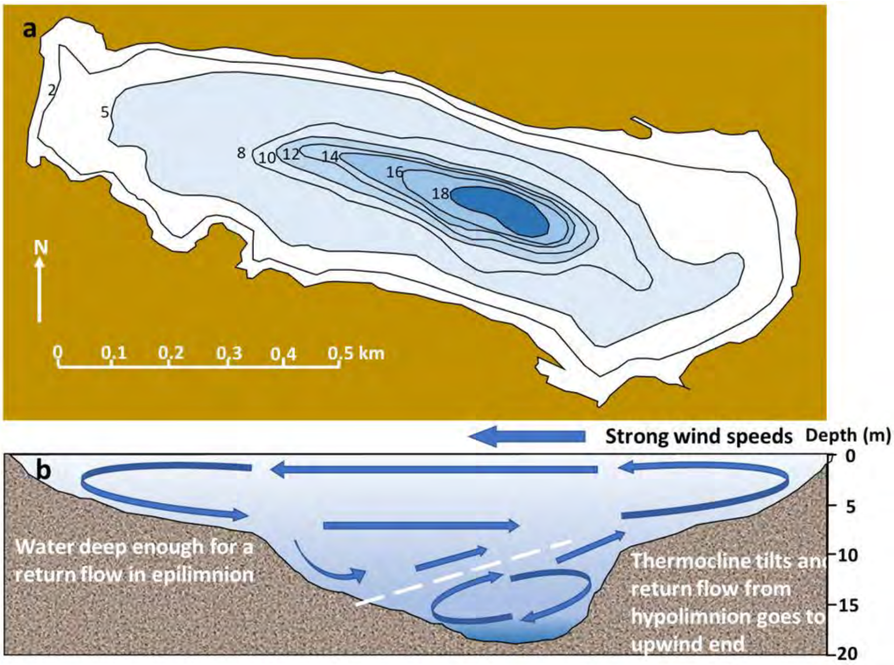

Figure 7-4: a) Bathymetry of Lake Camp, b) stylised circulation likely to occur when the lake is thermally stratified. Increasing wind strength will cause a surface current downwind and a return flow opposite to the wind along the top of the thermocline. The thermocline will tilt downwind, squeezing water out of the hypolimnion into the epilimnion along the bottom at the upwind end, and the lake will mix.

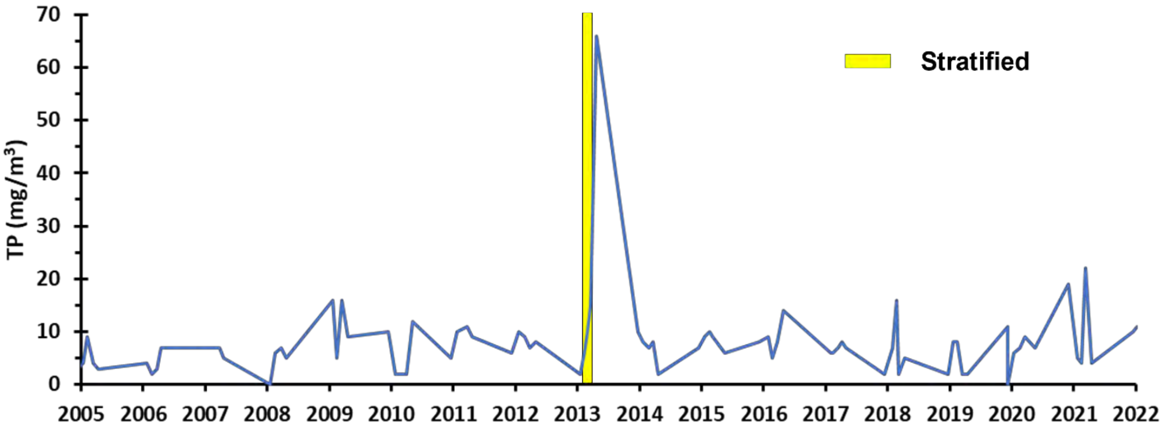

Figure 7-5: SoE monitoring timeseries SoE TP concentrations in Lake Camp showing stratified period.

SOE samples from 24th April 2013 had elevated TP concentrations (Figure 7-5), TN and turbidity but not chlorophyll-*a*. A detailed examination of the SOE data shows that samples were collected on 8th and 28th February, 20th March and 24th April. The samples in February and March were collected while the lake was strongly thermally stratified, when nutrient released from the sediments would be held within the hypolimnion. The lake mixed on 23rd March just after the 20th March sampling, and was not sampled again until 24th April by which time any phytoplankton growth would have senesced, leaving just the TP, TN and suspensoids (i.e., turbidity). No bottom water samples were collected. These would have shown any accumulation of DRP and NH_4-N in the hypolimnion.

Earlier and subsequent TP data show no other elevated concentration events (Figure 7-5), which might suggest there have been no other stratification events. However, the timing of the SOE sampling from the February 2013 event shows that the sampling frequency could easily miss an event.

In addition, perch also contribute to poor water clarity through bioturbation when present in high numbers in shallow lakes (section 6.5.2).

7.2.2 Potential mitigations

Our current understanding of internal processes are that the lake can thermally stratify and develop an anoxic hypolimnion, that can be broken up by wind generated currents, but these events occur infrequently. Hence, although an aeration system may reduce the incidence of diffusive P release, it would not be required for most of the year.

A whole lake application of a granular P-inactivation agent such as Phoslock® or Aqual-P (with caveats, see 6.2.1) could provide some protection against sediment P release.

7.2.3 Recommendations

The installation of a thermistor chain with top and bottom DO sensors would provide the information required to better interpret the SOE monitoring data. If a thermistor chain is not installed it is recommended that an estimate of HOD is taken by measuring sediment oxygen demand (SOD).

The SOE parameters measured should include pH and DRP. If possible, bottom water samples collected for DRP, NH_4-N and NO_3-N analyses should be collected on occasions where calm weather has prevailed for several days before sampling. Turbidity should be continuously recorded in the lake using turbidity sensors to identify if sediment resuspension is occurring during high wind events.

Additionally, taking targeted DRP samples from the deepest point, and an investigation of pH and DRP or DO during a prolonged still period in summer, and measuring the size of the legacy nutrient stores (total carbon, nitrogen and phosphorus) within the lake sediments are recommended (as detailed above, 7.1.3).

Conversations about perch, their impacts, consequences of their continued presence and management options should be initiated amongst stakeholders.

7.3 Lake Emma

7.3.1 Degradation

Lake Emma is reported to be polymictic (Bayer and Meredith 2020) – given its maximum depth is only 2.7 m, any stratification events are likely to be infrequent and short. The SOE monitoring data (Figure 7-6) shows there have been occasional events producing elevated chlorophyll-*a* concentrations. These chlorophyll-*a* spikes coincided with elevated TN and TP concentrations, suggesting either anoxic events may be mobilising DRP and fuelling cyanobacteria blooms (which fix atmospheric N), that wind resuspension may be mobilising DRP and DIN, or there were nutrient inputs from the catchment. Given the likely short duration of stratification events it is unlikely that anoxic conditions are created in the hypolimnion, but we have no data on the frequency and duration of stratification or on the rate of HOD.

Vegetation extended across most of the lakebed to a maximum depth of 2.1 m, limiting the extent of bare sediment that could readily be resuspended. Perch are present and may be contribute to poor water clarity through bioturbation, if present in high numbers (section 6.5.2).

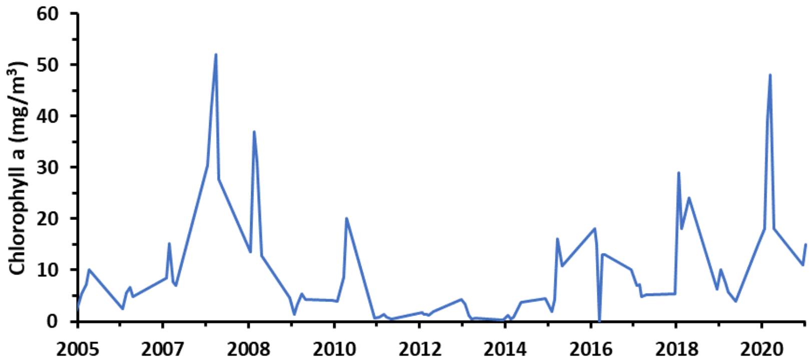

Figure 7-6: SOE time series chlorophyll-a data from Lake Emma.

7.3.2 Potential mitigations

A whole lake application of a granular P-inactivation agent such as Phoslock® or Aqual-P (with caveats, see 6.2.1) could provide some protection against sediment P release.

Stock should be excluded from the lake as they promote internal nutrient cycling by stirring up the sediment and can directly contribute nutrients as excreta.

7.3.3 Recommendations

Data focused on the frequency and duration of stratification events would enable a more accurate diagnosis of the causes of internal nutrient loading. If DO and temperature were recorded using high frequency sensors it may enable the calculation of HOD, enabling the duration of stratification required for anoxia to be predicted and an understanding of the frequency and duration of stratification events. Alternatively, sediment samples could be collected in summer to allow estimation of SOD (see Glossary for details).

Turbidity should be recorded continuously in the lake using turbidity sensors to identify if sediment resuspension is occurring during high wind events. This should include correlations with wind and rain events. The lake had reasonable water quality between 2011 and 2015 and the conditions that prevailed then need to be understood.

The SOE monitoring should include DRP and pH to enable the potential release of nutrients from beneath the macrophyte beds to be assessed. Additionally, taking targeted DRP samples from the deepest point, and an investigation of pH and DRP or DO during a prolonged still period in summer, and measuring the size of the legacy nutrient stores (total carbon, nitrogen and phosphorus) within the lake sediments are recommended (as detailed above, 7.1.3).

Conversations about perch, their impacts, consequences of their continued presence and management options should be initiated amongst stakeholders.

7.4 Lake Denny

7.4.1 Degradation

Bayer and Meredith (2020) report that Lake Denny is polymictic. However, with a max depth of 2.1 m any stratification events will be infrequent and short lived, suggesting that stratification-mediated anoxia in the hypolimnion is unlikely to be a mechanism contributing to internal nutrient loading. Denny is co-limited by N and P and this is evident as chlorophyll-a responds to increases in either TN or TP as shown in Bayer and Meredith (2020).

Catchment runoff will be contributing to the degrading water quality as a large plume of sediment and associated nutrients and organic matter was observed in the lake in March of 2018. This event may have imported P into the lake. Based on the available data it is difficult to assess the importance of the different mechanisms of internal nutrient cycling, but it is most likely that sediment resuspension and localised pH and DO mediated mobilisation events are contributing to nutrient availability. In addition, perch also contribute to poor water clarity through bioturbation when present in high numbers in shallow lakes (section 6.5.2).

7.4.2 Potential mitigations

Given that Lake Denny is co-limited by N and P it is important to control internal loading rates of both of these nutrients. Aqual-P may be a good option as it has a zeolite base that binds NH_4^+ as well as alum that binds DRP. The use of Aqual-P would reduce the nutrient loading associated with localised anoxic events and resuspension events, if they occur. The use of P-inactivation agent such as Phoslock® or Aqual-P has caveats (see 6.2.1).

7.4.3 Recommendations

Further information is required to establish the relative importance of potential mechanisms contributing to internal nutrient loading. It is recommended that an estimate of HOD is taken by measuring sediment oxygen demand (SOD).

The SOE variables measured should include pH and DRP. If possible, a bottom water sample for DRP, NH_4-N and NO_3-N analyses should be collected on occasions where calm weather has prevailed for several days before sampling. Turbidity should be continuously recorded in the lake using turbidity sensors to identify if sediment resuspension is occurring during high wind events.

Additionally, taking targeted DRP samples from the deepest point, and an investigation of pH and DRP or DO during a prolonged still period in summer, and measuring the size of the legacy nutrient stores (total carbon, nitrogen and phosphorus) within the lake sediments are recommended (as detailed in s 7.1.3).

Conversations about perch, their impacts, consequences of their continued presence and management options should be initiated amongst stakeholders.

7.5 Lake Heron

7.5.1 Degradation

Lake Heron is the deepest of the Ashburton Lakes, experiencing the longest periods of stratification (Figure 7-3) of any lake in this group, and is classified as seasonally stratified (Bayer and Meredith 2020). During the period of high frequency temperature and DO observation, the lake was stratified and there was a period of high wind that occurred in mid-December (2021) which partially mixed the lake. Despite long periods of stratification, the lake did not experience anoxia in its hypolimnion over the 43 days of monitoring. The lack of hypolimnion anoxia is likely due to the low oxygen demand within the lake. Its average rate of HOD was about 0.16 g O_2 $m^{-3}d^{-1}$ (Figure 7-7). At this rate, with an initial oxygen concentration of about 10 g m^{-3} it would take approximately 60 days of stratification before the hypolimnion became anoxic. The partial mixing event (20th December 2021) increased DO concentrations in the hypolimnion by approximately 5 mg/L. If this event had not occurred, anoxic conditions may have formed in the hypolimnion. This could greatly increase the internal loading of DRP in Lake Heron. We are unable to speculate on how often anoxic conditions form in Lake Heron's hypolimnion. Further investigation of the frequency of high wind events, similar to that which partially mixed the lake, will provide some insight into the likelihood of mixing events in the future relative to climate data.

Mixing was likely caused by a seiche that formed in the lake during the period of high winds. In such stratified conditions, the thermocline is likely to have tilted downwind and setup the lake for an internal seiche on the thermocline (Figure 7-9b). Because the wind event at that time was extreme (> 210 km/h), it was potentially strong enough, to push sufficient surface water into the deep basin to overwhelm the thermocline causing the tilt to reach the bottom, resulting in the lake mixing as shown (Figure 7-9 c). When the wind stopped, a barotropic seiche would have become active with a deeper return flow at the lakebed, disturbing the lake sediments along the full length of the shallow arm. The currents generated by the deeper return flow would likely be capable of sediment re-suspension when they are forced through the shallow waters outside of the lake's deeper basin. Without high frequency water level data from the ends of the lake and turbidity data, the effect of this potential mechanism of sediment re-suspension remains speculative.

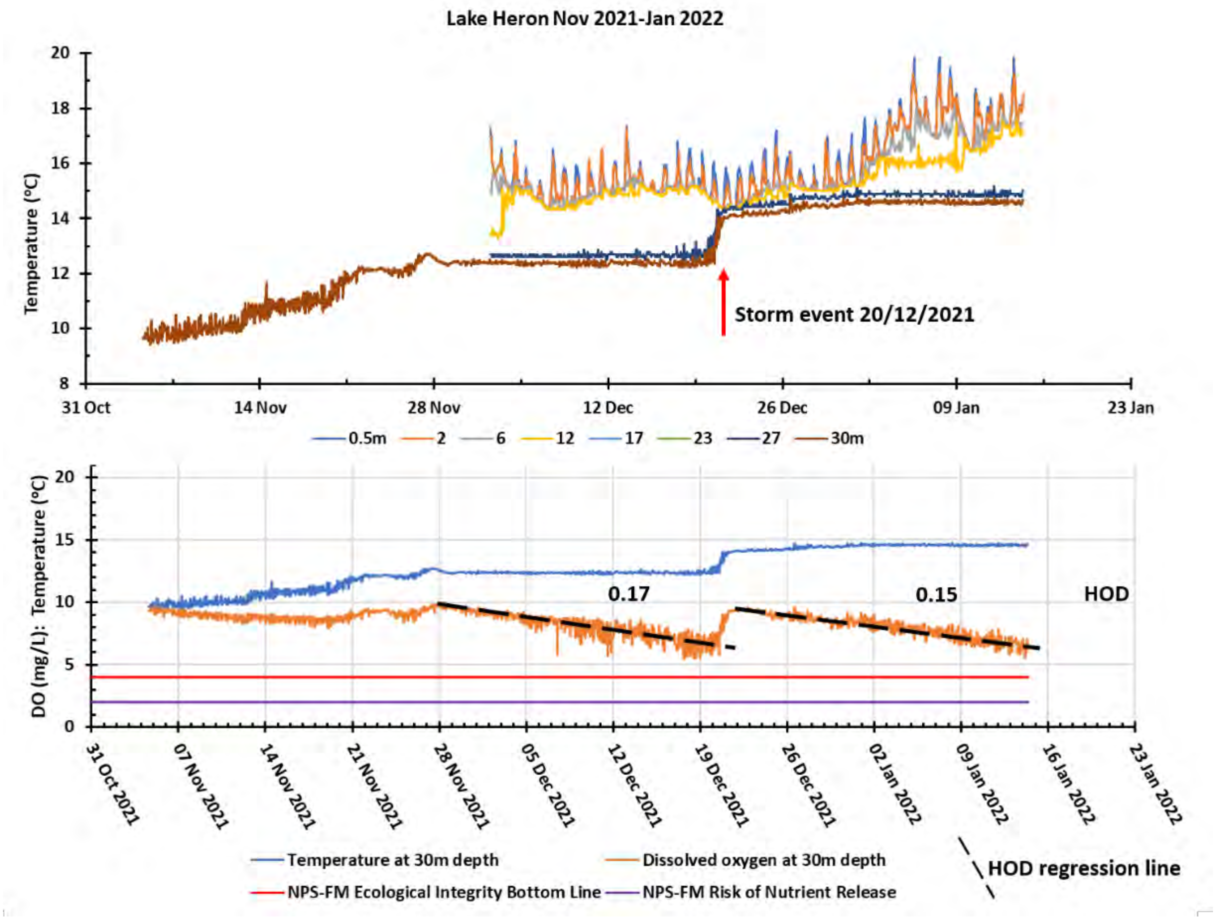

Figure 7-7: Lake Heron temperatures and bottom water dissolved oxygen (DO) at 32 m. Full depth lake mixing occurred when the bottom temperature rises rapidly to equal the surface temperature (upper graph). Regression lines were used to calculate hypolimnetic oxygen depletion (HOD) rates (mg of oxygen l^{-1} d^{-1}).

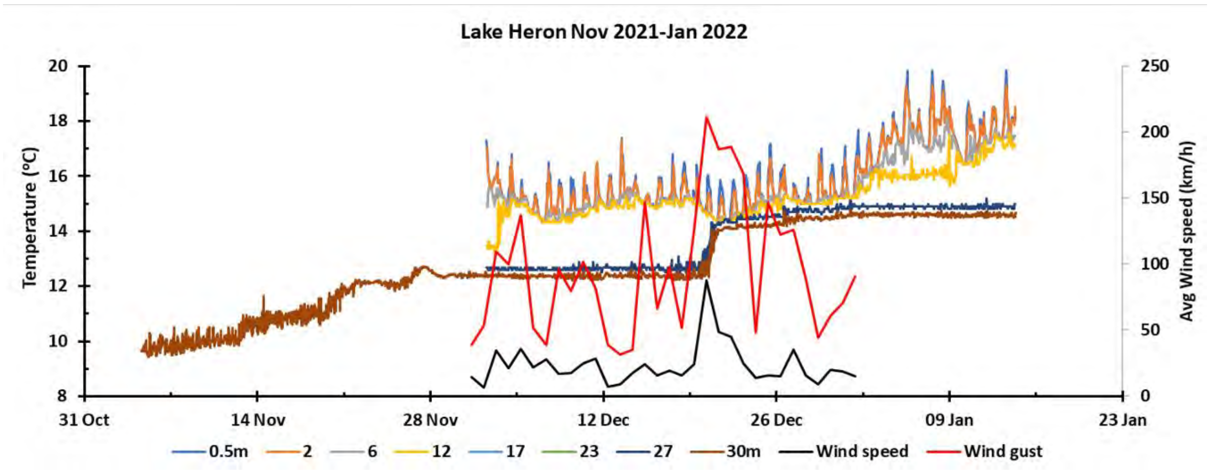

Figure 7-8: Daily average wind speed (black) and maximum wind speed of gusts (red) and thermal stratification in Lake Heron. Wind speed data from the Mount Potts Electronic Weather Station (EWS) and NIWA Virtual Climate Station Network (VCSN).

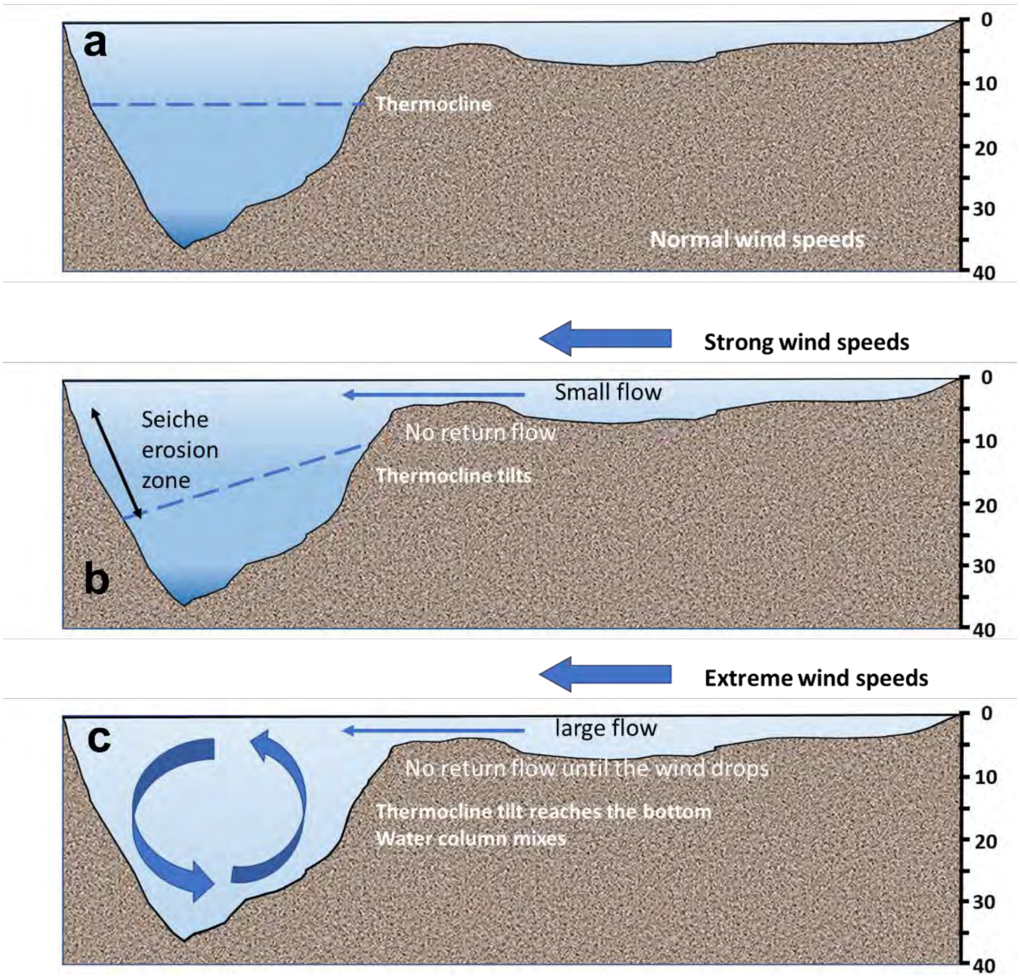

Figure 7-9: Stylised cross section of Lake Heron a) under calm conditions in summer (thermally stratified); b) under windy conditions from the northerly quarter; c) under extreme wind speed conditions.

7.5.2 Potential mitigations

Over the period that temperature and DO data was continually recorded in the lake, one mixing event that occurred during a period of high wind speeds. That the lake mixed to 32 m indicates that an internal seiche was formed which could cause sediment re-suspension during and after this event. Therefore, methods to increase sediment cohesion or protection from these currents may be appropriate. Such measures could take the form of wind breaks or flocculants. However, some of these measures (e.g., wind breaks) could lead to increased duration of stratification and potentially to sediment P release due to an anoxic hypolimnion. Coarse grained P inactivation agents such as Aqual-P or Phoslock® could provide protection against sediment P release (with caveats, see 6.2.1).

7.5.3 Recommendations

During the period that temperature and DO was continually recorded in the lake, no periods of anoxia in the hypolimnion were observed. Based on our estimated rate of HOD, the hypolimnion will become anoxic after ca. 55 days of stratification. In a warming climate, anoxia will become more likely. We recommend continuing the high frequency monitoring of temperature and DO in the lake to provide a better understanding of the role that stratification and hypolimnetic anoxia plays in determining lake water quality. Turbidity should be continuously recorded in the lake using turbidity sensors to identify whether sediment resuspension occurs during high wind events.

Sediment should be collected to investigate sediment bulk density and the concentrations of TP, TN and TOC to assess the size of the legacy nutrient pool within the Lake.

We also recommend targeted collection of samples from the deepest point for DRP analysis, and an investigation of pH and DRP or DO during a prolonged still period in summer are recommended (as detailed above, 7.1.3).

7.6 Lake Emily

7.6.1 Degradation

Lake Emily is shallow with a maximum depth of 2.3 m. It was reported by Bayer and Meredith (2020) as being polymictic – given its shallow depth, any stratification is likely to be short lived. Therefore, unless the HOD is extremely high it is unlikely that anoxia-mediated DRP releases contribute to internal nutrient loading in this lake. Wind mediated sediment resuspension could explain concurrent peaks in TN, TP, chlorophyll-a and turbidity observed within (shown in graphs of these variables presented in Bayer and Meredith (2020)), as could nutrient inputs from the catchment.

7.6.2 Potential mitigations

A whole-lake application of a granular P-inactivation agent such as Phoslock® or Aqual-P could provide some protection against sediment resuspension, as well as sequester mobilised DRP (and NH_4^+ in the case of Aqual-P) (with caveats, see 6.2.1).

7.6.3 Recommendations

Further investigation and monitoring is required in Lake Emily to gain a better understanding of the size of the legacy nutrient pool within the lake sediment and the mechanisms that are potentially contributing to its mobilisation. The collection of intact sediment cores and their analysis of bulk density, TP, TN and TOC contents will provide the concentration and areal content of sediment nutrients with the lake. This data would inform any subsequent P inactivation agent application.

Further, investigating the rate of SOD (as detailed above, 7.1.3) would aid in assessing the duration of stratification required for the hypolimnion to become anoxic. Turbidity should be continuously recorded in the lake using turbidity sensors to identify whether sediment resuspension is occurring during high wind events. Additionally, an investigation of pH and DRP or DO during a prolonged still period in summer is recommended (as detailed above, 7.1.3).

7.7 Māori East Lake, Ōtūwharekai (also Front and A)

7.7.1 Degradation

Māori East Lake is reported as being polymictic (Bayer and Meredith 2020), which seems likely given its shallow depth of 1.2 m. Stratification is likely to be of short duration, suggesting that stratification mediated hypolimnion anoxia is unlikely to be an important cause of internal nutrient cycling. Overall, the SOE data show good correlations between TN, TP, turbidity and chlorophyll-a. However, Bayer and Meredith (2020) note large fluctuations in TLI, primarily due to peaks in TP and phytoplankton biomass. In 2017 there was a large increase in both TP and chlorophyll-a, suggesting a mechanism of internal loading that only mobilised DRP, such as pH- or anoxia-mediated release. At other times, there have been concurrent increases in TN, TP, chlorophyll-a and turbidity suggesting nutrient inputs from the catchment.

7.7.2 Potential mitigations

The addition of a P-deactivation agent such as alum, Aqual-P or Phoslock® would provide protection against internal P loading (with caveats, see 6.2.1), likely limiting algal growth and reducing chlorophyll-a concentrations.

The high flushing rate of Māori East Lake suggests that ecological health should improve relatively rapidly under lower rates of external nutrient loading.

Stock should be excluded from the lake as they promote internal nutrient cycling by stirring up the sediment, while also contributing nutrients to the lake through excreta.

