Avon River Precinct Aquatic Ecology

Three years' post-rehabilitation activities
Prepared for the Christchurch City Council

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Cover photograph: Rehabilitation Site 3 at Christchurch Botanical Gardens, Hagley Park, © Boffa Miskell, 2017

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Executive Summary

The Christchurch City Council commissioned Boffa Miskell Limited to repeat an aquatic ecology survey of eight sites within the Avon River catchment, as part of long-term monitoring of the Avon River Precinct rehabilitation works. The purpose of this study was to determine whether the rehabilitation works had resulted in any measurable ecological changes, and what (if any) was limiting 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 three reference and five rehabilitation sites, in March 2017. The methodology employed repeated that of surveys in 2014 (one-year post-rehabilitation activities) and 2013 (baseline, prior to rehabilitation activities).

Only weak differences in habitat conditions between reference and rehabilitation sites were found. The most notable difference was greater substrate heterogeneity and larger substrate sizes, with less fine sediment present, at rehabilitation sites. This was consistent with the 2014 survey, one-year post-rehabilitation activities.

There were also significant, but subtle, differences in the macroinvertebrate communities found in reference and rehabilitation sites. These differences were largely due to greater or lesser numbers of individuals at sites, rather than the presence or absence of a particular macroinvertebrate taxon. However, the stony-cased caddisfly *Pycnocentrodes aureolus* and the stick-cased caddisfly *Hudsonema amabile* were both more abundant in rehabilitation sites than the reference sites.

There were also notable differences in the fish communities found both across years, and between rehabilitation and reference sites. More species and individuals of freshwater fishes were captured in 2017, compared to previous years. But of greater interest, bluegill bullies were both more abundant and found at a greater number of sites (including all rehabilitation sites) in 2017, compared to previous years. Lamprey were also found at a great number of sites in 2017, compared to previous years, and a single torrentfish was found at a rehabilitation site in 2017, but this species had not been recorded at any sites previously. Bluegill bullies and torrentfish are both classified as “at risk, declining”, while lamprey are “threatened, nationally vulnerable”. Both torrentfish and lamprey are only rarely encountered in the Avon River.

Overall, there are signs of ecological gains because of rehabilitation works in the ARP sites. However, limitations to further ecological recovery likely exist, and include a continued lack of large, emergent and submerged boulders in riffle habitat, on-going maintenance and removal of macrophytes and debris jams, limited variety and availability of in-bank habitat, lack of useable habitat in installed fish hotels due to seasonally fluctuating water levels, continuing stormwater, sediment, and contaminant inputs, and distance from source populations (particularly for aquatic insect species).

Rehabilitation works should be continued in Christchurch’s urban waterways, with a multi-faceted and goal-oriented approach for achieving the best outcomes.

Background

The Avon River / Ōtākaro is a spring-fed waterway, which is sourced from the western suburbs of Christchurch. It flows through the middle of the city centre before discharging into the Avon-Heathcote Estuary / Te Ihutai. As part of the Christchurch Re-build, the Avon River Precinct (ARP) anchor project included selected rehabilitation works in five sites in the Avon River. This work was conducted in 2013, with a vision to return the river to a more natural state and improve ecological conditions. The rehabilitation works undertaken included both in-stream and riparian works.

In-stream rehabilitation works included the construction of riffles and vegetated floodplains, removal of fine sediments, and addition of larger boulder substrates. The riparian work included riparian planting and construction of wetland floodplains along the river.

The Avon River's catchment is predominantly urban, with the river receiving storm water and run off from heavily urbanised areas with minimal riparian zones. The river received large amounts of liquefaction during the Canterbury earthquake events, which likely worsened conditions within the already fine-silt-dominated system.

A baseline survey was conducted by Boffa Miskell Limited (Boffa Miskell) in 2013 and prior to any rehabilitation works being undertaken[1]. The sites surveyed during the baseline study included three reference sites, where no rehabilitation works were to take place, and five rehabilitation sites where rehabilitation works were planned to be undertaken. In 2014, Opus resurveyed these sites, one year after the rehabilitation works had been completed. This report presents the findings of a survey conducted three years after the rehabilitation works were completed.

Scope

The Christchurch City Council (CCC) contracted Boffa Miskell to conduct an ecological survey of the ARP sites in March 2017 (three-years post-rehabilitation). This work was part of the long-term monitoring of the ARP river sites. The survey was conducted at the same 8 sites previously surveyed during the 'baseline' and 'one-year post-rehabilitation' surveys.

The purpose of this report is to:

- Describe the current ecological conditions of the sites along the Avon River with respect to riparian and in-stream habitat conditions, and macroinvertebrates and fish communities;
- Compare conditions in reference and rehabilitation sites three-years post-rehabilitation with those from the baseline and one-year post-rehabilitation surveys; and
- Discuss:
    - Potential reasons for any significant patterns and trends recorded;
    - The current success of the rehabilitation works; and
    - Any limiting factors of the rehabilitation works.

[1] The baseline survey was conducted prior to rehabilitation works at four sites in the Avon River. However, rehabilitation work had already been completed at Rehabilitation Site 1, downstream of the Antigua Boatsheds, known as Watermark.

Survey Methods

Site Locations

Riparian and in-stream habitat conditions, and macroinvertebrate and fish communities were assessed at 8 sites along the Avon River, during base-flow conditions in March 2017. The same field methods were used in this survey as during the baseline and one-year post-rehabilitation. Sites included 3 'reference' sites upstream of the ARP and where no rehabilitation works had been conducted; and the 5 'rehabilitation' sites within the ARP (Table 1; Figure 1).

The CCC provided Boffa Miskell with GPS locations of these sites, which had been previously sampled in 2013 (baseline, Boffa Miskell 2014) and in 2014 (one-year post-rehabilitation, Opus 2015). All sites included a riffle or fast-flowing run; upstream reference sites were selected to be representative (i.e. a reasonable comparison) of the 'rehabilitation' sites downstream.

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

Site name	Site Number	Northing	Easting
Avon River downstream of Mona Vale weir	Reference Site 1	5742492	2478634
Avon River at Carlton Mill Corner	Reference Site 2	5742834	2479764
Avon River in Hagley Park	Reference Site 3	5742010	2479390
Avon River near Durham Street	Rehabilitation Site 1	5741381	2480081
Avon River at Rhododendron Island	Rehabilitation Site 2	5741385	2480253
Avon River at Hereford Street	Rehabilitation Site 3	5741648	2480397
Avon River at Victoria Square	Rehabilitation Site 4	5741998	2480483
Avon River near Kilmore Street	Rehabilitation Site 5	5742329	2481261

Figure 1. Location of the 3 reference sites and 5 rehabilitation sites surveyed, as part of the ARP project, in March 2017.

Water Quality

Spot measures of basic water chemistry (pH, dissolved oxygen, conductivity) and water temperature were collected at each site using a hand-held EXO2 Sonde s/n water-quality meter.

Riparian and In-Stream Habitat

A variety of in-stream and riparian habitat parameters were recorded at each site on 8-10 March 2017, following the standard protocols of Harding et al. (2009) and Clapcott et al. (2011):

- Protocol 3 (P3) Quantitative protocol of Harding et al. (2009):
    - P3b: Hydrology and morphology procedure[2];
    - 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.

Full details of Protocol P3 (Harding et al. 2009), and SAM2 and SAM6 (Clapcott et al. 2011), including field-sheet templates, are provided in Appendices 1-3.

In summary, these habitat assessment methods involved measuring a range of riparian and in-stream physical habitat conditions at various distances across 6 equally spaced cross-sections established across the waterway every 10 m. The first (downstream most) cross-section at each site was located at the co-ordinates provided in Table 1.

In addition, the following parameters were measured at each of the first three (downstream) transects.

Total wetted width (m) was recorded at each of the first three transects, to give an average wetted width (m) for each site. Canopy cover (%), undercut bank extent (cm) (if present), extent of any overhanging vegetation (cm), ground cover (%), and general riparian vegetation conditions were recorded on the true left (TL) and true right (TR) banks along each of these transects, at each site.

Water depth (cm), soft sediment depth (cm), substrate composition (%), macrophyte depth (cm), percent cover, type (submerged or emergent) and dominant species of macrophytes, percent cover of organic material (leaves, moss, coarse woody debris), and percent cover and type of periphyton were measured at three locations (TL bank, mid channel and TR bank) along each of the three transects, at each site.

Soft sediment depth was determined by gently pushing a metal rod (10 mm diameter) into the substrate until it hit the harder substrates underneath. Substrate composition was measured within an approximately 20 x 20 cm quadrat randomly placed at each of the three locations along the three transects. Within each quadrat, the percent composition of the following sized

[2] P3b parameters were collected by Environment Canterbury's hydrologist. This was done at the same time as the other habitat assessments were conducted.

substrates was estimated: silt / sand (< 2 mm); gravels (2 – 16 mm); pebbles (16 – 64 mm); small cobbles (64 – 128 mm); large cobbles (128 – 256 mm); and boulders (> 256 mm).

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

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 therefore 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[3]) using two complimentary methods, on 8-10 March 2017.

Five replicate Surber samples (0.05 m^2, 500 µm mesh) were collected at each of the 8 sites. Surber samples were randomly collected from shallow riffles or fast-flowing runs, and the substrate was disturbed to an approximate depth of 5 cm.

In addition, a single and extensive composite kick-net (500 µm mesh) sample was collected from each site in accordance with protocols C1 and C2 of Stark et al (2001). Approximately 0.6 m^2 of stream bed was sampled at each site (i.e. each kick net sampled approximately 0.3 m x 2.0 m of stream bed), including sampling the variety of microhabitats present (e.g. stream margin, mid channel, undercut banks, macrophytes) to maximise the likelihood of collecting all macroinvertebrate taxa present at a site, including rare and habitat-specific taxa.

All macroinvertebrate samples were preserved, separately, in 70% ethanol prior to sending to Biolive, Nelson, 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.

Fish Community

Each site was revisited between 23 and 27 March 2017 (or 3 April 2017[4]) and the fish community was surveyed from within a reach of at least 50 m (i.e. the same survey reach as habitat and macroinvertebrate community were assessed) at each site. Each survey reach included the variety of habitats typically present at that site (e.g. stream margin, mid channel, undercut banks, macrophytes, silt, riffles). Survey reaches were divided into many subsections of approximately 2-3 m in length and electro-fished using multiple passes with a Kainga EFM 300 backpack mounted electric-fishing machine (NIWA Instrument Systems, Christchurch). Fish were captured in a downstream push net or in a hand (dip) net and temporarily held in buckets. All fish were then identified, counted and measured (length, mm) before being returned alive to the stream.

The habitat where fish were found was noted (e.g. under overhanging *Carex* plants, in macrophyte beds, in mid-channel fast riffles).

[3] The macroinvertebrate community was sampled at each site on the same day that the habitat assessment was conducted (i.e. prior to habitat assessments, but after basic water chemistry and temperature parameters were measured).

[4] The fish community of Rehab 5: Avon River near Kilmore Street was surveyed on 3 April 2017, as the river maintenance crew had cleared the macrophytes from the stream on 22 and 23 March 2017.

Data Analyses

Water Quality

A single measure of pH, dissolved oxygen, conductivity, and water temperature was measured at each site, in 2013 (baseline), 2014 (one-year post-rehabilitation), and 2017 (this study, three-years post-rehabilitation). Qualitative comparisons were made to detect any substantial changes in these parameters through time since rehabilitation activities.

Riparian and In-Stream Habitat

The multiple measures across transects, and at multiple transects within a site for water depth, soft sediment, substrate composition, macrophyte depth, percent cover of macrophytes, organic materials and periphyton, were averaged to give one value for each parameter per site.

A substrate index (SI), modified from Jowett and Richardson (1990), was calculated for each measure taken across the three transects at each site, using the formula:

SI = (0.06% boulder) + (0.05% large cobble) + (0.04% small cobble) + (0.03% pebble) + (0.02% gravel) + (0.01% silt / sand)

The calculated SI can range between 1 and 6, where an SI of 1 indicated 100% silt / sand and 6 indicated 100% boulders. That is, the larger the SI, the coarser the substrate and the better the habitat for macroinvertebrate and fish communities. Finer substrates generally provide poor, and often unstable, in-stream habitat. The multiple SIs calculated for each site (i.e. multiple values across three transects at each site) were averaged, to give one value per site.

Two-way analyses of variance (ANOVAs) were used to test for differences in select habitat conditions between reference and rehabilitation sites; and through time (i.e. baseline (Boffa Miskell 2014), one-year post-rehabilitation (Opus 2015), and three-year post-rehabilitation (this study)). The interaction term between 'rehabilitation treatment' and 'year' was also tested, to examine how rehabilitation works might have influenced parameters over time.

Response variables were $log(x+1)$ transformed where necessary to meet assumptions of normality and homogeneity of variances. ANOVAs were performed in R version 3.0.2 (The R Foundation for Statistical Computing 2013).

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 present 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)** – 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)** – 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.

ANOVAs were used to test for differences in averages[5] (1) between treatments (reference and rehabilitation sites); (2) among years (2013, 2015, and 2017); 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 3.0.2 (The R Foundation for Statistical Computing 2013).

A non-metric multidimensional scaling (or NMDS) ordination[6], with 1000 random permutations, using abundance data (averages from Surber samples) was used to determine if the macroinvertebrate community found was similar among the 8 sites surveyed, between reference and rehabilitation sites, and through time (i.e. baseline, one-year, and three-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 reference and rehabilitation sites; and among baseline (Boffa Miskell 2014); one-year post-rehabilitation (Opus 2015); and three-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[7].

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

[5] Averages were calculated using only Surber sample data.

[6] 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).

[7] 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).

(baseline, one-year, and three-years post-rehabilitation), similarity percentages (SIMPER) were calculated[8] to show which macroinvertebrate taxa were driving these differences.

NMDS, ANOSIM and SIMPER analyses were performed in PRIMER version 6.1.13 (Clarke and Warwick 2001; Clarke and Gorley 2006).

Fish Community

The total distance fished (in metres) at each site and the amount of time spent actively fishing (i.e. time displayed on the electro-fishing machine) were recorded. The fish capture data were then expressed as 'catch per unit effort' (CPUE), to standardise for differences in sampling effort among sites (i.e. total distance). CPUE was calculated by dividing the number of fish captured by the total area fished (i.e. total distance fished multiplied by average wetted width of a site), and extrapolated up to 100 m^2 for each site. CPUE was, therefore, expressed as number of fish captured per 100 m^2.

ANOVAs were used to test for differences in averages (1) between treatments (reference and rehabilitation sites); (2) among years (2013, 2015, and 2017); and (3) the interaction between treatment and year, in abundance (CPUE) 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 3.0.2 (The R Foundation for Statistical Computing 2013).

An NMDS ordination, with 1000 random permutations, using abundance data was also used to determine if the fish community found was similar among the 8 sites surveyed, between reference and rehabilitation sites, and through time (i.e. baseline, one-year, and three-years post-rehabilitation).

An analysis of similarities (ANOSIM), with 100 permutations, was then used to test for significant differences in fish community composition: between reference and rehabilitation sites; and among baseline (Boffa Miskell 2014); one-year post-rehabilitation (Opus 2015); and three-years post-rehabilitation (this survey).

If ANOSIM revealed significant differences in fish community composition (i.e. $R \neq 0$ and $P \leq 0.05$) between treatments (reference and rehabilitation sites), or among years (baseline, one-year, and three-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 6.1.13 (Clarke and Warwick 2001; Clarke and Gorley 2006).

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

Ecological Conditions

Site Descriptions

Reference Site 1: Avon River downstream of Mona Vale weir

Reference Site 1 was the most upstream site surveyed along the Avon River and was located downstream of the Mona Vale weir. At this site the river was, on average, 8.8 m wide and 23 cm deep, with a velocity of 0.4 m / s on the day of sampling. The site was largely run habitat with the true left (TL) side being significantly deeper along the base of the boulder bank. Only 15% of the stream bed was covered in macrophytes, however, there was high cover (37%) of organic matter present.

The TL bank was within residential housing with gardens extending to the water's edge and retaining walls along parts with scattered flax and *Carex secta*. The upstream extent of the TL bank was dominated by large boulders, likely placed there to prevent bank erosion at the end of Wood Lane. The true right (TR) bank was within the Christchurch Girls' High School grounds and was well vegetated with *Carex secta*, flaxes and other indigenous plantings and grasses, providing many areas of undercut banks. The submerged alga *Nitella hookeri* and macrophyte curly pondweed (*Potamogeton crispus*) were present at the site. Macrophyte cover was lower in 2017 than previous surveys. Filamentous algae were not found at this site, with only thin green recorded (13%) at this site.

The river bed was dominated by smaller cobbles and pebbles giving an overall average SI of 2.5. These substrates were only slightly embedded (average embeddedness score of 2) and moderate to loosely packed (average compactness score of 2).

Photo 1: Reference Site 1 – Avon River downstream of Mona Vale weir, looking upstream (left) and downstream (right).

Reference Site 2: Avon River at Carlton Mill Corner

Reference Site 2 was the second most upstream site surveyed along the Avon River and was located at the Carlton Mill corner on the north-western corner of Hagley Park. This site was, on average, 12.3 m wide and 26 cm deep, with a velocity on the day of sampling of 0.4 m / s.

The site was a combination of run and riffle habitat. Overall, the site was relatively shallow with a small, fast-flowing but deep area on the TL) side, under the road bridge. The average macrophyte cover was 24%[9]; organic cover was 19%.

At this site, the TL bank included a grass strip and footpath, and then the road. The riparian margin was dominated by grasses, with scattered *Carex* plants. The TR bank was also dominated by grasses with several flax and *Carex*. A large portion of the upstream extent of this site (approximately 20 m) was under the road bridge where the concrete footings of the bridge extended down the stream bed.

The only macrophyte present at this site was curly pondweed, which occurred in patches across the stream bed. There were no filamentous algae present at this site with only 20% coverage of thin green algae being recorded. The substrate here was dominated by cobbles and pebbles giving an overall SI of 2.3. These substrates were only slightly embedded (embeddedness score of 2) and moderate to loosely packed (compactness score of 2).

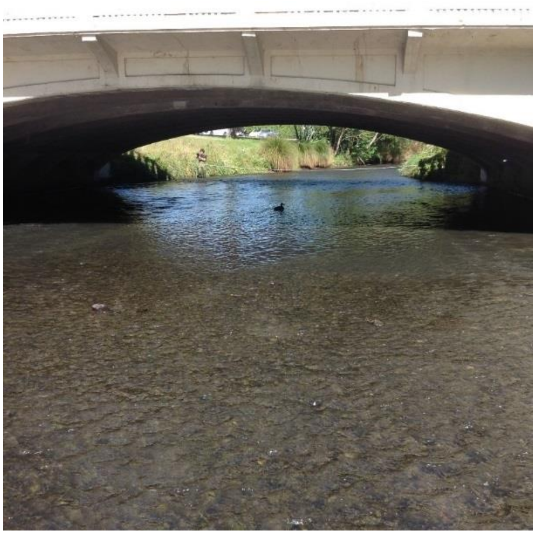


Photo 2: Reference Site 2- Avon River at Carlton Mill Corner looking downstream (left) and upstream (right).

