Fish in Stormwater Wetlands: A Pilot Study

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EXECUTIVE SUMMARY

This report describes results of a pilot study of fish communities in stormwater wetlands owned by Christchurch City Council. The study involved measuring basic habitat parameters, dissolved oxygen, temperature, and fish at a total of 21 wetland sites, 20 of which were stormwater wetlands. Fish were caught at all sites except for one site that had only 4 m^2 of surface water present. Six fish species were caught across the wetland sites, with native shortfin eel (*Anguilla australis*) found at all sites fish were caught. The prevalence of native fish at the sampled sites was notable, particularly given the presence of invasive rudd (*Scardinius erythrophthalmus*), tench (*Tinca tinca*) and goldfish (*Carassius auratus)* in other non-stormwater wetlands in the city.

Key design features of stormwater wetlands that determine fish presence include whether surface water is present, habitat considerations, and provision of fish passage into the wetland. Our observations indicate that even temporary surface water can provide fish habitat, with eels moving in to feed on terrestrial invertebrates when water levels are elevated and moving out as water levels drop. Flow restrictions at wetland outlets can act as migratory barriers, restricting the range of fish species that can enter a wetland. Culvert pipes were amongst the most common type of structures assessed and they ranged from presenting low to very high risk to fish passage. Low risk culverts were typically relatively short (tens, rather than hundreds of metres long), and had adequate water depth and slow velocities to provide passage for a range of fish species and life stages. Culvert attributes contributing to their classification as high or very high risk structures included very long lengths, shallow depths, swift velocities, and a vertical drop, or perch, at the outlet.

High water temperatures are likely limiting to sensitive fish species such as inanga (*Galaxias maculatus*) in many wetlands, while low dissolved oxygen concentrations may adversely affect all fish species in some wetlands. Limited water quality and sediment quality data indicate that concentrations of stormwater contaminants such as metals in Christchurch stormwater wetlands are sufficiently high to have adverse effects on the fish communities present.

We conclude it is likely that fish are being trapped in some stormwater wetlands in Christchurch. Traps may be in the form of poor water quality affecting fish fitness and community composition, or in the form of a physical trap, by fish being stranded or caught in piped networks.

We recommend that Christchurch City Council reviews the planning and design of stormwater wetlands to consider potential negative effects of stormwater wetlands on fish. This should include decisions around whether fish passage is provided for all new stormwater wetlands and whether fish passage should be altered for existing stormwater wetlands. Other recommendations include further sampling of water quality, sediment quality, and fish communities, and public education regarding invasive fish species and associated surveillance monitoring. Lastly, we recommend further study of fish in stormwater wetlands elsewhere in the country, given the national relevance of the issues raised in this pilot study.

1. INTRODUCTION

Urban stormwater wetlands include a range of stormwater treatment facilities, including first flush basins, retention basins, ponds, and polishing wetlands. The primary purpose of stormwater wetlands is to alleviate downstream flooding risk, by detaining peak flows, and to improve water quality, by removing stormwater contaminants such as metals. However, stormwater wetlands may also provide habitat for native fish species and some wetlands have been specifically designed to allow fish entry (e.g., Ho *et al.* 2018). The presence of native fish adds to the biodiversity of stormwater wetlands, but fish are in an environment designed to trap pollutants, which could reduce the health of the resident fauna (Zhang *et al.* 2020). Thus, stormwater wetlands may be 'ecological traps' (Tilton 1995) that attract native species, but reduce their health and fitness.

Published research regarding the fish fauna of urban stormwater wetlands is restricted to studies on the dwarf galaxias (*Galaxiella pusilla*) in Melbourne, Australia (Hale *et al.* 2019). The research found lower survival of native dwarf galaxias in stormwater wetlands than in nearby non-stormwater wetlands, and fish had delayed ovary maturation (Hale *et al.* 2018). Further research found that invasive mosquitofish (*Gambusia holbrooki*) heavily predated the native dwarf galaxias and that mosquitofish are often abundant in stormwater wetlands in Melbourne (Brown *et al.* 2018; Hale *et al.* 2019). The Australian dwarf galaxias is in the Galaxiidae family that includes many New Zealand native species, including inanga (*G. maculatus*). The findings from the Australian studies may therefore be equally applicable to New Zealand, but there have been no studies on the fish fauna of New Zealand stormwater wetlands to date.

This report describes the results of a pilot study investigating the fish communities of urban stormwater wetlands in Christchurch, New Zealand. The objective of the study was to answer the following questions, posed by Christchurch City Council (Council):

1. What fish are present in Council stormwater wetlands?
2. Are there design features that determine fish presence in stormwater wetlands?
    - If so, what are these features, and could they be incorporated into future designs to specifically include or exclude fish?
3. How successful are stormwater wetland designs that specifically exclude fish or provide for fish passage?
4. Are there any preliminary indications that fish communities in Council stormwater wetlands are adversely impacted by water quality, cyanobacteria blooms, fish passage, or habitat?
5. Are fish being trapped in Council stormwater wetlands?

Answering questions 2–5 could involve a comprehensive study that includes a multi-year programme of experimentation, which was not within the study scope. That is why the project is considered a pilot study, focussed on sampling fish communities and providing some guidance and preliminary answers to the above questions.

2. METHODS

2.1. Sampling Sites

A total of 21 wetlands were sampled for this study, including 20 urban stormwater wetlands, designed to attenuate and/or treat stormwater, and one urban pond (Site 20, Beckenham), which was not designed to receive stormwater[1] (Table 1, Figure 1). Sampling sites included stormwater wetlands in all the major catchments in Christchurch city. Sites were selected to include a range of wetland sizes, ages, and configurations, and were chosen in consultation with Council stormwater engineers.

Table 1: Wetlands sampled as part of this study. See table footer for dual language catchment names.

Site Number	Site Name	Catchment	Online?[1]	Easting (NZTM)	Northing (NZTM)
1	Sparks Wetland	Ōpāwaho	No	1567253	5175232
2	Sparks First Flush	Ōpāwaho	No	1566846	5175475
3	Bullers	Ōtākaro	Yes	1570878	5184743
4	Quaifes	Huritini	No	1563383	5174343
5	Wigram	Ōpāwaho	Yes	1565886	5177590
6	Ngā Puna Wai	Ōpāwaho	Yes	1565400	5177501
7	Eastman	Ōpāwaho	No	1566591	5174720
8	Halswell Downs	Ōpāwaho	No	1566403	5174419
9	Portlink	Ōpāwaho	No	1575042	5176765
10	Charlesworth	Linwood Canal	Yes	1575741	5178103
11	Clare Park	Ōtākaro	No	1574083	5184737
12	Prestons	Pūharakekenui	No	1572809	5187940
13	Burlington	Pūharakekenui	Yes	1570413	5186776
14	Alpine View	Ōtākaro	Yes	1573601	5186597
15	Ryman	Ōtākaro	No	1571146	5184337
16	Arthur Adcock	Piped direct to sea	No	1576299	5186033
17	Knights	Huritini	No	1562674	5175210
18	Douglas Clifford	Ōpāwaho	Yes	1565514	5175775
19	Spring Grove	Pūharakekenui	No	1570332	5188044
20	Beckenham	Ōpāwaho	No	1571546	5176717
21	Te Oranga Waikura	Ōpāwaho	No	1573668	5178723

Notes: [1] Online = wetlands with open waterways flowing into them; offline = upstream catchment is piped. Dual language catchment names: Ōpāwaho - Heathcote River; Ōtākaro - Avon River; Huritini - Halswell River; Pūharakekenui - Styx River.

[1] While Beckenham Pond was not designed for stormwater attenuation or treatment, it may receive some road runoff via a sump on Eastern Terrace. However, it is not referred to as a stormwater wetland in this report, as it is an ornamental pond by design.

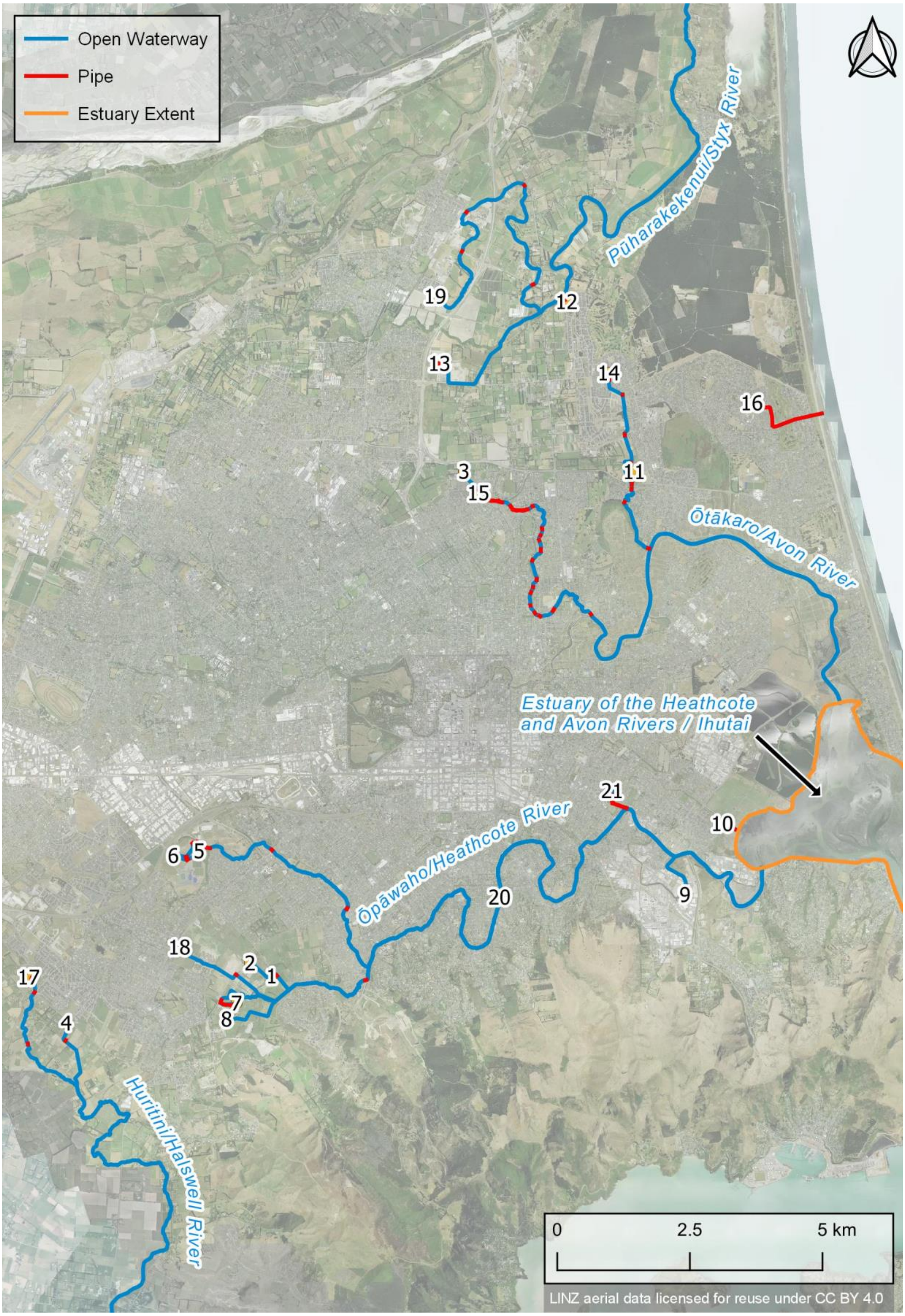

Figure 1: Stormwater wetland sampling sites and downstream waterways.

2.2. Field Methods

Sampling commenced on 9 March 2022 and was initially planned to be completed by the end of April 2022, to avoid negative impacts of lower temperatures on fish capture rates in the cooler months (Joy *et al.* 2013). However, after sampling nine sites, sampling for the remaining sites was delayed. The delayed sampling was in response to an event of high fish mortality at one site, which was suspected to relate to low dissolved oxygen levels. With the remaining sampling carried out over a cooler time of the year, the risk of potential fish deaths associated with high temperatures and low dissolved oxygen concentrations was reduced. All sampling was completed by 25 May 2022. The only exception to the March to May 2022 sampling window was Site 21 (Te Oranga Waikura), where the fish community had previously been sampled on 28 April 2021; all other parameters at this site were measured between March and May 2022.

2.2.1. Habitat

At each sampling site we took representative photographs, made notes on general habitat conditions, and drew a site map. Habitat measurements were made along three transects, each extending from the water's edge to the centre. For stormwater wetlands with multiple wetland cells, ponds, or basins, we sampled a single cell of the complex (e.g., a wetland cell or first flush basin), which was always the same cell that was sampled for water quality and fish. Water depths were measured along each transect, with the number of measurements varying depending on wetland size (from 6 to 12 measurements per transect; mean of 8).

At each transect, habitat measurements were taken at two points: one close to the water's edge and one near the centre of the wetland cell, pond, or basin. At each of the two points per transect, the following measurements were made:

- Visual estimate of substrate size, using the following size classes: silt/sand (<2 mm); small gravels (2–16 mm); large gravels (16–46 mm); small cobbles (64–128 mm),large cobbles (128–256 mm), boulders (256–4000 mm) and bedrock/hard surfaces (>4000 mm).
- Shade was measured using a spherical densiometer.
- Visual estimate of bed cover with deposited fine sediment (<2 mm diameter).
- Visual estimate of macrophyte cover (emergent and total) and species composition.
- Visual estimate of periphyton cover and composition.
    - Periphyton categories were adapted from those of Biggs & Kilroy (2000) and included: thin films (<0.5 mm thick), medium mats (0.5–3 mm thick), thick mats (>3 mm thick), short filamentous algae (<20 mm long) and long filamentous algae (>20 mm long).

2.2.2. Potential Fish Barriers

Wetland inlet, outlet, and flow control structures were assessed for fish passage using NIWA's Fish Passage Assessment Tool (FPAT). Structures were assigned a qualitative risk to fish passage score, ranging from very low to very high, following the qualitative risk classes and descriptions provided in Table F-2 of Franklin (2022). Structures that were unable to be assessed, due to lack of accessibility or uncertainty regarding their operation, were assigned a risk class following discussion with Council engineers.

2.2.3. Water quality

Dissolved oxygen (DO) and temperature were recorded at 15-minute intervals for a minimum 24-hour period at each wetland, using a calibrated YSI EXO3 multiparameter sonde. Each logger was attached to a steel 'waratah' post and positioned so that the sensors were located mid-depth in the deepest wadeable section of the wetland (maximum wadeable depth was approximately 1 m). We also took spot measurements of DO and water temperature using a handheld meter. However, we chose not to present these data as we found they were unrepresentative of temperature or DO data recorded by the data sondes.

