Number One Drain Restoration and Ecological Response

September 2020

Prepared for:
Christchurch City Council

Instream Consulting Limited
PO Box 28 173
Christchurch 8242

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EXECUTIVE SUMMARY

No. 1 Drain is an artificial waterway that flows through Christchurch Golf Club before entering Horseshoe Lake. Following damage to the bed and banks that occurred during the Canterbury earthquake sequence of 2010–2011, Christchurch City Council enhanced and restored a section of the waterway. Additionally, an in-line stormwater pond system was created, to improve downstream water quality and provide flood attenuation. This report assesses the effectiveness of ecological restoration activities by comparing ecological surveys conducted prior to restoration in May 2016 with post-restoration data collected in May 2020.

Restoration of the waterway involved realignment of the channel and reformation of the banks. Vertical concrete banks were replaced with more natural banks, native rushes were planted on the waters margin, and cobbles were added at some sites. Two in-line stormwater ponds were added that were planted with native rushes along their margins. The ponds also had five artificial floating wetlands, consisting of a floating base planted with native sedges.

Aquatic habitat improvements included increased substrate complexity, greater macrophyte cover, and more overhanging vegetation. Fine sediment cover remained high, with water velocities too low to prevent sedimentation. Habitat modifications were associated with substantially improved dissolved oxygen levels, which were very low during pre-restoration sampling. However, dissolved oxygen levels in the ponds may get much lower during summer, when water temperatures and plant respiration rates increase.

Macroinvertebrate communities responded positively to the restoration efforts, with increased taxa richness, diversity, MCI, and QMCI scores in 2020. Despite improvements in these metrics, the community remained dominated by pollution-tolerant taxa, with QMCI values below the LWRP Freshwater Outcome value of 3.5 at all sites.

A positive response was also observed in the fish community, with increased species richness. A total of five species were recorded, an increase of two from the baseline survey. Fish abundance remained low through the flowing sections but were high in the ponds. The ponds were also the preferred habitat of larger eels.

Overall, the restoration of No. 1 Drain has successfully enhanced aquatic habitat and improved the diversity of invertebrate and fish communities.

We recommend summer monitoring of dissolved oxygen, temperature, and the presence of toxic algae in the ponds, as there is a risk of elevated temperatures and low dissolved oxygen occurring over summer. Further monitoring of fish populations in the ponds is recommended during the summer, when adult and juvenile inanga may be present. Fish passage through the new tide gates in Horseshoe lake should be assessed, to determine if the new upstream habitat is being fully utilised by migratory species, especially inanga.

1. INTRODUCTION

Number One Drain (No. 1 Drain) is an artificial waterway that flows through the Christchurch Golf Club in the suburb of Shirley (Figure 1). The drain was previously concrete lined through the golf course, but Christchurch City Council (CCC) restored and enhanced it in response to damage to the bed and banks of the drain following the Canterbury earthquake sequence of 2010 and 2011. Restoration and enhancement activities included channel realignment and naturalisation. Two in-line stormwater ponds with floating wetlands were also included to provide flood attenuation and improve stormwater quality. The initial restoration activities were completed in November 2018 and the wetlands were constructed in September 2019.

Baseline aquatic ecology sampling was conducted in No. 1 Drain in May 2016, prior to any restoration work (Instream Consulting 2016b). Follow-up ecological monitoring was conducted in the drain and in the new stormwater ponds in May 2020, after all restoration activities were completed. This report compares the pre- and post-restoration sampling results, to assess how successful restoration activities have been in improving ecological health.

2. METHODS

2.1. Study Area

No. 1 Drain is piped upstream of Golf Links Road, then flows for approximately 500 m through Christchurch Golf Club before passing through a culvert under Horseshoe Lake Road. The waterway then flows along a reach approximately 240 m long before discharging into Broomfield Waterway and then Horseshoe Lake. The outlet of Horseshoe Lake is regulated by tide gates and a pump station. In 2016, numerous small drains, approximately 30 mm diameter and 1 m apart, entered both sides of No. 1 Drain along the golf course, but these were absent from the restored reaches in 2020.

Flow in No. 1 Drain is from land drainage during baseflow conditions. Prior to restoration activities, flows would rapidly increase in the drain following rainfall due to stormwater runoff from the piped headwaters upstream and from the adjacent golf course. Prior to the recent restoration work, there was no stormwater detention or treatment in the No. 1 Drain catchment, as residential development in the area pre-dated modern stormwater treatment design and regulations. Restoration activities included the addition of two stormwater ponds and floating wetlands, designed to attenuate flood flows and treat stormwater runoff.

Four locations along the flowing section of No. 1 Drain were sampled before and after restoration work (Figure 1, Table 1). The site locations during the follow-up survey were within 8 m of the baseline survey locations, except for Site 3. The downstream extent of Site 3 was shifted approximated 30 m downstream from its previous location. This was done so that the entire reach was in flowing habitat, downstream of the second pond.

Sites 1 and 4 were originally selected as control sites, upstream and downstream of proposed waterway restoration, respectively, while Sites 2 and 3 were within the restoration area and were proposed as treatment sites. However, some limited habitat restoration did in fact occur in Site 1, so it is now considered a treatment site.

Sites 1, 2, and 3 were all located on No. 1 Drain, while Site 4 was located further downstream on Broomfield Waterway because the section of No. 1 Drain immediately downstream of

Horseshoe Lake Road was too sluggish and overgrown with aquatic macrophytes during the baseline survey to serve as a good comparison for the upper sites.

Four additional sites were sampled in the new stormwater ponds. Sites A and D were sampled at the inlet and outlet of the ponds, respectively (Figure 2, Table 1). Site B was located alongside floating wetlands near the southeast bank of the first stormwater pond, while Site C was located along the northern bank of the second stormwater pond.

Baseline sampling occurred over 4 to 6 May 2016, while the follow-up survey occurred over 20 to 28 May 2020. Both the 2016 and 2020 sampling rounds followed unusually dry summer and autumn periods.

Figure 1: Locations of the flowing sampling sites (Sites 1–4) in No. 1 Drain. Note that aerial imagery is from 2015/16, so it does not show the realigned sections of the waterway or the ponds.

2.2. Water Quality

Dissolved Oxygen (DO), temperature, pH, and conductivity were measured in the field during both survey rounds. These measurements were collected with a Horiba U10 water quality meter during the baseline survey, and a Hanna (model HI 9829) water quality meter during the follow-up survey. On both occasions the water quality meters had been recently calibrated.

Figure 2: Pond sampling sites. Note that in this photograph the floating wetlands have just been planted and are still being moved into their final positions. Photograph taken 12 July 2019 (supplied by CCC).

Table 1: Study site locations. Coordinates mark the downstream end of each reach for stream sites and the central point for pond sites. Coordinates in New Zealand Transverse Mercator 2000 projection.

Site	Easting	Northing	Description
1	1572944	5183400	~4 m downstream of bridge, near workshop
2	1573039	5183402	Immediately upstream of culvert on upstream edge of “green”
3	1573293	5183482	Immediately upstream of footbridge on “rough” boundary
4	1573461	5183956	~50 m upstream of No. 2 Drain confluence
A	1573107	5183427	At the inlet of the most upstream pond
B	1573136	5183398	Alongside the floating wetland, on the southeast bank of the upstream pond
C	1573177	5183473	On the northern bank of the downstream pond
D	1573240	5183457	At outlet of the downstream pond

2.3. Habitat

Quantitative habitat data were collected from the four flowing sites (Sites 1–4), during both survey rounds. Qualitative habitat observations were made for the ponds during the follow-up survey, as standard stream habitat monitoring protocols are not applicable to pond systems, and there was no baseline data for comparison.

Quantitative habitat data were collected using a combination of Protocol P3 of Harding *et al.* (2009), sediment assessment methods 2 and 6 of Clapcott *et al.* (2011), and standard CCC protocols (Instream Consulting 2016a). With the exception of Site 1, each sampling site comprised a 50 m reach of stream, with habitat measurements generally made either as an average for the reach, or along each of five or six transects at 10 m intervals along the reach. Site 1 was only 30 m long, with transects spaced 5 m apart. This length was required due to lack of open drain habitat upstream of the restoration works. During the follow up survey, measurements could only be collected at the three most downstream transects of Site 2, due to golfing activities through the site.

The percentage contribution of run, riffle, and pool habitat was estimated visually for each sampling reach. The total length of the following habitat features were measured along both banks of each reach: gaps in riparian buffer, wetland soils, stable undercuts, livestock access, bank slumping, raw banks, rills/channels, and drains (see Harding *et al.* 2009 for details). Stream shading was measured at 20 random points along each reach using a spherical densiometer.

At 6 transects per site, the following bank features were measured: lower bank height (left and right), lower bank slope, depth of any bank undercuts, and length of overhanging vegetation within 0.3 m of the water surface. Water depth and velocity were measured at sufficient points along each transect to characterise changes in channel profile and velocity, with a minimum of five measurements per transect, as per protocol P3 of Harding *et al.* (2009). Velocity was measured using a calibrated Pygmy RS current meter during the baseline survey, and a Hach FH950.1 electromagnetic velocity meter during the follow-up survey.

At each of the 6 transects per site, substrate size and embeddedness was measured at 10 equidistant points. Embeddedness was assessed using a scale of 1 to 4, with 1 being not embedded and 4 being completed embedded (Harding *et al.* 2009). Fine sediment (<2 mm diameter) cover and depth were measured at 5 points along 6 transects per site. Fine sediment depth was measured by pushing a 10 mm diameter steel rod into the substrate until it hit harder substrates underneath. Sediment compactness was assessed once per transect, using a scale of 1 to 4, with 1 being very loose and 4 being tightly compacted (Harding *et al.* 2009).

At the left, centre, and right edge of 6 transects per site, the following data were recorded:

- Macrophyte cover, composition, and type (emergent and total).
- Periphyton cover and composition, using categories of Biggs and Kilroy (2000).
- Organic matter cover and type.

The width of each transect covered by macrophytes, periphyton, woody debris, and leaf packs were also recorded at 6 transects per site, as per protocol P3 of Harding *et al.* (2009).

Riparian vegetation cover was measured on each bank at five transects per site, at 0.5, 3, 7.5, and 20 m from the bank. Vegetation cover was recorded in each of the following height tiers: 0–0.3 m, 0.6–1.9 m, 2.0–4.9 m, 5–12 m, and > 12 m. Dominant vegetation was also recorded.

2.4. Macroinvertebrates

Benthic macroinvertebrates were collected from the flowing sites (Sites 1–4) using quantitative protocol C3 of Stark *et al.* (2001). Briefly, this involved disturbing the bed within a 0.1 m² area and collecting invertebrates in a 500 µm mesh net, with five replicate samples collected per site. Samples were preserved in 70% ethanol solution and were processed by Ryder

Consulting Limited (baseline survey) and Biolive Consultants (follow-up survey). In both instances, samples were processed using the full count with subsample option (protocol P3, Stark *et al.* 2001), and identified to species level where practical.

