Boffa Miskell

Kā Pūtahi Creek Ecology

Five years after the realignment works

Prepared for Christchurch City Council

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Executive Summary

The Christchurch City Council (CCC) commissioned Boffa Miskell Limited to conduct a repeat ecological survey of five sites within the Kā Pūtahi Creek following its realignment to avoid requiring multiple box culverts under the Northern Arterial Motorway in 2016. The monitoring programme aims to determine if the realignment and rehabilitation works has resulted in any measurable ecological changes and identify any limitations to the success of the rehabilitation works.

A variety of riparian and in-stream habitat variables, basic water quality measures and assessments of the macroinvertebrate and fish communities were made at two control sites and three sites within the realigned reach in March 2021. Methods used were generally the same as those used in the previous two surveys; baseline (2016, prior to realignment works) and one-year post-rehabilitation works (2017). This survey (2021) was undertaken five years following realignment and rehabilitation works.

Habitat conditions throughout the realignment sites were generally similar, though some key differences between realignment and control sites, and temporal changes in conditions, were evident. Compared to baseline conditions, the riparian vegetation is significantly improved, with a diverse array of indigenous plants becoming well established. Along the stream banks, sedges, toetoe and flax have become well established and overhang the stream increasing habitat availability further back. Further, these plants have helped to maintain stable banks and increase the buffer zone surrounding the waterway.

In-stream habitat improvements have also been observed since the baseline assessment, with increased habitat heterogeneity from the addition of cobble substrate. However, fine sediments have started to cover these coarser substrates added to the realigned channel since the ‘Year One’ survey. Similarly, macrophyte cover has increased at realigned sites since the ‘Year One’ survey and is comparatively higher than that in the control sites. The realigned and control reaches of Kā Pūtahi Creek are wide in places and velocity very slow, which leads to deposition of sediments that continue to enter the waterway upstream of the realigned reach.

While there were some subtle changes (improvement) in the macroinvertebrate community in response to rehabilitation works as shown by the 2017 surveying, the 2021 results show the community is now relatively similar among sites.

Macroinvertebrate abundance has increased since the baseline and ‘Year One’ surveys, though taxa found remain typically “pollution tolerant”. Crustaceans were the dominant group found across all sites in this survey. A shift in the macroinvertebrate community was observed between the baseline and ‘Year One’ surveys. However, the community found in 2021 has become more similar to the baseline condition at all sites. These changes are likely due to increasing cover of soft sediment on the cobble substrates of the realigned reach.

No significant changes in the fish community have been detected since the baseline survey, with a similar community being found in all surveys. The number of fish caught at realigned sites was similar to that found at control sites, and no new species or notable changes in abundance of any species have been evident. While giant bullies, which have been found in the previous two surveys, were not encountered in this survey, they are appeared to only have been present in low numbers and thus likelihood of capture is low. Further, different survey methods used (i.e., trapping rather than electric fishing) have different biases for capturing certain species (e.g., giant bully and inanga are underestimated using electric-fishing methods), which may have had some influence on the fish communities found. Habitat for giant bullies is available in Kā Pūtahi Creek, and the species may still be present.

Overall, realigned works have improved and diversified habitat at the realigned sites when compared to control sites and baseline conditions. However, high sediment loads and macrophyte growth in the catchment, and slow velocity in the wider parts of the waterway contributing to further deposition has resulted in a high cover of sediment in the realigned reach. As a result, habitat quality and availability for aquatic fauna (particularly aquatic insects, EPT fauna) is reduced, and is reflected in the lack of improvement of QMCI scores at these sites.

Limitations to further ecological improvement exist, including ongoing sediment inputs and high macrophyte growth in the catchment resulting in high rates of deposition, limited flow influenced by the wideness of the waterway, and in places, a lack of large boulders and logs providing greater habitat diversity.

Ongoing management and continuation of rehabilitation work is likely necessary to address issues and further habitat improvement gains throughout the Kā Pūtahi Creek.

Recommendations to improve the condition of the waterway include addressing upstream catchment-scale sediment inputs, managing macrophyte growth (e.g., through increasing channel shading) to Kā Pūtahi Creek and reducing the wetted width of the waterway to increase flow velocity by adding logs and boulders along the wetted edge of the stream.

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Appendices

1.0 Introduction

Waka Kotahi NZ Transport Agency constructed a new motorway, the Christchurch Northern Corridor (CNC), to link Christchurch City with State Highway 1. The CNC's original proposal involved installing two long box culverts to cross Kā Pūtahi Creek in Marshlands. The Christchurch City Council (CCC) realigned Kā Pūtahi Creek in 2016 to avoid these crossings. The realignment works replaced approximately 250 m of "oxbow" with 350 m of pool-riffle-run sequences, and reinstated *Carex*-dominated floodplains, with overhanging vegetation and varied in-stream and bank profiles (Shadbolt 2015).

To measure the ecological success of the realignment and enhancement works, CCC has been carrying out ecological monitoring of the creek. A baseline survey of the existing (pre-realignment) riparian and in-stream habitat conditions, and the macroinvertebrate and fish communities was carried out at four sites in 2016 (two sites within the oxbow to be realigned, and two control sites upstream and downstream of the realigned section) (Boffa Miskell, 2016). The first post-realignment survey occurred in 2017 (EOS Ecology, 2017), including the two control sites previously surveyed plus three sites within the newly realigned Kā Pūtahi Creek.

This report details the findings of habitat and faunal community surveys undertaken five years following the realignment of Kā Pūtahi Creek and compares the results to the previous surveys.

1.1 Scope

The CCC commissioned Boffa Miskell to resurvey the five sites within Kā Pūtahi Creek as part of its ongoing monitoring programme to measure ecological success of the realignment works. The survey was conducted in March 2021, five years following the realignment works.

The purpose of this report is to:

- Describe the current ecological conditions of the sites along Kā Pūtahi Creek;
- Compare conditions in control and realigned sites with those from the baseline, one-year and five-year post-realignment surveys; and
- Discuss any potential reasons for any significant patterns and trends recorded, and the current successes or limiting factors of the realignment works.

2.0 Survey Methods

2.1 Site Locations

The five monitoring sites were established during the baseline and 'Year One' surveys. These include the two control sites (one upstream and one downstream of the realigned reach) that were established and first surveyed in 2016 (prior to realignment, 'Baseline') and three sites within the realigned reach of Kā Pūtahi Creek (first surveyed in 2017, one year after the realignment works). A 50 m section was surveyed at each of the five sites. The upstream and downstream extent of each survey site is provided in Table 1, and locations of each site are shown in Figure 1.

Table 1. Site name, number, and co-ordinates of each of the sites surveyed in this study.

Site number	Location	Treatment	Upstream extent		Downstream extent	
			Easting	Northing	Easting	Northing
C1	Upstream of the realignment	Control	1570704	5187873	1570735	5187915
C2	Downstream of the realignment	Control	1570954	5188546	1570998	5188513
R1	Upstream end of realignment	Realigned	1570823	5188080	1570849	5188091
R2	Middle of realignment	Realigned	1570919	5188249	1570938	5188275
R3	Downstream end of realignment	Realigned	1570981	5188333	1571002	5188377

2.2 Water quality

Spot measures of water quality parameters were collected at each site using a Pro-DSS handheld water quality meter. Parameters measured were water temperature, pH, specific conductivity (µS / cm) and dissolved oxygen (mg / L and % saturation).

2.3 Riparian and in-stream habitat

Riparian and in-stream habitat was evaluated using the same methodologies established by the baseline survey (Boffa Miskell, 2016) and following standard protocols of Harding et al. (2009) and Clapcott et al. (2011):

- Protocol 3 (P3) Quantitative protocol of Harding et al. (2009)[1]:
  - P3b: Hydrology and morphology procedure;
  - P3c: In-stream habitat procedure; and
  - P3d: Riparian procedure.
- Sediment Assessment Methods of Clapcott et al. (2011):
  - Sediment Assessment Method 2 (SAM2) – in-stream visual estimate of % sediment cover; and
  - Sediment Assessment Method 6 (SAM6) – sediment depth.

In summary, these habitat assessment methods involved measuring a range of in-stream and riparian physical habitat conditions at various distances across six equally spaced transects established across the waterway every 10 m. The first (downstream most) and last (upstream most) transects were located at the co-ordinates provided in Table 1.

Parameters measured included:

- P3b: wetted width, water depth, sediment depth, water velocity

[1] Protocol 3 of Harding et al. (2009) specifies that two cross-sections should be located in each of riffle, run, and pool habitat. However, Kā Pūtahi Creek is dominated by slow-flowing run habitat, with riffles and pools being largely absent, so all six cross-sections at each site were established within run habitat.

- P3c: substrate size, substrate embeddedness, substrate compactness, extent of depositional or scouring zones, total cover of macrophytes, algae, leaf packs and large wood, bank cover
- P3d: buffer width and floodplain width, riparian vegetation type, height and cover, access to the stream, extent of bank erosion, and stream shading
- SAM2 & SAM 6: sediment cover and sediment depth.

Full details of Harding et al. (2009) P3 and Clapcott et al. (2011) SAM2 and SAM6, including field-sheet templates, are provided in Appendix 1.

Photographs of the upstream and downstream views of each site were also taken.

Habitat assessments took place from 4-8 March 2021.

2.4 Macroinvertebrate community

Macroinvertebrates (e.g., insects, snails and worms that live on the stream bed) can be extremely abundant in streams and are an important part of aquatic food webs and stream functioning. Macroinvertebrates vary widely in their tolerances to both physical and chemical conditions, and are used regularly in biomonitoring, providing a long-term picture of the health of a waterway.

The macroinvertebrate community was assessed at each site within the same 50 m reach where in-stream habitat was surveyed, from 4-8 March 2021.

Five replicate Surber samples (0.05 m^2, 500-µm mesh) were collected from each of the 4 sites following Protocol C3 of Stark et al. (2001). Surber samples were randomly collected from the most appropriate habitat available[2] at each site and disturbed to an approximate depth of 5 cm.

Macroinvertebrate samples were preserved in 70% ethanol prior to sending to Boffa Miskell's taxonomy laboratory for identification and counting in accordance with protocol P3 of Stark et al (2001). Macroinvertebrates were identified to species level, where possible, and thereafter to MCI level.

[2] Protocol C3 of Stark et al. (2001) recommends the use of a Surber sampler for quantitative sampling. However, the use of the Surber sampler can be ineffective in deep, low velocity areas as this sampling method relies on flow to wash organisms dislodged from the substrate into the net.

File Ref: BM200928_002_A4L_Monitoring.mxd

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0 100 m
1:5,000 @ A4

Data Sources:
Cadastre sourced from LINZ data service
Waterways sourced from Christchurch City Council
Projection: NZGD 2000 New Zealand Transverse Mercator

LEGEND
- Control sites
- Realignment sites
- Existing and realigned channel
- Old Channel

KĀ PŪTAHI CREEK AQUATIC ECOLOGY
2021 Monitoring Sites

Date: 14 June 2021 | Revision: 0

Plan prepared for Christchurch City Council by Boffa Miskell Limited

Project Manager: **tanya.blakely@boffamiskell.co.nz** | *Drawn:* AGa | *Checked:* TBl

Figure 1

2.5 Fish community

Each site was revisited between 9 and 11 March 2021 to assess the fish community.

To allow for comparisons between sites and among years, the fish community was assessed at all sites using a combination of fyke nets and Gee minnow traps. At each site, two fyke nets (baited with tinned cat food) and five Gee minnow traps (baited with marmite) were set within each of the 50 m survey reaches late in the afternoon and left overnight. The following morning, all fish captured were identified and measured to the nearest 5 mm before being returned alive to the stream.

Of the six sites surveyed, electric fishing was only suitable at Site C1, due to high sediment cover, low water velocity and high macrophyte cover at all other sites. Electric fishing data was not included in any analyses, however, findings from electric fishing at C1 are discussed.

Assessments of the fish community were conducted in accordance with Boffa Miskell's research and collection permit from the Department of Conservation (pursuant to section 26ZR of the Conservation Act 1987) and a Special Permit from the Ministry for Primary Industry (pursuant to section 97(1) of the Fisheries Act 1996.

2.6 Data analysis

Water quality

Water quality parameters were compared between treatments and across survey occasions, and also against the Freshwater Outcomes indicator values as set out in the Canterbury Land and Water Regional Plan 2015 (LWRP) and the CCC's Comprehensive Stormwater Network Discharge Consent (CSNDC, CRC214226).

Riparian and in-stream habitat assessments

The multiple measures across transects for the various riparian and in-stream habitat variables recorded at each site were averaged to give an average value for each parameter per cross-section. Cross-sections within a site were used as replicates in statistical analyses.

Two-way analyses of variance (ANOVAs) were used to test for differences in mean habitat conditions among sites. Response variables were *log* (x+1) transformed where necessary to meet assumptions of normality and homogeneity of variances. ANOVAs were performed in R version 3.5.1 (The R Foundation for Statistical Computing 2018).

Where appropriate, values were also compared with the Freshwater Outcomes indicator values as set out in the LWRP.

Macroinvertebrate community

The following macroinvertebrate metrics and indices were calculated to provide an indication of stream health:

- **Macroinvertebrate abundance** – the average number of individuals collected in the five replicate Surber samples collected at each site. Comparisons of abundance of macroinvertebrates among sites can be useful as abundance tends to increase in the presence of organic enrichment, particularly for pollution-tolerant taxa.

- **Taxonomic richness** – the average number of macroinvertebrate taxa recorded from the five Surber samples collected at each site. Streams supporting high numbers of taxa generally indicate healthy communities, however, the pollution sensitivity / tolerance of each taxon needs to also be considered.
- **EPT taxonomic richness** – the average number of Ephemeroptera (mayflies), Plecoptera (stoneflies) and Trichoptera (caddisflies) recorded from the five Surber samples collected at each site. These three insect orders (EPT) are generally sensitive to pollution and habitat degradation and therefore the numbers of these insects provide a useful indicator of degradation. High EPT richness suggests high water quality, while low richness indicates low water or habitat quality.
- **EPT taxonomic richness (excl. hydroptilids)** – the average number of EPT taxa excluding caddisflies belonging to the family Hydroptilidae, which are generally more tolerant of degraded conditions than other EPT taxa.
- **%EPT richness** – the percentage of macroinvertebrates that belong to the pollution-sensitive EPT orders found in the five Surber samples collected at each site, i.e. relative to total richness of all macroinvertebrates at each site. High %EPT richness suggests high water quality.
- **%EPT (excl. hydroptilids)** – the percentage of EPT taxa at each site, excluding the more pollution-tolerant hydroptilid caddisflies.
- **Macroinvertebrate Community Index (MCI-hb)**[3] – this index is based on the tolerance scores of Stark and Maxted (2007) for individual macroinvertebrate taxa found in the five Surber samples collected at each site. These tolerance scores, which indicate a taxon's sensitivity to in-stream environmental conditions, are summed for the taxa present at a site, and multiplied by 20 to give MCI-hb values ranging from 0 – 200.
- **Quantitative Macroinvertebrate Community Index (QMCI-hb)[3]** – this is a variant of the MCI-hb, which instead uses abundance data of the five replicate Surber samples. The QMCI-hb provides information about the dominance of pollution-sensitive species at a site.

Table 2 provides a summary of how MCI-hb and QMCI-hb scores were used to evaluate stream health.

Table 2. *Interpretation of MCI-hb and QMCI-hb scores for soft- bottomed streams (Stark & Maxted 2007).*

Stream health	Water quality descriptions	MCI	QMCI
Excellent	Clean water	>119	>5.99
Good	Doubtful quality or possible mild enrichment	100-119	5.00-5.90
Fair	Probable moderate enrichment	80-99	4.00-4.99
Poor	Probable severe enrichment	<80	<4.00

Note, the MCI and QMCI were developed primarily to assess the health of streams impacted by agricultural activities and should be interpreted with caution in relation to urban systems.

[3] The hard-bottom versions of the MCI and QMCI were used for Kā Pūtahi Creek as, although the bed of the waterway is now generally dominated by soft, fine sediments, it would once have been a coarse-substrate dominated / hard-bottom system. When using the MCI and QMCI for assessing ecosystem health, it's important to use the version (hard bottom versus soft bottom) most appropriate to the study system prior to human modification.

Two-way ANOVAs, with treatment and year as factors, were used to test for differences in averages: (1) between treatments (control and realigned sites); (2) among years (2016 – baseline, 2017 – 1-year post-realignment; and 2021 – 5-years' post-realignment); and (3) the interaction between treatment and year:

- macroinvertebrate abundance;
- taxonomic richness;
- EPT richness;
- EPT-except Hydroptilidae richness;
- MCI; and
- QMCI values.

Response variables were *ln (x+1)* transformed to meet assumptions of normality and homogeneity of variances. ANOVAs were performed in R version 3.5.1 (The R Foundation for Statistical Computing 2018).

A non-metric multidimensional scaling (or NMDS) ordination[4], with 999 random permutations, using abundance data (averages from Surber samples) was used to determine if the macroinvertebrate community found was similar among the 5 sites surveyed, between control and rehabilitation sites, and through time (i.e., baseline, one-year, and five-years post-rehabilitation).

