# Number One Drain Aquatic Ecology Survey

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# EXECUTIVE SUMMARY

No. 1 Drain is a concrete-lined artificial waterway that flows through the Christchurch Golf Club golf course in the suburb of Shirley. Christchurch City Council (CCC) is planning repairs and environmental enhancement work in No. 1 Drain, following damage to the banks and bed of the drain by earthquakes in 2010 and 2011. The purpose of this report is to describe the ecology of No. 1 Drain and provide restoration recommendations.

Within the golf course, No. 1 Drain has a concrete-lined channel with vertical banks approximately 1 m high and no perceptible flow. The drain is piped upstream of the golf course, but has natural banks downstream of the golf course. Within the golf course, riparian vegetation comprises mown grass and sparse exotic trees, with large areas of bare ground adjacent to the channel. In contrast, downstream of the golf course the banks are covered in long grass immediately beside the channel, and the true right bank is well covered with trees and shrubs.

Fieldwork on 4 and 6 May 2016 revealed moderately cool water temperatures, circum-neutral pH and moderate conductivity values; all values were typical for Christchurch urban streams. However, dissolved oxygen saturation was very low at all four sites sampled (16-34% saturation), and would likely be limiting for sensitive fish species such as brown trout.

Fine sediment (<2 mm diameter) covered most of the bed at all sites sampled, the only exception being where thin coverage exposed the concrete base in the golf course sites. The golf course sites had minimal fish cover, lacking any bank undercuts or macrophytes, with the primary fish cover being leaf packs and fine sediment deposits. Macrophytes covered on average 82% of the bed downstream of the golf course, and were the main source of cover for fish in this location. Macrophytes were dominated by exotic emergent species, particularly *Glyceria fluitans*.

The invertebrate community was dominated by pollution-tolerant taxa at all sites, particularly crustaceans, chironomid midge larvae, oligochaete worms, and molluscs. Pollution-sensitive EPT (Ephemeroptera, Plecoptera, and Trichoptera) species were represented by only one taxon, the cased caddisfly *Triplectides*, which was found in low numbers downstream of the golf course. Taxa richness was greater downstream of the golf course, mainly due to greater numbers of dipteran taxa.

Fish diversity was low overall, with only three species caught: shortfin eel (*Anguilla australis*), inanga (*Galaxias maculatus*), and upland bully (*Gobiomorphus breviceps*). Based on previous studies, longfin eel (*Anguilla dieffenbachii*) and common bully (*Gobiomorphus cotidianus*) likely also occur downstream of the golf course. Eels and inanga are both important mahinga kai, while longfin eel and inanga are also of conservation interest.

Overall, No. 1 Drain within the golf course provides minimal, poor quality aquatic habitat, and has degraded ecological values. Given the minimal flow in No. 1 Drain, and the soft substrates and low DO, we suggest that restoration should focus on improving wetland habitat, and improving stormwater discharged from the site, through the addition of stormwater detention and treatment facilities. We recommend that the primary flow path through the wetland has a v-shaped low flow channel, to maximise water depth and aquatic habitat during low flows, along with the addition of some pools to provide fish habitat. Riparian planting should occur up to the water edge, to help shade the waterway and provide cover for fish and invertebrates.

# 1. INTRODUCTION

No. 1 Drain is a concrete-lined artificial waterway that flows through the Christchurch Golf Club in the suburb of Shirley (Figure 1). Christchurch City Council (CCC) is planning restoration and enhancement work in No. 1 Drain, due to damage to the banks and bed of the drain caused by a series of large earthquakes in 2010 and 2011.

This report describes the results of an aquatic ecology survey of No. 1 Drain, undertaken as a pre-restoration baseline. The purpose of this report is to describe the current state of aquatic habitat, water quality and ecology, and provide recommendations for restoration.

# 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 Course along an open concrete channel. The golf course section of the drain includes one culvert, approximately 45 m long, immediately downstream of "Site 2" in Figure 1. Numerous small drains, approximately 30 mm diameter and 1 m apart, enter both sides of No. 1 Drain in its lower reaches, and possibly also the upper reaches, although this was difficult to verify due to greater water depths. Downstream of the golf course, No. 1 Drain passes through a road culvert under Horseshoe Lake Road and then enters a reach approximately 240 m long with a natural bed, before discharging into Broomfield Waterway and then Horseshoe Lake.

Flow in No. 1 Drain is from land drainage during baseflow conditions. A groundsman indicated that prior to the earthquakes the drain was very shallow, with less than 5 cm depth of water throughout the golf course, but that bed uplift following the earthquakes resulted in a backwatering effect and greater water depths in some sections. Flows rapidly increase following rainfall due to stormwater runoff from the piped headwaters upstream and from the adjacent golf course. There is currently no stormwater detention or treatment in the No. 1 Drain catchment, as residential development in the area predates modern stormwater treatment design and regulations.

Four ecology sampling locations were chosen in consultation with CCC (Figure 1, Table 1). Sites 1 and 4 were located upstream and downstream of the proposed waterway restoration, respectively, and were selected as control sites, while Sites 2 and 3 were within the restoration area and were treatment sites. 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 to serve as a good comparison for the upper sites.

Fieldwork was conducted during low flows over 4 to 6 May 2016, following an unusually dry summer and autumn period.


*Figure 1: No. 1 Drain and ecology sampling sites (numbered yellow circles). Blue lines indicate open waterways, while red lines typically indicate piped or culverted sections. Satellite Imagery from Google.*

*Table 1: Study site locations. Coordinates mark the downstream end of each reach.*

| Site | Easting | Northing | Description |
|---|---|---|---|
| 1 | 1572943 | 5183400 | ~4 m downstream of bridge, near workshop |
| 2 | 1573044 | 5183408 | Immediately upstream of culvert on upstream edge of “green” |
| 3 | 1573262 | 5183481 | Immediately upstream of footbridge on “rough” boundary |
| 4 | 1573461 | 5183956 | ~50 m upstream of No. 2 Drain confluence |

Note: Coordinates measured using the New Zealand Transverse Mercator 2000 projection.

## 2.2. Water Quality

Dissolved oxygen, temperature, pH, and conductivity were measured in the field using a recently-calibrated Horiba U10 water quality meter. No. 1 Drain and Broomfield Waterway are classified as a Spring-fed Plains-Urban stream 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.

## 2.3. Habitat

Habitat data were collected using a combination of Protocol PS of Harding et al. (2009), sediment assessment methods 2 and 6 of Clapcott et al. (2011), and standard CCC protocols (Instream Consulting 2016). 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 5 or 6 transects at 10 m intervals along the reach. Site 1 was only 30 m long, with transects spaced 6 m apart, as this was the only space available upstream of the proposed restoration area. Any potential barriers to fish passage were noted while walking along the stream.

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 1 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 5 measurements per transect, as per protocol P3 of Harding et al. (2009). Velocity was measured using a calibrated Pygmy RS current meter.

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 was 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 & 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 5 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 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. This method is typically used for stony-bottomed streams, and was considered appropriate for this site, based on naturally stony substrates present in nearby No. 2 Drain. Samples were preserved in 70% ethanol solution and were processed by Ryder Consulting Limited. Invertebrate samples were processed using the full count with subsample option, which is protocol P3 of Stark et al. (2001), and identified to species level where practical.