[9] The macrophytes may have been cleared from this site prior to our survey, as cover was estimated at 100% in December 2016.

Reference Site 3: Avon River in Hagley Park

Reference Site 3 was the most downstream of the reference sites, located within the Botanic Gardens, Hagley Park. On average, this site was 11.8 m wide and 23 cm deep, with a velocity of 0.35 m / s on the day of sampling.

Average macrophyte cover was just 6% across the site, all of which was the submerged macrophyte, curly pondweed. There was little organic material present in the stream, with only 5% organic cover being recorded. The TL bank was largely dominated by grasses and *Carex* plants with several smaller trees along the reach. The TR bank was confined by a retaining wall made from wooden planks, to the height of approximately 1.5 m above water level. Above and behind the wooden wall were grasses and larger *Carex* plants, which extended over the wall and provided the stream with areas of overhanging vegetation. There were also larger exotic trees scattered along the wider riparian zone throughout the reach, which provided some shading to the river.

The SI at this site was 2.2, the lowest across all reference and rehabilitation sites. The substrates were dominated by pebbles that were only slightly embedded (embeddedness score of 2), with similar compactness to the other reference sites (score of 2).


Photo 3: Reference Site 3 – Avon River in Hagley Park looking upstream (left) and downstream (right).

Rehabilitation Site 1: Avon River near Durham Street

Rehabilitation Site 1 was the most upstream of the rehabilitation sites, located near Durham Street and downstream of the Antigua Boatsheds, and within the Anchor Project of Watermark. The site was, on average, 9.9 m wide and 37 cm deep, with a velocity on the day of sampling of 0.28 m / s.

The river was largely run habitat, with a few deeper parts along the survey reach. Average macrophyte cover was 30% and organic cover of 15%. The TL bank was dominated by grasses, with a few larger exotic trees further from the water's edge. The upper part of the reach included an area of constructed floodplain wetland, which was planted with indigenous species including *Carex*, sedges, flaxes and ferns. The constructed wetland was approximately 20 m long, which equated to around 30% of the entire survey reach. The TR bank was entirely constructed wetland with boulders and indigenous plantings. There were a few large chestnut trees further back from the water's edge, which provided shading to the river channel. Over time the plantings undertaken as part of the rehabilitation[10] works have formed a relatively dense riparian buffer, particularly on the TR bank (Photo 3).

The macrophytes present at this site were Canadian pondweed (*Elodea canadensis*), and curly pondweed, and the alga *Nitella hookeri*. These macrophytes occurred in large beds within the river channel. Macrophytes were not recorded as being present in the baseline survey, however, were recorded as relatively high coverage in the Year 1 survey (56%) (Opus 2015). It's important to note that macrophytes are regularly cleared from waterways within Christchurch City, and the baseline study was conducted after this activity had occurred.

Filamentous algae were present at this site, with an average cover of 13.5%. Filamentous algae cover has not changed over the three surveys (2013, 2014, 2017) at this site.

The substrate at this site was dominated by cobbles and pebbles giving an overall SI of 2.8. Substrate composition has remained similar at this site over time. These substrates were only slightly embedded (embeddedness score of 2) and moderate to loosely packed (compactness score of 2).

Photo 4: Rehabilitation Site 1 - Avon River near Durham Street looking upstream (left) and downstream (right).

[10] It is important to note that the rehabilitation works at Rehabilitation Site 1 – Avon River near Durham Street had been completed 5 months prior to the baseline survey took place. Therefore, caution should be used when comparing 2014 and 2017 data to the baseline (2013) information.

Rehabilitation Site 2: Avon River at Rhododendron Island

Rehabilitation Site 2 was the second most downstream of the rehabilitation sites, and was located upstream of Rhododendron Island and immediately downstream of the recently built Canterbury Earthquake National Memorial. At this site the river was, on average, 10.2 m wide and 32 cm deep with a velocity on the day of sampling of 0.32 m / s.

The upstream extent of the reach had very deep water with large boulders on the stream bed. A riffle was constructed, as part of the rehabilitation works, approximately halfway along the study reach. This continues downstream eventually becoming deeper and forming run habitat. The TL bank had a constructed floodplain along the entire reach, with boulders placed along the toe and indigenous plantings, overlaying weed matting, extending out to a grassed and paved walking / recreation area. The downstream extent of the TR bank had boulders along the toe, and a garden had been recently planted with indigenous species. The upstream extent of the TR bank was the recently completed Canterbury Earthquake National Memorial (Photo 5), which included hollow spaces and boulders at the water's interface and overhanging the river channel. However, at the time of surveying, the river's water level was below these spaces and so did not provide obvious useable aquatic habitat.

The only macrophyte present at this site was curly pondweed, which occurred in large beds throughout the reach, particularly in the deeper water. The substrate here was dominated by cobbles and the large boulders in the upstream extent of the reach, which gave an overall SI of 3.4. The SI calculated in 2017 was markedly greater than that recorded in the baseline survey, (baseline survey SI: 1.4). These substrates were only slightly embedded (embeddedness score of 2) and moderate to loosely packed (compactness score of 2).

Total sediment cover was high in 2013 (baseline survey), but extremely low in 2014 (Year 1 survey). There was slightly more sediment cover in 2017, than 2014, which could be (in part) attributed to the recent construction works associated with the Canterbury Earthquake National Memorial completed early 2017.

Several PVC pipes were observed underneath the memorial wall. These were thought to be a combination of stormwater pipes draining nearby tree pits and any seepage under the wall, as well as those constructed as "fish hotels". In March 2017, the river (water levels) were below many of these pipes, however, when this site was revisited in May 2017 the "fish hotels" were partially submerged (Photo 5).

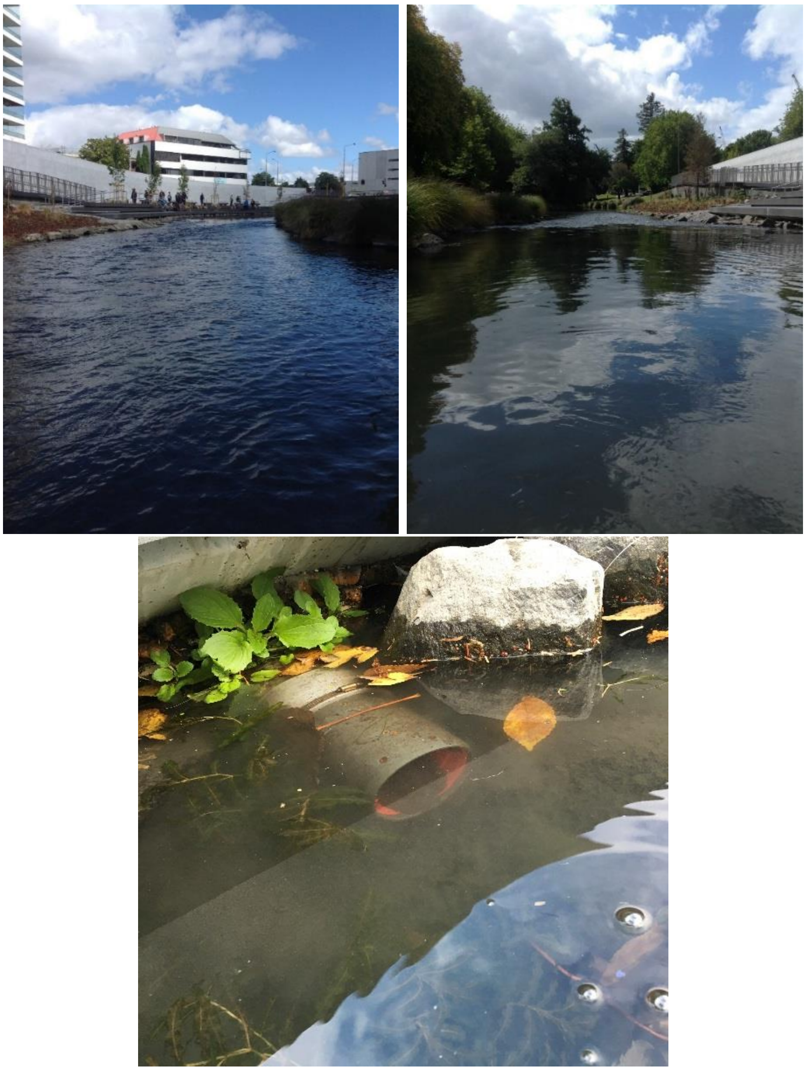


Photo 5: Rehabilitation Site 2 - Avon River at Rhododendron Island looking upstream toward the Canterbury Earthquake National Memorial (top left) and downstream (top right); and pipes used to create "fish hotels" underneath the memorial wall (bottom). Note, the photo of the "fish hotel" pipe was taken in May 2017, when water levels were much higher than during the survey of March 2017.

Rehabilitation Site 3: Avon River at Hereford Street

Rehabilitation Site 3 was located at the Hereford Street Bridge, immediately upstream of Mill Island. At this site, the river was, on average, 14.2 m wide (the widest of all reference and rehabilitation sites) and 29 cm deep, with a velocity of 0.28 m / s on the day of sampling. The main flow type at this site was run habitat, with a deep pool located at the upstream extent of the sample reach.

The first three transects (i.e. the downstream most 20 m) of the sample reach were under the Hereford Street Bridge. For the remainder of the site, the riparian zone on the TL side was a constructed wetland with indigenous plantings, and boulders placed along the toe. There were some larger exotic trees that provided shade to the river channel. The TR bank also had a constructed wetland with boulders along the toe, but there was a much narrower strip of planting, than on the TL, due to the steep bank leading up to the road. At the upstream extent of the survey site, large steps and a formal access area of “The Terraces” has been constructed along the TR bank.

There were only small patches of macrophytes at this site, dominated by curly pondweed. Macrophyte abundance was lower in 2017, than previous years. Both brown and green (short and long) filamentous algae were present at this site, along with think brown and thick green mats of algae.

The substrate at this site was dominated by cobbles, with an overall SI of 3.1. This was similar to previous surveys. These substrates were only slightly embedded (embeddedness score of 2) and moderate to loosely packed (compactness score of 2).

Photo 6: Rehabilitation Site 3 - Avon River at Hereford Street looking upstream (left) and downstream under the Hereford Street Bridge (right).

Rehabilitation Site 4: Avon River at Victoria Square

Rehabilitation Site 4 was located upstream of Victoria Square, and immediately upstream of the Armagh Street Bridge. At this site, the river was the deepest of all sites, with an average water depth of 45 cm. On average, the site was 9.1 m wide, with a velocity of 0.3 m / s on the day of sampling. The main flow type at this site was run habitat, with deeper pools at both the downstream and upstream extent of the survey reach.

Much of the TL bank had dense plantings for approximately 3 m from the water's edge up to the old Provincial Court building. Boulders lined the toe of the TL bank. The upstream extent of the TL bank had large, established *Carex* grasses, which overhung the river. The TR bank also had large, established *Carex* grasses planted along the water's edge, and overhanging the river. Outside of this immediate riparian margin, mown grass led to Oxford Terrace. A constructed wetland was located within the downstream extent of the TR bank, with boulders along the toe.

Macrophyte cover was high at this site, with an average cover of 64%, and dominated by large clusters of curly pondweed and the macroalga *Nitella*, in the middle of the river channel. Patches of watercress (*Nasturtium officinale*) were present along the margins. Long green filamentous algae, while present on exposed cobbles along the margins, were not abundant, with an average cover of 2%.

The substrate at this site was dominated by cobbles, with some boulders, with an average SI of 3.5. This was greater than the SI calculated for this site in 2013 and 2014. The substrates were only slightly embedded (embeddedness score of 2) and moderate to loosely packed (compactness score of 2). Sediment depth was greatest in 2017, compared to the previous surveys[11].

Photo 7: Rehabilitation Site 4 – Avon River at Victoria Square looking upstream (left) and downstream (right).

[11] However, this may in part be due to a couple of measures of very high sediment depth within the site. Generally, sediment depth was low.

Rehabilitation Site 5: Avon River near Kilmore Street

Rehabilitation Site 5 was the most downstream site and was located downstream of the Kilmore Street Bridge, and of the Firefighters' Memorial. Here, the river was, on average, 11.9 m wide and 33 cm deep, with a velocity of 0.39 m / s on the day of sampling. The main flow type at this site was run habitat.

Constructed wetlands were located along both the TL and TR banks, with boulders along the toe. The wetland vegetation extended to a grassed roadside verge on both sides. There were several large oak trees scattered along the reach, which provided shading to the river.

This site had a macrophyte cover of 34%. Macrophytes at this site were largely dominated by large clusters of curly pondweed, with smaller areas of the macroalga *Nitella hookeri* through the mid channel. Macrophyte cover appears to be temporally variable at this site, with high macrophyte cover recorded during the baseline survey, but no macrophytes found in 2014 (Year 1 survey). However, this is almost certainly due to the macrophyte maintenance in the Avon River. Macrophytes were cleared from this site before sampling commenced in 2014, but were cleared after the survey in both 2013 and 2017. Long green, and short and long brown filamentous algae were present at this site. Thin and thick brown algae were also present.

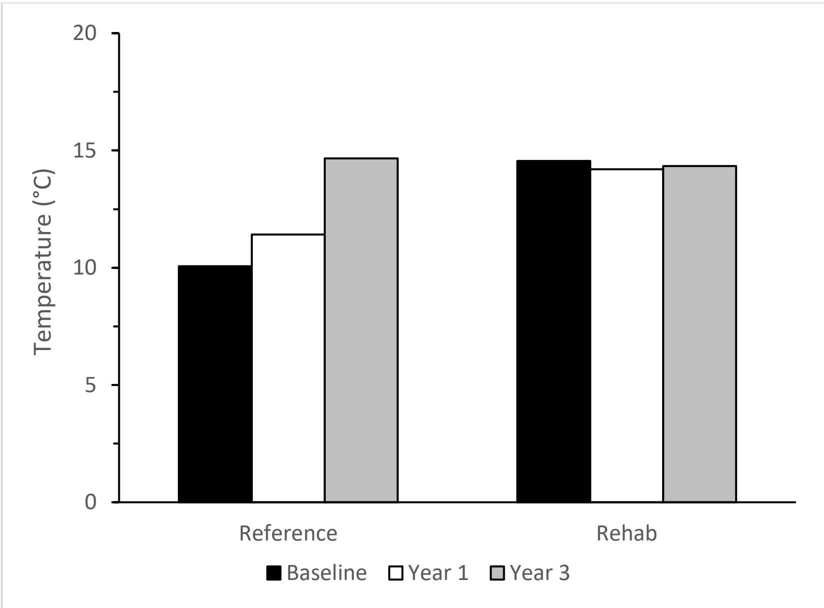
The substrate was dominated by large cobbles and boulders and gave the highest overall SI across all sites of 4.3. The SI calculated for this site in 2017 was markedly greater than that calculated in 2013 (SI of 1.3) but similar to that measured in 2014 (SI of 3.7). The substrates were only slightly embedded (embeddedness score of 1) and moderate to loosely packed (compactness score of 2).

Photo 8: Rehabilitation Site 5 – Avon River at Kilmore Street looking upstream (left) and downstream (right).

General Habitat Conditions

Water Quality

Water temperature was lower (cooler) in the reference sites in 2013 (baseline) and 2014 (one-year post-rehabilitation), than 2017 (three-years post-rehabilitation), and in the rehabilitation sites more generally (Figure 2). However, it's important to note that water temperature was only measured on one occasion at each site, on each sampling occasion, and that it can fluctuate both daily and seasonally.



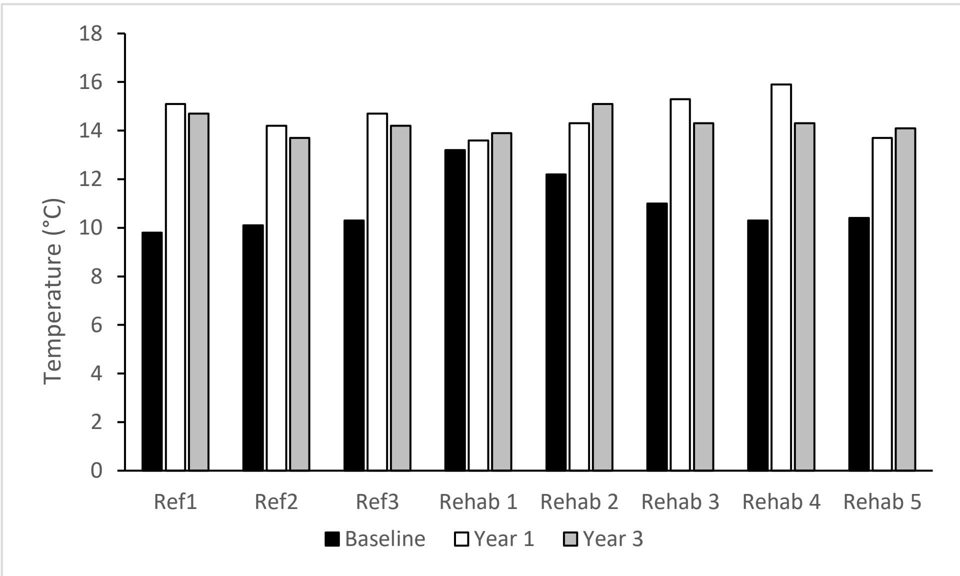
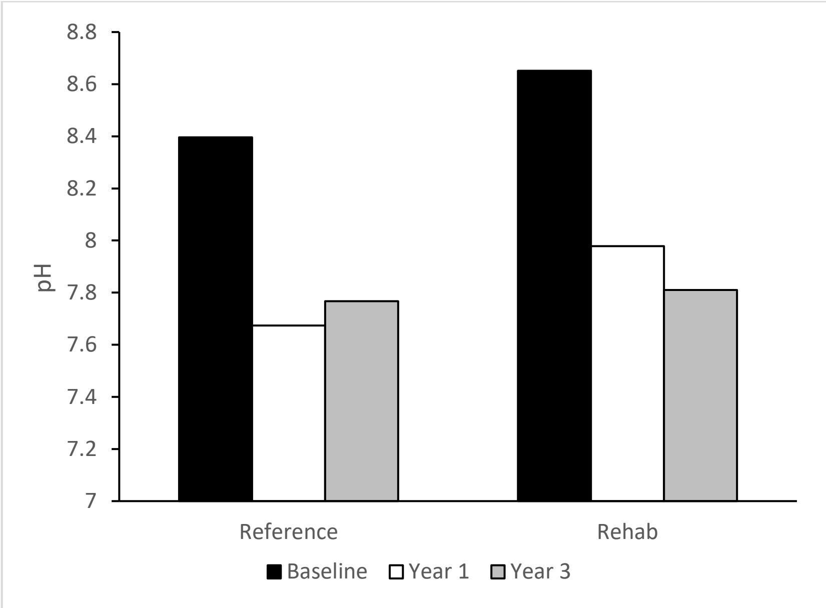


Figure 2: Water temperature (°C) measured at the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

pH was greatest in 2013 (baseline), while pH was more similar (and circum-neutral) in 2014 (Year 1) and 2017 (Year 3) in both reference and rehabilitation sites (Figure 3). However, on all three survey occasions, pH was circum-neutral (to slightly alkaline) and likely within parameters tolerant of most aquatic fauna.