NMDS ordinations rank sites such that distance in ordination space represents community dissimilarity (in this case using the Bray-Curtis metric). Therefore, an ordination score (an *x* and a *y* value) for the entire macroinvertebrate community found at any site can be presented on an *x-y* scatterplot to graphically show how similar (or dissimilar) the community at a site is from that found at another site. Ordination scores that are closest together are more similar in macroinvertebrate community composition, than those further apart (Quinn and Keough 2002).

An analysis of similarities (ANOSIM), with 100 permutations, was then used to test for significant differences in macroinvertebrate community composition: between control and realigned sites; and among baseline (Boffa Miskell, 2015); one-year post-rehabilitation (EOS Ecology, 2017); five-years post-rehabilitation (this survey).

It is helpful to view ANOSIM results when interpreting an NMDS ordination. An NMDS ordination may show that communities appear to be quite distinct (i.e., when shown graphically, sites could be quite distinct from one another in ordination space), but ANOSIM results show whether these differences are in fact statistically significantly different[5].

If ANOSIM revealed significant differences in macroinvertebrate community composition (i.e., $R \neq 0$ and $P \leq 0.05$) between treatments (control and realigned sites), or among years (baseline,

[4] Goodness-of-fit of the NMDS ordination was assessed by the magnitude of the associated 'stress' value. A stress value of 0 indicates perfect fit (i.e., the configuration of points on the ordination diagram is a good representation of actual community dissimilarities). It is acceptable to have a stress value of up to 0.2, indicating an ordination with a stress value of <0.2 corresponds to a good ordination with no real prospect of misleading interpretation (Quinn & Keough 2002).

[5] ANOSIM is a non-parametric permutation procedure applied to the rank similarity matrix underlying the NMDS ordination and compares the degree of separation among and within groups (i.e., treatment or years) using the test statistic, R. When R equals 0 there is no distinguishable difference in community composition, whereas an R-value of 1 indicates completely distinct communities (Quinn & Keough 2002).

one-year, and five-years post-rehabilitation), similarity percentages (SIMPER) were calculated[6] to show which macroinvertebrate taxa were driving these differences.

NMDS, ANOSIM and SIMPER analyses were performed in PRIMER version 7.0.13 (Clarke and Warwick 2001; Clarke and Gorley 2006).

Fish community

The fish capture data were expressed as 'catch per unit effort' (CPUE), to enable any future comparisons of fish community information that may use different methods or sampling effort. CPUE was calculated by dividing the number of fish captured by the total number of traps and nets deployed at a site. CPUE was, therefore, expressed as number of fish per trap per night.

ANOVAs were used to test for differences in averages: (1) between treatments (control and realigned sites); (2) among years (2016 – baseline; 2017 – 1-year post-realignment; and 2021 – 5-years post-realignment); and (3) the interaction between treatment and year of abundance and total richness of fish captured.

Response variables were *ln (x+1)* transformed to meet assumptions of normality and homogeneity of variances. ANOVAs were performed in R version 3.5.1 (The R Foundation for Statistical Computing 2018).

An NMDS ordination, with 999 random permutations, using abundance data was also used to determine if the fish community found was similar among the sites surveyed, between control and realigned sites, and through time (i.e., baseline, one-year, and five-years post-rehabilitation).

An analysis of similarities (ANOSIM), with 100 permutations, was then used to test for significant differences in fish community composition: between control and realigned sites; and among baseline (Boffa Miskell 2016); one-year post-rehabilitation (EOS Ecology 2017); and five-years post-rehabilitation (this survey, 2021).

If ANOSIM revealed significant differences in fish community composition (i.e., $R \neq 0$ and $P \leq 0.05$) between treatments (control and realigned sites), or among years (baseline, one-year, and five-years post-rehabilitation), similarity percentages (SIMPER) were calculated to show which fish species were driving these differences.

NMDS, ANOSIM and SIMPER analyses were performed in PRIMER version 7.0.13 (Clarke and Warwick 2001; Clarke and Gorley 2006).

[6] The SIMPER routine computes the percentage contribution of each macroinvertebrate taxon to the dissimilarities between all pairs of sites among groups.

3.0 Ecological Conditions

3.1 Site descriptions

3.1.1 Control sites

The control sites, C1 and C2, were established in the baseline survey, and are located upstream (C1) and downstream (C2) of the realigned reach of Kā Pūtahi Creek (Figure 1).

C1 (Photo 1) is located within an area of mature exotic trees and alongside the new CNC. Here, the creek is an average of 1.3 m wide, shallow (average of 15.4 cm) and is dominated by run and pool habitat. There is a small section of riffle (about 3 m long), which was the only riffle seen at any of the survey sites. The water velocity was faster than at other survey sites, however, was still slow at 0.02 m / s. Riparian vegetation on both banks is dominated by exotic trees and shrubs, including elder and crack willow. The riparian buffer extends for approximately 60 m on the true left (TL) bank and 8 m on the true right (TR) bank, providing near-complete shading for the creek along the surveyed reach. Wandering willy provides groundcover in some places along the 50 m reach, however, the banks are typically bare earth. In some places, the banks are eroding into the waterway.

There were no macrophytes along the survey reach at C1 in the 2021 survey, compared to cover recorded in the 'Year One' survey (10.8%) and a very small amount of cover in the baseline survey (0.6%). No algae was recorded along the reach, nor any log jams or large boulders. There was less woody debris and leaf packs (average of 1.2 cm and 2.8 cm respectively, across an average width of 1.34 m) present in 2021 than observed in the 'Year One' survey (22.5 cm woody debris and 5 cm leaf pack across an average width of 1.8 m).

The stream bed substrates were dominated by silt and clay, with occasional patches of cobbles. The substrates had an average compactness score of 2.3, where although dominated by finer substrates, the riverbed substrates were mostly loose, with little compaction. Substrate was moderately embedded with an average score of 2.7. Fine sediment cover was slightly lower in the 2021 survey than previously, with 83.5% average cover, compared to 95% and 93% recorded in the baseline and 'Year One' surveys respectively. Soft sediment was 10.75 cm deep on average in this survey.

Photo 1. Site C1 view looking upstream (left) and downstream (right).

Site C2 (Photo 2) is located downstream of the realigned reach of the Kā Pūtahi Creek and runs through farm paddocks. Here, the waterway is entirely run habitat that is on average 2.9 m wide and 22.1 cm deep. On the day of surveying, water flowed slowly with an average velocity of 0.02 m/s.

It is important to note that measuring velocity at this site was difficult due to a high cover of macrophytes and debris restricting the available clear water to measure velocity.

Both TL and TR banks were vegetated with rank grass and intermittent mature exotic trees (poplars and willows) along the wetted edge. These trees provide shading to the stream along most of the reach. The root systems from these trees have created significant areas of stable undercuts, with 31 m (of the total 50 m of stream length surveyed) of undercut habitat present on the TR, and 14 m of undercut on the TL. The riparian buffer was 0.5-1.0 m wide on both banks. The waterway is fenced on both sides, however, stock may be able to access the waterway on the true left as there were areas where the fence had fallen over.

Macrophyte and algae cover was higher than that seen in previous surveys, with filamentous algae covering approximately 20% of the bed (determined from an average measure across the transects). Woody debris was prevalent along the reach and covered more of the reach than observed in the baseline and 'Year One' surveys.

The substrate at C2 was almost entirely silt / sand and had a compactness score of 4 indicating very compact substrate. Soft sediment cover and depth measurements reflected this, with an average sediment depth of 36.5 cm and cover of 100%. Soft sediment at C2 was the deepest of any site and was similar to that seen in previous surveys.

Photo 2. Site C2 view looking upstream (left) and downstream (right).

3.1.2 Realigned reach sites

The realigned section of the Kā Pūtahi Creek is approximately 350 m long. Three monitoring sites were established in the 'Year One' survey (EOS Ecology, 2017) in the upstream, middle and downstream sections of the realigned stream. This is the second survey undertaken at the three sites, five years following the realignment of the waterway.

Site R1 (Photo 3) is the upstream-most of sites along the realigned section of Kā Pūtahi Creek.

The waterway at R1 was an average of 4.4 m wide and 21.4 cm deep. Dense macrophytes and very slow water flow in areas clear of macrophytes prevented water velocity from being accurately measured. The habitat is approximately half run and half pool habitat.

The planted riparian vegetation at R1 has become well established and sedges (*Carex* species) dominate the vegetation community along the wetted edge. Other species along the waters' edge include flaxes and toetoe. These sedges at the downstream end of survey reach overhang the waterway creating in-stream habitat, however, water levels were low at the time of the survey and no vegetation overhung the water in the upstream part of the survey reach. Very little shading is present over the stream, with some overhanging sedges providing a small amount of shading immediately underneath them. Taller growing vegetation is set back from the wetted edge and provides limited shade at times to the channel. In the upstream-most half of the survey reach, macrophytes (watercress) entirely covered the width of the waterway restricting water flow (Photo 3, downstream view).

A construction phase stormwater discharge pipe from the adjacent subdivision has been installed since the 1-year post-realignment survey. While we did not observe this in our survey, Belinda Margetts (Principal Waterways Ecologist visited the site on 16 July 2021 and noted that there were no obvious effects from the discharge. The pipe discharges to ground (on the bank) rather than into the waterway directly (Photo 3).

The substrate at this site is predominantly cobbles covered by a significant amount of sediment. 100% of the reach was covered by sediment, compared to 52% at the 'Year One' survey. Sediment was an average depth of 4.6 cm, compared to 0.5 cm in the 'Year One' survey. Here, fine sediment was not overly compact over and around the cobbles and the substrate was slightly embedded, with an average score of 2.3. Sediment and cobbles were moderately compact with an average compactness score of 2.7.

Photo 3. Site R1 view looking upstream (top left) and downstream (top right); construction-phase stormwater discharge pipe (bottom).

Site R2 (Photo 4) is in the middle of the realigned reach of the Kā Pūtahi Creek. The survey reach largely follows a bend which is relatively narrow at the downstream end, widening at the upstream end. The waterway has an average width of 3 m and average depth of 25.8 cm.

Habitat conditions at R2 are very similar to those at R1. The upper portion of the survey reach was smothered by dense macrophytes (watercress) and velocity data was not able to be collected. Downstream of the macrophytes, velocity was negligible. While the water is moving very slowly, the habitat is almost entirely pool habitat (84%), with about 8 m of run habitat present within the 50 m survey reach.

The riparian vegetation at R2 is in very good condition, with sedges, flaxes and toetoe having grown to overhang the waterway providing good habitat for aquatic fauna. While again, there is limited shading provided by taller vegetation over the channel, some shading is provided by these trees and shrubs at times. The banks are well stabilised by the riparian vegetation. Where the waterway widens in the upper part of the survey reach, watercress is prevalent, restricting water flow. Macrophytes were present, on average, across 20.6% of transects, and algae over 8.2%.

The substrate at this site is, again, small cobbles covered by fine sediment. Sediment covered the entirety of the site in a thin layer that was no deeper than 1 cm at any point measured. The substrate is slightly embedded with an embeddedness score of 2 and is moderately loose with an average compactness score of 2.2.

Photo 4. Site R2 view looking upstream (left) and downstream (right).

Site R3 (Photo 5) is located at the downstream end of the realigned section of Kā Pūtahi Creek. The waterway is very wide (average width 7.9 m), and there is little to no flow, meaning habitat is entirely pool or still habitat.

The riparian vegetation is well established, with large sedges, flaxes and toetoe alongside the wetted edge on both banks. These sedges provide overhanging vegetation, creating habitat for aquatic fauna. Taller growing vegetation, such as tī kōuka / cabbage trees and *Pittosporum* species, is set behind the sedges, primarily on the TR bank. These taller species have grown well to stabilise banks and provide some shading to the waterway at times. However, due to the very wide nature of the channel, shading to the waterway from these trees will remain limited.

Most of the survey reach was covered by common duckweed (*Lemna disperma*), with dense curly-leaved pondweed (*Potamogeton crispus*) throughout the water column beneath.

The substrate at R3 is dominated by small cobbles. Sediment covers the entire site in a thin layer over cobbles, with an average depth of 1 cm. There was no fine sediment cover observed in the 'Year One' survey. The substrate is not embedded, with an average embeddedness score of 1.3. The substrate is mostly loose and easily moved, with an average compactness score of 1.8.

Photo 5. Site R3 view looking upstream (left) and downstream (right).

3.2 General habitat conditions

3.2.1 Water quality

Table 3 provides a summary of the water quality parameters measured at the five sites in March 2021.

Table 3. Water quality parameters measured in Kā Pūtahi Creek in 2021, five years after realignment works.

Site number	Date	Time	Temperature °C	pH	Specific conductivity µS / cm	DO mg/L	DO %
C1	4/03/2021	10:36	14.9	7.17	125.4	6.47	63.9
C2	5/03/2021	09:02	13.8	7.18	199.7	2.28	22.0
R1	4/03/2021	13:51	17.7	7.26	125.1	6.97	73.2
R2	5/03/2021	13:24	15.3	7.41	127.9	7.93	79.1
R3	5/03/2021	11:15	14.1	7.45	155.3	5.81	56.5

Water temperature varied across sites (Table 3) and were relatively similar to those recorded in previous surveys. Temperature can fluctuate both daily and seasonally, so it is important to note that water temperature was only measured once at each site on each sampling occasion.

pH was circum-neutral at all sites and is likely to be within a tolerable range for aquatic fauna. pH levels at all sites were also within the LWRP guidelines of between 6.5 and 8.5. There were no discernible differences in pH in any year between control and realigned sites.

Specific conductivity was variable across sites in this survey (2021). Conductivity was variable through time, with levels recorded in this survey being lower than that measured in the 'Year

One' survey (2017). Conductivity was only measured once at each site in each survey and can fluctuate both daily and seasonally, which is important to consider when comparing this data.

Dissolved oxygen levels were relatively low at all sites, with very low dissolved oxygen (22%) at C2 and the remaining sites ranging from 56% to 79% dissolved oxygen saturation and 2.28 to 7.93 mg / L. The LWRP sets a guideline of 70% as the minimum acceptable value for "spring-fed – plains waterways" and "spring-fed – plains – urban waterways", for which only two sites were better than this minimum threshold (R1 and R2).

It is important to note that DO was also only measured once at each site. DO can fluctuate both diurnally and seasonally and is also impacted by other factors such as cover of macrophytes. However, high levels of sediment and large proportions of detritus could also have a role, and the low DO concentrations at sites measured could be a real result. If this is the case, these low DO levels may be adversely affecting in-river fauna.

3.2.2 Riparian and in-stream habitat

Wetted width

The average wetted width at realigned sites in the 2021 survey was 509.2 cm, compared to 213.6 cm at control sites (Figure 2). Wetted width was relatively similar to the previous (2017) survey, and realigned sites were similar in average wetted width to that of the sites within the oxbow (baseline survey). Wetted width at control sites in the baseline study were greater than measured after the realignment, in 2017 and 2021. This was due to greater wetted width measured at Site C2. A similar trend is seen for water depth, as discussed below.

When all data from the three surveys was analysed, there was no significant difference in wetted width between treatments (i.e., control vs realigned sites) ($F_{1,9}$ = 0.615, P = 0.187), nor between surveys ($F_{2,9}$ = 0.964, P = 0.255), due to the high variability in wetted width of the waterway.

There was no significant interaction between treatment and year for the wetted width (Treatment: Year interaction: $F_{1,9}$ = 0.021, P = 0.799).

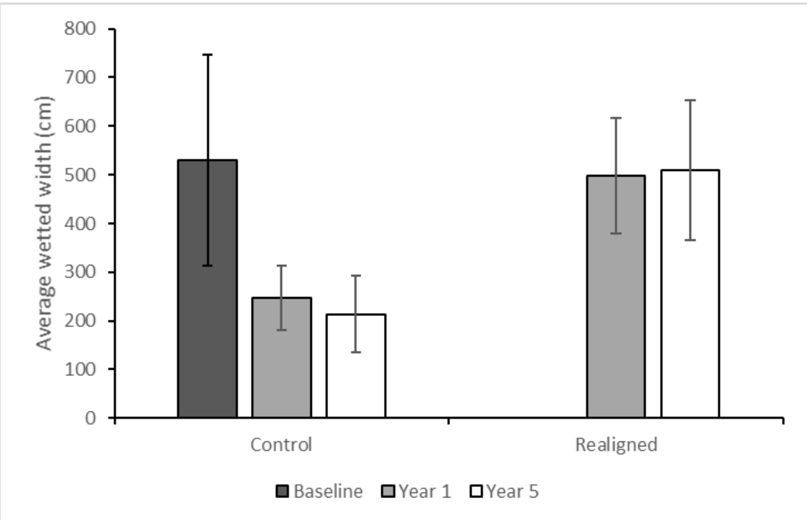


Figure 2. Average (±1SE) wetted width (cm) measured at the control and realigned sites for the baseline study (2016, black bars), one-year post-rehabilitation (2017, grey bars), and five-years post-rehabilitation (this survey – 2021, white bars).

Water depth

The average water depth at realigned sites in this survey was 26.3 cm, compared to 20.5 cm at control sites (Figure 3). It's important to note that water depth was significantly greater at Site C2 in the baseline study, compared to that measured in 2017 and 2021. Following realignment water depth is relatively similar between realigned and control reaches; when analysing all data from the three surveys, there was no significant difference in water depth between treatments ($F_{1,9}$ = 0.003, P = 0.868), nor between surveys ($F_{2,9}$ = 0.618, P = 0.115). There was no interaction between treatment and year for water depth (Treatment: Year interaction: $F_{1,9}$ = 0.024, P = 0.656).