7.7.3 Recommendations

An assessment of the size of the legacy nutrient pool (bulk density and the concentrations of TP, TN and TOC), its mobility during sediment re-suspension events and the effectiveness of different P-inactivation agents during re-suspension events is recommended (see Lake Clearwater for more details) as is an assessment of SOD. Turbidity should be continuously recorded in the lake using turbidity sensors to identify if sediment resuspension is occurring during high wind events.

We recommend the addition of DRP and pH to SOE monitoring – their inclusion will enable assessment of the role of pH-mediated DRP releases.

An investigation of pH and DRP or DO during a prolonged still period in summer, is also recommended (as detailed above, 7.1.3).

7.8 Māori West Lake, Ōtūwharekai (also Back and B)

7.8.1 Degradation

Māori West Lake was reported to be up to 2.6 m deep (de Winton and Burton 2017), and polymictic (Bayer and Meredith 2020). At these depths, stratification in the lake is likely to be infrequent and brief, which makes anoxia in the hypolimnion unlikely. However, this assessment is speculative – no data exist to provide insight into stratification or HOD/SOD in the lake. Similarly, wind-driven resuspension of sediment is a potential source of nutrients, however this seems unlikely given the entire lakebed was covered in vegetation in 2017 (de Winton and Burton 2017). Nutrient inputs alone may be driving the high TLI.

7.8.2 Potential mitigations

The addition of a P-inactivation agent (with caveats, see 6.2.1) would likely increase the health of the lake by limiting the availability of phosphorus, thereby minimising phytoplanktonic growth.

7.8.3 Recommendations

Further monitoring is recommended to increase our understanding of the mechanisms that contribute to internal nutrient loading rates, including:

- assessment of sediment bulk density and the size of the legacy nutrient pool (TN, TP and TOC),
- assessment of sediment oxygen demand to better our assessment of the likelihood of anoxia occurring during periods of stratification (as detailed above, 7.1.3),
- continuously recording turbidity to identify if sediment resuspension is occurring during high wind events,
- addition of DRP and pH to the SOE monitoring, to enable assessment of the role of pH-mediated DRP releases, and
- targeted collection of samples for DRP analysis, and an investigation of pH and DRP or DO during a prolonged still period in summer (as detailed above, 7.1.3).

8 Summary and recommended actions

The Ōtūwharekai Action Plan is intended to be a structured and well-evidenced programme of work to stop further degradation of the lakes and wetlands of the Ōtūwharekai catchment that delivers a pathway to restoration.

The in-lake mitigations component (Part 2, this report):

- identifies which lakes may require in-lake mitigation,
    - by examining lake condition and restoration goals,
- assesses feasible mitigation options on a lake-by-lake basis,
    - identifying likely causes of degradation, and priorities for mitigation (ecological values and pressures),
    - reviews the available options and quantifies expected lake health outcomes, and
- recommends a course of action for each lake likely to halt the degradation, and deliver a pathway to restoration.

Targets have independently been set for TLI and LakeSPI for the lakes. The most recent data indicates that all lakes currently fail to meet those targets, with the exception of Lake Camp, which currently meets its LakeSPI target. There has been a steady decline in water quality (TLI) since annual seasonal monitoring began (see Bayer and Meredith 2020).

Improving lake water quality while native plants are still present provides the best chance for improvement of lake ecological condition. Once the native macrophytes are lost (an inevitable consequence of continued water decline), water clarity will decrease (impacting adversely on light climate), making recovery of native vegetation more difficult. An increase in the number of invasive species that establish in the lakes, or an increase in their proportion of macrophytes will also make lake restoration more costly and challenging, and the probability of success will diminish significantly.

Most of the Ōtūwharekai lakes have scored high, or high to moderate, for their ecological values, the exception being Lake Denny (moderate to low), with pressures and threats in the high to moderate category. These scores indicate that although most of the lakes still have high ecological values, they are also at risk (e.g., Lake Heron scores high for both categories). These circumstances support the requirement for urgent implementation of mitigation action to restore and protect all of the lakes.

However, by using the combined scores for ecological value and risk, a mechanism exists whereby the lakes may be ranked from highest to lowest as follows:

Heron > Māori East > Emily = Clearwater > Māori West > Camp = Emma > Denny

This (or an alternate) ranking can be used to prioritise actions, for example by focussing mitigation actions on the highest ranked lakes first to protect them from further decline.

Several approaches have been identified and discussed to improve water quality, including the use of P-inactivation agents, flocculants and biosecurity planning. These in-lake interventions provide ‘ambulance at the bottom of the cliff’ solution scenarios – they may go some way to generating short-term improvements, while incurring on-going management costs. They do not however address the on-going inputs of nutrients from external sources. Reducing external nutrient inputs to the lakes would allow natural flushing of nutrients with outflowing lake water to take place over time.

The efficacy of the mitigation options identified and described earlier cannot be assured, because limited data exist to establish with certainty the in-lake causes of degradation for each lake. Recommendations for targeted monitoring and data collection will provide additional information to reduce this uncertainty; once this information is available, identification of the most useful mitigation options will be likely and assessment of the feasibility of in-lake mitigations could be improved.

8.1 Recommended actions

1. Turn off the tap - reduce catchment contaminant loads (see Kelly et al. 2021; Part 3 report). Significant reduction of nutrient and sediment inputs to all lakes is essential. This is particularly important given that we cannot change the climate change trajectory – the predicted increased frequency of extreme weather events (e.g., strong winds) and increasing temperatures will continue, and negatively impact the ecology of the lakes. Coupling these climate factors with unchecked nutrient loads, may lead to exceedance of a threshold where the cumulative stressors are too great for macrophytes to persist. Arguably some lakes are already at that threshold. Should the native plants be lost, the opportunities to restore the lakes will be greatly reduced (relative to potential from the current state). The only sustainable "lever" currently available to improve lake condition is reduction of the nutrient and sediment loads delivered from the respective lake catchments.

2. Manage in-lake nutrient issues where they exist (e.g., with sediment capping/P inactivation). Without also reducing ongoing input of phosphorus to the lakes, this approach is likely to largely represent a "band-aid" approach. Reapplication will be necessary over time, leading to ongoing expense. In-lake P concentration reductions may temporarily be achieved by application of P-inactivation agents, but without catchment interventions, the P content will be replenished with the sediment transported into the lake in subsequent rainstorms. However, once external loads are reduced, the potential may exist for internal load events to occur, e.g., should low dissolved oxygen concentration conditions occur. Sediment capping agents may provide protection against these events, or reduce the ecological effects of these internal load events.

   There are caveats to the use of P inactivation agents that must be considered (and in some cases resolved), before they are used; these include: adequate understanding of the potential for off-target impacts on macrophytes, and understanding whether the presence of plants will limit ability to effectively place these products on the sediment.

   The TN content in lake water may be reduced through uptake by macrophytes (but much of this nitrogen may be returned to lake following plant senescence and decay). Effective management and protection of the macrophytes in these lakes will be key. That will require reduction of sediment inputs from the lake catchments and internal and external nutrient loading. Denitrification will also reduce the nitrate-N fraction of TN in lake waters (converting nitrate-N to N_2, which is subsequently lost to the atmosphere), and capping agents such as Aqual-P are able to bind ammoniacal-N.

3. Prevent any new incursions of alien invasive species (AIS). Continue the ECan surveillance programme and design a targeted community awareness campaign for the Ōtūwharekai lakes, focusing on prevention of new incursions of invasive species, and reducing the risk of spreading non-native fish and plants that are currently present in some of the lakes to other lakes.

   Investigate the appetite/will amongst mana whenua, mandated agencies with management responsibility, and other stakeholders, to support a programme for the removal of perch.

4. Continue to monitor the lakes (as in Bayer and Meredith 2020), with additional monitoring as recommended in Section 8.2.

8.2 Monitoring recommendations

Further monitoring is recommended to increase our understanding of the mechanisms that are contributing to internal nutrient loading. The additional monitoring is prioritised below, with those items at the top considered as higher priority than those at the bottom of the list. Unless otherwise stated, this list applies to all lakes.

1. Measure the sediment bulk density and sediment TN, TP and TOC to understand the magnitude of the legacy nutrient content of each lake, and when necessary calculate an appropriate P inactivation agent dose.

2. Continuously record turbidity to identify if/when sediment resuspension is occurring. Turbidity loggers provide a relatively easy and cheap way to monitor turbidity ("cloudiness" of water) and enable correlation with weather events to understand what wind speeds cause sediment resuspension, and to determine how frequently these resuspension events occur.

3. Continuous monitoring of DO and temperature at multiple depths to understand the frequency of stratification and anoxia events. These monitoring actions should be prioritized for the deeper lakes (Heron, Clearwater and Camp), where such monitoring has already been undertaken, and where stratification is likely to be more prolonged, and likely to have an impact on the timing and magnitude of the internal P load.

4. Assessment of sediment oxygen demand (SOD) to better understand the likelihood of anoxia occurring during periods of stratification. SOD is the major contributor to hypolimnetic demand (HOD); it could be quantified in the laboratory to gain a better understanding of the likelihood of stratification-induced anoxia, instead of requiring continuous in-lake DO monitoring. The latter approach is preferred, because it provides a direct measurement of DO, but is likely to be considerably more expensive.

5. Addition of DRP and pH to the suite of water quality variables currently measured in the SOE monitoring programme. These data will enable assessment of the role and importance of pH-mediated DRP releases. Selection of the most effective P inactivation agent will also require pH data. The DRP samples should be taken from both the surface and from the deepest part of each lake to help determine whether anoxia is driving internal P loading during periods of stratification.

6. An investigation of pH and DRP or DO during a prolonged still period in summer within and outside of macrophyte beds.

7. Preparation or refinement of lake nutrient budgets would be helpful for understanding the health trajectory of the lakes and to assess the likely impact that any potential intervention might have on lake condition.

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10 Glossary of abbreviations and terms

DRP	Dissolved reactive phosphorus. The dominant soluble and bioavailable form of phosphorus
LakeSPI	Lake Submerged plant indicators. A biomonitoring method used to assess the ecological condition of lakes
SOD	Sediment oxygen demand. SOD can be measured by placing a known mass of sediment into an air-tight container and filling the container with lake water so there is no gaseous head space. The demand for oxygen is then measured by monitoring the decline in oxygen concentration through time. Sediment bulk density can then be used to convert the mass of sediment into an areal measure of oxygen demand.
TLI	Trophic level index
TN	Total nitrogen
TOC	Total organic carbon
TP	Total phosphorus

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Appendix A Lake location map

Figure A-1: Location of eight Ōtūwharekai lakes. Source Burton et al. 2022.

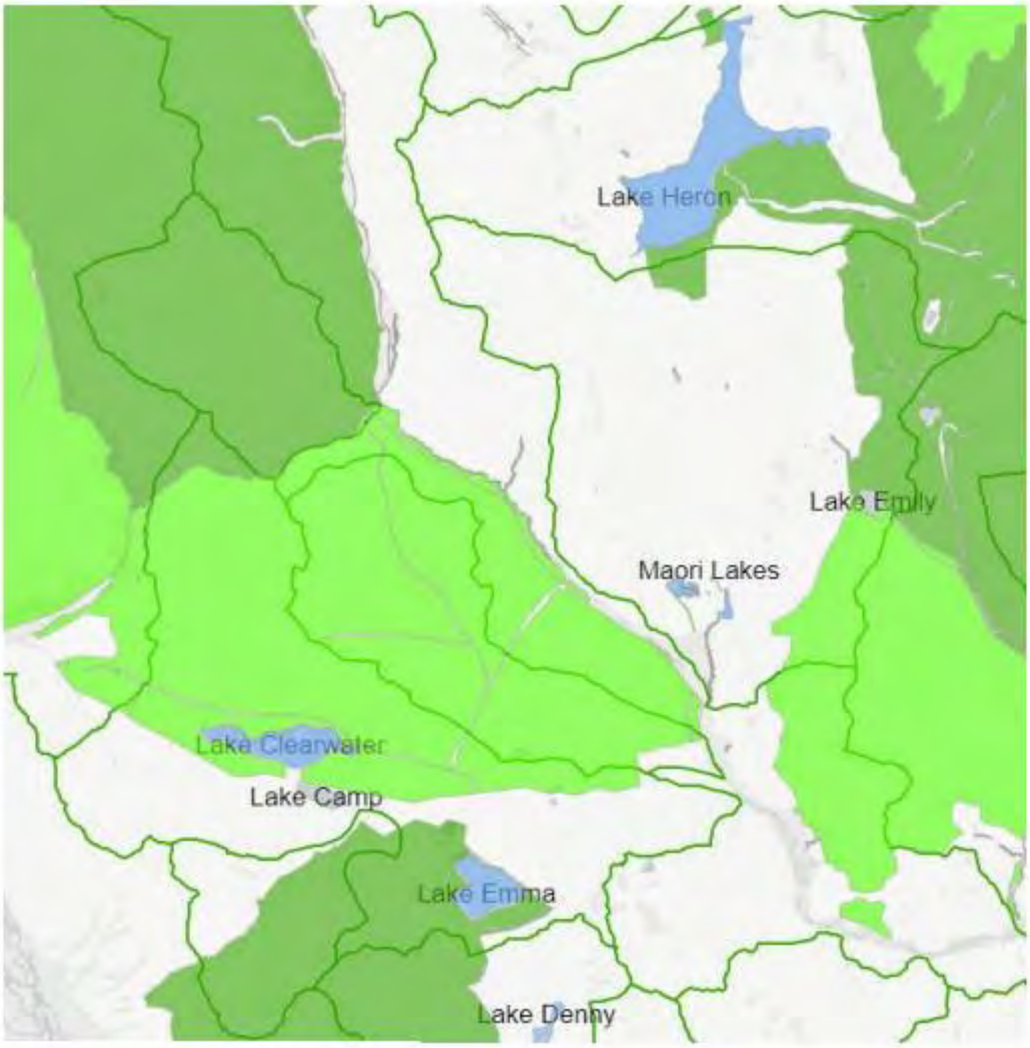

Figure A-2: Map showing public conservation land in the Ashburton Lakes Basin. Source: Bayer and Meredith 2020. Pale green are Stewardship Areas, and dark green represents Conservation Park.

Appendix B Lake catchment landcover maps

Maps are based on 2018 landcover classes

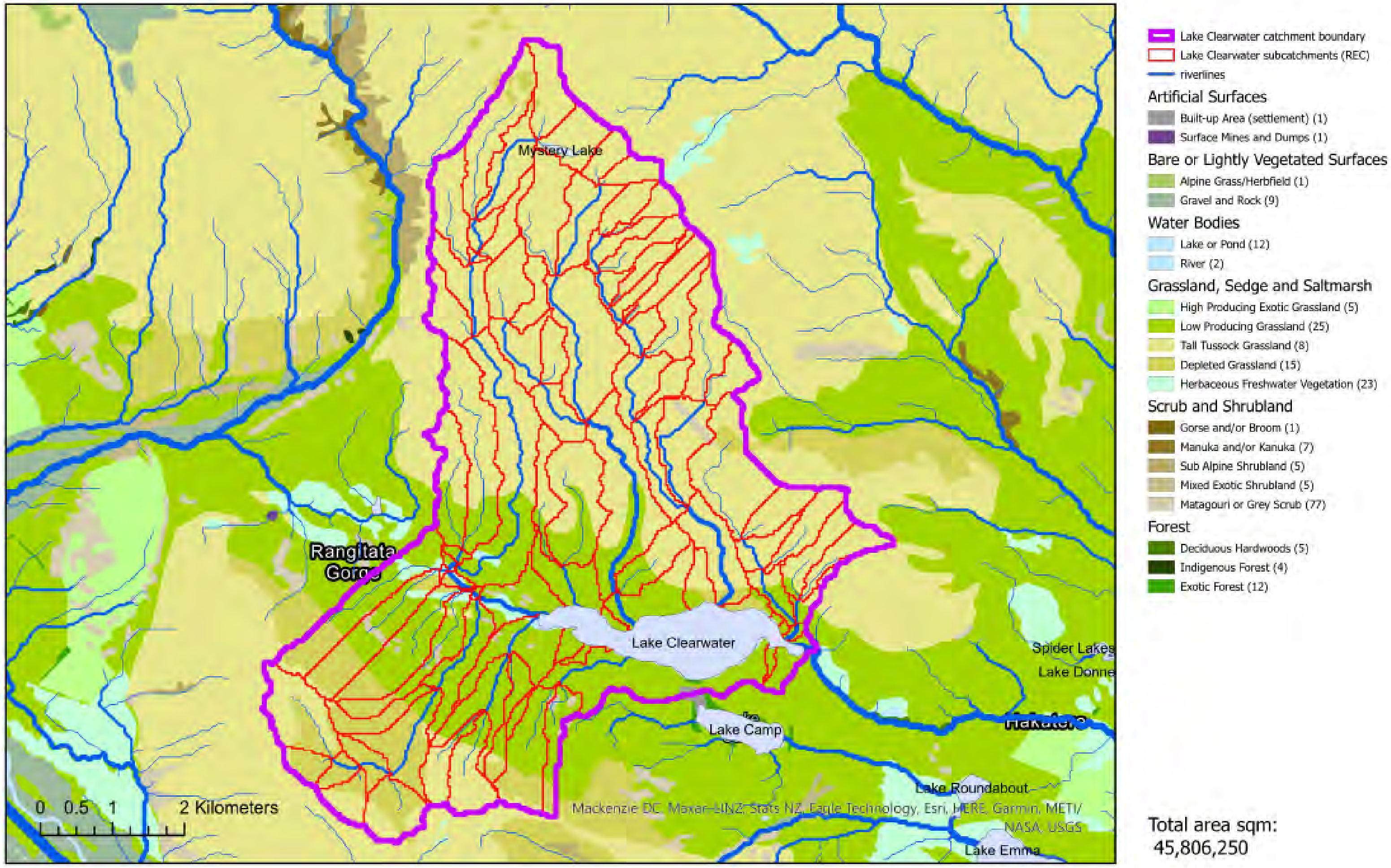

Figure B-1: Catchment map for Lake Clearwater.

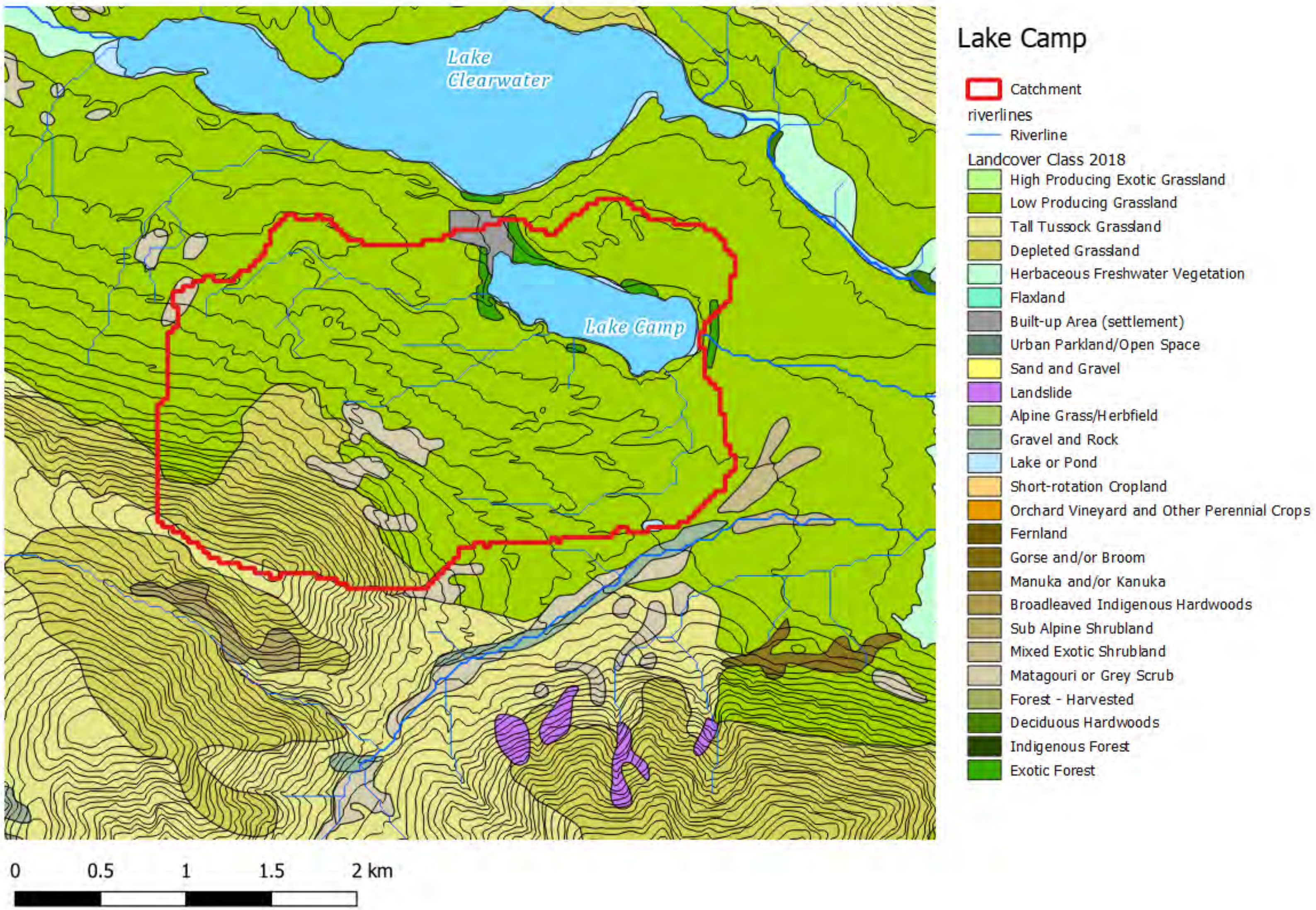

Figure B-2: Catchment map for Lake Camp.

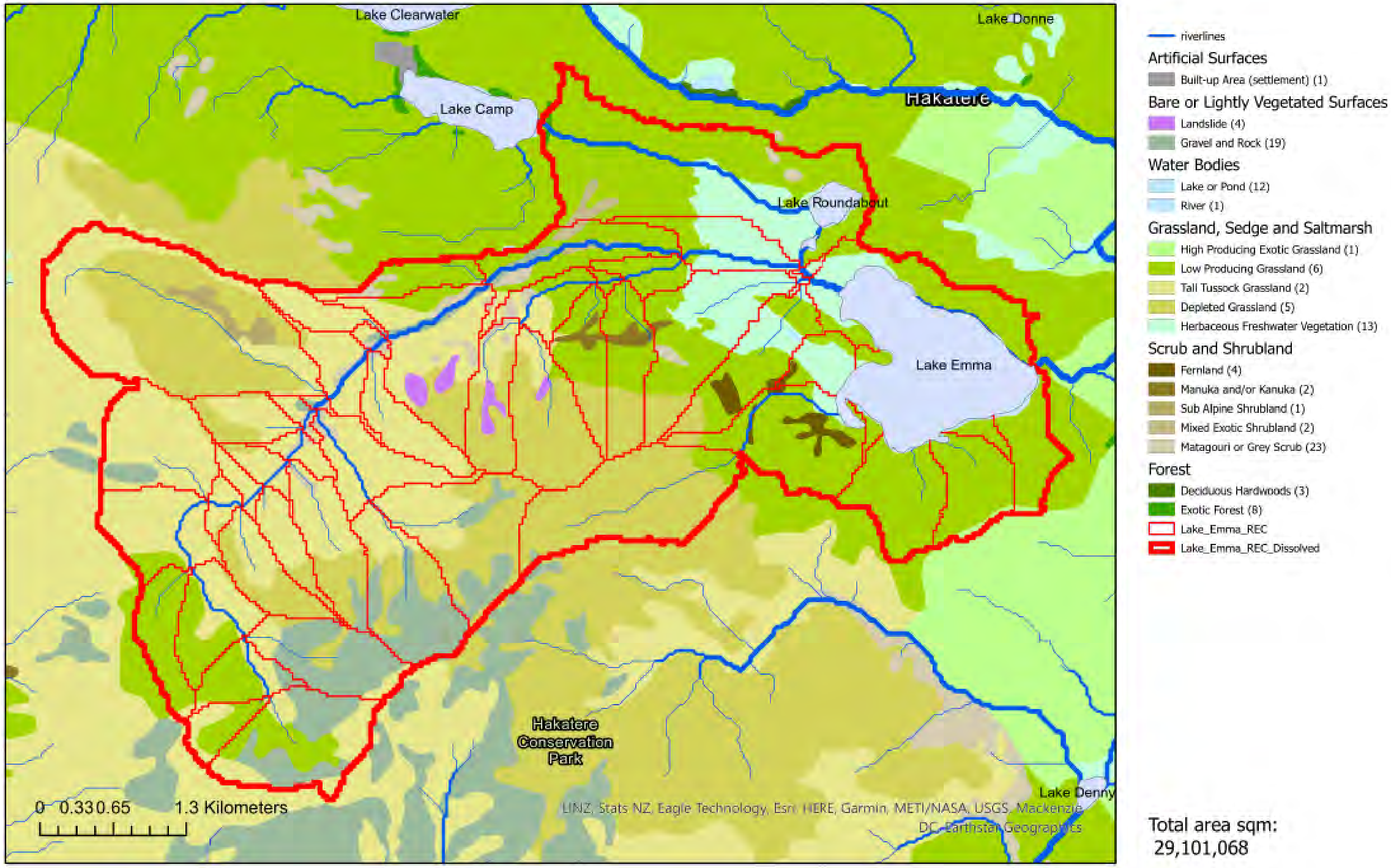

Figure B-3: Catchment map for Lake Emma.

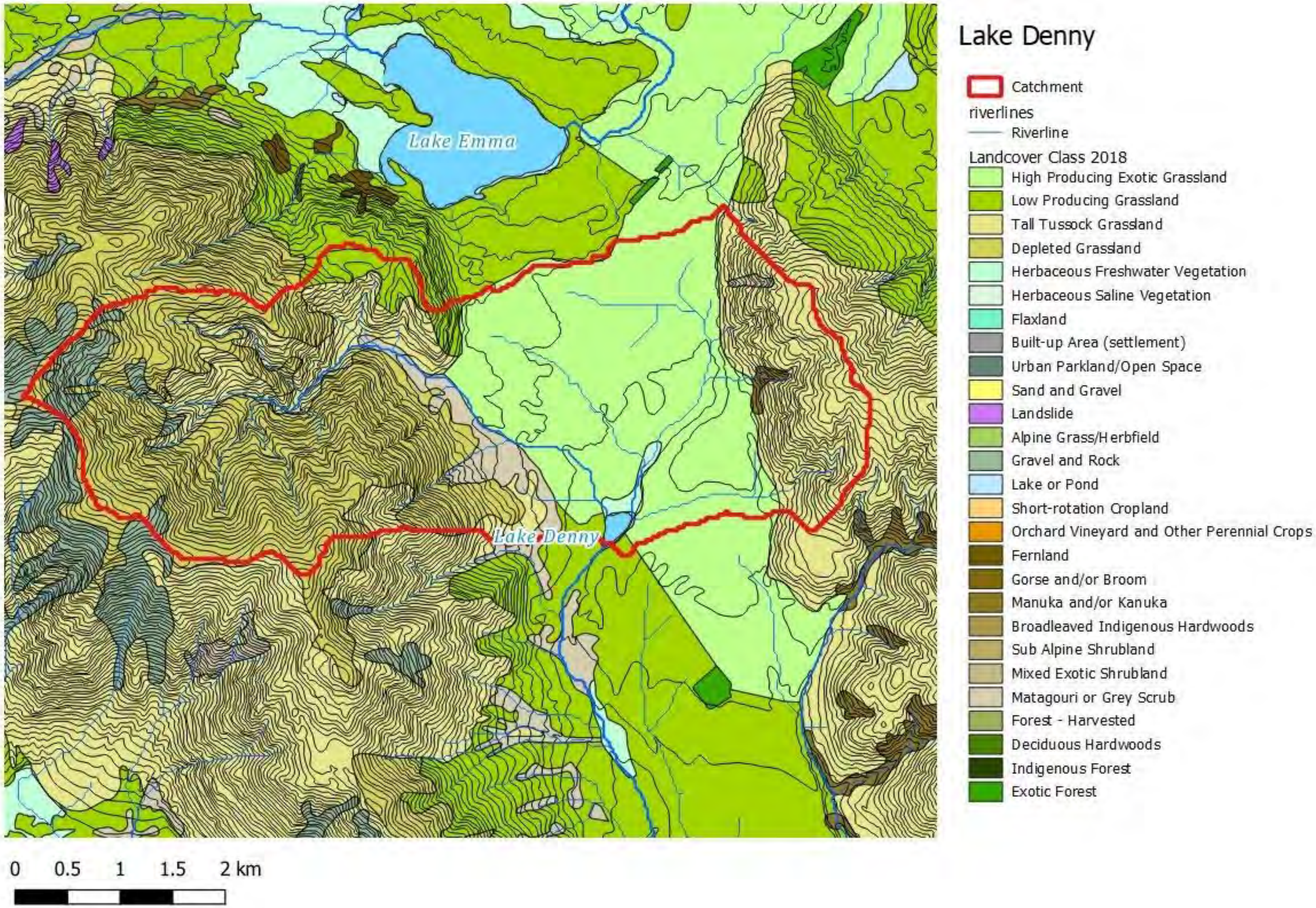

Figure B-4: Catchment map for Lake Denny.