2.2.4. Fish

The initial plan was to sample the fish community using overnight trapping with five unbaited, fine mesh fyke nets[2] per sampling site over March and April 2022. This sampling method commenced on 8/3/2022 and was used at nine sites, but ceased when large numbers of dead eels were found in fyke nets at Site 18 (Douglas Clifford) on 24/3/2022 (despite air gaps being present at the top of each net). Very low oxygen levels at the site prompted concern that fine mesh fyke nets may be harmful at low oxygen sites, due to reduced flow-through and the large numbers of fish they can catch. Therefore, we subsequently altered the fishing methods by halting fish sampling at all sites until DO monitoring data had been reviewed. Subsequently, five unbaited fine mesh fyke nets were deployed overnight to sample the fish community at sites with DO concentrations above 2 mg/L throughout the 24-hour monitoring period. Five unbaited coarse mesh fyke nets[3] and eight unbaited Gee minnow traps (mesh size 6.4 mm) were deployed overnight at sites with DO concentrations that fell below 2 mg/L during the monitoring period. Fishing of the remaining sites resumed on 27 April 2022 using the updated methods. The only exception was Site 21 (Te Oranga Waikura), where fishing data was available from previous sampling undertaken using unbaited fine mesh fyke nets on 28/4/2022. All fish sampling was completed by 25/5/2022.

Fewer fyke nets and minnow traps were deployed at sites with very shallow water depths and/or area to sample. These sites included Site 9 (Portlink), where 4 coarse mesh fykes and 7 minnow traps were deployed, and Site 17 (Knights), where only 1 minnow trap was deployed. Site 18 (Douglas Clifford) was originally sampled using 5 fine mesh fykes, but it was re-sampled later because it was suspected the original catch was influenced by high water levels due to recent rainfall. Water levels were too shallow on subsequent fishing to deploy fyke nets, so only 8 minnow traps were deployed.

The catch efficiency of fine mesh fykes versus coarse mesh fykes plus minnow traps was compared at Site 1 (Sparks Wetland), Site 5 (Wigram) and Site 21 (Te Oranga Waikura). Fine mesh fyke net sampling occurred on 9/3/2022 at Site 1, 27/4/2022 at Site 5, and 28/4/2021 at Site 21. Coarse mesh fyke and minnow trapping occurred over 5–12/5/2022 for all three sites. The relative catch efficiency of fine mesh fyke nets and minnow traps was compared at four sites (Sites 3, 10, 12, and 13), where five fine mesh fykes and eight minnow traps were deployed at each site. See Table 2 for a summary of fishing methods used at each site.

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[2] 3 mm mesh (trap and leader); trap opening 600 mm wide x 550 mm high; single leader 4.8 m long x 550 mm high; net 3.5 m long (excluding leader); two internal traps, separated by 22 mm mesh plastic grid to exclude larger fish. Similar to prototype described by Joy *et al.* (2013).

[3] 15 mm mesh, measured knot to knot (trap and leader); trap opening 600 mm wide x 500 mm high; single leader 2.8 m long x 500 mm high; net 2.8 m long (excluding leader); one internal trap, with no exclusion grid.

All caught fish were identified and counted. Lengths were measured for at least the first 50 individuals of each fish species caught at each site. Once 50 fish were measured, the remainder of that species was measured until the processing of that net was complete. All other individuals in the remaining nets were only counted. After processing, all fish were returned immediately to the waterways they were caught in.

Table 2: Fishing methods for the 21 wetlands sampled. Method types are as follows: F = fine mesh fyke nets, C = coarse mesh fyke nets, M = minnow traps, + = methods used on the same sampling date, and commas separate methodologies used on different sampling dates.

Site No.	Site Name	Method Type
1	Sparks Wetland	F, C+M
2	Sparks First Flush	F
3	Bullers	F+M
4	Quaifes	F
5	Wigram	F, C+M
6	Ngā Puna Wai	F
7	Eastman	F
8	Halswell Downs	F
9	Portlink	C+M
10	Charlesworth	F+M
11	Clare Park	C+M
12	Prestons	F+M
13	Burlington	F+M
14	Alpine View	C+M
15	Ryman	F
16	Arthur Adcock	C+M
17	Knights	M
18	Douglas Clifford	F, M
19	Spring Grove	C+M
20	Beckenham	C+M
21	Te Oranga Waikura	F, C+M

2.3. Data Analysis

Habitat transect data (depths, periphyton, shade, substrate composition, and macrophytes) were averaged for each transect and then the average of the three transects was taken, for comparison amongst sites. Only one transect fitted into the small area of water present at Site 17 (Knights), therefore habitat means were based on a single transect at that site.

A fish barrier risk score was ascribed for each sampling site, based on the lowest risk barrier assessed that a fish would have to navigate to enter the wetland from downstream of the wetland complex. The fish barrier risk score for each wetland did not consider other potential barriers downstream of the wetlands.

Dissolved oxygen and temperature monitoring data from data loggers were trimmed to a 24-hour period for each site, prior to plotting and preparing summary statistics. Spot water quality measurements were tabulated.

We used GIS data from the Council's open source 3-Waters Asset Database to calculate a variety of variables that may potentially determine fish presence in a wetland. Distance to coast was calculated by tracing the path of each stormwater wetland to its coastal outlet along the WaterCourse and StormWater GIS layers (including piped and open waterways). Distance to coast was calculated because fish taxa richness typically declines with increasing distance from the coast (McIntosh and McDowall 2004), as many of our native fish species are diadromous (i.e., they migrate between freshwater and the sea as part of their life history). The total length of piped waterway downstream of each wetland site was calculated by summing all lengths of piped waterway between stormwater wetland sampling sites and coastal outlets, using the SwPipe GIS layer. Wetland age was derived from a combination of the SwBasin and SwOutlet GIS layers. Local knowledge was used to adjust wetland age data where there were discrepancies in wetland age data between wetland assets (e.g., different commissioning dates for outlet pipes versus the wetland itself). Wetland area was calculated from the SwBasin GIS layer, with manual checks and adjustments made for sites with considerably smaller surface water area than indicated in the GIS layer.

Total catch data was compared amongst sites using results from the most representative sampling method used at each site. Thus, fine mesh fyke net data was used for all sites where fine mesh fyke nets were deployed, except for Site 18 (Douglas Clifford), where we used minnow trapping data. That was because we considered the fyke net data unrepresentative at this site, due to the fyke nets being set during elevated water levels following rainfall. The final data set for comparison included a total of 13 sites with fine mesh fyke data, five sites with five coarse mesh fykes and eight minnow traps, one site (Site 9, Portlink) with four coarse mesh fykes and seven minnow traps, one site (Site 18, Douglas Clifford) with only eight minnow traps, and one site (Site 17, Knights) with only a single minnow trap.

The New Zealand Freshwater Fish Database (NZFFD) was searched for records within each of the sampled wetlands, to determine whether any additional fish species had previously been recorded. We also reviewed NZFFD fish data from other wetlands in Christchurch that do not receive stormwater, for comparison with our data. For this comparison, we searched for NZFFD records from Travis Wetland, Lakes Albert and Victoria in Hagley Park, Halswell Quarry Ponds 1 and 3, and ponds in the Groynes recreational area.

Fish community composition was summarised using non-metric multidimensional scaling (NMDS) ordination, using the Vegan R Package (Dixon 2003; Oksanen *et al.* 2007). For this test, Bray-Curtis dissimilarity index was used to identify community differences among sites. Environmental explanatory variables were fit against the ordination using the envfit function (also from the vegan package). The envfit function fits habitat variables against the ordination axis, generating an r (goodness of fit) value and a p value via permutations (we used 999 permutations). Individual species abundances were also then correlated against the ordination in the same fashion. In addition to the summary output of the envfit tests, we examined the ordination plots with the habitat and species abundance vectors overlayed, to determine whether this revealed any meaningful trends.

Spearman rank correlation was used to explore potential relationships between the response variables of total fish abundance, fish taxa richness, shortfin eel abundance, and median shortfin eel length, and the following potential explanatory variables: median and minimum DO; mean depth; pond area; distance to coast; length of downstream pipe; emergent and total macrophyte cover (edge and centre separately); filamentous algae cover (edge and centre); and percent shade (edge and centre). Kruskal Wallis tests were run to determine if fish barrier risk had a significant effect on the above response variables.

3. RESULTS

3.1. Habitat

Sample sites varied in wetted area from a minimum of 4 m² at Site 17 (Knights) to a maximum of 39,826 m² at Site 5 (Wigram), with most sites ranging from 1,000 to 9,000 m² (Table 3). Site 17 (Knights), which had the smallest wetted area of 4 m², was unusual amongst the sites sampled, as it was essentially a dry basin with minimal surface water present. Wetlands were generally shallow, with a mean depth of 0.4 m across all sites. The shallowest mean depth was 0.06 m at Site 18 (Douglas Clifford) and the deepest was 0.79 m at Site 1 (Sparks First Flush, Table 3). See Figure 2 for representative site photographs and Appendix 1 for photographs of all the wetland sampling sites.

Distance to the coast ranged from a minimum of 0.4 km at Site 10 (Charlesworth) to a maximum of 35.6 km at Site 17 (Knights), with most sites falling between 5 and 25 km from the coast (Table 3). The total length of downstream piped waterway ranged from 6 m at Site 10 (Charlesworth) to 1,694 m at Site 16 (Arthur Adcock), with a median of 98 m. Site 16 was notable for being completely piped from the pond outlet to the coast. For all other wetlands downstream waterways were mostly open, with pipes comprising <6% of the total waterway length.

Most of the wetlands sampled have been recently created, with 18 of the 21 wetlands <20 years old, 15 of them <10 years old, and seven of them <5 years old (Table 3). Site 20 (Beckenham), a non-stormwater wetland was the oldest wetland sampled, at 67 years, while the oldest stormwater wetland sampled was Site 5 (Wigram), at 29 years.

The substrate at most wetland sites was covered in fine sediments (<2 mm diameter). Exceptions to this were the margins of Site 13 (Burlington), where the substrate comprised large cobbles (128–256 mm) and boulders (256–4,000 mm), and Site 14 (Alpine View), where large gravels (16–64 mm) dominated at the wetland margin. At Site 15 (Ryman) the substrate throughout the basin was comprised of ballast (crushed, gravel-sized stones), which was covered by a thin layer of fine sediment.

Riparian vegetation at most wetland sites comprised a narrow (<10 m) band of native plantings, dominated by low-growing sedges, especially *Carex secta* (Figure 2). Beyond the band of native plantings, the ground cover was typically either mown grass (particularly associated with wet ponds within a larger flood detention basin) or tracks and roads.

Nearly all sites were poorly shaded, with site means of <30% shade, reflecting the broad nature of stormwater wetlands, coupled with the fact that many are very new, and lack established trees. Where shade was present, it was typically greatest near the wetland edge (Figure 3) and was associated with overhanging sedges (*Carex* spp.). A notable exception was Site 17 (Knights), where mature *Carex* throughout the basin resulted in high shading in the wetland centre. Site 20 (Beckenham) also had relatively high shading at the wetland centre, due to a combination of mature exotic trees and the pond being relatively small and therefore easily shaded.

Table 3: Habitat characteristics of the wetlands sampled.

Site No.	Site Name	Area (m^2)	Mean Depth (m)	Wetland Age (years)	Fish Barrier Risk	Distance From Coast (km)	Length Piped Downstream (m)
1	Sparks Wetland	1,931	0.79	2.8	Low	17.3	33
2	Sparks First Flush	4,642	0.59	2.8	Low	18.8	39
3	Bullers	8,922	0.23	3.0	Low	16.0	795
4	Quaifes	2,733	0.21	3.8	High	34.0	19
5	Wigram	39,826	0.54	29.2	Very High	21.0	433
6	Ngā Puna Wai	27,205	0.29	3.0	Very High	20.2	409
7	Eastman	1,485	0.33	9.2	Very High	19.8	326
8	Halswell Downs	7,415	0.42	7.7	High	18.0	30
9	Portlink	1,872	0.16	9.9	Very High	3.9	27
10	Charlesworth	4,473	0.40	27.0	Medium	0.4	6
11	Clare Park	8,156	0.24	5.8	Medium	8.5	69
12	Prestons	4,624	0.31	7.4	Medium	12.8	160
13	Burlington	5,183	0.36	0.9	Medium	16.2	45
14	Alpine View	1,939	0.53	10.2	Low	10.4	116
15	Ryman	1,548	0.18	9.2	Very High	15.2	786
16	Arthur Adcock	2,541	0.43	19.2	Very High	1.7	1,694
17	Knights	4	0.10	5.8	High	35.6	60
18	Douglas Clifford	546	0.06	12.0	Medium	19.4	190
19	Spring Grove	1,141	0.27	5.9	Very High	21.2	80
20	Beckenham	867	0.39	67.2	High	9.8	31
21	Te Oranga Waikura	720	0.67	4.4	Low	4.8	243

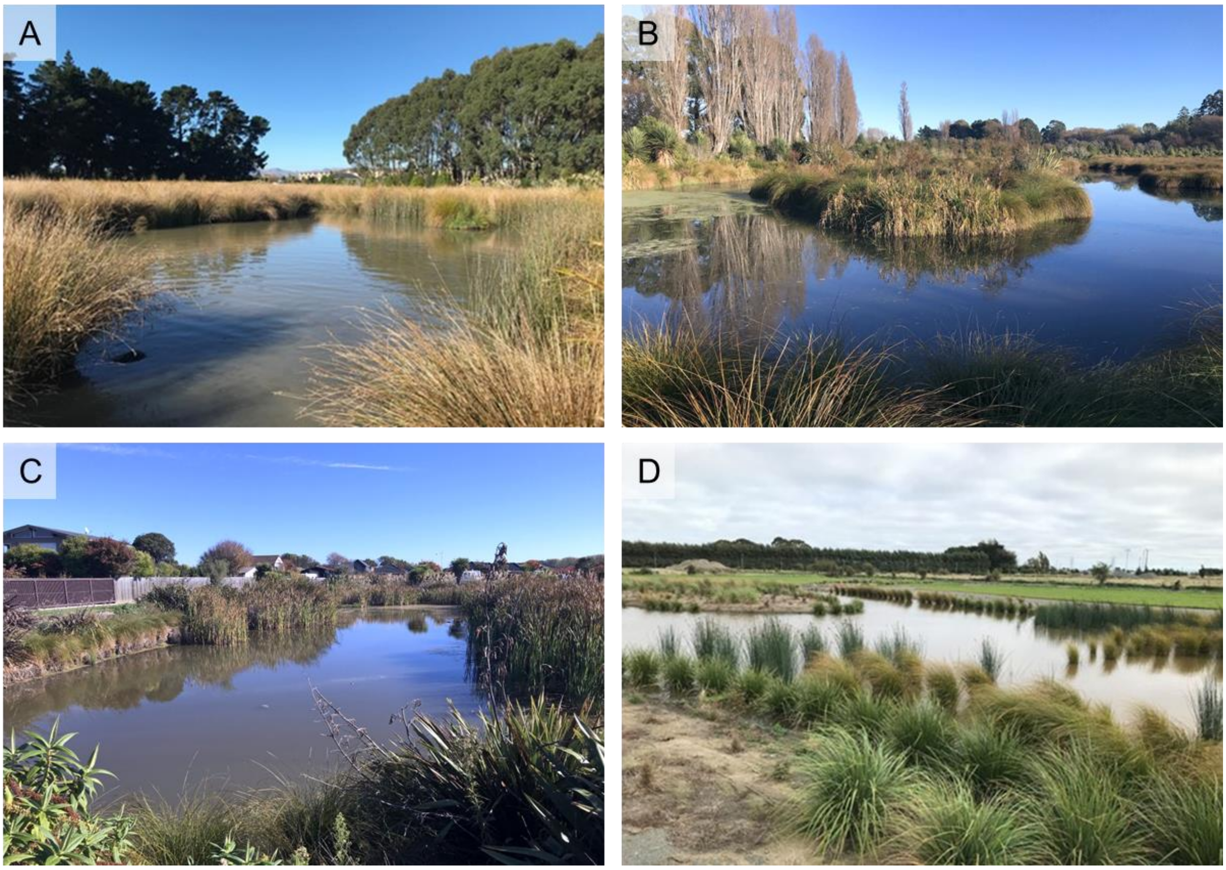

Figure 2: Examples of some of the wetlands sampled. A) Site 3 (Bullers); (B) Site 11 (Clare Park); Site 15 (Ryman); Site 8 (Halswell Downs).