It's unclear why water depth at Site C2 was so much greater in the baseline study than in 2017 and 2021. Wetted width was also greater at Site C2 in the baseline study, so it's likely there was a channel blockage downstream affected wetted width and water depth at C2. This may have been removed prior between the baseline and Year 1 survey, or more likely channel capacity was affected by the oxbow resulting in a wide wetland-like system at C2 prior to livening of the realignment (refer to Figure 1 for location of sites).

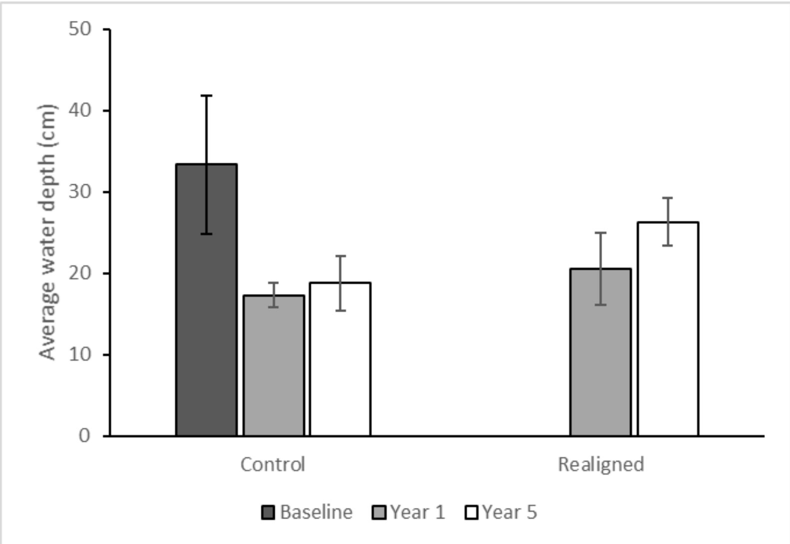


Figure 3. Average (±1SE) water depth (cm) measured at the control and realigned sites for the baseline study (2016, black bars), one-year post-rehabilitation (2017, grey bars), and five-years post-rehabilitation (this survey – 2021, white bars).

Macrophyte cover

There was higher cover of macrophytes at the realigned sites, than control sites in 2021, with an average of 60.7% cover in realigned sites and just 3.2% cover at control sites. This difference was significant ($F_{1,9}$ = 72.02, P < 0.05). While macrophyte cover at realigned sites in 2021 appeared much higher than in other surveys, there was no significant difference in macrophyte cover between surveys ($F_{2,9}$ = 0.618, P = 0.115).

There was no significant interaction between year sampled and treatment (Treatment: Year interaction: $F_{1,9}$ = 36.82, P = 0.052) (Figure 4).

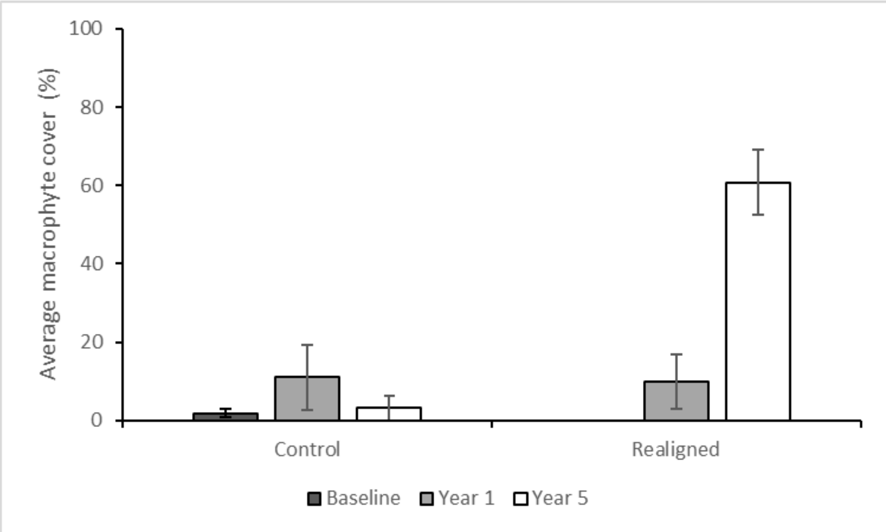


Figure 4. Average (±1SE) macrophyte cover (%) measured at the control and realigned sites for the baseline study (2016, black bars), one-year post-rehabilitation (2017, grey bars), and five-years post-rehabilitation (this survey – 2021, white bars).

Embeddedness & compactness

There was a significant difference in embeddedness between treatments ($F_{1,9}$ = 15.51, P < 0.001), where embeddedness was higher at control sites than at realigned sites. There was no significant difference in embeddedness between surveys ($F_{2,9}$ = 0.52, P = 0.344). This is reflected by the high sediment cover and absence of cobble materials at control sites.

There was no interaction between year and treatment when embeddedness data was analysed (Treatment: Year interaction: $F_{1,9}$ = 0.504, P = 0.161).

There was a significant difference in compactness between treatments ($F_{1,9}$ = 1.216, P < 0.001), where compactness was higher at control sites than at realigned sites (Figure 5). There was no significant difference in average compactness between surveys ($F_{2,9}$ = 0.113, P = 0.06).

There was also an interaction between year sampled and treatment for compactness, where realigned sites in both 2017 and 2021 had a lower average compactness than that observed at control sites, with the exception of control and realigned sites in 2021, which were not significantly different (Treatment: Year interaction: $F_{1,9}$ = 0.208, P < 0.01) (Figure 5).

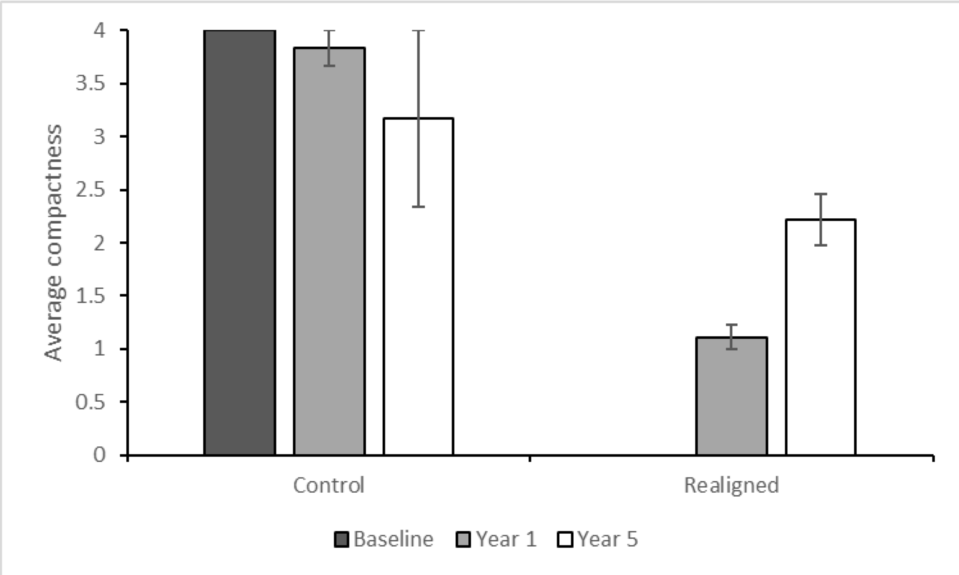


Figure 5. Average (±1SE) compactness measured at the control and realigned sites for the baseline study (2016, black bars), one-year post-rehabilitation (2017, grey bars), and five-years post-rehabilitation (this survey – 2021, white bars).

Soft sediment cover

Soft sediment cover at realigned sites was on average 100% of each reach, with control sites averaging 91.8% cover. There was no significant difference in sediment cover between treatments ($F_{1,9}$ = 0.06, P = 0.757), nor between surveys ($F_{2,9}$ = 3.24, P = 0.124). Sediment cover at realigned sites has increased from 17.3% average cover at the time of the 'Year One' survey to 100% cover in 2021.

There was no interaction between year sampled and treatment for sediment cover, where realigned sites in 2017 had much lower average compactness than that observed at any other time (Treatment: Year interaction: $F_{1,9}$ = 2.32, P = 0.08 (Figure 6).

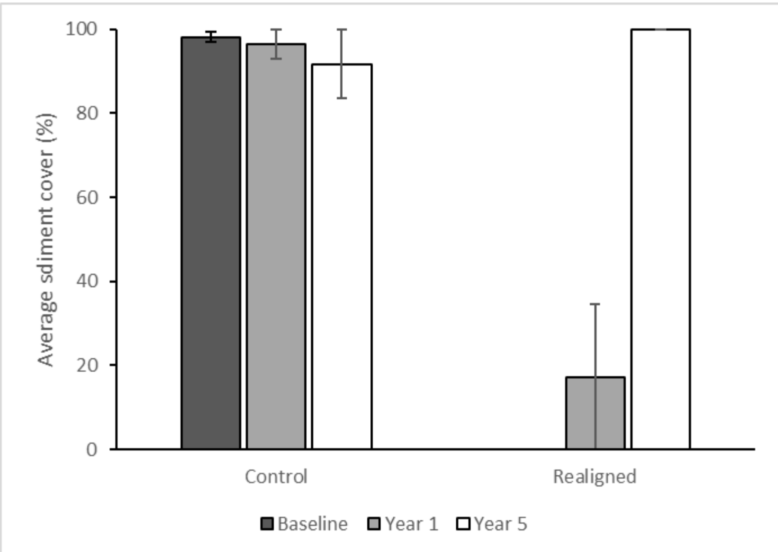


Figure 6. Average (±1SE) sediment cover (%) measured at the control and realigned sites for the baseline study (2016, black bars), one-year post-rehabilitation (2017, grey bars), and five-years post-rehabilitation (this survey – 2021, white bars).

Sediment depth

At realigned sites in 2021, average sediment depth was 2.2 cm, compared to an average of 23.6 cm at control sites. There was a significant difference in sediment depth between treatments ($F_{1,9} = 23.62$, $P < 0.001$), where sediment was significantly deeper at control sites than at realigned sites (Figure 7). There was no significant difference in average sediment depth between surveys ($F_{2,9} = 1.57$, $P = 0.355$), nor any interaction between year and treatment (Treatment: Year interaction: $F_{1,9} = 0.281$, $P = 0.534$).

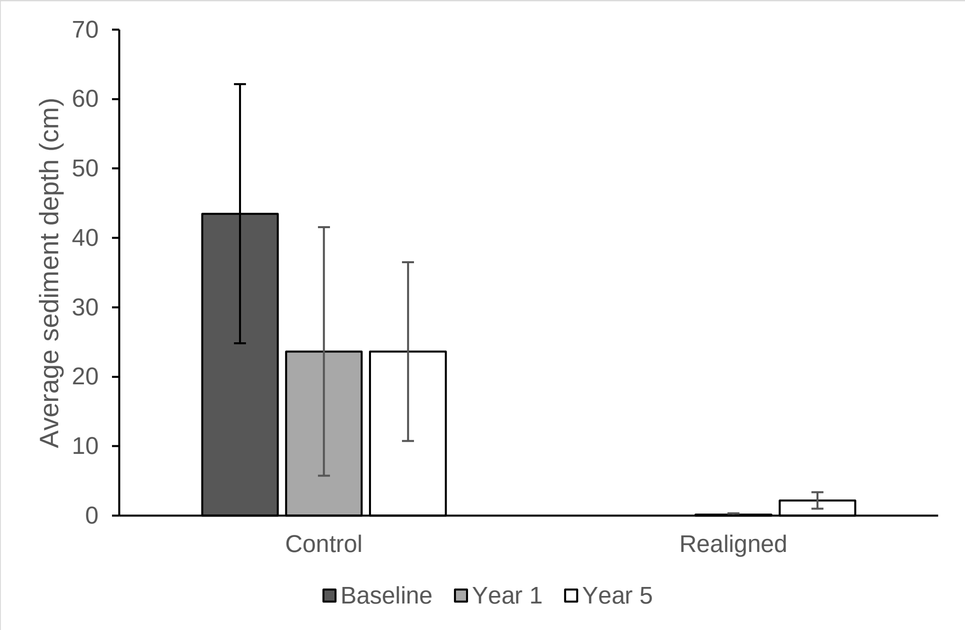


Figure 7. Average (±1SE) sediment depth (cm) measured at the control and realigned sites for the baseline study (2016, black bars), one-year post-rehabilitation (2017, grey bars), and five-years post-rehabilitation (this survey – 2021, white bars).

Algae

Average algae cover was 8.2% at realigned sites and 27.9% at control sites in 2021. This is compared to 64.5% cover at realigned sites and 1.6% cover at control sites in 2017. There was, however, no significant difference in algae cover between treatments ($F_{1,9} = 22.15$, $P = 0.124$), nor between surveys ($F_{2,9} = 12.23$, $P = 0.481$) (Figure 8). There was no interaction between treatment and year (Treatment: Year interaction: $F_{1,9} = 36.82$, $P = 0.056$).

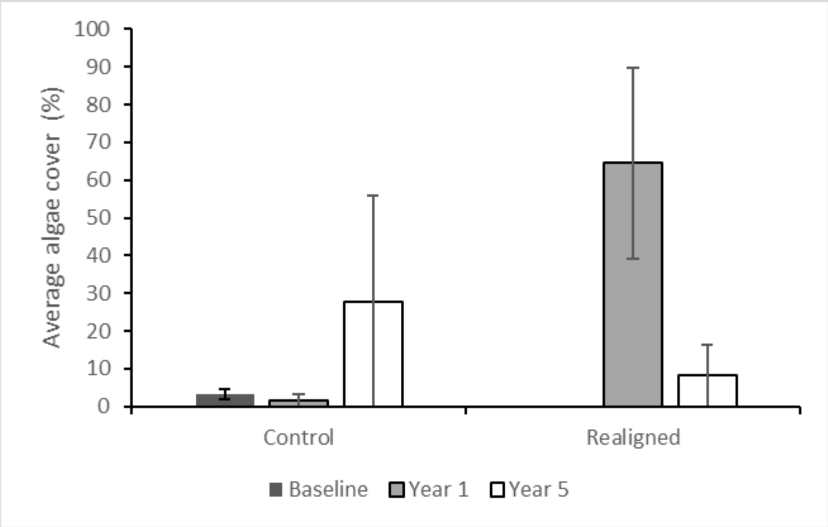


Figure 8. Average (±1SE) algal cover (%) measured at the control and realigned sites for the baseline study (2016, black bars), one-year post-rehabilitation (2017, grey bars), and five-years post-rehabilitation (this survey – 2021, white bars).

Vegetation and canopy cover

Vegetation cover was assessed from the wetted edge of the waterway for 20 m at each transect. Vegetation cover at realigned sites has increased substantially since the 'Year One' survey. Low cover vegetation (0-0.3 m) has increased from 21.7% (± 11.5 S.E.) in 2017 to 70.1% (± 10.8 S.E.) in 2021. Small shrubs and grasses (0.3-1.9 m of height) covered 7.9% (± 5.8 S.E.) in the 'Year One' survey, with an increase to an average of 68.7% (± 9.2 S.E.) cover in this survey. In this survey, vegetation cover 2.0-4.9 m high was 10.6% (± 5.4 S.E.) on average, and vegetation 5-12 m high was present on average in 2.2% (± 0.7 S.E.) of the area surveyed. No vegetation had grown over 2 m at the time at the time of the 'Year One' survey.

Canopy cover was measured at 20 points at each site. Average canopy cover at realigned sites in the 'Year Five' survey was 2.9%, compared to 88.1% at control sites. In 2017, average canopy cover at realigned sites was 5.4%. This may be due to slight variation in transect location rather than a decline in canopy cover.

Velocity

While water was visibly moving in places, velocity in the realigned reach of Kā Pūtahi Creek was negligible, and unable to be measured in 2021 in most places. High macrophyte cover further contributed to this.

Velocity was very slow at control sites, with an average of 0.02 m / s (±0.002 SE). Measuring velocity was also difficult at control sites due to high cover of woody debris and filamentous algae.

3.3 Macroinvertebrate community

3.3.1 Overview in 2021

A total of 46,318 individuals belonging to 49 taxonomic groups were collected in all Surber samples across the five sites in 2021.

The most diverse group was true flies (Diptera), for which there were 15 different taxa, followed by caddisflies (Trichoptera) with 9 taxa, crustaceans with five taxa, molluscs with four taxa and Odonata and true bugs (Hemiptera) each with three taxa. Aquatic worms (Annelida) were represented by two taxa, and the remaining macroinvertebrate groups were represented by a single taxon (mites, Acarina; spiders, Arachnids; Hydra, Cnidaria; beetles, Coleoptera; mayflies, Ephemeroptera; roundworms, Nematoda; Nemertea; and flatworms, Platyhelminthes).

While crustaceans (made up by ostracods (seed shrimp) and *Paracalliope* amphipods) were not particularly diverse, they were numerically dominant (i.e., the most abundant group). Snails (Mollusca) were the next most dominant group numerically, followed by worms, then true flies. Some taxa were present in only very low numbers, with only one or two individuals representing a taxon in some cases.

Crustaceans were also the dominant macroinvertebrate group, making up 55% of all macroinvertebrates collected from the five sites. The most abundant taxon was the freshwater seed shrimp, which made up a significant proportion of communities found at each site. Molluscs made up the next largest proportion of the community at 28%. True flies were the most diverse group of taxa collected, but only made up 3.5% of the macroinvertebrate community found.