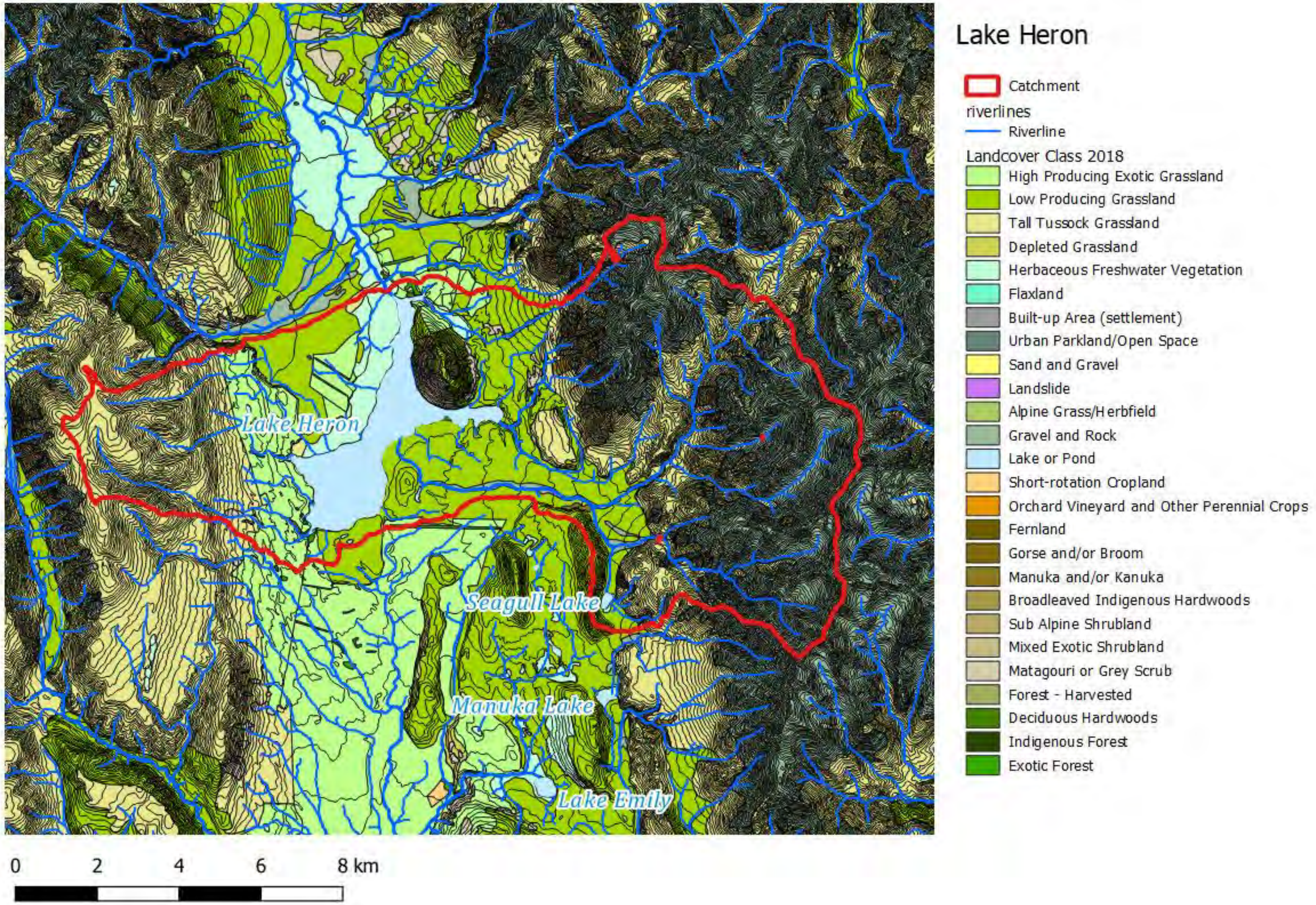

Figure B-5: Catchment map for Lake Heron.

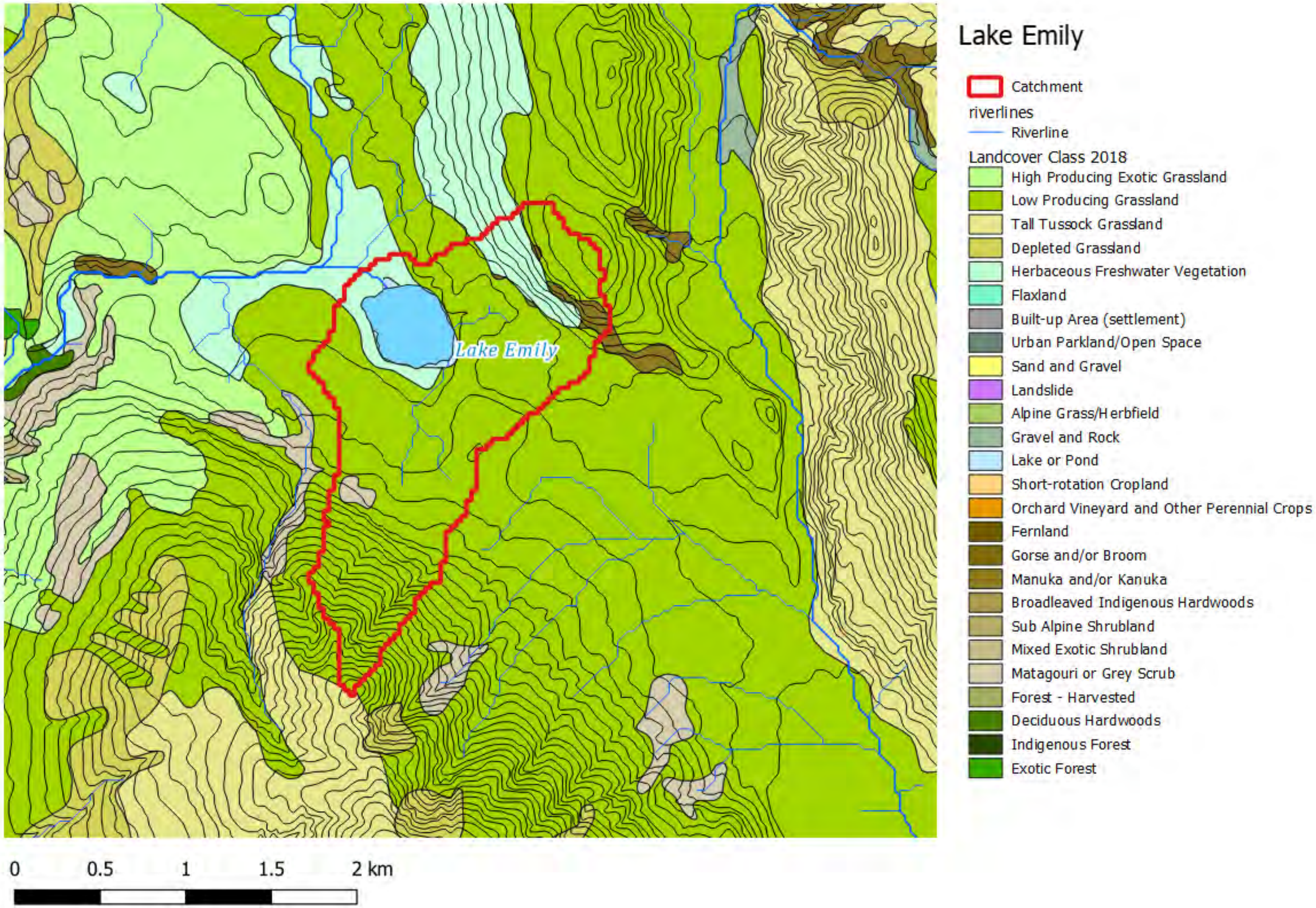

Figure B-6: Catchment map for Lake Emily.

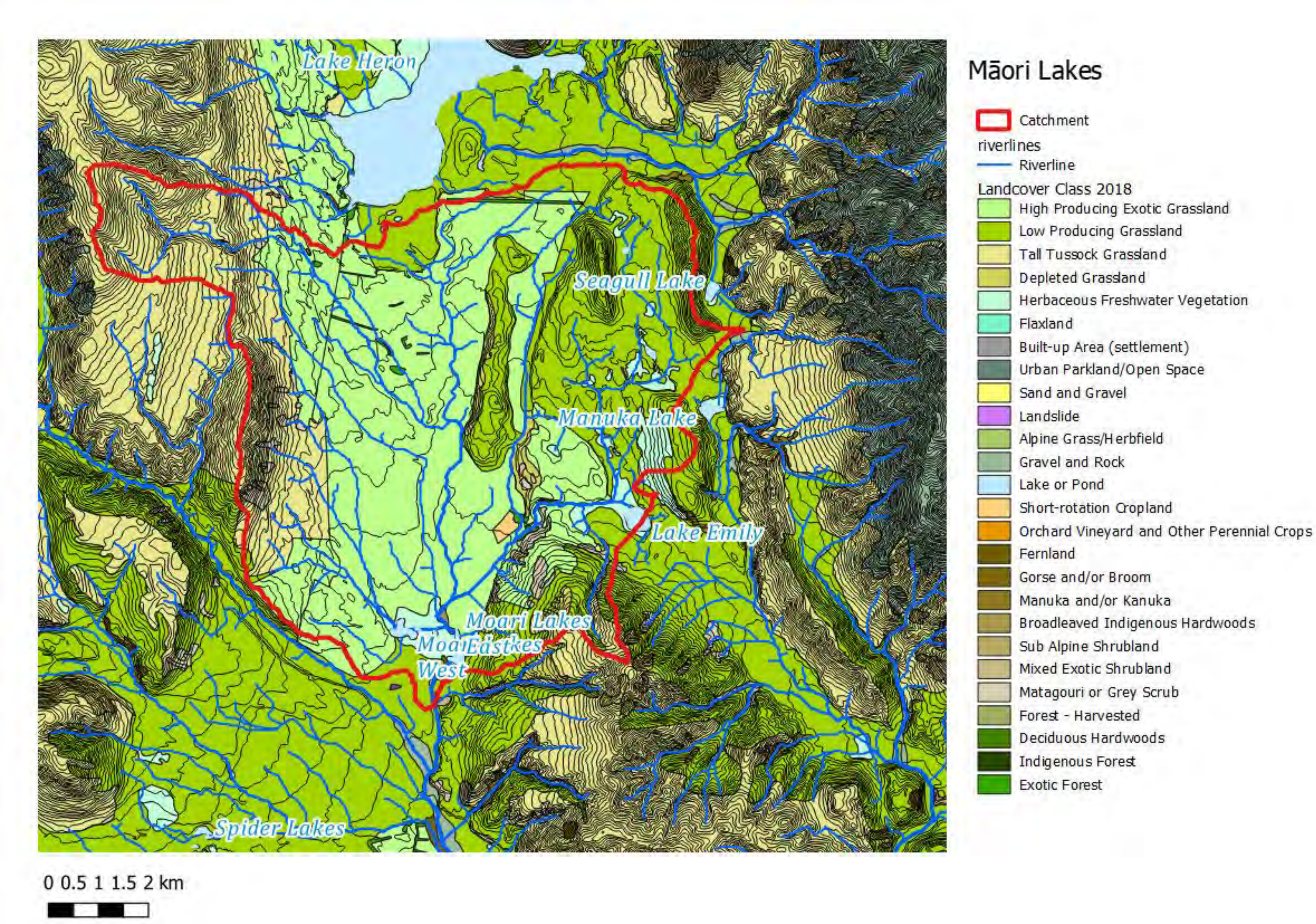

Figure B-7: Catchment map for Māori Lakes East and West.

Appendix C High naturalness waterbodies

Table C-1: High naturalness waterbodies. Source CLWRP (2019)

Description of High Naturalness Waterbodies from the CLWRP (table 13, pg 314) Lake Emily. Outstanding natural features and landscapes that include a regionally significant wetland complex. Habitat of threatened/endangered indigenous birds and freshwater species including eel and freshwater mussel. High visual amenity value. Māori Lakes. Outstanding natural features and landscapes. Habitat of threatened/endangered indigenous birds, including crested grebe and Australasian bittern. Inflows and outflows high habitat value for maintaining longfinned and shortfinned eel and galaxidae and the sport fish, brown trout. Outflows high habitat value for Chinook salmon spawning and freshwater mussels. High visual amenity value. Lake Clearwater. Outstanding natural features and landscapes including a regionally significant red tussock wetland. Habitat of threatened/endangered indigenous birds and Recommended Area for Protection. High habitat value for indigenous fish such as galaxidae, eel, freshwater mussels and the sport fish brown trout. High visual amenity value. Lake Camp. Outstanding natural features and landscapes. High habitat value for longfinned eel, freshwater mussel, crested grebe and the sports fish rainbow trout Lake Emma. Outstanding natural features and landscapes including pedestal *Carex secta* and *Schoenus* wetlands. Habitat of threatened/endangered indigenous birds. High habitat value for indigenous fish such as galaxidae, eel, freshwater mussels and the sports fish brown trout. High visual amenity value.

Appendix D

Recommendations for future management sourced directly from Te Rūnanga o Arowhenua et al. (2010)

1. That all waterways which continue to be important for food gathering are managed and enhanced for food gathering quality into the future. 2. That increased protection and enhancement of waterways through the development of native riparian and wetland buffer zones be investigated and implemented. 3. Greater advocacy, rates relief and other economic methods for the protection and enhancement of native riparian and wetland buffer zones and vegetation patches in currently poor or un-vegetated or un-fenced areas on private land. 4. Specific restoration, pest and weed eradication and exotic species control in and around all lakes, including the use of tī kouka, houhi, kowhai, maukoro, mikimiki, beech and aruhe and other native plants that prove to compete well with, or can be planted underneath willow and other exotic species invading lakes and wetlands. This should consider the removal of pest fish from specific areas. The following lakes and sites should be a priority: - Ōtūwharekai (East) / Lower Māori Lake - Kirihonuhonu / Lake Emma - The Oliver Stream area of Ō Tū Roto / Lake Heron - The Swin river access area of Ō Tū Roto / Lake Heron - Te Puna a Taka / Lake Clearwater; and - Ōtautari / Lake Camp. 5. Specific measures to control siltation/sedimentation and E. coli contamination of Ōtūwharekai (East) / Lower Māori Lake and further protection of Ōtūwharekai (West) / Upper Māori Lake, including the potential purchase of surrounding land, the control of exotic species, and the development of better buffers, particularly around the road edge corner of Lower Māori Lake and incoming water ways of both lakes. 6. Consideration for the complete and ongoing removal of exotic fish from the Māori Lakes and work towards making the lake complex a native fish only area. 7. Further tuna/eel monitoring surveys and investigation to understand the potential of an annual cultural harvest, particularly at the Māori Lakes. 8. Further investigation and control of human and agricultural pollution at Te Puna a Taka / Lake Clearwater and the Oliver Stream area of Ō Tū Roto / Lake Heron. 9. Support for future wānanga and hui to reconnect tangata whenua with the Ōtūwharekai / Ashburton Lakes Area, particularly around future interpretation and cultural harvest opportunities of both tuna, raupō (for mokihi), hua kaki anau (black swan eggs) and other mahinga kai. 10. Investigation into the habitat requirements of, and future possibilities (including specific sites), for the reintroduction of eastern buff weka into the area. 11. Greater research into the impacts of, and solutions for, treating and dealing with non-point and point source pollution of waterways in the area. 12. Continued regular monitoring, including cultural assessments, to understand the success, or otherwise, of future management and development of the catchment

Appendix E Lake ecological value assessment

Ecological values assessment	Clearwater	Camp	Emma	Denny	Heron	Emily	Māori east	Māori west
VALUE RATING	**high**	**high**	**high-moderate**	**moderate to low**	**very high**	**high-moderate**	**high-moderate**	**high-moderate**
SCORE TOTAL <4 is 'low'; 4 to 6 is 'moderate to low'; 7 to 8 is 'moderate'; 9 to 10 'high to moderate'; 11 to 12 is 'high'; 13 to 20 is 'very high'	**12.5**	**11**	**9**	**6**	**14**	**9.5**	**10**	**10**
Habitat Area. Area (ha); depth (m). Scores assigned for area were: >100ha is 3; 10-100ha is 2; <10ha is 1; and 1ha or less is rank 0; and for the depth as: >25m is 3, 10-25m is 2, 10-2m is 1. and less than 2m is 0.	197ha, 19m, (3,2)	45ha, 18.9m, (2, 2)	160ha, 2.7m (3,1)	5ha, 1.8m (1,0)	695ha, 36.2m (3,3)	19ha, 2.3m (2,1)	9ha, 1.3m (1,0)	10ha, 1.8m (2,0)
SCORE	*2.5*	*2*	*2*	*1*	*3*	*1.5*	*1*	*1*
Buffering. Native vegetation cover % in the catchment immediately adjacent to lake; Wetland extent as a % of lake area; Emergent extent as a % of lake perimeter (emergent bands >20m).	Unfarmed tussock est. at 90%, adjacent land classed as low productive, now in DOC estate (1); Wetlands ca 20% of lake area (2); Emergent little to none (0); (1.2.0)	Mostly low productive grassland, ca. half now in DOC estate (0); Wetland % (0); Perimeter emergent % (0); (0.0.0)	Mainly productive grassland adjacent to lake (0); Wetlands present and reserved ca. 80% (2); Emergent perimeter ca. 25% (1); (0.2.1)	High productive grassland (0); Wetland on N. shore and W. of the lake ca. 66% perimeter and same area as lake (2); Emergent perimeter 66% (2); (0.2.2)	Large areas of productive land adjacent to lake 75% (native beyond the productive zone) (1); Wetland extent ca. 33% of lake area (2); Emergent extent % of perimeter (0); (1.2.0)	Productive grassland and conservation estate (1); Large wetland margin linked to 50ha swamp (3); Ca. 75% perimeter has emergent (2); (1.3.2)	Productive grassland beyond the wetland (0); Wetland complex (3); Wetland margin (3); (0.3.3)	Productive grassland beyond the wetland (0); Wetland complex 100ha (3); Wetland margin (3); (0.3.3)
SCORE	*1*	*0*	*1*	*1*	*1*	*2*	*2*	*2*
Water quality. TLI. [Data supplied by Ecan (in Ecan.xlsx)]	eutrophic	mesotrophic	eutrophic	supertrophic	mesotrophic	eutrophic	eutrophic	eutrophic
SCORE	*1*	*2*	*1*	*0*	*2*	*1*	*1*	*1*

Ecological values assessment	Clearwater	Camp	Emma	Denny	Heron	Emily	Māori east	Māori west
Diversity. Total number of indigenous species is scored as: >20 species – 3; 15-20 species – 2; 5-14 species – 1; <5 species – 0.	16	14	11	2	20	10	6	6
SCORE	*2*	*1*	*1*	*0*	*2*	*1*	*1*	*1*
Integrity. NCI scores for lakes were classed as >75% – Rank 3; >50-75% – Rank 2; >20-50% or 1-20%– Rank 1; 0%– Rank 0.	54	66	38	9	49	25	50	43
SCORE	*2*	*2*	*1*	*1*	*1*	*1*	*1*	*1*
Endangered species. Each Nationally Threatened taxa scored 5, declining species 2 and other At-Risk and new to New Zealand species. 1. These were summed and lakes with an endangered species score >15 becomes 3; 5-15 becomes 2; <5 >0 becomes 1 and 0 when no endangered taxa were recorded.	Longfin tuna present in low abundance (2); *Gobiomorphus hubbsi* (2); kākahi present (2); *Triglochin palustre* (5); score of 11	Longfin tuna present in low abundance (2); *Galaxias brevipinnis* (2); kākahi (2); *Isolepis basilaris* (2); *Carex decurata* (2); score of 10	Longfin tuna present in low abundance (2); kākahi present (2); score of 4	kākahi present (2)*;* score of 2	Longfin tuna present (2); *Galaxias brevipinnis* (2); Kākahi present (2); *Ranunculus brevis* (5); *Carex cirrhosa* (5); score of 16	kākahi present (2)*;* score of 2	Longfin tuna present (2); *Galaxias brevipinnis* (2); *Galaxias vulgaris* (2); kākahi present (2); *Ranunculus macropus* (5); score of 13	Longfin tuna present (2); *Galaxias brevipinnis* (2); *Galaxias vulgaris* (2); kākahi present (2); *Ranunculus macropus* (5); score of 13
SCORE	*2*	*2*	*1*	*1*	*3*	*1*	*2*	*2*
Key Species. Presence of living kākahi/mussels adds an additional point to the score.	Present, deformed soft shells.	Mass mortality event in 2013. Dense patches in 2021.	Present at low density 10/sqm	Present up to 96/sqm	Present, average density 52/sqm	Present, decreased density, thickened shells	Present, good condition, 19/sqm	Present, 65/sqm
SCORE	*1*	*1*	*1*	*1*	*1*	*1*	*1*	*1*
Connectivity, including to other lakes and wetlands. NB: All close to one another, and all generate a point.	y	y	y			Y (50ha wetland, drains via Jacobs Stream to Māori lakes)	y	y
SCORE	*1*	*1*	*1*	*1*	*1*	*1*	*1*	*1*

Appendix F Lake pressures and threat assessment

Pressure / threat assessment	Clearwater	Camp	Emma	Denny	Heron	Emily	Māori east	Māori west
RATING. A score of 13-15 is 'low'; 9-<12 is 'moderate' and <9 is 'high'. NB: max. pressure / threat is indicated by a zero score	**high**	**high**	**high**	**high**	**high**	**moderate**	**moderate**	**high**
TOTAL PRESSURE/THREAT **SCORE**. Maximum of 15	**6**	**4**	**6**	**5**	**6**	**9**	**9**	**6**
Biosecurity - Pest plants. Invasive submerged weed pressure was scored as follows based on LakeSPI Invasive Impact Index (III): III<10% - 3; III>10 <50% - 2; III >50% -1; no vegetation - 0.	49	31	71	74	56	74	49	72
SCORE	*2*	*2*	*1*	*1*	*1*	*1*	*2*	*1*
Biosecurity - Pest and non-native fish. Fish pressure was scored as follows based on the presence of the highest impact species: non present– 3; FRAM <20 – 2; FRAM 20-25 – 1; FRAM >25 – 0.	Perch FRAM score 31, Brown trout FRAM score 26; Rainbow trout FRAM score 23; Tench FRAM score 14	Perch FRAM score 31, Brown trout FRAM score 26, Rainbow trout FRAM score 23	Brown trout FRAM score 26, Rainbow trout FRAM score 23; Perch FRAM score 31	Perch FRAM score 31	Brown trout FRAM score 26, Rainbow trout FRAM score 23, Chinook salmon score 18	*no data or no non-native fish?*	Brown trout FRAM score 26	Brown trout FRAM score 26
SCORE	*0*	*0*	*0*	*0*	*0*	*3*	*0*	*0*
Spread/incursion risk. Risk of spread was scored as: not accessible & no motorboats permitted -3, accessible off-roading -2, accessible & no motorboats - 1, proximity to main road & concrete boat ramp – 0.	Motorboats not permitted, row boats allowed, concrete boat ramp	Motorboat permitted; concrete boat ramp	Boat access to the lake is poor, but fishing is allowed from moored boats	Off road access, no boat ramp	Nature refuge, no motorboats allowed, row boats only; access at several points along the lake	Off road access, no boat ramp	Accessible (i.e., road goes right by the lake), no boat ramp; not attractive for boating	Off road access, no boat ramp
SCORE	*1*	*0*	*2*	*3*	*2*	*3*	*3*	*3*

Pressure / threat assessment	Clearwater	Camp	Emma	Denny	Heron	Emily	Māori east	Māori west
Eutrophication - Pasture nutrient pressure based was on % pastoral land coverage from the FENZ. Pasture nutrient pressure scores were: <1% - 3; 1-25% - 2; >25-50% -1; >50% - 0.	26.78	72.53	24.3	32.9	29.17	82	38.85	73.2
SCORE	*1*	*0*	*2*	*1*	*1*	*0*	*1*	*0*
In-lake enrichment pressure was scored as follows: Chl a (med) < 2 mg/m^3 – 3; Chl a 2-5 mg/m^3 – 2; Chl a >5-12 mg/m^3 – 1; Chl a >12 mg/m^3 – 0.	4.3	2.4	9.2	18.1	4.4	2.9	1.3	4.1
SCORE	*2*	*2*	*1*	*0*	*2*	*2*	*3*	*2*

NIWA
Taihoro Nukurangi

Ōtūwharekai Potential Actions

Task 3 - Catchment Interventions

Prepared for Ministry for the Environment
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Executive summary

The brief for this report, the third component of potential actions for the Ōtūwharekai (Ashburton Lakes) Action Plan, is to address "*Catchment interventions beyond on-farm reductions – Identify opportunities for suitable interventions and appraise the likelihood of success on a lake-by-lake basis, prioritising the most critical lakes first*."

We initially reviewed the lake contaminant load reduction targets in terms of catchment losses, and resultant loading rates for the lakes and natural wetlands, both for the current situation and after proposed target reductions in lake loading are achieved. Although there are considerable uncertainties in the modelled loadings, they illustrate the relative magnitude of nutrient loadings to the lakes and wetlands, and the corresponding load reductions needed in the catchment.

Nitrogen (N) and phosphorus (P) losses from the Ōtūwharekai catchments were on average, similar to typical rates for hill and high-country beef and sheep farming in the region, ranging from 0.8-9.2 kg N/ha/y and 0.03-0.9 kg P/ha/y. However localised areas of intensive agriculture in close proximity to many of the lakes and wetlands generate elevated rates of nutrient loss.

Current areal mass loadings to the lakes, estimated from CLUES modelling, range from 34-1,671 kg N/ha lake/y and 1.2-61 kg P/ha lake/y. This shows that around half of the lakes are receiving areal nutrient loadings close to, or above agronomic fertiliser application rates commonly applied to intensive fodder cropping areas in the region.