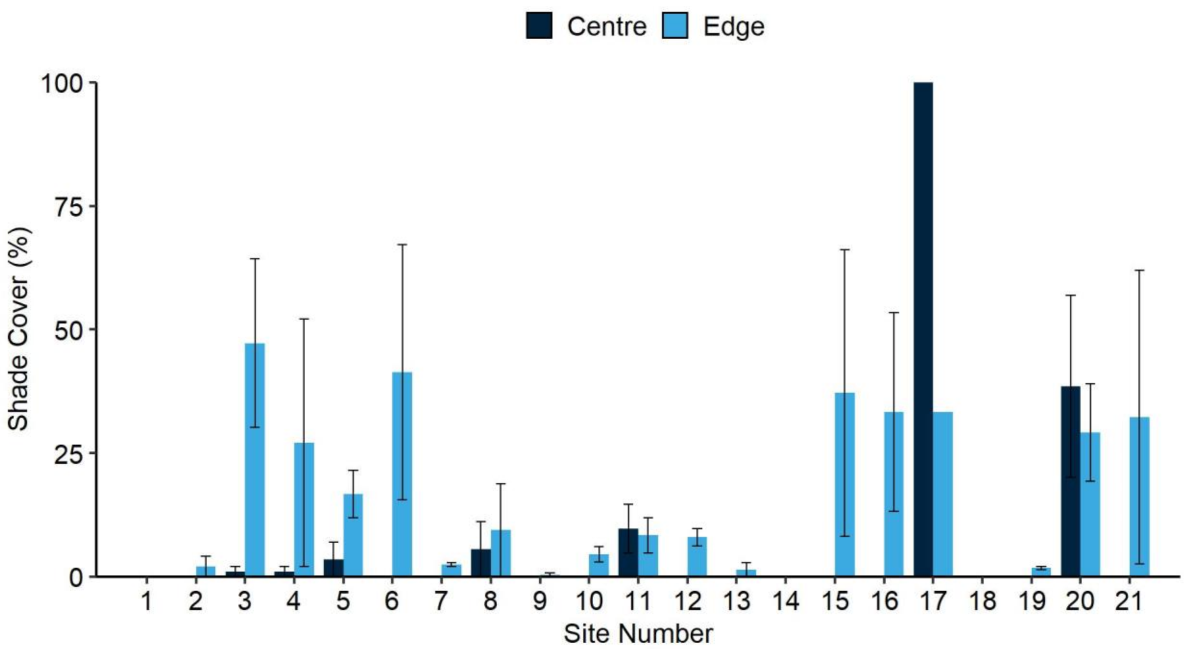

Figure 3: Mean (1±SE) shade percent for edge and centre measurements at all sites. No error bars are shown for Site 17 because only a single transect was measured at that site.

Total macrophyte cover was often high, but it varied considerably amongst sites (Figure 4). Macrophyte cover was typically greatest towards the edge of wetlands, where emergent macrophytes were found (Figure 5). Site 6 (Ngā Puna Wai) was the only site where no macrophytes were recorded, possibly due to high turbidity observed at the site. Common duckweed (*Lemna disperma*) is a floating native species that was common at many sites and covered the entire wetland surface at Sites 14 (Alpine View) and 21 (Te Oranga Waikura). Another native floating species, Pacific azolla (*Azolla rubra*) was also common. Other regularly encountered macrophytes included the exotic macrophyte watercress (*Nasturtium officinale*), an emergent species found mostly around the water's edge, and the exotic curly pondweed (*Potamogeton crispus*), a submerged species.

Periphyton cover was low at most sites, reflecting the dominance of fine sediments, relatively high macrophyte cover, and possibly also high turbidity at some sites. However, where periphyton was present, it was dominated by long filamentous algae (>20 mm long) and bed cover was high at some sites (Figure 6). When present, long green filamentous algae was often found suspended within the water column, and very rarely found confined to the substrate. Filamentous algae cover was greatest towards the centre of Site 3 (Bullers), where it covered 100% of the bed.

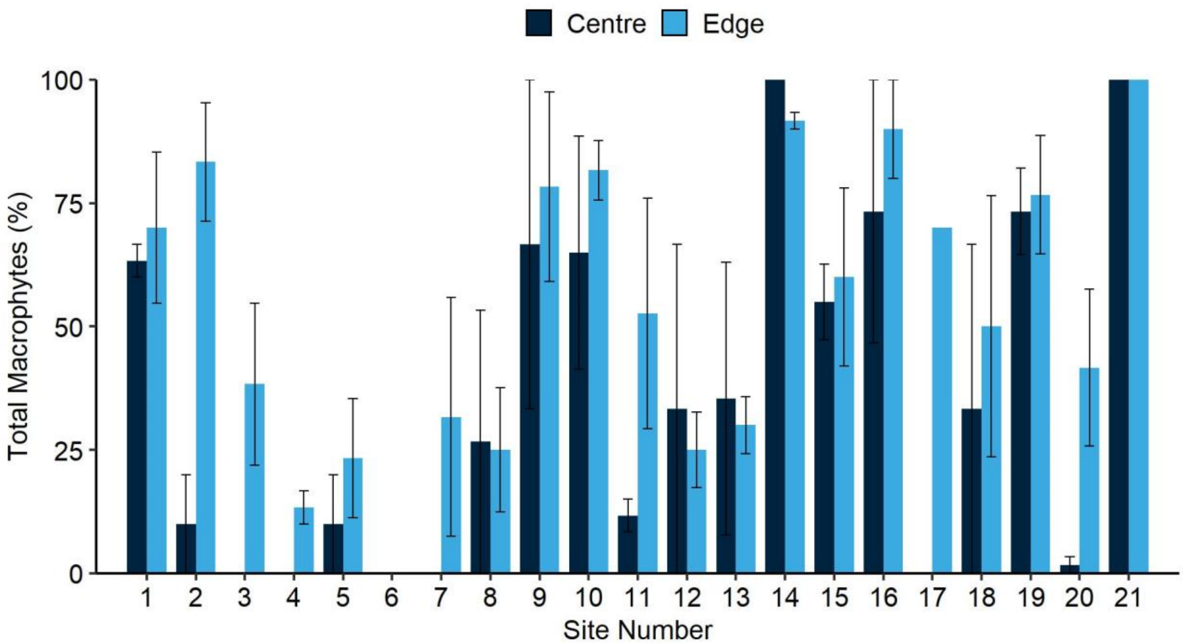

Figure 4: Mean (1±SE) total macrophyte cover for edge and centre measurements at all sites. No error bars are shown for Site 17 because only a single transect was measured at that site.

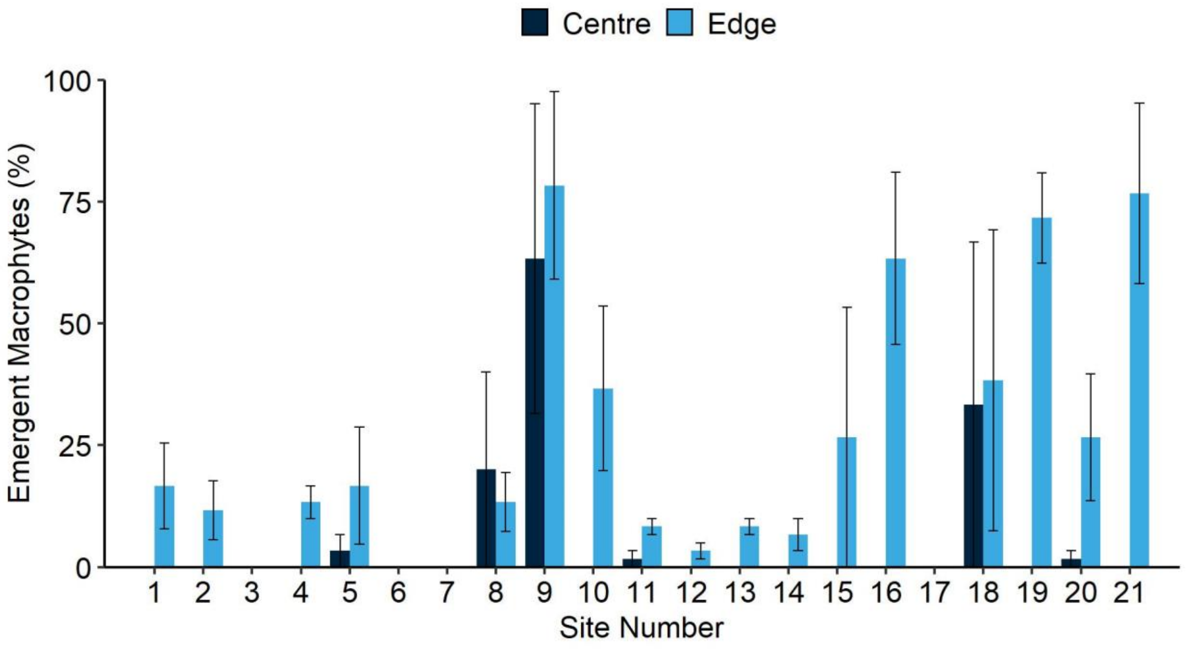

Figure 5: Mean (1±SE) emergent macrophytes cover for edge and centre measurements at all sites

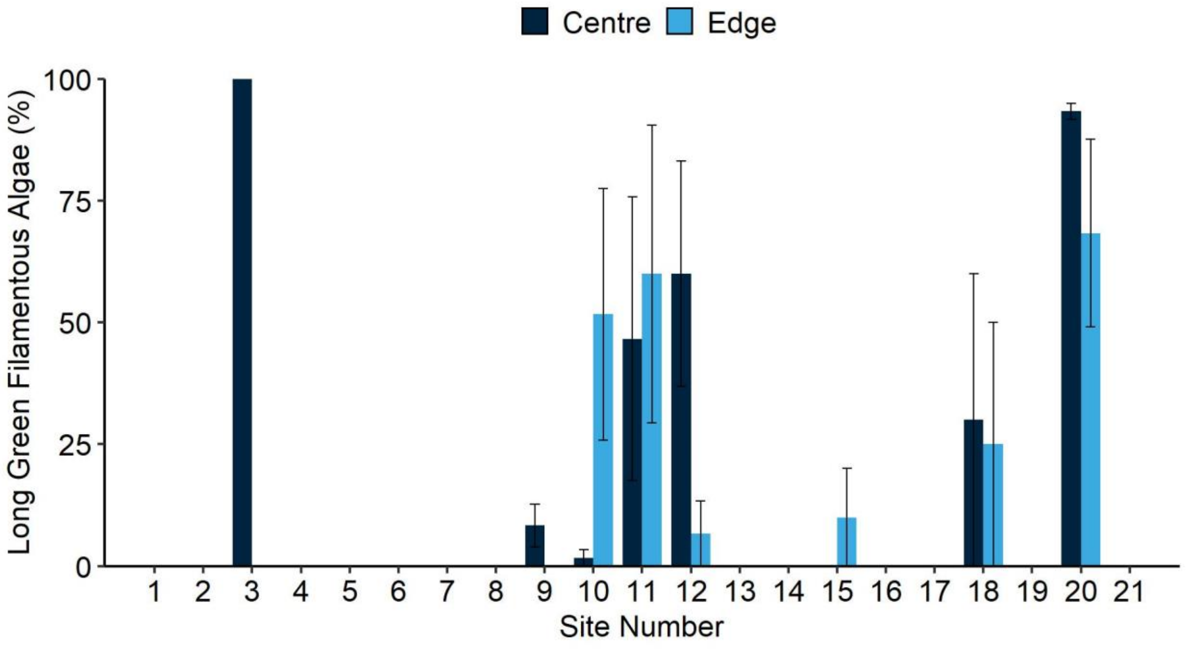

Figure 6: Mean (1±SE) long green filamentous algae cover for edge and centre measurements at all sites.

Overall, the wetlands sampled provided sufficient aquatic habitat to potentially support fish populations. The only exception was Site 17 (Knights), where there was minimal surface water present. Aquatic macrophytes and wetland plants provide good potential fish cover at many locations, but there is a general lack of larger wood or larger cobbles and boulders that could provide additional habitat complexity. Notable exceptions include Site 21 (Te Oranga Waikura), where logs have been placed throughout the wetland, and Site 3 (Bullers), where boulders and tree stumps have been installed.

3.2. Potential Fish Barriers

Of the 21 wetlands sampled, downstream structures presented a low risk to fish passage at five sites, medium risk at five sites, high risk at four sites, and very high risk at seven sites (Table 3). Pipe culverts were the most common structures, while other structures included gates, valves, weirs, and narrow flow constrictions. Examples of low-risk structures include the outlet of Site 21 (Te Oranga Waikura), where there is adequate depth and low velocities to provide fish passage, and a culvert joining wetland cells at Site 13 (Burlington), where baffles have been placed to enhance fish passage (Figure 7). An example of a very high-risk structure is the outlet of Site 16 (Arthur Adcock), which comprises a weir with multiple steps (Figure 7). Other very high-risk structures were the combination of grates and piped network draining the retention pond outlets at Site 7 (Eastman) and Site 18 (Spring Grove). The outlet grates had 35 mm spacings between the bars at both wetlands (Figure 7). See Appendix 2 for details of all 51 structures assessed for fish passage across the wetland sites.

Figure 7: Examples of structures assessed for fish passage risk at the wetlands sampled. These include a very high-risk structure at A) Site 16 (Arthur Adcock), a medium-risk structure at B) Site 18 (Douglas Clifford), and low risk structures at C) Site 21 (Te Oranga Waikura) and D) Site 13 (Burlington).

3.3. Water Quality

Mean water temperatures recorded over a 24-hour period ranged from a low of 12.7 °C at Site 17 (Knights) in early April, to a high of 20.7 °C at Site 2 (Sparks First Flush) in early March (Table 4). Maximum temperatures ranged from a low of 13.9 °C at Site 17 (Knights) to a high of 22.4°C at Site 1 (Sparks Wetland) in early March. Preferred water temperatures are <20 °C for some native freshwater fish species, such as inanga (*Galaxias maculatus*; Richardson *et al.* 1994). Seven sites recorded maximum temperatures >20 °C, and a further three sites had maximum temperatures of 19–20 °C. Given the autumn timing of temperature measurement, it is likely temperatures exceed 20 °C during the warmer summer months at many of the sites sampled.

Table 4: Summary statistics of temperature (Temp.) and dissolved oxygen (DO) measured over 24 hours.