At the pond sites (Sites A–D), invertebrate samples were collected using semi-quantitative kicknet approach. Briefly this involved collecting a single kicknet sample from the range of available habitats present, in proportion to the habitat types present, and covering a total area of approximately 0.6 m^2. Samples were again preserved in a 70% ethanol solution and processed by Biolive Consultants. Samples from the pond sites were processed using 200 fixed counts with a scan for rare taxa (protocol P2, Stark *et al.* 2001).

2.5. Fish

The fish community at each flowing site (Sites 1–4) was sampled using a Kainga EFM 300 backpack electrofishing machine. Following standard CCC protocols (based on those of Joy *et al.* 2013), the range of habitats present at each site were sampled using a single pass. Stunned fish were either scooped up with a hand net or caught in a stopnet downstream of the catching electrode.

At each of the pond sites (Sites A–D), the fish community was sampled by setting five unbaited fine-mesh fyke nets. At the inlet and outlet sites (Sites A and D, respectively), fyke nets were positioned to capture fish moving into, or out of, the ponds. At Site B, fyke nets were positioned around the edges of one of the floating wetlands. At Site C, the nets were spaced out around the bank, away from the floating wetlands.

Captured fish were transferred to a bucket, then identified, counted, and measured (fork length, mm). Captured eels were anesthetised using an ethanol-clove oil solution to assist the measuring process. Eels were given time to recover from the aesthetic, before being returned to their resident habitats along with the rest of the caught fish.

2.6. Data Analyses

All statistical tests were undertaken using R (R Core Team 2013).

2.6.1. Water Quality

No. 1 Drain and Broomfield Waterway are classified as a Spring-fed Plains-Urban streams under Environment Canterbury's Land and Water Regional Plan (LWRP). Dissolved oxygen data were compared against the LWRP freshwater outcome of a minimum of 70% saturation for Spring-fed Plains-Urban streams. Temperature data were not compared against guidelines, as they were likely cooler than typical summer temperatures. As there is no LWRP outcome value for pH, the LWRP Receiving Water Standard (Schedule 5) range of 6.5–8.5 was used to provide context.

2.6.2. Habitat

Habitat data collected at multiple locations per transect were averaged to get a mean value for each transect. Similarly, data collected separately for each bank were averaged to get a mean value per transect. This is necessary as observations within a transect are not independent and are therefore pseudo-replicates.

Habitat variables were compared amongst sites and years by running two-way, Type III, ANOVA. The Type III sums of squares approach was selected due to the unbalanced nature of the data, a result of the incomplete sampling of Site 2 during the follow-up survey. ANOVA assumptions of normality and homogeneity were tested by running Shapiro-Wilk tests and examining histograms and diagnostic plots, prior to running ANOVA. Where appropriate, data were transformed to meet these assumptions. When the assumptions could not be met with parametric transformations, the data were rank transformed. The models were first run including both the restoration sites and the downstream control, to account for temporal variation not associated with the treatment effect (restoration). The models were then run a second time, excluding the downstream control. Improved significance of year or site x year interaction effects in the second run was used to infer responses associated with the restoration activities.

Bed cover with filamentous algae, macrophytes, and fine sediment were compared against LWRP freshwater outcomes for Canterbury waterways. Relevant outcomes for Spring-fed Plains-Urban streams are <30% cover of long filamentous algae, <30% cover with emergent macrophytes, <60% cover with total macrophytes, and <30% fine sediment cover.

2.6.3. Macroinvertebrates

Several macroinvertebrate taxa were identified to the species level in one of the sampling rounds, but only to the genus level in the other sampling round. This was a result of macroinvertebrate processing being carried out by different taxonomists during each sampling round. To allow for comparisons of invertebrate communities to be made between years, the taxonomic resolution had to be reduced to the genus level in such cases. This affected the following species: *Austrosimulium australense, Chironomus zealandicus, Corynoneura scutellate, Enochrus tritus, Ferissia neozelanica, Gyraulus corinna. Physella (Physa) acuta, and Triplectides cephalotes*.

The following biological indices were calculated from the raw invertebrate data for each of the sampling sites:

Taxa Richness: The number of different invertebrate taxa (families, genera, species) at a site. Richness may be reduced at impacted sites, but is not a strong indicator of pollution. Taxa richness may also reflect heterogeneity of available habitat.

Shannon Diversity Index: The Shannon Diversity Index provides a single value that incorporates the components of taxa richness and relative abundances of taxa. It is calculated as:

$$H' = \sum p_i \times \ln(p_i)$$

Where pi is the proportion of individuals in taxa i. Therefore, the maximum Shannon Diversity value for a community is determined by the number of species present (H_{max}= ln(richness)), while the realised value is determined by both the richness and relative abundances. Communities with high numbers of taxa, with individuals distributed evenly amongst these taxa, score the most highly in this index. Conversely, communities with lower numbers of taxa, and greater differences in abundance amongst taxa, score the lowest in this index.

Increases in Shannon Diversity may be associated with greater ecosystem stability and can be indicative of increased habitat heterogeneity. See Gallardo *et al.* (2011) for examples of the Shannon Diversity index being used in freshwaters.

Pielou Evenness: The relative abundances of all taxa within a community (Heino *et al.* 2007). This is calculated as:

$$J' = \frac{H'}{H_{max}}$$

Where H' is the Shannon Diversity Index, and H_{max} is the maximum possible score for a given community, described above. Subsequently, J' is bounded between 0–1, where a community with abundances perfectly even amongst taxa would score a 1, and a community with great disparity amongst taxa abundances would score towards zero.

Increases in Pielou Evenness may indicate greater ecosystem stability, and an environmental condition is not strongly favouring some taxa over others. See Heino *et al.* (2007) for examples of Pielou Evenness use in freshwaters.

%EPT: The percentage of all individuals collected made up of pollution-sensitive Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly) taxa. %EPT is typically reduced at polluted sites, and is particularly sensitive to sedimentation.

EPT Taxa Richness: The total number of EPT taxa. EPT richness is typically more negatively affected by pollution than overall taxa richness.

%EPT and EPT Taxa Richness Excluding Hydroptilidae: Both EPT indices were calculated with and without the hydroptilid caddisflies *Oxyethira* and *Paroxyethira*. Unlike most EPT taxa, hydroptilid caddisflies are relatively pollution-tolerant and can be very abundant, skewing EPT indices.

MCI and QMCI: The Macroinvertebrate Community Index and the Quantitative MCI (Stark 1985). Invertebrate taxa are assigned scores from 1 to 10 based on their tolerance to organic pollution. Highest scoring taxa (e.g., many EPT taxa) are the least tolerant to organic pollution. The MCI is based on presence-absence data: scores are summed for each taxon in a sample, divided by the total number of taxa collected, then multiplied by a scaling factor of 20. The QMCI requires either total counts or percentage abundance data: MCI scores are multiplied by abundance for each taxon, summed for each sample, then divided by total invertebrate abundance for each sample. MCI and QMCI scores were calculated using tolerance scores for hard-bottomed streams for the flowing sites. MCI and QMCI scores can be interpreted as per the quality classes of Stark and Maxted (2007) as summarised in Table 2. QMCI scores were also compared against the LWRP freshwater outcome QMCI score of 3.5 for Spring-fed Plains-Urban streams.

Table 2: Interpretation of the MCI and QMCI scores (from Stark and Maxted 2007).

Quality Class	MCI	QMCI
Excellent	>119	>5.99
Good	100-119	5.00-5.90
Fair	80-99	4.00-4.99
Poor	<80	<4.00

Taxa richness, Shannon Diversity, and Pielou Evenness were compared statistically between the pond sites, and amongst sites and years at the flowing sites, using Type I ANOVA. Using the same diagnostic methodology described in the habitat analysis, the assumptions of ANOVA were checked, and data transformed where appropriate. In addition to these metrics, abundance, MCI, and QMCI were statistically compared amongst sites and years at the flowing sites, using two-way Type I ANOVA. We did not calculate MCI and QMCI values for the pond sites as these metrics are intended for streams and rivers and are not suitable for ponds and lakes (Stark and Maxted 2007). EPT data were not compared statistically due to very low numbers being recorded (see results section).

Invertebrate community composition was compared amongst the pond sites, and amongst the flowing sites, between years, using non-metric multidimensional scaling (NMDS), a form of ordination. The ordination was based on a Bray-Curtis dissimilarity matrix, using square-root transformed data and the Ecodist package in R. Spearman rank correlation was used to reveal which taxa were most closely correlated with NMDS axis scores.

2.6.4. Fish

Fish abundance data was not transformed prior to comparison. At the pond sites, the standardised fishing effort allowed for direct comparison of catches between sites. At the stream sites, fishing efforts were variable between years and sites. Measured in m^2, the fishing efforts were (2016/2020): Site 1 (34.4/46.6), Site 2 (33.3/95.8), Site 3 (36.0/59.5), Site 4 (61.6/48.3). While distance fished was standardised at each site between years, changes to wetted widths resulted in different sizes of fished areas between years. At Site 2, a large change in wetted width resulted in a much larger fished area in 2020. Fish abundances were not standardised to the fished areas for comparison due to very low catch numbers at some sites, where small changes in fish catch can lead to misleadingly large changes in fish per unit area.

Fish abundance was compared to the baseline survey to assess changes in fish density. Statistical analyses were not possible due to low fish abundances and a lack of a suitable level of spatiotemporal replication.

3. RESULTS

Summarised ANOVA statistics are provided in Appendix 1.

3.1. Water Quality

3.1.1. Flowing Sites

Water temperature was moderately cool during both during both surveys, ranging from 13.5–14.4°C in the baseline survey, and 10.0–13.4°C in the follow up survey (Figure 3). Temperatures at the two most upstream sites, Sites 1 and 2, were comparable, both between sites and survey rounds. Temperatures at the downstream sites, Sites 3 and 4, were comparable between sites, but had reduced by at least 3.6°C compared to the baseline survey.

Compared to the baseline survey, DO concentration and saturation more than doubled across all sites, except for Site 4, the downstream control. Despite the increase in DO, only Site 3 complied with the LWRP Freshwater Outcome of >70% saturation. Consistent with the previous survey, conductivity increased in a downstream direction. However, conductivity was lower at all sites compared to the baseline survey. The pH values were consistent among sites and survey rounds, all falling within the LWRP Receiving Water Standards range of pH 6.5-8.5.