Lake nutrient reduction targets of up to 83% for N and up to 93% for P have been proposed for some of the lakes to meet the Environment Canterbury Land and Water Plan objectives. If the proposed reductions in nutrient inflows were achieved, they would significantly reduce areal nutrient loads to the lakes, but loadings to the Māori Lakes (N and P) and Lake Denny (N) would still remain high (>150 kg N/lake ha/y and >50 kg P/ lake ha/y).

Natural wetland areas, ranging from ~10-370 ha, are present in all catchments except Lake Camp's. Their location at the land-water interface on the edge of waterways, seepage zones and lakes, suggests they could play a significant role in nutrient attenuation and protection of lake water quality. However, there is uncertainty regarding the proportion of surface-water and groundwater passing through, rather than under or around them. Additionally, these nationally significant fen wetlands are key sites in the Department of Conservation's Arawai Kākāriki Wetland Restoration Programme and have intrinsic ecological values which require protection.

In most of the lake catchments the estimated average areal nutrient loads to the natural wetlands (especially for N) are well above critical nutrient load limits recommended for maintenance of their ecological integrity (maximum 40 kg N/ha /y and 20 kg P/ha /y). Nutrient load reductions proposed to meet the lake targets would substantially reduce risks to the fen wetlands, assuming they occurred upstream of the wetlands. However, some wetlands would still receive nutrient loads above recommended upper limits.

If the nutrient attenuation capability of the natural wetlands continues to be utilised without reduction of incoming loads there are likely to be changes in their ecology and integrity in the long-term, particularly in areas where nutrient-rich inflows are concentrated. This is expected to cause a reduction in the diversity of plant communities and a change to taller-growing plant communities adapted to higher nutrient conditions, such as raupo (bulrush, *Typha orientalis)*. There is also a greater risk of exotic weed invasion across the wetlands under elevated nutrient loadings.

Information from two limited hydrogeological investigations undertaken in parts of Ōtūwharekai's complex glacio-fluvial landscape were used, along with experience from similar montane terrain, to characterise the key flow pathways involved in transport of different contaminants and identify opportunities for interception and treatment. Surface water flow-paths, particularly ephemeral streams, are expected to be most important for mitigating sediment and phosphorus losses. The priority flow path for nitrogen is expected to be via leaching to groundwaters flowing through the permeable, underlying glacial deposits. These may emerge directly into wetlands and lakes or, particularly in the larger, low-gradient valleys, be intercepted by streams.

Suitable interventions for reducing contaminant loads to the wetlands and lakes, include both enhancement of natural attenuation processes and implementation of additional mitigation options. The most suitable interventions identified for contaminant attenuation in the Ōtūwharekai are:

- Enhancement of riparian zones by livestock exclusion and planted buffers.
- Natural and constructed wetlands intercepting surface-waters and shallow groundwaters.
- Detainment bunds and sedimentation traps intercepting ephemeral surface water flows.
- Woodchip denitrification walls intercepting groundwater flows.
- Improved wastewater management in the Lake Clearwater settlement and lake-side camping areas.

Recommendations on appropriate interventions are provided on a lake-by-lake basis and guidance provided as to where they should preferentially be located. However, it should be noted that many of these interventions have not been tested in high-country settings previously and may require adaption and testing to ensure their sustainability and verify their efficacy.

The reduction of nitrogen loads achievable will depend to a large extent on what proportion of groundwaters can be intercepted at shallow depth as they enter lakes, wetlands, and riparian zones where they can interact with organic-rich soils and plants. We currently lack this specific flow-path information, limiting our ability to specifically target mitigations to the critical locations and predict their overall efficacy. Substantial benefits could be realised by incorporating the site-specific local knowledge of farmers and iwi, supplemented with some focussed hydrogeological investigations in key parts of the catchments. This would enable improved identification of key source areas, transport pathways, interception points and, ultimately, the proportion of contaminant load that can be attenuated and that which cannot. This information would enable better targeting of interventions and improved certainty of outcomes.

We thus, recommend in the next stage of this project, working collaboratively with landowners, iwi and other stakeholders to further evaluate the suitability of the various interventions recommended, and develop site-specific integrated approaches to implement them. Such interventions will need to be considered along with changes in on-farm management to achieve the environmental objectives sought for each catchment.

1 Introduction

1.1 Scope

The Ōtūwharekai Action Plan is intended to be a structured and well-evidenced programme of work to stop further degradation of the lakes and wetlands of the Ōtūwharekai catchment and deliver a pathway to restoration. This report focusses on catchment interventions to reduce contaminant loads to the lakes and wetlands, the third strand of work listed below:

1. Farm System (AgResearch) – This is a peer review of the estimated impact of the proposed farm system mitigation actions developed by Environment Canterbury, including, review of the suitability of mitigation approaches and the quantities of nutrient (nitrogen (N) and phosphorus (P)) reductions predicted (and matched to the nutrient reductions (N and P) required for individual lakes).

2. In-lake mitigations (NIWA) – Identify which lakes may need/require in-lake mitigation and assess feasible options on a lake-by-lake basis including, reviewing the available range of mitigations (flocculating agents, sediment capping, aeration, etc.), and quantifying expected impacts/projections on lake health outcomes.

3. Catchment interventions beyond (on-farm[1]) reductions (NIWA) – Identify suitable interventions and appraise the likelihood of success on a lake-by-lake basis, prioritising the most critical lakes first.

4. Cost and Impact (NIWA) – This piece looks across the proposed mitigations in 1, 2 and 3, above, costing them. This involves comprehensive consideration of impact on economic activity on farms, loss of income, cost of physical mitigations (both on farm and in-lake) as well as the cost of long-term rehabilitation and restoration. A cost-benefit analysis of the mitigations available across the catchment will help prioritise mitigations in terms of impact on lake and ecosystem health.

We have focussed our attention on the catchments of Lakes Heron, Clearwater, Camp, Emily, Emma, Denny and Māori East and West (Appendix A) and the 4 large farms delineated in Appendix B.

1.2 Approach

Having some first-hand knowledge of the Ōtūwharekai lakes and wetlands and their catchments from previous work in the area, our approach was to:

1. Understand and prioritise current water quality issues and targets in the catchment using available knowledge, key contaminants of concern, and the magnitude of load reductions required.

2. Identify likely contaminant source areas, the dominant flow paths involved in contaminant transport, and potential interception points within the catchments.

3. Assess potential interventions within the catchment to increase attenuation of contaminant loads before they flow into the lakes, considering both enhancement of

[1] Added for clarification.

natural attenuation processes and implementation of appropriate additional mitigations.

4. Make recommendations as to the most relevant within-catchment interventions and the steps that would be required to deliver the improvements required.

2 Lake and wetland water quality and contaminant load reduction targets

2.1 Lakes

Bayer and Meredith (2020) provide a comprehensive overview of the water quality information available for each of the Ōtūwharekai Lakes. In general, all 8 lakes are showing signs of deteriorating water clarity and ecological condition due to elevated nutrient loadings. Lake water TN : TP ratios indicate they are primarily phosphorus-limited, except for Lake Denny which appears to be co-limited by both N and P. However, both Bayer and Meredith (2020) and Hofstra et al. (2022) recommend that both N and P should be managed wherever possible.

All of the lakes require reductions in catchment N loads to meet water quality targets under the Environment Canterbury Land and Water Plan. Four of the smaller lakes also require reductions in P loads. Based on required percentage nutrient load reductions required (Table 5, Kelly *et al.* 2021) the priority for catchment nutrient load reductions is:

Denny (N & P) > Emma (N & P) > Clearwater (N) > Camp (N) > Maori East (N) > Maori West (N) > Emily (N & P) > Heron (N)

Current nutrient losses across the Ōtūwharekai landscape estimated using the CLUES model do not appear to be particularly high when averaged across the whole lake catchments, ranging from 0.8-9.2 kg N/ha/yr and 0.03-0.89 kg P/ha/yr (Table 2-1 and Table 2-2). These loss rates are generally within the range typical for high and hill country sheep and beef farm typologies for Marlborough-Canterbury regions (4 and 8 kg N/ha/yr, and 0.3 and 0.5 kg P/ha/yr for high and hill country, respectively; Monaghan *et al.* 2021). However, a large proportion of the catchments are in the conservation estate and the active areas of agriculture (i.e., those that generate the bulk of the nutrient losses) are located mainly in the broader valleys close to the wetlands and lakes. The catchments with notably higher apparent mean nutrient loss estimates are Lake Camp (9.2 kg N/ha/yr) and Heron (0.89 kg P/ha/yr). The high loss estimate for Lake Camp is surprising given the preponderance of low producing grassland and lack of intensive agriculture in the catchment (see Appendix D). It is possible that in the CLUES modelling the nutrient load estimated for the settlement has been incorrectly included in the Lake Camp catchment rather than for Lake Clearwater. Review of the LCDB5 landcover values suggests that this is the case.

Despite these uncertainties, the estimates of nutrient loading relative to lake area are instructive, illustrating the relative magnitude of nutrient loadings which range from 34–1,671 kg N/ha/yr and 1.2–61 kg P/ha/yr (Table 2-1 and Table 2-2). Mean areal loading rates are above 100 kg N/ha/yr for (in ascending order) Lakes Camp (117), Denny (1,120) and Māori Lake (1,671), and above 50 kg P/ha/yr for the Māori Lakes (55) and Denny (61). Overall, all but one of the lakes showed areal N loading rates close to, or well above, those commonly applied to agricultural cropping areas in the region (~60 kg N/ha /y). The two highest lake areal P loadings were also well above those commonly applied to crops in the region (20-30 kg P/ha /y).

If the nutrient reduction targets proposed to reduce in-lake nutrient concentrations to meet Environment Canterbury Land and Water Plan Objectives (CLWPO) (Kelly *et al.* 2021) were achieved, catchment loadings to the lakes would be substantially lower, reducing lake loading rates to 8-896 kg N/ ha/yr and 0.4-55 kg P/ ha/yr respectively (Table 2-1 and Table 2-2). However, Lakes Denny and

Māori would still have areal N loading rates above 100 kg N/ha/yr, and the Māori Lakes would have a mean areal P loading >50 kg P/ha/yr.

Water quality and associated ecological issues relating to in-lake mitigation options in the Ōtūwharekai Lakes have been reviewed in the Task 2 report by Hofstra et al. (2022). Based on the available information, Hofstra et al. (2022) proposed the following priority for lake restoration:

Heron > Clearwater > Māori East > Māori West > Camp > Emily > Emma > Denny

Ōtūwharekai has an immense cultural significance for Ngai Tahu whanau as an important area of seasonal mahinga kai and as a major travelling route between settlements on the eastern coast of Te Waipounamu (the South Island) and Te Tai Poutini (the West Coast). A cultural health assessment of the area (Te Rūnanga o Arowhenua *et al.* 2010), noted the need for solutions to reduce diffuse and point source pollution of waterways in the area, and called for increased protection and enhancement of waterways through the development of native riparian and wetland buffer zones. Recommendations for future management, based on a cultural health assessment by Te Rūnanga o Arowhenua are listed in Appendix B , and priorities for action identified in that assessment are repeated below for easy reference:

- Ōtūwharekai (East) / Lower Māori Lake,
- Kirihonuhonu / Lake Emma,
- The Oliver Stream and Swin river access areas of Ō Tū Roto / Lake Heron,
- Te Puna a Taka / Lake Clearwater; and
- Otautari / Lake Camp.

Table 2-1: Summary of current and target catchment nitrogen losses and consequent areal loading rates to lakes and wetlands. Catchment, lake and wetland areas from LCDB5. Target rates are based on the proposed nutrient reduction targets calculated by Kelly *et al.* (2021) to meet the desired lake TLI values (Bayer and Meridith, 2020). Values shown in red exceed upper levels recommended for wetland ecosystem protection (40 kg N/ha/y).

Lake	Catchment area[1] (ha)	Lake area (ha)	Wetland area (ha)	Current loss/loading rates				Proposed target rates					
				Catchment TN loss (T/y)	Mean catchment TN loss rate (kg/ha/y)	Mean lake TN loading rate (kg/ha/y)	Mean wetland TN loading rate (kg/ha/y)	Percentage catchment TN loss reduction (%)	Catchment TN loss reduction (T/y)	Mean catchment TN loss rate reduction (kg/ha/y)	Mean catchment TN loss rate (kg/ha/yr)	Mean lake TN loading rate (kg/ha/yr)	Mean wetland TN loading rate (kg/ha/yr)
Heron	10 460	713	370	30	2.9	42	81	4	1.2	0.11	2.8	40	78
Clearwater	4 373	208	49	12	2.7	58	248	69	8.3	1.89	0.9	18	77
Camp	564	44	0	5	9.2	117		52	2.7	4.78	4.4	56	
Denny	1 851	6	10	7	3.8	1 120	666	83	5.8	3.13	0.6	191	113
Emily[2]	1 556	21	99	1	0.8	56	12	30	0.4	0.23	0.5	39	8
Emma[3]	2 113	189	167	6	3.1	34	39	75	4.9	2.29	0.8	8	10
Māori combined	8 211	39	238	66	8.0	1671	276	46	30.5	3.72	4.3	896	148

[1] Excluding lake area.
[2] Lake Emily's catchment area has been revised based on advice from ECAN, significantly increasing its overall contributing catchment.
[3] Excludes additional linked catchment area for Lake Camp in upper catchment.

Table 2-2: Summary of current and target catchment phosphorus losses and consequent areal loading rates to lakes and wetlands. Catchment, lake and wetland areas from LCDB5. Target rates are based on the proposed nutrient reduction targets calculated by Kelly *et al.* (2021) to meet the desired lake TLI values (Bayer and Meridith, 2020). Values shown in red exceed upper levels recommended for wetland ecosystem protection (20 kg P/ha/y).

				Current loss/loading rates				Proposed target rates					
Lake	**Catchment area[1] (ha)**	**Lake area (ha)**	**Wetland area (ha)**	**Catchment TP loss (T/y)**	**Mean catchment TP loss rate (kg/ha/y)**	**Mean lake TP loading rate (kg/ha/y)**	**Mean wetland TP loading rate (kg/ha/y)**	**Percentage catchment TP loss reduction (%)**	**Catchment TP loss reduction (T/y)**	**Mean catchment TP loss rate reduction (kg/ha/y)**	**Mean catchment TP loss rate (kg/ha/yr)**	**Mean lake TP loading rate (kg/ha/yr)**	**Mean wetland TP loading rate (kg/ha/yr)**
Heron	10 460	713	370	9.28	0.89	13.0	25.1	0	0	0	0.89	13.0	25.1
Clearwater	4 373	208	49	1.83	0.42	8.8	37.7	46	0.84	0.19	0.23	4.7	20.4
Camp	564	44	0	0.18	0.32	4.1	-	0	0	0	0.32	4.1	-
Denny	1 851	6	10	0.38	0.21	60.9	36.2	93	0.35	0.19	0.01	4.3	2.5
Emily[2]	1 556	21	99	0.05	0.03	2.4	0.5	31	0.02	0.01	0.02	1.7	0.3
Emma[3]	2 113	189	167	0.23	0.11	1.2	1.4	67	0.15	0.07	0.04	0.4	0.5
Māori combined	8 211	39	238	2.17	0.26	55.1	9.1	0	0	0	0.26	55.1	9.1

[1] Excluding lake area.
[2] Lake Emily's catchment area has been revised based on advice from ECAN, significantly increasing its overall contributing catchment.
[3] Excludes additional linked catchment area for Lake Camp in upper catchment.

2.2 Wetlands

Relatively large areas of natural wetland[2], ranging from ~10 ha to 370 ha, are present in all catchments except Lake Camp. Proportionally, wetlands make up ~1-6% of the catchment area of Lakes Clearwater, Māori, Heron, Emily and Emma. The ability of wetlands to attenuate sediment, nutrients and other contaminants flowing from land to surface waters is well-established (Johnston, 1991; Mitsch, 1992; Jordan *et al.* 2011; Hansen *et al.* 2018). Given the strategic presence of these wetland areas at the land: water interface, within and alongside waterways and seepage zones on the margins of the lakes, they likely play a significant role in localised protection of lake water quality.

The Ōtūwharekai wetlands are considered to be one of the best intact examples of inter-montane wetland systems in New Zealand, and have been selected as one of the five flagship sites in the Department of Conservation's Arawai Kākāriki wetland restoration programme (Macdonald and Robertson, 2017). As such, protection of their ecological integrity should be an important goal of catchment management. Elevated inputs of contaminant to these naturally nutrient-poor wetlands have the potential to affect the composition and productivity of their wetland vegetation communities (Bedford *et al.* 1999), and their biogeochemical and ecological functioning (Hemond and Benoit, 1988; Zedler and Kercher, 2005).

The lakes most commonly show P-limitation, whereas the wetland plant tissue N : P ratios and observed responses to experimental nutrient additions in the wetlands draining to Lake Clearwater indicate N-limitation in the predominant wet fens (Burge *et al.* 2020). In the absence of specific information from other wetlands in the area we have assumed the results of this study are broadly representative of the situation in other Ōtūwharekai wetlands.

A diverse range of wetlands occur in the area (Hooson, 2015; Burge *et al.* 2020). In brief, the main wetland vegetation in the low nutrient fens is bog rush (*Schoenus pauciflorus*) and low stature sedges (e.g., *Carex diandra*, *C. gaudichaudiana*). In deeper pools and lake margins, taller-growing purei (*Carex secta*) are common, with raupo (*Typha orientalis*) dominating in some higher-nutrient areas, in particular in the Māori Lakes and Lake Emma. Red tussock (*Chionochloa rubra*) is common in ephemeral seeps, swales and riparian areas around the wetlands as well as in surrounding lowland areas.

Small plot fertilisation experiments involving N (equivalent to 35 and 70 kg/ha/yr either as nitrate or ammonium), P (20 kg/ha/yr) or N+P additions over ~3 years to plant communities in the fens northwest of Lake Clearwater caused relatively small changes in *Schoenus pauciflorus*, *Carex diandra*, or *Chionochloa rubra* dominated communities (Burge *et al.* 2020). However, after three years of fertilisation at the higher rates of nitrogen application tested, these communities showed significant increases in above-ground vegetative biomass and reductions in species diversity. Diversity losses were predominantly of short-statured forb[3] species (both native and introduced), likely due to increased cover of taller growing sedges. Responses to increased nutrient supply can often be gradual in relatively intact natural wetland plant communities such as these, due to their inherent low growth rates, associated with tolerance of the harsh montane climate, flooded soils and low natural nutrient supply (Chapin *et al.* 1986; Gough *et al.* 2000). However, these slow changes can lead to significant cumulative changes in community composition, biomass and functioning over

[2] based on the herbaceous freshwater vegetation category of LCDB 5; see Appendix D.
[3] Any herb species that is not a grass or grass-like.

decadal timescales. They are also likely to increase the susceptibility of these wetland plant communities to invasion and spread of faster-growing, more nutrient-demanding species (Sorrell *et al.* 2004), including natives such as raupo (*Typha orientalis*) and introduced species such as grey willow (*Salix cinerea*) and monkey musk (*Erythranthe guttata*, formerly *Mimulus moschatus*). These nutrient demanding species are already present in restricted areas, and others may be introduced from outside the area.

Based on the evidence above we would expect gradual changes in the composition of the natural wetland plant communities to occur in the areas subject to elevated nutrient loads (particularly of nitrogen). Impacts are most likely down-gradient of source areas (e.g., more intensive cropping and pasture areas) in zones where nutrient-rich seeps or flows enter and preferentially flow through wetlands. Changes in plant litter nutrient concentrations and decomposition dynamics as a consequence of larger nutrient inputs may also affect rates of nutrient cycling, carbon sequestration and emission of greenhouse gases (Wang *et al.* 2014).

Much of the increased nutrient inputs to the wetlands would be incorporated in the wetland plants, litter and soils (especially P), while microbial transformation would return other contaminants to the atmosphere (predominantly N). If nutrient loads to the wetlands can be kept to reasonable levels (see below), then the extent of changes in wetland composition and functioning are likely to be more limited in magnitude and extent, which may be able to be seen as a tolerable low-cost outcome.

Different wetland types are able to tolerate a wide range of nutrient loadings and there are no well-established numerical criteria, limits or concentrations that define the ideal water quality for different natural wetland types. Appropriate nutrient criteria are therefore commonly determined based on natural reference conditions for a specific wetland type (USEPA, 2008; Sorrell, 2010; Verhoeven, 2014). Fens such as those in Ōtūwharekai are fed by precipitation, groundwater flow and surface water flow, and are normally considered to have intermediate sensitivity to N enrichment (Morris, 1991; Koerselman and Verhoeven, 1992; Bobbink *et al.* 1998; Clarkson and Peters, 2010).

One approach to maintaining wetland condition would limit nutrient loads input to the wetlands below the levels associated with measurable changes in vegetation reported by Burge et al. (2020); i.e., upper levels ≤40 kg N/ha/y and ≤20 kg P/ha/y. This N load is consistent with the critical limit for fens proposed by Hefting et al. (2013). However, research carried out along gradients of atmospheric N deposition in Europe and analysis of meta-datasets on the effects of increased deposition has identified a critical N load of 25 kg N/ha/y for fens (Bobbink et al. 1998; Verhoeven, 2014). Furthermore, Richardson and Quian (1999) have calculated critical loading rates for P at 10 kg P/ha/y, based on the analysis of a large database of wetlands enriched with nutrients, and Hefting et al. (2013) have proposed protective P loading rates of 5 kg P/ha/y. We have used the lower of these N and P values as indicative protective (i.e., precautionary) limits.

Figure 2-1 and Figure 2-2 compare the mean current and future target nutrient loadings to the wetlands in each Ōtūwharekai catchment. Wetlands in all catchments on average currently receive nitrogen levels above those recommended for protection of their ecological heath, and four catchments receive phosphorus levels on average above those recommended. Target reductions proposed to achieve desired lake TLIs appear to be insufficient to achieve recommended maximum N loading rates for wetland protection in 4 catchments (Heron, Clearwater, Denny and Māori), and insufficient in 2 wetlands (Heron and Clearwater) for achieving maximum P guideline loading rates for wetland protection. The target lake catchment nutrient load reductions proposed would provide protective P levels for wetlands in 3 catchments (Emma, Emily and Denny), but not achieve protective levels for N.

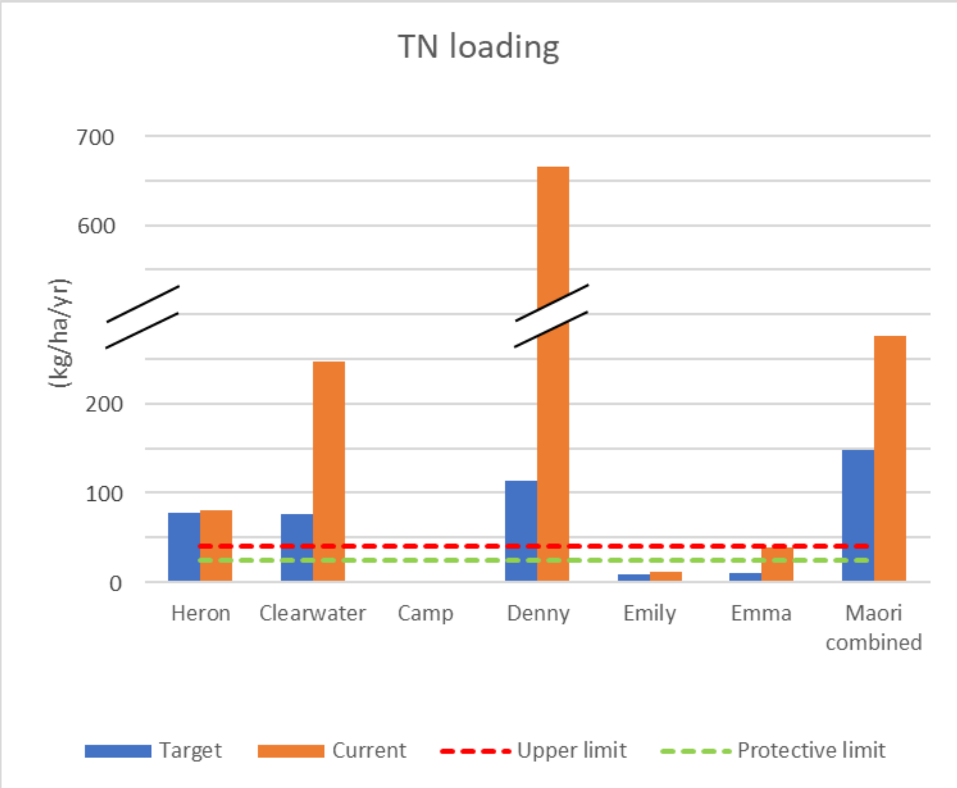

Figure 2-1: Comparison of current and target mean TN loadings to wetlands compared to proposed upper and protective nutrient limits for ecological integrity. See Table 2-1 for further information.

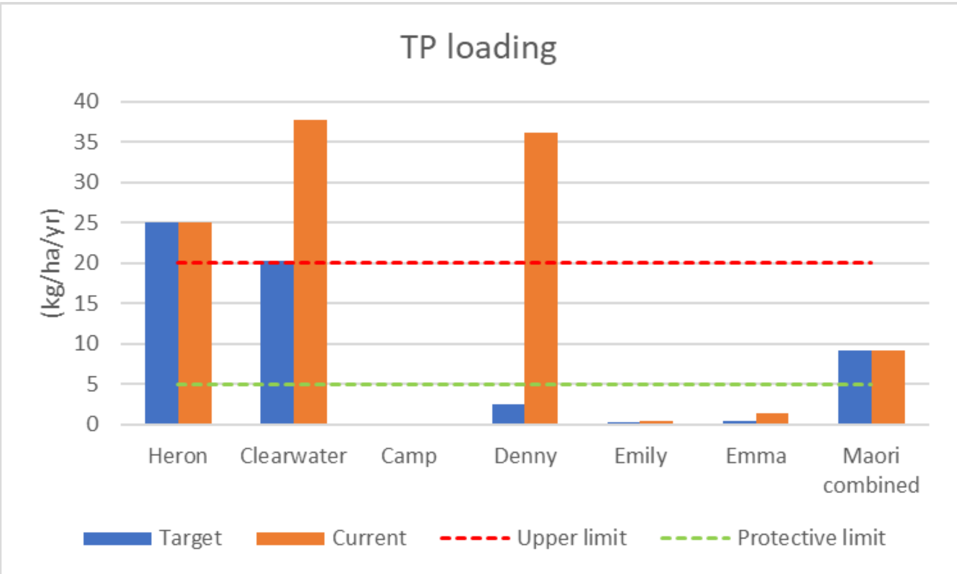

Figure 2-2: Comparison of current and target mean TP loadings to wetlands compared to proposed upper and protective nutrient limits for ecological integrity. See Table 2-1 for further information.

3 Catchments

3.1 Catchment characteristics and flow paths

Ōtūwharekai is a complex glacio-fluvial terrain. Flow paths in alpine landscapes and aquifers such as these are renowned for their complexity and heterogeneity (Somers and McKenzie, 2020). Geomorphological, sedimentological and geochronological investigations in the Lake Clearwater Basin by Evans (2008) describe a >100 m thick wedge of glacial and paraglacial sediments in the valley preserved from at least 3 glacial phases over more than 180,000 years. Multi-layered deposits of morainic, fluvial, and lacustrine sediments, loess and paleosols of widely differing hydraulic conductivity fill the valleys in complicated arrangements orchestrated by phases of glacial expansion and contraction. Geomorphological investigations by Mabin (1984, 1987) in the vicinity of Lake Heron suggest a broadly similar situation in both main basins and in the other lake catchments considered in this report. Limited investigations of soils in the catchments have focussed either on agricultural fertility (e.g., Whitley *et al.* 2018), or on soil genesis and evolution from a geological perspective (e.g., Rodbell, 1990). They provide only limited understanding of soil hydrological properties or likely flow paths through the catchments of interest.