Site No.	Site Name	Date[1]	Mean DO (mg/L)	Min DO (mg/L)	Mean Temp. (°C)	Max Temp. (°C)
1	Sparks Wetland	9/03/2022	9.5	6.2	18.5	22.4
2	Sparks First Flush	9/03/2022	9.3	7.6	20.7	22.3
3	Bullers	3/04/2022	9.4	6.3	17.0	19.0
4	Quaifes	14/04/2022	6.9	5.5	12.9	15.3
5	Wigram	29/03/2022	9.9	7.6	17.1	19.0
6	Ngā Puna Wai	28/03/2022	6.5	2.7	16.4	18.6
7[2]	Eastman	22/05/2022	14.0	8.9	18.8	22.3
8	Halswell Downs	23/03/2022	8.7	7.0	18.5	20.1
9	Portlink	30/03/2022	2.9	1.8	18.5	21.1
10	Charlesworth	30/03/2022	7.4	5.1	18.4	19.7
11	Clare Park	3/05/2022	1.8	1.0	15.1	17.3
12	Prestons	4/04/2022	10.7	5.4	18.1	20.4
13	Burlington	5/05/2022	3.6	3.3	15.3	16.2
14	Alpine View	4/04/2022	3.9	0.9	17.2	18.3
15	Ryman	3/04/2022	7.1	4.9	18.5	21.4
16	Arthur Adcock	5/04/2022	0.8	0.2	15.6	16.0
17	Knights	9/04/2022	3.8	2.7	12.7	13.9
18	Douglas Clifford	25/03/2022	0.5	0.1	15.0	17.6
19	Spring Grove	9/04/2022	6.8	5.1	14.4	15.6
20	Beckenham	11/04/2022	4.3	2.6	13.6	15.2
21	Te Oranga Waikura	12/04/2022	0.2	0.1	13.2	14.1

Note: [1] Date is when the water quality logger was deployed. [2] Temperature data from Site 7 is from 10/3/2022; DO data was discarded from that date, due to concerns over dubious readings.

Mean DO concentrations recorded over a 24-hour period ranged from a low of 0.2 mg/L at Site 21 (Te Oranga Waikura) in mid-April to a high of 14.0 mg/L at Site 7 (Eastman) in early May (Table 4). Minimum DO concentrations were also lowest at Site 21, with a low of

0.06 mg/L, while the highest daily minimum DO concentration was 14 mg/L at Site 7. The National Policy Statement for Freshwater Management 2020 (NPSFM, Ministry for the Environment 2020) has a national bottom line of 4 mg/L for protection of aquatic biota in lakes and rivers. While some sites were characterised by high DO concentrations that were consistently well above 4 mg/L, others had moderate to low concentrations that dropped below 4 mg/L for at least part of the day (Figure 8). Ten of the 21 sites recorded minimum DO concentrations below 4 mg/L, with some sites sitting well below 4 mg/L for the entire 24-hr period.

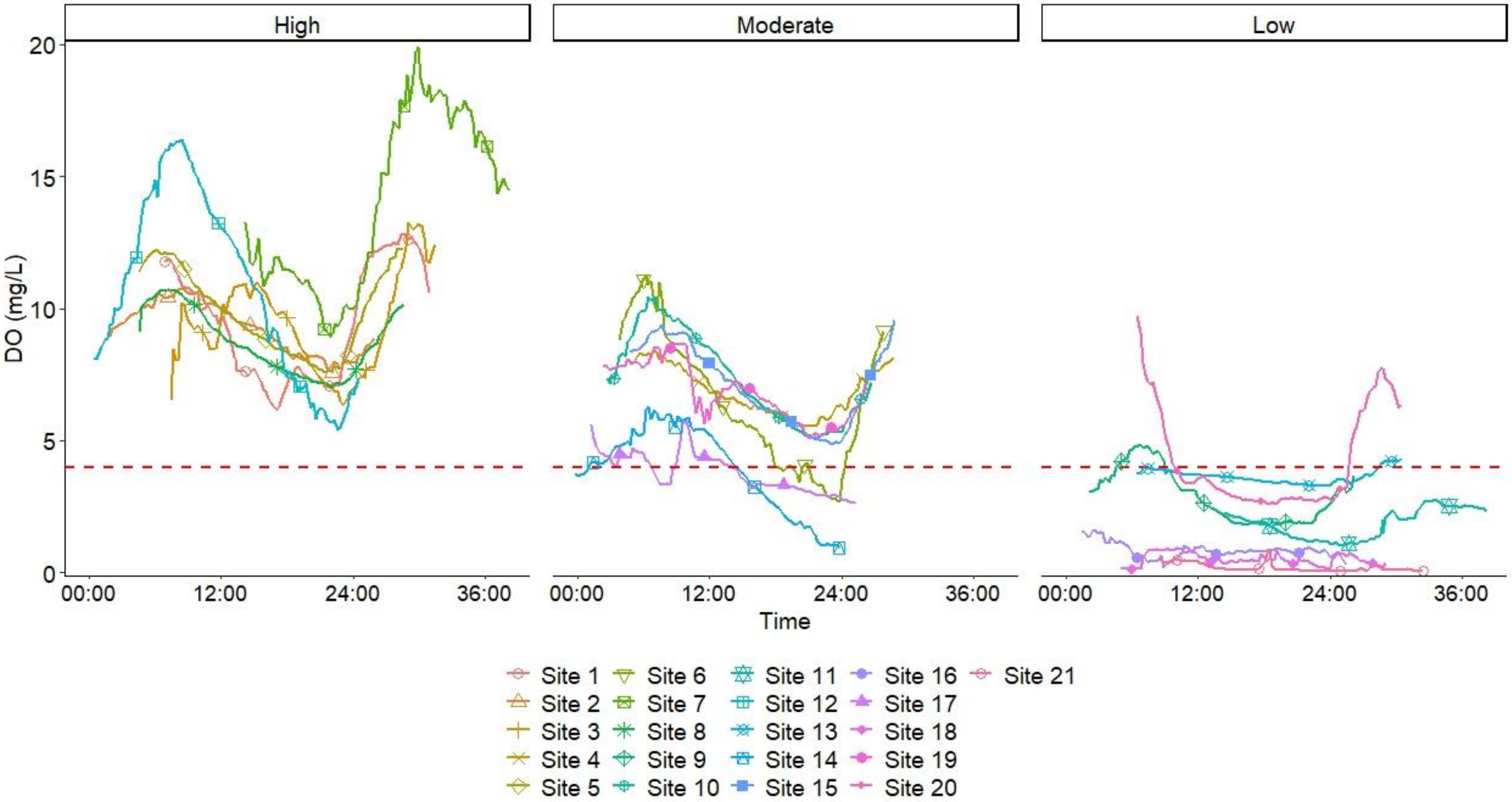

Figure 8: Dissolved oxygen (DO) concentrations recorded over a 24 hour period at each sampling site. Sites are even split into those with relatively high, moderate, and low median DO concentrations over the 24 hour period. Red dashed line indicates NPSFM national bottom line of 4 mg/L.

3.4. Fish

3.4.1. Methods Comparison

Fine mesh fyke nets caught more fish and a greater range of fish species and life stages than coarse mesh fykes and minnow traps combined (Figure 9). For example, common bully (*Gobiomorphus cotidianus*) and juvenile bullies (*Gobiomorphus sp.*) were caught by fine mesh fykes, but not by coarse mesh fykes and minnow traps at Site 1 (Sparks Wetland). Similarly, longfin eel (*Anguilla dieffenbachii*) were caught using fine mesh fykes, but were not caught using coarse mesh fykes and minnow traps at Site 5 (Wigram). In addition, coarse mesh fykes caught fewer smaller-sized eels (Figure 10). Both fine mesh fyke nets and minnow traps were used to sampled fish at five sites (Table 2). No additional species were caught using minnow traps than were caught by fine mesh fyke nets at each of the five sites. Overall, these data indicate that fish diversity was likely underestimated at those sites where coarse mesh fykes and/or minnow traps were used to sample the fish community, when compared with sites sampled using fine mesh fyke nets.

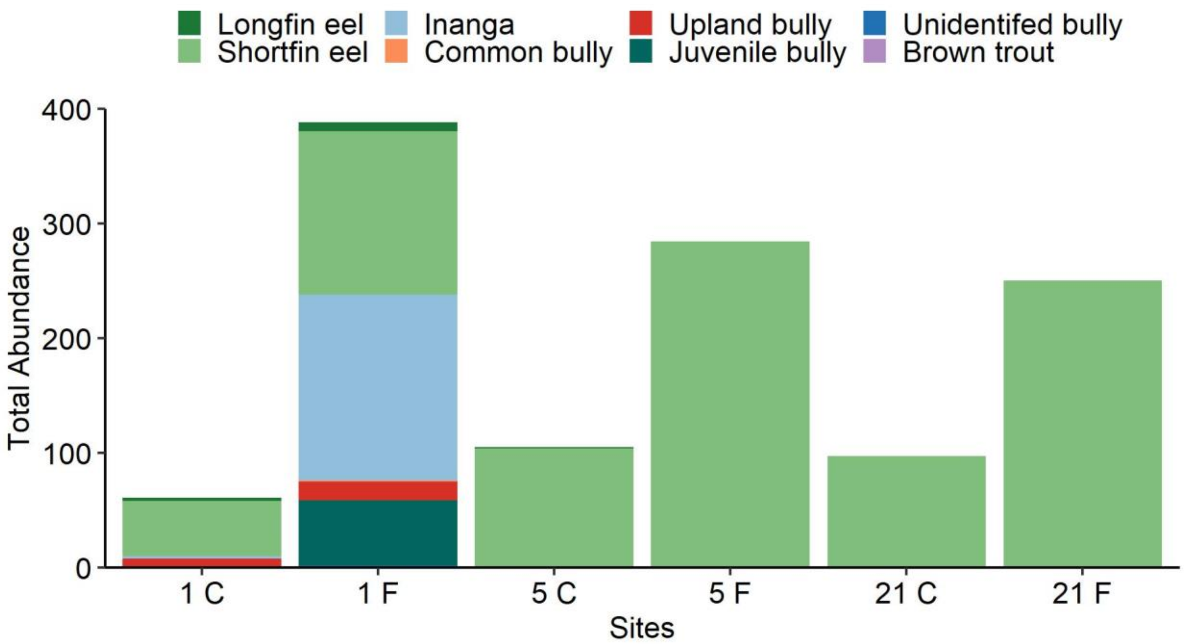

Figure 9: Comparison of the fish catch between coarse mesh fyke nets and minnow traps (C) and fine mesh fyke nets (F), at sites where both fishing methods were used.

As noted in Section 2.2.4 above, a fish kill at Site 18 (Douglas Clifford) resulted in a subsequent change of fishing methods for sites with low oxygen concentrations. Fine mesh fyke nets were initially deployed when water levels in the basin were elevated following rain two days previous. The five fyke nets caught a total of 404 shortfin eels (*A. australis*). Many of the eels had distended guts and large numbers of earthworms were regurgitated in the fyke nets and on the measuring boards. In addition, many of the eels were observed gulping for air in the recovery bins after processing. Dissolved oxygen concentrations were very low, with a

24-hr mean of 0.5 mg/L. We returned to re-fish the site when water levels were much lower and found depths were too shallow to deploy fyke nets. Eight minnow traps were set and they caught a single shortfin eel and three upland bully (*G. breviceps*). We only used the minnow trap data for subsequent comparison amongst sites because it was clear the earlier conditions with elevated water levels were not representative of baseflow conditions.

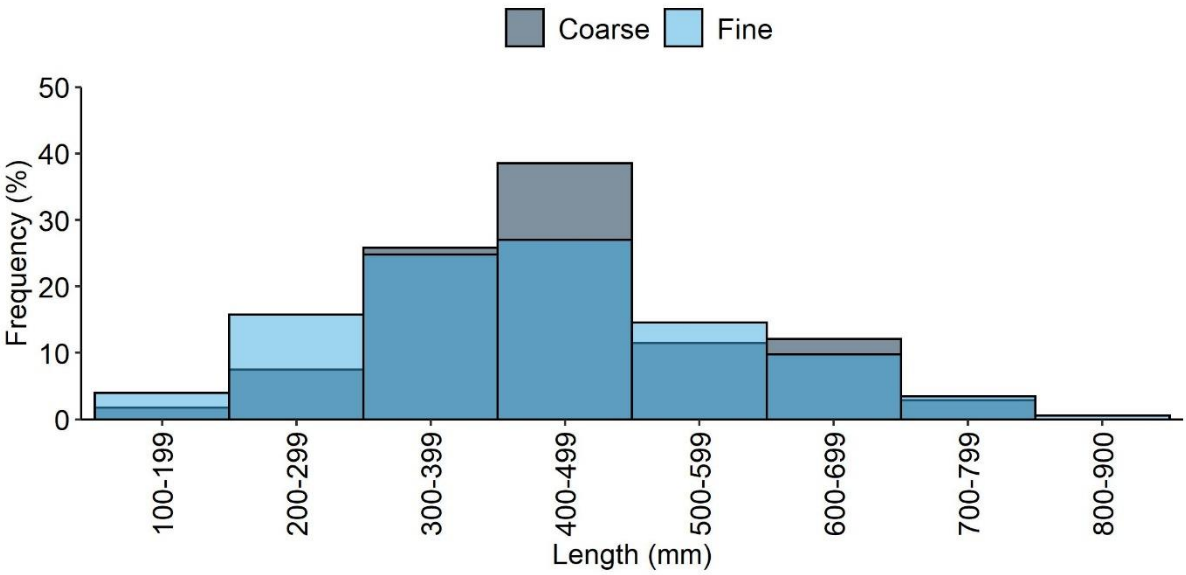

Figure 10: Comparison of the size distribution of shortfin eels caught in fine mesh and coarse mesh fyke nets.

3.4.2. Fish Community Composition

Fish were caught at all sites except for Site 17 (Knights), which had minimal surface water (Table 3). A total of six species were caught across the wetland sites, including native shortfin eel, longfin eel, inanga, common bully, and upland bully, and introduced brown trout (*Salmo trutta*). Shortfin eels were caught at all 20 of the sites where fish were found and they were the most abundant species overall, with a total of 2,945 caught across the wetlands. Site 2 (Sparks First Flush) had the largest catch of shortfin eels, with 802 caught in five fine mesh fyke nets (Figure 11). Inanga were the second-most abundant species, with a total of 344 caught across five sites, although they were most abundant at Site 10 (Charlesworth) and Site 1 (Sparks Wetland). Common bully was the third most abundant species, with a total of 131 caught across three sites, with the greatest catch at Site 10 (Charlesworth). Upland bully were found at six sites, but only 56 were caught across the sites. Longfin eel were caught at five sites and they were also uncommon, with a total of 19 caught. The largest fish caught overall was a longfin eel measuring 1,218 mm, from Site 1 (Sparks Wetland). A single brown trout was caught at Site 5 (Wigram). See Appendix 3 for a summary of fishing results for all sites and fishing methods.

Taxa richness was low overall, with a maximum of five species recorded at Site 1 (Sparks Wetland), and four species recorded at Site 10 (Charlesworth) and Site 13 (Burlington, Figure 12). There were 10 sites where shortfin eels was the only species caught. As noted in Section 3.4.1, fishing method affected the total catch and it is likely that more species were present at some of the sites that were only sampled with coarse mesh fykes and/or minnow traps.