## 2.5. Fish

The fish community at each site 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. Caught fish were transferred to a bucket, then identified, counted, and measured (fork length, mm), before being returned alive to the stream.

## 2.6. Data Analysis

### 2.6.1. 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. Median habitat data from each site were compared statistically using the Kruskal-Wallis test (a nonparametric equivalent to ANOVA), because ANOVA assumptions of normality and homogeneity of variances could not be satisfied, even after data transformation. All statistical tests were undertaken using R statistical software (R Core Team 2016).

Bed cover with filamentous algae, macrophyte cover, 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.

### 2.6.2. Macroinvertebrates

The following biological indices were calculated from the raw invertebrate data:

**Taxa Richness: T**he 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.

**%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. We used calculated site MCI and QMCI scores using the tolerance scores for hard-bottomed streams (Stark & Maxted 2007), as we assumed that substrates are naturally hard-bottomed in the area, based on data from nearby No. 2 Drain. MCI and QMCI scores can be interpreted as per the quality classes of Stark & 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 MCI and QMCI scores (from Stark & 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 and total abundance data were compared statistically amongst sites using ANOVA. The Kruskal-Wallis test was used to compare QMCI and MCI data amongst sites, as these data were not normal, even following transformation. EPT data were not compared statistically, due to very low numbers being recorded (see results section).

Invertebrate community composition was compared amongst sites using non-metric multi-dimensional 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.

#### 2.6.3. Fish

Electric fishing data were converted to abundance per 100 $m^2$ of stream surveyed. Abundance per 100 $m^2$ is a measure of catch per unit of effort (CPUE). Results were compared with data reported for nearby waterways (Taylor & McMurtrie 2003, James 2012, Boffa Miskell 2014), and Freshwater Fish Database records. No statistical analyses were conducted, as the data are not quantitative and too few taxa were captured at each site to calculate meaningful fish community indices.

# 3. RESULTS

## 3.1. Water Quality

At all sites temperatures were moderately cool (<15°C), pH was around neutral (pH=7) to slightly alkaline (pH>7), and conductivity moderate (225-395 µS/cm), and within the range of typical values for spring-fed streams in Christchurch (Margetts & Marshall 2015). Dissolved oxygen (DO) saturation was below the LWRP freshwater outcome of 70% saturation at all the sites, with a minimum of 16% measured at Site 1 and a maximum of 34% at Site 4 (Table 3). DO was amongst the lowest recorded from Christchurch waterways, and was well below the minimum of 70% saturation recorded at the outlet of nearby Horseshoe Lake from monthly samples in 2014 (Margetts & Marshall 2015). Only very tolerant invertebrate and fish species can tolerate such low DO levels.

An oily sheen was observed on the water surface throughout much of Sites 3 and 4 (Figure 2). 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. The sheen is most likely from wetland seepage, as there are no historic landfills under the sampling sites, the sheen occurred both within and downstream of the golf course, and No. 1 Drain is within an area characterised by heavier soils that was covered in extensive wetland prior to land drainage.

*Table 3: Water quality at ecology sampling sites, measured on 4 & 6 May 2016.*

| Site | pH | Conductivity (µS/cm) | Dissolved oxygen (mg/L) | Dissolved oxygen (%) | Temperature (°C) | Time |
|---|---|---|---|---|---|---|
| 1 | 7.08 | 225 | 2.19 | 23 | 14.4 | 10:42 am |
| 2 | 7.21 | 226 | 1.66 | 16 | 13.5 | 3:00 pm |
| 3 | 7.71 | 284 | 2.57 | 25 | 14.4 | 10:05 am |
| 4 | 7.25 | 395 | 3.52 | 34 | 13.8 | 3:15 pm |

*Figure 2: Oily sheens on the water at Site 3 (left) and Site 4 (right).*

## 3.2. Habitat

Representative site photographs are shown in Figure 3 and additional photographs are provided in Appendix 1. Results of all statistical tests are in Appendix 2.

No. 1 Drain within the golf course is narrow, straight, concrete-lined and has near-vertical walls (Figure 3). At Site 2, the top of the south bank was leaning over the channel, due to earthquake damage, and wooden bracing was in place to stop the bank from completely toppling over (Figure 3). There was a significant difference in lower bank angle amongst sites (P<0.001), with mean bank angles of 88 to 99 degrees for Sites 1 to 3, and a mean of 21 degrees at Site 4. Bank heights also differed significantly amongst sites (P<0.001), with Sites 1 to 3 having uniform bank heights of 1.1 m, while Site 4 bank height was on average 0.4 m and was more variable.  Bank undercuts were absent from all sites, due to the concrete walls at Sites 1 to 3 and the relatively low bank angles at Site 4.


*Figure 3: Representative photographs of the four ecology sampling sites.*

Riparian vegetation cover differed significantly amongst sites (P<0.05) for most combinations of tier height and distance from the channel assessed (see Appendix 2 for details). Overall, riparian ground cover at the golf course sites was dominated by bare ground and mown grass closest to the channel, and mown grass with some sparse exotic trees further from the channel (Figure 4). Large areas of bare ground adjacent to the channel provided minimal buffering at the golf course sites. At Site 4 long grass dominated ground cover close to the channel, and on the true right bank rank grass graded into mature *Carex* and a mix of taller shrubs and trees, including willow, dense blackberry and the occasional fern. The most statistically significant pattern was that Site 4 had greater cover of long grass (0.3-1.9 m tall) and shrubs (2.0-4.9 m) than the other sites (P<0.001).