The alpine and montane climate of the area during winter is well described by Boraman (2011): "…for much of the winter months the upper basins are covered with snow. The lower basin(s) would be covered in snow several times a year. There are occasional seasonal large snowfall that covers the basin for prolonged periods of time" … "During the winter months the wetlands areas tend to freeze up and the release of water is slowed into the lakes. Over winter, the Māori Lakes surface is often completely frozen over, this may last for several weeks at a time." In contrast during summer and autumn the upland areas of the catchment dry out significantly, hastened by low seasonal rainfall and frequent winds.

Hydrological and nutrient load studies in the Lake Clearwater catchment (Caruso *et al.* 2013; Wadworth-Watts *et al.* 2013) estimated that surface run-off and groundwater flow supplied approximately equal proportions of N and P load to Lake Clearwater and its associated wetlands. Water resource investigations by Boraman (2011) estimated around 35% of the inflows to the Māori Lakes entered directly as groundwater. He noted water bubbling out of the silt on the eastern shores and thought it likely this would be typical over the lakebed. The remaining 65% of inflow entered the Māori Lakes as surface flows, with the interconnected wetland and streams largely fed by shallow groundwaters. From these studies and our experience in the catchment, we have inferred the following regarding contaminant flow pathways. The high-county areas are relatively steep and rocky with shallow soils, favouring surface runoff. Eroded channels on the steep upper slopes link via ephemeral channels into alluvial terraces on the toe-slopes of the hills that intergrade with morainic deposits. We expect the toe-slopes, morainic terraces and valley bottom areas to be relatively free-draining, with significant infiltration to groundwater. Much of this groundwater re-emerges down slope into streams, wetlands or directly into the lakes. This is consistent with experience in other, similar landscapes (Vincent *et al.* 2019; Somers and McKenzie, 2020), and also with broad-scale hydrological modelling studies of groundwater recharge potential in the region informed by lithology, slope, aspect, land use, soil and drainage density (Singh *et al.* 2019).

In the Clearwater catchment, Whisky Stream is the only perennial stream flowing through farmland and the wetland towards the lake. Three ephemeral streams flow from natural tussock grassland catchment on the north side of the lake and wetlands. Caruso *et al.* (2013) measured decreases in flow in Whiskey Stream with distance downstream, suggesting that it was losing water to groundwater. They also noted the very porous alluvial and colluvial material underlying and adjacent to the streams and wetlands. A wider range of streams (Swin, Mellish, Olliver, Triangle and Dunbar) flow into Lake Heron, mostly originating in the mountainous high country. To the south of Lake Heron, in the wide basin flowing to the Māori Lakes. Boraman (2011) noted that that flows in Jacob's Stream on the eastern side of the catchment responded relatively quickly to rainfall events. These flow pulses were attenuated by at least 10-12 hours during passage through the wetlands and lakes. In contrast, flows from Gentleman Smith's Stream were less responsive with a prolonged flow recession, presumably due to a greater proportion of groundwater baseflow.

Together these observations suggest a strong connectivity between surface and ground waters, and the potential for parafluvial and hyporheic zones associated with the streams to function as preferential flow paths through the landscape, even when the streams are not flowing at the surface. Such flows have the potential to be intercepted to some extent by deeper-rooting riparian vegetation and wetlands, but we have insufficient information to determine for the Ōtūwharekai catchments what proportion of flows are currently or potentially able to be intercepted in these zones.

The studies of Caruso *et al.* (2013); Wadworth-Watts *et al.* (2013) provide useful hydrological estimates for the Lake Clearwater catchment , but have necessarily relied on a relatively limited dataset involving only 3 synoptic sampling events and continuous flow records over 4 months, with nearly 40% of catchment surface flows unmonitored. Discrepancies in their flow budgets across the basin led them to speculate that that the groundwater component of flows were potentially larger than 50%. Furthermore, we suspect that they may have underestimated the importance of nitrate-N losses to groundwater, as this is commonly the dominant loss pathway for grazed agricultural fields (McDowell *et al.* 2008). We therefore expect that the bulk of the nitrogen will enter the larger lakes (Clearwater and Heron) and associated wetlands via groundwater. Based on hydrological investigations of the Māori Lakes over approximately 2 years (Boraman, 2011) we expect that for lakes in the wider lower-gradient basins (Māori Lakes, and Lakes Emma and Denny), a greater proportion of groundwater flows are intercepted by streams, and so a greater proportion of contaminants (~two thirds) will enter these lakes in surface waters.

Where natural wetlands occur upslope of lakes, it is uncertain what proportion of groundwater flows will be passing through them (and so directly influenced by them), skirt around, or flow beneath them. This will depend on the nature and layering of the underlying glaciofluvial materials, and the extent to which the organic materials accumulating in the base of the wetlands have sealed them off from the underlying aquifer. Alternatively, the wetlands may be exporting organic carbon that promotes microbial denitrification in downstream waterways and aquifers. These uncertainties significantly limit our ability to identify the suitability and predict the efficacy of various mitigation options in terms of achieving the nitrogen load reductions required.

Knowledge of the pathway of groundwater flows into the lakes is also important for determining whether they may be intercepted near the lake edge (for example by deep-rooting riparian trees or in littoral wetlands) or released directly into deeper water where submersed macrophyte beds may intercept them. Slow seepage through organic lake and wetland sediments is likely to provide opportunities for effective denitrification, while higher velocity, concentrated outflows from springs into the lakes will not. In relatively homogeneous hydro-geological conditions, upward seepage to a lake decreases exponentially with distance from the lake shore (McBride and Pfannkuch, 1975; Lewandowski *et al.* 2015), but horizontal and vertical heterogeneities in the hydraulic conductivity of the aquifer and surficial lake sediments can markedly alter this pattern (Cherkauer and Nader, 1989;

Karan *et al.* 2014). Although divers have noted outflows of subsurface springs down to depths of around 12 m within Lake Heron (Hofstra et al. 2022), no systematic investigations of their extent or their implications for broader-scale flow paths through the catchments have been made.

For phosphorus, we have assumed that the majority of P will be being transported by surface streams during stormflows, and be associated with suspended particulates. The bulk of sediment will likely be generated in the steeper upstream parts of the catchment, but the most P-enriched sediments are likely to be generated closer to the lakes in areas of intensive land-use.

3.2 Diffuse catchment sources

Increased catchment contaminant loadings to the lakes and wetlands are likely to be a legacy of historical vegetation change, erosion and agricultural activity. Early Māori colonisation of the area resulted in an increase in fires from the early 14th century and change from predominantly forest to tussock grasslands (Molloy *et al.* 1963; McWerthy *et al.* 2010; McGlone, 2001). European use of the area for grazing starting in the 1950s; activities included repeated cycles of burning, over-sowing with exotic grasses and N-fixing legumes, application of lime and fertilisers, grazing by livestock (sheep and beef) and introduced pests (rabbits, hares, deer) (Acland, 1930; O'Connor, 1982, 1983; McSaveney and Whitehouse, 1989; Hoy and Isern, 1995; Scott *et al.* 1996).

Repeated firing of the landscape, in particular, is likely to have caused substantial changes in the balance and cycling of nutrients by volatilizing carbon and nitrogen and pyro-mineralizing phosphorus in plants and soil (Pellegrini *et al.* 2015). As a consequence a substantial proportion of C and N accumulated in plants and soils would have been lost to the atmosphere, and P would have been lost via runoff and leaching, with substantial fractions of P lost from the catchment deposited downstream in wetlands and lakes (Gimeno-García *et al.* 2000; Chen *et al.* 2010; Nocentini *et al.* 2022).

High and hill-country soils in the region are characteristically shallow and acidic with low fertility (N, P and S) and high levels of free aluminium limiting the growth and establishment of legumes (Haynes and Williams, 1993). Broadscale aerial top-dressing of grazed areas with phosphorus, sulphur, lime and molybdenum has been undertaken, specifically aimed at promoting N-fixing by clover and related pasture species (O'Connor, 1983; Maxwell *et al.* 2016). An additional consequence has been the stimulation of nitrogen-fixing shrubs such as the native matagouri, (*Discaria toumatou*) (Daly, 1967), which is widespread through the Ōtūwharekai catchments, and invasive exotic shrubs, such as hawthorn, broom and gorse (Williams *et al.* 2010). Fixation rates of 67 kg N/ha/yr have been measured for matagouri (Daly, 1967). It grows preferentially in and near ephemeral water courses (see Appendix J) and on shingle fans (Daly, 1967), which makes its nitrogen supply to soil readily mobile during stormflow and hyporheic flow (subsurface flows through porous streambeds).

Although large-scale burning of the land is now not purposely employed as a land management practice, agricultural intensification has occurred in restricted areas of these farms in recent years, involving cultivation and increased fertilisation for intensive fodder production and livestock grazing. Examination of the land cover statistics for the lake catchments between 1996 and 2018 (Appendix D) only show significant changes in Lake Heron's catchment, with a 747ha conversion of low producing grassland (24% reduction) to high-producing exotic grassland (4.75-fold increase). However, significant areas of fodder cropping and intensive pasture have also been observed by the authors in the last decade in the Lake Clearwater catchment on the south side of Hakatere Potts Rd (see Figure 3-1 and Appendix G). Google Earth images (2006-2020) also show evidence of more recent cycles of cultivation, cropping and intensive pasture development in these areas of the Lake Clearwater catchment (Appendix F). This brings into question the accuracy of the land cover estimates in the Land Cover Database (Landcare Research, 2018), and may explain some of the discrepancies noted by Kelly *et al.* (2021) between modelled catchment loads and measured in-lake nutrient concentrations.

Figure 3-1: Areas of improved pasture in rotation with fodder cropping on toe-slopes southwest of Lake Clearwater. Improved area in centre background. Photo taken from Hakatere Potts Rd, November 2012 (Photo: K.A. Bodmin, formerly NIWA).

3.3 Sewage sources

The only concentrated area of human habitation in the lake catchments is the fishing hut settlement situated between Lakes Clearwater and Camp (Figure 3-2). There are also a number of lodges operating in the Lake Heron catchment, and camping grounds with on-site toilet facilities on the shores of Lakes Clearwater, Camp and Heron.

The Lake Clearwater settlement currently consists of an estimated 195 huts and a camping ground in which up to 50 or more sites may be occupied (Marias, 2021). The settlement appears to have been established sometime before 1964 when it was around one quarter of its current size and actively expanding (Appendix H). By 1982, the settlement appears to have stabilised close to its current size. However, a number of huts have been extended or rebuilt in recent years and are increasingly occupied for extended periods (Marias, 2021).

There is an intermittently flowing drain and shallow gully system passing from Lake Camp through the settlement to Lake Clearwater with a significant downward gradient through the area towards Lake Clearwater and 13 m difference in lake surface elevation (Land Information New Zealand, 2019). It is likely that nutrients and other contaminants discharged to ground in wastewater from these sites would flow to Lake Clearwater. Over the last approximately 65 years, the majority of wastewater from this settlement has been discharged on-site via pit latrines (Marias, 2021). We presume these will have been supplemented by greywater seepage pits. Marias (2021), reports that many redevelopments have involved the removal of traditional long-drop toilets and replacement with holding tanks (which we assume operate essentially like septic tanks discharging to seepage pits). However, many long-drop toilets still remain. Recently the Ashburton District Council has required hut-owners to install holding tanks and have wastewaters transported out of the catchment, but this process appears to still be in early transition (Marias, 2021).

The Hut Tenant Association estimated an occupancy of 8 people/dwelling for 8 weeks of the year (Marias, 2021), however a number of the huts are advertised on internet rental sites, suggesting the period of usage could be considerably greater for some. Occupancy of the camping area has been estimated at 35 sites for 9 months of the year with average of 4 people/site (Marias, 2021). This would appear to be a high estimate based on hut occupancy estimates and apparent occupation visible from satellite imagery (Google Earth).

Nutrient losses from pit latrines and septic tanks discharging to seepage pits are highly variable depending on a wide range of factors including intensity and variability of usage, diet of users, climate, soil type, depth to groundwater and distance to discharge point or surface water (Graham and Polizzotto, 2013; Dzwairo, 2018). We have assessed likely annual contributions of nitrogen and phosphorus based on reductions in nutrients commonly observed in septic tanks and associated seepage fields, taking into account the free-draining nature of the alluvial soils in the area based on information from comprehensive reviews (Siegrist *et al.* 2000; Lusk *et al.* 2017b). These estimates appear to be broadly consistent with the limited data available for pit latrines elsewhere (e.g., Gill *et al.* 2009b; Nyenje *et al.* 2013; Dzwairo, 2018) . Our estimates of nutrient loads lost to Lake Clearwater from these sources (see calculations in Appendix I) are in the range of ~400-1,200 kg N/yr and 120-350 kg P/yr, depending on occupancy in the range of 10-30% of days per year (i.e., 37-110 d/y). Attenuation is likely to reduce the quantity of nutrient that could potentially reach Lake Clearwater to about 340-1,005 kg N/yr and 10-30 kg P/y. Our estimates for N load to the lake are 1.36 to 4-fold higher than those previously estimated by Wadworth-Watts *et al.* (2013) of up to 250 kg/yr. Their estimate was based on 10% annual occupancy by 8 people/dwelling/d for 225 dwellings, using a generalised estimate of nutrient lost per dwelling derived from a region-wide assessment of nutrient loss to groundwater from on-site septic tank sewage treatment systems in Canterbury (11 kg TN/yr, Loe, 2012). Comparison of our estimates for settlement wastewater loads with overall catchment estimates from CLUES modelling (12,010 kg N/yr and 1,183 kg P/yr, Kelly *et al.* 2021) show they could potentially account for 3-8% of total N loads and 1-2% of P loads to Lake Clearwater, for occupancy in the range of 10-30% respectively.

Figure 3-2: Google Earth image of Lake Clearwater fishing hut settlement. Lake Clearwater is visible to the north (top) and Lake Camp to the south. The campground is in the top left of the settlement, close to the Lake Clearwater shoreline. Google Earth image, 19 Jan 2018.

3.4 Climate change impacts to consider

NIWA's climate change projections for Canterbury mountain and hill country (Macara *et al.* 2020; Tonkin and Taylor, 2022) over the next ~20- 70 years predict minimal change in overall annual precipitation as temperatures warm, but reduced depths of snow and ice. The seasonality of rainfall is likely to increase, with winter rainfall more strongly associated with storm events, and earlier and less sustained snowmelt. This is likely to translate to even greater variability in catchment run-off (the difference between precipitation and evapotranspiration), with reductions in mean annual low flow, increased incidence of erosion, and greater risk of drought and fire. This will likely affect the ecology and distribution of alpine, subalpine, montane and wetland plant communities, and the functioning and ecology of the lakes. Both positive effects (shorter and less harsh winters, longer growth season) and negative effects (seasonal droughts, number of hot days, weed and pest survival and proliferation) for agriculture in Ōtūwharekai are likely, with significant potential to increase pressure on water resources, and also to encourage farming intensification with increased contaminant losses.

4 Potential interventions to manage catchment contaminant loads

A wide range of potential edge-of-field and flow-path mitigation options exist with different applicability, targeting different contaminant types and presenting different costs, benefits and disbenefits (McKergow *et al.* 2007; McDowell *et al.* 2013; Tanner, 2020). Here we have focussed on those we consider have potential application in Ōtūwharekai.

4.1 Wastewater management

The wastewater contaminant load from the Lake Clearwater settlement is reasonably small, but constitutes a readily manageable nutrient source. It has been estimated to account for around 3% of the N load to the lake, assuming 10% occupancy, but has the potential to increase up to 8% of the N load at 30% occupancy. Actions to reduce inputs of nutrients from this composite wastewater source are already in motion (Marias, 2021). The approach appears to be capture of toilet waste in holding tanks for transport and disposal outside the catchment (Marias, 2021). It is not known whether this will address greywater disposal in the settlement, which is of lesser concern as a nutrient source. Additional approaches could include limitations on occupancy, further intensification of housing or generation of additional wastewater in the area. Alternatives to transporting all wastewaters out of the catchment would be adoption of waterless toilet systems (e.g., composting, vermiculture or incineration), or advanced on-site or communal wastewater treatment incorporating nutrient removal and land application (Tanner *et al.* 2012; Bay of Plenty Regional Council, 2022).

4.2 Livestock exclusion

Caruso et al. (2013) recorded a high fraction of organic N in surface inflows to Lake Clearwater, and considered this to be from animal excreta derived from agricultural land. In addition, they noted cattle congregating in the ephemeral channels, resulting in pugging, erosion of the banks and stream bed, as well as direct deposition of faecal wastes along the channels and bed. These wastes and sediments are readily mobilised during rain events, thus acting as a 'point-source' of pollutants discharged to natural waterways and wetlands downstream (MfE, 2001).

The majority of the intensively grazed and fodder-cropped areas around the Ōtūwharekai lakes are classed as 'low slope' (average slope less than or equal to 10 degrees across the area of land parcel used for grazing) under the current Resource Management (Stock Exclusion) Regulations 2020[4]. This requires exclusion of cattle, deer and pigs from the beds of lakes, rivers (>1 m wide) and wetlands (>500 m^2), with a minimum set-back of 3 m by July 2025. Exclusion of such livestock by fencing of these waterways, strategic areas of flood plains, and the shores of lakes and wetlands would reduce the potential for them to act as sources of nutrients and faecal contaminants, and reduce the potential for trampling, compaction and disturbance of streambanks, sediment yield and shoreline erosion (Trimble and Mendel, 1995; Bilotta *et al.* 2007; McKergow *et al.* 2012; Dodd *et al.* 2016; O'Callaghan *et al.* 2019). Provision of alternative stock drinking water facilities would be required where access to waterbodies was restricted.

Sheep grazing is likely to result in lower nutrient losses and reduced erosion than grazing of cattle or deer (McDowell and Wilcock, 2008; Hoogendoorn *et al.* 2011), and as a consequence does not require fencing under the current Resource Management (Stock Exclusion) Regulations 2020. Sheep also tend to be less attracted to wet and boggy areas of the landscape (Putfarken *et al.* 2008). Sheep may be the only livestock grazing option available where fencing of streams, lakes and wetland is not

[4] https://environment.govt.nz/acts-and-regulations/regulations/stock-exclusion-regulations/

feasible or cost-effective. In such areas, strategic provision of water troughs or tree shelter to encourage relocation of livestock camping areas (which contribute a high proportion of nutrient losses; Betteridge *et al.* 2010) to areas with lower risk of waterway contamination may be appropriate interventions (Lindsay Mathews, pers. comm.).

4.3 Riparian buffers

A riparian buffer is a strip of land which separates agricultural activities from waterways, waterbodies and wetlands. Riparian buffers can provide a wide array of benefits, such as sediment and nutrient retention, stream shade and bank stabilisation, provision of natural woody debris and habitat for native plant, insect and animal species[5] (Howard-Williams *et al.* 1986; Smith, 1989; Parkyn *et al.* 2005; McKergow *et al.* 2016) (Figure 4-1).

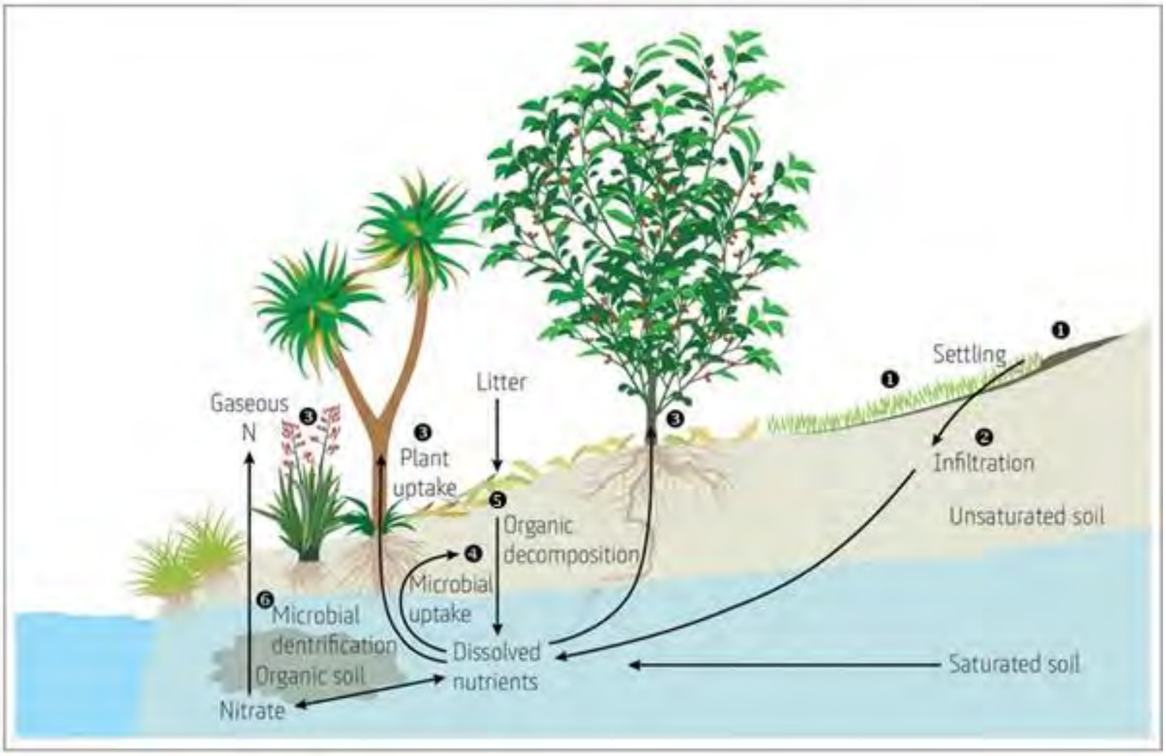

Figure 4-1: Cross section through a riparian buffer showing key sediment and nutrient removal processes. For surface run-off the main processes are (1) settling and (2) infiltration of water and dissolved contaminants following settling of particulates. For dissolved nutrients the main processes are (3) uptake through roots into plant tissues, (4) microbial uptake, (5) immobilisation as soil organic matter and (6) microbial denitrification. From McKergow *et al.* (2022).

If a significant proportion of groundwater and hyporheic flows within the broader catchment are sufficiently shallow, riparian planting alongside permanent and ephemeral waterways in the lower gradient areas of the catchment and around the margin of lakes will be able to interact with them, reducing nutrient in shallow groundwater through plant uptake and microbial denitrification and other biogeochemical processes (Lewandowski and Nützmann, 2010). Further information on groundwater pathways and zones of interaction with surface-waters is necessary to be able to assess the efficacy of such strategies. Planting with appropriate deep-rooting, non-nitrogen-fixing plants

[5] e.g., fringing stream vegetation is important as breeding habitat for some native fish species.

that provide suitable N uptake, productivity and litter characteristics (e.g., Franklin *et al.* 2019) will be important to maximise performance.

Establishment of riparian shrubs in the harsh landscape of Ōtūwharekai may require special techniques. The species selected will need to be able to survive the extreme climatic conditions of frost, snow cover, wind and cold, as well as summer dryness. Protection will initially also be required against browsing by rabbits, hares, deer etc. Although willows and other introduced riparian trees have potential to provide useful ecosystem services more rapidly than native species, they are likely to be less acceptable than native species in this natural environment, and may impose greater weed risks. Use of the native woody species that grew in this region before human disturbance should also be considered, whilst taking into account the influence of expected climate changes in the region (Macara *et al.* 2020; Tonkin and Taylor, 2022). For preliminary evaluation we recommend only implementing planted riparian buffers focussing on lower gradient areas of the catchment (average slopes of 10 degrees or less), and assuming performance is in the lower performance band proposed by McKergow *et al.* (2022), i.e., assuming a 30% reduction in sediment and N, and a 27% reduction in P for a 5 m wide buffer (5% of 100 m slope length) (Figure 4-2).

It should be noted, however, that extensive areas of riparian trees are likely to increase evapotranspiration rates, reducing water yields and flux into the lakes (Salemi *et al.* 2012; Satchithanantham *et al.* 2017; Marttila *et al.* 2018), which may reduce flushing and increase residence times in the lakes.

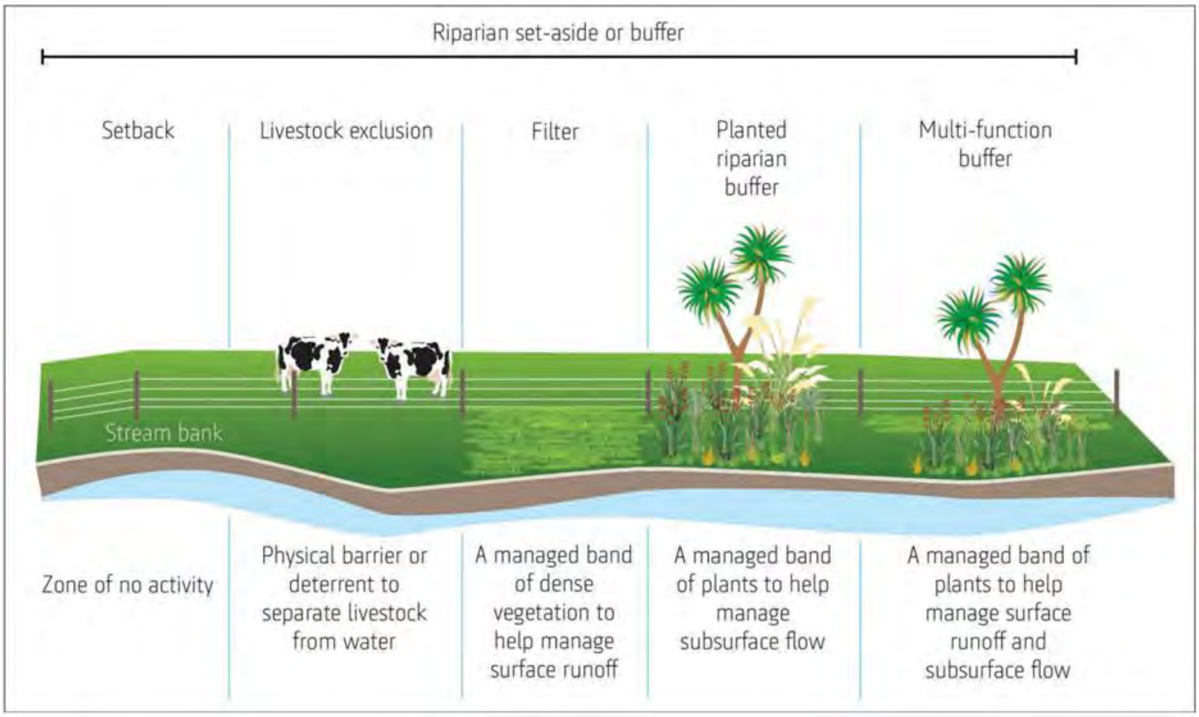

Figure 4-2: The range of common riparian buffer types. From McKergow *et al.* (2022).