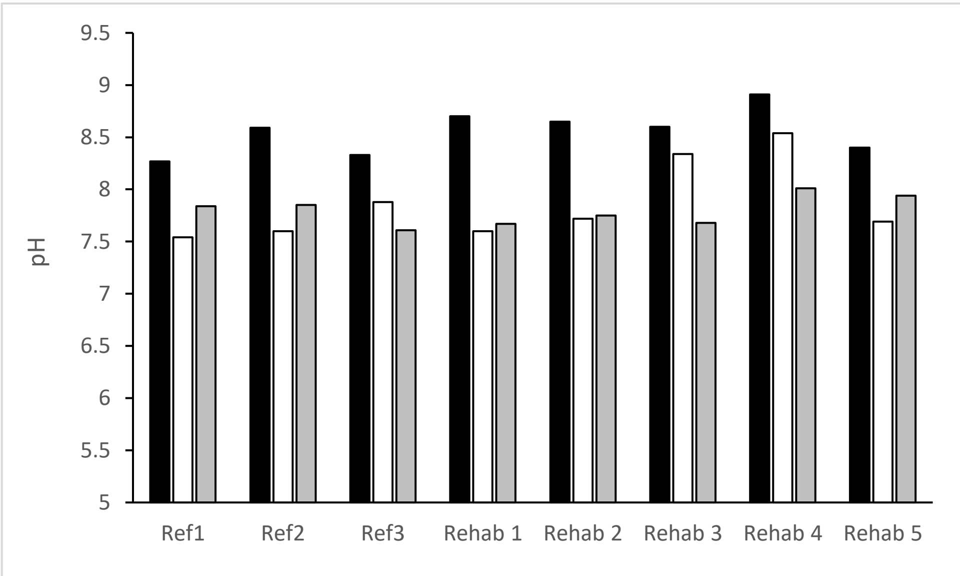
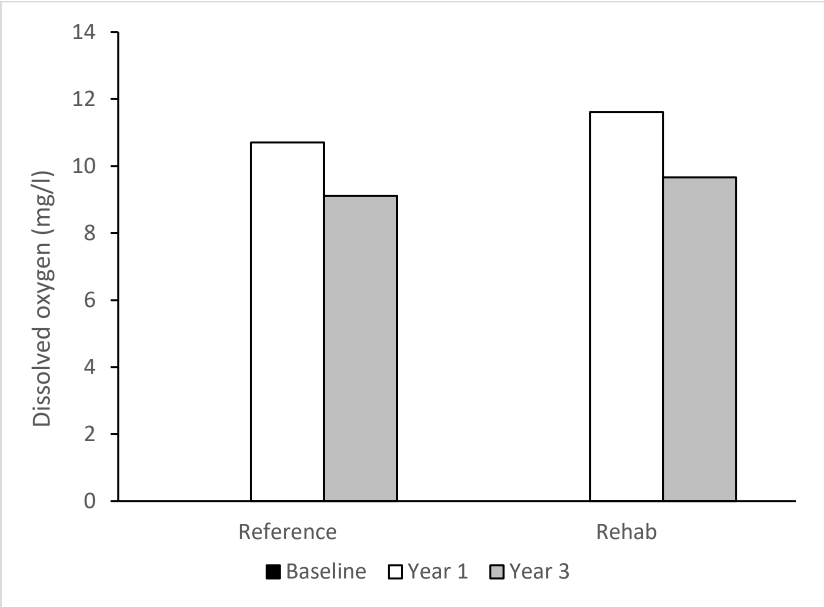


Figure 3: pH measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

Dissolved oxygen (DO) was marginally lower in 2017 than 2014, but no difference was observed between reference and rehabilitation sites (Figure 4). DO was not measured in 2013 (baseline). It's important to note that dissolved oxygen was only measured on one occasion at each site, on each sampling occasion, and that it can fluctuate both daily and seasonally.



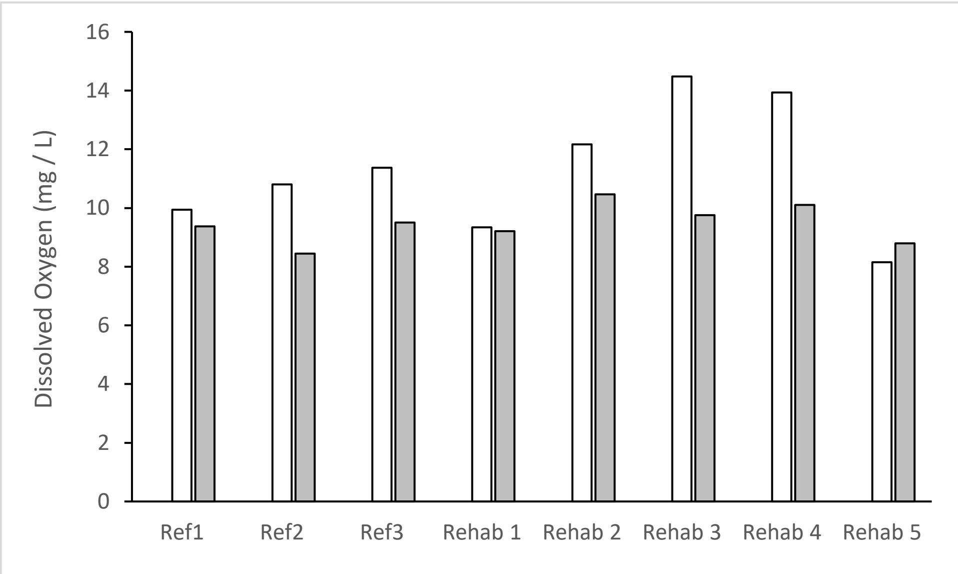
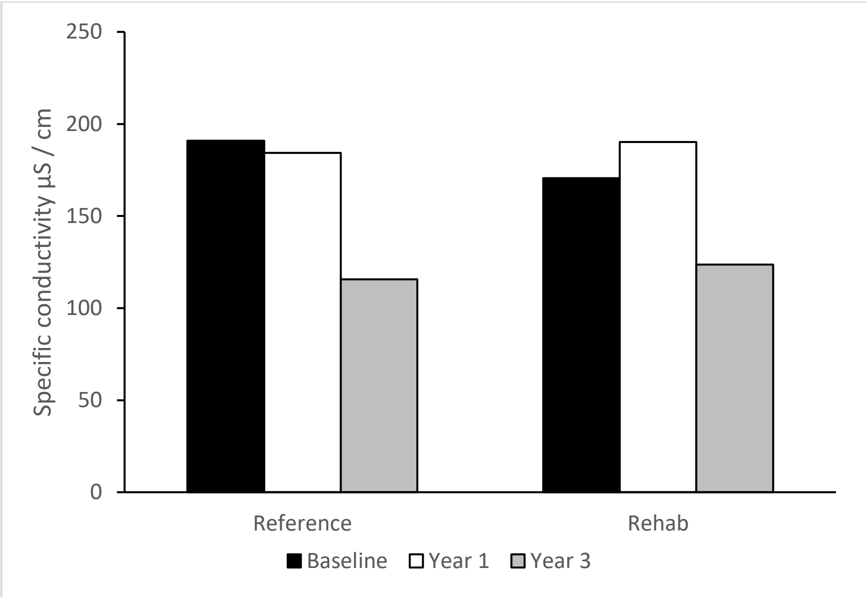


Figure 4: Dissolved oxygen (mg/L) measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

Specific conductivity was similar in most sites between baseline and one-year post-rehabilitation, but was lower in 2017 (three-years post-rehabilitation) (Figure 5). Although variable through time, conductivity measured on all three occasions was comparable to levels measured in urban streams around Christchurch. Moreover, it's important to note that conductivity was only measured on one occasion at each site, on each sampling occasion, and that it can fluctuate both daily and seasonally.



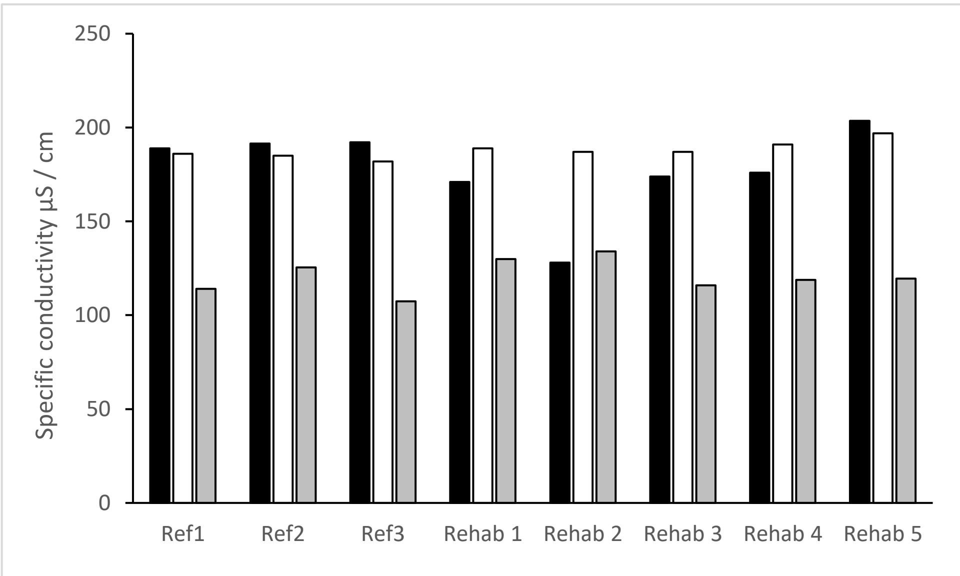
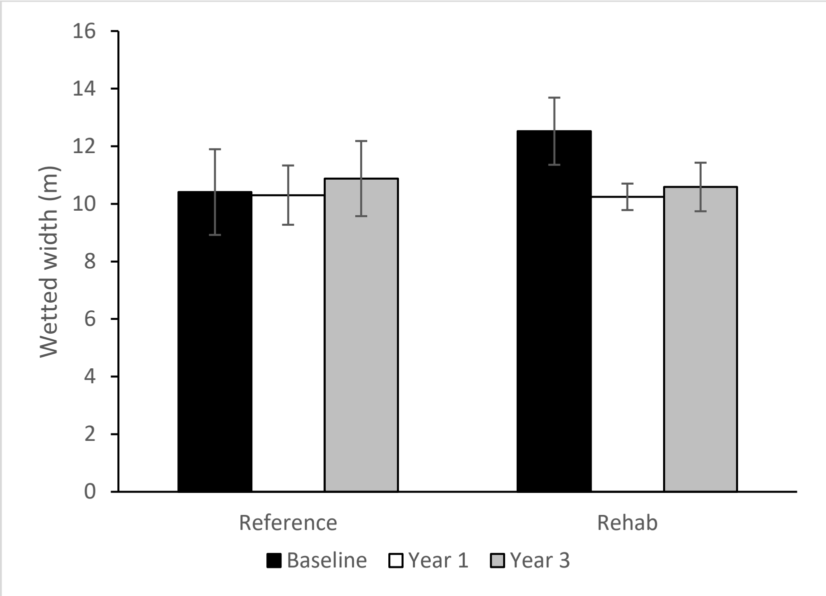


Figure 5: Specific conductivity measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

Riparian and In-Stream Habitat

Overall, rehabilitation activities did not greatly influence the average wetted width of the river (i.e. there was no statistically significant difference between average wetted width in reference or rehabilitation sites, average width 10.5 m and 11.1 m, respectively) ($F_{1,18}$ = 0.463, P = 0.505). Wetted width also didn't differ among years ($F_{2,18}$ = 1.080, P = 0.361) (Figure 6).



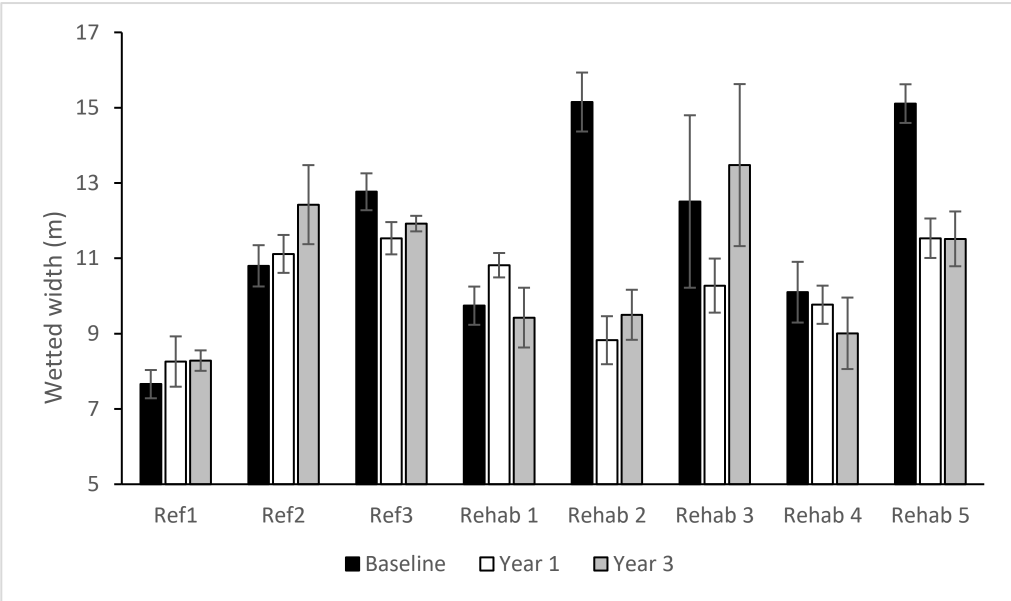
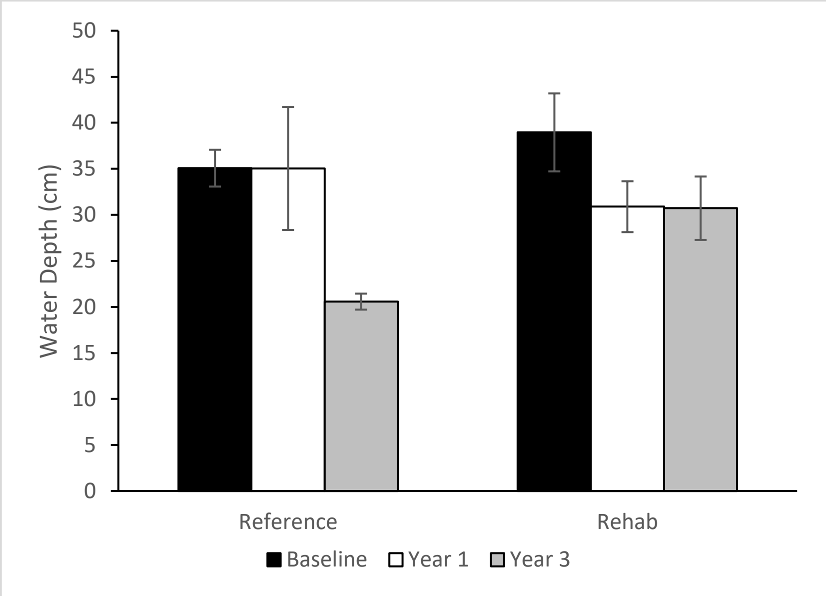


Figure 6: Average (±1SE) wetted width (m) measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars). Note the Note the width data was supplied by Environment Canterbury.

Average water depth was also found to be similar between reference (average 30.2 cm) and rehabilitation (average 33.2 cm) sites ($F_{1,18}$ = 1.053, P = 0.318), but decreased over time, with shallower water depths measured in 2017 (average 26.9 cm) compared to that measured in 2013 (baseline; average 37.5 cm) and 2014 (average 32.4 cm) ($F_{2,18}$ = 3.863, P = 0.040) (Figure 7).



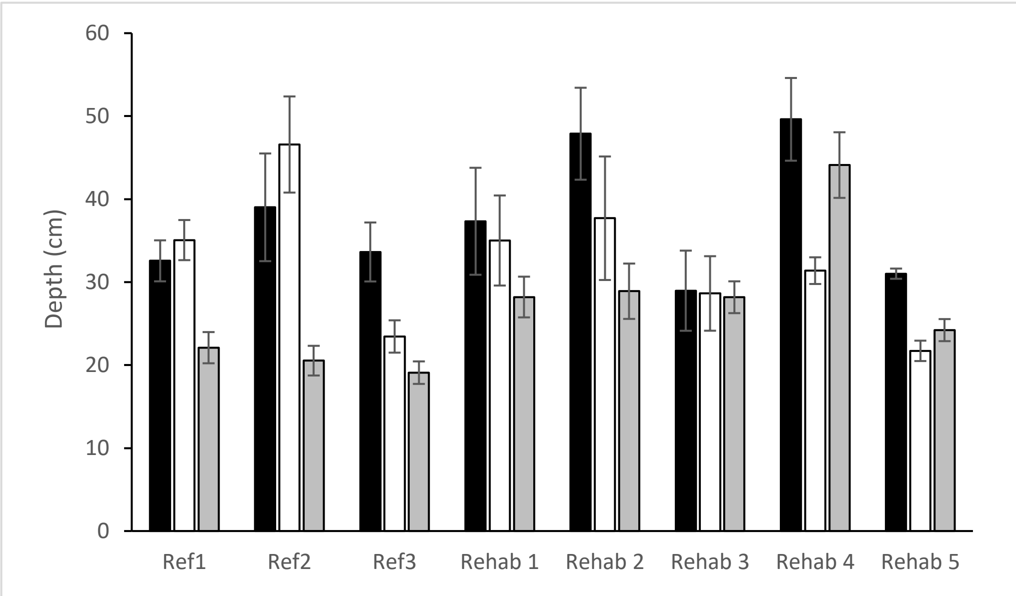
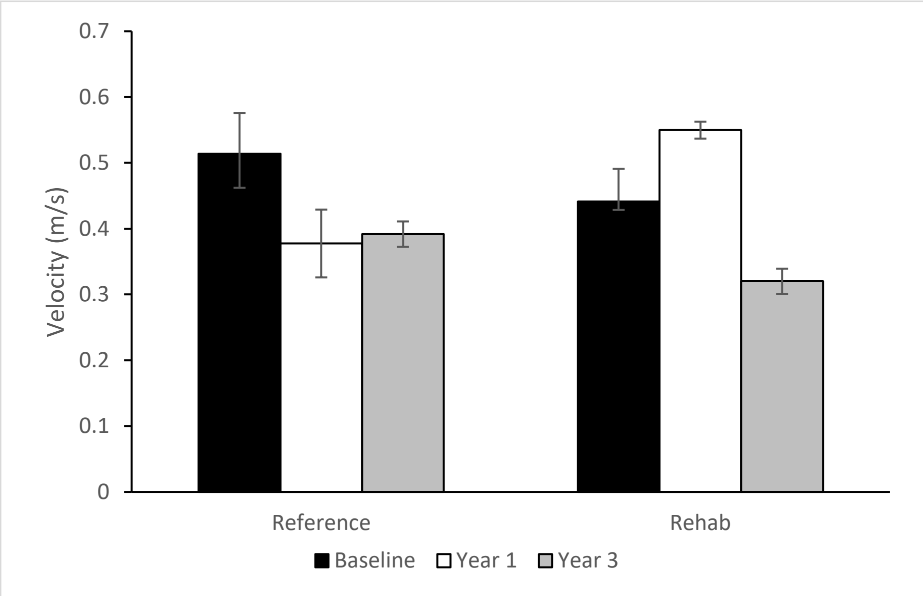


Figure 7: Average (±1SE) water depth (cm) measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

Velocity was found to be variable across sites and through time (Figure 8). However, there was no significant difference in average velocity measured at reference and rehabilitation sites ($F_{1,18}$ = 0.060, P = 0.810). Average velocity was significantly different among years ($F_{2,18}$ = 8.060, P = 0.003), where the average velocity recorded in 2017 (0.34 m / s) was lower than that recorded in both 2013 and 2014 (0.46 m / s and 0.48 m / s, respectively). Noting that water depth was also lower in 2017, than previous years, it is possible that these lower water levels influenced average velocity in 2017. The significant treatment:year interaction effect ($F_{2,18}$ = 6.649, P = 0.007) highlighted that average velocity at reference sites generally was similar in 2014 and 2017, but greater in 2013 (baseline). While average velocity was greatest in 2014, and lowest in 2017, in rehabilitation sites. It's difficult to ascertain if this is due to rehabilitation works or a result of slight differences in sampling locations (i.e. minor differences in transect locations) within each site among years.