There were several taxa found at all sites surveyed, including the highly abundant seed shrimps, the snails *Potamopyrgus* sp. and *Physa* sp., and aquatic worms (Oligochaetea).

3.3.2 Total abundance

Macroinvertebrate abundance varied significantly among sites, with between 2,138 and 20,812 individuals collected in each Surber sample.

Average macroinvertebrate abundance, as determined from Surber samples, differed slightly (but not significantly) between treatments (i.e., between control and realigned sites) ($F_{1,9} = 1.92$, $P = 0.054$), where macroinvertebrate abundance was higher at realigned sites than at control sites. However, no significant differences in abundance were detected over time ($F_{2,9} = 1.03$, $P = 0.31$), nor any interaction between treatment (i.e., control versus realigned) and time sampled (Treatment: Year interaction: $F_{1,9} = 0.63$, $P = 0.24$).

3.3.3 Taxonomic richness

Taxonomic richness was somewhat variable across sites in this survey, ranging on average from 11 to 19 taxa per site.

The average number of macroinvertebrate taxa did not differ between control and realignment sites ($F_{1,9} = 24.23$, $P = 0.186$). Nor was there a significant difference in taxonomic richness among the survey years ($F_{2,9} = 2.21$, $P = 0.912$), or any interaction between treatment and year for taxonomic richness (Treatment: Year interaction: $F_{1,9} = 0.15$, $P = 0.913$).

3.3.4 EPT richness and percent composition

The EPT insect orders (Ephemeroptera, mayflies; Plecoptera, stoneflies; and Trichoptera, caddisflies) are generally sensitive to pollution and habitat degradation and are useful indicators of stream health. High EPT richness suggests good water and habitat quality, while low EPT richness suggests poorer water quality and degraded stream health. Caddisflies and mayflies have been the only EPT taxa found in Kā Pūtahi Creek in all surveys, including this one.

There was a total of 9 caddisfly taxa collected in the 2021 survey. This included *Pycnocentria* and *Polyplectropus* for each of which only one individual was collected at Control Site 1. The average number of EPT taxa collected ranged between one and eight taxa per site (average range was 0.4 to 3.0 taxa per site). Caddisfly diversity was lowest at Realignment Site 2 which had only one caddisfly taxa - the pollution tolerant taxon *Oxyethira*. Realignment Site 3 had only two caddisfly taxa present, both of which belonged to the pollution-tolerant caddisfly family Hydroptilidae (*Oxyethira* and *Paroxyethira*). EPT taxa richness was highest at Control Site 1, where a total of 8 taxa were found, with an average of three taxa in each of the five Surber samples.

Of the caddisfly taxa collected, the pollution-tolerant *Oxyethira* was the most abundant caddisfly found, and was found at every site except Control 1. The other pollution-tolerant caddisfly taxa, *Paroxyethira*, was found in low numbers at Realignment Sites 1 and 3, with four and eight individuals found, respectively. No other EPT taxa were collected at Sites R2 and R3, with low numbers (2-8 individuals) of *Oecetis* and *Triplectides* found at R1, and *Triplectides* at C2.

One mayfly taxon, *Deleatidium*, was found in low numbers at Control Site 1 (3 individuals). Mayflies were not collected at any other site in 2021. *Deleatidium* was collected at Control Site 1 in and 2017 but not in 2016, and never at any other site previously.

Average EPT richness did not differ between treatments (i.e., between realignment and control sites) nor across years ($F_{1,9}$ = 1.9, P =0.19, $F_{2,9}$ = 2.21, P = 0.91).

When the pollution tolerant taxa *Oxyethira* and *Paroxythira* were excluded from the analysis, no significant difference in EPT taxa richness was found between realigned and control sites ($F_{1,9}$ = 0.96, P = 0.96), nor a significant difference in average EPT taxa among years ($F_{2,9}$ = 1.03, P = 0.24). There was also no interaction between treatment and year (Treatment: Year interaction: $F_{1,9}$ = 0.535, P = 0.216).

EPT made up a very small proportion of communities in all surveys. EPT taxa made up 0.1 - 0.9% of communities in the realigned reach and 3.1-3.3% at control sites in 2021. This is compared to a maximum of 0.7% EPT taxa in the baseline survey (2016), and between 0.3 and 2.9% of the community in the 2017 survey.

Mayflies have only ever been detected at Control Site 1 in very low numbers - one individual in 2017 and four in 2021 (which equated to 0.14% of the community at that site).

3.3.5 Macroinvertebrate Community Index

MCI and QMCI scores are a measure of stream, or ecological, health with higher scores indicating greater ecological condition. The hard-bottom versions of the MCI and QMCI were used for Kā Pūtahi Creek as, although the bed of the waterway is now generally dominated by soft, fine sediments, it would once have been a coarse-substrate dominated / hard-bottom system. When using the MCI and QMCI for assessing ecosystem health, it’s important to use the version (hard bottom versus soft bottom) most appropriate to the study system prior to human modification.

MCI

MCI scores were somewhat variable across surveys and between realignment and control sites (Figure 9). When data across all surveys was compared, there was no significant difference between realignment and control sites’ average MCI scores ($F_{1,9}$ = 158.7, P = 0.063 (Figure 9).

All sites in all years had MCI scores below 80, indicating "poor" stream health with "probable or severe enrichment" (based on the water quality categories of Stark and Maxted 2007) (Figure 8). There was no significant difference in MCI score averages over time ($F_{2,9} = 283.3$, $P = 0.56$).

There was a significant interaction between year sampled and treatment, where realigned sites in 2017 had a higher average MCI score than that observed at realigned sites in 2021 (Treatment: Year interaction: $F_{1,9} = 185.2$, $P < 0.05$) (Figure 9).

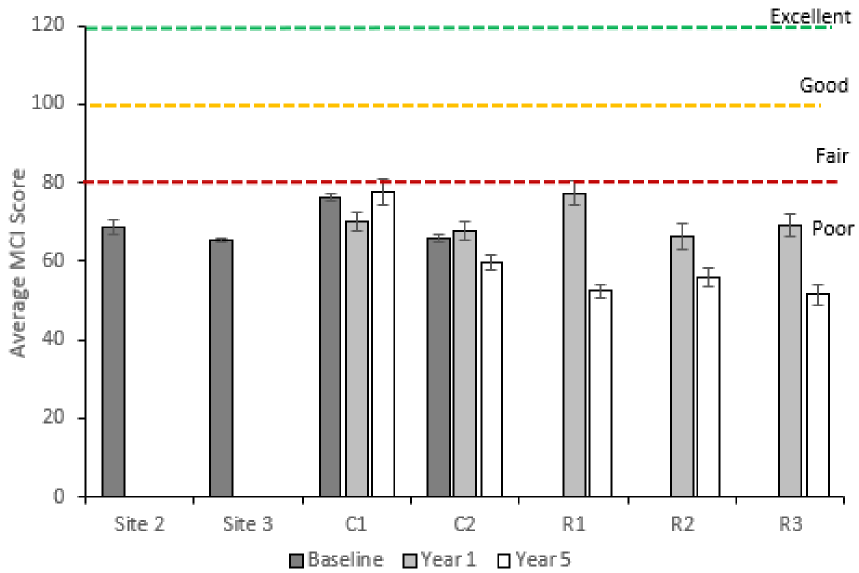


Figure 9. Average (±1SE) Macroinvertebrate Community Index (MCI) measured at the control and realigned sites for the baseline study (2016, black bars), one-year post-rehabilitation (2017, grey bars), and five-years post-rehabilitation (this survey – 2021, white bars). The dashed lines indicate the water quality categories of Stark and Maxted (2007). See Table 2 for further information.

QMCI

QMCI, the quantitative variant of the MCI, is often considered a better indicator of "health" than MCI as it also accounts for abundances of macroinvertebrate taxa, while MCI only accounts for presence and can be biased by rare but sensitive taxa or vice versa.

QMCI scores from this (2021) survey, showed stream health at all sites was, on average, "poor", with the average score ranging from 2.2 to 3.3 (Figure 11). There was a significant difference in mean QMCI between realignment and control sites across all years, where QMCI scores were higher at control sites on average than at realigned sites ($F_{1,9} = 0.62$, $P < 0.05$).

There was also a significant difference in QMCI through time ($F_{2,9} = 1.76$, $P < 0.01$), where the 'Year One' (2017) survey had a significantly higher QMCI scores than the 2021 survey (Figure 11).

There was no interaction between treatment and year for QMCI score (Treatment: Year interaction: $F_{1,9} = 0.14$, $P = 0.287$).

QMCI scores of all sites in all surveys fell below the CSNDC receiving environment attribute target levels and the LWRP freshwater outcomes, where the minimum QMCI score for "spring-fed – plains waterways is 5 (Figure 11). Moreover, QMCI scores from sites C1 and R3 in Year 1 were the only sites that met the "spring-fed – plains – urban waterways" target of 3.5 (Figure 11).

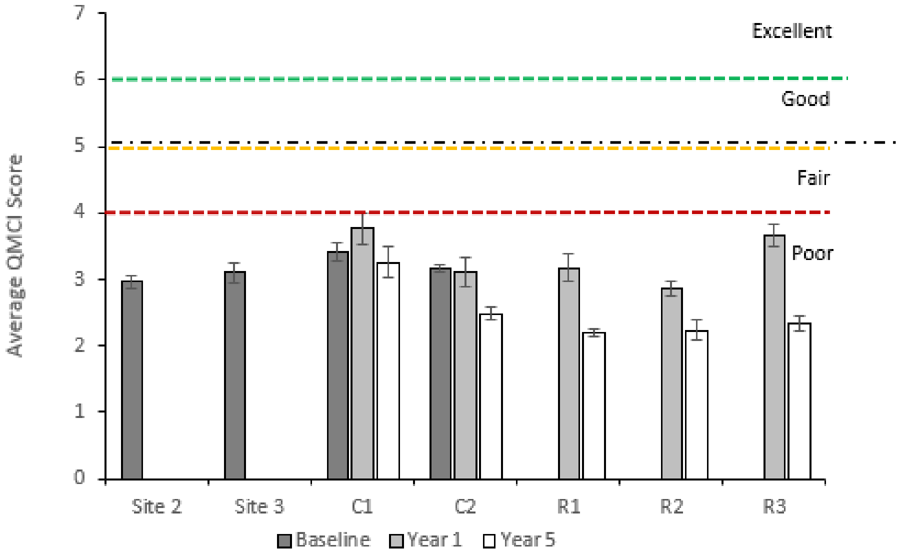


Figure 10. Average (±1SE) Macroinvertebrate Community Index (MCI) measured at the control and realigned sites for the baseline study (2016, black bars), one-year post-rehabilitation (2017, grey bars), and five-years post-rehabilitation (this survey – 2021, white bars). The dashed lines indicate the water quality categories of Stark and Maxted (2007). See Table 1 for further information. The black dash-dot lines indicate the lower limit QMCI scores of the CCC's Comprehensive Stormwater Network Discharge Consent and the LWRP guidelines for "spring-fed – plains waterways" (QMCI 5).

3.3.6 Changes in community composition

Macroinvertebrate community composition has changed somewhat over time and has been largely driven by changes in the relative dominance, rather than the absence, of taxa.

The macroinvertebrate community in this survey (2021) was dominated by crustaceans and snails in both realigned and control sites (Figure 11). In the baseline study, the proportion of crustaceans was 50%, while in 2017 the proportion of crustaceans in the community was considerably lower at 12% and 11% at control and realigned sites, respectively. Molluscs also made up a similar proportion of the community in 2021 (19% and 28% at control and realigned sites, respectively) when compared to the baseline survey (29%). The proportion of crustaceans observed in control sites, and true flies at realigned sites in 2017 were considerably higher (77% and 33%, respectively) than that seen at other treatments both previously (2016), or subsequently (2021) (Figure 11).

In general, the 2021 community composition (in terms of general taxonomic group composition) is similar to that seen in the baseline survey (2016) (Figure 11). The communities found in 2017 and 2021 include all taxa found in the baseline survey, however, a number of taxa found in the 2017 survey were not found in the 2021 survey, including the beetle *Antiporous*, the true flies Empididae and Muscidae and the caddisfly *Pycnocentrodes*. Likewise, in 2021 several taxa were collected that that have not been found in previous surveys, including the hemipteran / true bugs *Anisops*, the damselfly *Austrolestes*, the true flies *Austrosimulium*, Chironomidae, Ephydridae, *Hexatomini* and Psychodidae, and the beetle Hydrophilidae. In general, these taxa were found in relatively low numbers (i.e., no more than 9 individuals at each site).

The presence of these taxa could be due to rehabilitation activities and changes to available habitat; damselfly nymphs and the true bugs, for example, are likely responding to the slower-flowing water with overhanging vegetation. More importantly, because these taxa are "rare", they will have had little impact on the analysis of community composition or QMCI results.

Despite an apparent initial difference in the relative abundance of taxon groups in 2017 from the baseline survey, the community found in the 2021 survey is more reflective of that found in the baseline survey (Figure 11).

Caddisfly relative abundance in this survey was generally unchanged from the previous (2017 survey), with 3% and 0% for control and realigned sites, respectively. In 2017, caddisflies made up 2% and 1% of the community at control and realigned sites, respectively. Mayflies have only ever been detected at Control Site 1 in very low numbers (one individual in 2017 and four in 2021), and so have contributed little to overall community composition, or community differences between control and realigned sites.

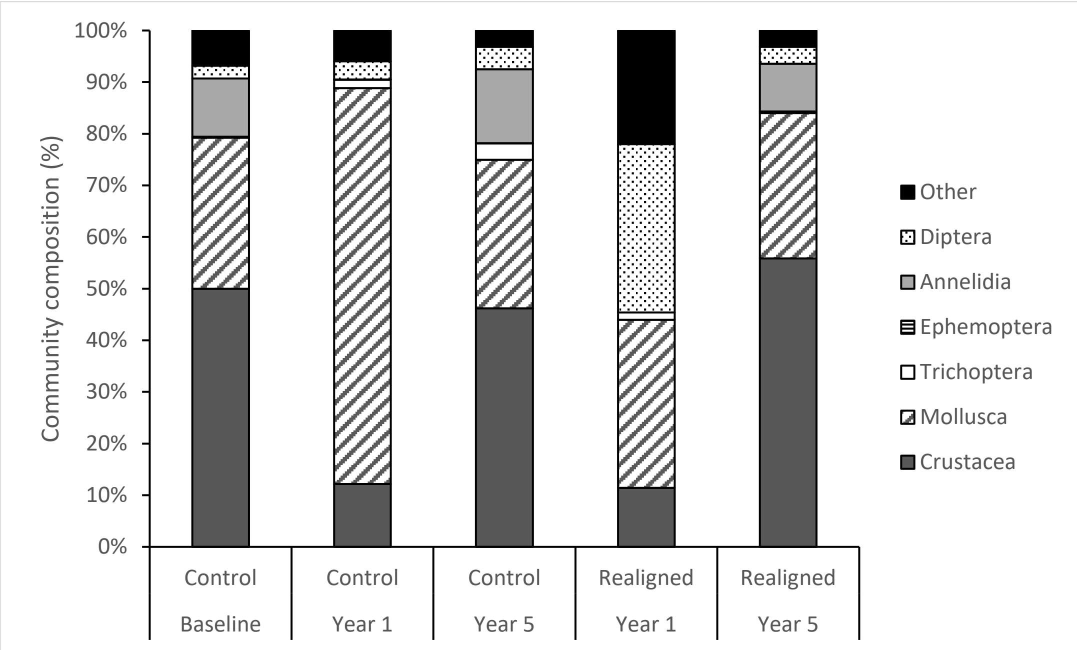


Figure 11. Average macroinvertebrate community composition (%) found at control and realigned sites for the baseline study (2016, black bars), one-year post-rehabilitation (2017, grey bars), and five-years post-rehabilitation (this survey – 2021, white bars).

The NMDS ordination, confirmed by the ANOSIM results, detected slight but significant differences in the macroinvertebrate community through time (ANOSIM R = 0.635, P = 0.04) (Figure 12, Appendix 4), which was due to weak (not significant) differences between the 2016 and 2017 macroinvertebrate communities (R = 0.679, P = 0.067). There were also weak but non-significant differences between 2017 and 2021, and 2016 and 2021 (Appendix 4).

The NMDS and ANOSIM indicate an initial shift in community composition since baseline survey, though a potential subsequent shift in community becoming more similar to that observed in the baseline survey.

SIMPER indicated that these significant differences in community composition were largely due to differences in the average number of occurrences of some taxa (i.e., greater or lesser numbers of individuals), rather than the presence or absence of a particular taxon among sampling occasions. For example, ostracods (freshwater seed shrimp) and *Potamopyrgus* snails were most abundant in 2021, where they were 22 and 8 times more abundant than in the 2017 samples, respectively. Ostracods were substantially more abundant in both the 2016 and 2021 surveys, compared to the 2017 survey. Worms were also in higher abundances in the 2021 survey compared to other surveys.

In addition, ostracods and aquatic worms were less abundant in the 2021 survey than in the baseline (2016) survey and were also strong drivers of differences in community composition.