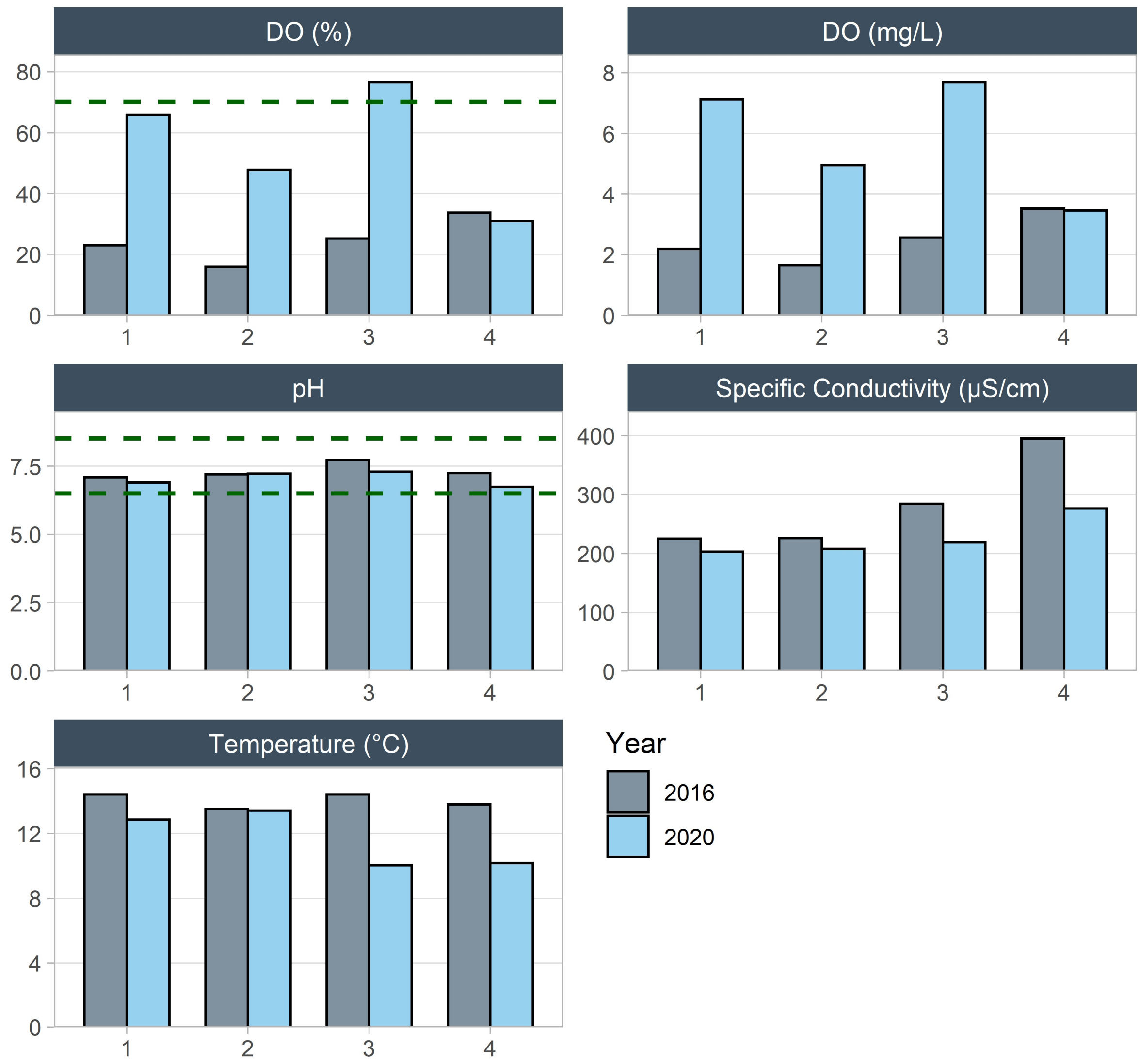

Figure 3: Measured physio-chemical parameters at the flowing sites (Sites 1–4). Green dashed lines indicate LWRP freshwater outcomes and receiving environment standards, which are detailed in the methods.

An oily sheen was observed on the water surface over much of the length of Site 1 (Figure 4). Such a sheen was noted during the baseline survey, over Sites 3 and 4. As discussed by Instream Consulting (2016b), oily sheens can be associated with natural seepage from wetland soils containing low DO, but can also be caused by other factors, such as hydrocarbon spills or landfill leachate. As the sheen has been observed during both survey rounds, and the drain is within area characterised by heavier soils that was covered in extensive wetland prior to land drainage, the source is most likely natural. Iron floc was also abundant through Sites

1 and 2, indicative of iron rich, low DO ground water (Figure 4). Iron floc and natural oily sheens often co-occur.

Figure 4: An oily sheen on the water surface at Site 1 (left) and iron floc at the downstream of Site 2 (right).

3.1.2. Pond Sites

Temperature decreased through the pond system, reducing by 3.2°C from the inlet (Site A) to the outlet (Site D) (Table 3). Conductivity followed the same pattern, with a reduction of 60 µS/cm through the pond systems. Conversely, DO saturation and concentration generally increased through the system. Oxygen saturation increased by from 47.3% at the inlet to 61.0% at the outlet.

Table 3: Spot measurements of water quality at pond sites.

Site	pH	Conductivity (µS/cm)	Dissolved oxygen (mg/L)	Dissolved oxygen (%)	Temperature (°C)	Time
A	6.85	223	5.02	47.3	13.04	12:30
B	6.79	208	4.56	40.6	10.68	11:55
C	6.97	174	6.44	56.6	10.03	11:38
D	7.16	163	6.98	61	9.82	11:20

3.2. Habitat

3.2.1. Flowing Sites

Representative site photographs, before and after restoration, are shown in Figure 5. Note that the 2020 photograph at Site 4 represents the state of the site during the habitat survey, after recent weed clearing and bank trimming. The macroinvertebrate and fish sampling were completed prior to these instream and bank maintenance works.

Two-way type III ANOVA identified many significant habitat differences between sites and years, as well as significant interactions between sites and years (see Appendix 1).

During the restoration process, much of the channel through the golf course was realigned, involving complete reforming of the banks. For Sites 2 and 3, this meant a change in bank form, from near vertical and overhanging concrete walls to lower angle engineered soil walls. These engineered soil walls comprised steel mesh cages filled with a mix of soil and stone, enclosed in geofabric and planted with grass (Figure 5). At Site 1, a partially inundated fresh-plain was created and planted with rushes (*Juncus sp.*) on the southern (true right) bank, in front of new, near-vertical wooden and concrete walls. The banks of Site 4 were not altered by the restoration process, but pugging of the lower banks was evident, likely from recent instream weed removal and bank trimming (Figure 5). Lower bank heights did not change significantly between survey rounds when all sites were examined simultaneously ($p=0.13$), however, bank slopes were significantly less during the follow-up survey ($p<0.001$), with the greatest reduction in slope occurring at Sites 2 and 3 (Figure 6).

The restoration works significantly increased the wetted width of sites within the golf course (Sites 1–3, $p<0.001$, Figure 6). Greater variation in widths was also present post-restoration, with wetted widths within the golf course ranging from 1.1–1.2 m during the baseline survey and 1.6–3.2 m during the follow-up survey (Figure 6). Water depth was substantially reduced at Sites 1 and 2 compared to the baseline survey, while intermediate increases in water depth were observed at Sites 3 and 4. ANOVA identified that water depth only varied significantly when the control site (Site 4) was excluded from the model ($p = 0.006$), and there was a significant interaction effect between site and year both when including ($p < 0.001$) and excluding Site 4 ($p<0.001$), indicating that restoration did not affect the water depth at all sites equally.

Water velocity within the golf course (Sites 1–3) was previously below detectable limits (<0.06 m/s, the Pygmy meter detection limit), or completely lacked flow (Figure 6). During the follow-up survey, velocities were not equal between sites ($p<0.001$), increasing in a downstream direction, and were significantly higher than during the baseline survey ($p<0.001$). Water was clearly backed-up and stagnant in the three golf course sites in 2016 and there was more visible movement in 2020. However, little can be concluded from comparison of water velocities between years at Sites 1-3, because velocities were below detection limits for the velocity meter used in 2016 and they were only detected in 2020 because a more sensitive instrument was used that had a lower detection limit. Regardless of whether there was a statistically significant difference in water velocity between years, water velocity remained very low in 2020, with a mean of less than 0.05 m/s within the golf course, and 0.10 m/s at Site 4.

2016 2020

Site 1

Site 2

Site 3

Site 4

Figure 5: Representative photographs of the flowing sites during the baseline (2016) and follow-up (2020) surveys.

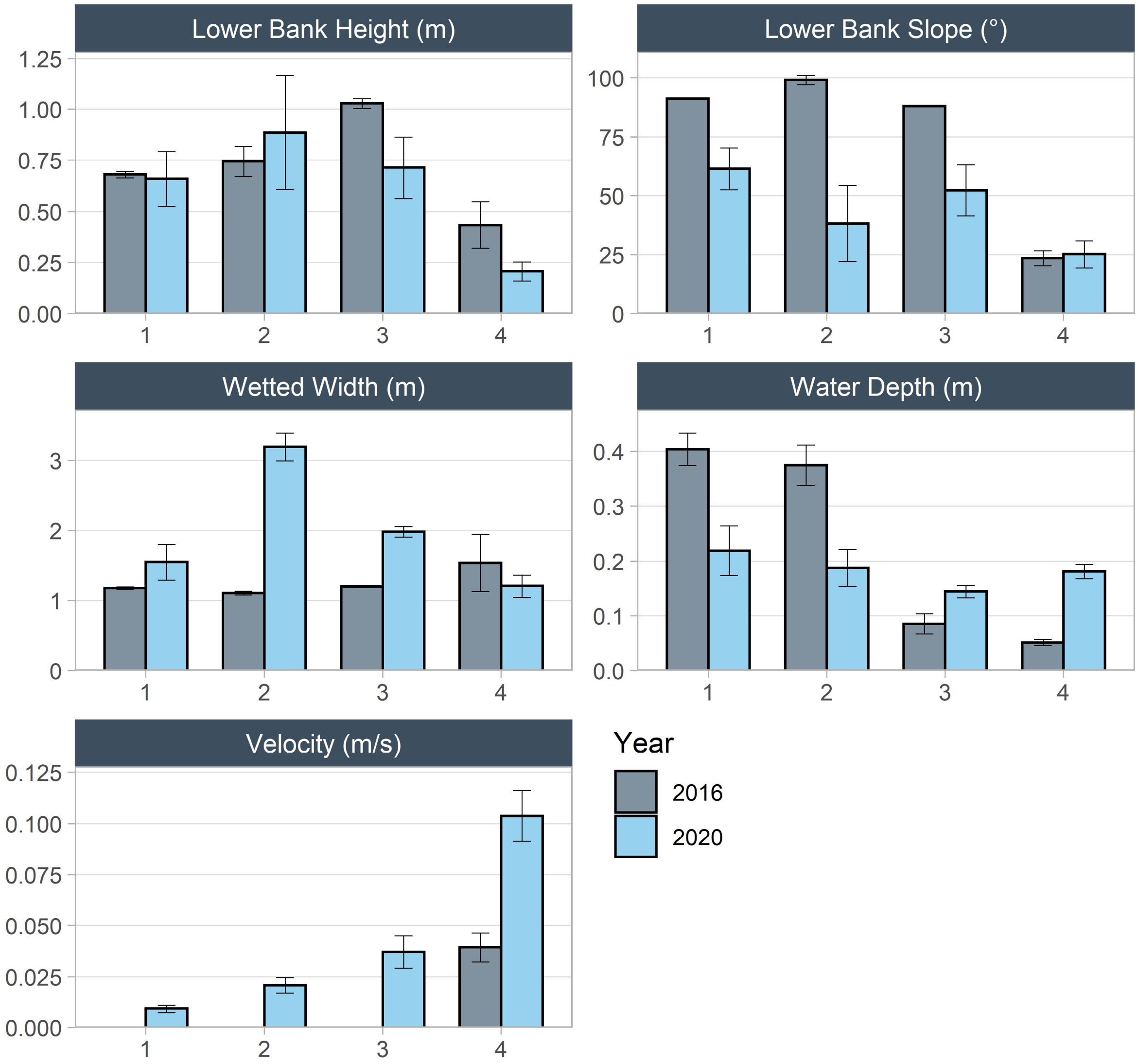

Figure 6: Mean values (± 1S.E.) of selected physical bank and water properties for each sampling site during the baseline (2016) and follow-up (2020) surveys.