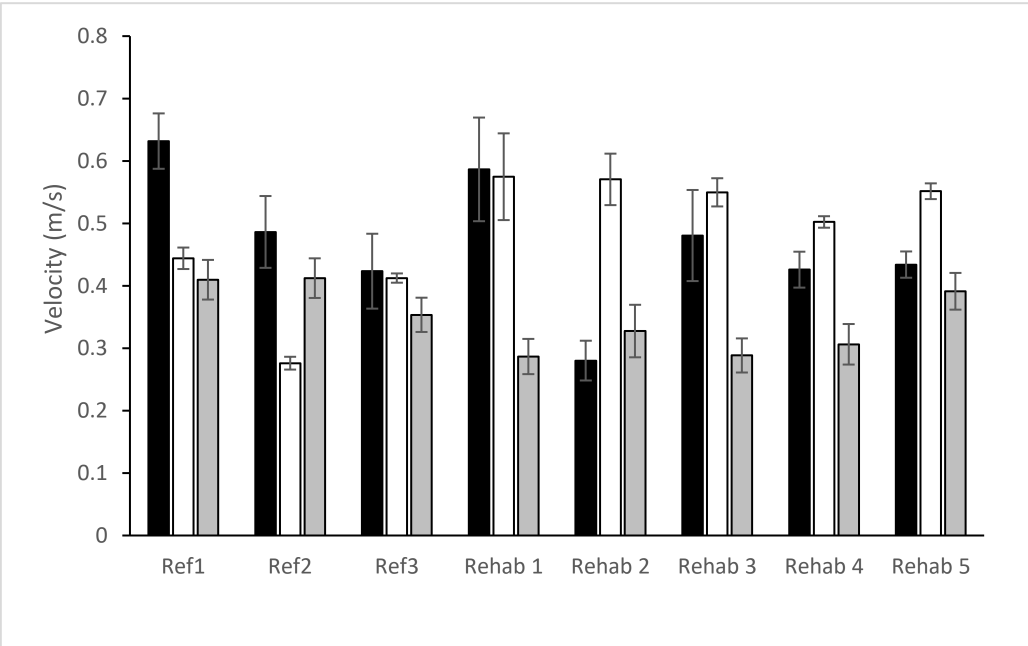
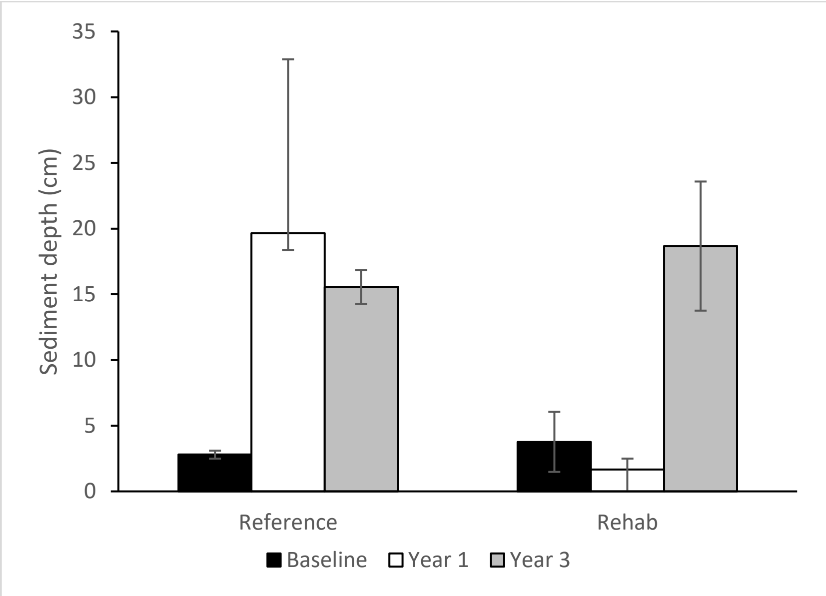


Figure 8: Average Velocity (±1SE) measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

Sediment depth was significantly greater at reference sites, than rehabilitation sites ($F_{1,18}$ = 7.231, P = 0.015) (Figure 9), with average depths of 8.2 cm in reference sites and 2.2 cm in rehabilitation sites. While there was no statistically significant difference among sediment depths measured in 2013, 2014, and 2017 ($F_{2,18}$ = 1.261, P = 0.307), sediment depth was slightly greater (but highly variable across sites) in 2014. This was likely due to slight differences in measurement methodology[12].



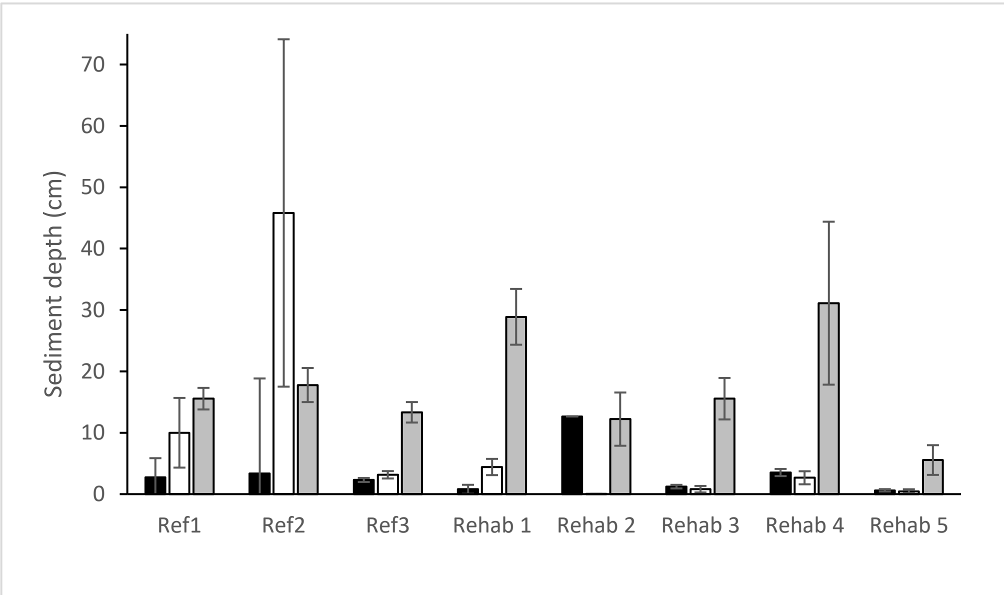


Figure 9: Average (±1SE) sediment depth (cm) measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

[12] Sediment depth during the baseline (2013) and this study (2017) was measured by gently pushing a 10 mm wading rod through the soft layer of sediment until hard substrate was reached below. Where macrophytes were present, sediment depth underneath the macrophytes was measured. A similar method was used in 2014 (Opus 2015), however, where macrophytes covered the area to be surveyed, sediment depth was not measured.

Sediment cover was highly variable across sites, and over the different surveys (Figure 10Figure 10) and although there was slightly greater sediment cover (on average) in reference sites (28% cover), than in the rehabilitation sites (18% cover), this difference was not statistically significantly different ($F_{1,18} = 2.278$, P = 0.149). Sediment cover was estimated to be slightly lower in 2014, but there was no significant effect of year overall ($F_{2,18} = 2.489$, P = 0.111).

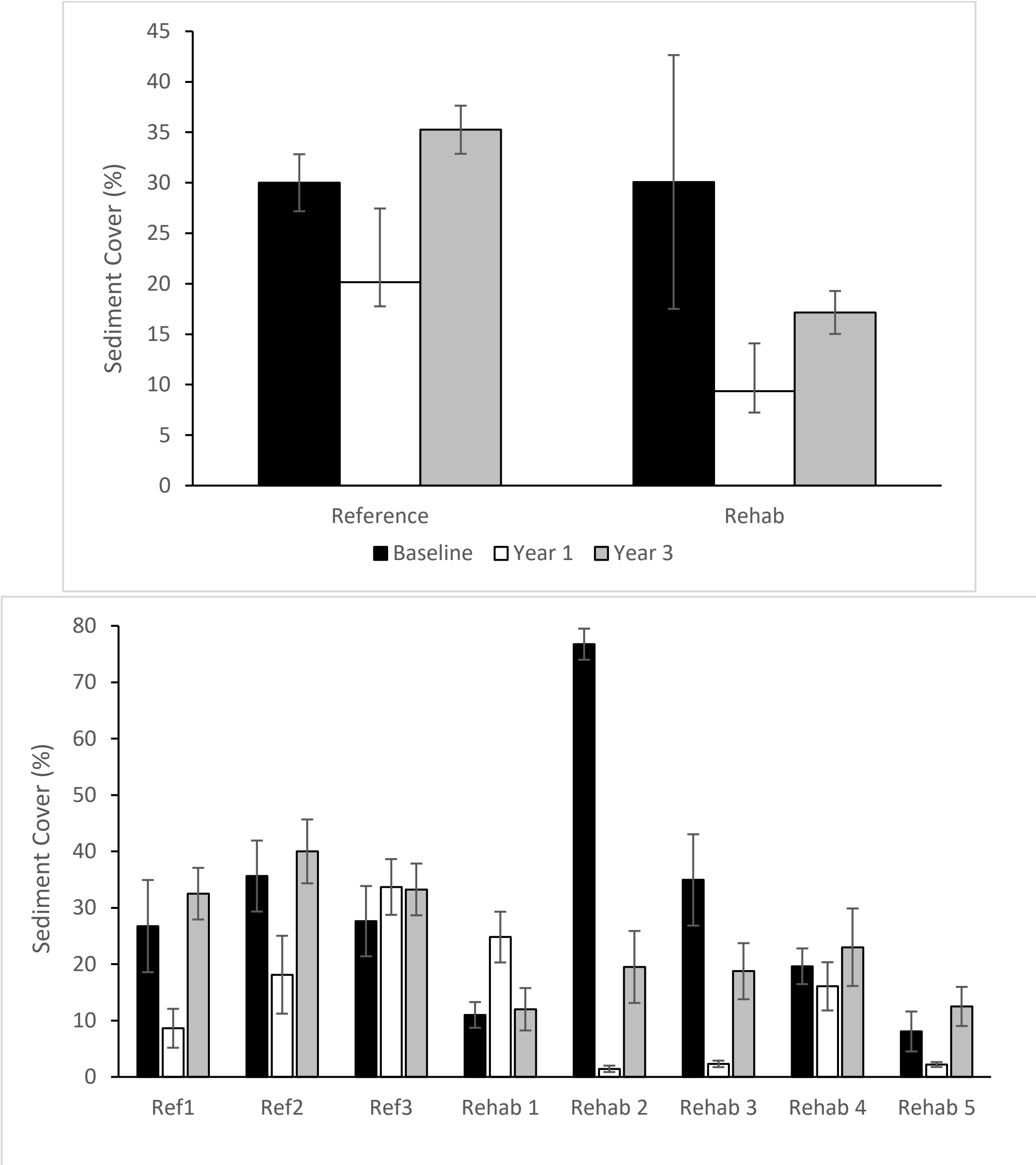
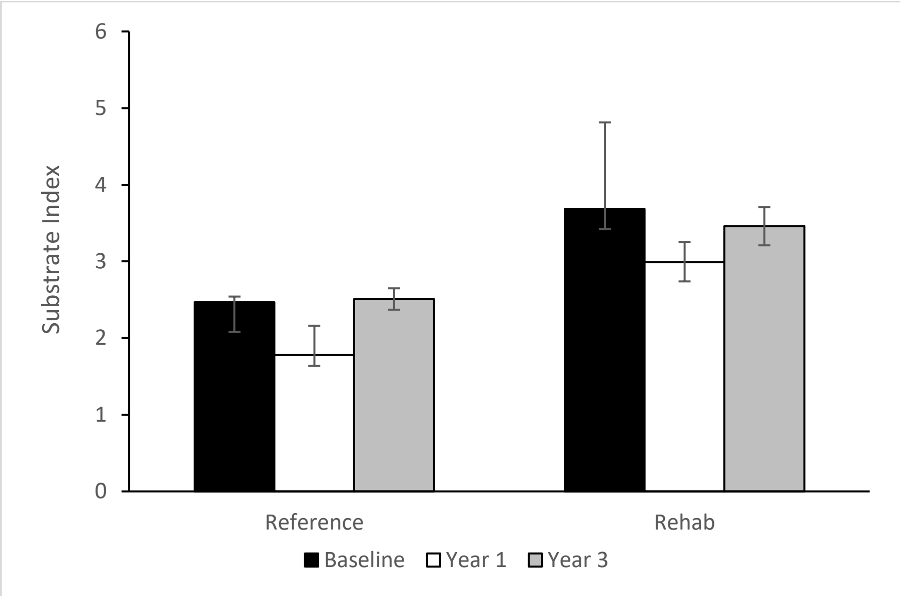


Figure 10: Average (±1SE) sediment cover (%) measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

The Substrate Index was significantly greater in rehabilitation (average of 2.2), than reference sites (average of 3.3) ($F_{1,18}$ = 9.138, P = 0.007), indicating that rehabilitation sites had coarser substrates present (Figure 11). While there were subtle differences through time, where substrate index at rehabilitation sites increased slightly with time, these differences were not statistically significant (year: $F_{2,18}$ = 1.590, P = 0.231; treatment:year interaction: $F_{2,18}$ = 2.816, P = 0.086) (Figure 11).



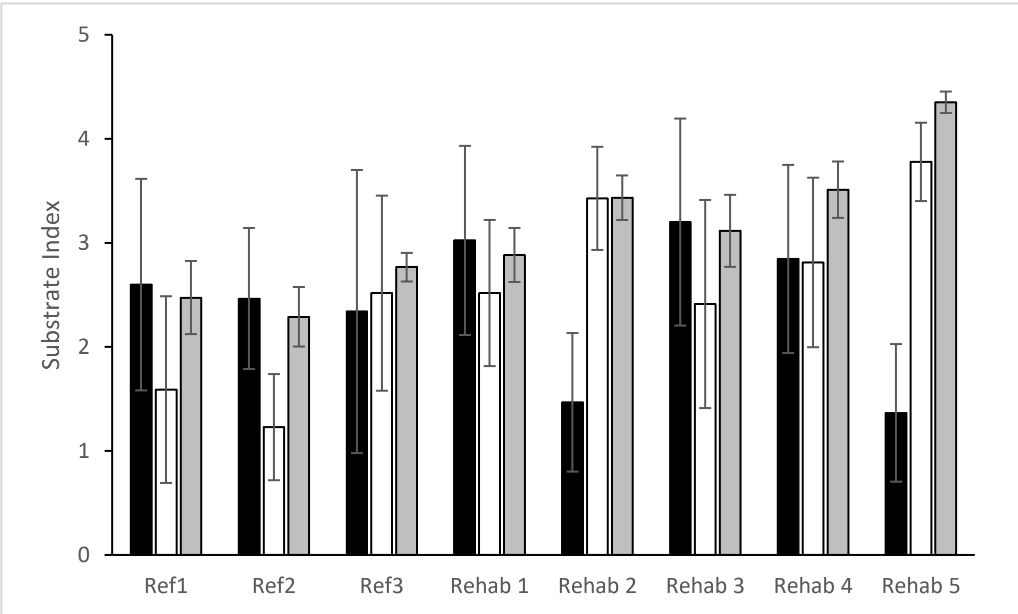
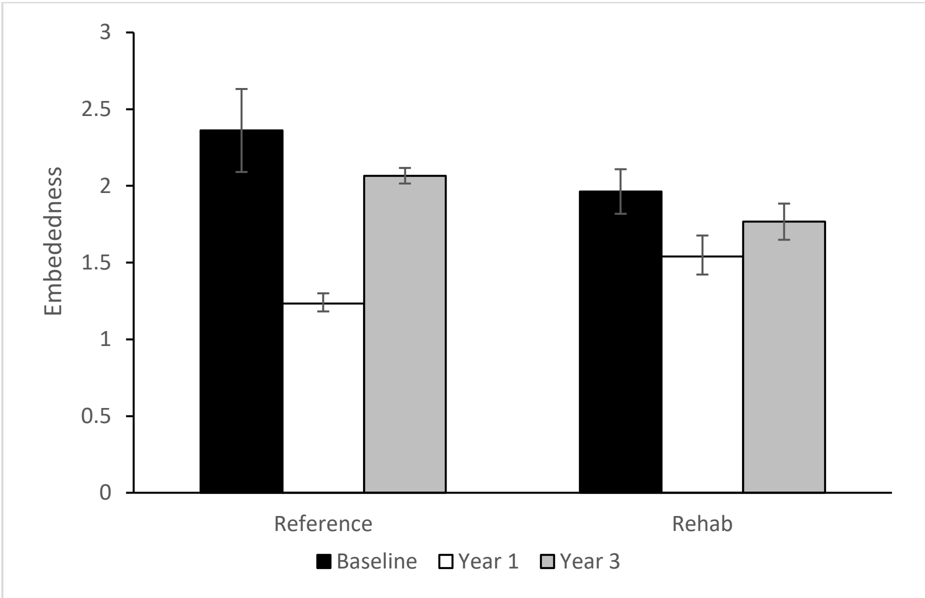


Figure 11: Average (±1SE) Substrate Index measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

Embeddedness, which is the degree to which coarse particles are surrounded by fine particles and indicates the availability of interstitial spaces between coarser particles, was not different between reference and rehabilitation sites ($F_{1,18}$ = 1.102, P = 0.308). There was a statistically significant effect of year, where embeddedness measures in 2014 where lower than those in 2013 (baseline) and 2017 (this study) ($F_{2,18}$ = 11.34; P <0.001) (Figure 12). However, there was a slight difference in the way embeddedness was measured in 2013 (baseline) compared to the other years[13]. Moreover, these differences in average embeddedness estimated in the three survey years may not equate to biologically relevant differences.