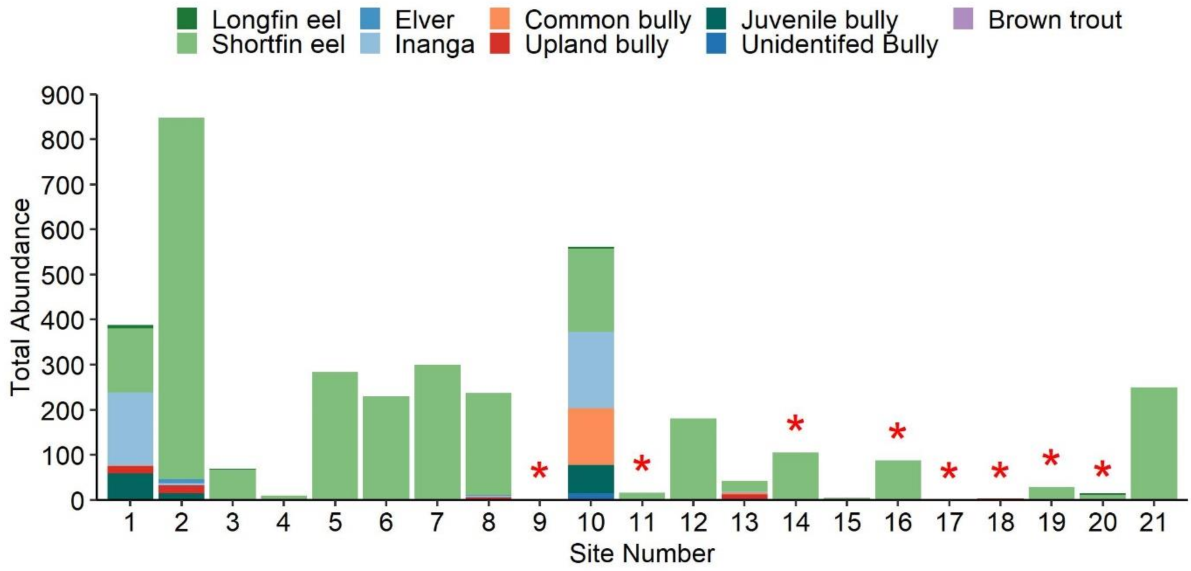

Figure 11: Abundance of fish species caught at each wetland site. Data are from fine mesh fyke nets, except those marked with asterisks, which were sampled using a combination of coarse mesh fyke nets and minnow traps.

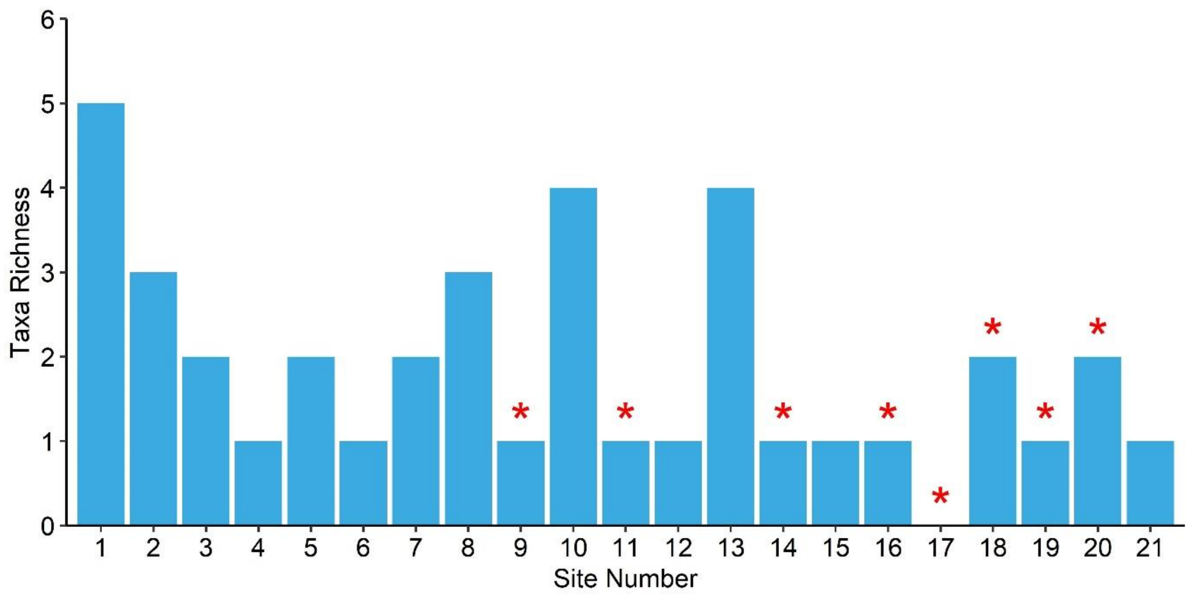

Figure 12: Fish taxa richness at each wetland site. Data are from fine mesh fyke nets, except those marked with asterisks, which were sampled using a combination of coarse mesh fyke nets and minnow traps.

Eight of the 21 sampling sites had previous fishing records in the New Zealand Freshwater Fish Database (NZFFD). Shortfin eels were recorded in the NZFFD at all eight sites. No additional species, beyond those caught during our survey, were recorded at five of the eight sites. The NZFFD search yielded additional taxa for three sites: unidentified bully (*Gobiomorphus sp.)* at Site 12 (Prestons); longfin eel, inanga, and a single goldfish (*Carassius auratus)* at Site 16 (Arthur Adcock); and a single longfin eel at Site 21 (Te Oranga Waikura).

The NZFFD search for fish sampling records[4] from other Christchurch non-stormwater wetlands yielded records for Travis Wetland, the Groynes Pond 5 (the largest lake in the Groynes Recreational Area), and Halswell Quarry Pond 1 (the largest pond in Halswell Quarry Park). There were no fish records for Victoria Lake and Lake Albert in Hagley Park or Halswell Quarry Pond 3. Fish records for Travis Wetland included native shortfin eels, giant bully (*G. gobioides*), common bully, inanga, and Canterbury mudfish (*Neochanna burrowsius*), and the invasive noxious species rudd (*Scardinius erythrophthalmus*). However, we are aware that a rudd monitoring and eradication programme has been carried out in Travis Wetland, and there have been no records of this species since 2010. Fish records for the Groynes Pond 5 included native shortfin eels and longfin eels, and exotic brown trout, rainbow trout (*Oncorhynchus mykiss*), and the invasive exotic species, tench (*Tinca tinca*). Fish records for Halswell Quarry Pond 1 included shortfin eels, common bully, and rudd.

We did not catch giant bullies, Canterbury mudfish, rudd, tench, or rainbow trout at any of the sites we surveyed, nor were there NZFFD records for them at the wetlands we sampled. We note that Canterbury mudfish were translocated to Travis Wetland for conservation purposes in 2010 and 2021[5]. Rudd, tench, rainbow trout, and brown trout will have been introduced into waterways for recreational fishing and aesthetic purposes. Overall, the three non-stormwater wetlands generally had higher taxa richness than found at our sampled wetlands, but the greater taxa count was mainly comprised of exotic species with no conservation value.

3.4.3. Factors Affecting Fish Communities

We expected greater taxa richness at sites located closer to the coast, as this is a general characteristic of New Zealand's native fish fauna, due to the prevalence of diadromy (McIntosh and McDowall 2004). There was clearly an influence of proximity to coast, with Site 10 (Charlesworth), the site closest to the coast (0.4 km), having relatively high species richness (four species), and also large numbers of inanga and the largest numbers of common bully recorded at any site (both inanga and common bully are diadromous). Site 9 (Portlink) was the second-closest to the coast (3.9 km), but had low taxa richness, with only shortfin eels recorded. The lack of species diversity at Site 9 was likely due to the presence of an outlet structure assessed as presenting a very high risk to fish passage. The three wetlands that recorded four or five taxa all had outlet structures assessed as presenting low to medium risk to fish passage; none of them were graded as high or very high risk. Site 1 (Sparks Wetland), which had the highest species richness (five species) is 17.3 km from the coast, but the fish passage risk was assessed as low.

[44] The NZFFD includes records of sampling effort and records include sites where no fish were found. Therefore, the absence of an NZFFD record means no fish sampling has occurred, not that fish are absent.

[5] https://traviswetland.org.nz/about-travis/events/2012-mudfish-release/

Ordination on the fish community at the 13 sites with fine mesh fyke netting data yielded a stress value of 0.06, indicating a good representation of the dissimilarity matrix in two dimensional space (Clarke 1993). There were no statistically significant ($p<0.05$) or ecologically meaningful envfit correlations between axis scores and environmental variables. However, pond area, downstream pipe length, and shade at edge were nearly significant ($p=0.053–0.077$).

There were several statistically significant spearman rank correlations between environmental variables and the response variables of total fish abundance, fish taxa richness, shortfin eel abundance, and median shortfin eel length. Total abundance was positively correlated ($p<0.01$) with mean depth (Figure 13); taxa richness was correlated positively with mean depth ($p<0.05$) and negatively with percent edge shade ($p<0.01$, Figure 14); and shortfin eel abundance was positively correlated with mean depth ($p<0.05$). In addition, shortfin eel length and abundance were negatively correlated to each other ($p<0.05$, Figure 15). Fish abundance and taxa richness showed a weak, positive correlation ($p=0.054$). Plots of the fish data categorised by barrier score suggest that ponds with the highest richness and abundance had low barrier risk, but high abundance and richness did not occur at all sites with low barrier risk scores (Figure 16). Kruskal-Wallis tests revealed no significant effect of barrier risk score on any of the response variables ($p>0.05$).

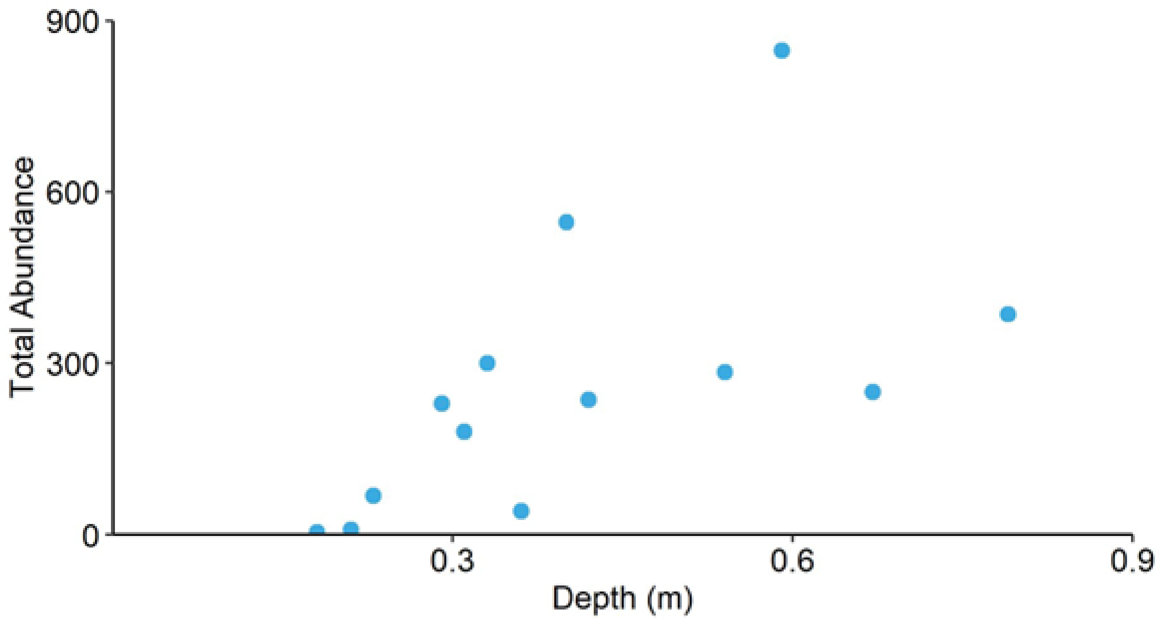

Figure 13: Relationship between mean water depth and total fish abundance for 13 sites with fine mesh fyke data.

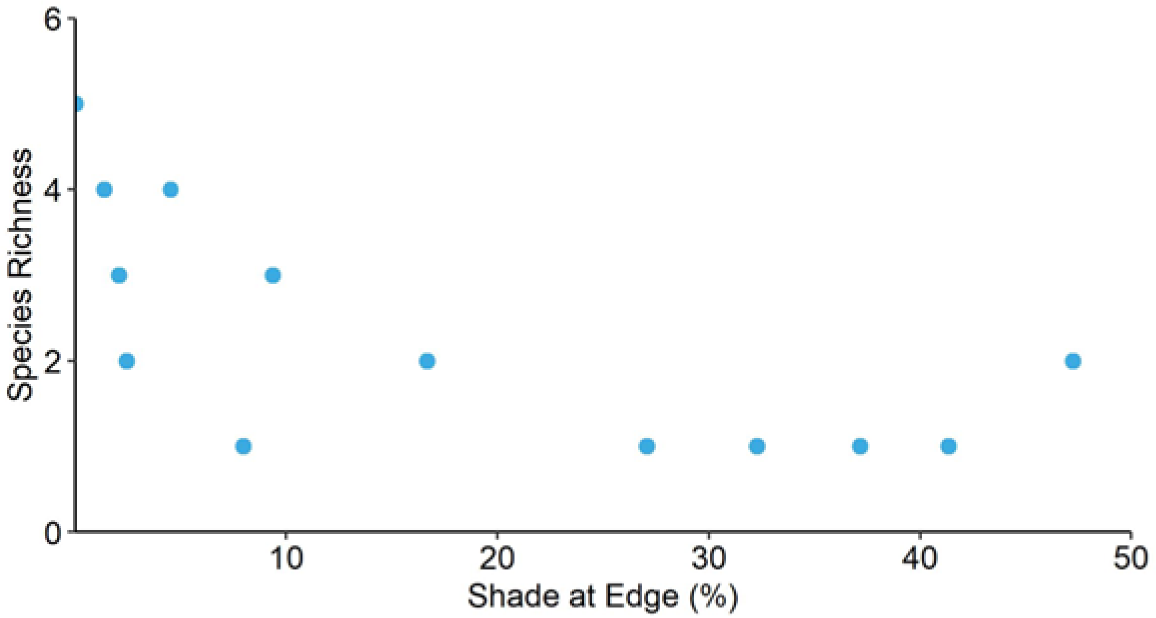

Figure 14: Relationship between shade and fish species richness for 13 sites with fine mesh fyke data.