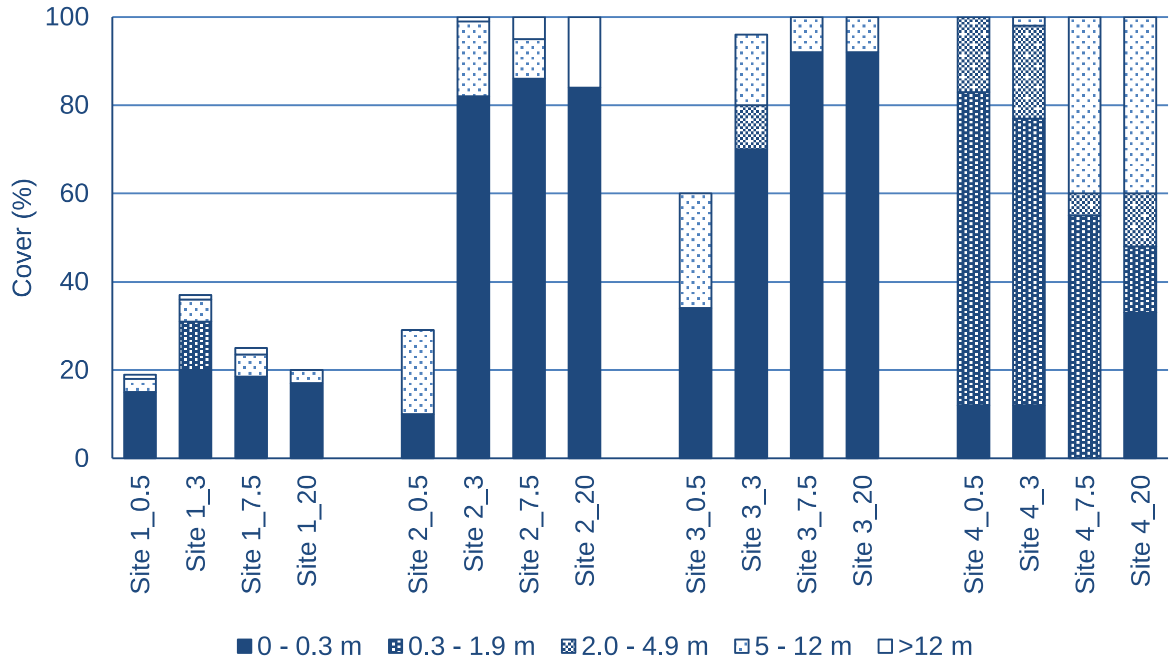


*Figure 4: Percent ground cover with different vegetation tier heights (indicated by bar legend colours) at 0.5, 3.0, 7.5, and 20 m distance (indicated by x-axis label suffixes) from the water edge at each site. Locations with less than 100% vegetation cover had bare ground comprising the remaining area.*

Channel shading differed significantly amongst sites (P<0.01), with shading greatest at Site 1 (mean = 46% shade), from a combination of sparse exotic trees and bridges, and lowest at Site 4 (7% shade), due to a lack of tall trees adjacent to the channel (Table 4).

The amount of vegetation overhanging the channel differed significantly amongst sites (P<0.001), with minimal or no vegetation overhanging the golf course sites, and an average of 40 cm of vegetation (mainly grass) overhanging the water at Site 4 (Table 4). The lack of natural banks and overhanging vegetation at the golf course sites provide no habitat for terrestrial invertebrates and no fish cover.

Channel widths did not differ significantly amongst sites (P=0.09), although the three golf course sites were characterised by uniform channel widths of approximately 1.1 m, whereas Site 4 was slightly wider (mean 1.5 m) and considerably more variable in width (Table 4)

Mean water depths differed significantly amongst sites (P<0.001), and ranged from 5 cm at Site 4 to 40 cm at Site 1 (Table 4). The golf course groundsman indicated that the drain had uniformly shallow water depths prior to the earthquakes, and that the increased depths in the upper reaches of the drain were caused by bed uplift during the earthquakes.

There was no perceptible flow at all of the golf course sites, with water velocities less than 0.06 m/s (the Pygmy meter detection limit) at all of Sites 1 to 3. Site 4 also had minimal flow, and velocities were near zero amongst beds of emergent macrophytes. However, velocities of 0.1 to 0.2 m/s were detected in narrow, open water sections at Site 4.

*Table 4: No. 1 Drain habitat characteristics. Data are site means.*

| Parameter | Site | | | |
|---|---|---|---|---|
| | **1** | **2** | **3** | **4** |
| Width (m) | 1.18 | 1.11 | 1.20 | 1.54 |
| Depth (m) | 0.40 | 0.38 | 0.09 | 0.05 |
| Bank height (m) | 1.08 | 1.12 | 1.11 | 0.43 |
| Bank angle (degrees) | 91 | 99 | 88 | 24 |
| Buffer width (m) | 0.1 | 0.0 | 60.9 | 56.7 |
| Shade (%) | 46 | 17 | 33 | 7 |
| Overhanging vegetation (cm) | 1 | 0 | 0 | 39 |
| Fine sediment cover (%) | 100 | 77 | 100 | 93 |
| Fine sediment depth (cm) | 8 | 4 | 16 | 20 |
| Leaf packs (cm) | 31 | 62 | 27 | 3 |
| Algae (cm) | 5 | 8 | 80 | 0 |
| Macrophytes (cm) | 0 | 0 | 0 | 113 |

Fine sediment (<2 mm diameter) dominated bed sediments at all sites, with fine sediment cover at all sites at least double the LWRP freshwater outcome of 30% cover (Table 4). There were no significant differences in fine sediment cover amongst sites (P=0.065). Fine sediment depth differed significantly amongst sites (P=0.006), with mean depths of around 4-8 cm at Sites 1 and 2, and sediment depths of 16-20 cm at Sites 3 and 4. Examination of sediments at Site 3 revealed a fine organic layer overlaying fine sand, with an anoxic black sand layer underneath.

Macrophyte cover differed significantly amongst sites (P<0.001), with little or no macrophyte cover at the golf course sites and an average of 82% cover at Site 4 (Figure 5). All of the Site 4 macrophytes were emergent species, with the exotic *Glyceria fluitans* being particularly common. Exotic starwort (*Callitriche stagnalis*) was also relatively abundant, while native duckweed (*Lemna* sp.), and azolla (*Azolla rubra*) were sparse. The lack of macrophytes at the golf course sites was likely due to the lack of natural banks for macrophytes to grow on.

Periphyton cover was sparse at Sites 1, 2, and 4, while Site 3 had moderate coverage with thin green films and short green filamentous algae (Figure 6). Bed coverage with long green filamentous algae was <10% at all sites, and well below the LWRP freshwater outcome of 30% bed cover (Figure 5, Table 4). No periphyton was observed at Site 4, presumably due to the dominance of macrophytes and soft sediments.

Bed cover with organic matter ranged from 33% at Site 1 to 94% at Site 3, and mainly comprised a mix of fine detritus, leaves and small sticks. However, there was significantly greater bed coverage with leaf packs at the golf course sites than Site 4 (P<0.001), due to the litter contribution from large deciduous trees and the minimal water velocity at the golf course sites.

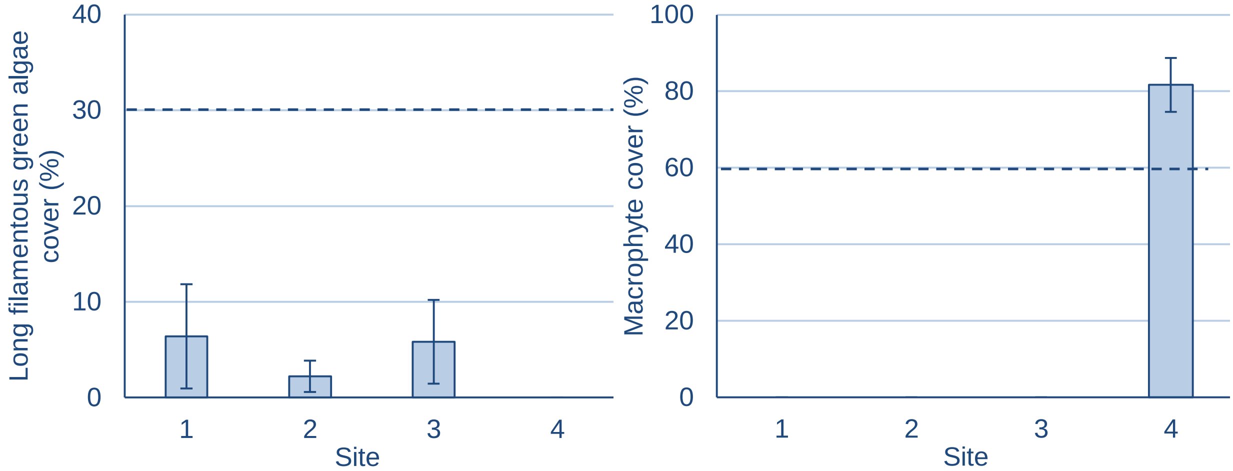


*Figure 5: Mean (±1 SE) bed cover with long green filamentous algae (left) and total macrophyte cover (right). Dashed lines indicate LWRP freshwater outcomes.*

No major impediments to fish passage were observed, although water depths at Sites 3 (mean depth = 0.09 m) and 4 (mean depth = 0.05 m) are likely too shallow to provide significant habitat or passage for larger-bodied species such as adult brown trout (*Salmo trutta*).