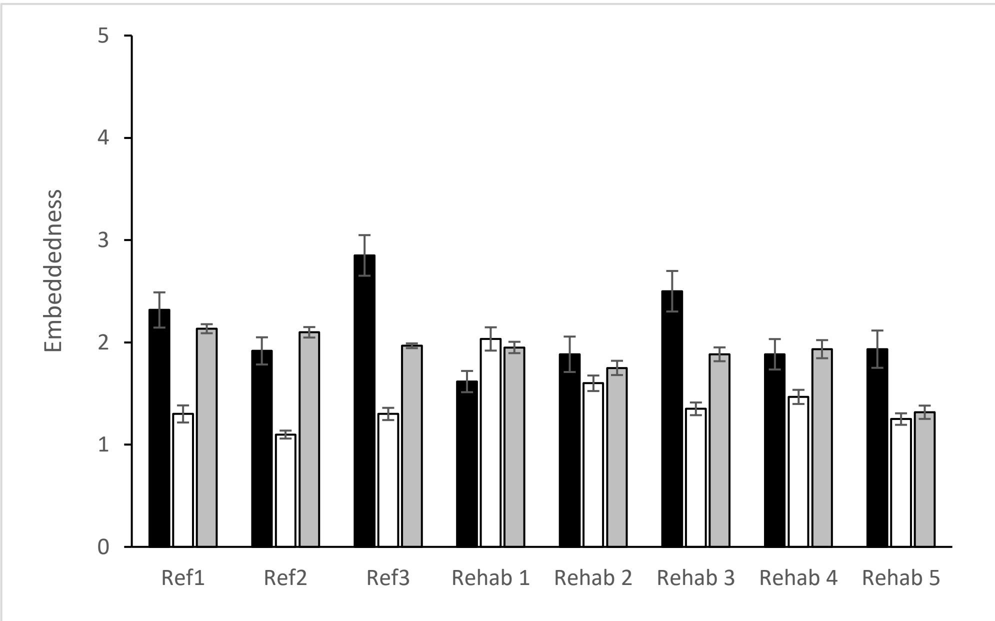


Figure 12: Average (±1SE) embeddedness measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

[13] Embeddedness was estimated on a categorical scale of 1-4 in 2013, but a categorical scale of 1-5 in 2015 and 2017, where an increasing score corresponds with increased embeddedness.

Compactness showed a similar relationship to embeddedness, where there was no significant difference in average compactness in reference and rehabilitation sites ($F_{1,18}$ = 0.109, P = 0.745). There was a detectable difference in compactness over time, where average compactness was greatest in 2013 (baseline average 3.4) compared to the 2014 (average 2.1) and 2017 (average 2.0) studies ($F_{2,18}$ = 14.196, P <0.001) (Figure 13).

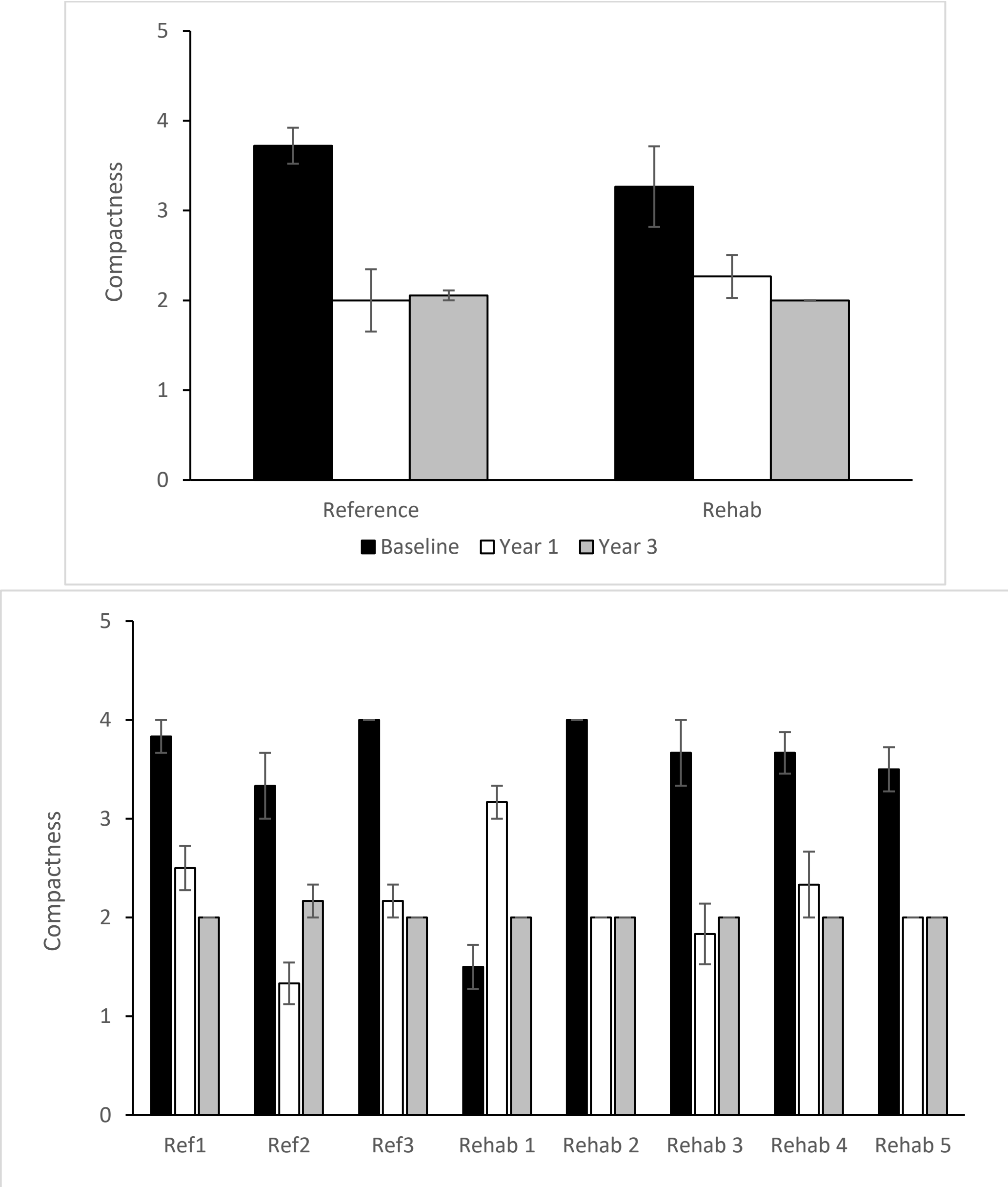


Figure 13: Average (±1SE) compactness measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

Macrophyte cover was highly variable among sites and through time (Figure 14). There were no significant differences in average macrophyte cover between reference and rehabilitation sites ($F_{1,18}$ = 0.390, P = 0.540) (average cover 28.3 % and 28.7 %, respectively) or among years ($F_{2,18}$ = 0.039, P = 0.962) (average cover 30 %, 29 % and 25 % for 2013, 2014 and 2017, respectively). This is not surprising given that macrophytes are regularly cleared from the Avon River, so any differences that might be expected due to rehabilitation treatment are likely to be masked by clearing activities.

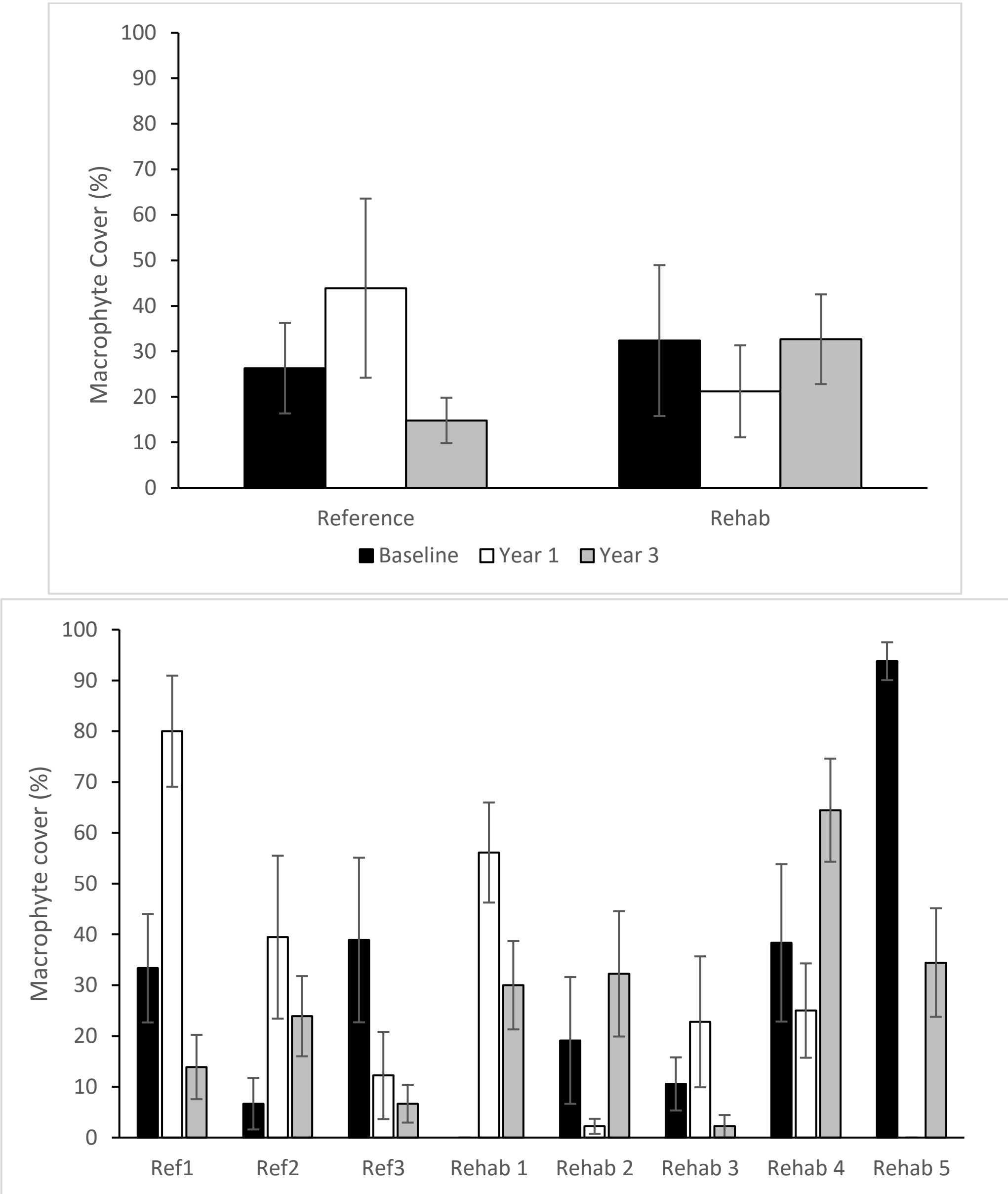


Figure 14: Average (±1SE) macrophyte cover measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

Algal cover was also variable among sites and through time (Figure 15). Average algal cover was greater in rehabilitation (49%), than reference (28%), sites ($F_{1,18} = 4.758$, $P = 0.043$), and was greatest in 2017 (average 59%), than the 2013 and 2014 (average of 34% and 29%, respectively) surveys ($F_{2,18} = 3.930$, $P = 0.038$).

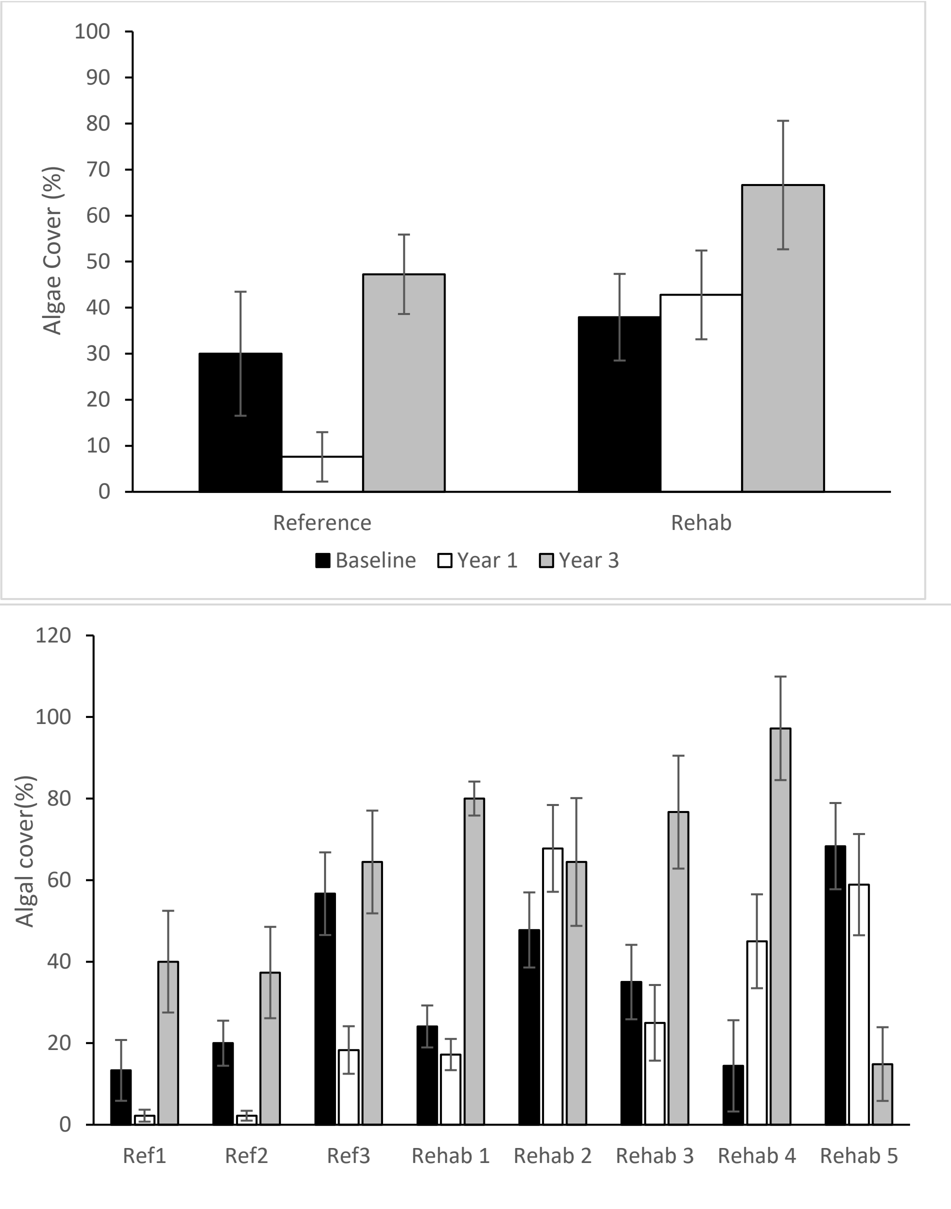


Figure 15: Average (±1SE) algal cover (%) measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

Macroinvertebrate Community

Overview

A total of 66,304 individuals, belonging to 41 taxonomic groups, were collected from all Surber and kick-net samples collected in March 2017 (i.e. from all eight sites surveyed). The most diverse groups were the true flies (Diptera) and the caddisflies (Trichoptera), both of which were represented by 10 different taxa. The next most diverse groups were the freshwater snails and bivalves (Mollusca) with 6 taxa, followed by crustaceans, with 4 taxa. Aquatic beetles (Coleoptera) and Annelida (worms and leeches) were each represented by two taxa. All other macroinvertebrate groups were represented by a single taxon (e.g. aquatic mites, Acarina; *Hydra*, Cnidaria; springtails, Collembola; true bugs, Hemiptera; damselflies, Odonata; and flatworms, Platyhelminthes).

Although crustaceans were not the most diverse group, they were numerically dominant (i.e. the most abundant group). Caddisflies and snails and bivalves were the next most dominant (abundant) group, followed by aquatic worms, and true flies. A number of macroinvertebrate taxa were only found in low numbers, or only one individual was ever encountered.

The freshwater amphipod *Paracalliope fluviatilis* was the most abundant species across all sample sites, with over 40,000 individuals collected. The stony-cased caddis *Pycnocentrodes aureolus*, the native mud snail *Potamopyrgus antipodarum*, seed-shrimp ostracods, and the stick-cased caddis *Hudsonema amabile* were also highly abundant.

The crustaceans dominated the macroinvertebrate community, making up 60% of all macroinvertebrates collected from the eight sites. Although caddisflies and true flies were the most diverse, they each only made up a small proportion of the total sample (15% and 4% of all macroinvertebrates sampled, respectively).

There were many taxa that were found at all sites surveyed. This included the most abundant taxa, *Paracalliope fluviatilis*, *Potamopyrgus antipodarum*, and *Pycnocentrodes aureolus*.

Total abundance

Macroinvertebrate abundance varied among the sites, ranging from 1,070 to 9,704 individuals collected in the Surber samples, with an additional 2,554 – 13,057 collected in the kick-net samples.

Macroinvertebrate abundance, as determined from Surber samples, did not differ between reference and rehabilitation sites ($F_{1,18}$ = 0.006, P = 0.939). However, there were minor (and significant) differences in the number of macroinvertebrates collected over time, with more macroinvertebrates collected in 2017 and 2014, than 2013 ($F_{2,18}$ = 3.578, P = 0.049) (Figure 16). Abundance increased with time (year) at reference sites, but for rehabilitation sites abundance was greatest in 2014, yet similar in 2013 and 2017 (Treatment:Year interaction: $F_{2,81}$ = 3.829, P = 0.041) (Figure 16).