Instream habitat features were markedly different both between sites and among years. In 2016, substrates were highly homogenous and dominated by fine sediment (<2 mm) at Sites 1, 3, and 4 (Figure 7). At Site 2, exposed concrete was present during the baseline survey, which was recorded as 4000 mm as per protocol P3 (Harding *et al.* 2009), greatly inflating the particle size score at this site. With the exception of this concrete, Site 2 was dominated by fine sediment during the baseline survey. The restoration activities introduced larger substrate particles to the waterway, especially at Site 3, where fines previously dominated, average particle sizes neared 100 mm (or "cobble" size, using the Wentworth scale, Figure 7). Changes in substrate size were determined to be statistically significant among sites ($p<0.001$) and years ($p=0.01$), regardless of the inclusion of Site 4 in the model.

Irrespective of changes in particle size, fine sediment cover remained high, with no significant effect of year on this parameter, even when the control site was excluded from the model ($p=0.20$). Fine sediment cover at all sites remained well above the LWRP Freshwater

Outcome of 30%. However, fine sediment depth was greatly reduced across all sites in 2020 (Figure 7), with survey year having a significant effect on this parameter, both when Site 4 was included ($p=0.001$) and excluded ($p<0.001$).

Overhanging vegetation was substantially more abundant during the follow-up survey at Sites 2 and 3, with a minor increase observed at Site 1 (Figure 7). ANOVA determined that survey year had a significant relationship with this parameter ($p<0.001$), as well as identifying a significant site x year interaction ($p<0.001$). The only site that was not observed to have higher levels of overhanging vegetation was Site 4, where recently stream maintenance had substantially reduced the length of the riparian vegetation (Figure 5). Changes in riparian vegetation also influenced the level of shading. The effect of the restoration on shading was not consistent between sites, indicated by the significant interaction term in both ANOVA models ($p<0.001$). Shade declined substantially at Sites 1 and 3, increased at Site 2 and remained relatively unchanged at Site 4.

Total macrophyte cover changed significantly between years ($p<0.001$), and there was a significant site x year interaction effect ($p<0.001$). Sites 1 and 3 had substantially greater macrophyte cover during the follow-up survey, while a slight increase occurred at Site 2. Only the downstream control saw a reduction in macrophyte cover, which can be attributed to recent macrophyte clearing activities prior to the habitat survey (Site 4, Figure 7). None of the sites within the golf course (Sites 1–3), nor the downstream control, were above the LWRP outcome value of 60% total macrophyte cover. The macrophyte community was dominated by *Nitella* and *Potamogeton crispus* at Site 1, *Elodea canadensis* and monkey musk (*Mimulus moschatus*) at Site 3, and *Glyceria fluitans* at Site 4.

3.2.2. Pond Sites

Representative photographs of each of the pond sites are provided in Figure 8.

Site A was located at the inlet to the pond system. Water was conveyed under the fairway to the pond system *via* two approximately 55 m long pipe culverts (Figure 9). Water through the pipes was of adequate depth and sufficiently low velocity to allow for fish passage. Substrate was dominated by pebbles (16–64 mm), with some silt/sand (<2 mm) and small cobble (64–128 mm) also present. Macrophytes were abundant, consisting of *Elodea canadensis* (80%) and *P. crispus* (20%). The substrate and macrophytes were smothered in long green filamentous algae and iron floc (Figure 10).

Site B was located alongside one of the floating wetlands. The substrate was similar to Site A and was also covered in fines. The floating wetlands provided the only substantial shade for the pond system. Roots from the *Carex* planted on top of the floating platform had begun to grow down into the water column but were reasonably sparse.

The substrate at Site C (bank site) was also dominated by pebbles, with small amounts of small cobble also present. Fine sediment covered approximately 85% of the substrate. *E. canadensis* was abundant with some *P. crispus* also present. Shading was near zero at this site.

Some larger substrates (large cobbles and boulders) were present at the bank edges at Site D (pond outlet). Consistent with the other pond sites, fine sediments nearly completely covered the substrate at Site D. Macrophyte cover was high, dominated by *E. canadensis*, with some *P. crispus*, *Nitella*, and *Lemna minor.* Water was retained in the pond system by a small restriction outlet pipe, which appeared to be clear of any blockages and was wide enough to provide fish passage.

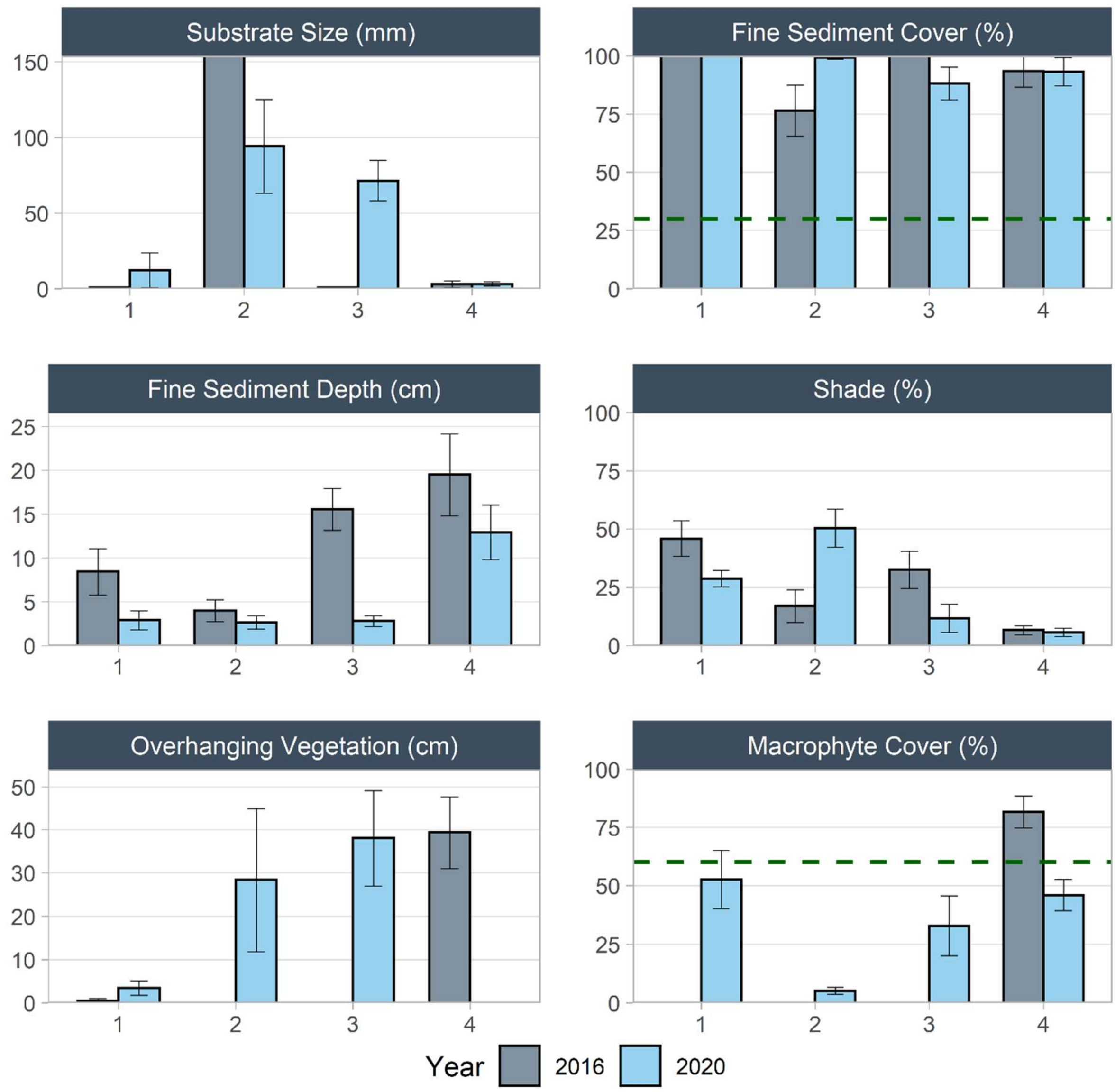

Figure 7: Mean values (± 1S.E.) of selected instream habitat and cover features during the baseline (2016) and follow-up (2020) surveys. Also included are the relevant standards for comparison (green dashed line), which are detailed in the methods.

Figure 8: Representative photographs of each of the pond sampling sites: A) Inlet, B) Floating wetland, C) Bank, D) Outlet.

Figure 9: Inlet culverts to the pond system.

Figure 10: Long filamentous green algae and iron floc smothering macrophytes and substrate at site A.

3.3. Macroinvertebrates

3.3.1. Flowing Sites

A total of 44 distinct taxa were recorded across the four flowing sites during the follow-up survey, a 52% increase from the 29 taxa recorded in the baseline survey. Pollution-tolerant taxa remained dominant, with none of the 10 most abundant taxa in 2020 having an MCI score over 5 (Figure 11). EPT taxa were better represented than during the baseline survey, where only a single EPT taxon was recorded. Five EPT species were recorded in 2020, two of which were the pollution tolerant *Oxyethira albiceps* and *Paroxyathira*. Excluding these two taxa, EPT taxa abundance remained low, with relative abundance of EPT being below 1.5% at all sites.

Consistent with the baseline survey, ostracod crustaceans were the most abundant taxa, totalling 24% of the macroinvertebrates collected (Figure 11). Oligochaete worms also remained highly abundant, representing 14% of macroinvertebrates collected. *Chironomus zealandicus,* which was the second most abundant taxa during the baseline survey, was greatly reduced in 2020 and did not fall into the top ten most abundant taxa. Many of the most abundant taxa during the follow-up survey were relatively small constituents of the community during the baseline survey. Notably, *O. albiceps*, which was not identified during the baseline survey, was the fourth most abundant taxon during the current survey, with a relative abundance of 10%. Other abundant taxa that had substantial increases in relative abundance in 2020 included *Gyraulus snails*, and Cladocera and copepod crustaceans.