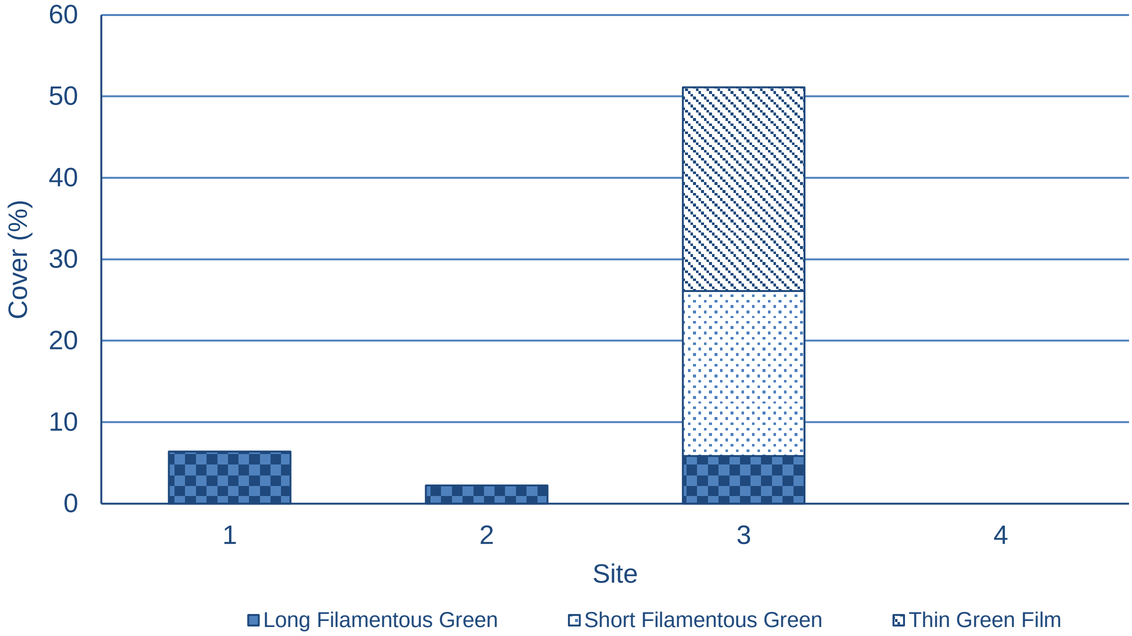


*Figure 6: Cover and composition of major periphyton groups.*

### 3.3. Macroinvertebrates

A total of 29 invertebrate taxa were recorded across the four sites and the invertebrate community was numerically dominated by pollution-tolerant taxa at all sites. The five most abundant taxa across all sites were ostracod crustaceans (27% of total abundance) the blood-worm *Chironomus zealandicus* (21%), oligochaete worms (17%), the common mudsnail *Potamopyrgus antiposarum* (13%), and sphaeriid molluscs (11%). Other common but less abundant taxa included *Gyraulus* snails, cladocerans, copepods, and chironomid midge larvae (Figure 7). Overall, invertebrate community composition was similar to that recorded previously from other drains in the area (Taylor & McMurtrie 2003, James 2012, Boffa Miskell 2014).

Pollution-sensitive EPT taxa were represented by only one taxon, the cased caddisfly *Triplectides*, which was found in low numbers at Site 4. Hence, mean percent EPT abundance was below 1% and EPT taxa richness was below 5% at all sites.

Taxa richness varied significantly amongst sites (P<0.001), and taxa richness was greatest at Site 4, which had a mean of 17 taxa per 0.1 m², while all of the golf course sites had a mean of 10 taxa per 0.1 m² (Figure 8). Higher taxa richness at Site 4 was primarily due to greater numbers of dipteran taxa being present. Mean invertebrate abundance ranged from 552 individuals per 0.1 m² at Site 2 1,250 per 0.1 m² at Site 3, but differences amongst sites were not statistically significant (P=0.0938), due to considerable variation between samples within sites (Figure 8).

MCI scores were very low, did not differ significantly amongst sites (P=0.1492), and were indicative of poor water quality or habitat at all sites (Figure 8). QMCI scores differed significantly amongst sites (P=0.0397), although all sites had low mean QMCI scores in the range of 1.9 to 2.7, and indicative of poor conditions (Figure 8). At all sites, mean QMCI scores were below the LWRP freshwater outcome QMCI score of 3.5 for Spring-fed Plains-Urban streams. Low MCI and QMCI scores reflected the dominance of pollution-tolerant taxa.