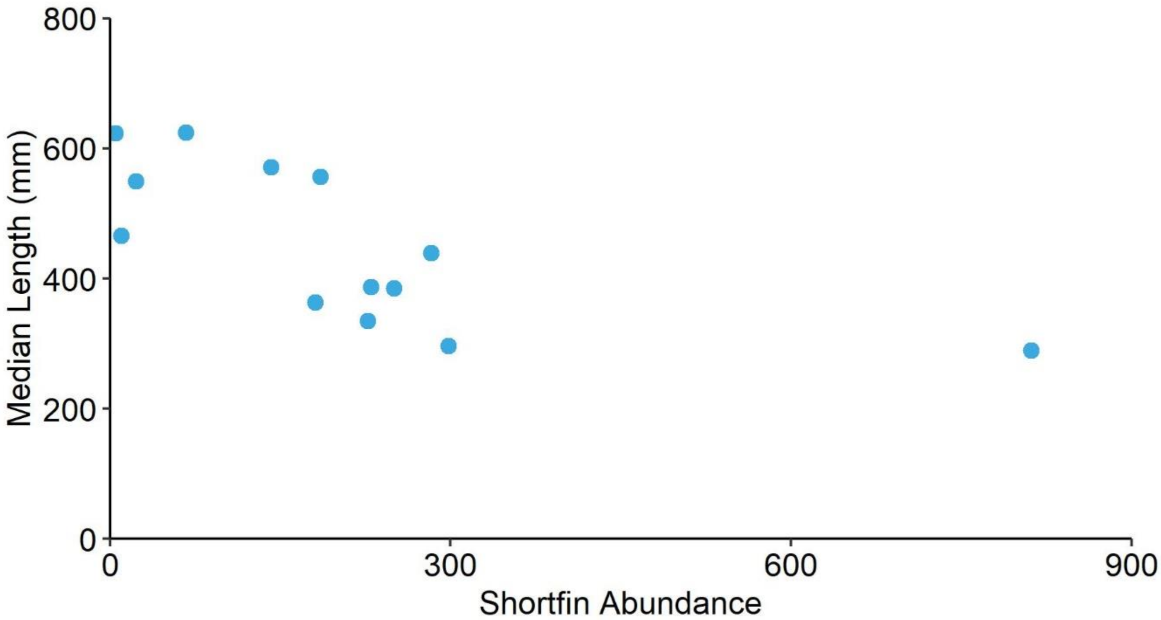

Figure 15: Relationship between shortfin eel abundance and length for 13 sites with fine mesh fyke data.

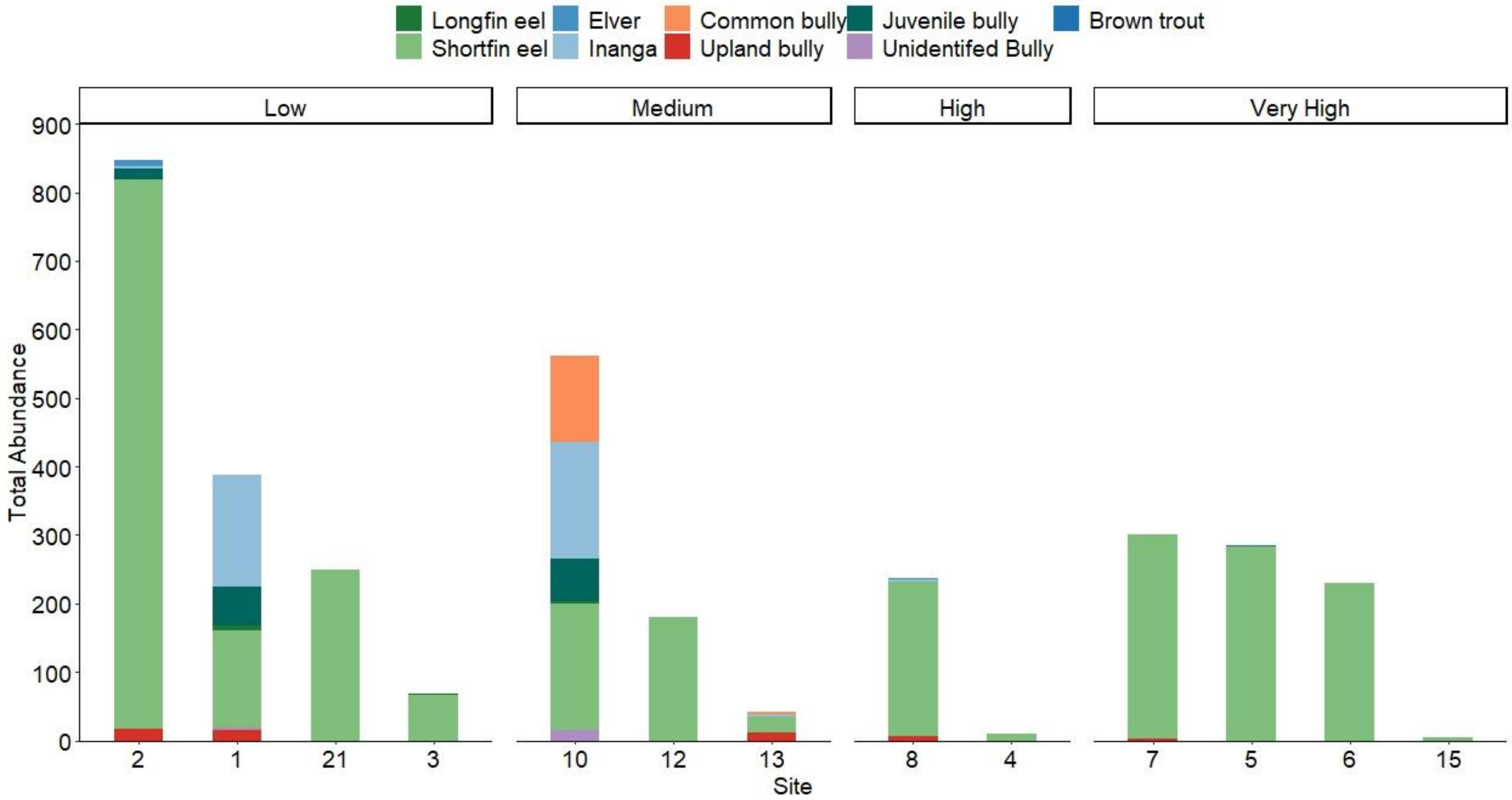

Figure 16: Fish community composition for sites sampled with fine mesh fyke nets. Sites are ordered by fish passage risk for each wetland.

4. DISCUSSION

4.1. What Fish are Present in Council Stormwater Wetlands?

We caught fish at 20 of the 21 wetland sites sampled. Site 17 (Knights) was the only site where no fish were caught, due to a lack of surface water. Shortfin eels were found in all wetlands where fish were caught, and they were often the only species caught. The data from this survey strongly suggest that fish – particularly shortfin eels – will be present in all of Christchurch's stormwater wetlands that have sufficient surface water.

Five of the six fish species caught are native species. Longfin eels and inanga have an At Risk – Declining conservation status, while shortfin eels, upland bully, and common bully are classified as Not Threatened (Dunn *et al.* 2018). The only exotic species caught was a single brown trout. Further sampling would likely turn up additional fish species, but our results indicate that the abundance of exotic species was very low relative to native species. The prevalence of native fish at the sampled sites was notable, particularly given the presence of invasive rudd and tench in other non-stormwater wetlands in the city. Goldfish, which is also an invasive species, had previously been recorded from one of the sites we sampled (Site 10, Arthur Adcock), but only a single individual had been recorded. Invasive fish species are common in urban ponds and wetlands in New Zealand, where they are introduced for aesthetic reasons (e.g., goldfish) and to support sports fisheries (e.g., rudd and tench). Why native species dominated in our study is unclear, but it may mean that there are fewer intentional translocations of exotic fish into stormwater wetlands compared to other wetlands around the city. Invasive fish species have the potential to negatively impact water quality, aquatic habitat, and native biodiversity (Collier and Grainger 2015), so keeping such species out of wetlands should be a management priority. This can be done by a combination of public education and surveillance monitoring.

Stormwater wetlands had lower species richness compared to the three non-stormwater urban wetlands with historic fish records. However, the greater species count at non-stormwater wetlands was mainly comprised of exotic species. The only exceptions were records of Canterbury mudfish, which have a Threatened – Nationally Critical conservation status, in Travis Wetland, and giant bully, which have an At Risk – Naturally Uncommon status (Dunn *et al.* 2018), also in Travis Wetland. Canterbury mudfish were translocated to Travis Wetland and we would not expect them to occur in artificially created stormwater wetlands. However, giant bully are diadromous and they could reasonably be expected to occur in Council stormwater wetlands, given suitable water quality and habitat conditions, and provided there were no major fish passage barriers downstream.

4.2. What Design Features Determine Fish Presence in Stormwater Wetlands?

Key design features of stormwater wetlands that determine fish presence include whether surface water is present, habitat considerations, and provision of fish passage into the wetland. These physical design features are discussed further in the following paragraphs. Wetland design may also influence fish presence and community composition via impacts on water quality, and this is discussed in Section 4.4 below.

Fish were caught at all stormwater wetlands that had sufficient surface water present. While no fish were caught at Site 17 (Knights), the wetted area to sample was restricted to a puddle with an area of 4 m² and a mean depth of 0.10 m. The second-smallest wetland was Site 18 (Douglas Clifford), which had a wetted area of 546 m² during fish sampling under baseflow conditions, when upland bullies and shortfin eels were caught. The influx of shortfin eels into the Site 18 wetland following rainfall, coupled with their distended guts and regurgitated earthworms, suggests that eels were opportunistically feeding on terrestrial earthworms during elevated water levels. This is supported by the observation of shortfin eels feeding opportunistically on terrestrial earthworms during inundation of pasture around Lake Ellesmere (Ryan 1986). Similarly, in a study in Lake Pounui, the contribution of terrestrial invertebrates to the diet of shortfin and longfin eels increased from <6% during non-flood conditions up to 80–83% during a flood period when terrestrial vegetation was inundated (Jellyman 1989). These studies and our own observations suggest that eel abundances in stormwater wetlands may vary greatly in response to water level, with eels moving in to feed on terrestrial invertebrates when water levels are elevated and moving out as water levels drop. This indicates that even temporary surface water can result in fish being present in stormwater wetlands. It also suggests that there is potential for fish to be stranded in stormwater wetlands as water levels recede. The topic of fish stranding is discussed further in Section 4.5 below.

There were few strong, ecologically meaningful correlations between habitat variables and fish abundance and diversity. Deeper wetlands generally had greater taxa richness, greater abundance of shortfin eels, and greater total fish abundance, although there was considerable variation in this general pattern. There was also a general trend of reduced shortfin eel length at higher eel densities. Sites with very high shortfin eel abundance also had low densities of smaller-bodied fish species, such as bullies and inanga that could be potential prey items for larger eels. These general trends suggest that predation pressure on all fish, including smaller eels, may be high in the deeper stormwater wetlands with high shortfin eel densities. However, this suggestion is not supported by strong data and requires confirmation via a more detailed investigation into eel diets in stormwater wetlands.

The outlets of stormwater wetlands typically have various types of constrictions that are designed to detain water. These flow constrictions can also act as migratory barriers, restricting the range of fish species that can enter a wetland. Culvert pipes were amongst the most common type of structures assessed and they ranged from presenting low to very high risk to fish passage. Low risk culverts were typically relatively short (tens, rather than hundreds of metres long), and had adequate water depth and slow velocities to provide passage for a range of fish species and life stages. Culvert attributes contributing to their classification as high or very high risk structures included very long lengths, shallow depths, swift velocities, and a vertical drop, or perch, at the outlet. Shortfin eels were the only diadromous fish species caught at sites with very high risk outlet structures. Juvenile eels are exceptional climbers and they can scale formidable vertical barriers (Franklin *et al.* 2018), which explains their presence upstream of structures that present a very high risk to fish passage.

Some diadromous species that lack strong climbing abilities were occasionally found upstream of structures that were assessed as high or very high risk. For example, inanga had previously been recorded at Site 16 (Arthur Adcock), which has a weir (assessed as very high risk) and a series of steps at the pond outlet (Figure 7), and it is piped for 1.7 km from the pond outlet to the sea. Inanga are generally considered to be weak climbers (Franklin *et al.* 2018), so their presence at Site 16 is perplexing. One possible explanation is that juvenile inanga (whitebait) can access the wetland when very high tides or heavy seas coincide with high pond levels, resulting in water levels backing-up through the pipe and over-topping the weir. Backwatering events may similarly explain the presence of inanga at Site 8 (Halswell Downs), where the valve outlet was assessed as a high risk structure and did not appear to provide fish passage for inanga.

Based on a review of purpose-built fish exclusion structures in natural waterways (Charters 2013), excluding all fish species – including shortfin eels – from a stormwater wetland would likely involve the combination of a perched and undercut outlet, preferably with a large drop (≥1.5 m), a sharp edge to the outlet lip, and no wetted margins for climbing species. Providing a large drop may seem impractical at many stormwater facilities in Christchurch, where the topography is often very flat, but it could be achieved in at least some locations using an outlet weir with a sharp and undercut crest.

Designing a wetland with no surface water present during baseflow conditions would also prevent the establishment of any permanent fish population. This is impractical in many locations in Christchurch that have high groundwater levels and are naturally prone to surface ponding. However, the outlet could be designed so that there is no surface water present during baseflow conditions. This might involve discharging via a channel filled with a porous gravel filter media.

In summary, key features that could be incorporated into future stormwater wetland designs to specifically include or exclude fish relate to the presence of surface water and the relative fish passage risk of downstream structures. Wetland design will need to vary according to local conditions, and the desired outcomes, both in terms of water quality treatment and flood mitigation, and whether fish are wanted in the wetland. We conclude that if wetlands are designed to intentionally exclude fish in Christchurch, they should incorporate design features that are known to restrict fish presence, while acknowledging that some fish may still enter the wetland. Design features that will restrict fish presence include: long culverts with shallow, swift water; perched and undercut outlets; vertical, sharp-edged weirs; flap gates; and lack of surface baseflow at the outlet that may attract fish into the wetland. Fish relocation may need to be considered for those tenacious fish that are able to navigate past the barriers put in their way, depending on water quality and habitat conditions within the wetland. All wetland designs, including those intended to exclude fish, should aim to facilitate downstream fish passage to avoid trapping any fish that do manage to enter the wetland.

4.3. How Successful are Wetland Designs that Address Fish Passage?

There was little readily available information to confirm whether Council stormwater wetlands had been specifically designed to provide for fish passage. Fish passage was clearly not a consideration for some of the earliest stormwater wetlands, such as Site 5 (Wigram), based on their complex outlet structures. Fish passage would also not have been a consideration for stormwater wetlands such as Site 17 (Knights), which lack surface water during baseflow conditions. However, we are unaware of any Council stormwater wetlands that were specifically designed to exclude fish.

At least three of the stormwater wetlands we sampled have been commissioned within the last five years and they were specifically designed to provide fish passage. These wetlands are Site 3 (Bullers), Site 13 (Burlington), and Site 6 (Ngā Puna Wai). All three are examples of ‘online’ stormwater wetlands, with open waterways flowing into them (as opposed to ‘offline’ wetlands with a piped network upstream). The following paragraphs discuss how successful each of the fish passage designs are, based on our assessment of the outlet structures and the fish community present.

Site 3 (Bullers) has a fish-friendly flap gate at its outlet, which is designed to remain open during higher flows, and we assessed as presenting a low risk to fish passage. We caught both shortfin and longfin eels in the wetland. There are NZFFD records of common bully and upland bully from the adjacent Buller Stream. There is adequate potential habitat present in the adjacent stormwater wetland for upland and common bullies, and the outlet is unlikely a barrier for bullies. The absence of bullies in the wetland may therefore be due to other factors, such as unsuitable water quality (a maximum temperature of 19 °C was recorded in early April) or predation.

Site 13 (Burlington) was designed to provide fish passage and habitat for inanga and shortfin eel, which have been recorded in the adjacent Watsons Drain (Ho *et al.* 2018). We note that there are also NZFFD records of upland bully from Watsons Drain. However, upland bully are non-diadromous, so provision of fish passage for upland bully is not a priority for wetland design. At Burlington, a wetland flows into a detention basin via a pipe culvert, and the detention basin discharges into Horners Drain via an outlet orifice, pipe culvert and naturalised channel. We assessed the outlet orifice and culvert as presenting a medium risk to fish passage and the culvert between the wetland and basin as a low risk, due to the presence of spoiler baffles to enhance fish passage (Figure 7). We caught shortfin eel, inanga, common bully, and upland bully during our fish survey. The species richness of four put Burlington in the top three out of the 21 wetlands we sampled. This confirms that the fish passage designs were successful.