4.4 Wetlands

4.4.1 Natural

Many of the Ōtūwharekai lakes support significant areas of littoral wetland (Lakes Emily, Māori East and Māori West, Emma (Hooson, 2015; Bayer and Meredith, 2020) and/or are fringed by wetlands. These wetlands can quell wave action and stabilise sediments along the shoreline. They can also attenuate nutrients flowing towards the lakes in surface waters and in shallow lateral or emerging groundwater flows (Haertel *et al.* 1995; Bratli *et al.* 1999; Gibbs and Matheson, 2001; Sollie and Verhoeven, 2008) (Figure 4-3). In addition, in some catchments (e.g., Lake Clearwater), inflows pass through natural wetlands that will attenuate nutrients passing through them derived from the catchment (Knox *et al.* 2008). The functioning of these natural wetland areas would likely benefit from livestock exclusion and pest control (rabbits, hares and rodents). There may also be opportunities to remediate wetland functioning through small-scale modifications, for example where flows have been diverted or channelised (e.g., culverts). Deeper groundwaters flowing in alluvial layers below these wetlands will not be intercepted and so are not likely to be influenced by the wetlands.

Use of existing low-nutrient status natural wetlands for contaminant attenuation may, however, impact negatively on their ecology and natural character, and be incompatible with their presence in a nature conservation reserve. We have proposed restricting nutrient loadings to these wetlands to limit impacts (see Section 5.1).

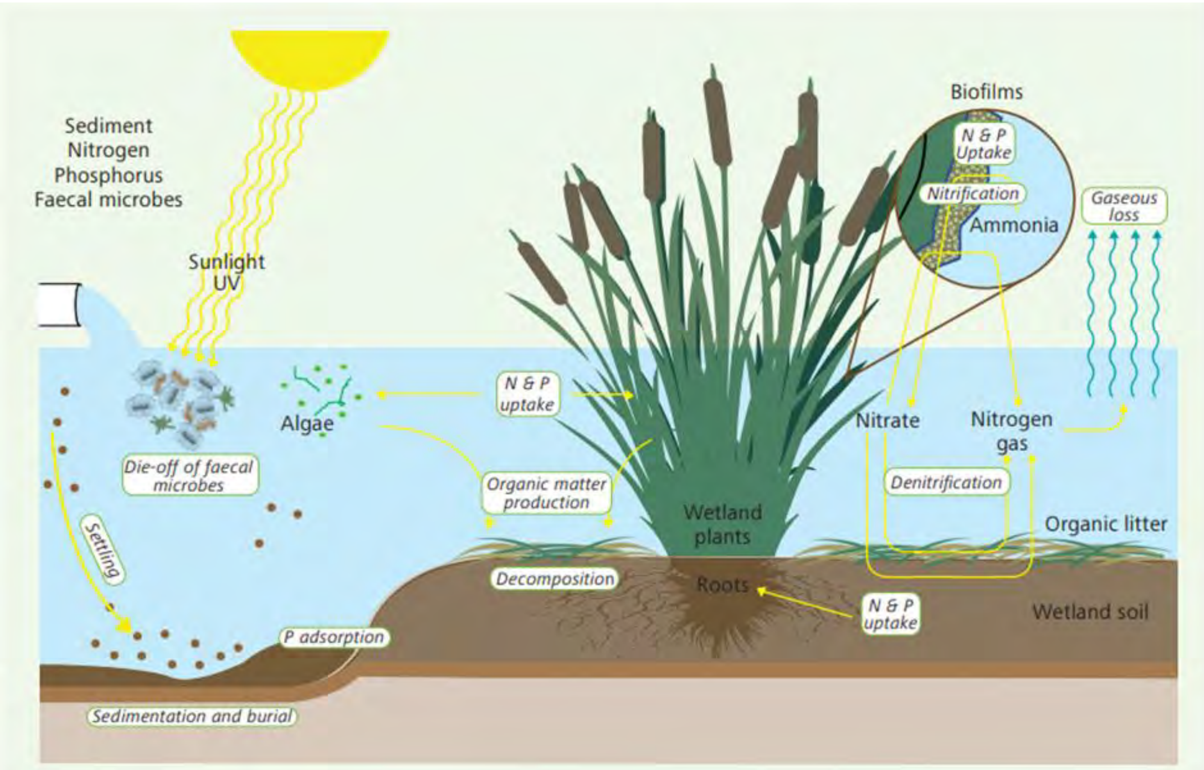

Figure 4-3: Key nutrient removal processes in a surface-flow wetland. From Tanner et al. (2022).

4.4.2 Modified natural

There is potential to enhance the sediment and nutrient removal performance of existing natural wetlands in strategic locations by enhancing dispersion and retention of flows using simple earthen bunds, rock and timber structures to create a stepped cascade of wetland cells. These options would need to be investigated further and their appropriateness evaluated relative to expected costs and benefits.

4.4.3 Constructed

Constructed wetlands are a recognised technology for reducing nutrient fluxes from agricultural land into adjacent waterways. Wetlands combine slow water velocities and specialised aquatic plant species to enhance sediment settling, microbial denitrification, plant uptake and adsorption of nutrients. Once established, a well-designed constructed wetland can sustainably remove nutrients and sediments with little or no human intervention for many decades (Kadlec and Wallace, 2009). A key feature in the success of a constructed wetland will be placement in the landscape in areas where high amounts of nutrients being transported in surface (or near surface) waters may be intercepted.

Various design features can be incorporated within a constructed wetland to optimise performance, depending on the hydrology, landscape features and characteristics of incoming pollutants. These include adjusting wetland size relative to the contributing catchment loads, use of sedimentation basins, single or multiple cell wetlands, and appropriate plant species choice (Figure 4-4). A guide for wetland design and construction has recently been produced for the New Zealand agricultural sector (Tanner *et al.* 2022).

The Ōtūwharekai climate will limit the types of plants that can survive in constructed wetlands and reduce their expected nitrogen removal performance, which is temperature sensitive. Preliminary assessment of nutrient removal performance is possible through modelling, but further studies are required to identify appropriate wetland plant species, adapt constructed wetland design, and properly quantify their annual nutrient removal performance. Given these constraints, for preliminary assessment we will assume performance is at the bottom of the range for cool climate constructed wetlands (Tanner *et al.* 2022); i.e., 22 % TN reduction for a wetland size of 2% of contributing catchment area, and 28% for a wetland size of 4% of contributing catchment area.

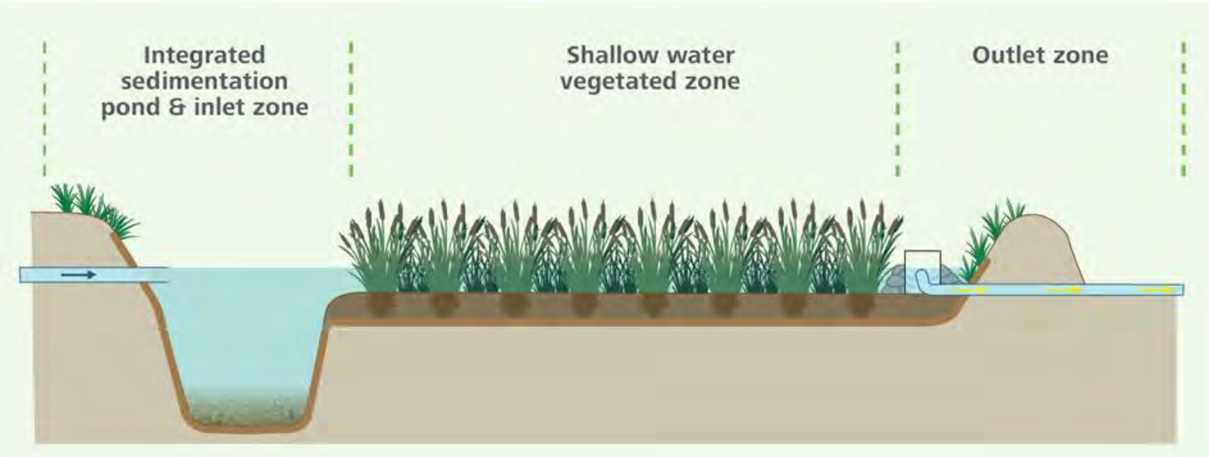

Figure 4-4: Schematic longitudinal section of a constructed wetland incorporating an initial sedimentation pond. From Tanner et al. (2022).

4.5 Denitrification walls and bioreactors

Denitrification walls are a subsurface trench filled with an organic carbon source (e.g., wood chips or sawdust) that intercepts shallow groundwater flows (Figure 4-5). Similarly, a denitrifying bioreactor is a lined bed filled with porous organic carbon media (most commonly woodchips), through which subsurface tile drainage flows (Schipper *et al.* 2010). The organic carbon acts as an energy source for microbial denitrification, in a similar manner to decomposing plant leaf litter in wetlands and riparian buffers.

Studies in New Zealand have found more than 95% nitrate removal for denitrification walls over more than 5 years operation, with no detectable reduction in the added total carbon (Schipper and Vojvodic-Vukovic, 2001). High rates of nitrate removal by denitrification walls have been recorded in USA over more than 15 years (Robertson *et al.* 2008). Similarly, bioreactors have been shown to be effective in nitrate removal by microbial denitrification and anammox processes, as well as reducing faecal bacteria and viruses (Schipper *et al.* 2010; Rambags *et al.* 2016; Rambags *et al.* 2019). For preliminary analysis, sustainable nitrate removal rates in woodchip bioreactors of ~1.5 g NO_3-N /m^3 woodchip material /d can be assumed due to the cool temperatures during snow melt. The media in both denitrification walls and bioreactors can also be chemically or thermally modified or adsorbent materials added to enhance phosphorus removal. Ballantine and Tanner (2010) have identified allophane, Papakai tephra, limestone and alum as materials with the most potential as soil amendments for P retention in the New Zealand agricultural landscape.

A denitrifying wall needs to be dug into the ground where it will intercept groundwater flows, whereas flows are directed into a bioreactor, either as a surface or subsurface (e.g., tile drain) flow. These technologies can be placed close to the source of nutrient pollution (before it has entered deeper groundwater pathways) or close to the lakes or groundwater (where groundwater returns to the surface). As with any of the suggested technologies, it is imperative that these technologies intercept flows containing nitrate and do not restrict groundwater flow through them, causing groundwaters to flow under or around them. In situations such as at Lake Clearwater, where deep underground lenses of gravel appear to be a significant hydrological pathway, it may not be possible to adequately intercept these flows close to the lakeside.

In the absence of information to the contrary, natural rates of denitrification in groundwater are expected to be low in the permeable subsoils common at Ōtūwharekai. It is possible to increase rates of in-situ microbial denitrification by injecting or dosing a suitable carbon source – liquid (e.g., methanol) or gaseous (e.g., methane) which acts as electron donor – directly into a flowing aquifer (Huno *et al.* 2018; Zhao *et al.* 2022). In practice, increasing rates of in-situ groundwater denitrification is hard to achieve, due to aquifer mixing issues and biofouling by the microbial biofilms whose growth and associated gas bubble formation is stimulated (Baveye *et al.* 2001). This option is not considered suitable or practical for immediate application – significant aquifer investigations, modelling and associated research will be required to evaluate and commission this approach. It is therefore not considered further in the current evaluation.

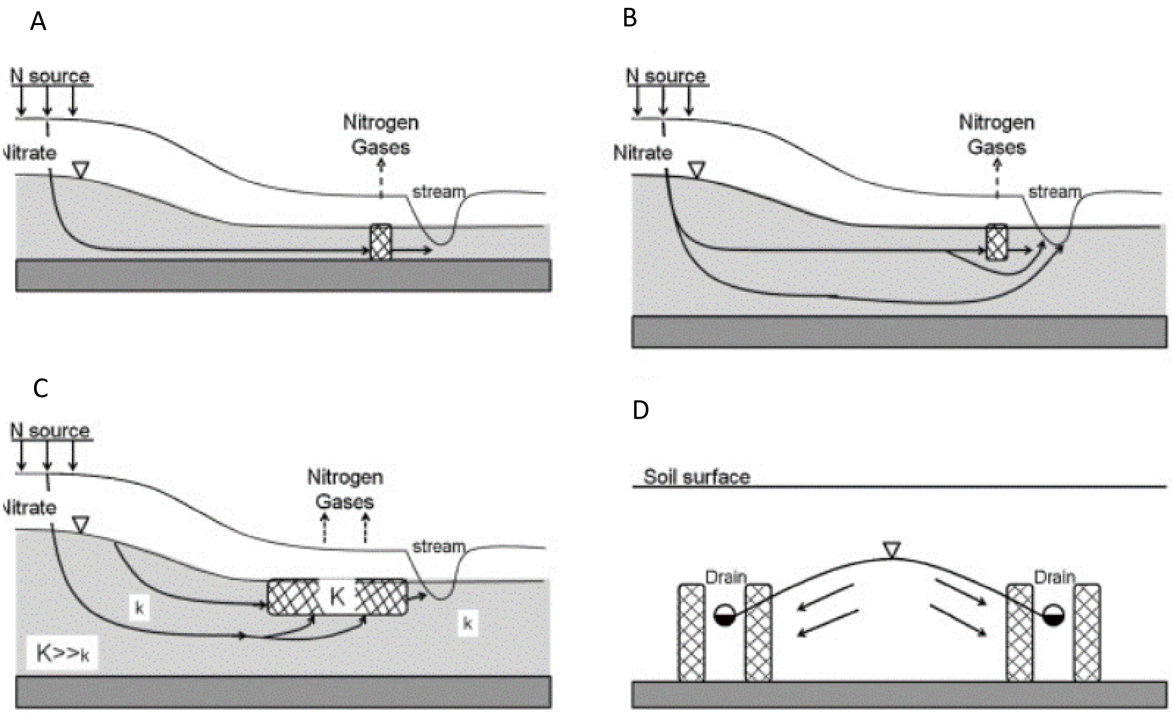

Figure 4-5: Various configurations of denitrification walls and beds intercepting groundwater flows. The walls and beds contain woodchips or other carbon-rich media. Cross-sections of: A. denitrification wall where groundwater flows are shallow, constrained by an impervious layer (aquiclude); B. denitrification wall not installed to the full depth of the impervious layer, so only intercepts a proportion of the groundwater flow; C. high permeability layer installed in upper part of aquifer causing upwelling of groundwater into a denitrification bed, and (D) denitrification walls installed on either side of a subsurface tile drain. From Schipper *et al.* (2010).

4.6 Sediment traps/basins

Sedimentation traps are "pond-like" depressions constructed to slow sediment-laden water (Figure 4-6). Reducing velocities and turbulence enables entrained particles and associated nutrients to settle out of the water flow. Sedimentation traps can be built in-line (directly in a water flow path) or off-line (where peak water flows are diverted into a basin adjacent to the main water path). The primary determinant of sediment trap efficiency is its size relative to the flows it is receiving. Large, deep (>2 m) traps allow sediments to settle out of the flow. For inorganic sediments, large particles will settle more readily than fine particles, however fine particles (silts and clays) carry proportionately more nutrients due to their higher surface area to weight ratio.

General guidance on sediment trap design is provided in Hudson (2002). Information on suspended sediment characteristics is required to evaluate their potential efficacy. The sizes of sedimentation traps are limited by the practicality of emptying them (no wider than the width an excavator can reach), and the potential for wind resuspension of captured sediments (in overly large traps).

Sedimentation traps will also capture organic particles, which often have much higher nutrient levels, however organic particles are less dense than inorganic particles. Thus, sedimentation traps are sometimes linked with constructed wetlands, which are more able to filter out these low-density particles.

In the Ōtūwharekai catchments, relatively few waterways are likely to be suitable for use as sedimentation traps, but there may be potential to locate them at strategic locations such as below road culverts (e.g., draining high intensity agricultural areas above the Lake Clearwater wetlands), where flows are channelised and readily intercepted.

Figure 4-6: A range of sedimentation traps with differing degrees of landscape fit. Sedimentation traps can be built solely with functionality in mind (left), or integrated with wetlands (two examples on right). The sedimentation traps are arrowed.

4.7 Detainment bunds

Detainment bunds provide temporary impoundment of stormflows (Paterson *et al.* 2020). They entail construction of a low-level dam wall or bund place across the flow path of an ephemeral stream, such that, when stormflows do occur, these build up behind the bund (Figure 4-7 and Figure 4-8). Water is impounded for periods up to 3 days (to minimise pasture damage), after which the waters are manually released via a drain (Levine *et al.* 2019; Levine *et al.* 2021a; Levine *et al.* 2021b). Temporarily impounding the stormflow water and associated sediments, faecal wastes and particulate (and to a lesser extent, dissolved) nutrients behind the bund promotes infiltration into the soil profile and settling of suspended particulate material and associated contaminants. These systems have been successfully trialled on dairy farms in the Rotorua region on semi permeable volcanic soils, where much removal was attributed to the infiltration observed. Investigations are beginning in other New Zealand regions (Northland, King Country and Otago) to test efficacy on less permeable soils and in combination with passive flocculant additions.

The applicability of this technology in the Ōtūwharekai catchments is uncertain. The gravelly soils in the area may not be suitable for bund/ impoundment construction, as they may lack the cohesion necessary for structural integrity. Also, while these detainment bunds are designed to work with ephemeral flows, they are generally intended for short duration flows associated with storm events. Many of the ephemeral flows in this catchment are likely to be for much greater periods (associated with snow melt), under which conditions this technology has not been tested. To reduce farmer management requirements, we would recommend detainment bunds be operated with constrained continuous outflows to temporarily detain and slow outflow, rather than as fill and release basins. This is similar to the concept of peak run-off control (Figure 4-9) as proposed by Marttila *et al.* (2010).

On the basis that this technology is untested in this landscape, and the significant number of unknown variables contributing to a successful outcome, application of this technology would ideally require monitoring to adjust operation and assess its efficacy. For preliminary assessment we recommend assuming 50% reduction in sediment load and associated particulate P loads from ephemeral surface-flows.

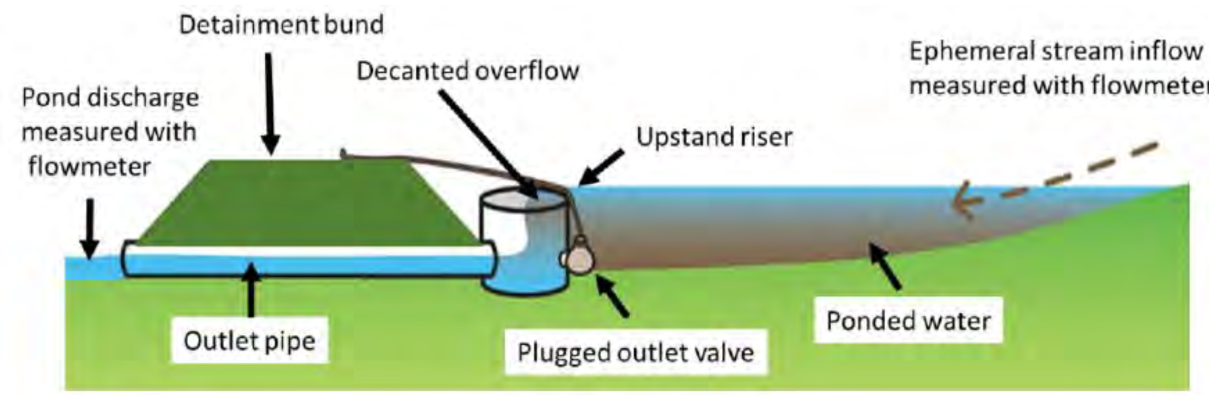

Figure 4-7: Schematic of Detainment Bund and ephemeral stream inflow. Ponding builds up behind the detainment bund. Note upstand riser and discharge pipe features. from Levine *et al.* (2020).

Figure 4-8: Ponded drainage temporarily impounded behind a detainment bund created from a causeway road. From Clarke *et al.* (2013).

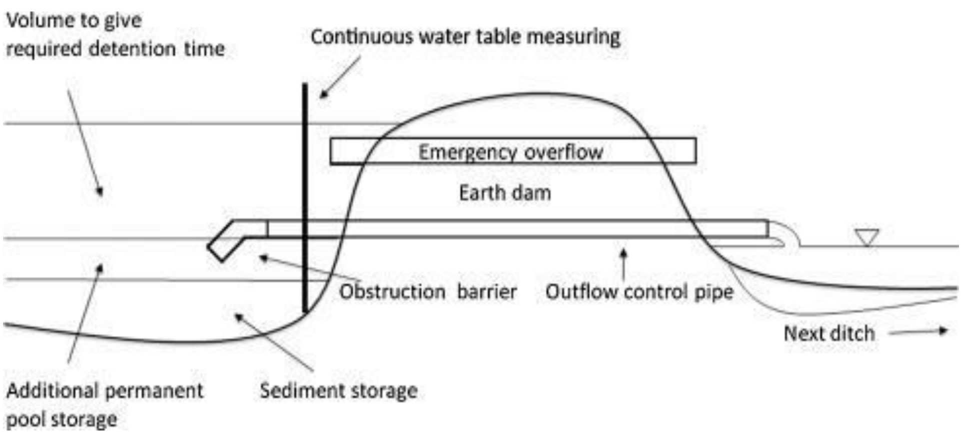

Figure 4-9: Schematic of peak run-off structure used to control discharge rates from ephemeral stream and ditch flows. (Marttila *et al.* 2010).

5 Recommendations

5.1 Catchment intervention priorities

A range of different criteria can be applied to prioritise catchment interventions; for instance: highest ecological value, most able to be protected, most at risk of change, most impacted, greatest likelihood of success. We have been asked to prioritise catchment interventions appraising the likelihood of success of suitable interventions on a lake-by-lake basis, prioritising the most critical lakes first.

5.1.1 Lake Heron

Based on load reduction targets required to meet the Environment Canterbury Land and Water Plan Objectives (CLWPO) (Kelly *et al.* 2021), Lake Heron is the closest to meeting its nutrient reduction targets, requiring a 4% reduction in N and currently meeting its TP target, but still well above its chlorophyll-*a* TLI target. It also ranks high based on ecological values.

Focussing on groundwater flows as the main flow path for N entry to the lake, retirement and planting, where appropriate, of lake, wetland, and stream margins on the western side of the lake, down gradient of intensive farming areas, is the easiest option (Figure E-3). However, as discussed above, there are significant uncertainties as to what proportion of the groundwater would be able to be intercepted and ameliorated. The prime option to intercept and treat groundwater flow paths would be to construct woodchip denitrification walls along a large section of the western and north-western inflow pathways and shore of the lake. This could require up to 4 km of denitrification wall, but it may be possible to reduce this length substantially if further investigation identified zones of highest flow and N load, which would be targeted for denitrification.

Management of P loading to the lake needs to focus on reducing sediment inputs from the higher intensity areas on the western side of the lake – sediments from this area are likely to convey the bulk of the P load.

Sediment loadings are likely to be highest in the streams that originate in the high country. Stock exclusion, planting and enhancement of riparian and associated natural wetland areas (e.g., Dunbar, Triangle and Mellish Streams) are likely to be effective mitigation options to consider in order to reduce sediment inputs to the lake.

5.1.2 Lake Emily

Lake Emily has a TN reduction target of 30% (0.4 T/yr or 1.5 kg/ha/yr) and a TP reduction target of 31% (0.02 T/yr or 0.01 kg/ha/yr). Low producing grassland is the main landcover (Figure E-6) representing 78% of the catchment (Appendix D). The lake has a moderate TN loading rate and a relatively low TP loading rate. Significant wetland areas are associated with the lake which on receive nutrient loads below critical ecological thresholds.

Achieving proposed nutrient targets for the lake is likely to require several actions:

- remediation of any obvious source areas of sediment, complemented by stock exclusion,
- enhancement of the lake margins and existing natural wetlands to the north of the lake would seem to be appropriate options to achieve the required level of additional P trapping and N reduction, and

- improvements in farm nutrient and grazing management, to minimise the ongoing mobilisation of contaminants (dissolved and particulate).

5.1.3 Māori Lakes

The West and East Māori Lakes have N reduction targets of 34 and 48% respectively, and both currently meet their TLI targets for P. These lakes are more mesotrophic to eutrophic in nature, with a large catchment to lake ratio and extensive wetland areas around and between the lakes (Figure 5-1, Figure E-7). Seventy percent of the Māori Lakes catchment is classified as low producing grassland according to LCDB5 (Appendix D), but available evidence suggests it is relatively intensively grazed by sheep, cattle and deer (draft Environment Canterbury Environmental Plans for Farm 1, and Google Earth Imagery).

Riparian retirement and planting appear to be viable interventions down-gradient of the high intensity agricultural areas upstream of the Māori lakes. Further investigations are warranted to determine the relative N concentrations in Jacob's and Gentlemen Smith's Streams and other surface-waters to the north of the lakes. These groundwater-fed inflows to the streams are likely to be dominant nutrient pathways into the lakes, and potentially able to be intercepted. Denitrification walls may allow interception and treatment of emerging shallow groundwaters before they enter these streams. Alternatively stream inflows may be able to be redirected to flow into and through existing littoral wetlands areas, to better utilise their natural attenuation potential.

Figure 5-1: Extensive raupo wetlands fringing Māori Lakes. Dead willows remaining after herbicide control visible in the background. (Photo Chris Tanner, Feb 2012).

5.1.4 Lake Clearwater

Target TN and TP reductions for Lake Clearwater of 69% and 46% respectively are substantial, requiring reductions of 8.3 T/yr of N and 0.84 T/yr of P (~1.9 kg N/ha/yr r and 0.23 kg P/ha/yr). N and P loadings to the fens to the west of the lake are also significantly above recommended guidelines (see Figure 2-1Figure 2-2).

Stopping nutrient inputs to the lake from the fishing hut settlement and camping ground is an obvious place to start, although this appears to be a relatively minor component of the overall nutrient load to the lake. The next obvious area to focus attention is the more intensive cropping and grazing areas west of the lake and south of the valuable fen wetland (Figure E-1). Both surface (P and sediment) and groundwater (N) flows need to be intercepted. Potential interventions for ephemeral surface flows include:

- sedimentation ponds, detainment bunds and/or constructed wetlands above or below culverts crossing Hakatere Potts Rd, and
- riparian enhancement in the Whiskey Stream catchment to reduce sediment-associated P (and N in surface water). This will require additional investigations in the catchment to identify critical source areas and appropriate actions.

To intercept the predominant N load conveyed in groundwater, installation of a denitrification wall approximately parallel with the road has the greatest potential. This would likely need to be ~0.8 km in length, and its most appropriate location (likely close to the wetland) would need to be identified after further investigation and consultation with land managers.

5.1.5 Lake Camp

This lake is surrounded by low producing grassland (Figure E-2) and small areas of exotic forest, which have reduced in extent in recent years. The catchment drops relatively steeply down to the lake edge with no significant areas of wetland. Current modelling suggests that a 52% reduction in N loading to the lake is required, and no reduction in P loading. However, it appears that a significant portion of the nutrient load arising from the fishing hut settlement has been assigned to the Lake Camp catchment. We think it is more likely that with 13 m of elevation difference between Lakes Clearwater and Camp (higher elevation, Land Information New Zealand, 2019), the direction of groundwater flow would be towards Lake Clearwater. Incorrect apportionment of nutrient inputs may be creating an incorrect catchment loading in the CLUES modelling for Lake Camp (excessive) and Clearwater (insufficient). If we are wrong and the direction of flow is largely toward Lake Camp, then this may explain the elevated nutrient concentrations measured in Lake Camp, and the lack of obvious indications of groundwater contamination along the shore of Lake Clearwater. Notwithstanding this, it appears from lake water quality records that nutrient reductions are also required in other inputs to Lake Camp to maintain its water quality.

We recommend that nutrient reduction efforts be focussed on the main ephemeral streams entering the lake from the western and southern sides of the lake. The incised valleys may also be significant pathways for groundwater flow through the catchment. Options to reduce sediment and nutrient loads include: 1) creation of a series of small detention ponds in the lower parts of these valleys, 2) trenching in woodchip denitrification walls across or within these valleys, or planting of deep-rooted trees in the riparian zones along these shores. It will be challenging to make such interventions sufficiently robust to withstand flood flow events in these relatively high-gradient, erosive stream channels.