There was also a significant difference in community composition between realigned and control sites (R = 0.625, P = 0.01) (Appendix 4). These differences were again predominantly driven by differences in abundances of some taxa, including *Potamopyrgus*, which was around four times more prevalent at control sites than realigned sites. Ostracods, non-biting midges (Orthocladiinae), water boatmen (*Sigara*), the freshwater snails *Physa* and *Gyraulus,* and aquatic worms were more prevalent at realigned sites than at control sites (these groups were 3-5 times more prevalent at realigned than control sites) (see Appendix 4 for further details on SIMPER results).

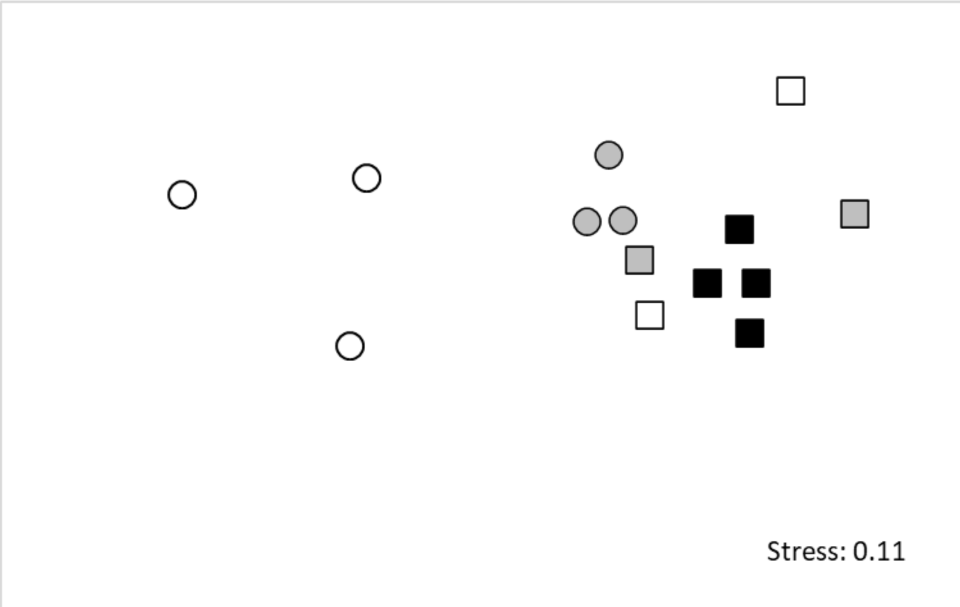


Figure 12. Non-metric multidimensional scaling (NMDS) ordination based on a Bray-Curtis matrix of dissimilarities calculated from macroinvertebrate data collected from the four sites surveyed in 2016 (baseline survey – black), and five sites surveyed in 2017 (one-year post-rehabilitation – white), and 2021 (five-years post-realignment, grey; this study). Control sites are shown as squares; realigned sites are shown as circles. Axes are identically scaled so that sites closest together are more similar in community composition, than those further apart. The significance of differences in community dissimilarity was confirmed using Analysis of Similarities (ANOSIM).

3.4 Fish community

A total of 168 individuals, belonging to five different species, were captured using traps and nets at the five sites in Kā Pūtahi Creek in March 2021. At each site, two baited fyke nets and five baited Gee-minnow traps were set.

The species caught, in descending order from most to least abundant, were: shortfin eel (*Anguilla australis*), common bully (*Gobiomorphus cotidanus*), upland bully (*G. breviceps*), inanga (*Galaxias maculatus*), longfin eel (*A. dieffenbachii*) and juvenile eels (*Anguilla* spp.). Longfin eel and inanga are classified as At Risk - Declining, while the other species captured are Not Threatened (Dunn et al. 2018). Fish of all species were of varying sizes, with both adults and juveniles of all species found suggesting recruitment is occurring.

Electric fishing was only a suitable survey method at Site C1. Twenty shortfin eels were caught during electric fishing, with an additional 21 eels (unidentified species) and 11 adult inanga seen but not caught. Fish caught using electric fishing methods were not included in any analysis to prevent biases in comparing different fishing methods[7]. Further, juvenile eels were excluded from the analysis as they were not able to be identified to species level.

3.4.1 Total abundance and species richness

Species richness was consistent across sites, with an average of five species being found at each site. R1 had the highest species richness with five species captured. R2 and C2 had the lowest species richness with three species captured.

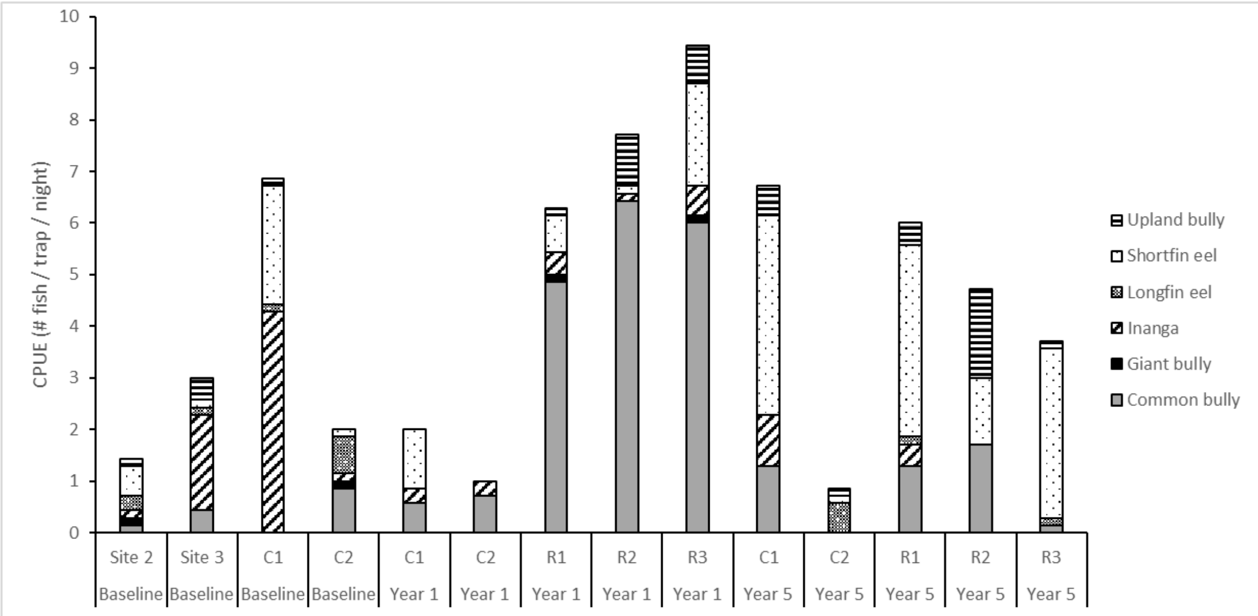
Shortfin eels and upland bullies were found at all sites, and common bullies found at all sites except C2. Shortfin eel was the most abundant species numerically, with 86 individuals found across the five sites. Inanga were found at two sites in this survey (R1 and C1), though had been found at all sites in the previous ('Year One') survey. Species found were very similar to those found in previous surveys, however, giant bullies (*G. gobioides*) were not found at any site in the 'Year Five' (2021) survey, despite having been found (in low numbers) in both the baseline and 'Year One' surveys.

Longfin eels were found at three sites in this survey, including one individual at each of sites R1 and R3. Four large individuals (length > 500 mm) were caught at C2. Longfin eels were not found in the 'Year One' survey at any site, though were in similar numbers to the 2021 survey in the baseline survey.

Total abundance was much lower at C2 than at other sites where only six fish were caught in traps. At the other four sites, between 26 and 47 fish were caught. However, when all data was considered, there was no significant difference in total abundance between years ($F_{2,9} = 0.135$, $P = 0.748$) (Figure 13). There was a significant difference in number of fish caught at control and realigned sites, with more fish found at realigned than control sites ($F_{1,9} = 1.78$, $P < 0.05$) (Figure 13).

A similar number of species was caught in control and realigned sites ($F_{1,9} = 0.381$, $P = 0.457$), however, there was a difference in species richness between years ($F_{2,9} = 8.0$, $P < 0.05$). Post-hoc Tukey testing showed a significant difference in the number of species caught between the baseline and following two surveys ($P < 0.05$), where more species were caught in the baseline survey than subsequently. There was no significant difference in number of species caught in the 'Year One' and this survey ('Year Five').

[7] Electric fishing surveys that were conducted in the 'Year One' survey were not repeated as the accumulation of sediment and slow velocity in the realigned channel and at C1 meant the method was likely to be ineffective. Data from electric fishing was not included in analysis of the fish community.



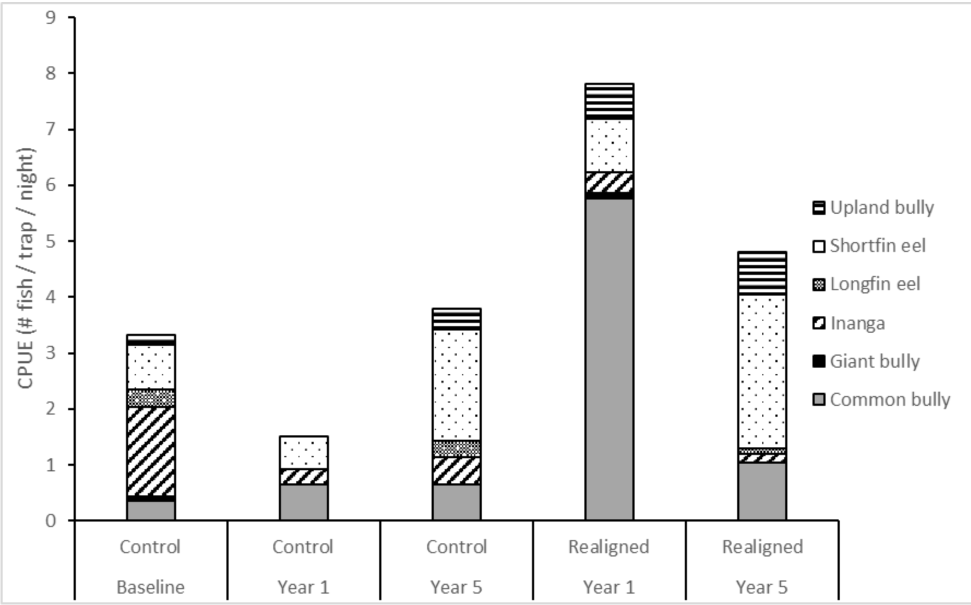


Figure 13. Total abundance of fish, separated by species, found at each site (above) and on average at control and realigned sites (below). Numbers are show as catch per unit effort (CPUE): number of fish per trap per night fished.

3.4.2 Community composition

The NMDS ordination, confirmed by the ANOSIM results, showed only very slight difference in the fish community composition between survey occasions (ANOSIM R = 0.466, P = 0.01) (Figure 15), which was due to a significant difference between the 2017 and 2021 (R = 0.714, P = 0.033) (Appendix 5). There were no detectable significant differences in fish community between 2016 and 2017, or 2016 and 2021 (Appendix 5).

SIMPER indicated that this difference in community composition between 2017 and 2021 was due to differences in the average number of occurrences of common bully, shortfin eel and inanga. Common bully and inanga were found in greater numbers in 2017, than 2021; the opposite was found for shortfin eel (Appendix 5). Of interest, longfin eels were not found in the 'Year One' survey but were found in both the baseline (2016) and this survey (2021). Longfin eel was not a significant driver of differences in community composition.

There was a significant, albeit weak, difference in community between treatments (i.e., control vs realigned sites) (ANOSIM R = 0.5, P = 0.05) (Appendix 5).

SIMPER analysis again showed this was due to differences in the average number of occurrences of common bully, shortfin eel and upland bully; these species were all found in greater numbers in realigned than control sites (Appendix 5).

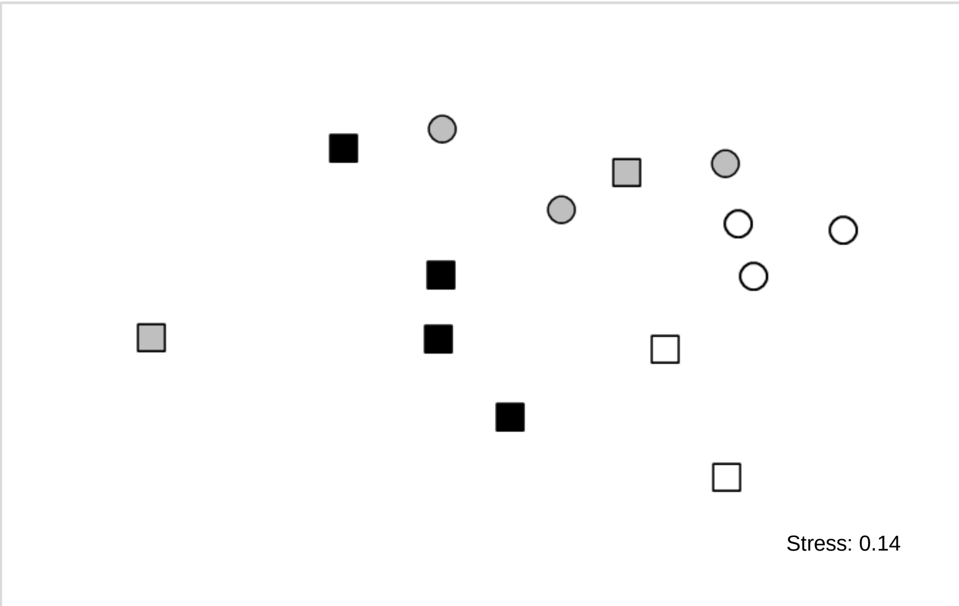


Figure 14. Non-metric multidimensional scaling (NMDS) ordination based on a Bray-Curtis matrix of dissimilarities calculated from fish abundance (CPUE) data collected from the four sites surveyed in 2016 (baseline survey – black), and five sites surveyed in 2017 (one-year post-rehabilitation – white), and 2021 (five-years post-realignment, grey; this study). Control sites are shown as squares; realigned sites are shown as circles. Axes are identically scaled so that sites closest together are more similar in community composition, than those further apart. The significance of differences in community dissimilarity was confirmed using Analysis of Similarities (ANOSIM).

4.0 Discussion

4.1 Water quality

In general, there were no discernible differences in water quality measures between control and realigned sites, nor between surveys. As noted previously, the water quality parameters measured can vary diurnally and seasonally and only single measures of each parameter were taken at each site in each survey.

The realignment and associated rehabilitation work (including planting and bank reshaping) that would have been expected to improve shading and reduce runoff have likely not become established enough to provide these functions effectively. The catchment receives a variety of inputs from both rural and urban sources that have a significant influence on water quality in Kā Pūtahi Creek.

The measured temperature, pH, and conductivity in the 2021 survey were all within or above recommended freshwater outcomes set out in the LWRP and the attribute target levels of the CSNDC; and are within ranges aquatic life can tolerate. Dissolved oxygen was notably low at C2, and only two sites reached the minimum guideline (70%) for spring-fed plains waterways as set in the LWRP. High levels of sediment and large proportions of detritus likely contributed to the lower DO measured at C2. However, as this measurement was particularly low and was not reflected at any other site or in any other survey, this may have been due to a meter error.

4.2 Habitat

The realignment of Kā Pūtahi Creek presented an opportunity to 'naturalise' a section of the waterway and incorporate a variety of features to diversify the available habitat. The realigned reach was designed to include a variety of bank heights and grades, flow habitats, and introduce logs, wetland areas and extensive indigenous planting in the riparian area.

Habitat along the now-retired oxbow reach of Kā Pūtahi Creek was limited for aquatic fauna (Boffa Miskell 2016). The channel was silt-dominated with little to no flow, in-stream habitat was heterogenous (i.e. few logs, cobbles or macrophytes to provide variable habitat) and banks were unstable in places. The surrounding riparian vegetation was comprised of exotic trees along one bank and grasses along the other, giving little buffer from the surrounding agricultural land. The macroinvertebrate communities found were indicative of poor stream health, and the fish community was of low species diversity. Nevertheless, the waterway supported two fish species of conservation concern (inanga and longfin eel) despite the poor habitat quality.

Both the control and realigned reaches are wide with relatively shallow and slow flowing water.

Restoration planting as part of the realignment works has greatly improved the riparian zone when compared to baseline conditions. Since the realignment work was completed in mid-2016, the planted vegetation has grown substantially. Although canopy cover has not changed since 2017, this survey in 2021 was the first time that vegetation heights over 2 m have been measured, where 12.8% of the riparian margin (up to 20 m from the water's edge) has vegetation in the 2 m to 12 m high categories.

Banks are stable along the realigned reach, and there are few gaps in vegetation cover. *Carex* sedges, flaxes and toetoe planted along the immediate riparian margins have grown to overhang the waterway, creating in-stream habitat for fauna and provide some shading to the waterway. Shading from over the realigned section of Kā Pūtahi Creek is variable, with narrower reaches receiving significant shading from overhanging sedges and flaxes, while wider sections receive limited shading from riparian planting. While shading will remain limited where the stream is very wide, taller tree species should continue to grow and increase shading to the waterway. Importantly, the indigenous vegetation along the riparian zone greatly increases the availability of diverse high quality terrestrial habitat alongside the creek, and provides potential habitat and refuge for birds, insects and lizards.