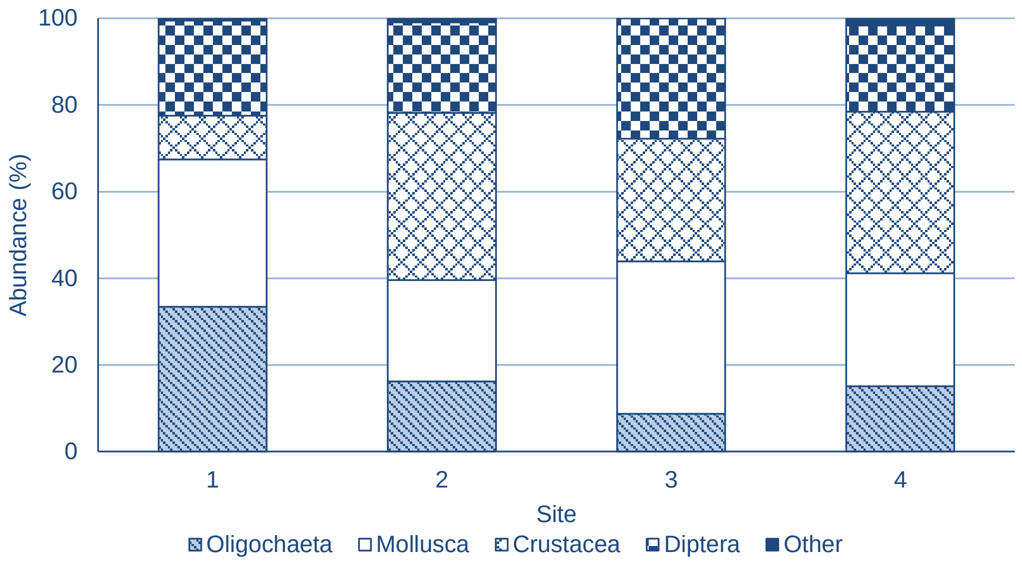


*Figure 7: Relative abundance of major invertebrate taxa at each site.*

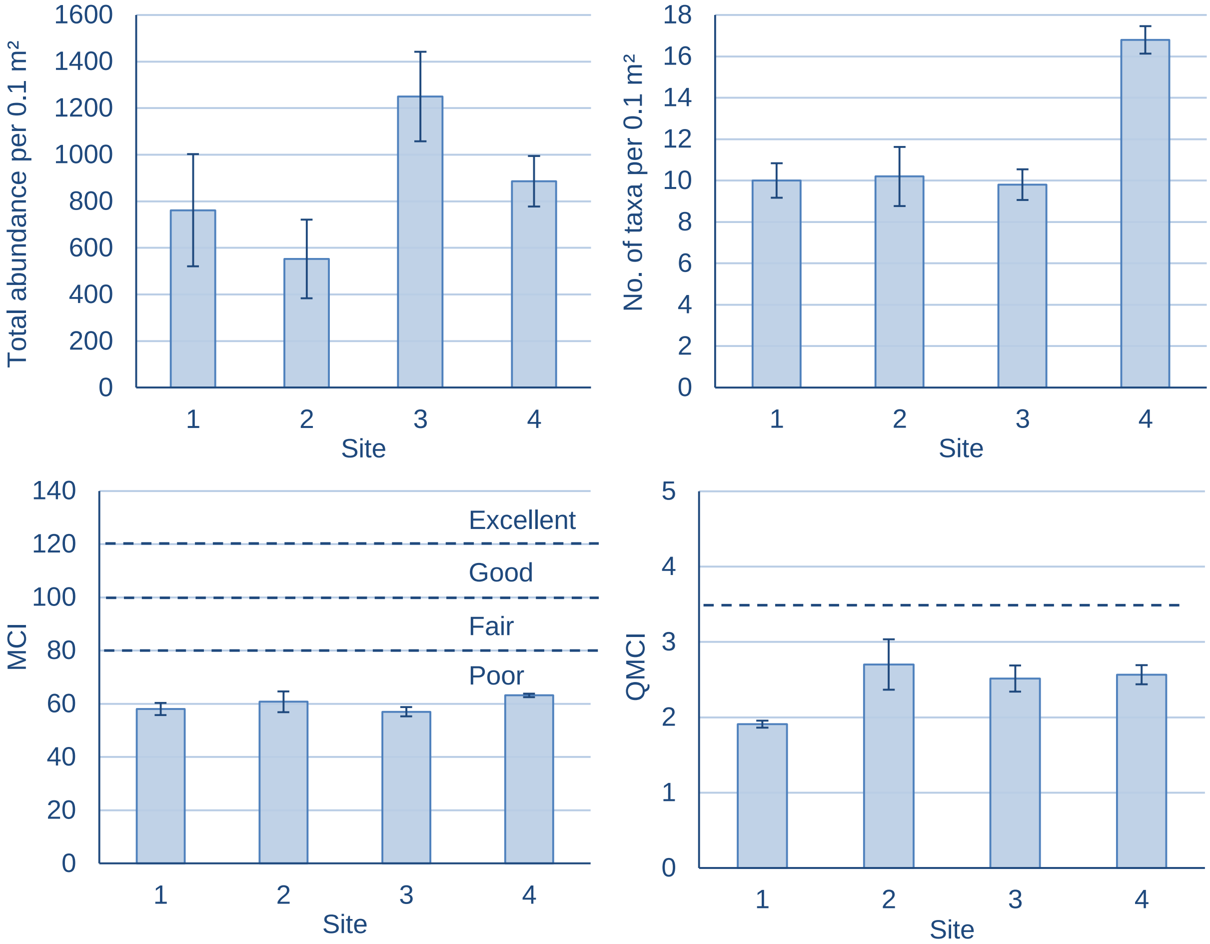


*Figure 8: Mean (±1 SE) total invertebrate abundance, taxa richness, MCI, and QMCI scores. Dashed lines on the MCI plot indicate quality classes from Stark & Maxted (2007), while the dashed line on the QMCI plot indicates the LWRP freshwater outcome of 3.5.*

Ordination of the invertebrate community yielded a two-dimensional solution with moderate stress (0.21), indicating a potentially suspect relationship between the original dissimilarity matrix and distance in ordination space (Clarke 1993). However, invertebrate correlation coefficients with each axis compare favourably with patterns observed in raw invertebrate data (Appendix 3), so the ordination appears a reasonable representation of community dissimilarity amongst samples. The ordination plot (Figure 9) showed considerable overlap between sites, indicating no strong differences in community composition amongst sites. However, samples from Sites 1 and 2 tend towards the right of the ordination plot, reflecting a greater abundance of sphaeriid bivalves and fewer snails (Mollusca), chironomid midge larvae, and ostracods (Figure 9).