Although there is reasonable fish passage into the stormwater wetlands at Burlington, the fish community present may be adversely affected by other environmental factors in the wetland. We recorded a minimum DO concentration of 3.3 mg/L at Burlington in early May (Table 4), which is below the NPSFM national bottom line of 4 mg/L. Dissolved oxygen concentrations will drop further over summer, so it is likely that sensitive fish and invertebrate species may be affected by low DO concentrations in the wetland complex. Another factor potentially affecting native fish in Burlington is the introduction of invasive fish species. We understand that ornamental fish (presumably goldfish) were recently introduced into the wetland. As noted in the previous section, goldfish are an invasive species and they can negatively affect water quality and native fish species. Furthermore, goldfish have been implicated in worsening the effects of toxic algal blooms, due to the activation of toxic algae as they pass through goldfish guts (Morgan and Beatty 2007). We did not detect any goldfish or any other exotic fish species during our survey of Burlington. Regardless, we recommend against any further introduction of exotic fish species, as well as commencing a surveillance and eradication programme to get rid of any invasive species that are already present.

The stormwater wetland at Site 6 (Ngā Puna Wai) is a recent extension of the existing Wigram Retention Basin (Site 5). The Wigram basin was built nearly 30 years ago, and it had a complicated outlet, comprising a mix of weirs, valves, and pipes that created a formidable fish barrier. Despite this, shortfin eels were abundant in the wetland, although there was limited opportunity for mature eels to migrate downstream. The Ngā Puna Wai extension included provision for downstream passage for fish from the combined Wigram and Ngā Puna Wai wetlands, but upstream passage was not enhanced. The new outlet includes a weir, gate valve, and approximately 360 m long concrete pipe that discharges into the Heathcote River. We assessed the outlet as presenting a very high risk to fish passage, based on the barrier it presents to fish migrating upstream. However, the new outlet does provide improved passage for adult eels migrating downstream to spawn. Wigram basin has high levels of stormwater contaminants and the implications of this for the fish community are discussed in the following section.

In summary, the three stormwater wetlands we reviewed that incorporated fish passage into their designs all appear to have met their design briefs. As noted above and discussed further below, it is highly likely that poor water quality affects the fish community to varying extents in each of the three wetlands.

4.4. Are Fish Adversely Affected by Water Quality or Other Factors?

Research in Australia indicates that fish communities in stormwater wetlands could be adversely affected by accumulated stormwater contaminants and other factors, such as invasive species (Hale *et al.* 2019). Data on stormwater contaminant levels was unavailable for most of the wetlands we sampled at the timing of writing. However, recent water quality monitoring at Site 17 (Knights) and Site 12 (Prestons) confirmed each stormwater wetland was effective at removing over 50% of metals, suspended sediment, and some nutrients prior to discharge (Allan 2022). Similarly, water quality monitoring in the Halswell Retention Basin in Springs Halswell Reserve, a wetland not sampled in the current study, recorded a decline in many common urban contaminants at the outlet, when compared to the inlet (Margetts and Marshall 2020). In addition, these contaminants were generally found in higher concentrations in the Halswell Retention Basin than in waterway monitoring sites around the city. These studies confirm that local stormwater wetlands are fulfilling their function of improving discharge water quality, by accumulating contaminants within the wetland complex.

The degree of contaminant accumulation will presumably vary according to various factors including upstream landuse, wetland age and its configuration. A survey of sediment quality in the Heathcote River catchment found median sediment zinc concentrations at Site 5 (Wigram) were approximately 1,600 mg/kg, or four times the environmental guideline of 410 mg/kg[6], and over three times sediment concentrations in the river downstream (Oddy 2019). The catchment upstream of Site 5 is dominated by industrial landuse, with several zinc electroplaters, so the level of zinc contamination in the wetland is likely higher than in other stormwater wetlands that principally drain residential landuse. Sampling of fish tissues for heavy metals in Christchurch waterways to date has focussed on lower catchment rivers and the estuary (McMurtrie 2015; McMurtrie 2019), so impacts on fish living within stormwater wetlands is currently unknown. However, the limited water and sediment quality data indicate that concentrations of stormwater contaminants in Christchurch stormwater wetlands are sufficiently high to have adverse effects on the fish communities present, and this warrants further investigation.

The limited monitoring of DO and temperature we undertook suggests that low DO concentrations and high summer water temperatures are likely contributing factors to the dominance of shortfin eels at the stormwater wetlands we sampled. Conversely, the absence of more sensitive species such as inanga at most sites sampled likely reflects a combination of fish passage restrictions and poor water quality. Temperature monitoring in autumn suggested many of the wetlands will exceed 20 °C in the warmer summer months. By comparison, preferred water temperatures for different life stages of inanga are 18.1–18.8 °C,

[6] GV-high default guideline value from the Australian and New Zealand guidelines for fresh and marine water quality; https://www.waterquality.gov.au/anz-guidelines.

whereas shortfin eels have preferred temperatures of 26.9 °C (Richardson *et al.* 1994). In a review of dissolved oxygen criteria for New Zealand freshwater fish, Franklin (2014) stated that waterways with consistently low dissolved oxygen concentrations (<3 mg/L) are likely to limit the presence of sensitive fish species such as inanga and brown trout, and instead be dominated by tolerant species, such as shortfin eel and exotic species such as goldfish. In addition, fish communities at wetland sites with intermediate dissolved oxygen concentrations (<6 mg/L) are likely to suffer chronic health effects, such as slower growth rates, and reduced fecundity or recruitment (Franklin 2014).

Cyanobacteria blooms have previously been detected in some of the stormwater wetlands we sampled, including Site 4 (Quaifes) and Site 16 (Arthur Adcock). Toxic cyanobacteria can adversely affect fish communities, as well as being harmful to humans and other animals, and they are often associated with waterbodies with elevated nutrients and low dissolved oxygen levels (Wood *et al.* 2017; Huisman *et al.* 2018). Furthermore, as noted in the previous section, invasive goldfish may exacerbate the toxic effects of cyanobacterial blooms (Morgan and Beatty 2007). Cyanobacterial blooms and goldfish have been recorded at Site 16, so there is the potential for adverse impacts on the native fish community at this site. However, it is uncertain with the limited information available whether cyanobacteria are negatively affecting fish communities in Council stormwater wetlands.

In summary, there is sufficient data to conclude poor water quality is likely adversely affecting fish communities in Christchurch stormwater wetlands, and this warrants further investigation.

4.5. Are Fish Being Trapped in Council Stormwater Wetlands?

An ecological trap occurs when animals mistakenly select habitats where their fitness is reduced (Robertson and Hutto 2006). Several reviews have concluded that constructed wetlands may either function as ecological traps or as important nodes of aquatic habitat and biodiversity (Clevenot *et al.* 2018; Hale *et al.* 2019; Zhang *et al.* 2020). That is because constructed wetlands generally accumulate pollutants that may be toxic to resident fauna, but they may also represent a habitat that is otherwise scarce in a highly modified environment. Hale *et al.* (2018) concluded that stormwater wetlands were likely ecological traps for the native fish *G. pusilla* in Melbourne, because stormwater wetlands represent a large proportion of habitats available, they are attractive to animals, and they reduce their fitness. Thus, the likelihood of Council stormwater wetlands being ecological traps may be greater in catchments where natural wetland habitats are uncommon compared to stormwater wetlands.

Although wetlands were historically widespread in Christchurch, most wetlands were drained as the city developed. There are still some areas of wetland in the Avon River catchment, most notably around Travis Wetland. However, in the Heathcote River catchment, there are very few natural wetlands and a comparatively large number of stormwater wetlands. There is evidence of high levels of heavy metal contamination in older stormwater wetlands in the upper Heathcote River catchment, as well as low DO concentrations and high temperatures that may affect fish communities (see Section 4.4 above). It is therefore likely that some stormwater wetlands in Christchurch are ecological traps, particularly those in the Heathcote catchment.

In addition to acting as ecological traps, some stormwater wetlands in Christchurch may also be physical traps for native fish species. We observed shortfin eels below the grates of basins that are joined by pipes at Site 7 (Eastman). As water levels drop below the basin invert, eels become trapped in the connecting pipes. It is uncertain whether the pipes completely dry out, but the water quality in the pipes is clearly poor, as we measured a DO concentration of 0.22 mg/L at a newly constructed access manhole on 18/5/2022. Fish that opportunistically move into temporarily flooded wetland habitat, as was observed at Site 18 (Douglas Clifford), may also become stranded as flood waters recede. Eels are cryptic species, so it may be difficult to determine the extent of fish stranding if there is abundant vegetation cover or mud, as they will tend to bury into mud and vegetation, rather than lying exposed on the surface. This contrasts with other large-bodied species such as trout, which are often found on the sediment surface during stranding events.

In summary, it is likely that fish are being trapped in at least some stormwater wetlands in Christchurch. Traps may be in the form of poor water quality affecting fish fitness and community composition, or in the form of a physical trap, by fish being stranded or caught in piped networks.

5. CONCLUSIONS AND RECOMMENDATIONS

Fish were present at 19 of the 20 stormwater wetlands we sampled. The data from this survey strongly suggest that fish will be present in all of Christchurch's stormwater wetlands that have sufficient surface water. Our data confirm that water does not need to be permanent for fish to be present, with observations of fish moving into temporarily inundated stormwater basins. Shortfin eels were present at all sites that fish were found and they were the most abundant species caught. Results of this pilot study suggest that the fish community of Christchurch stormwater wetlands is restricted by structures that present barriers to fish migration. The three stormwater wetlands we reviewed that incorporated fish passage into their designs all met their design briefs. Poor water quality likely limits wetland suitability for sensitive fish species, such as inanga. We conclude that many stormwater wetlands may be ecological traps.

We recommend the Council reviews its current stormwater planning process to consider the potential negative effects of stormwater wetlands on fish, and that this is done as a priority. National policy directives requiring fish passage past most instream structures is prompting some councils to provide for fish passage past all structures, including stormwater wetlands. Our data indicate that fish entry into stormwater wetlands needs to be carefully considered, given the potential for adverse ecological effects on the resident fish population. Fish may provide important ecological functions in stormwater wetlands, such as predating on the larvae of mosquitos and other insect pests (Christchurch City Council 2003). However, the functional benefits of fish in wetlands needs to be weighed up against potential ecological costs to the fish themselves, in terms of their health and fitness. These considerations must be made in conjunction with Environment Canterbury, who typically require fish passage as a condition of consent for stormwater facilities.

Based on the findings discussed in the preceding sections, we recommend the following:

- Council reviews the planning and design of stormwater wetlands to consider potential negative effects of stormwater wetlands on fish. This should include decisions around whether fish passage is provided for all new stormwater wetlands and whether fish passage should be altered for existing stormwater wetlands.
- When practicable, offline stormwater wetland designs should be selected over online systems. Offline facilities are preferred as fish may be excluded from the stormwater treatment facility, when required, without preventing access to existing upstream habitats.

- Review potential outlet designs for specifically excluding and including fish from stormwater wetlands. The review can build on preliminary data provided in this report.
- Sample fish communities using fine mesh fyke nets at all stormwater wetlands that have not yet been sampled using this method. It is important to understand what fish are present in stormwater wetlands to inform future management, and fine mesh fyke nets proved the most efficient method at sampling the fish community. We found fine mesh fyke nets could be used in even low oxygen conditions minimal no fish mortalities, provided water levels were not elevated by recent rainfall.
- Sample non-stormwater wetlands using the same methods, to compare fish community structure.
- Use public education and surveillance monitoring to prevent invasive fish species from becoming an issue in stormwater wetlands. Eradication of invasive species from stormwater wetlands should be a priority. This should be done in collaboration with the Department of Conservation and Environment Canterbury, who both have interests in pest species management.
- Monitor dissolved oxygen concentrations and water temperature at stormwater wetlands over the summer months. If dissolved oxygen and temperature conditions are unfavourable for fish, review options for remedying. Solutions may involve retrofitting the wetland to improve water quality conditions or to exclude fish, as well as relocating fish out of the wetland.
- Sample water and sediment concentrations of metals and other common stormwater contaminants in stormwater wetlands with fish (i.e., where surface water is present).
- Undertake more detailed studies of fish communities in stormwater wetlands, to better understand fish health. Investigations may include analysis of metals in fish tissues, measuring fish growth rates, and analysis of gut contents.
- National research into fish in stormwater wetlands, given the national relevance of the issues raised in this pilot study.

6. REFERENCES

Allan, L. (2022). Knights Stream and Prestons Stormwater Facility monitoring 2020-2021 annual report. *Report prepared by PDP for Christchurch City Council, June 2022*.

Biggs, B. J. F., and Kilroy, C. (2000). 'Stream periphyton monitoring manual'. (Prepared for the Ministry for the Environment. NIWA: Christchurch.)

Brown, T. R., Coleman, R. A., Swearer, S. E., and Hale, R. (2018). Behavioral responses to, and fitness consequences from, an invasive species are life-stage dependent in a threatened native fish. *Biological Conservation* **228**, 10–16. doi:10.1016/j.biocon.2018.10.003

Charters, F. (2013). Waterway barrier design for protection of native aquatic values. *Report prepared by University of Canterbury for Department of Conservation, October 2013*.

Christchurch City Council (2003). 'Waterways, wetlands and drainage guide - Ko Te Anga Whakaora mō Ngā Arawai Rēpō. Part B: Design'. (Christchurch City Council: Christchurch, New Zealand.)

Clarke, K. R. (1993). Non-parametric multivariate analyses of changes in community structure. *Australian Journal of Ecology* **18**, 117–143.

Clevenot, L., Carré, C., and Pech, P. (2018). A review of the factors that determine whether stormwater ponds are ecological traps and/or high-quality breeding sites for amphibians. *Frontiers in Ecology and Evolution* **6**, 1–12. doi:10.3389/fevo.2018.00040

Collier, K. J., and Grainger, N. P. J. (Eds.) (2015). 'New Zealand invasive fish management handbook'. (University of Waikato and Department of Conservation: Hamilton, New Zealand.)

Dixon, P. (2003). VEGAN, a package of R functions for community ecology. *Journal of Vegetation Science* **14**, 927–930.

Dunn, N. R., Allibone, R. M., Closs, G. P., Crow, S. K., David, B. O., Goodman, J. M., Griffiths, M., Jack, D. C., Ling, N., Waters, J. M., and Rolfe, J. R. (2018). Conservation status of New Zealand freshwater fishes, 2017. *New Zealand Threat Classification Series 24*, 11.

Franklin, P. (2022). Fish Passage Assessment Tool mobile application user guide - version 2.0. *Prepared for Ministry for Business Innovation & Employment and Ministry for the Environment by NIWA, February 2022*.

Franklin, P. A. (2014). Dissolved oxygen criteria for freshwater fish in New Zealand: a revised approach. *New Zealand Journal of Marine and Freshwater Research* **48**, 112–126.

Franklin, P., Gee, E., Baker, C., and Bowie, S. (2018). New Zealand fish passage guidelines. *NIWA Client Report 2018019HN*, 229.