Although, representing a much smaller nutrient load, improvements in wastewater management facilities at the camping ground along the northern side of Lake Camp should also be considered, primarily focussed on reducing N leaching.

5.1.6 Lake Emma

This lake experiences excessive levels of algae biomass (chlorophyll-*a*) and requires large reductions in nutrient levels. The target proposed for this lake requires a substantial reduction in nutrient losses from the catchment, i.e., 75% N and 67% P. This level of reduction would require reductions of 4.8 T N/yr and 0.15 T P/yr, or average catchment loss rates to reduce by ~2.3 kg N/ha/yr, and 0.07 kg P/ha/yr. The abundance of natural wetlands and predominance of low-producing grassland (Figure E-4) suggest this lake should be in better condition than indicated by recent water quality records. Bayer *et al.* (2021) note that this lake is likely to be impacted by internal loading as a result of summer stratification, resulting in seasonal deoxygenation of the bottom waters (hypolimnion) that drives release of iron-bound P from the lake sediments. This substantial internal load has the potential to substantially slow the lake response (improvement), should nutrient loadings from the catchment be reduced.

Fencing of wetlands and the inflow stream from Lake Roundabout may provide some benefit. However further investigation is warranted to better understand the flow and nutrient inputs to this lake and the role of lake internal nutrient loading before further recommendations are possible.

5.1.7 Lake Denny

Lake Denny is a relatively small lake which suffers from excessive levels of algae biomass (chlorophyll-*a*) and requires the largest reductions in nutrient levels of all the lakes considered in the present study, 83% for N and 93% for P. This equates to 5.8 T N/yr and 0.35 T P/yr. The LCDB5 landcover estimates (Appendix D) indicate that ~78% of its catchment is low-producing grassland, and more than 10% as natural wetland (Figure E-5). Such catchment characteristics would generally be associated with a lake with better water quality. However, the available landcover information does not appear to accurately reflect current practices within the catchment. Substantial areas of high producing grassland and possibly cropping are evident in the catchment north of the lake in recent Google Earth images.

The fringing wetlands which appear to intercept the main surface inflows to the lake are large compared with the lake itself, however they are small compared with the contributing catchment and may not be as extensive as suggested by the LCDB.

Potential interventions recommended include fencing inflowing streams to the north, which pass through high producing grasslands. These should have wide riparian margins/plantings, with strategic, targeted placement of constructed wetlands or denitrification walls following detailed survey and investigation. Better understanding of how surface and groundwater flows and nutrient inputs interact with existing natural wetlands are also required before further recommendations can be made.

5.2 Proposed implementation steps

1. Convene a collaborative, multi-landowner and stakeholder workshop informed by DOC, ECAN, and iwi goals to prioritise the most valuable lake and wetland areas in each catchment, and identify those that have the most realistic chance of being able to be sustainably maintained at or below maximum recommended nutrient loadings.
    - Focus actions to protect priority areas.
    - De-intensify farming activities or apply mitigations in areas where there is the highest risk of impacts, and greatest amelioration potential.
2. Manage livestock access and cropping in vulnerable zones.
    - Exclude all livestock, cultivation and fertilisation from a wide (i.e., ≥100m) buffer zone around all lakes, streams and prioritised wetlands.
    - Exclude livestock, other than sheep, from all wetlands, floodplains and streams greater than 1 m wide and/or 30 cm deep by 2025 to meet the RMA (stock exclusion) Regulations 2020 on all land with localised[6] slopes ≤ 10°.
3. Target sediment and P reductions primarily in ephemeral surface flows.
    - Identify key sediment and P source areas and down-stream hotspots in the catchment, particularly those upstream from intensive agricultural operations in lake and wetland catchments.
    - Implement most cost effective and efficacious edge-of-field or flow path interventions to intercept key inflows. These include:
        - Natural wetland protection.
        - Sediment traps and detainment structures.
        - Riparian buffers.
        - Constructed and modified wetlands.
4. Target N reductions primarily in shallow groundwaters.
    - Map expected downslope/stream paths of groundwater flow from all intensively managed agriculture towards lakes and prioritised wetlands.
    - Determine the depth zones of major groundwater flow paths into key lakes and wetlands and identify optimal zones of interception.
    - Implement most cost effective and efficacious edge-of-field or flow path interventions to intercept shallow groundwaters, subsurface drainage (if any) and surface run-off. These include:
        - Riparian buffers.

[6] i.e., pertaining to the immediate landscape, not averaged across adjacent areas of high country

- Constructed and modified wetlands.
- Denitrification walls or bioreactors.

5. Fast-track a sustainable solution for wastewater management at the Lake Clearwater fishing settlement and carry out an evaluation in all the lake catchments of wastewater facilities at all lodges and lake-side camping areas with toilet facilities.

   - Either provide for complete removal of sewage wastes from catchment (administered by council), or communal treatments system able to reliably provide advanced treatment with ≥ 70% N reduction.

6 Conclusions

The shallow lakes and fen wetlands in Ōtūwharekai originated in a nutrient-poor, inter-montane glacio-fluvial landscape and are sensitive to elevated nutrient inputs. They have been exposed to significant anthropogenic disturbance, particularly during the last 170 years of high-country farming, which has recently been intensified close to the lakes. These land use changes have created a legacy of nutrient and sediment stores within the lakes and their catchments, which will require significant mitigation over a prolonged period. Skeletal soils and deep porous aquifers favour predominant groundwater flows, with limited attenuation of nitrogen. Areas of natural palustrine, lacustrine and riverine wetlands occur in many catchments, which have significant potential to attenuate nutrient fluxes, but it is uncertain how much they interact with emerging groundwaters. In the wider, lower gradient valleys (e.g., Māori, Emma and Denny) we expect a higher proportion (around two thirds) of groundwater will be intercepted by streams before they enter the wetlands and lakes. In the larger lakes (Heron, Clearwater and Camp), a half or more of the wetland and lake inflows may be via direct groundwater springs and seeps, markedly reducing opportunities for interception and treatment.

The evidence of increasing lake nutrient levels and declining water quality indicates that these wetland areas are unable to effectively mitigate the current losses of nutrients from farming and other activities in these catchments. This may be due to a lack of effective interception and interaction with the groundwaters entering the lakes (as noted above), and/or the ambient temperatures that occur when ground and surface waters are mainly mobilised. Snow and ice-melt is likely to be highly pulsed and occur when temperatures are low, reducing potential rates of microbial transformation (e.g., denitrification) and plant uptake of nutrients. Furthermore, wetland plant tissue nutrient ratios and small-scale fertilisation experiments show that the natural fen wetlands in the catchment are themselves vulnerable to increased nutrient loadings. The practicality and cost-effectiveness of the catchment interventions to reduce catchment contaminant zones will depend on: i) the extent to which the contaminant attenuation capability of the natural wetlands is able to be utilised or enhanced to protect the lakes, or ii) whether protection of the natural wetlands themselves is prioritised.

The lack of specific information on the relative importance of surface- and groundwater flow paths in each catchment and their degree of interaction with wetlands, and riparian and littoral zones in the lakes limits our ability to fully determine the most appropriate mitigations and predict their likely efficacy. Targeted investigations of these factors would greatly help to resolve these issues and reduce uncertainty.

The most relevant edge-of-field and flow pathway interventions for application in Ōtūwharekai are livestock exclusion and riparian buffers, natural and constructed wetlands, denitrification walls and bioreactors, sediment traps and detainment bunds. However, it should be recognised that because of the relatively harsh montane environment (cold wind-swept winters and hot dry summers), most of these options will not be able to be implemented “off the shelf”. They will require adaption to ensure they are sustainable in the local environment, e.g., suitable riparian and wetland plant species and establishment techniques will need to be identified and tested.

To better target edge-of-field and flow-path interventions, and ensure they provide the environmental benefits sought, we recommend adopting an adaptive ‘learning by doing’ approach informed by:

- preliminary investigations to fill key knowledge gaps regarding groundwater and nutrient flow-paths, and
- detailed farm plans tailored to the landscape in consultation with farmers, iwi and experienced land management officers.

7 Acknowledgements

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Appendix A Location of the Ōtūwharekai Lakes

Figure A-1: Location of eight Ōtūwharekai lakes. Source Burton *et al.* (2022).

Appendix B Recommendations from the cultural health assessment

Recommendations for future management sourced directly from Te Rūnanga o Arowhenua *et al.* (2010)

1. That all waterways which continue to be important for food gathering are managed and enhanced for food gathering quality into the future.

2. That increased protection and enhancement of waterways through the development of native riparian and wetland buffer zones be investigated and implemented.

3. Greater advocacy, rates relief and other economic methods for the protection and enhancement of native riparian and wetland buffer zones and vegetation patches in currently poor or un-vegetated or un-fenced areas on private land.

4. Specific restoration, pest and weed eradication and exotic species control in and around all lakes, including the use of tī kouka, houhi, kowhai, maukoro, mikimiki, beech and aruhe and other native plants that prove to compete well with, or can be planted underneath willow and other exotic species invading lakes and wetlands. This should consider the removal of pest fish from specific areas. The following lakes and sites should be a priority: - Ōtūwharekai (East) / Lower Māori Lake - Kirihonuhonu / Lake Emma - The Oliver Stream area of Ō Tū Roto / Lake Heron - The Swin river access area of Ō Tū Roto / Lake Heron - Te Puna a Taka / Lake Clearwater; and - Ōtautari / Lake Camp.

5. Specific measures to control siltation/sedimentation and *E. coli* contamination of Ōtūwharekai (East) / Lower Māori Lake and further protection of Ōtūwharekai (West) / Upper Māori Lake, including the potential purchase of surrounding land, the control of exotic species, and the development of better buffers, particularly around the road edge corner of Lower Māori Lake and incoming water ways of both lakes.

6. Consideration for the complete and ongoing removal of exotic fish from the Māori Lakes and work towards making the lake complex a native fish only area.

7. Further tuna/eel monitoring surveys and investigation to understand the potential of an annual cultural harvest, particularly at the Māori Lakes.

8. Further investigation and control of human and agricultural pollution at Te Puna a Taka / Lake Clearwater and the Oliver Stream area of Ō Tū Roto / Lake Heron.

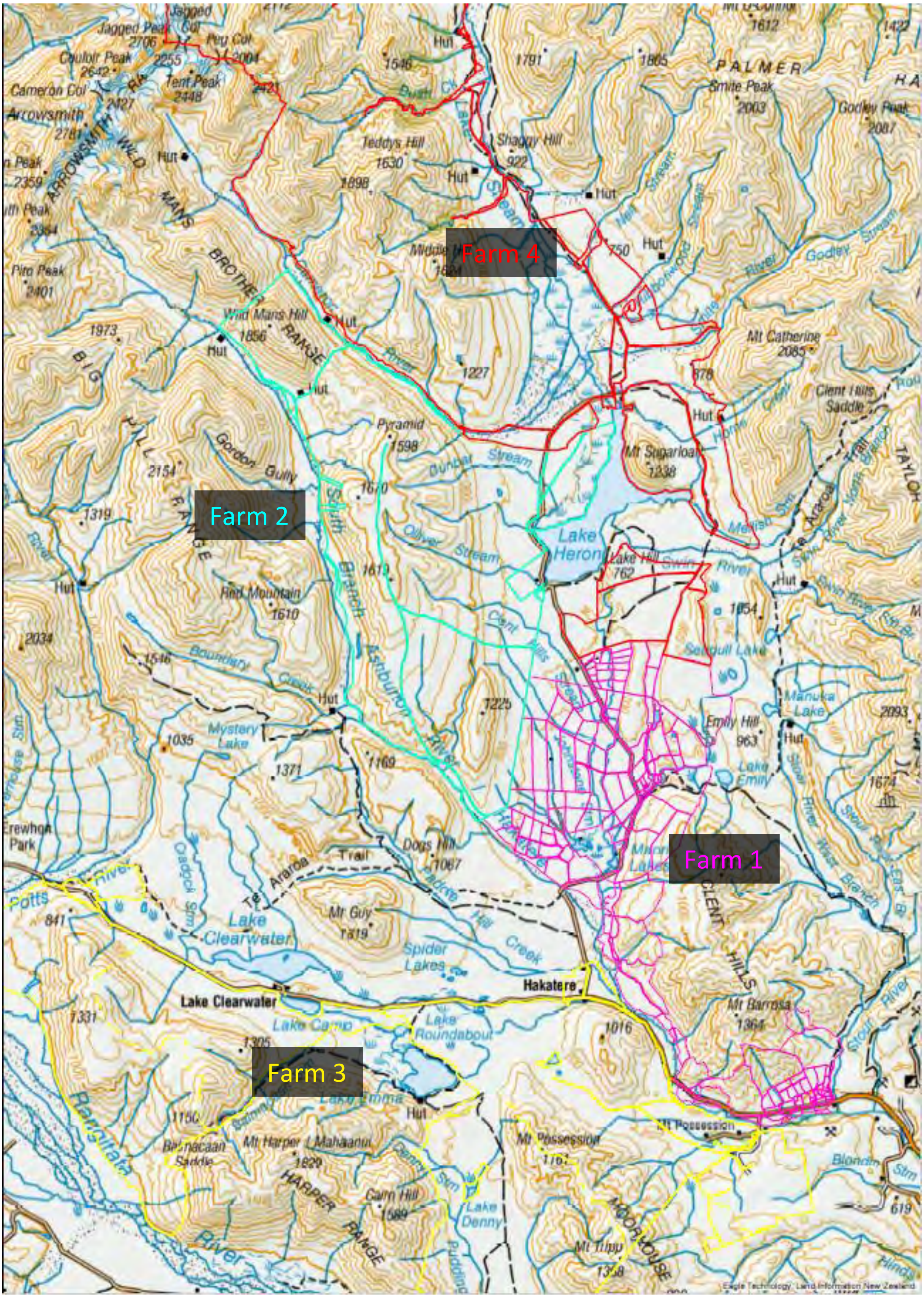
9. Support for future wānanga and hui to reconnect tangata whenua with the Ōtūwharekai / Ashburton Lakes Area, particularly around future interpretation and cultural harvest opportunities of both tuna, raupō (for mokihi), hua kaki anau (black swan eggs) and other mahinga kai.

10. Investigation into the habitat requirements of, and future possibilities (including specific sites), for the reintroduction of eastern buff weka into the area.

11. Greater research into the impacts of, and solutions for, treating and dealing with non-point and point source pollution of waterways in the area.

12. Continued regular monitoring, including cultural assessments, to understand the success, or otherwise, of future management and development of the catchment

Appendix C Location of the four farms in the lake catchments



Appendix D Summary of lake catchment landcover in 2018

Based on LCDB 5 (Landcare Research, 2018).

Row Labels	Lake Heron		Lake Clearwater		Lake Camp		Lake Denny		Lake Emily		Lake Emma		Maori Lakes	
	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%	ha	%
Lake or Pond	712.97	6.4%	208.26	4.5%	44.33	7.3%	6.24	0.3%	20.85	1.3%	189.49	6.5%	39.37	0.5%
Herbaceous Freshwater Vegetation	369.58	3.3%	48.52	1.1%		0.0%	10.49	0.6%	98.77	6.3%	167.26	5.7%	238.01	2.9%
Alpine Grass/Herbfield	6.12	0.1%												
Sub Alpine Shrubland	255.4	2.3%									14.32	0.5%		
Matagouri or Grey Scrub	153.65	1.4%	35.47	0.8%	18.65	3.1%	62.97	3.4%	0.23		35.87	1.2%	90.43	1.1%
Manuka and/or Kanuka	56.14	0.5%					6.19	0.3%	0.15		11.99	0.4%	4.48	0.1%
Tall Tussock Grassland	3 320.43	29.7%	2 713.07	59.2%	3.09	0.5%	427.44	23.0%	0.15		746.65	25.7%	1 399.64	17.0%
Low Producing Grassland	2 332.97	20.9%	1 153.81	25.2%	435.72	71.6%	59.47	3.2%	1 223.75	77.6%	403.92	13.9%	5 805.44	70.4%
Depleted Grassland	58.65	0.5%	417.17	9.1%	92.79	15.3%	635.13	34.2%	0.46		1 096.74	37.7%	155.85	1.9%
High Producing Exotic Grassland	946.14	8.5%					604.8	32.6%	118.48	7.5%	5.37	0.2%	296.57	3.6%
Mixed Exotic Shrubland											13.55	0.5%	14.82	0.2%
Broadleaved Indigenous Hardwoods	3.09												7.58	0.1%
Deciduous Hardwoods	4.46										1.62	0.1%	8.73	0.1%
Built-up Area (settlement)			2.96	0.1%	5.21	0.9%								
Exotic Forest	25.68	0.2%	2.18		8.5	1.4%					1.40	<0.1%	31.5	0.4%
Gorse and/or Broom or Fernland	3.24										19.57	0.7%		
Gravel or Rock or Landslide	2 924.11	26.2%					44.93	2.4%	2.86	0.2%	202.34	7.0%		
Surface Mine or Dump	0.63													
Grand Total	11 173.28		4 581.42		608.29		1 857.65		1 577.03		2 910.1		8 250.47	

Appendix E Lake catchment landcover maps

Maps are based on 2018 landcover classes in LCDB5 .

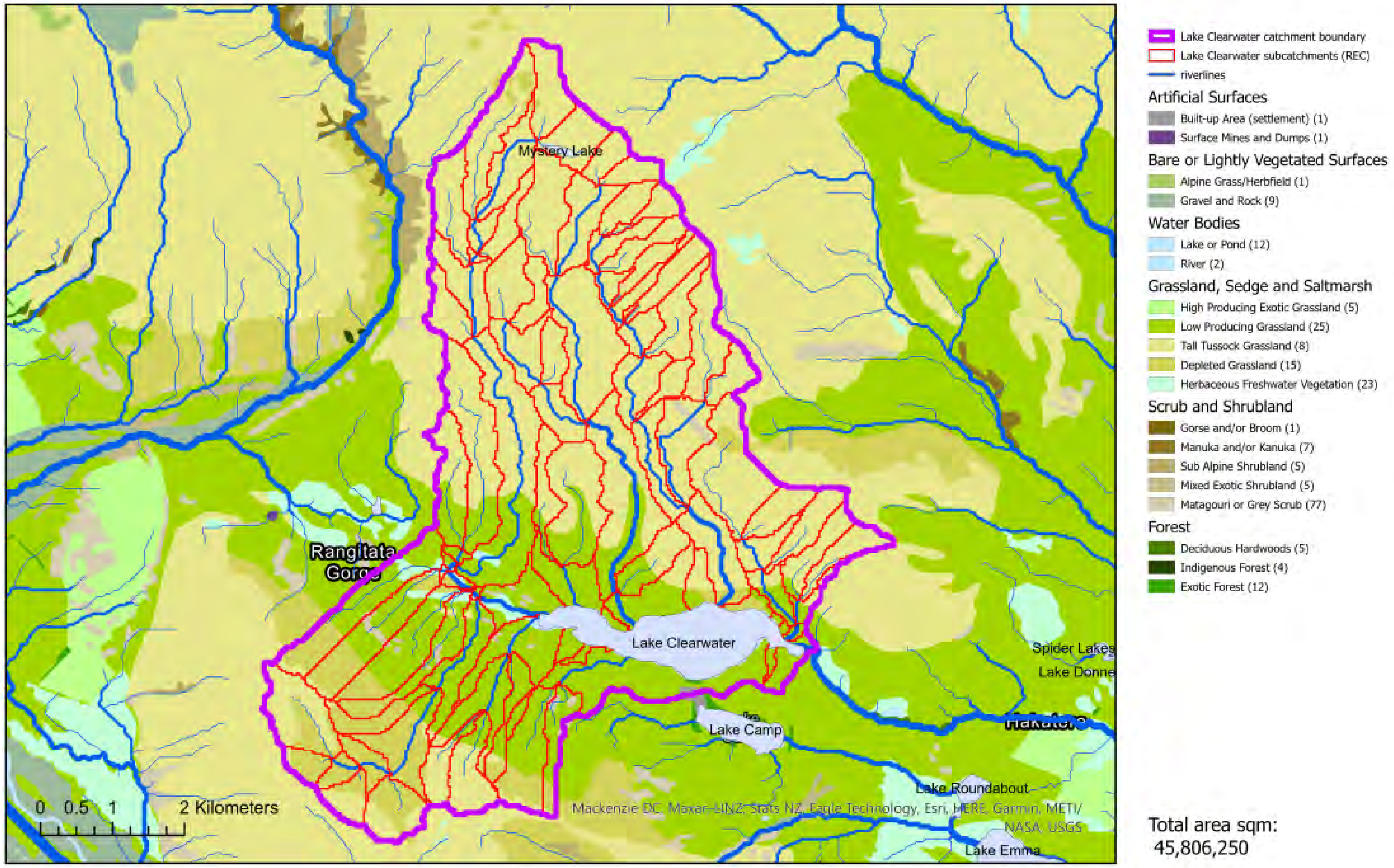


Figure E-1: Catchment map for Lake Clearwater.

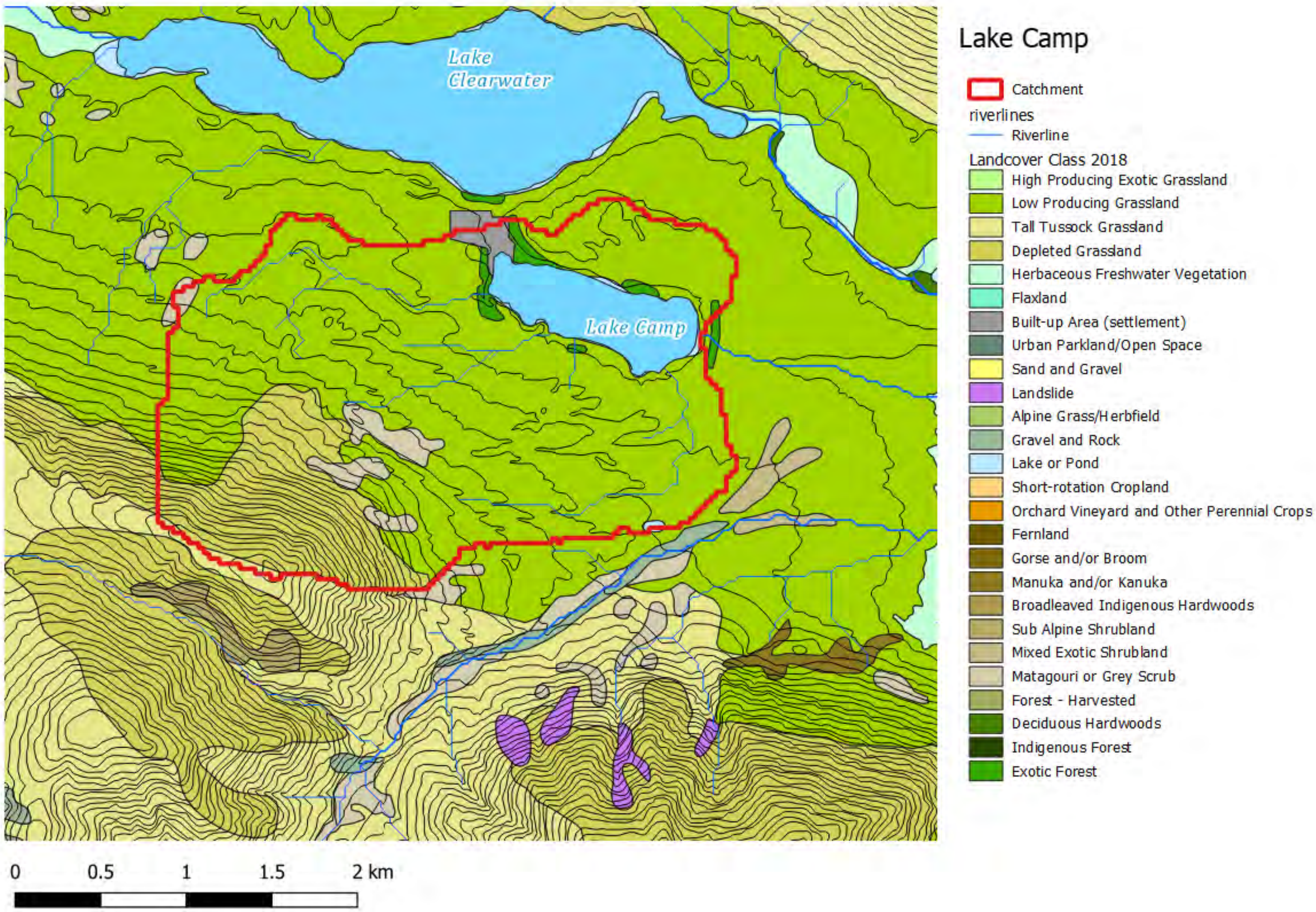


Figure E-2: Catchment map for Lake Camp.

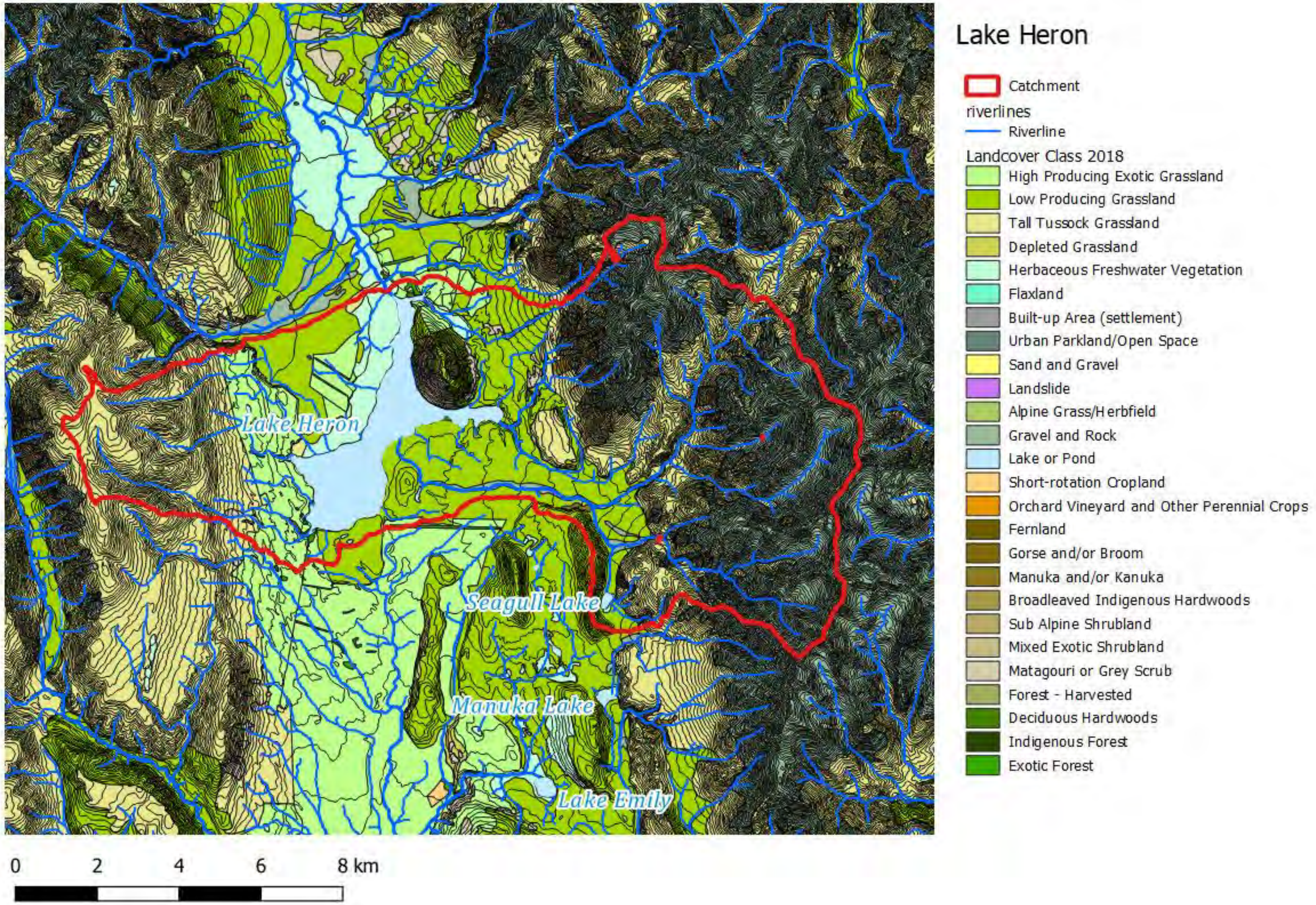


Figure E-3: Catchment map for Lake Heron.