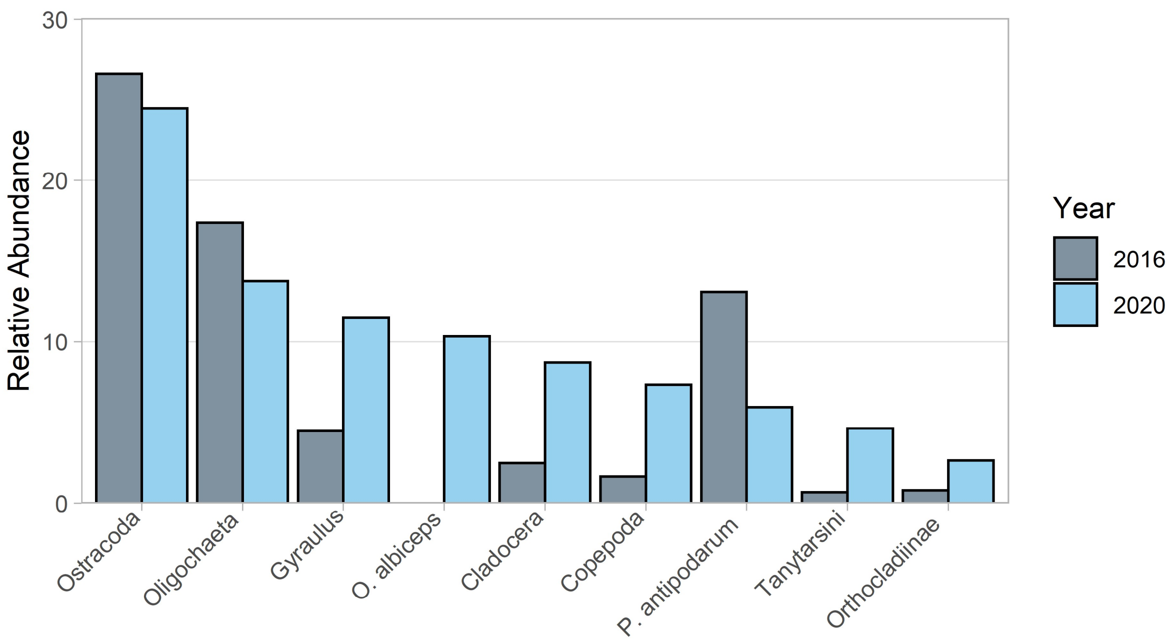

Figure 11: The relative abundances of the ten most abundant taxa in the follow-up (2020) survey, compared to their relative abundances in the baseline (2016) survey.

Macroinvertebrate abundance was greatest overall at Site 3 and abundance more than double at Site 3 between sampling years (Figure 12). This was associated with a significant difference in macroinvertebrate abundance among sites ($p<0.001$) and a weak, but not statistically significant year x site interaction ($p=0.083$ including Site 4). Abundance was lower and changed comparatively little between years at the other flowing sites.

Taxa richness, Shannon diversity, MCI, and QMCI all increased at Sites 1–3 in 2020, but decreased or showed little change at the downstream control (Site 4, Figure 12). These trends were reflected in significant site x year interactions ($p<0.05$) when Site 4 was included in the analysis. Despite increased MCI and QMCI scores at Sites 1–3, all sites remained in the 'Poor' quality class of Stark and Maxed (2007) for MCI and QMCI, and below the LWRP Freshwater Outcome of 3.5 for QMCI.

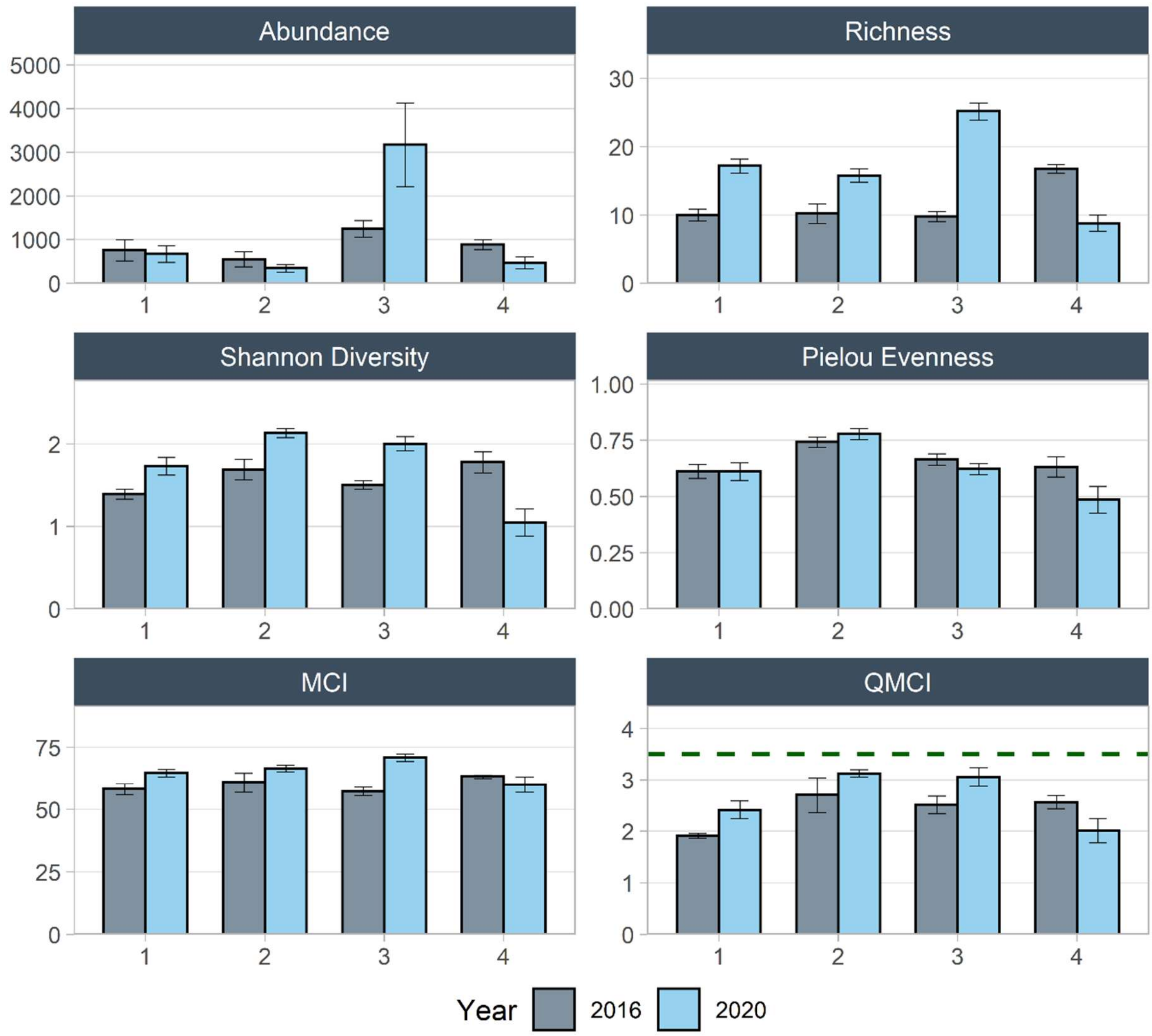

Figure 12: Mean values (± 1S.E.) for calculated macroinvertebrate metrics during the baseline (2016) and follow-up (2020) surveys. Also included are the relevant standards for comparison (green dashed line), which are detailed in the methods.

NMDS ordination of the macroinvertebrate communities revealed both spatially and temporally distinct community organisations (Figure 13). The sampled communities were relatively less distinct (more homogenous) during the baseline survey, indicated by the tight grouping of 2016 samples in Figure 13. During the baseline survey, communities at Sites 2–4 were relatively indistinguishable in ordination space. The community at Site 1 was slightly separated in 2016, driven by a greater relative abundance of Sphaeriidae molluscs, and lower abundance of *Potamopyrgus antipodarum* snails, when compared with the other 2016 sites.

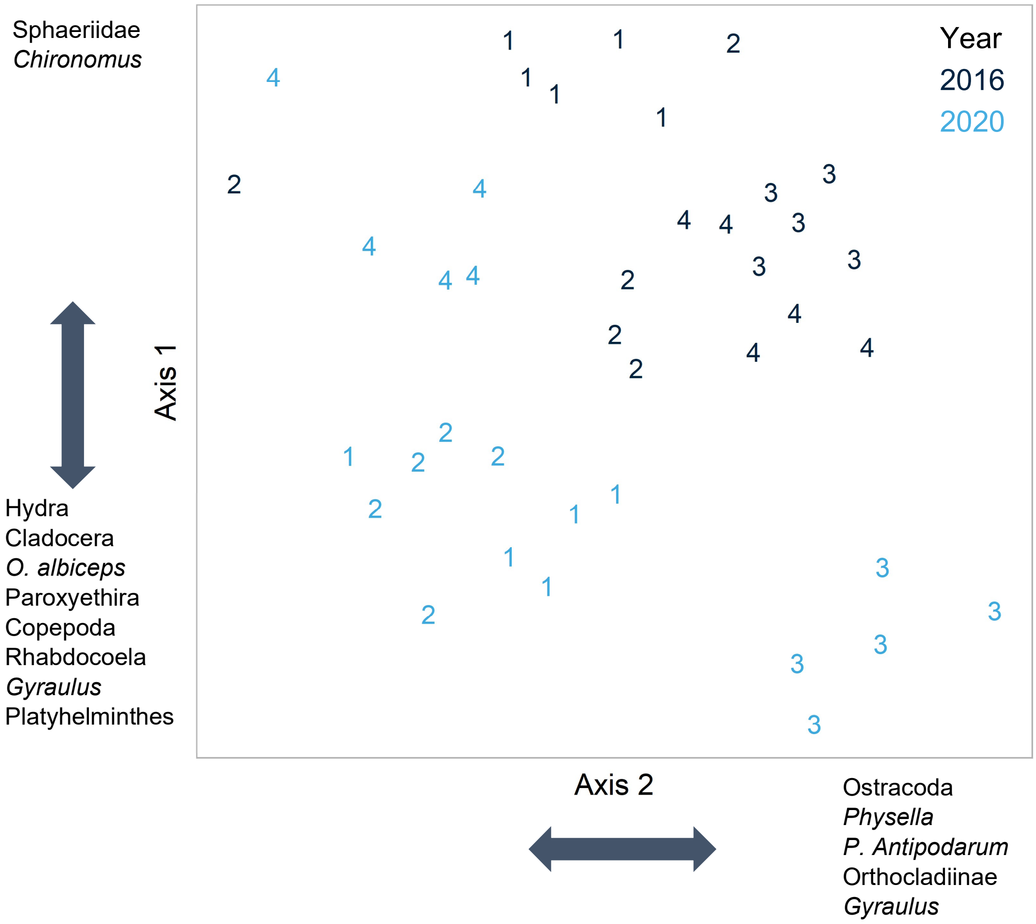

Figure 13: NMDS plot of all invertebrate samples from the flowing sites. Sample year indicated by colour and site indicated by number. Species that were strongly correlated with each axis (p<0.01) and identified as substantial drivers of site differences among years are included on each axis. Plot stress is 0.19.