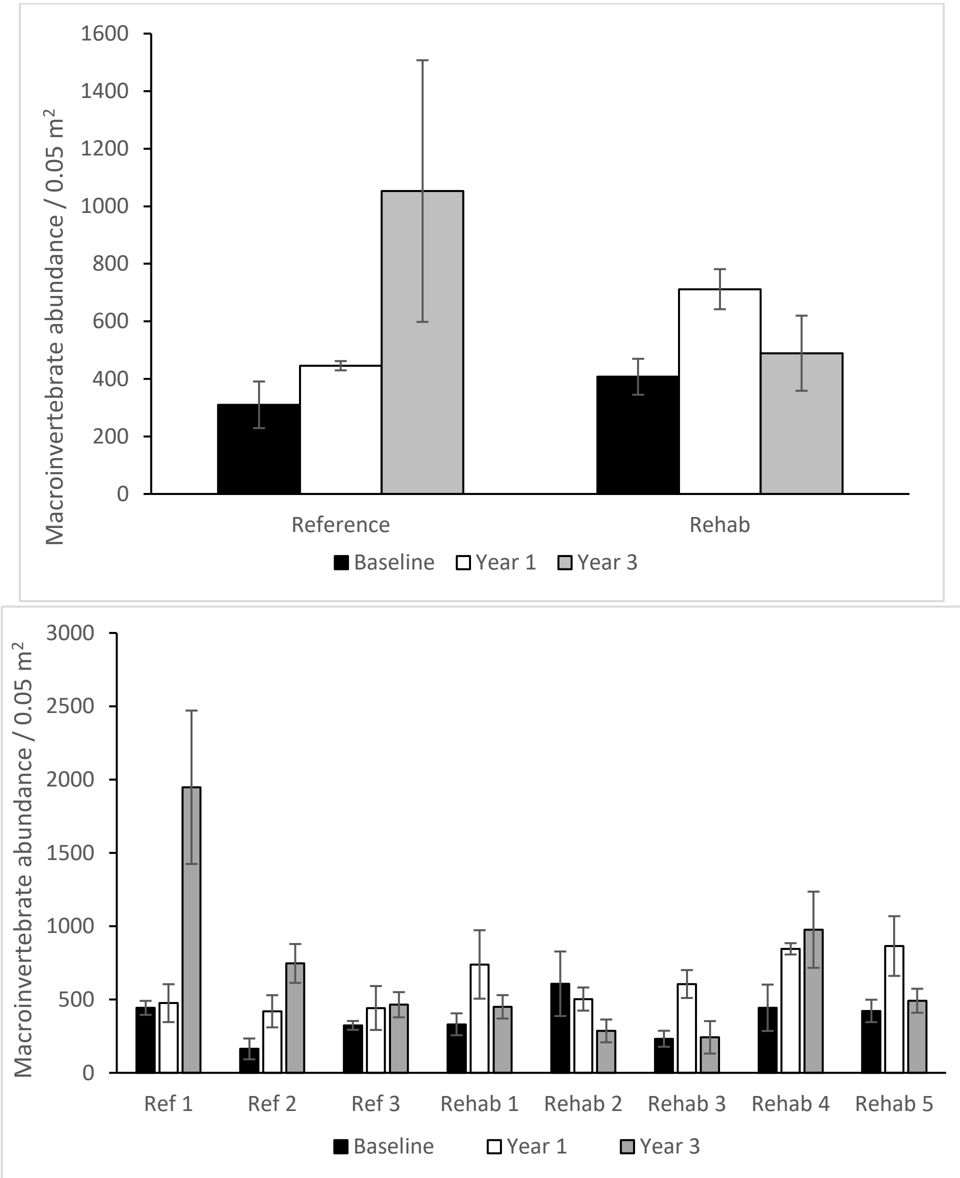


Figure 16: Average (±1SE) macroinvertebrate abundance from Surber samples measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

Taxonomic richness

Taxonomic richness was also variable among sites, ranging from 18 to 31 taxa per site. The kick-net samples collected, on average, 8 additional taxa that were not found in the Surber samples.

Taxonomic richness from Surber samples did not differ between reference and rehabilitation sites (average taxon richness: 15.51 and 15.53, respectively) ($F_{1,18} < 0.001$, $P = 0.992$).

There was, however, a significant difference among sample years, with an increase in the number of macroinvertebrate taxa collected through time (average taxon richness: 2013 – 11.9, 2014 – 16.55, and 2017 - 18.125) ($F_{2,18} = 7.556$; $P = 0.004$) (Figure 17).

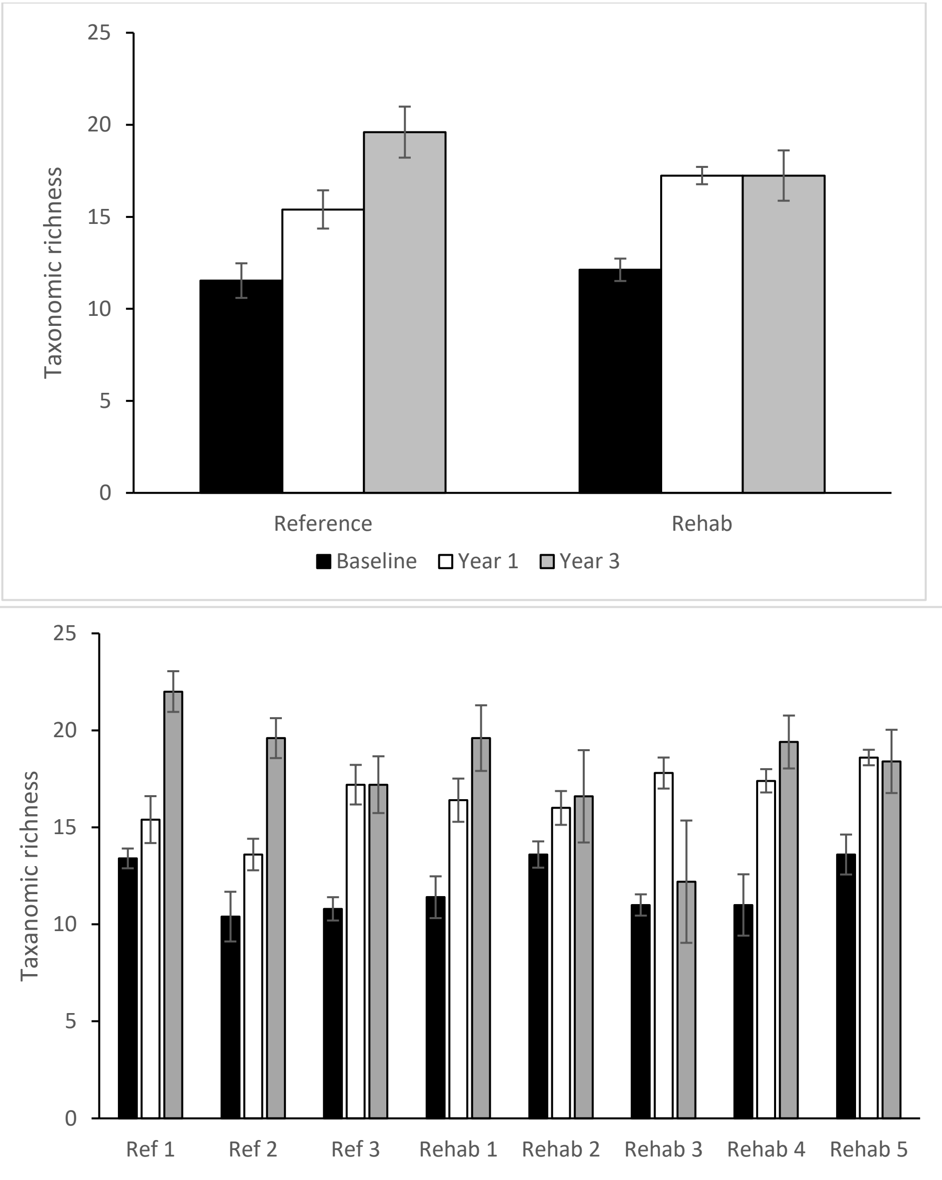


Figure 17: Average (±1SE) macroinvertebrate taxonomic richness from Surber samples measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars).

EPT richness

The EPT insect orders (Ephemeroptera, mayflies; Plecoptera, stoneflies; and Trichoptera, caddisflies) are generally more sensitive to pollution and habitat degradation, than other taxa, and are useful indicators of stream health. High EPT richness suggests good water and habitat quality where low EPT richness indicates poor water and habitat quality. Caddisflies were the only of the EPT taxa found in the Avon River across all surveys; no mayflies or stoneflies were collected at any sites.

There was a total of 10 caddisfly taxa collected in the 2017 survey, but this included a single larva of the family Oeconesidae that was only ever found in the kick-net sample in Reference Site 2. The average number of EPT taxa collected in the Surber samples ranged between 3 and 8 per site. Rehabilitation Site 3 showed the lowest caddisfly diversity, with an average of just three taxa collected in the five Surber samples. Rehabilitation Site 1 and Reference Site 3 each had an average of six caddisfly taxa collected from five Surber samples. All other sites had an average of seven or eight caddisfly taxa collected found in the Surber samples.

Hudsonema amabile and *Pycnocentrodes aureolus* were the most abundant of the caddisflies and were encountered at every site. The pollution tolerant caddisflies (family Hydroptilidae), *Oxyethira* and *Paroxythira,* were also present at all eight sites in 2014 and 2017, but not in 2013 (Figure 18).

Average EPT richness did not differ between reference and rehabilitation sites ($F_{1,18}$ = 0.969, P = 0.337), but there was a significant difference in EPT richness among years ($F_{2,18}$ = 3.658, P = 0.046). EPT richness was greater in 2017 (the Year 3 survey), than the previous two surveys (Figure 18). When hydroptilids, the more pollution sensitive EPT taxa, were excluded there were no significant differences in EPT richness observed between site or survey year.

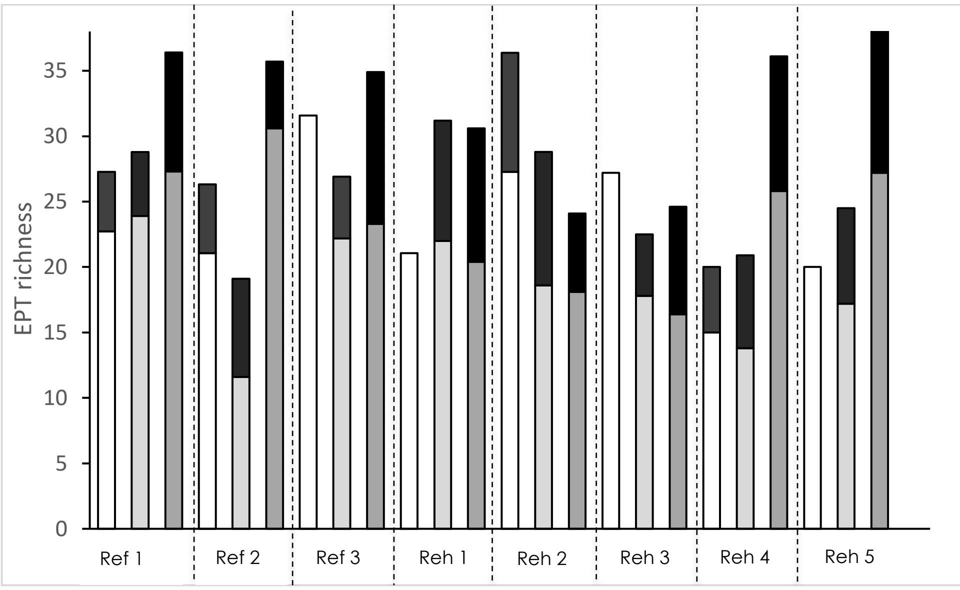
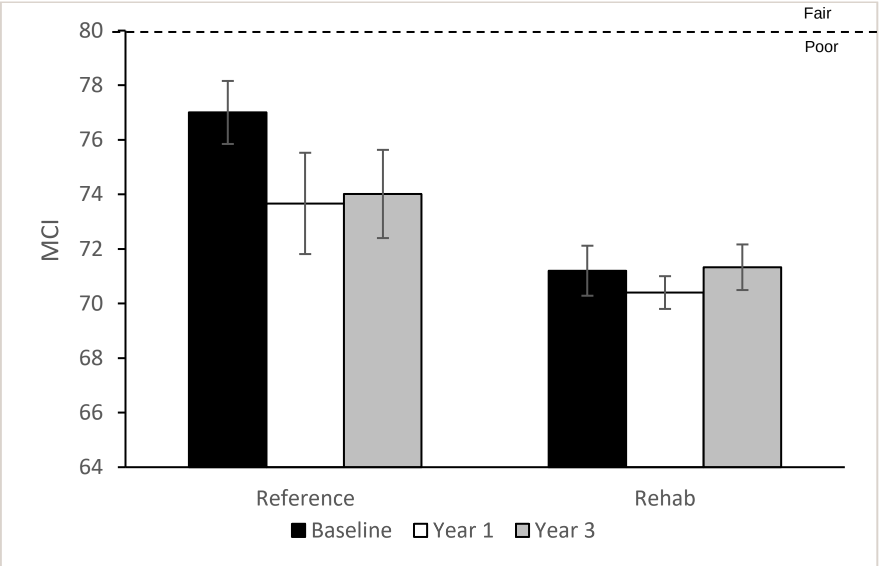


Figure 18: Average EPT richness (light bars) and hydroptilid richness (dark bars) collected from Surber samples at all eight sites from the baseline survey (Boffa Miskell 2014), 2014 (Opus 2015), and this survey (2017) – shown from left to right. Note, the vertical dotted line is for visual aid to differentiate between sites. SE bars are not shown.

Macroinvertebrate Community Index

MCI and QMCI scores are a measure of stream, or ecological, health with higher scores indicating greater ecological condition.

MCI scores were variable among sites, with reference sites having marginally higher MCI scores, than rehabilitation sites ($F_{1,18} = 18.87$, $P < 0.001$) (average MCI of 74 and 70, respectively). All sites surveyed (across 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 19) and there was no difference in MCI scores over time ($F_{2,18} = 18.879$, $P < 0.001$).



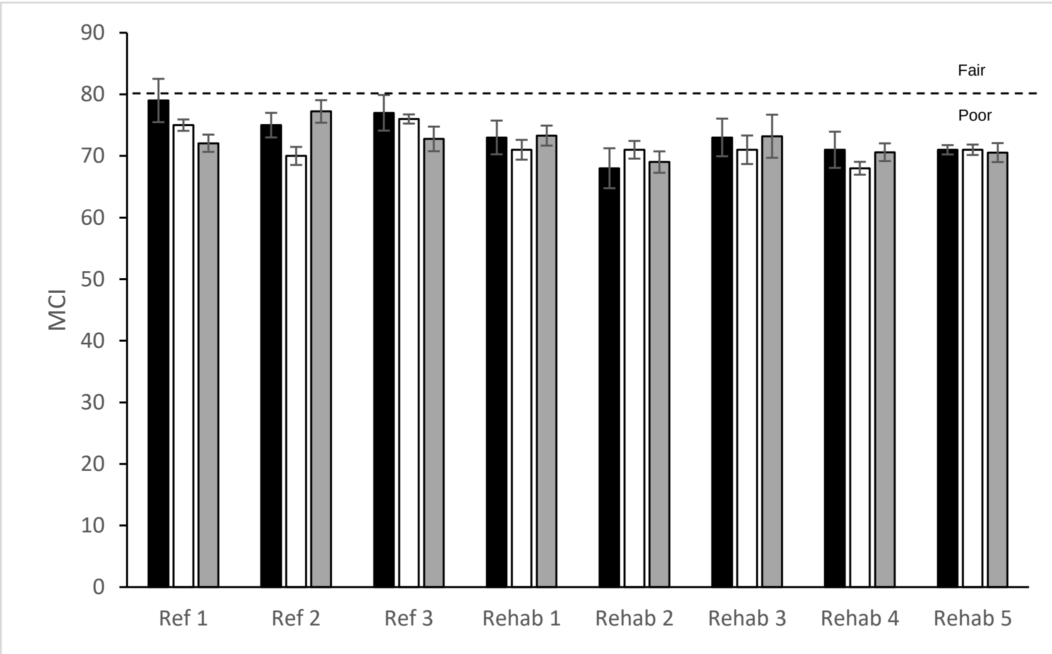


Figure 19: Average (±1SE) Macroinvertebrate Community Index (MCI) measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars). The dashed line indicates the water quality categories of Stark and Maxted (2007), where "poor" = "probable severe enrichment" and "fair" = "probable moderate enrichment". The "good" = "doubtful quality or possible mild enrichment", and "excellent" = "clean water" categories are outside of the scale of this figure. See Table 2 for further information.

QMCI, which is considered a better indicator of “health” than MCI, as it considers both abundance and presence of macroinvertebrate taxa, showed slightly different results. All sites, except Rehabilitation Site 1, surveyed in 2017 had QMCI scores of 4, or above, indicating “fair” stream health with “probable mild pollution”. Rehabilitation Site 1 had “poor” stream health with “probable or severe enrichment” (based on the water quality categories of Stark and Maxted 2007) (Figure 20).

There was no significant difference between the QMCI scores at reference (average QMCI of 3.9) and rehabilitation sites (average QMCI of 3.7) ($F_{1,18}$ = 0.442, P = 0.514). There were, however, significant differences observed among survey years ($F_{2,18}$ = 8.90, P = 0.002). QMCI scores were lowest in 2013, compared to 2014 and 2017 surveys (Figure 20).

These differences may indicate an improvement in stream health, with QMCI scores increasing in 2014 and 2017. However, it is important to note that QMCI scores can be highly variable through time. This is because abundances of macroinvertebrates can vary / change due to a range of disturbances including both natural (e.g. floods) and anthropogenic perturbations (e.g. nutrients / stormwater discharges). Moreover, Increases in QMCI scores were detected in both reference and rehabilitation sites (Figure 20). Differences in QMCI of the magnitude detected in this study (between baseline (2013) and post-rehabilitation (2014 and 2017) may not reflect ecologically relevant change in macroinvertebrate community composition.

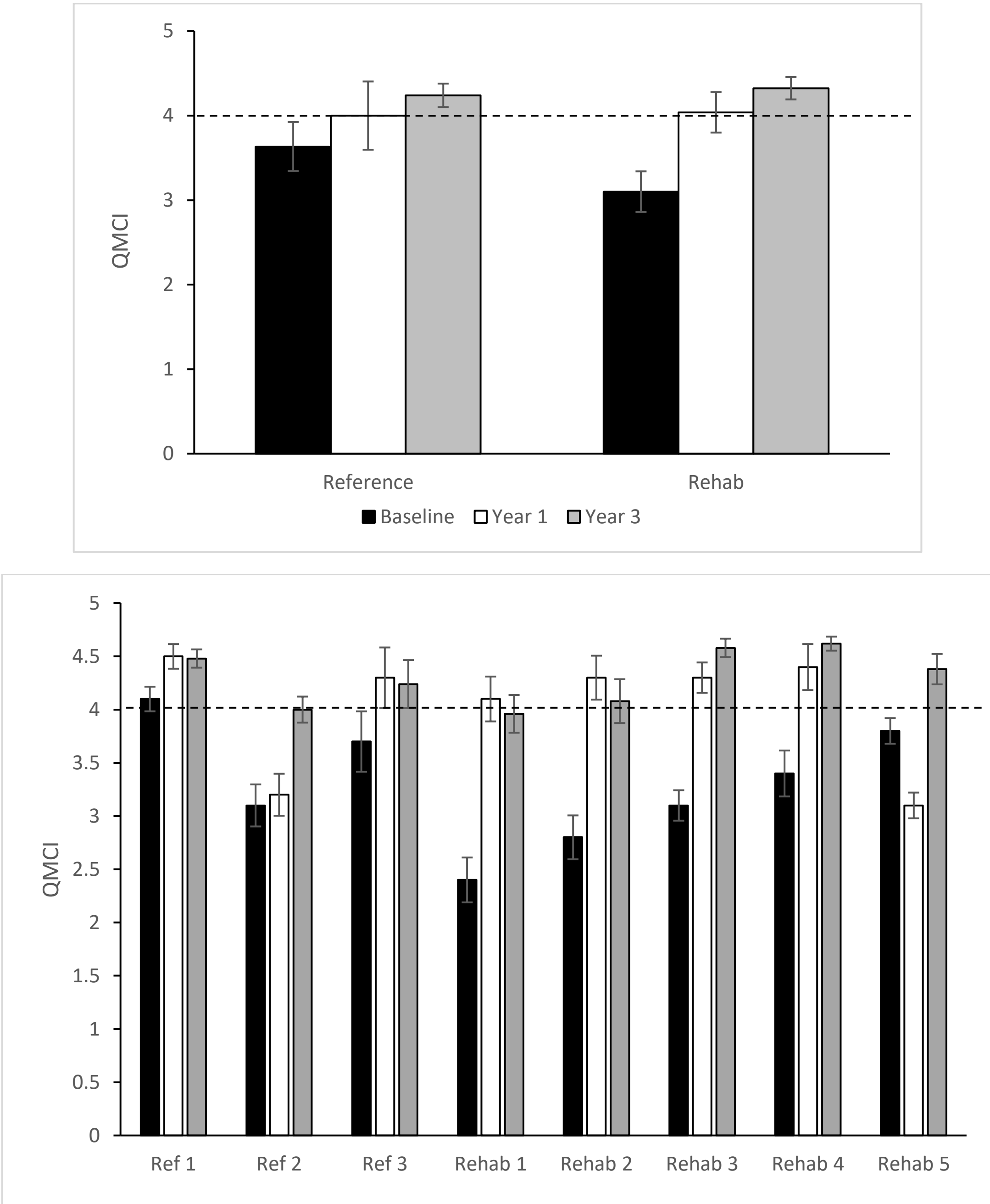


Figure 20: Average (±1SE) QMCI measured across the reference and rehabilitation sites (top) and at each of six transects, at each of the eight Avon River Precinct sites (bottom) for the baseline study (2013, black bars), one-year post-rehabilitation (2014, white bars), and three-years post-rehabilitation (2017 – this study, grey bars). The dashed line indicates the threshold between the "poor" and "fair" water quality categories of Stark and Maxted (2007), where "poor" = "probable severe enrichment" is below QMCI of 4, and "fair" = "probable moderate enrichment" is QMCI 4.00 – 4.99. The "good" = "doubtful quality or possible mild enrichment", and "excellent" = "clean water" categories are outside of the scale of this figure. See Table 2 for further information.