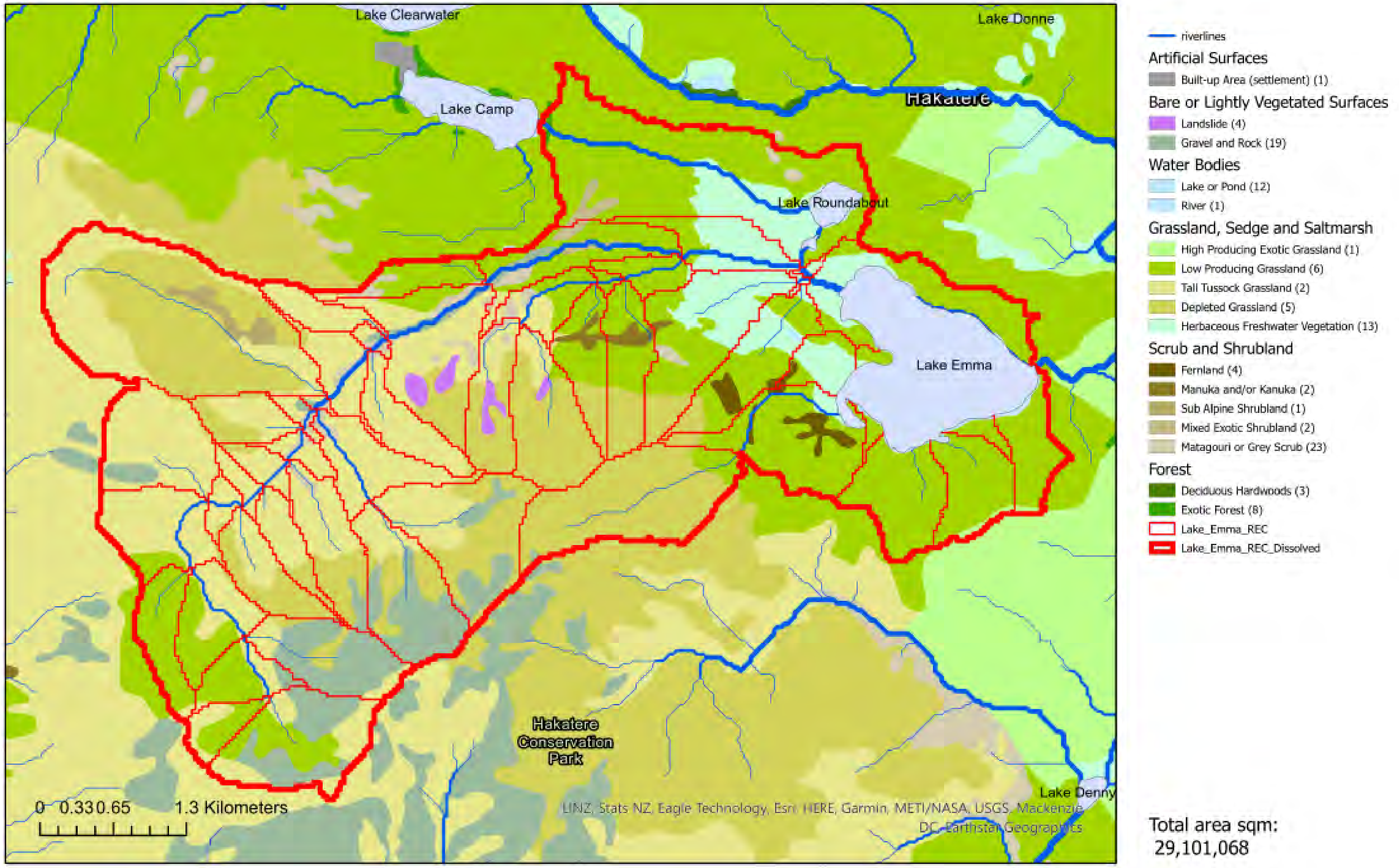


Figure E-4: Catchment map for Lake Emma.

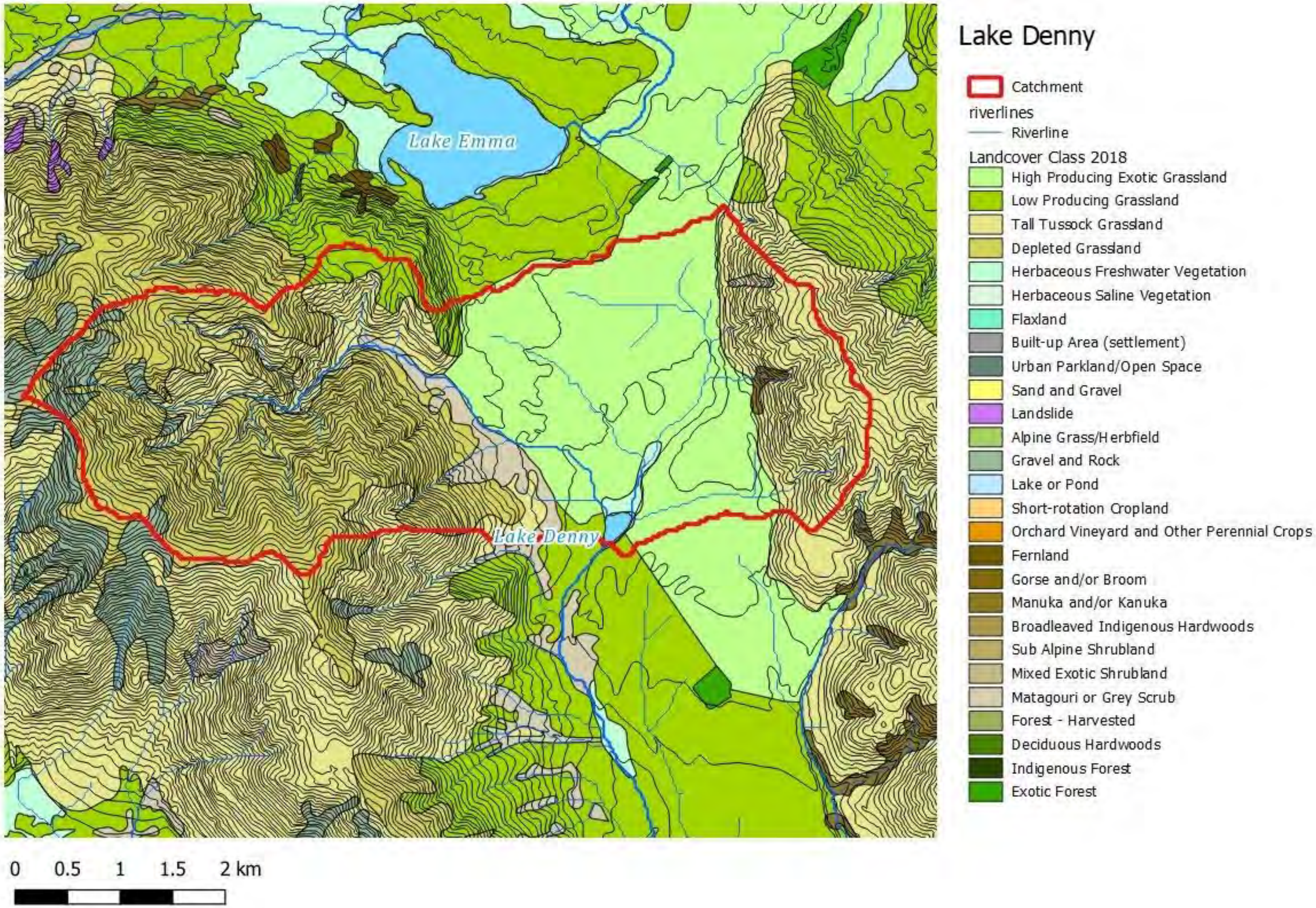


Figure E-5: Catchment map for Lake Denny.

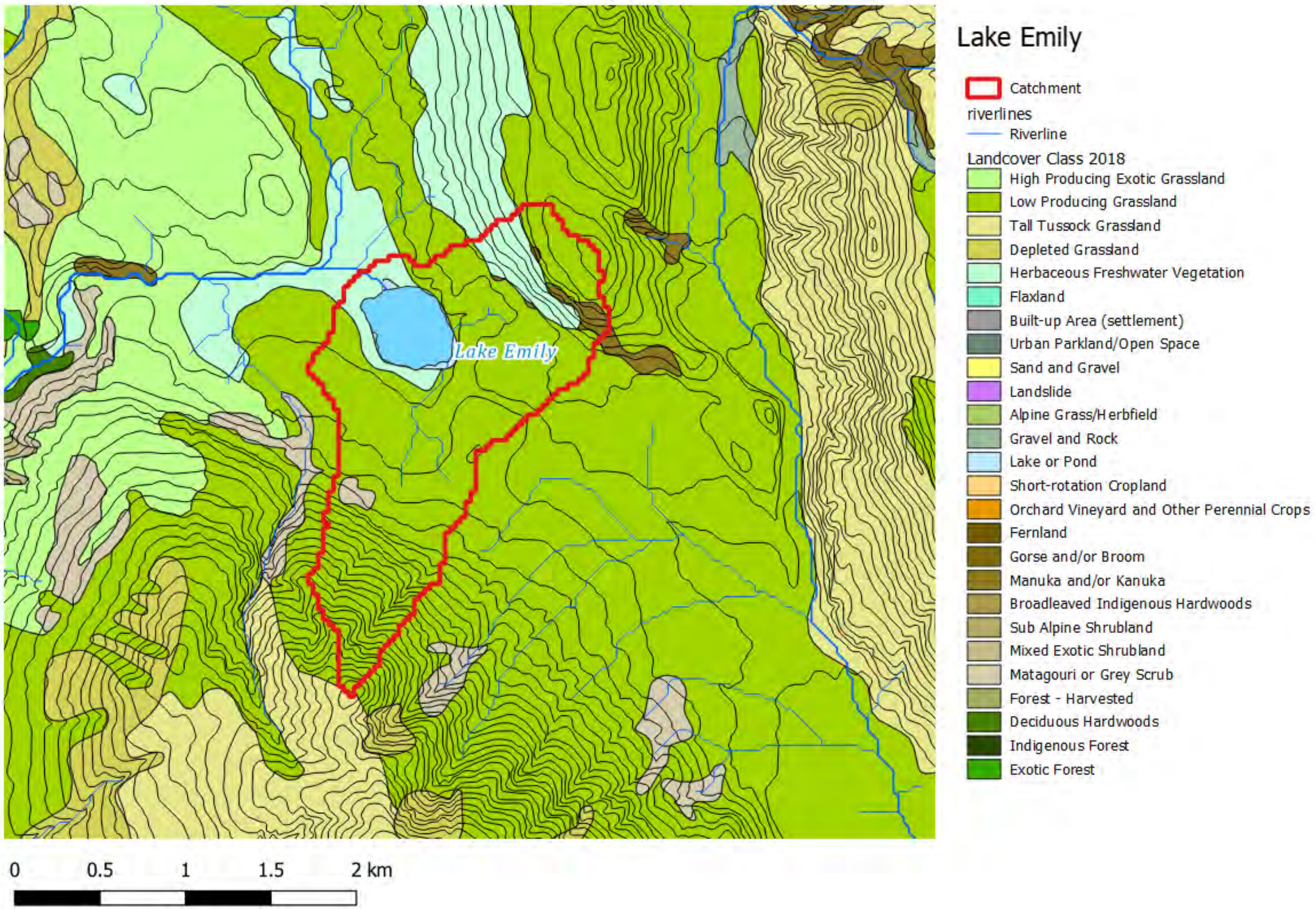


Figure E-6: Catchment map for Lake Emily.

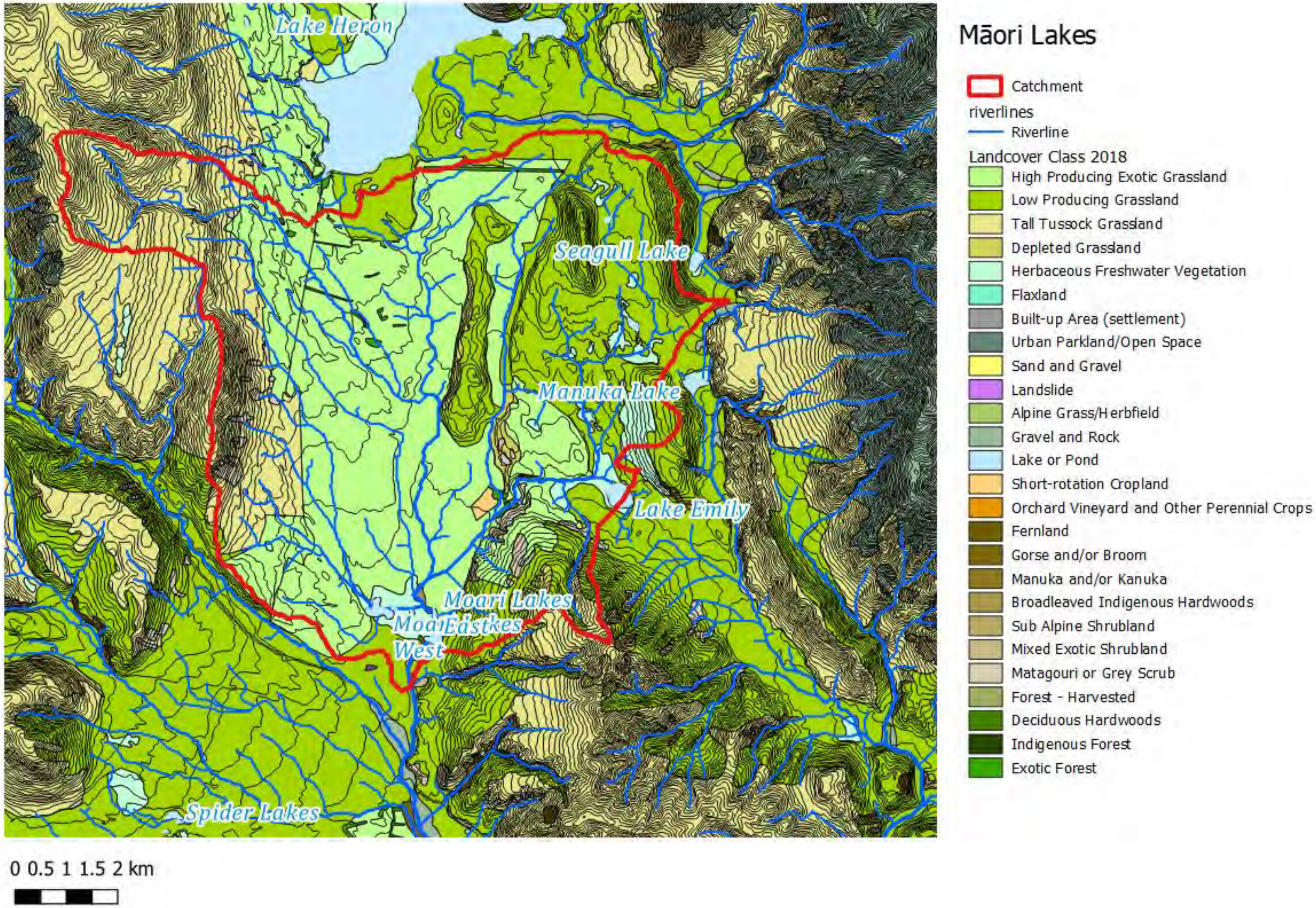


Figure E-7: Catchment map for Māori Lake East and West.

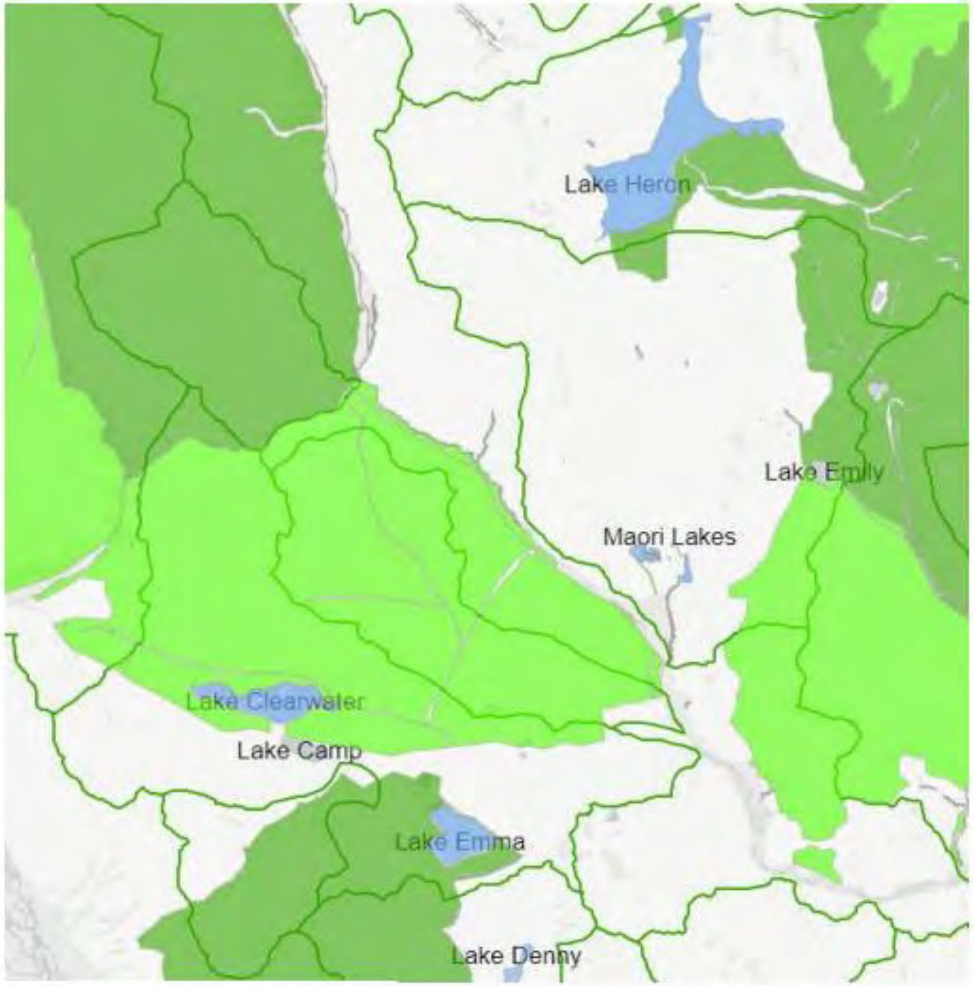


Figure E-8: Map showing public conservation land in the Ashburton Lakes Basin. (Bayer and Meredith 2020). Green lines are catchment boundaries, land blocks in pale green are Stewardship Areas, and dark green represents Conservation Park.

Appendix F Aerial images of recent agricultural activity in the Lake Heron catchment

Figure F-1: Lake Heron catchment showing areas of agricultural development in March 2019. Google Earth Image.

Figure F-2: Lake Heron catchment showing areas of agricultural development in February 2006. Google Earth Image.

Appendix G Agricultural activity in the Lake Clearwater catchment

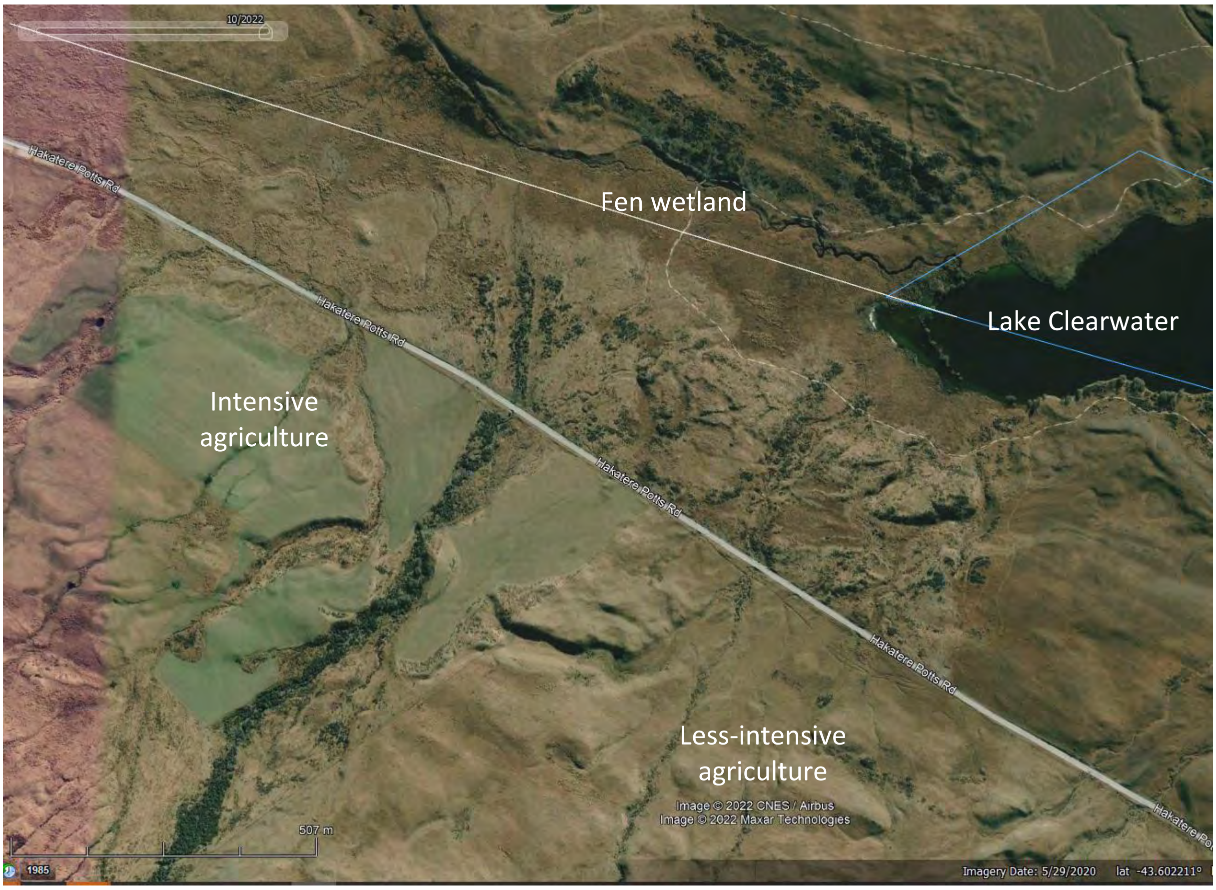


Figure G-1: Areas of more intensive agricultural activity to the southwest of Lake Clearwater in October 2020. These areas, visible as greener smooth areas which have transitioned from fodder cropping to pasture, drain NE towards Hakatere Potts Rd into an area of fen wetland which flows to Lake Clearwater. From Google Earth Imagery. Note: browner strip on right of image is from May 29 2020, remainder from October 2020.

Figure G-2: Same areas as above on December 23 2018, showing areas of fodder cropping. Fodder cropping areas show as bright green. From Google Earth.

Figure G-3: Same areas as above on 19 January 2018, showing new areas under cultivation. From Google Earth.

Figure G-4: Same areas as above on 2 September 2006, before agricultural intensification. From Google Earth

Figure G-5: Fodder cropping areas draining to wetlands to southwest of Lake Clearwater viewed from across the valley (Photo: C.C. Tanner, Feb 2012)

Figure G-6: Aerial view looking southwest towards wetlands to west of Lake Clearwater at high water level. (Photo: Marty Flannagan, NIWA, Nov 2012).

Figure G-7: Intensive pasture southwest of Lake Clearwater. (Photo: Chris Tanner; Feb 2013).

Figure G-8: View south-east from within the Lake Clearwater wetlands. Note the bright green pasture areas on toe-slopes of hills in the background that drain down into these areas. The copper brown coloured sedge is bogrush (*Schoenus pauciflorus*) and the green one, predominantly a form of purei (*Carex diandra*). (Photo: Chris Tanner, NIWA, Feb 2013).

Figure G-9: The yellow-flowered monkey musk growing in Whiskey Stream as it flows from under the road culvert from cropping and improved pasture areas. Abundant monkey musk (*Erythranthe guttata*, formerly *Mimulus moschatus*), like watercress (Cox and Rutherford, 2012; Fernandez-Going *et al.* 2013), is considered as an indicator of nitrate-rich flowing waters. (Photo: Chris Tanner, Jan 2011).

Appendix H Development of the Lake Clearwater Settlement

Figure H-1: Lake Clearwater fishing hut settlement February 1964. Retrolens.nz image (snip from Crown 1580-3730-14).

Figure H-2: Lake Clearwater fishing hut settlement April 1966. Retrolens.nz image (snip from Crown 2019-C-13).

Figure H-3: Lake Clearwater fishing hut settlement February 1982. Retrolens.nz image (snip from Crown 8390-E-16).

Figure H-4: Lake Clearwater fishing hut settlement February 1986. Retrolens.nz image (snip from Crown 8595-A-24).

Appendix I Sewage nutrient load calculations for Lake Clearwater Settlement

Percentage occupancy	Huts				Campground					N load per person [2]	P load per person [2]	Total source load		Estimated load to lake	
	No Huts[1]	Average number of occupants	Days per year	Hut person days	No Camp sites [1]	Average number of occupants	Days per year	Camping person days	Total person days	(g N /person /day)	(g P /person /day)	TN load (kg/yr)	TP load (kg/yr)	TN load [3] (kg/yr)	TP load [4] (kg/yr)
10%	195	4	36.5	28 470	35	4	36.5	5 110	33 580	12	3.5	403	118	343	10
30%	195	4	110	85 800	35	4	90	12 600	98 400	12	3.5	1 181	344	1 004	29

[1] From Marias (2021).
[2] Based on generalised estimates for New Zealand on-site wastewater by Potts and Ellwood (2000).
[3] Assumes no significant reduction in the pit (<5% reported for septic tanks, Lusk *et al.* 2017a) and ~15% reduction in N leached based on average for septic tank discharge fields (Siegrist *et al.* 2000)
[4] Assumes 15% average P retention in sludge (average for septic tanks, Lusk *et al.* 2017a) and 90% reduction in P leached during soil passage of septic tank effluent (minimum observed after 1 m infiltration in gravelly silt soils by Gill *et al.* 2009a).

Appendix J Matagouri presence in ephemeral stream channels

Figure J-1: Aerial image of part of catchment draining to Lake Camp showing matagouri and other woody shrubs along stream channels. Lake Camp situated in top right corner of photo. Note matagouri (tūmatakuru, *Discaria toumatou*) is a native non-leguminous nitrogen-fixing species.