Macroinvertebrate communities during the follow-up survey were much more heterogenous, forming more distinct groups in ordinal space (Figure 13). Generally, the resampled communities had negative shifts along Axis 1. The largest community shift occurred at Site 3, which ended up being the most distinct community in 2020. This shift was driven by substantial changes in the abundances of numerous taxa, as well as the disappearance of previously abundant taxa. *Chironomus*, a genus that was originally prevalent at Site 3, with 1819 individuals in 2016, was not recorded at the site during the follow-up survey. Conversely, *O. albiceps,* Cladocera, Tanytarsini chironomids, and *Paroxyethira* were not recorded at Site 3 during the baseline survey, but were numerous during the follow-up survey, with abundances of 2375, 1926, 845, and 750, respectively.

Sites 2 and 3 both had negative shifts along Axis 1 between 2016 and 2020, resulting in very similar communities during the 2020 survey. For these sites, major community shifts included a relatively large reductions in ostracods, Sphaeriidae, and *Chironomus*. Both sites were characterised in 2020 by high abundances of oligochaetes and copepods.

Site 4, the downstream control, was the only site to change little in ordination space over time (Figure 13). The community similarity between 2016 and 2020 samples at Site 4 was largely driven by high abundances of oligochaetes and ostracods, which made up 53% and 34% of the total collected animals at this site, respectively. The most substantial difference between macroinvertebrate communities in 2016 and 2020 at Site 4 was in the abundance of *Chironomus.* While *Chironomus* accounted for, on average, 19% of the individuals in the 2016 samples, only 0.6% of individuals collected at Site 4 in 2020 were *Chironomus*.

3.3.2. Pond Sites

A total of 35 distinct invertebrate taxa were recorded across the four pond sites. One-way ANOVA confrormed there was no signicant difference among the pond sites with regards to taxa richness (p=0.11), Shannon diversity (p=0.58), or Pielou Eveness (p=0.42).

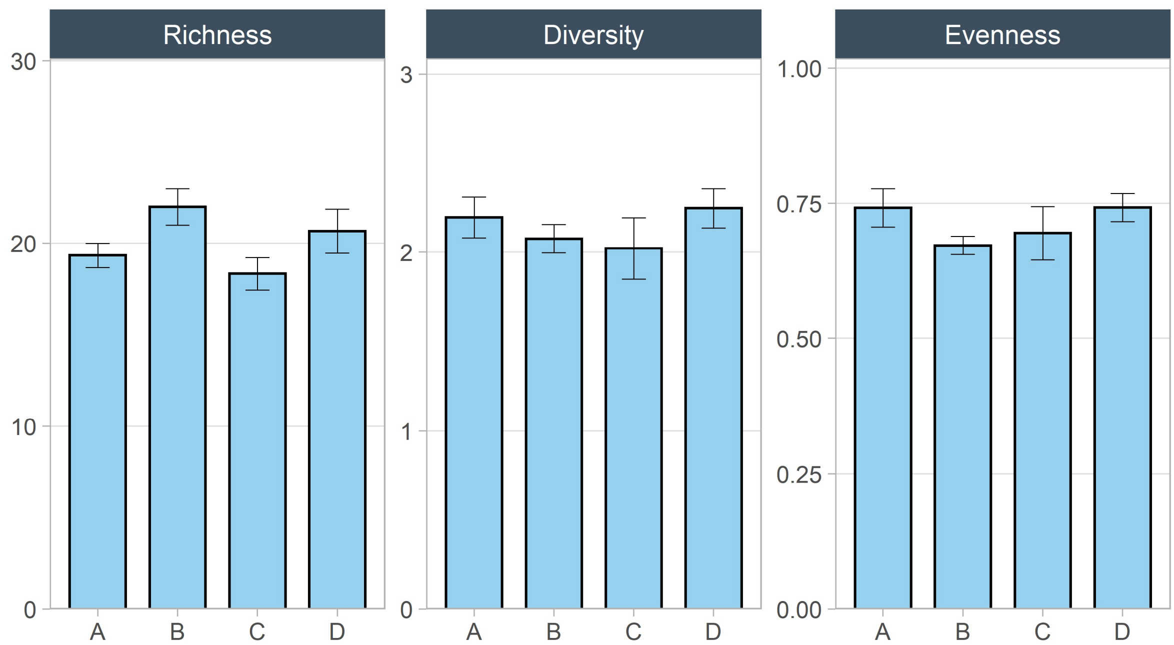

Figure 14: Site level mean values (± 1S.E.) of select macroinvertebrate metrics at the pond sites during the 2020 survey.

Ordination of the pond macroinvertebrate communities revealed a lack of site-specific community structure, with intra-site ordination distances appearing to be only slightly shorter than inter-site distances. Consequently, there were few significant taxa associations with the ordination Axis. Of these associations, they have limited ecological significance. Ceratopogonidae, while being strongly correlated with Axis 2, represented less than 3% of individuals across all sites, and thus, is not substantially contributing to site orientations in ordination space. Similarly, copepod relative abundances only varied from 9–19% between sites, contributing little to inter-site community differences. Tanytarsini chironomids, however, appear to significantly contribute to site separation, with Sites C and D having higher relative abundance of this taxa (35% and 16% respectively), than Sites A and B (10% and 5%).

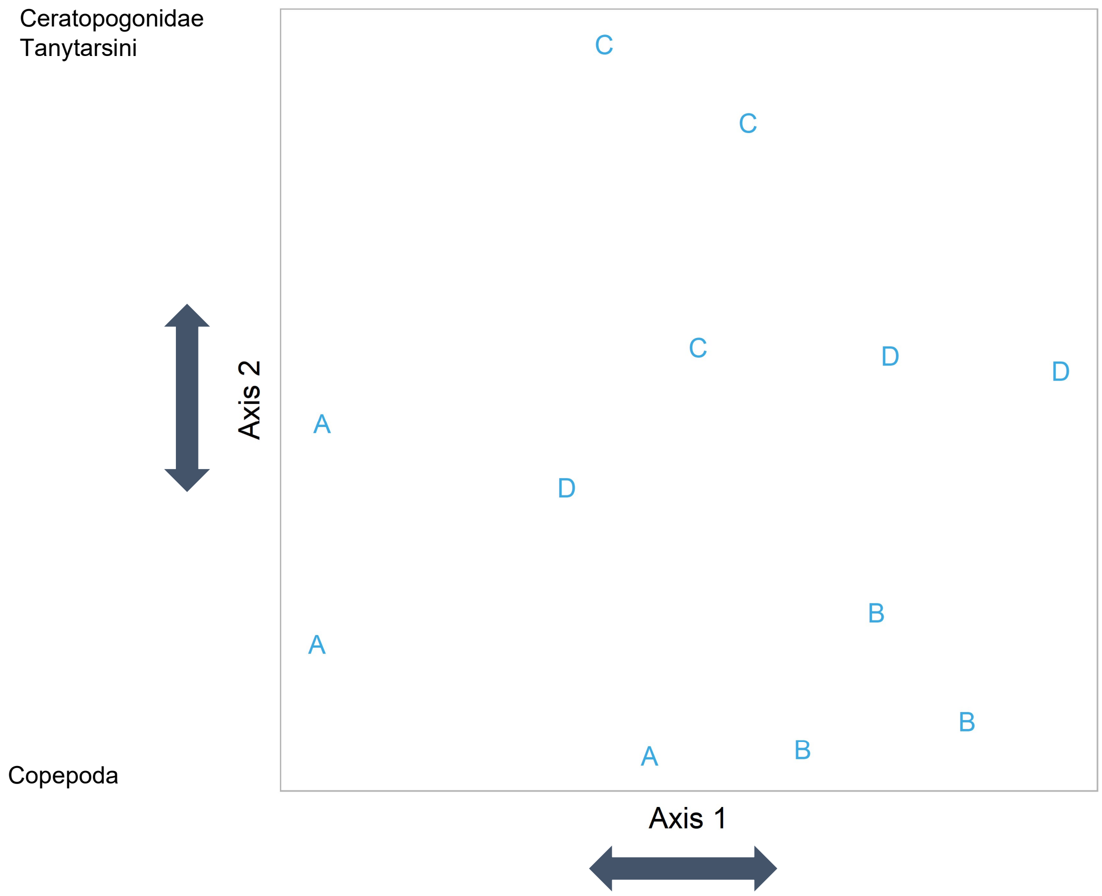

Figure 15: NMDS plot of all invertebrate samples from the pond sites. Species that were strongly correlated with an axis (p<0.01) are labelled. Plot stress is 0.17.

3.4. Fish

3.4.1. Flowing Sites

Fish diversity was low in 2016 and remained low in 2020 at flowing sites, with only two species recorded: shortfin eel (*Anguilla australis*) and upland bully (*Gobiomorphus breviceps*). Both shortfin eel and upland bully are native species that are not threatened (Dunn *et al.* 2018). All shortfin eels in the follow-up survey were within the previously recorded size range of 142–489 mm, with the exception of three individuals, which measured 134, 492, and 580 mm. Median length of shortfin eels was similar between sampling dates, with a median of 255 mm in 2016 and 288 mm in 2020.

The only species not captured in 2020 that was recorded in the baseline survey was inanga (*Galaxias maculatus*). Inanga have an At Risk – Declining conservation status (Dunn *et al.* 2018). However, inanga were recorded in the pond system in 2020 (see below).

Fish abundances at flowing sites within the golf course were low, with slight decreases at Sites 2 and 3, although abundances were well up at Site 1. Consistent with the baseline survey, fish abundance was greatest at Site 4, by some margin.

Figure 16: Fish abundances at flowing sites during the baseline (2016) and follow-up (2020) surveys.

3.4.2. Pond Sites

Fish diversity was higher in the ponds than in the flowing sections of the golf course, with five species recorded: inanga, shortfin eel, upland bully, longfin eel (*Anguilla dieffenbachii*), and common bully (*G. cotidianus*). Longfin eel have an At Risk – Declining conservation status, while common bully are not threatened (Dunn *et al.* 2018). Shortfin eel and upland bully were the most abundant species recorded, followed by inanga (Figure 17). Relative abundance of each fish species did not vary substantially among sampling sites.