Community composition

There were some differences in community composition over time, which were largely due to variance in relative dominance (percent abundance) of different taxa over time. The 2014 macroinvertebrate survey was dominated by caddisflies, whereas the baseline survey community was dominated by crustaceans and true flies. The macroinvertebrate community in 2017 was dominated by crustaceans, particularly the taxa *Paracalliope* and Ostracoda. Conversely, community composition appeared to be relatively similar between reference and rehabilitation sites through time (Figure 21).

Nevertheless, the relative abundance of molluscs and worms detected in the rehabilitation sites decreased, between 2013 (22% and 15%, respectively) and 2017 (14% and 4%, respectively). There was also a notable shift in the percent contribution of crustaceans and caddisflies between 2013 and 2017 surveys (Figure 21).

The reference sites had lower relative abundances of molluscs, than the rehabilitation sites in all years, while relative abundances of crustaceans were higher in reference than rehabilitation sites in all years, expect 2017. Caddisfly relative abundance in the reference sites varied through time; it was lowest in 2013 (13%) and 2017 (17%), and highest in 2014 (32%).

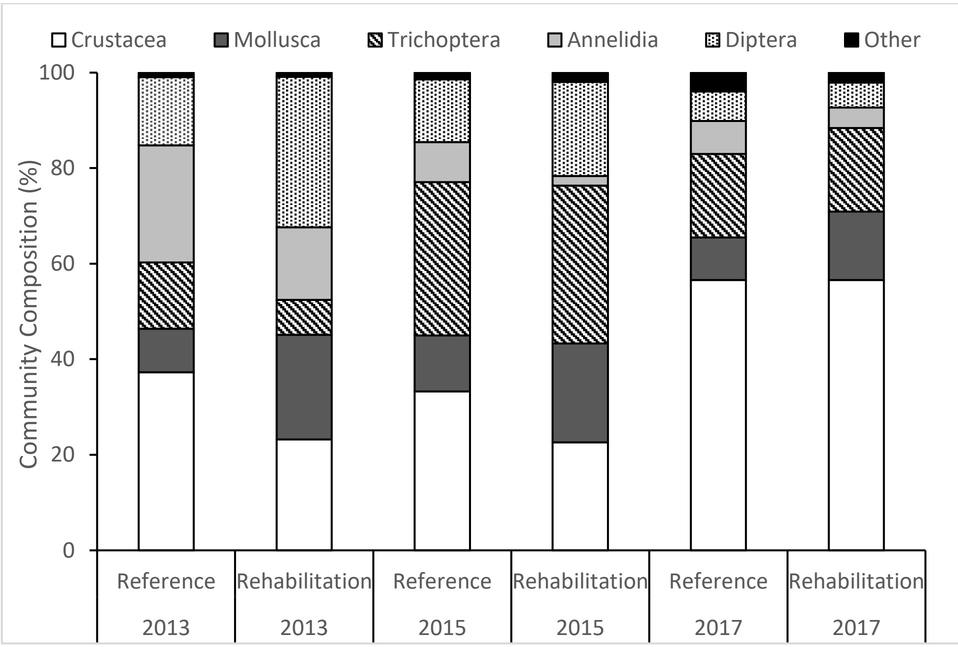


Figure 21: Average macroinvertebrate community composition (%) found at reference and rehabilitation sites across the baseline survey (2013), one-year post-rehabilitation (2014) and three-years post-rehabilitation (2017) surveys.

The NMDS ordination, confirmed by the ANOSIM results, also indicated that there were statistically significant differences in the macroinvertebrate community found through time (ANOSIM R = 0.457; P = 0.001) (Figure 22). The macroinvertebrate communities were more similar within each survey occasion (year), than among years (2017 v. 2014 R = 0.365, P = 0.016; 2017 v. 2013 R = 0.595, P = 0.003; 2014 v. 2013 R = 0.499, P = 0.012).

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 from one sampling occasion. For example, the amphipod *Paracalliope fluviatilis* was considerably more abundant in the 2017 survey than the previous 2014 and 2013 surveys. The caddisfly *Pycnocentrodes aureolus* and the snail *Potamopyrgus antipodarum* were more abundant in 2014, whereas orthoclad midge larvae and oligochaete worms dominated the macroinvertebrate community in 2013 (see Appendix 4 for further details on SIMPER results).

There were also significant, but subtle, differences between reference and rehabilitation sites (ANOSIM R = 0.265; P = 0.026) when combining all survey years. SIMPER again indicated that these significant (but subtle) differences in community composition (between rehabilitation and reference sites) 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 macroinvertebrate taxon. For example, *Paracalliope fluviatilis* was more abundant in the reference, than rehabilitation sites; the stony-cased caddisfly *Pycnocentrodes aureolus* was more than twice as abundant in rehabilitation sites than reference sites; and the cased caddisfly *Hudsonema amabile* was more abundant in rehabilitation sites than the reference sites (see Appendix 4 for further details on SIMPER results).

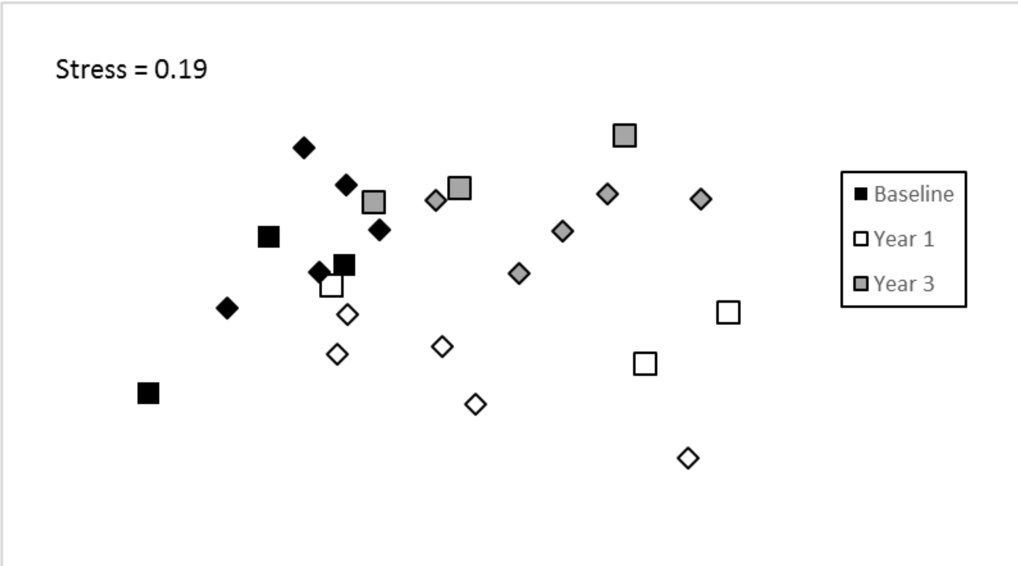


Figure 22: Non-metric multidimensional scaling (NMDS) ordination based on a Bray-Curtis matrix of dissimilarities calculated from macroinvertebrate abundance data collected from the eight sites surveyed in 2013 (baseline survey – black squares; Boffa Miskell 2014), 2014 (one-year post-rehabilitation – white squares; Opus 2015) and 2017 (three-years post-rehabilitation – grey squares; this study). Reference sites are shown as squares; rehabilitation sites are shown as diamonds. Axes are identically scaled so that sites closest together are more similar in macroinvertebrate composition, than those further apart. The significance of differences in community dissimilarity was confirmed using Analysis of Similarities (ANOSIM).

Fish Community

Overview

A total of 918 individuals, belonging to 11 different species, were captured at the eight sites in the Avon River, in March 2017. The species caught, in descending order from most to least abundant, were: upland bully (*Gobiomorphus breviceps*), common bully (*G. cotidanus*) bluegill bully (*G. hubbsi*), shortfin eel (*Anguilla australis*), longfin eel (*A. dieffenbachii*), giant bully (*G. gobioides*), lamprey (*Geotria australis*), smelt (*Retropinna retropinna*), torrentfish (*Cheimarrichthys fosteri*), and the introduced brown trout (*Salmo trutta*). Lamprey is classified as Threatened, Nationally Vulnerable; longfin eel, bluegill bully, and torrentfish are listed as at Risk, Declining, while the other species captured are Not Threatened (Goodman et al. 2014).

Total abundance and species richness

Species richness was relatively similar with an average of six species being found at each site. Rehabilitation Site 3 (in 2017) had the highest species richness (9 species), whereas Rehabilitation Site 5, Rehabilitation Site 2, and Reference Site 3 had the lowest richness with 6 species being found in 2017.

Common bully, upland bully, and both longfin and shortfin eels were found at all eight sites surveyed. Lampreys were uncommon, with only 4 individuals captured, one individual and each of three sites, and two individuals at one site. Smelt and torrentfish were the least abundant species and each species was found at only one site.

Upland bullies were the most commonly occurring species, found at all sites in relatively high numbers. Common bullies and giant bullies were also found at every site; common bullies were in relatively high in numbers, whereas giant bullies were found in low numbers with only one individual being found at Rehabilitation Sites 1, 2, and 5.

Bluegill bullies were the next most common species, and were found at all rehabilitation sites, but were found at only one of the reference sites. Both longfin and shortfin eels were found at all sites in relatively similar numbers.

Compared to previous years, more species of freshwater fish were found in the eight Avon River sites in 2017 (three-years post-rehabilitation) survey than the other years ($F_{2,18} = 4.87$, $P = 0.020$). However, sampling effort was marginally greater at a number of, but not all, sites in 2017 and 2014, than 2013. An average of 5 species was captured in the baseline (2013) survey, while 6 (on average) species were found in the Year 1 (2014) survey, compared to an average of 7 species in 2017. Marginally more species were found at rehabilitation than reference sites, when all years were combined, but this difference was not statistically significantly different ($F_{2, 18} = 1.62$, $P = 0.220$).

Most importantly, some freshwater fish species were found at rehabilitation sites in 2017, but were not found (or were in lower numbers and frequency) in 2014 and 2013 (baseline) (Figure 23). For example, bluegill bullies were found, in low abundances, in Rehabilitation Sites 4 & 5 in 2013 (baseline), and again in relatively low abundances in Rehabilitation Sites 1, 2, 4, & 5 in 2014. In 2017, bluegill bullies were found in all five of the rehabilitation sites and in greater numbers than previously recorded. A single torrentfish was found at Rehabilitation Site 4 in 2017. It was caught mid channel, within a macrophyte bed of curly pondweed. Torrentfish were not detected in either the 2013 or 2014 surveys (Figure 23).

Lamprey were also found in greater numbers, and more sites, in 2017, compared to previous years. Lamprey were not found in 2013 survey; two individuals were recorded at one rehabilitation site in 2015; and a total of four individuals were captured at two rehabilitation and one reference site in 2017.

Significantly more fish (i.e. number of individuals) were captured at rehabilitation sites, than reference sites ($F_{1,18} = 4.534$, $P = 0.047$). The number of fish found in 2017 was also greater than in 2013 and 2014 ($F_{2,18} = 28.44$, $P < 0.001$) (Figure 23).

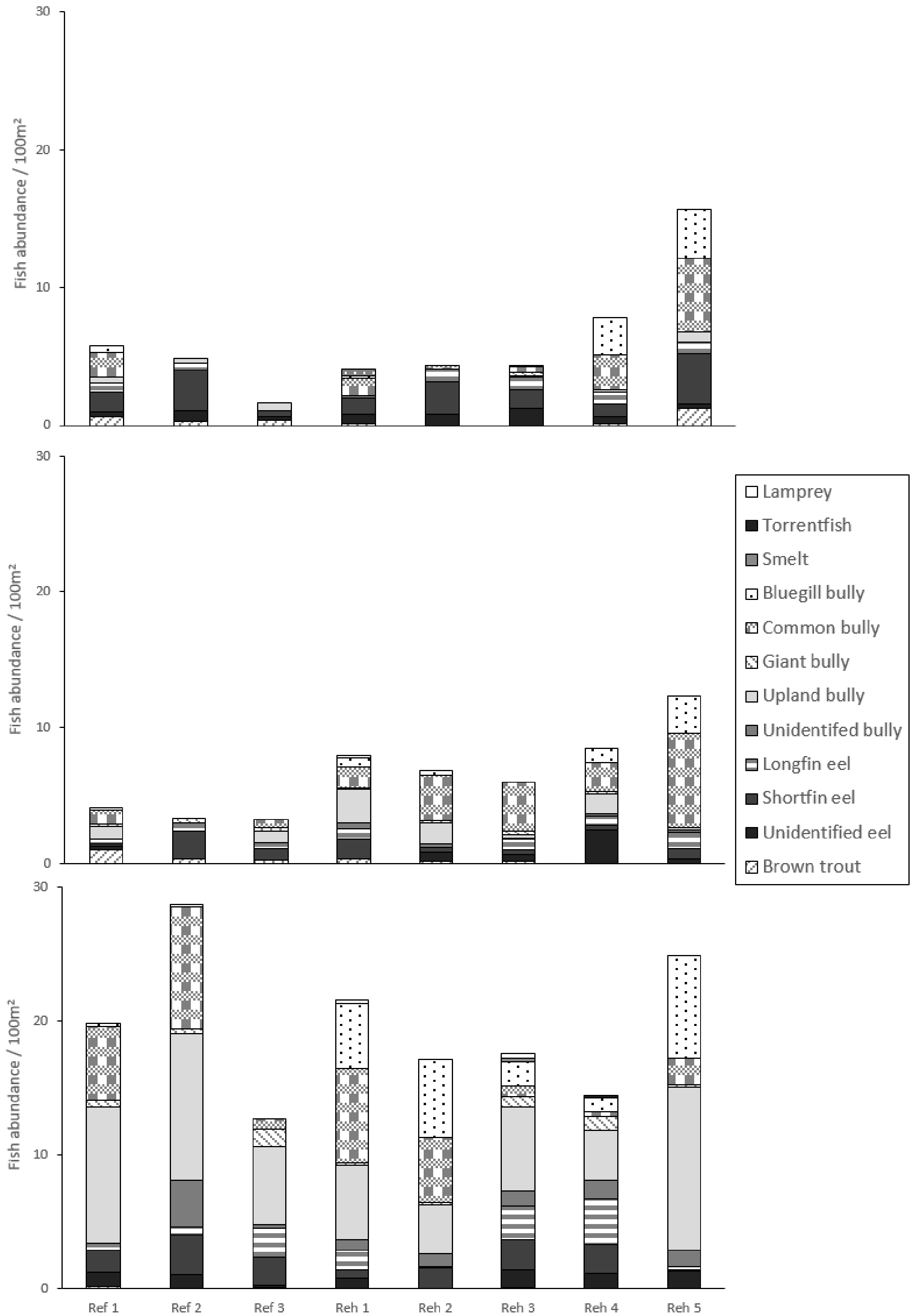


Figure 23: Total abundance of fish, separated by species, captured at each of the eight sites surveyed in 2013 (baseline, top); 2014 (one-year post-rehabilitation, middle); and 2017 (three-years post-rehabilitation, bottom). Numbers are show as catch per unit effort (CPUE): per 100 m^2 of waterway surveyed using electric fishing.

Community composition

The NMDS ordination, confirmed by the ANOSIM results, indicated that there were significant differences in the fish communities found through time (ANOSIM R = 0.583, P = 0.001) (Figure 24). The fish community found in 2017 was different to that of 2013 and 2014 (2017 v. 2014 R = 0.750, P = 0.002; 2017 v. 2013 R = 0.682, P = 0.002). The fish communities in 2013 and 2014 were not statistically significantly different (R = 0.243, P = 0.061).

The main drivers of the differences between the 2017 and 2013 & 2014 fish communities were generally due to differences in abundances of upland bully, common bully, bluegill bully, shortfin eel, and longfin eel. Common, bluegill and upland bullies were more abundant in 2017, than 2013 and 2014, and dominating the 2017 community at some sites (particularly the rehabilitation sites). Of greatest interest was the increasing abundances of bluegill bully, an At Risk, Declining species.

Several new species were also found (or found in greater numbers) in 2017 (e.g. torrentfish, lamprey), however, the presence of these species were not main drivers of community differences (see Appendix 5 for further details on SIMPER results).

ANOSIM indicated there were also significant differences between reference and rehabilitation sites (ANOSIM R = 0.241, P = 0.028). Again, these differences were due to changes in average number of occurrences of some species (i.e. greater or lesser numbers of individuals), rather than the presence or absence of particular species from one sampling occasion. For example, common bullies and bluegill bullies were markedly more abundance in rehabilitation than reference sites (see Appendix 5 for further details on SIMPER results). It’s also noteworthy that a single torrentfish was detected in the 2017 surveys at Rehabilitation Site 4, Avon River at Victoria Square, but never found at any other site or during other sampling occasions.