In order to maintain the quality of habitat along Kā Pūtahi Creek, some additional management is likely required. Some pest tree species have grown on the banks of the realigned waterway, including grey willow, alder and crack willow. Control has taken place in some areas, including on the flood plain slightly upstream of R1. However, some secondary death of planted *Carex* sedges has also occurred here. Using a broadleaf specific herbicide to control pest trees should avoid this secondary plant mortality. Some mortality of taller tree species has also occurred in the wider riparian zone, and replacement planting should be considered. Shading from canopy-forming or taller trees is essential for limiting excessive macrophyte growth, and will be an important consideration given that macrophyte cover is high at realigned sites.

In fact, macrophyte cover throughout the realigned reach was very high, covering an average of 60.2% of the assessed reaches – above the acceptable total macrophyte cover of the LWRP and exceeding the attribute target level in the CSNDC (maximum of 50% total cover of macrophytes for spring-fed – plains waterways).

Downstream (i.e., at R3), macrophytes were primarily submerged curly-leaved pondweed and Canadian pondweed, while at upstream sites, dense watercress beds affected flow. Coupled with already very slow water velocity, the extensive macrophyte cover is further slowing water velocity, and likely contributing to the accumulation of fine sediment in places. As riparian vegetation continues to grow, increased shading should prevent such extensive growth of the macrophytes. However, in areas where the waterway is very wide or potential shading is still many years away, active management (i.e. selective macrophyte clearance) is likely required.

While the substrate of the new channel was initially dominated by cobbles and gravels (EOS Ecology 2017), finer sediments have accumulated substantially throughout the realigned channel. High sediment inputs from rural sources upstream and very slow water velocity throughout allow suspended sediments to deposit in the waterway and are the primary causes of the sediment accumulation. Nevertheless, the substrates in the realigned reach are less embedded than at the control sites; compactness is greater at control sites. It is anticipated that further sediment will accumulate throughout the realigned reach, continuing to bury the stony substrate.

4.3 Macroinvertebrate communities

The macroinvertebrate community found in the 2021 survey was significantly different in composition to that found in 2017, but relatively similar to community composition found in 2016. It appears that the shift in macroinvertebrate community (2017) was due to initial in-stream changes due to realignment activities (e.g., clean gravels, open water, riffle habitat). True flies and molluscs dominated the community in the realigned reaches in 2017. The community in 2021 has become more similar to the community in the control reach and 2016 (dominated by crustaceans and molluscs).

The macroinvertebrate community found has remained dominated by “pollution-tolerant” crustaceans and snails, which are typically indicative of poorer quality aquatic habitat and are characteristic within waterways in the wider Christchurch region. Since the baseline survey, no new notable taxa (including of higher ecological value or sensitivity) have been found in the realigned sites.

All sites remain in “poor” health as indicated by MCI and QMCI scores. Through time, the average MCI scores found at each site have remained relatively similar at all sites. However, when taking relative abundance into account, the average QMCI has declined at realignment sites since the baseline survey; QMCI scores at control sites were greater than at realigned sites in 2021. No notable changes have occurred since the baseline survey at control sites, with scores indicating all sites have been of “poor” health (range of 2.2 - 3.3). The freshwater outcome ecological health indicator of the LWRP and the attribute target level of the CSNDC is 5 for “spring-fed – plains waterways” such as Kā Pūtahi Creek.

Analysis of the composition of species at control versus realigned sites showed a small number of taxa differ in abundance between the treatments.

It is important to note that macroinvertebrate communities can be variable through time and are sensitive to disturbances. Of note is that all ‘Year One’ sites supported a significantly different community composition to that observed in both the baseline survey and this survey. Further analysis of the data may reveal an insight into the factors influencing the macroinvertebrate community. The shift in community could be as a result of several factors, including changes to the habitat available, water chemistry attributes, and general conditions caused by the time of year and the weather preceding the survey. The ‘Year One’ survey took place within a year of the realignment works and thus successional changes within the faunal communities have likely taken place since the new channel was livened. Additionally, the accumulation of sediments and increase in macrophyte cover within the realigned reach of Kā Pūtahi Creek, and thus reduced availability of stony substrate that is clear of sediment has reduced suitable habitat for pollution-sensitive EPT taxa.

4.4 Fish communities

The fish communities found at all sites within Kā Pūtahi Creek have remained relatively similar in each survey. Further, a comparable number of fish were caught in each survey, with only small changes in abundances of common bully and shortfin eel found between the ‘Year One’ and ‘Year Five’ surveys.

Electric fishing surveys that were conducted in the ‘Year One’ survey were not repeated as the accumulation of sediment and slow velocity in the realigned reaches meant the method was likely to be ineffective. Electric fishing was also not appropriate at C2 due to deep sediment.

It is worth noting that different survey methods (e.g., trapping versus electric fishing) have different biases for capturing certain species. For example, giant bully and inanga are underestimated using electric-fishing methods. Conversely trapping will likely detect presence of inanga but underestimate eels (Joy et al. 2013). These biases in species detection may have had some influence on the results of the fish communities found.

More fish were caught at realigned sites than at the downstream control site (C2) in 2021, though a similar number of fish were caught at the upstream control site (C1) compared to the realigned sites. This may be a reflection of the habitat quality at the downstream control site, which had deep soft sediment substrate, and high cover of filamentous algae. There wasn’t, however, a significant difference in the total abundance of fish caught at control sites compared to realigned sites overall.

There has been a slight change in the community composition detected. However, no new species have been found in the creek since the baseline survey, and differences in community detected have been largely driven by changes in abundances of fish species. Giant bullies were not found in the 2021 survey; previously this species had only been found in low numbers. While giant bullies are underestimated by electric fishing methods (Joy et al. 2013), this species is often only detected in low numbers and can be difficult to detect even when present. It appears that habitat suitable for giant bullies is still available (e.g., undercut banks with overhanging vegetation), and this species is likely present.

No clear trends have arisen in the fish community in Kā Pūtahi Creek at both control and realigned sites. Nevertheless, habitat along the realigned reach is much more diverse than was available in the original channel. As the habitat develops in the realigned reach of the waterway, such as increasing overhanging vegetation and increasing organic matter entering the reach, habitat availability will likely continue to diversify and be reflected in the species present and their abundance. However, the increased habitat diversity achieved through the realignment works may be limited by the high levels of sediment continuing to enter the waterway and settling along the realigned reach, clogging interstitial spaces that would otherwise provide good habitat for aquatic fauna.

4.5 Current successes and limitations of rehabilitation works

Realigning Kā Pūtahi Creek presented a unique opportunity to increase the “naturalness” and aquatic habitat diversity and availability in the waterway. While riparian and in-stream conditions have been improved considerably throughout the reach, the ecological condition of the waterway has remained in “poor” health when considering the macroinvertebrate communities found (as indicated by MCI and QMCI scores). There are likely a number of limiting factors contributing to this, as discussed below.

Habitat

The works to create wetland floodplains and extensive riparian planting have become further established since the ‘Year One’ survey and thus increased available habitat to both fish and macroinvertebrate fauna in places. When compared to baseline conditions, habitat diversity and availability is significantly improved, and should continue to improve.

Heterogeneity of substrate and in-stream habitat is incredibly important for aquatic fauna, with variety providing diverse options for refuge, feeding and breeding habitat. Many macroinvertebrate species require cobbles clear of algae and macrophytes as habitat and for laying eggs, while fish species such as bullies use stable boulders as breeding habitat. Terrestrial fauna, including insects and birds, can also use these features as refuge and habitat.

However, while creating such conditions has been successful in parts of the waterway, success in maintaining such conditions has been somewhat limited. While gravels and cobbles were added as part of the channel construction, sediment has almost entirely covered the length of the realigned stream, limiting the availability of suitable habitat for a wider range of species.

Further, some areas of the creek have been constructed to be very wide, resulting in very slow water velocity through these sections. Subsequently, deep sediment has accumulated in places, in addition to extensive macrophyte cover throughout the channel. While some of this excess macrophyte growth will reduce as riparian vegetation grows and shades the waterway, the riparian vegetation will not entirely shade the wider sections of the waterway, even once fully established.

It is probable that the riparian wetlands and vegetation provide an improved buffer to urban runoff into the river. Incorporating analysis of water chemistry measures, such as dissolved metals, nutrient levels and other common urban inputs into waterways, could provide a more complete understanding as to the influence of the rehabilitation works on stream health, and in turn, the faunal communities within it.

Within the realigned reach, macrophyte removal should be considered as a short-term solution to help address these issues. We understand that this has already been carried out (B. Margetts, pers. com. July 2021). Consideration should also be made to reduce the width of the waterway in places. This could involve adding boulders and logs to the edges of the creek to reduce the wetted width. In addition to narrowing the channel, this would also increase habitat diversity available to aquatic fauna. However, these modifications to the channel are unlikely to increase water velocity (and thereby create riffle habitat) as there is not sufficient fall. We understand that following our survey some work was undertaken to investigate the addition of logs to the widest areas of the channel in order to narrow the wetted width of the waterway, but this is not going to be carried out (due to insufficient fall to create riffle habitat) (B. Margetts, pers. com. July 2021).

Management at the catchment scale is likely needed to address the high sediment load within the waterway. Improvements could include fencing and riparian planting of the waterway along its length and identifying and addressing point sources of sediment and contaminant inputs (i.e. stormwater discharges).

Fauna

The fish community found within Kā Pūtahi Creek has remained relatively unchanged, both in terms of species richness and abundance, since the waterway was realigned. Species composition has remained the same, with the exception of giant bullies not being found in 2021. As previously discussed, giant bullies have only been caught in low numbers previously and may be present still.

A shift in the macroinvertebrate community has been observed since the baseline survey in 2016, where freshwater seed shrimp (ostracods) and snails such as *Potamopyrgus* have fluctuated in prevalence, though remained dominant. The proportion of EPT taxa (here only made up by caddisflies and infrequent mayflies) in the community has remained very low. Water quality, as indicated by MCI and QMCI scores, has remained poor at the realigned sites in 2021. No clear trends can yet be determined as the 2021 survey was only the second monitoring occasion in the realigned reach of Kā Pūtahi Creek.

Colonisation of the river by new macroinvertebrate taxa of higher ecological value or sensitivity was not detected in this survey (2021). The limitations that have contributed to this are likely to be complex. For example, the habitat available to macroinvertebrates that favour cobbly substrate has decreased, not increased, since 2017. Further, aquatic insect colonisation via aerial colonisation from other waterways may also be limited for a variety of reasons. Colonisation by way of aerial dispersal of adult insects with winged adult life stages, such as EPT taxa, is likely more difficult in the semi-urban environment where cross-catchment dispersal paths may be disrupted by the general urban environment (i.e. road crossings, buildings etc.) (Blakely et al. 2006).

A further barrier to recolonisation is a general lack of source populations by taxa, such as mayflies and stoneflies. Still, likelihood of colonisation by EPT taxa into Kā Pūtahi Creek is higher than that for waterways outside of the Styx River catchment; the Styx River hosts EPT taxa that are generally absent from other catchments within the wider Christchurch urban area (e.g. the Heathcote and Avon River catchments) (Boffa Miskell 2020; Blakely et al. 2006).

The high density of macrophytes in the realigned channel was identified in places as potentially affecting flow throughout the reach. Periodic and selective clearing of macrophytes should be considered, however, it should be noted that fauna (fish and macroinvertebrates) are likely to be present in the macrophytes when cleared, leading to some mortality of freshwater fauna. As shading to the channel increases as riparian vegetation grows, the need for macrophyte clearance should become incrementally less, as shade will prevent excessive macrophyte growth. An adaptive management approach should be adopted by assessing the frequency, timing and general methods of macrophyte clearance, and adjusting over time as necessary. Targeting problematic areas and leaving macrophytes and debris where they are not likely to cause flooding problems may also help to maintain diverse habitat for aquatic fauna.

5.0 Recommendations

The realignment work undertaken in Kā Pūtahi Creek aimed to avoid the need for two large box culvert crossings for the construction of the Christchurch Northern Corridor. The CCC used this opportunity to enhance the waterway and improve the habitat available in the relatively channelised and degraded system.

Overall, significant improvements have been made to the riparian habitat along the realigned channel, when compared to conditions found in the baseline survey. Varied indigenous species provide diverse habitat along the streams' banks, as well as stabilising the banks and providing some shading to the waterway.

An initial improvement to in-stream health (as indicated by the macroinvertebrate community index) was apparent in 2017, one year after the realignment works. However, this 2021 survey, five years after the realignment works indicates little difference in macroinvertebrate community or stream health as a result of the stream realignment and enhancement. This is likely to be, in part, due to high levels of sediment deposition on the cobble bed within the realigned channel.

Additional rehabilitation works, which could further enhance ecological improvements within the realigned reach of Kā Pūtahi Creek could include:

- Introduce a macrophyte management programme to remove excess macrophytes.
    - Macrophyte management should be undertaken in such a way as to avoid mortality of fish and other freshwater fauna and should be selective (i.e., not all of the macrophyte should be removed) so as to maintain habitat heterogeneity and variability throughout the channel.
- Multi-faceted approach to stormwater management, to reduce the inputs of sediments and contaminants to the river.
    - Identify, isolate/treat, and monitor fine sediment inputs in Kā Pūtahi Creek catchment. Fine sediment accumulation is significantly impacting the availability of habitat at realignment sites.
    - Consider removing excess sediment with a tool (such as 'Sand Wand') to increase available stony habitat for aquatic fauna.
- Add logs and boulders along the edge of the waterway, particularly where the channel is very wide, to concentrate flow and increase and diversify habitat in the waterway[8].
- Address plant mortality in the wider riparian zone of the realigned reach. In the wider riparian zone (i.e., 5-10 m back from the waterway) there has been some death of planted trees. Replacing these should take place to ensure the function of the riparian zone as a filter for overland and underground runoff is achieved.
- Plant tall and / or canopy forming trees closer to the water's edge, to shade the channel keeping water temperatures cool, inputting appropriate leaf litter resources, and limiting excessive macrophyte growth.
- Continue to control pest tree species such as grey willow, crack willow and alder.

[8] We understand the placement of logs along the wetted margins of Kā Pūtahi Creek was considered but will not be undertaken as the fall is not sufficient to increase water velocities and addition of logs alone will not create riffle habitat (B. Margetts, pers. comm July 2021).

- Any weed spraying should be done with broadleaf-specific herbicide to avoid secondary death of *Carex* plants. Larger trees may require other treatment, such as drilling and filling or cutting and pasting with an appropriate herbicide.
- These poisoned trees should be left standing to fall naturally to increase habitat complexity and increase organic matter in the area; and avoid disturbance to stream banks.
- Replacement planting of trees that have died should be considered where mortality has resulted in large gaps in the vegetation cover.

- A large amount of rubbish appeared to have been dumped around site C1. We recommend this rubbish be collected to prevent it, or related contaminants, entering the waterway.

Monitoring programme

At present the monitoring programme methods are very rigorous, however, not all methods are well suited to the waterway's conditions. For example, the Surber sampler is designed for riffle-type habitat with cobble substrate. Kā Pūtahi Creek, while historically hard-bottomed, has become a soft-sediment dominated waterway and is very deep and relatively slow flowing in places. Use of the standard kick-net methodology for macroinvertebrate sampling (as used in the CCC standard methods) may be more appropriate than Surber sampler for this waterway. Limitations in comparisons to previous data will need to be carefully considered, however. Collection of a single kick-net sample will provide information on community composition but does not provide the same information as replicate Surber samples.