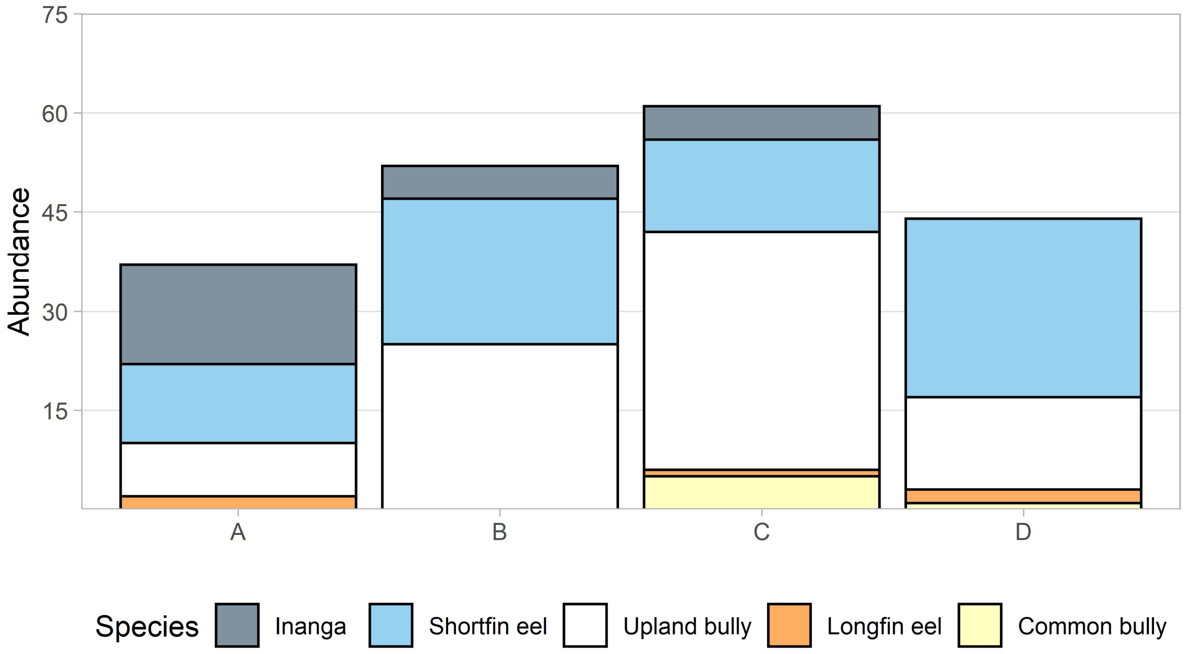

Figure 17: The abundances of fish species caught at each pond site during the follow-up (2020) survey. Data are the total of five fyke nets per site.

Shortfin eels were larger in the ponds than in the flowing sections, ranging in length from 63 mm to 671 mm, with the median length being 409 mm (Figure 17). The sizes of shortfin eels in the ponds were normally distributed, with few eels under 200 mm or over 600 mm.

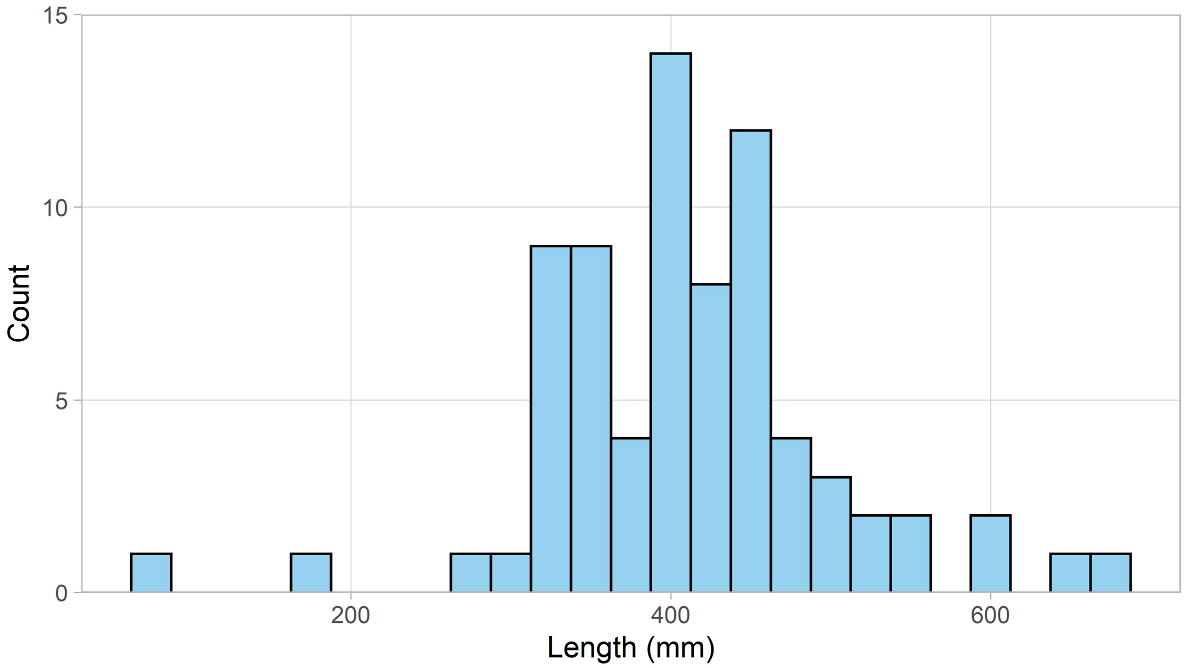

Figure 18: Size frequency of shortfin eels caught from pond sites in 2020.

4. DISCUSSION

Restoration of No. 1 Drain has substantially improved aquatic habitat, which is associated with a more diverse invertebrate community and improved MCI and QMCI scores. Impacts on fish communities have been less marked at flowing sites, but the addition of new pond habitat has greatly increased fish abundance and diversity.

Low levels of DO was identified as a factor likely limiting biota during the baseline survey (Instream Consulting 2016b). In 2016, DO levels as low as 1.66 mg/L were recorded. It is likely that even lower levels were occurring during the warmer months, due to the combined impacts of greater respiration from plants and algae, and lower oxygen saturation in warmer water. For context, Dean and Richardson (1999) found dissolved oxygen levels of 1 mg/L resulted in a mortality rate of 22% within 12 hours for adult inanga and 47% for adult common bullies, over the same duration. Juvenile common bullies were identified as being even more sensitive to low oxygen conditions with 100% mortality after 4 hours at 1 mg/L. Sublethal effects are likely to occur at the levels previously recorded in No. 1 Drain, which may include increased ventilation rates, reduced feeding, and altered predator avoidance responses (Franklin 2014). This hypoxic condition of No. 1 Drain was likely the result of substrate and water velocity properties. The sediment in No. 1 Drain was observed to be particularly dark in colour, indicative of high organic matter content. Respiration of the microbes involved in the breakdown of this organic matter consumes oxygen in the water and, without adequate flow through (as was the case in No. 1 Drain), the water became hypoxic. Restoration of No. 1 Drain substantially raised the dissolved oxygen levels within this section of the waterway. The lowest recorded value within the restored section of No. 1 Drain (Sites 1–3) during the follow-up survey was 4.95 mg/L. Increased oxygen levels were likely due to greater flow through the system, resulting from the reformed channels, providing fresh oxygenated water and preventing hypoxia due to stagnation.

Despite the improvement in dissolved oxygen levels, only Site 3 was above the LWRP outcome value of 90% dissolved oxygen saturation. To achieve this level at all sites, water velocities must be increased to prevent stagnation, or features that improve surface aeration (such as riffles) must be included. Creation of riffle sections is often a priority for restoration projects, because they improve habitat heterogeneity, but riffles are impractical in the setting of No. 1 Drain. As described in the baseline survey, the golf course section of No. 1 Drain lacks the flow and gradient to create swift flows or riffle environments (Instream Consulting 2016b). However, adjustment to the water level in the pond system may provide some benefit. The in-line pond system serves as a hydraulic control, where the water level in the ponds determines the water level and velocity of the flowing sections upstream. Reduction of the water level in the pond could, therefore, increase the velocities upstream by preventing backfilling into the channelled sections.

Regardless of the current water level, the new ponds appear to provide some additional water quality benefits. Water exiting the pond system was about 14% more saturated with oxygen, likely due to photosynthesis from macrophytes that were abundant within the ponds. Furthermore, both conductivity and temperature dropped consistently in a downstream direction through the pond system. This is indicative of cool ground water infiltrating into the ponds, diluting the surface water. The cooling effect of the ground water, in combination with the photosynthesis of the macrophytes, raised the absolute amount of oxygen in the water by 39%, with water exiting the pond system holding 6.98 mg/L of oxygen. While the ponds appear to be providing water quality benefits to No. 1 Drain, it is important to remember that spot measurements represent a snapshot of water quality. During the warmer, drier months and at night, the benefits of these ponds may be reduced, and may even be reversed. Respiration of the numerous macrophytes in the ponds likely depletes oxygen levels overnight. During the warmer months, the slow-moving water in the ponds, in combination with limited shade, may result in elevated water temperatures, which would further lower oxygen levels. The moderating effect of the intercepting ground water may also be lessened during the drier months, when the local water table drops, reducing the level of groundwater intrusion into the ponds. It is not possible to determine how significant these mechanisms may be without summer monitoring of temperature and dissolved oxygen. However, there is the possibility that the limited flow and potentially warm temperatures within the ponds could raise the risk of toxic algal blooms. The proximity of the pond system to Horseshoe Lake, a culturally significant area, increases the importance of avoiding and monitoring blooms. Toxic algal blooms in the ponds would provide a threat to dogs allowed in the water at Horseshoe Lake, as well as to people through secondary contact while participating in paddle sports. Reduction of the water level in the ponds over summer would shorten water residence time, allowing less time for water to heat up, which may reduce the risk of toxic algal blooms. While increased water velocity is associated with substantial improvements in water quality, there has been little change in the range of hydraulic habitats available. The flow character has remained homogenous, with only slow run habitat present within the golf course, reflecting the low channel grade across the site. Velocities were inadequate to prevent build-up of fines, with fine sediment cover nearing 100% at all sites. As described above, it is unlikely that velocities capable of clearing fines and preventing further settlement can be achieved in this section of No. 1 Drain, given the low flow volume and gentle gradient. However, imported cobbles have provided a large increase in substrate complexity, relative to the homogenous fine sediment or bare concrete that formed the bed of the waterway prior to restoration.