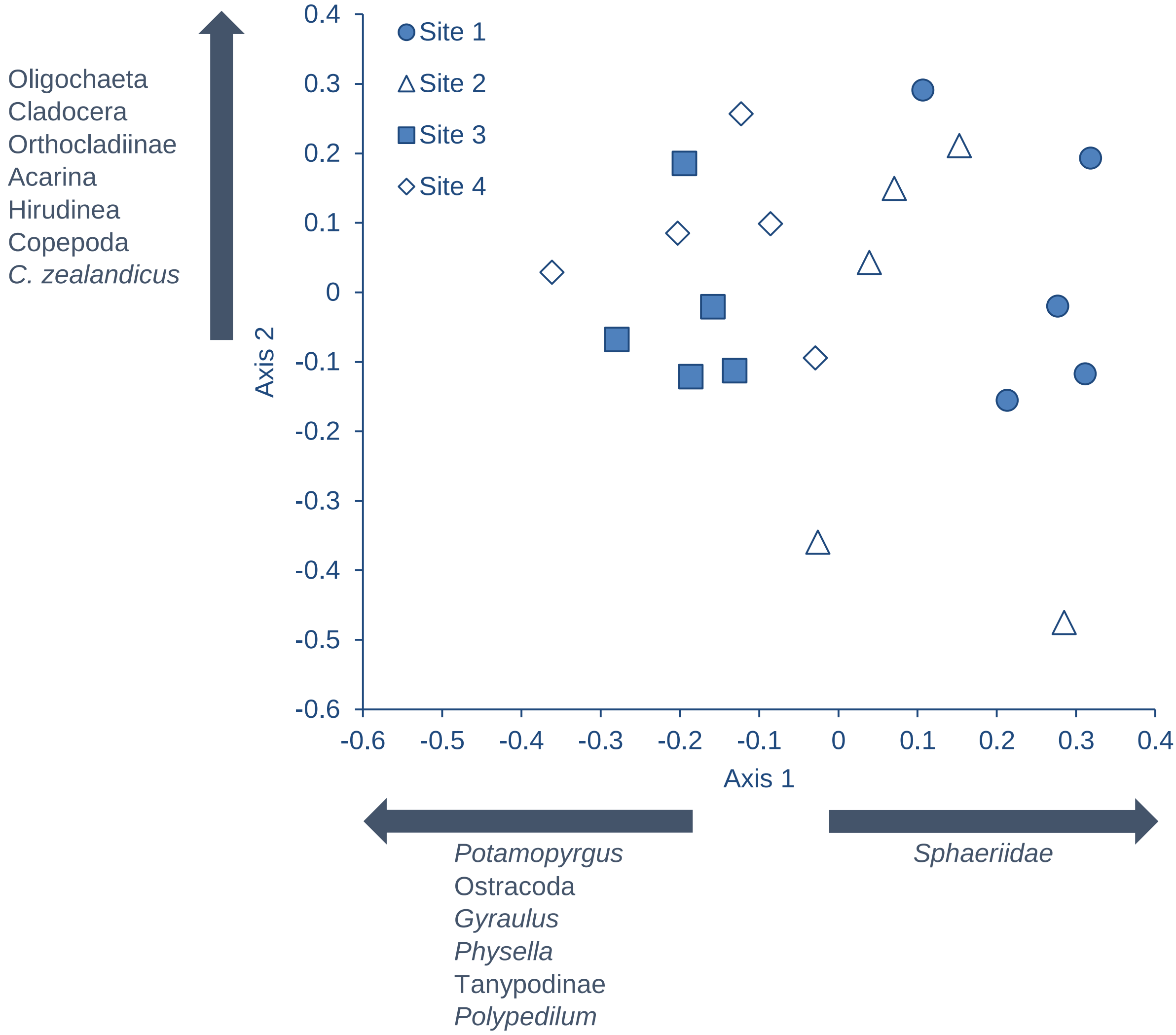


*Figure 9: NMDS ordination of the invertebrate community (stress = 0.21). Each symbol represents a replicate sample from the four sites. Invertebrate taxa significantly correlated (P<0.05) with each axis are shown.*

## 3.4. Fish

Fish diversity was low overall, with only three species caught: shortfin eel (*Anguilla australis*), inanga (*Galaxias maculatus*), and upland bully (*Gobiomorphus breviceps*). No fish were caught at Site 1, although a single large eel was observed prior to fishing. Only shortfin eel were caught at Sites 2 and 3, but an adult inanga was observed at Site 3 (Table 5). Eel abundance at Site 4 was double that at Sites 2 and 3 (Figure 10). Shortfin eel are a common native species that are also important mahinga kai. Upland bullies are common native species, while native inanga are of conservation interest because they are classified as At Risk – Declining, and their juveniles comprise the annual whitebait catch that is of cultural and recreational value.

Other fish species likely to occur in the vicinity of No. 1 Drain include common bully (*Gobiomorphus cotidianus*), and longfin eel (*Anguilla dieffenbachii*), based on nearby Freshwater Fish Database records and other reports (Taylor & McMurtrie 2003, James 2012, Boffa Miskell 2014). Native longfin eel are important mahinga kai and have an At Risk

– Declining status, while native common bully are not threatened (Goodman et al. 2014). Bluegill bully (*Gobiomorphus hubsii*) occur in the lower reaches of nearby No. 2 Drain (Taylor & McMurtrie 2003, James 2012), which is of conservation interest because of their At Risk – Declining status, and because they are very uncommon in Christchurch waterways. However, bluegill bully are unlikely to occur in No. 1 Drain because their preferred habitat is swift, stony riffles, and water velocities are too low and substrates are too fine for bluegill bullies in No. 1 Drain.

*Table 5: Fish caught at each sampling site. Fish seen, but not caught are marked "*".*

| Common name | Scientific name | Site 1 | Site 2 | Site 3 | Site 4 |
|---|---|---|---|---|---|
| Shortfin eel | *Anguilla australis* | * | 3 | 4 | 15 |
| Upland bully | *Gobiomorphus breviceps* | | | | 1 |
| Inanga | *Galaxias maculatus* | | | * | 1 |
| **Total** | | **0** | **3** | **4** | **17** |
| Area fished ($m^2$) | | 35 | 33 | 36 | 62 |

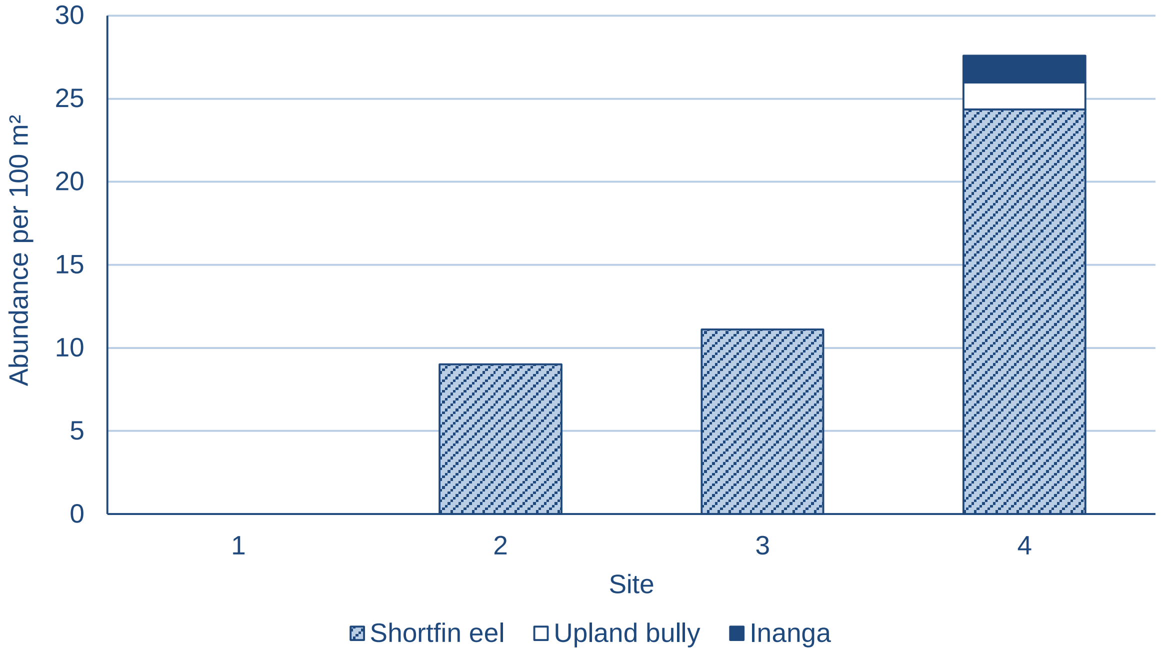


*Figure 10: Fish species and abundance at the four sampling sites.*

# 4. DISCUSSION

All of the No. 1 Drain sites surveyed within the golf course had very poor habitat quality, with minimal variation amongst the sites. The golf course sites had either mown grass or bare ground adjacent to the waterway, providing poor riparian buffering and little shade. The straight concrete channel within the golf course results in a lack of variation in instream habitat and provides no habitat for aquatic plants to establish. Deposits of fine sediment and leaves provide the only cover for fish and invertebrates in the golf course. Downstream of the golf course, natural banks allow macrophytes to establish, which provide habitat for a more diverse invertebrate fauna and cover for fish.