Hale, R., Coleman, R., Sievers, M., Brown, T. R., and Swearer, S. E. (2018). Using conservation behavior to manage ecological traps for a threatened freshwater fish. *Ecosphere* **9**, 1–13. doi:10.1002/ecs2.2381

Hale, R., Swearer, S. E., Sievers, M., and Coleman, R. (2019). Balancing biodiversity outcomes and pollution management in urban stormwater treatment wetlands. *Journal of Environmental Management* **233**, 302–307. doi:10.1016/j.jenvman.2018.12.064

Ho, M., Stone, M., and Taylor, M. (2018). Fish passage design for ecological connectivity to a stormwater facility. *Water New Zealand Stormwater Conference*, 20.

Huisman, J., Codd, G. A., Paerl, H. W., Ibelings, B. W., Verspagen, J. M. H., and Visser, P. M. (2018). Cyanobacterial blooms. *Nature Reviews Microbiology* **16**, 471–483. doi:10.1038/s41579-018-0040-1

Jellyman, D. J. (1989). Diet of two species of freshwater eel (Anguilla spp.) in Lake Pounui, New Zealand. *New Zealand Journal of Marine and Freshwater Research* **23**, 1–10. doi:10.1080/00288330.1989.9516334

Joy, M., David, B., and Lake, M. (2013). 'New Zealand freshwater fish sampling protocols. Part 1 - Wadeable rivers and streams'. (The Ecology Group - Institute of Natural Resources, Massey University: Palmerston North.)

Margetts, B., and Marshall, W. (2020). Surface water quality monitoring report for Christchurch City waterways: January – December 2019. *Christchurch City Council Report*.

McIntosh, A. R., and McDowall, R. M. (2004). Fish communities in streams and rivers. In 'Freshwaters of New Zealand'. (Eds J. Harding, P. Mosley, C. Pearson, and B. Sorrell.) (New Zealand Hydrological Society and New Zealand Limnological Society: Christchurch.)

McMurtrie, S. (2019). Food Safety of fish & shellfish in Ōtautahi/Christchurch – 2019 survey. *EOS Ecology Report No. ENV01-18103-01, July 2019*.

McMurtrie, S. (2015). Heavy metals in Christchurch fish and shellfish: 2014 survey. *EOS Ecology Report No. 08002-ENV01-04, prepared for Environment Canterbury, May 2015*.

Ministry for the Environment (2020). National Policy Statement for Freshwater Management 2020 (NPS-FM).

Morgan, D. L., and Beatty, S. J. (2007). Feral Goldfish (Carassius auratus) in Western Australia: A case study from the Vasse River. *Journal of the Royal Society of Western Australia* **90**, 151–156.

Oddy, G. (2019). Contaminants within urban waterways sediment and the implications for flood protection projects. *New Zealand Stormwater Conference and Expo*.

Oksanen, J., Kindt, R., Legendre, P., O'Hara, B., Stevens, M. H. H., Oksanen, M. J., and Suggests, M. (2007). The vegan package. *Community ecology package* **10**, 719.

Richardson, J., Boubée, J. A. T., and West, D. W. (1994). Thermal tolerance and preference of some native New Zealand freshwater fish. *New Zealand Journal of Marine and Freshwater Research* **28**, 399–407.

Robertson, B. A., and Hutto, R. L. (2006). A framework for understanding ecological traps and an evaluation of existing evidence. *Ecology* **87**, 1075–1085. doi:10.1890/0012-9658(2006)87[1075:AFFUET]2.0.CO;2

Ryan, P. A. (1986). Seasonal and size-related changes in the food of the short-finned eel, Anguilla australis in Lake Ellesmere, Canterbury, New Zealand. *Environmental Biology of Fishes* **15**, 47–58. doi:10.1007/BF00005388

Tilton, D. L. (1995). Integrating wetlands into planned landscapes. *Landscape and Urban Planning* **32**, 205–209. doi:10.1016/0169-2046(95)07001-B

Wood, S. A., Puddick, J., Fleming, R., and Heussner, A. H. (2017). Detection of anatoxin-producing Phormidium in a New Zealand farm pond and an associated dog death. *New Zealand Journal of Botany* **55**, 36–46. doi:10.1080/0028825X.2016.1231122

Zhang, C., Wen, L., Wang, Y., Liu, C., Zhou, Y., and Lei, G. (2020). Can constructed wetlands be wildlife refuges? A review of their potential biodiversity conservation value. *Sustainability* **12**, 1–18. doi:10.3390/su12041442

APPENDIX 1: SITE PHOTOGRAPHS FROM 2022

Figure 1: Site 1, Sparks Wetland.

Figure 2: Site 2, Sparks First Flush.

Figure 3: Site 3, Bullers.

Figure 4: Site 4, Quaifes.

Figure 5: Site 5, Wigram.

Figure 6: Site 6, Ngā Puna Wai.

Figure 7: Site 7, Eastmans.

Figure 8: Site 8, Halswell Downs.

Figure 9: Site 9, Portlink.

Figure 10: Site 10, Charlesworth.

Figure 11: Site 11, Clare Park.

Figure 12: Site 12, Prestons.

Figure 13: Site 13, Burlington.

Figure 14: Site 14, Alpine View.

Figure 15: Site 15, Ryman.

Figure 16: Site 16, Arthur Adcock.

Figure 17: Site 17, Knights.

Figure 18: Site 18, Douglas Clifford.

Figure 19: Site 19, Spring Grove.

Figure 20: Site 20, Beckenham.

Figure 21: Site 21, Te Oranga Waikura.

APPENDIX 2: FISH BARRIER ASSESSMENTS

Table 1: All structures assessed for fish passage downstream of fish sampling locations. Key barriers are the highest risk structures affecting passage into each wetland.

Site No.	Site Name	Key Barrier?	Risk to Passage	Structure type	Survey Date	Council ID	FPAT ID	East (NZTM)	North (NZTM)
1 & 2	Sparks Wetland & First Flush	Y	Low	Culvert or pipe	18/05/22	SwPipe 94288	143328	1567307	5175242
2	Sparks First Flush		Low	Weir dam or flow restriction	18/05/22	WcWeirs 251	143321	1566884	5175492
2	Sparks First Flush		Low	Weir dam or flow restriction	18/05/22	WcWeir 250	143325	1566924	5175607
2	Sparks First Flush		Low	Bridge	18/05/22	No ID	143326	1566872	5175537
2	Sparks First Flush		Low	Culvert or pipe	18/05/22	SwPipe 94287	143327	1567236	5175206
2	Sparks First Flush		Low	Weir dam or flow restriction	18/05/22	WcWeirs 253	143329	1567252	5175259
3	Bullers	Y	Low	Gate or valve	4/01/21	SwValve 624	133185	1570896	5184672
3	Bullers		Low	Weir dam or flow restriction	19/05/22	WcWeirs 249	143357	1570911	5184683
3	Bullers		Low	Weir dam or flow restriction	19/05/22	WcWeirs 245	143358	1570767	5184847
3	Bullers		Low	Weir dam or flow restriction	19/05/22	WcWeir 246	143359	1570930	5184798
4	Quaifes	Y	High	Weir dam or flow restriction	28/03/22	No ID	142902	1563423	5174342
4	Quaifes		Very High	Weir dam or flow restriction	28/03/22	No ID	142903	1563351	5174307
4	Quaifes		High	Weir dam or flow restriction	28/03/22	No ID	142904	1563398	5174331
4	Quaifes		High	Other	28/03/22	SwPipeID 92592	142905	1563406	5174000
4	Quaifes		Very High	Other	28/03/22	SwPipeID 92593	142906	1563375	5174025
4	Quaifes		High	Other	28/03/22	SwPipeID 92594	143118	1563304	5174090
5 & 6	Wigram & Ngā Puna Wai		Medium	Culvert or pipe	10/01/20	WcWeirs 5	1624	1565716	5177810
5 & 6	Wigram & Ngā Puna Wai	Y	Very High	Culvert or pipe	11/12/19	SwPipe 96031	1875	1565762	5177734
5 & 6	Wigram & Ngā Puna Wai		Very High	Other	12/12/19	No ID	1877	1566041	5177631

Site No.	Site Name	Key Barrier?	Risk to Passage	Structure type	Survey Date	Council ID	FPAT ID	East (NZTM)	North (NZTM)
5 & 6	Wigram & Ngā Puna Wai		Low	Culvert or pipe	18/05/22	SwPipe 96026	143331	1565645	5177478
5 & 6	Wigram & Ngā Puna Wai		Very Low	Bridge	18/05/22	No ID	143332	1565745	5177789
5 & 6	Wigram & Ngā Puna Wai		Medium	Culvert or pipe		SwPipe 96025	No FPAT Record	1565667	5177422
7	Eastman	Y	Very High	Culvert or pipe	17/05/22	SwPipe 87528	143362	1566576	5174720
8	Halswell Downs		Low	Culvert or pipe	17/05/22	No ID	143317	1566518	5174428
8	Halswell Downs		Very High	Culvert or pipe	17/05/22	SwPipe 97921	143456	1566359	5174336
8	Halswell Downs		Low	Culvert or pipe	30/05/22	SwPipe 97924	143735	1566374	5174455
8	Halswell Downs	Y	High	Gate or valve	17/05/22	SwPipe 97925	143878	1566514	5174434
9	Portlink	Y	Very High	Culvert or pipe	22/05/22	SwPipe 74014	143455	1575029	5176785
10	Charlesworth	Y	Medium	Culvert or pipe	18/05/22	SwPipe 4042	143341	1575770	5178129
11	Clare Park		High	Culvert or pipe	19/05/22	SwPipe 92579	143353	1574050	5184542
11	Clare Park		Low	Culvert or pipe	19/05/22	SwPipe 92577	143354	1574087	5184720
11	Clare Park	Y	Medium	Culvert or pipe	19/05/22	SwPipe 92573	143355	1574018	5184904
11	Clare Park		Low	Weir dam or flow restriction	25/05/22	SAP IE000000000011367606	143515	1573980	5184871
12	Prestons		Medium	Culvert or pipe	11/05/22	SwPipe 84352	143263	1572854	5187892
12	Prestons		Low	Culvert or pipe	11/05/22	SwPipe 92584	143264	1572812	5188060
12	Prestons	Y	Medium	Culvert or pipe	12/05/22	SwPipe 84354	143265	1572780	5187986
13	Burlington		Low	Culvert or pipe	12/05/22	SwPipe 96998	143266	1570328	5186793
13	Burlington		Low	Culvert or pipe	12/05/22	SwPipe 96982	143267	1570421	5186786
13	Burlington	Y	Medium	Culvert or pipe	24/05/22	SwPipe 96988	143497	1570565	5186716
14	Alpine View	Y	Low	Culvert or pipe	28/04/22	SwPipe 68570	143259	1573600	5186553
14	Alpine View		Low	Culvert or pipe	28/04/22	SwPipe 81350	143260	1573581	5186646
14	Alpine View		Low	Culvert or pipe	11/05/22	SwPipe 6868	143261	1573604	5186485
15	Ryman	Y	Very High	Culvert or pipe	19/05/22	SwPipe 82033	143356	1571179	5184338
16	Arthur Adcock	Y	Very High	Weir dam or flow restriction	18/05/22	SwPipe 41502	143342	1576340	5185985
17	Knights		Very High	Pump	28/03/22	PS0214 Marshs Rd SW	142900	1562739	5175172

Site No.	Site Name	Key Barrier?	Risk to Passage	Structure type	Survey Date	Council ID	FPAT ID	East (NZTM)	North (NZTM)
17	Knights	Y	High	Other	28/03/22	SwPipe 1419	142901	1562723	5175196
18	Douglas Clifford		Low	Culvert or pipe	18/05/22	SwPipe 23607	143319	1565444	5175727
18	Douglas Clifford	Y	Medium	Culvert or pipe	18/05/22	SwPipe 78909	143877	1565567	5175670
19	Spring Grove	Y	Very High[1]						
20	Beckenham	Y	Very High	Culvert or pipe	22/05/22	SwPipe 72093	143454	1571560	5176712
21	Te Oranga Waikura		Low	Weir dam or flow restriction	31/01/21	SwFlowRestriction 49	134107	1573765	5178713
21	Te Oranga Waikura	Y	Low	Culvert or pipe	28/04/21	SwPipe 92927	136472	1573687	5178644

Note: [1] There are multiple pipe outlets from Site 19 (Spring Grove), which are all perched and present a very high risk to passage. No single structure was assessed.

APPENDIX 3: FISH DATA

Table 1: Fine mesh fyke net fish sampling results. Data are total fish count from five nets, with the size range (mm) in brackets. Asterisks indicate sites where minnow traps were deployed at the same time (minnow trap data are reported separately in Table 3).

Site No.	Site Name	Shortfin Eel	Longfin Eel	Elver	Inanga	Common Bully	Upland Bully	Juvenile Bully	Brown Trout
1	Sparks Wetland	142 (295–759)	8 (459–1218)		162 (49–92)	1 (39)	16 (36–53)	57 (21–40)	
2	Spark Road First Flush	802 (156–620)		9 (106–155)	5 (66–82)		17 (37–61)	15 (30–40)	
3*	Bullers	67 (387–739)	2 (595–771)						
4	Quaifes	10 (308–822)							
5	Wigram	210 (245–767)							1 (537)
6	Ngā Puna Wai	230 (110–661)							
7	Eastman	298 (144–685)					2 (40–43)		
8	Halswell Downs	226 (179–631)		1 (144)	4 (66–82)		6 (42–59)		
10*	Charlesworth	185 (204–841)	4 (635–1040)		169 (36–95)	118 (38–80)		77 (24–60)	
12*	Prestons	181 (261–581)							
13*	Burlington	23 (287–702)			3 (89–144)	3 (56–66)	11 (42–57)		
15	Ryman	5 (427–763)							
18	Douglas Clifford	404 (122–881)							

Table 2: Coarse mesh fyke net sampling results. Data are total fish count, with the size range (mm) in brackets. Results are from five nets, except Site 9, where only four nets were used. Minnow traps were also deployed at all sites, but minnow trap data are reported separately in Table 3.

Site No.	Site Name	Shortfin Eel	Longfin Eel	Brown Trout
1	Sparks Wetland	48 (351–776)	3 (680–1048)	
5	Wigram	88 (357–717)	1 (391)	1 (534)
9	Portlink	1 (432)		
11	Clare Park	8 (472–740)		
14	Alpine View	105 (289–792)		
16	Arthur Adcock	88 (295–655)		
19	Spring Grove	29 (161–606)		
20	Beckenham	11 (365–638)	4 (525–845)	
21	Te Oranga Waikura	164 (245–783)		

Table 3: Minnow trap fish sampling results. Data are total fish count, with the size range (mm) in brackets. Data are totals from eight traps, except for Site 9, where seven traps were used, and Site 17, where one trap was used.

Site No.	Site Name	Shortfin Eel	Inanga	Common bully	Upland bully
1	Sparks Wetland		2 (70–77)		8 (42–46)
3	Bullers				
5	Wigram				
9	Portlink				
10	Charlesworth Pond[1]		1	8	
11	Clare Park				
12	Prestons				
13	Burlington			1 (51)	1 (51)
17	Knights				
18	Douglas Clifford	1 (324)			3 (48–63)

Note: [1] Lengths were not measured for the inanga and common bullies caught in the minnow traps at Charlesworth Pond. This was because 50 individuals of each of these species had already been measured from the fyke nets.