Reduced bank slopes and riparian *Juncus* plantings close to, and in some cases within, the waterway provide increased fish cover, reflected by the increased overhanging vegetation. The only site that did not increase in overhanging vegetation was the downstream control (Site 4), a direct result of extensive bank trimming prior immediately prior to the habitat survey. Despite greater overhanging vegetation at the golf course sites, shade was only increased at Site 2. At Sites 1 and 3, reductions in shade relates to removal of large overhead trees and reforming of steep banks reducing bank shade, and may also be affected by survey methodology. Following conventional densiometer methodology, the shade measurement was recorded at hip height. Much of the planted *Juncus* was below hip height at Sites 1 and 3, and was subsequently not included in the shade measurement. Juncus were taller at Site 2 and therefore resulted in greater measured shade. Shading was sufficient to maintain relatively low to moderate macrophyte cover at all of the sites. Such levels of macrophyte cover would provide additional habitat complexity and cover for fish and invertebrates compared to pre-restoration, when macrophytes were absent. Macrophytes and algae were particularly abundant within the ponds, which was expected, given the lack of shading. This is unlikely to change with time, given the low height of the species included in riparian plantings, even when fully developed. Taller trees would need to be planted to provide adequate shade, but this would be impractical in the middle of a golf course.

Following restoration, macroinvertebrate communities were more diverse, and while still being dominated by pollution tolerant taxa, there was an increase in the relative number of more sensitive taxa (increased MCI) and an increase in the relative abundance of sensitive taxa (increased QMCI). These changes likely reflect a combination of improved water quality and habitat. With improved water quality, especially the increase in DO, the waterway has the potential to host more sensitive species. However, ordination of the communities at the flowing sites revealed distinct community structures among many of the sites. This indicates that the macroinvertebrate communities have likely diverged in response to increased habitat heterogeneity amongst and within sites.

The fish community within No. 1 Drain also appears to have responded positively to the restoration efforts. Fish taxa richness was up, which is associated with higher levels of dissolved oxygen and greater habitat heterogeneity. Fish numbers remained low at the flowing sites, relative to the downstream control. However, the large majority of the catch at all sites was shortfin eel, for which the flowing sites provide low quality habitat. Conversely, the ponds provide a large amount of high-quality shortfin eel habitat, which is where the real benefit of the restoration was seen in terms of fish abundance. A total of 194 fish were captured between the pond sites. This represents a large increase in total fish abundance and biomass in No. 1 Drain.

Fish densities and diversity were likely underestimated by fyke netting the ponds in May 2020. That is because the survey was conducted in May, so that it was comparable to the baseline survey, but this is outside the recommended sampling period of 1 December to 30 April (Joy *et al.* 2013). That is because fish may be less active and less susceptible to capture during cooler months. In addition, downstream migration of adult inanga for spawning may have resulted in fewer inanga being caught than may be expected during the summer months.

Regardless of the impacts of timing of the fish survey, the addition of pond habitat, with high macrophyte cover, floating wetland cover, and relatively higher water quality, has added a sizeable (approximately 4,700 m²) amount of open water fish habitat, that is being well utilised. The addition of deep ponds also seems to have benefited the larger size classes of eels, with the median sized shortfin eel being greater in the ponds, when compared the flowing sites, as well as being greater than the baseline survey. This is consistent with the literature on eel habitat preferences, which has shown a preference for high cover (bank, macrophyte, and instream debris), and slow, deep (>0.3 m) habitats, by eels over 500 mm (Jellyman *et al.* 2003). These larger eels (>500 mm), were not present within the golf course during the baseline survey, but 15 eels >500 m were recorded during the follow-up survey, 14 of which were in the ponds.

Within the ponds there was no discernible habitat preference by the captured species, with similar abundances recorded at each of the sampling sites. We speculated that the floating wetlands may provide high quality fish cover among the hanging roots of the planted *Carex*. While the roots are currently sparse and unlikely to provide substantial fish cover, the plantings are young, and the value of the roots as cover will presumably increase as the plants develop. The floating wetlands will, however, provide substantial benefits during the summer, shading large portions of the ponds that would otherwise be exposed.

The richness and abundance of fish species in within No. 1 Drain will be partly determined by any downstream restrictions on fish passage. The downstream culvert, which runs eastward under Horseshoe Lake Road, was of adequate size, with a small amount of hydraulic head, and is not a threat to fish passage, evident from the increased abundance of migratory species (both eel species, inanga, and common bully). The five tide gates at the outlet to Horseshoe Lake have recently been repaired (completed January 2019) and now include a single fish-friendly gate. While this clearly allows some upstream migration of young inanga, the effectiveness of this instalment is unclear. The effect of the pump station on downstream eel migration is also not known, however, being of Archimedes screw design, mortality of eels that travel through the pump is likely much lower than through pumps with a propeller-type design (Buysse *et al.* 2013).

Overall, the restoration of No. 1 Drain can be considered successful. Most facets of the waterway have displayed improved values, including water quality, instream habitat, and the presence of a more diverse and abundant invertebrate and fish community. While this community may still lack sensitive species relative to a natural system, the success of the restoration must be framed in the context of an artificial urban waterway, that is still actively receiving urban stormwater runoff. While some attributes of the waterway are unlikely to improve through time, including water velocity, dissolved oxygen, and fine sediment cover, the full benefits of the restoration efforts may not yet be observable. Riparian vegetation is still developing in the lower sections of the waterway, and given appropriate management, will provide future increases in shade and cover. Instream habitat within the ponds may also still be developing, through the natural accumulation of leaves and other organic matter, which may provide further fish cover. Finally, assessment and remediation of downstream fish barriers would ensure that the restored habitat in No. 1 Drain is accessible, which could result in improvements in fish abundance and diversity in the future.

5. RECOMMENDATIONS

- Maintenance of riparian plantings, especially in the sections downstream of the ponds.
- Summer monitoring of:
  - Water quality within the ponds using a dissolved oxygen and temperature logger. This would help determine whether hypoxic conditions are occurring, that may be limiting to the aquatic fauna.
  - Presence of toxic algae. The risk of toxic algal blooms occurring in ponds and lakes increases with greater nutrient levels and lower oxygen levels, and the risk is typically greatest during the warmer summer months.
  - Fish within the ponds. As both the baseline and the current survey were carried out in May, many inanga may have already migrated downstream to spawn, resulting in artificially low numbers. The ponds have the potential to provide good habitat for this species and it would be useful to know if this habitat is being well utilised.
- Investigate the feasibility of reducing pond water levels to shorten pond water residence time and increase water velocities upstream.
- Monitoring of typical stormwater contaminants (e.g., zinc, lead, and copper) to assess the effectiveness of the ponds at improving stormwater quality.
- Assessment of effectiveness of the fish friendly tide gate in Horseshoe Lake. It is currently unknown whether the single fish-friendly tide gate is sufficient to provide unimpeded fish passage. Hence, it is unknown whether this structure is limiting the use of the new habitat in No. 1 Drain by fish species such as inanga.

6. ACKNOWLEDGEMENTS

This project was entirely funded by Christchurch City Council. Thank you to Belinda Margetts and Kamal Narang from CCC for instigating the project and for organising site access. Special thanks also to Christchurch Golf Club, for allowing access to their course and for the loan of their Gator to carry field equipment between sites.

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APPENDIX 1: STATISTICS SUMMARY

Table 1: Summary of statistics from Type III two-way ANOVA run on collected habitat variables at flowing sites. ANOVA run including and excluding the control site (Site 4).

Parameter	Data Transformation	P-value (Including Site 4)	P-value (Excluding Site 4)
Algae (cm)	Ranked	Site: <0.001 Year: 0.09 Site*Year: 0.09	Site: <0.001 Year: 0.11 Site*Year: 0.068
Fine Sediment Cover (%)	Ranked	Site: 0.051 Year: 0.21 Site*Year: 0.014	Site: 0.012 Year: 0.204 Site*Year: 0.003
Fine Sediment Depth (cm)	Square root	Site: <0.001 Year: 0.0012 Site*Year: 0.16	Site: 0.09 Year: <0.001 Site*Year: 0.13
Leaf Pack (cm)	Incl. Site 4: Ranked Excl. Site 4: Log+1	Site: <0.001 Year: <0.001 Site*Year: 0.007	Site: 0.11 Year: 0.003 Site*Year: 0.12
Lower Bank Height (m)	Incl. Site 4: Ranked Excl. Site 4: None	Site: <0.001 Year: 0.13 Site*Year: 0.13	Site: 0.18 Year: 0.51 Site*Year: 0.16
Lower Bank Slope (°)	None	Site: <0.001 Year: <0.001 Site*Year: <0.001	Site: 0.017 Year: <0.001 Site*Year: 0.007
Macrophyte Cover (%)	Ranked	Site: <0.001 Year: <0.001 Site*Year: <0.001	Site: <0.001 Year: <0.001 Site*Year: <0.001
Water Depth (m)	Incl. Site 4: Square root Excl. Site 4: Ranked	Site: <0.001 Year: 0.45 Site*Year: <0.001	Site: <0.001 Year: 0.006 Site*Year: <0.001
Organic Cover (%)	Incl. Site 4: None Excl. Site 4: Square root	Site:<0.001 Year: 0.008 Site*Year: 0.16	Site: <0.001 Year: 0.04 Site*Year: 0.23
Overhanging Vegetation (cm)	Ranked	Site: 0.010 Year: <0.001 Site*Year: <0.001	Site: 0.035 Year: <0.001 Site*Year: 0.002
Shade (%)	Ranked	Site: <0.001 Year: 0.87	Site: <0.001 Year: 0.85

Parameter	Data Transformation	P-value (Including Site 4)	P-value (Excluding Site 4)
		Site*Year: <0.001	Site*Year: <0.001
Velocity (m/s)	Ranked	Site: <0.001 Year: <0.001 Site*Year: <0.001	Site: <0.001 Year: <0.001 Site*Year: 0.001
Wetted Width (m)	Ranked	Site: 0.15 Year: <0.001 Site*Year: 0.002	Site: 0.001 Year: <0.001 Site*Year: 0.001
Substrate Size (mm)	Ranked	Site: <0.001 Year: 0.003 Site*Year: <0.001	Site: <0.001 Year: 0.014 Site*Year: <0.001

Table 2: Summary of statistics from Type I two-way ANOVA run on calculated macroinvertebrate metrics. ANOVA run including and excluding the control site (Site 4).

Parameter	Data Transformation	P-value (Including Site 4)	P-value (Excluding Site 4)
Abundance	Log transformation	Site: <0.001 Year: 0.073 Site*Year: 0.082	Site: <0.001 Year 0.502 Site*Year: 0.222
Richness	Square-root transformation	Site: 0.004 Year: <0.001 Site*Year: 0.002	Site: 0.003 Year: <0.001 Site*Year: <0.001
Shannon Diversity	None	Site: >0.001 Year: 0.069 Site*Year: <0.001	Site: 0.0018 Year: <0.001 Site*Year: 0.654
Pielou Evenness	None	Site: >0.001 Year: 0.14 Site*Year: 0.091	Site: >0.001 Year: 0.92 Site*Year: 0.41
MCI	Squared transformation	Site: 0.41 Year: <0.001 Site*Year: 0.003	Site: 0.36 Year <0.001 Site*Year: 0.09
QMCI	None	Site: <0.001 Year: 0.095 Site*Year: 0.018	Site: 0.0012 Year 0.004 Site*Year: 0.95