In addition to having generally poor physical habitat, lack of flow throughout No. 1 Drain is likely to be a major factor affecting biological communities. Swift water velocities are generally preferred by pollution-sensitive EPT taxa, as well as riffle-dwelling fish species such as bluegill bully. Thus, the presence of bluegill bully and greater abundance of EPT taxa in the lower reaches of nearby No. 2 Drain reflects the combination of coarser substrates and greater flows in that location. However, the addition of coarse substrates to No. 1 Drain is unlikely to enhance habitat for EPT taxa or bluegill bully, due to a lack of flow and associated low water velocities. Low DO concentrations throughout No. 1 Drain would also limit the biota to more tolerant species.

Overall, the golf course section of No. 1 Drain currently has low ecological value, but it does have potential for ecological enhancement. The key values of the golf course section are that it is mostly an open reach (as opposed to being piped) and that it is connected to more natural reaches downstream.

# 5. RESTORATION RECOMMENDATIONS

As discussed in the previous section, the golf course section of No. 1 Drain currently has poor aquatic habitat. In addition, minimal flow and low DO concentrations likely further limit the biota. Given the lack of flow and gentle gradient, we suggest that the restoration goal for the site should be to improve wetland values, and downstream water quality, rather than attempting to create a swift, stony-bottomed stream. Downstream water quality could be improved through the addition of a detention basin and wetland for stormwater treatment within the golf course, as proposed by CCC.

We do not see any particular value in replacing the 45 m culvert in the upper reaches of the golf course with an open section of waterway. So-called “daylighting” has been shown to increase the abundance of sensitive EPT taxa in some New Zealand streams (Neale & Moffett 2016). However, we consider daylighting is unlikely to benefit the upper reaches of No. 1 Drain, given the lack of flow and adequate velocities for EPT taxa, and the lack of an upstream or nearby source of colonists. We consider that restoration efforts could be better spent on improving general wetland values and treating stormwater, as outlined below.

We recommend the actions listed below to improve wetland values and improve downstream water quality. These actions add to and expand on the general recommendations of Taylor & McMurtrie (2003).

- **Stormwater treatment.** The addition of a stormwater detention pond and wetland will help treat stormwater from existing residential areas, where there is currently no treatment. This should help improve downstream water quality and hydrology.
- **Natural banks.** The existing concrete channel with vertical walls should be replaced with more natural construction materials (e.g., rock or geotextile bags) with the banks battered back to limit erosion. Natural banks provide habitat for riparian plants and animals, and allow macrophytes to take root, which in turn provide habitat and cover for fish and invertebrates.
- **Low flow habitat.** A v-shaped low flow channel should be created throughout the length of the golf course, to maximise depths in flowing sections for invertebrates and fish.
- **Deeper pools.** Including pools in the channel design will provide some deeper aquatic habitat for fish species such as eels and inanga, while also helping to trap fine sediment before it is transported further downstream.
- **Riparian planting.** Native vegetation should be planted up to the water's edge, to shade-out nuisance macrophytes and provide habitat for fish and invertebrates. Increased stream shading should reduce the need for regular drain clearance, while plants on the lower banks will overhang the water and provide habitat and localised shading.
- **Fish salvage.** All fish should be removed from the affected length of drain prior to any major instream channel works or realignment.
- **Monitor success.** Water quality and ecological monitoring should be undertaken following completion of the restoration activities, to evaluate success of the works. As a minimum, monitoring should include: measurement of common stormwater contaminants (e.g., suspended sediment and metals) upstream and downstream of the works, before and after completion of the stormwater treatment facilities; measuring riparian vegetation cover and species composition along the restored channel; and sampling fish upstream, along, and downstream of the restored channel.

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# APPENDIX 1: SITE PHOTOGRAPHS

*Figure 1: Site 1, view upstream from the bottom of the reach.*

*Figure 2: Site 1, view downstream from the top of the reach.*

*Figure 3: Site 2, view upstream from the bottom of the reach.*

*Figure 4: Site 2, view downstream from the top of the reach.*

*Figure 5: Site 3, view upstream from the bottom of the reach.*

*Figure 6: Site 3, view downstream from the top of the reach.*

*Figure 7: Site 4, view upstream from the bottom of the reach.*

*Figure 8: Site 4, view downstream from the top of the reach.*

## APPENDIX 2: STATISTICS SUMMARY

All analyses except for invertebrate taxa richness and abundance used Kruskal-Wallis for site comparisons, due to large differences in variance between groups (i.e., violation of ANOVA assumption of homogeneity of variances). ANOVA was used for invertebrate taxa richness and abundance, as ANOVA assumptions were met. For post-hoc comparisons, sites sharing the same horizontal bars have medians (or means in the case of ANOVA) that do not differ significantly (i.e., p ≥0.05). Non-significant post-hoc comparisons are indicated as "ns".

| Parameter | P-value | Post-hoc comparisons |
|---|---|---|
| Width (m) | 0.0939 | ns |
| Depth (m) | **0.0005** | 4 3 2 1 |
| Lower Bank Height (m) | **0.0043** | 4 1 3 2 |
| Lower Bank Angle (degrees) | **0.0005** | 2 1 3 4 |
| Fine Sediment Cover (%) | 0.0649 | ns |
| Fine Sediment Depth (cm) | **0.0063** | 2 1 3 4 |
| Shade (%) | **0.0019** | 4 2 3 1 |
| Algae (cm) | **0.0082** | 4 1 2 3 |
| Macrophytes (cm) | **<0.0001** | 1 2 3 4 |
| Macrophyte cover (%) | **<0.0001** | 1 2 3 4 |
| Organic matter (%) | **0.0029** | 1 4 2 3 |
| Leaf packs (cm) | **0.0104** | 4 3 1 2 |
| Overhanging vegetation (cm) | **0.0008** | 2 3 1 4 |
| Riparian buffer width | **0.0011** | 2 1 4 3 |
| Invertebrate Abundance* | 0.0938 | ns |
| Invertebrate Taxa Richness | **0.0002** | 3 1 2 4 |
| MCI Scores | 0.1492 | ns |
| QMCI Scores | 0.0397 | ns |

| Vegetation Height (m) Tier | Distance from Bank (m) | P-value | Post-hoc comparisons |
|---|---|---|---|
| 0 – 0.3 m | 0.5 | 0.3860 | 2 4 1 3 |
| | 3.0 | **0.0363** | 4 1 3 2 |
| | 7.5 | **0.0010** | 4 1 2 3 |
| | 20.0 | **0.0014** | 1 4 2 3 |
| | | | |
| 0.3 – 1.9 m | 0.5 | **0.0003** | 1 2 3 4 |
| | 3.0 | **0.0016** | 2 3 1 4 |
| | 7.5 | **0.0003** | 1 2 3 4 |
| | 20.0 | **0.0003** | 1 2 3 4 |
| | | | |
| 2.0 – 4.9 m | 0.5 | **0.0028** | 1 2 3 4 |
| | 3.0 | **0.0205** | 1 2 3 4 |
| | 7.5 | **0.0003** | 1 2 3 4 |
| | 20.0 | **0.0003** | 1 2 3 4 |
| | | | |
| 5 – 12 m | 0.5 | 0.4953 | 4 1 2 3 |
| | 3.0 | 0.7921 | 4 1 2 3 |
| | 7.5 | **0.0265** | 1 3 2 4 |
| | 20.0 | **0.0032** | 2 1 3 4 |
| | | | |
| >12 m | 0.5 | 0.3916 | 2 1 3 4 |
| | 3.0 | 0.5497 | 3 4 1 2 |
| | 7.5 | 0.5487 | 3 4 1 2 |
| | 20.0 | **0.0187** | 3 4 1 2 |