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Appendix 1: Protocol 3 (P3), Harding et al., 2009

P3b field form

Site code		Site name	
Assessor		Date	

Reach assessment

Wetted width (m)		
Reach length (m)		
	Easting	Northing
Reach start		
Reach end		

Pool	Maximum depth(m)	Sediment depth (m)	Crest depth (m)
1			
2			
3			
4			
5			
6			

Meso-habitat length (m)					
Rapid	Run	Riffle	Pool	Backwater	Other

Plan diagram of the site (include significant land marks, access points, N direction, direction of stream flow, location of roads, rough scale)

Notes/comments

Cross sections

Run	Location^{+}									Water depth below staff gauge										
	LBF	LB$_1$	LB$_2$	LB$_3$	WE	1	2	3	4	5	6	7	8	9	10	WE	RB$_3$	RB$_2$	RB$_1$	RBF
Offset (m)																				
Depth (m)																				
Velocity	0	0	0	0	0											0	0	0	0	0

$^{+}$ ‘head’, ‘middle’ or ‘tail’ of run
LBF = left bank full, LB = left bank (for bank offsets record distance between ground and transect line in depth row), WE = water’s edge

Run	Location^{+}									Water depth below staff gauge										
	LBF	LB$_1$	LB$_2$	LB$_3$	WE	1	2	3	4	5	6	7	8	9	10	WE	RB$_3$	RB$_2$	RB$_1$	RBF
Offset (m)																				
Depth (m)																				
Velocity	0	0	0	0	0											0	0	0	0	0

Run	Location^{+}									Water depth below staff gauge										
	LBF	LB$_1$	LB$_2$	LB$_3$	WE	1	2	3	4	5	6	7	8	9	10	WE	RB$_3$	RB$_2$	RB$_1$	RBF
Offset (m)																				
Depth (m)																				
Velocity	0	0	0	0	0											0	0	0	0	0

Riffle	Location^{+}									Water depth below staff gauge										
	LBF	LB_1	LB_2	LB_3	WE	1	2	3	4	5	6	7	8	9	10	WE	RB_3	RB_2	RB_1	RBF
Offset (m)																				
Depth (m)																				
$^{+}$ 'head', 'middle' or 'tail' of run																				

Riffle	Location^{+}									Water depth below staff gauge										
	LBF	LB_1	LB_2	LB_3	WE	1	2	3	4	5	6	7	8	9	10	WE	RB_3	RB_2	RB_1	RBF
Offset (m)																				
Depth (m)																				

Riffle	Location^{+}									Water depth below staff gauge										
	LBF	LB_1	LB_2	LB_3	WE	1	2	3	4	5	6	7	8	9	10	WE	RB_3	RB_2	RB_1	RBF
Offset (m)																				
Depth (m)																				

Pool	Location[+]									Water depth below staff gauge										
	LBF	LB_1	LB_2	LB_3	WE	1	2	3	4	5	6	7	8	9	10	WE	RB_3	RB_2	RB_1	RBF
Offset (m)																				
Depth (m)																				

[+] 'head', 'middle' or 'tail' of run
LBF = left bank full, LB = left bank (for bank offsets record distance between ground and transect line in depth row), WE = water's edge

Pool	Location[+]									Water depth below staff gauge										
	LBF	LB_1	LB_2	LB_3	WE	1	2	3	4	5	6	7	8	9	10	WE	RB_3	RB_2	RB_1	RBF
Offset (m)																				
Depth (m)																				

Pool	Location[+]									Water depth below staff gauge										
	LBF	LB_1	LB_2	LB_3	WE	1	2	3	4	5	6	7	8	9	10	WE	RB_3	RB_2	RB_1	RBF
Offset (m)																				
Depth (m)																				

P3c field form

Site name		Site code	
Assessor		Date	

Riffle 1	Cross-section	Wetted width (m)									
		1	2	3	4	5	6	7	8	9	10
	Substrate size										
	Embeddedness										
	Compactness										
	Depositional & scouring (cm)										
	Macrophytes (cm)										
	Algae (cm)										
	Leaf packs (cm)										
	Woody debris (cm)										
	Large boulders & log jams (count)										
	Bank cover (m)	Left bank					Right bank				

Riffle 2	Cross-section	Wetted width (m)									
		1	2	3	4	5	6	7	8	9	10
	Substrate size										
	Embeddedness										
	Compactness										
	Depositional & scouring (cm)										
	Macrophytes (cm)										
	Algae (cm)										
	Leaf packs (cm)										
	Woody debris (cm)										
	Large boulders & log jams (count)										
	Bank cover (m)	Left bank					Right bank				

Run 1	Cross-section	Wetted width (m)									
		1	2	3	4	5	6	7	8	9	10
	Substrate size										
	Embeddedness										
	Compactness										
	Depositional & scouring (cm)										
	Macrophytes (cm)										
	Algae (cm)										
	Leaf packs (cm)										
	Woody debris (cm)										
	Large boulders & log jams (count)										
	Bank cover (m)	Left bank					Right bank				

Run 2	Cross-section	Wetted width (m)									
		1	2	3	4	5	6	7	8	9	10
	Substrate size										
	Embeddedness										
	Compactness										
	Depositional & scouring (cm)										
	Macrophytes (cm)										
	Algae (cm)										
	Leaf packs (cm)										
	Woody debris (cm)										
	Large boulders & log jams (count)										
	Bank cover (m)	Left bank					Right bank				

Pool 1	Cross-section	Wetted width (m)									
		1	2	3	4	5	6	7	8	9	10
	Substrate size										
	Embeddedness										
	Compactness										
	Depositional & scouring (cm)										
	Macrophytes (cm)										
	Algae (cm)										
	Leaf packs (cm)										
	Woody debris (cm)										
	Large boulders & log jams (count)										
	Bank cover (m)	Left bank					Right bank				

Pool 2	Cross-section	Wetted width (m)									
		1	2	3	4	5	6	7	8	9	10
	Substrate size										
	Embeddedness										
	Compactness										
	Depositional & scouring (cm)										
	Macrophytes (cm)										
	Algae (cm)										
	Leaf packs (cm)										
	Woody debris (cm)										
	Large boulders & log jams (count)										
	Bank cover (m)	Left bank					Right bank				

P3d field form

Site name		Site code	
Assessor		Date	

	Buffer width (m)		Land slope		Distance to stopbank (m)		Distance to floodplain (m)	
Cross-section	LB	RB	LB	RB	LB	RB	LB	RB
1								
2								
3								
4								
5								

Riparian vegetation	Distance from LB (m)				Distance from RB (m)			
Cross-section 1	0.5	3	7.5	20	0.5	3	7.5	20
Native vegetation	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N
Veg tier height								
0 - 0.3 m								
0.3 - 1.9 m								
2.0 - 4.9 m Shrubs								
5 - 12 m Subcanopy								
>12 m Canopy								
Cross-section 2								
Native vegetation	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N
Veg tier height								
0 - 0.3 m								
0.3 - 1.9 m								
2.0 - 4.9 m Shrubs								
5 - 12 m Subcanopy								
Cross-section 3								
Native vegetation	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N
Veg tier height								
0 - 0.3 m								
0.3 - 1.9 m								
2.0 - 4.9 m Shrubs								
5 - 12 m Subcanopy								
Cross-section 4								
Native vegetation	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N
Veg tier height								
0 - 0.3 m								
0.3 - 1.9 m								
2.0 - 4.9 m Shrubs								
5 - 12 m Subcanopy								
Cross-section 5								
Native vegetation	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N	Y/N
Veg tier height								
0 - 0.3 m								
0.3 - 1.9 m								
2.0 - 4.9 m Shrubs								
5 - 12 m Subcanopy								
>12 m Canopy								

	Left bank	Right bank
Gaps in buffer		
Wetland soils		
Stable undercuts		
Livestock access		
Bank slumping		
Raw bank		
Rills/Channels		
Drains (count)		

Shading of water				

Notes

Appendix 2: Sediment Assessment Method 2 (SAM2), Clapcott et al., 2011

Sediment Assessment Method 2 – In-stream visual estimate of % sediment cover

Rationale	Semi-quantitative assessment of the surface area of the streambed covered by sediment. At least 20 readings are made within a single habitat
Equipment required	• Underwater viewer - *e.g.*, bathyscope (www.absolutemarine.co.nz) or bucket with a Perspex bottom marked with four quadrats • Field sheet
Application	Hard-bottomed streams
Type of assessment	Assessment of effects
Time to complete	30 minutes
Description of variables	
% sediment	A visual estimate of the proportion of the habitat covered by deposited sediment (<2 mm)
Useful hints	Work upstream to avoid disturbing the streambed being assessed. Mark a four-square grid on the viewer to help with estimates – determine the nearest 5% cover for each quadrat. Calculate the average of all quadrats as a continuous variable following data entry. More than five transects may be necessary for narrow streams, to ensure 20 locations are sampled.

Field procedure

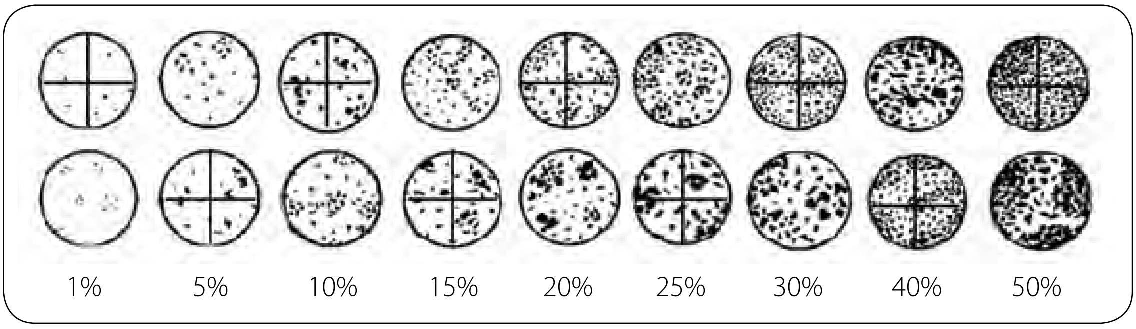
- Locate five random transects along the run.
- View the streambed at four randomly determined locations across each transect, starting at the downstream transect.
- Estimate the fine sediment cover in each quadrat of the underwater viewer in increments (1, 5, 10, 15, 20 …100%).
- Record results in the table below.
- Repeat for four more transects so that 20 locations are sampled in total.

Note: Estimation of cover in each quadrat is important during training but may not be necessary for experienced viewers – instead one measurement per location could be recorded.

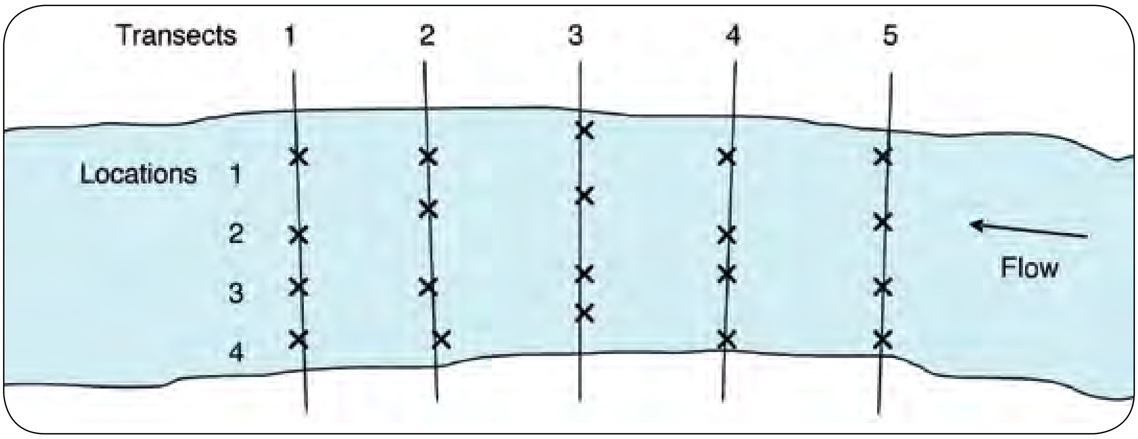
% sediment	Transect 1		Transect 2		Transect 3		Transect 4		Transect 5	
Location 1	Q1	Q2								
	Q3	Q4								
Location 2										
Location 3										
Location 4										

Useful images

Digital examples of percent cover of sediment on the streambed as seen through an underwater viewer.



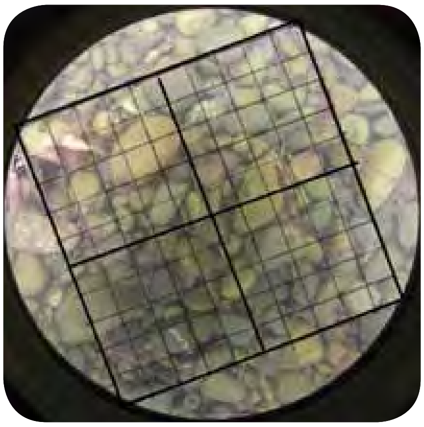
An example of viewer locations (x) for the in-stream visual assessment of sediment.



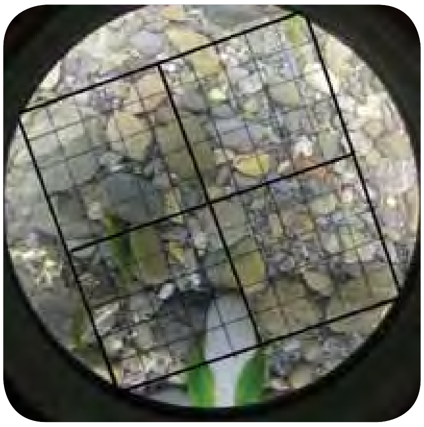
Real examples of percent cover of sediment on the streambed as seen through an underwater viewer.

1%

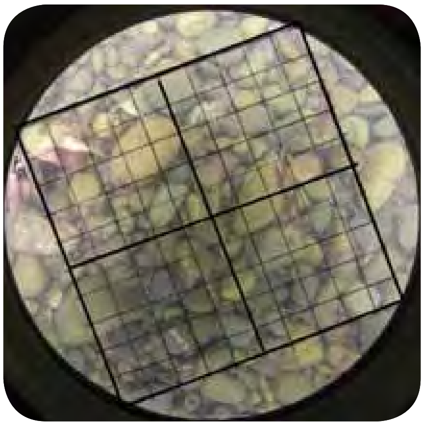
1%



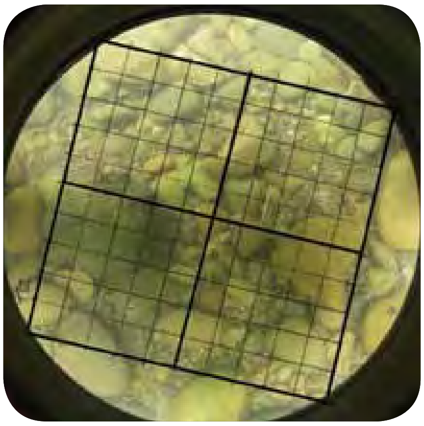
5%



5%



10%



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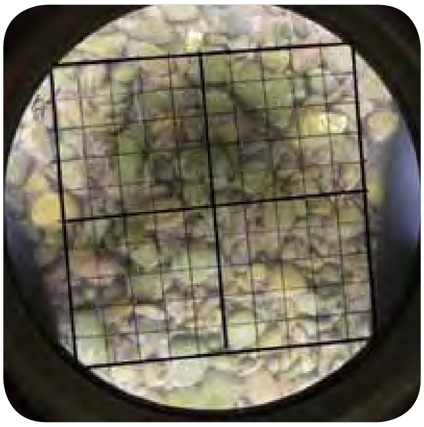
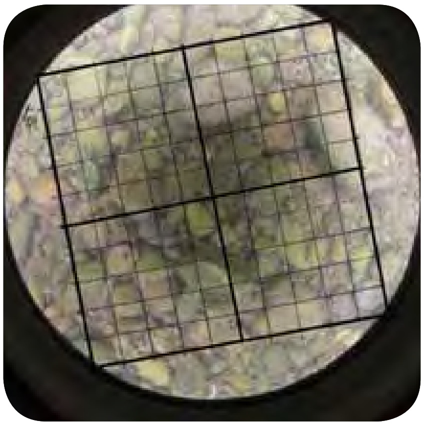


15%



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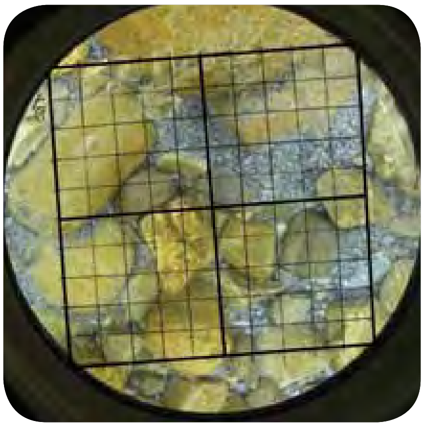
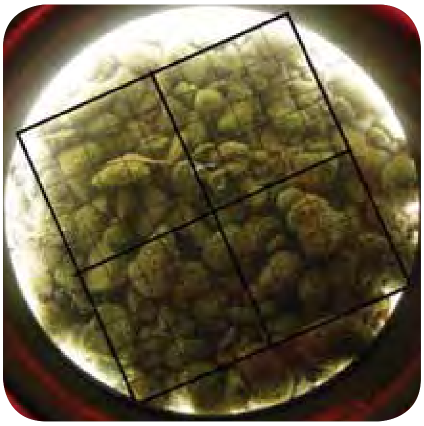
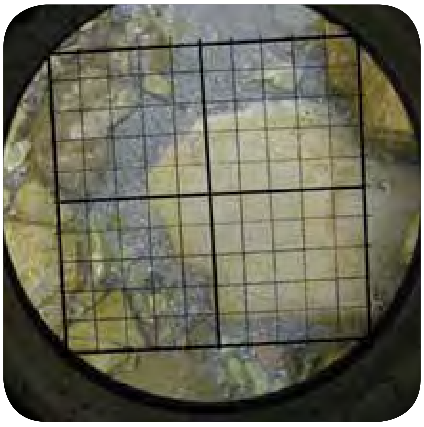


20%

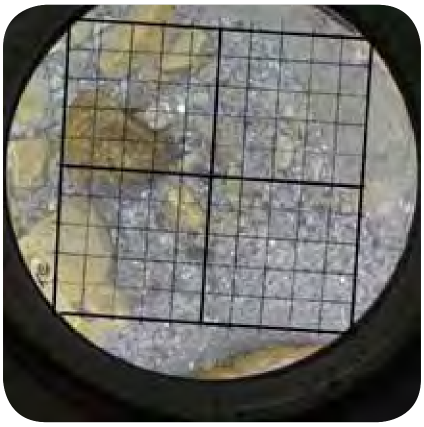


20%

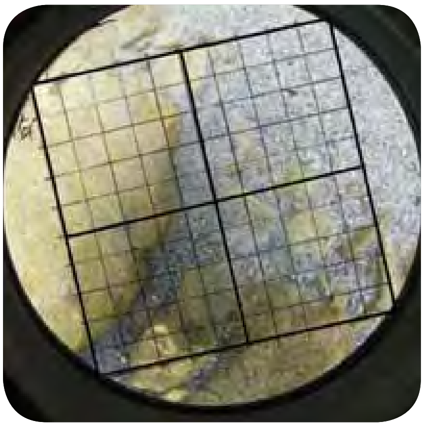


25%

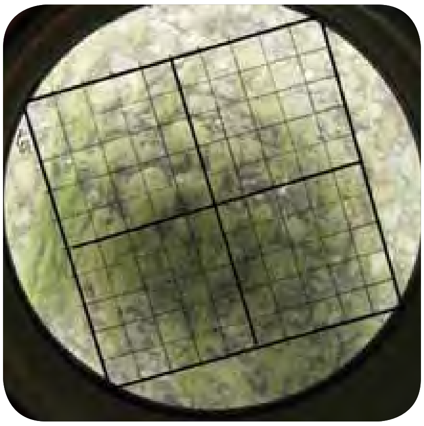
30%



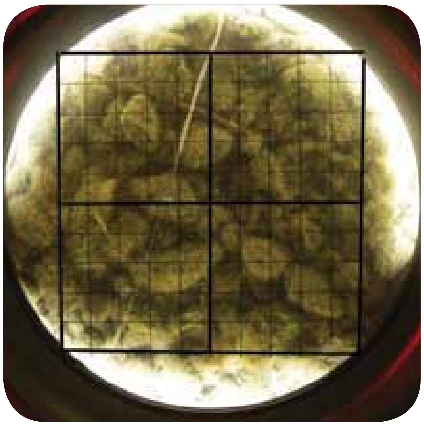
40%



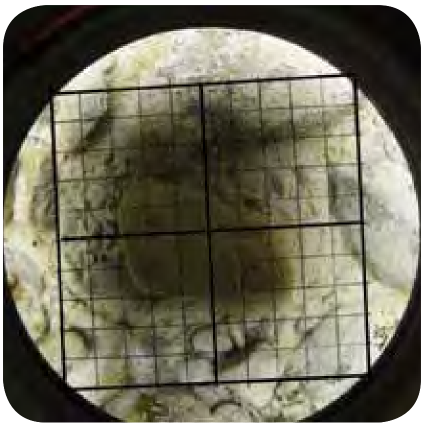
50%



90%



100%



Appendix 3: Sediment Assessment Method 6 (SAM6), Clapcott et al., 2011

Sediment Assessment Method 6 –Sediment depth

Rationale	Quantitative assessment of the depth of sediment in a run habitat. At least 20 readings are made within a single habitat
Equipment required	• Ruler or ruled rod • Field sheet
Application	Hard-bottomed streams
Type of assessment	Assessment of effects
Time to complete	30 minutes
Description of variables	
Sediment depth (mm)	A measure of the depth of sediment (mm).
Useful hints	Determine the sampling grid first to ensure an even cover of edge and midstream locations. Move upstream to avoid disturbing the streambed being assessed. Calculate the average depth for each site. This method is usually only suitable when fine sediment is visible from the stream bank.

Field procedure

- Start downstream and randomly locate five transects along the run.
- Measure the sediment depth (mm) at four randomly determined locations across each transect and record depth in the table below.

Depth (mm)	Transect 1	Transect 2	Transect 3	Transect 4	Transect 5
Section 1					
Section 2					
Section 3					
Section 4					

Appendix 4: SIMPER & ANOSIM results – macroinvertebrate community

ANOSIM
Analysis of Similarities

Two-Way Crossed - AxB

Resemblance worksheet
Name: Resem2
Data type: Similarity
Selection: All

Factors

Place	Name	Type	Levels
A	Year	Unorderec	3
B	Treatment	Unorderec	2

Year levels
2016
2017
2021

Treatment levels
Control
Realigned

Tests for differences between unordered Year groups
(across all Treatment groups)
Global Test
Sample statistic (Average R): 0.635
Significance level of sample statistic: 0.4%
Number of permutations: 999 (Random sample from 2100)
Number of permuted statistics greater than or equal to Average R: 3

Pairwise Tests

Groups	R Statistic	Significanc Level %	Possible Permutati	Actual Permutati	Number >= Observed
2016, 2017	0.679	6.7	15	15	1
2016, 2021	0.571	13.3	15	15	2
2017, 2021	0.629	10	30	30	3

Tests for differences between unordered Treatment groups
(across all Year groups)
Global Test
Sample statistic (Average R): 0.625
Significance level of sample statistic: 1%
Number of permutations: 100 (All possible permutations)
Number of permuted statistics greater than or equal to Average R: 1

Outputs
Plot: Graph15
Plot: Graph16
Worksheet: Resem3

SIMPER
Similarity Percentages - species contributions

Two-Way Analysis

Data worksheet
Name: Macros_sqrt
Data type: Abundance
Sample selection: All
Variable selection: All

Parameters
Resemblance: S17 Bray-Curtis similarity
Cut off for low contributions: 70.00%

Factor Groups

Sample	Year x Treatment
2016 C1	2016 x Control
2016 Site 2	2016 x Control
2016 Site 3	2016 x Control
2016 C2	2016 x Control
2017 C1	2017 x Control
2017 C2	2017 x Control
2017 R1	2017 x Realigned
2017 R2	2017 x Realigned
2017 R3	2017 x Realigned
2021 C1	2021 x Control
2021 C2	2021 x Control
2021 R1	2021 x Realigned
2021 R2	2021 x Realigned
2021 R3	2021 x Realigned

Examines Year groups
(across all Treatment groups)
Group 2016
Average similarity: 74.75

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Ostracoda	17.01	18.48	7.16	24.72	24.72
Potamopyrgus	11.85	10.63	2.86	14.23	38.95
Oligochaeta	8.96	8.98	5.26	12.01	50.96
Sphaeriidae	6.69	6.2	2.19	8.29	59.25
Cladocera	6.62	5.99	3.47	8.02	67.27
Hydra	5.26	5.35	4.33	7.16	74.43

Group 2017
Average similarity: 49.16

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Physa	11.88	6.29	1.96	12.8	12.8

Sigara	6.23	6.23	1.71	12.68	25.48
Orthocladiinae	9.55	5.41	1.7	11.01	36.49
Oligochaeta	6.09	3.25	1.87	6.61	43.1
Chironomus	5.24	2.62	3.15	5.32	48.42
Potamopyrgus	8.92	2.49	0.5	5.07	53.49
Ostracoda	6.08	2.43	1.12	4.95	58.44
Tanytarsini	3.13	2.18	1.41	4.44	62.88
Platyhelminthes	3.35	2.06	1.21	4.18	67.06
Tanypodinae	2.69	1.8	3.52	3.66	70.73

Group 2021
Average similarity: 68.26

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Ostracoda	28.03	16.86	3.76	24.7	24.7
Potamopyrgus	17.47	10.08	13.55	14.77	39.47
Oligochaeta	12.92	8.21	6.06	12.03	51.51
Gyraulus	7.79	3.81	1.46	5.58	57.09
Sphaeriidae	5.88	3.57	3.16	5.23	62.32
Platyhelminthes	3.99	2.92	2.95	4.27	66.59
Physa	4.56	2.6	2.82	3.81	70.41

Groups 2016 & 2017
Average dissimilarity = 44.84

	Group 2016	Group 2017				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Potamopyrgus	11.85	8.92	8.28	1.38	18.46	18.46
Ostracoda	17.01	6.08	5.89	2.96	13.12	31.58
Oligochaeta	8.96	6.09	3.85	3.01	8.58	40.17
Hydra	5.26	5.56	2.89	3.51	6.45	46.62
Sphaeriidae	6.69	1	2.56	1.83	5.71	52.33
Cladocera	6.62	2.76	2.25	1.43	5.01	57.34
Nematoda	0	2.02	2.07	3.07	4.62	61.96
Physa	3.6	11.88	2.02	1.5	4.51	66.47
Platyhelminthes	3.98	3.35	1.61	1.7	3.6	70.08

Groups 2016 & 2021
Average dissimilarity = 36.76

	Group 2016	Group 2021				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Ostracoda	17.01	28.03	4.02	1.98	10.95	10.95
Cladocera	6.62	3.95	3.01	1.45	8.17	19.12
Potamopyrgus	11.85	17.47	2.49	1.44	6.79	25.91
Hydra	5.26	1.32	2.33	2.92	6.33	32.24
Sphaeriidae	6.69	5.88	1.8	1.94	4.88	37.12
Paracalliope	4.28	3.19	1.72	1.25	4.69	41.81
Nematoda	0	3.68	1.65	16.47	4.5	46.31
Oligochaeta	8.96	12.92	1.59	1.29	4.33	50.64

Oxyethira	0.39	2.33	1.48	1.09	4.02	54.66
Physa	3.6	4.56	1.28	1.38	3.49	58.14
Platyhelminthes	3.98	3.99	1.28	1.22	3.48	61.62
Gyraulus	0.11	7.79	1.23	0.99	3.36	64.98
Chironomus	3.5	3.73	1.22	1.49	3.33	68.31
Copepoda	2.19	1.86	1.11	0.99	3.01	71.32

Groups 2017 & 2021
Average dissimilarity = 56.37

	Group 2017	Group 2021				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Ostracoda	6.08	28.03	9.61	1.76	17.04	17.04
Potamopyrgus	8.92	17.47	7.76	1.95	13.76	30.8
Oligochaeta	6.09	12.92	3.35	1.84	5.94	36.74
Physa	11.88	4.56	3.26	1.19	5.78	42.52
Orthocladiinae	9.55	2.31	2.85	1.06	5.06	47.58
Gyraulus	2.45	7.79	2.49	1.24	4.42	52
Hydra	5.56	1.32	2.12	0.77	3.75	55.75
Sphaeriidae	1	5.88	2.09	1.9	3.7	59.45
Cladocera	2.76	3.95	1.65	1.22	2.93	62.38
Platyhelminthes	3.35	3.99	1.62	2.07	2.87	65.25
Sigara	6.23	3.18	1.53	1.72	2.71	67.96
Paracalliope	4.51	3.19	1.52	1.12	2.7	70.66

Examines Treatment groups
(across all Year groups)
Group Control
Average similarity: 69.01

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Ostracoda	14.14	15.84	2.89	22.95	22.95
Potamopyrgus	14.38	10.54	3.34	15.27	38.22
Oligochaeta	7.51	8.06	3.02	11.68	49.9
Sphaeriidae	5.03	5.47	1.94	7.93	57.83
Cladocera	4.79	4.65	1.6	6.73	64.56
Hydra	3.1	4.2	1.74	6.08	70.64

Group Realigned
Average similarity: 61.01

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Ostracoda	20.92	10.22	1.05	16.75	16.75
Oligochaeta	11.81	5.88	1.89	9.63	26.39
Physa	11.06	5.38	1.91	8.82	35.21
Sigara	7.14	5.24	1.68	8.58	43.79
Potamopyrgus	10.73	4.96	0.91	8.14	51.93
Orthocladiinae	8.89	3.86	1.13	6.32	58.25
Gyraulus	7.45	3.53	1.95	5.79	64.05
Chironomus	5.29	2.54	3.9	4.17	68.22

Tanytarsini	4.58	2.45	2.93	4.01	72.23	

Groups Control & Realigned
Average dissimilarity = 54.07

	Group Control	Group Realigned				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Potamopyrgus	14.38	10.73	7.31	1.05	13.52	13.52
Ostracoda	14.14	20.92	5.89	1.23	10.89	24.41
Orthocladiinae	1.26	8.89	3.29	0.97	6.09	30.5
Physa	3.78	11.06	3.27	0.9	6.05	36.55
Sigara	0.83	7.14	3.05	1.53	5.63	42.18
Gyraulus	0.87	7.45	2.75	1.09	5.09	47.28
Oligochaeta	7.51	11.81	2.69	1.31	4.98	52.25
Hydra	3.1	5.11	2.06	0.6	3.81	56.07
Paracalliope	4.33	3.5	1.87	1.54	3.45	59.52
Cladocera	4.79	3.62	1.74	1.34	3.22	62.74
Platyhelminthes	3.91	3.56	1.73	2.1	3.2	65.94
Tanytarsini	0.77	4.58	1.51	1.92	2.79	68.73
Chironomus	3.39	5.29	1.39	1.28	2.57	71.3

Appendix 5: SIMPER & ANOSIM results – fish community

ANOSIM
Analysis of Similarities

Two-Way Crossed - AxB

Resemblance worksheet
Name: Resem5
Data type: Similarity
Selection: All

Factors

Place	Name	Type	Levels
A	Year	Unordered	3
B	Treatment	Unordered	2

Year levels
2016
2017
2021

Treatment levels
Control
Realigned

Tests for differences between unordered Year groups
(across all Treatment groups)
Global Test
Sample statistic (Average R): 0.466
Significance level of sample statistic: 1%
Number of permutations: 100 (Random sample from 2100)
Number of permuted statistics greater than or equal to Average R: 0

Pairwise Tests

Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
2016, 2017	0.5	13.3	15	15	2
2016, 2021	0.107	46.7	15	15	7
2017, 2021	0.714	3.3	30	30	1

Tests for differences between unordered Treatment groups
(across all Year groups)
Global Test
Sample statistic (Average R): 0.5
Significance level of sample statistic: 5%
Number of permutations: 100 (All possible permutations)
Number of permuted statistics greater than or equal to Average R: 5

Outputs
Plot: Graph21
Plot: Graph22

SIMPER
Similarity Percentages - species contributions

Two-Way Analysis

Data worksheet
Name: Data6
Data type: Other
Sample selection: All
Variable selection: All

Parameters
Resemblance: S17 Bray-Curtis similarity
Cut off for low contributions: 70.00%

Factor Groups

Sample	Year x Treatment
2016 C1	2016 x Control
2016 C2	2016 x Control
2016 Site 2	2016 x Control
2016 Site 3	2016 x Control
2017 C1	2017 x Control
2017 C2	2017 x Control
2017 R1	2017 x Realigned
2017 R2	2017 x Realigned
2017 R3	2017 x Realigned
2021 C1	2021 x Control
2021 C2	2021 x Control
2021 R1	2021 x Realigned
2021 R2	2021 x Realigned
2021 R3	2021 x Realigned

Examines Year groups
(across all Treatment groups)
Group 2016
Average similarity: 56.23

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Inanga	2.77	15.57	1.62	27.69	27.69
Shortfin eel	2	13.11	3.16	23.31	51.01
Longfin eel	1.41	12.26	3.71	21.8	72.81

Group 2017
Average similarity: 77.96

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Common bully	4.65	45.48	9.78	58.34	58.34
Inanga	1.51	14.35	1.47	18.41	76.75

Group 2021

Average similarity: 56.38

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Shortfin eel	3.82	29.88	2.03	53	53
Upland bully	1.84	12.1	5.11	21.46	74.45

Groups 2016 & 2017
Average dissimilarity = 52.57

	Group 2016	Group 201				
Species	Av.Abund	Av.Abunc	Av.Diss	Diss/SD	Contrib%	Cum.%
Shortfin eel	2	1.96	12.37	1.78	23.53	23.53
Inanga	2.77	1.51	11.84	1.21	22.53	46.06
Longfin eel	1.41	0	10.7	2.1	20.35	66.4
Common bully	1.3	4.65	6.92	1.39	13.17	79.57

Groups 2016 & 2021
Average dissimilarity = 47.96

	Group 2016	Group 202				
Species	Av.Abund	Av.Abunc	Av.Diss	Diss/SD	Contrib%	Cum.%
Inanga	2.77	0.88	14.06	1.26	29.31	29.31
Shortfin eel	2	3.82	11.05	1.27	23.05	52.36
Common bully	1.3	2.09	9.21	1.4	19.2	71.56

Groups 2017 & 2021
Average dissimilarity = 50.40

	Group 2017	Group 202				
Species	Av.Abund	Av.Abunc	Av.Diss	Diss/SD	Contrib%	Cum.%
Common bully	4.65	2.09	16.48	1.98	32.69	32.69
Shortfin eel	1.96	3.82	12.45	1.55	24.7	57.39
Inanga	1.51	0.88	7.19	1.45	14.26	71.65

Examines Treatment groups
(across all Year groups)
Group Control
Average similarity: 53.77

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Inanga	2.07	15.25	1.37	28.37	28.37
Shortfin eel	2.13	11.32	1.96	21.05	49.41
Common bully	1.55	10.81	0.75	20.1	69.52
Longfin eel	0.96	9.19	1.45	17.1	86.62

Group Realigned
Average similarity: 74.09

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Common bully	4.41	31.55	1.73	42.58	42.58

Shortfin eel	3.31	23.36	1.51	31.52	74.1

Groups Control & Realigned
Average dissimilarity = 46.90

Species	Group Control Av.Abund	Group Rea Av.Abu	Av.Diss	Diss/SD	Contrib%	Cum.%
Common bully	1.55	4.41	17.26	1.76	36.8	36.8
Shortfin eel	2.13	3.31	11.82	1.21	25.2	62.01
Upland bully	0.84	2.01	8.43	1.4	17.97	79.97

About Boffa Miskell

Boffa Miskell is a leading New Zealand professional services consultancy with offices in Auckland, Hamilton, Tauranga, Wellington, Christchurch, Dunedin and Queenstown. We work with a wide range of local and international private and public sector clients in the areas of planning, urban design, landscape architecture, landscape planning, ecology, biosecurity, cultural heritage, graphics and mapping. Over the past four decades we have built a reputation for professionalism, innovation and excellence. During this time we have been associated with a significant number of projects that have shaped New Zealand's environment.

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