# APPENDIX 3: RAW INVERTEBRATE DATA AND INDICES

| Taxon | MCI | Site 1 | | | | | Site 2 | | | | | Site 3 | | | | | Site 4 | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | a | b | c | d | e | a | b | c | d | e | a | b | c | d | e | a | b | c | d | e |
| ACARINA | 5 | 2 | | | 1 | | | 1 | 1 | 5 | 1 | | | | | | | 1 | | | |
| CNIDARIA | | | | | | | | | | | | | | | | | | | | | |
| *Hydra* species | 3 | | | | | | | 1 | | | | | | | | | | | | | |
| COLEOPTERA | | | | | | | | | | | | | | | | | | | | | |
| *Enochrus tritus* (Hydrophilidae) | 5 | | | | | | | | | | | | | | | | | | | | 1 |
| COLLEMBOLA | 6 | | | | | | | | | | | | | | | | | | | 14 | |
| CRUSTACEA | | | | | | | | | | | | | | | | | | | | | |
| Cyclopoida (Copepoda) | 5 | 2 | | 1 | 8 | | | 151 | | 81 | 35 | 3 | | 2 | | | | 1 | | | |
| Ostracoda | 3 | 208 | 11 | 24 | 117 | 52 | 6 | 115 | 78 | 260 | 214 | 214 | 261 | 230 | 419 | 623 | 294 | 215 | 778 | 357 | 118 |
| Cladocera | 5 | 6 | | | | | | 301 | | 107 | 12 | | | | | | | 2 | | | |
| DIPTERA | | | | | | | | | | | | | | | | | | | | | |
| *Austrosimulium australense*-group | 3 | | | | | | | | | | | | | | | | | 1 | 1 | | |
| *Chironomus zealandicus* | 1 | 306 | 99 | 112 | 87 | 87 | 28 | 178 | 32 | 108 | 120 | 215 | 131 | 554 | 694 | 225 | 149 | 149 | 60 | 140 | 78 |
| *Corynoneura scutellata* | 2 | | | | | | | | | | | | | | | | | 3 | | 1 | |
| Hexatomini | 5 | | | | | | | | | | | | | | | | | | | | 1 |
| Muscidae | 3 | | | | | | | | | | | | | | | | 1 | | 1 | | 1 |
| Orthocladiinae | 2 | 2 | | | 2 | | | 4 | | | | | 1 | | 1 | | 19 | 53 | 38 | 9 | 6 |
| *Paradixa fuscinervis* | 4 | | | | | | | | | | | | | | | | | | 1 | | |
| *Polypedilum* species | 3 | | | | | | | | | | | | | | | | 6 | 1 | 8 | 5 | |
| Tanypodinae | 5 | | | 3 | | | | 1 | | | 5 | | | 1 | 6 | 4 | 4 | 11 | 2 | 2 | 1 |
| Tanytarsini | 3 | | | | | | | | | | | | | | | | 36 | 50 | 12 | 12 | 6 |
| *Zelandotipula* species | 6 | | | | | | | | | | | | | | | | 1 | | | | |
| HEMIPTERA | | | | | | | | | | | | | | | | | | | | | |
| *Microvelia macgregori* | 5 | | | | | | | | | | | | | | | | | | | 1 | |
| HIRUDINEA | 3 | 4 | 1 | 6 | | | | 1 | | 3 | 2 | | | 1 | | | 7 | 15 | 3 | 1 | 2 |
| MOLLUSCA | | | | | | | | | | | | | | | | | | | | | |
| *Ferissia neozelanica* | 3 | 4 | 1 | 7 | | 1 | | 2 | | | 1 | | 15 | 4 | 1 | 1 | | | | | |
| *Gyraulus corinna* | 3 | 15 | 6 | 7 | | 1 | 5 | 28 | 1 | 24 | 37 | 23 | 181 | 104 | 23 | 80 | 73 | 22 | 96 | 20 | 24 |
| *Physella (Physa) acuta* | 3 | | | 4 | 1 | | | | | | | 3 | 2 | 3 | 4 | 7 | 17 | 10 | 6 | 2 | 14 |
| *Potamopyrgus antipodarum* | 4 | 12 | 1 | 1 | 3 | 4 | | 27 | 104 | 49 | 66 | 435 | 120 | 170 | 325 | 370 | 152 | 94 | 116 | 56 | 150 |
| Sphaeriidae | 3 | 374 | 154 | 130 | 502 | 64 | 4 | 45 | 31 | 44 | 65 | 18 | 12 | 53 | 95 | 7 | 54 | 26 | 6 | 86 | 44 |
| NEMATODA | 3 | | | | 1 | 1 | | | | | | | | | 1 | | 4 | 5 | 1 | 2 | |
| OLIGOCHAETA | 1 | 523 | 84 | 170 | 501 | 92 | 25 | 107 | 6 | 130 | 101 | 59 | 80 | 57 | 342 | 68 | 98 | 323 | 55 | 92 | 86 |
| PLATYHELMINTHES | 3 | | | | 1 | | 1 | | 1 | 6 | | | | | 1 | | 5 | 7 | | | |
| TRICHOPTERA | | | | | | | | | | | | | | | | | | | | | |
| *Triplectides cephalotes* | 5 | | | | | | | | | | | | | | | | 1 | | 4 | | 1 |
| | | | | | | | | | | | | | | | | | | | | | |
| Total abundance | | 1458 | 357 | 465 | 1224 | 302 | 69 | 962 | 254 | 817 | 659 | 970 | 803 | 1179 | 1912 | 1385 | 921 | 989 | 1188 | 800 | 533 |
| Taxa richness | | 12 | 8 | 11 | 11 | 8 | 6 | 14 | 8 | 11 | 12 | 8 | 9 | 11 | 12 | 9 | 17 | 19 | 17 | 16 | 15 |
| EPT richness | | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 1 | 0 | 1 | 0 | 1 |
| %EPT abundance | | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0 | 0.1 | 0 | 0.3 | 0 | 0.2 |
| MCI-hb | | 63 | 53 | 62 | 60 | 53 | 47 | 66 | 58 | 65 | 68 | 58 | 51 | 62 | 57 | 58 | 64 | 63 | 61 | 63 | 65 |
| QMCI-hb | | 1.88 | 1.98 | 1.81 | 2.05 | 1.83 | 1.46 | 3.38 | 3.12 | 2.95 | 2.59 | 2.89 | 2.62 | 2.11 | 2.09 | 2.85 | 2.62 | 2.11 | 2.88 | 2.54 | 2.67 |