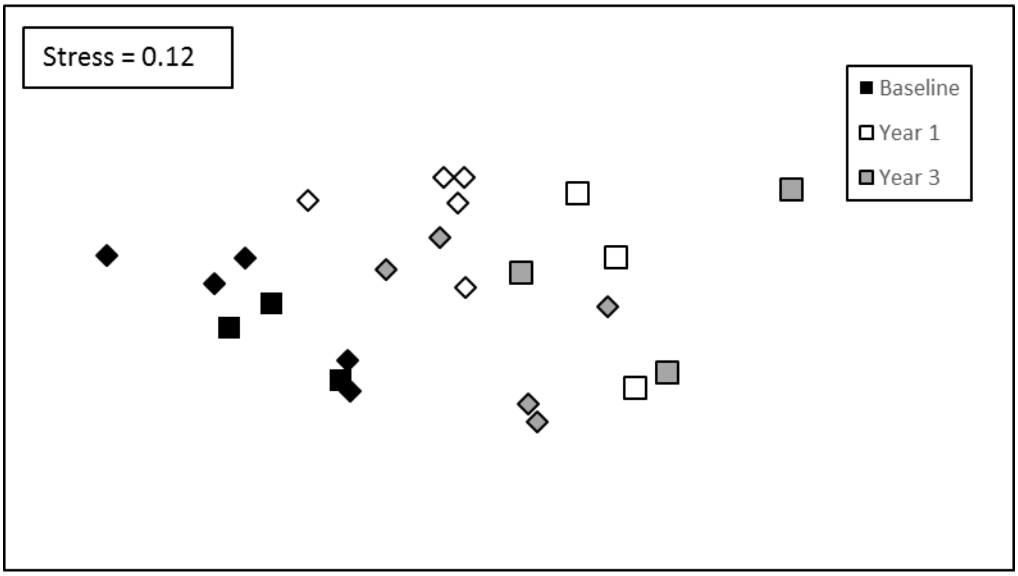


Figure 24: Non-metric multidimensional scaling (NMDS) ordination based on a Bray-Curtis matrix of dissimilarities calculated from fish abundance (CPUE) data collected from the eight sites surveyed in in 2013 (baseline survey – black squares; Boffa Miskell 2014), 2014 (one-year post-rehabilitation – white squares; Opus 2015) and 2017 (three-years post-rehabilitation – grey squares; this study). Reference sites are shown as squares; rehabilitation sites are shown as diamonds. Axes are identically scaled so that sites closest together are more similar in macroinvertebrate composition, than those further apart. The significance of differences in community dissimilarity was confirmed using Analysis of Similarities (ANOSIM).

Discussion

Riparian and in-stream rehabilitation works were undertaken at five sites along the Avon River in 2013 / 2014, as part of the ARP 'anchor project' and Christchurch Rebuild. The main objective of this work was to return the river to a more natural state and to increase water quality and ecological health. Three surveys have been completed as part of the monitoring programme, including a baseline survey in 2013 (Boffa Miskell 2014a), the one-year post-rehabilitation survey 2014 (Opus 2015), and the three-years' post-rehabilitation survey in 2017 (this study).

The results of this ecological survey (2017) indicate that all sites were largely of "poor" to "fair" ecological health, as indicated by the macroinvertebrate community indexes (MCI & QMCI). The macroinvertebrate communities were dominated by pollution-tolerant taxa, such as snails, ostracods, and midge larvae. However, some subtle changes in river, or ecosystem, health have been noted and may be, in part, due to the rehabilitation works. Slight changes to in-stream and riparian habitat conditions were apparent, as well as subtle differences in macroinvertebrate community composition and notable differences in the fish communities.

Water quality

There were no marked differences in water quality observed between reference and rehabilitation sites. The basic water quality parameters recorded were only recorded on one occasion at each site, in 2013, 2014, and 2017. However, these parameters (DO, pH, temperature, conductivity) fluctuate both daily and seasonally. Moreover, some of the habitat rehabilitation activities that might be expected to improve water quality conditions (e.g. riparian plantings improving stream shading and therefore moderating water temperatures) may still require time to modify conditions of the aquatic ecosystem. Stormwater, and other inputs, still enter the river, bringing run off from the urban catchment. All of which can impact water quality and stream health.

Habitat characteristics

In-stream and riparian habitat conditions that were the main foci of the rehabilitation work were constructed wetlands, creation of riffle habitat, and the removal of fine sediment embedding bed substrates. It is noteworthy that a reduction in sediment depth, over time, was detected at the rehabilitation sites. Sediment depth was also lower in the rehabilitation, than the reference, sites. Together, these indicate that the cleaning of substrates (i.e. the removal of fine sediments entrained within the cobbles and gravels) may have had a lasting effect in rehabilitation sites.

The embeddedness and compactness measures further reinforce this, with embeddedness and compactness remaining low in 2017. This may mean coarser substrate and interstitial spaces continue to be more readily available for aquatic fauna, such as for macroinvertebrate taxa to graze on, fish to lay eggs, etc. Substrate index also appeared to be greatest in rehabilitation sites, which presumably can also be attributed to the removal of fine sediment from in-stream substrates, as well as the addition of larger boulders along the wetted edge / toe of river bank.

Both water depth and velocity were lower in 2014 and 2017, than 2013 (baseline). The one-year post-rehabilitation survey (Opus 2015) was conducted during a 'dry' year, however, depth and velocity in the three-years' post-rehabilitation survey have decreased again. The cause of this decrease over time could be attributed to dry summer conditions over the past few summers indicating periods of low flows around the survey time in March. Another, more widespread, cause of the decline in water levels and velocity could be the impact of ongoing groundwater abstraction that occurs in Canterbury, which may be impacting spring-fed, lowland rivers such as the Avon River. However, CCC and Environment Canterbury have investigated causes of low river levels over the previous summersd. Low winter rainfall and the associated reduced groundwater recharge was determined to as having the greatest influence, with groundwater abstraction being only a small contributing factor (Greg Burrell, CCC Waterways Ecologist, pers. comm.).

Macroinvertebrate communities

There were only subtle changes in macroinvertebrate community detected, with macroinvertebrate abundance, taxonomic richness, EPT richness, and QMCI scores all increasing through time. These differences may indicate an improvement in stream health over time, with QMCI scores increasing in 2014 and 2017. However, it is important to note that many of these parameters can be highly variable through time. Abundances of macroinvertebrates can vary / change due to a range of disturbances including both natural (e.g. floods) and anthropogenic perturbations (e.g. nutrients / stormwater discharges). Moreover, Increases in QMCI scores were detected in both reference and rehabilitation sites, and differences of the magnitude detected in this study may not reflect ecologically relevant change in macroinvertebrate community composition.

There were no detectable differences in macroinvertebrate abundance, taxonomic richness, and EPT richness attributable to the ARP rehabilitation activities. MCI scores were, however, greater in reference than rehabilitation sites, yet all sites remained (both over time and at sites where rehabilitation activities had taken place) within the “poor” or “fair” water quality categories of Stark and Maxted 2007. In general, the macroinvertebrate communities continued to be dominated by taxa typical of more degraded waterways.

When the entire macroinvertebrate community was considered, there were minor differences detected between rehabilitation and reference sites. Some macroinvertebrate taxa appeared to have responded to changes to in-stream conditions at rehabilitation sites. For example, the stony-cased caddis *Pycnocentrodes aureolus* was more than twice as abundant in rehabilitation sites than reference sites; and the cased caddisfly *Hudsonema amabile* was more abundant in rehabilitation sites than the reference sites.

Fish communities

There were notable differences in the fish communities found both across years, and between rehabilitation and reference sites.

More species and individuals of freshwater fishes were captured in 2017, compared to previous years. Of greater interest, bluegill bullies were both more abundant and found at a greater number of sites (including all rehabilitation sites) in 2017, compared to previous years.

Lamprey were also found at a great number of sites in 2017, compared to previous years, and a single torrentfish was found at a rehabilitation site in 2017, but this species had not been recorded at any sites previously.

Fish abundance was significantly different over time as well as significantly different between treatment, with rehabilitation sites showing greater abundance than reference sites and higher numbers being observed over time. Species richness of fish was only significantly different over time and not between reference and rehabilitation sites. This may have been due to slight differences in sampling effort, with marginally greater area surveyed, across all sites, in 2017 (and 2014) than 2013. Nevertheless, rehabilitation sites had slightly greater fish species richness, than reference sites, but this difference was not statistically different.

Current success of rehabilitation works

This ecological assessment of the Avon River Precinct sites indicates that the ARP sites were generally of poor or fair ecological health (based on the macroinvertebrate community index). In-stream rehabilitation works have included the construction of riffles and wetland floodplains along the river, which are likely to provide habitat for aquatic fauna (particularly fishes) during times of flood. The wetland floodplains also contribute to improved terrestrial biodiversity and ecosystem health, and may provide important habitat for certain life stages of aquatic fauna, such as resting sites for winged adult aquatic insects.

While there were few marked differences between reference and rehabilitation sites, detectable differences in sediment depth and substrate index were noted. This indicates that rehabilitation works, which included cleaning of bed substrates, may have had a lasting effect on in-stream conditions. More importantly, differences, albeit subtle ones, have been detected in aquatic fauna.

The stony-cased caddis *Pycnocentrodes aureolus* and the stick-cased caddis *Hudsonema amabile* were both more abundant in rehabilitation sites than the reference sites. This was also found to be the case in 2014, one-year post-rehabilitation activities. This may indicate that the coarser substrates, due to removal of fine sediments during the rehabilitation activities, has resulted in a response from the biological community.

However, no new macroinvertebrate taxa, taking advantage of newly created habitat, were found in the rehabilitation sites in 2017, nor in 2014 (Opus 2015). It may be that rehabilitation works have not lead to substantial enough changes to riparian and in-stream communities for macroinvertebrates that were once present in the catchment to recolonise (Suren and McMurtrie 2005). A lack of substantial change in the macroinvertebrate community found in rehabilitation sites could also be an artefact of limited, or no, source populations of additional clean-water taxa (e.g. mayflies and caddisflies) surrounding the restored sites. Mayflies, which were once present in the Avon River catchment (e.g. *Deleatidium* sp.; Robb 1992) are now limited to a few semi-rural waterways on the outskirts of Christchurch's metropolitan area (e.g. Ōtūkaikino Creek, Styx River, and the wider other spring-fed tributaries in the Waimakariri River catchment). Recent studies have shown that river restoration success (and colonisation by 'new' clean-water macroinvertebrate taxa) depends on the presence of source populations within relatively close proximity (Sundermann et al. 2011).

Moreover, EPT taxa (and other aquatic insects) have a winged adult stage and are likely to face a multitude of anthropogenic barriers to dispersal in urban environments, which can all have implications for recruitment. For example, road crossings (i.e. culverts), light pollution (many of our caddisfly species are nocturnal), and the probable confusion of the built environment (e.g. concrete, which when wet reflects polarised light that confuses aquatic insects, tall buildings with few riparian 'markers' for species to navigate along and between waterways) may all disrupt adult aquatic insect flight (Blakely et al. 2006).

Aerial dispersal of adults can be a particularly important (but not the only) route for aquatic insects to recolonise waterways. Adult aquatic insects have a variety of species-specific methods for laying eggs in a waterway. Some insects approach a stream almost like a top-dressing pilot and broadcast lay their egg masses into the water. While others have very specific water velocity oviposition (egg laying) substrate types. Some aquatic insect species deposit egg masses on the undersides of submerged boulders in stream channels, others specifically select emergent boulders with very specific water velocities, or in the middle of riffle habitat. The size of the (submerged or emergent) boulder is extremely important to some species, while for others water velocity is more critical (Reich and Downes 2003). Despite the in-stream rehabilitation work carried out at the ARP sites, there is still a lack of large emergent boulders. The successful recruitment of aquatic insects, which in turn provide food sources for freshwater fishes, is (in part) dependent on the availability of suitable oviposition habitat.

Although only very subtle differences in macroinvertebrate communities were detected, indicating a relatively weak response to rehabilitation works, the fish community continued to show a response to rehabilitation works. In particular, bluegill bullies, an open-bed, fast-water species, were found at all of the rehabilitation sites (more than in 2014) and in greater numbers than previous surveys. This is an exciting result, given this species is classified as at risk, declining, and is considered locally rare in Christchurch, with only small, isolated populations known from a few riffles within Christchurch's urban waterways. For example, small populations are known to occur in Cashmere Stream (at the Cashmere Road Bridge riffle (EOS Ecology 2013)), in No. 2 Drain (EOS Ecology 2012), and the Avon River (Boffa Miskell 2014b, Opus 2015, and this study).

Another exciting find in the 2017 survey was the single torrentfish, recorded at Rehabilitation Site 4: Victoria Square. Like the bluegill bully, torrentfish specialise in living in very swift cobble habitat. Torrentfish are very rarely found in Christchurch's waterways, with the only other known record from Cashmere Stream in 2008 (EOS Ecology 2013). It's plausible that the newly created rapid and riffle habitat of the ARP sites may provide suitable habitat for torrentfish to inhabit. Time will tell if this find was a one-off record of a single individual, or if this species will persist.

Lamprey were also found at more sites in the 2017, than previous surveys, and there appeared to be a propensity towards rehabilitation sites. While found throughout New Zealand, lamprey is a *threatened, nationally vulnerable* species. With a rather interesting lifecycle they spend most of their lives at sea, attached to other fish and feeding off their hosts blood. However, the adults spawn in freshwater, and the larvae may spend as much as four years in our rivers and streams before heading to the ocean. Until recently, little was known about their breeding habitats and they were thought likely to occur in stony-bottomed streams. However, a small population was recently found in a silt-dominated timber-lined 'drain' waterway, a tributary of the Styx River in Christchurch.

Limitations to success

As many of New Zealand's freshwater fishes, and all (except upland bully) found in this study, are diadromous, meaning they spend part of their lives in the sea, and part in freshwater. Thus, there may be opportunities for 'new' species to colonise the ARP sites via the Avon-Heathcote estuary. If in-stream conditions created by the rehabilitation works continue to persist (e.g. clean gravels, fast riffles), the macroinvertebrate and fish community is likely to continue to improve, and new species, or species previously present in the Avon River catchment, may colonise the river.

Distance to source populations

However, it is important to have realistic expectations for these likely improvements. For example, the distance to source populations of clean-water aquatic insect taxa is great (more than 5 km), and colonisation of some species (e.g. mayflies, caddisflies) is likely to be limited (Sundermann et al. 2011). It may be that certain macroinvertebrate taxa (and particularly aquatic insects) will not be able to recolonise the ARP sites without 'stepping stone' habitats of rehabilitated waterways scattered between the source population and the Avon River. Rehabilitation efforts need to continue to focus on a variety of waterways within Christchurch, including small tributary waterways that may assist in providing colonisation pathways.

Importance of boulder habitat

While some submerged boulders have been added to the river channel at rehabilitation sites, there remains limited availability of large emergent boulders. These emergent boulders, which protrude from riffles in the middle of the channel, are an important oviposition resource for many aquatic insect species. Although we have limited knowledge on the spawning habitat requirements of torrentfish and bluegill bullies, we do know that large, stable boulders provide important egg-laying substrates for bullies. More boulders, both emergent and submerged, should be added to the rehabilitation sites to provide this egg-laying habitat (Blakely and Harding 2005).

Macrophyte and debris maintenance

Like many of Christchurch's urban waterways, macrophytes and debris jams are regularly cleared from the Avon River to manage drainage capacity and flood flow conveyance. However, in urban stream systems, macrophytes can provide valuable cover and habitat in an often otherwise homogenous environment. For example, many fishes, including bluegill bullies, were found in high numbers in macrophyte beds in the Avon River. Although no freshwater crayfish (kēwai / kōura, *Paranephrops zealandicus*) were found at any of the ARP (reference or rehabilitation) sites, this at risk, declining species has been found in macrophyte beds in the Heathcote River (Boffa Miskell 2015) and Cashmere Stream (EOS Ecology 2013). Macroinvertebrates and fish can be entangled in macrophytes and debris jams and end up unintentionally being removed along with the unwanted macrophytes and debris jams. Unfortunately, without a salvage programme fauna are unlikely to find their way back to the waterway, and are left stranded and perish. Changes to the current maintenance regimes of macrophyte cover and debris jams in Christchurch's waterways may be an important part of improving ecological health, particularly in areas where little other diversity of habitat or cover is available.

Fish hotels and natural fish cover

Fish hotels have been included in the rehabilitation design at (site 2) the Canterbury Earthquake National Memorial. These 'fish hotels' were designed to provide habitat for eels and other large fishes. The inclusion of anthropogenic materials (cordyline pipe, concrete blocks, large water-filled hollows, etc.) are an excellent inclusion in situations such as this. The design of the memorial wall meant that hard surfaces (i.e. steps) were built to the water's edge along the true right bank at Rehabilitation Site 2. It was, therefore, not possible to include natural in-bank diversity for fauna (e.g. root balls, tree stumps). However, when the site was surveyed in March 2017, the water level of the river was not high enough to inundate the fish hotels, which likely left little useable aquatic habitat. When the site was revisited in May 2017, when winter baseflows are generally higher than summer baseflows, the fish hotels were either partially or fully covered by the water. It will be useful to monitor these fish hotels, observing water levels throughout the year, to determine the frequency and duration that these provide useable aquatic habitat. The diversity and availability of in-bank habitat for fauna, including root balls, tree stumps, earth banks, overhanging vegetation could be increased at the rehabilitation sites, and more generally along much of the Avon River. Unlined, earth banks are also important habitat for kēwai / kōura that burrow into banks. Where there are site constraints, such as at the memorial wall, anthropogenic materials could be more regularly incorporated into the river bank.

Stormwater, sediment, and contaminant inputs

Although the ARP rehabilitation works have resulted in measurable (albeit subtle) improvements in habitat conditions and macroinvertebrate and fish communities, untreated stormwater inputs, which bring sediments and contaminants, may continue to limit in-stream recovery. Stormwater treatment devices, such as tree pits, swales, and rain gardens, are being installed in many new developments in the central city as well as the wider Avon River catchment. This needs to continue with a focus on reducing the quantity of sediment and contaminant inputs into the catchment, and retrofitting existing drainage and stormwater connections.

Conclusions

Overall, the rehabilitation works at the ARP sites have resulted in major aesthetic and more minor ecological improvements. Increased velocities now provide a perceptible babbling noise, and purposeful places for the public to access the river and interact with large eels (e.g. 'tame' eels at the Terraces). The constructed wetland floodplains and riparian plantings also contribute to increased terrestrial biodiversity and provide habitat for terrestrial and aquatic fauna, including winged adult aquatic insects.

Ecological gains may arguably be small, but nevertheless important. Some caddisfly taxa have become more abundant at rehabilitation sites overtime, and bluegill bully populations are now found at all rehabilitation sites. The exciting find of a single torrentfish, at one rehabilitation site, is also to be celebrated. However, it is unclear whether the habitat and water velocities of the ARP sites is suitable to sustain a torrentfish population. This will be unfolded through future surveying of the ARP sites as part of this long-term monitoring programme.

Additional rehabilitation activities, which could further enhance the ecological response, may include:

- Addition of large, emergent and submerged, boulders in riffle habitat;
- Changes to the macrophyte and debris jam maintenance regime;
- Increased variety and availability of in-bank habitat, including root balls, tree stumps, earth banks, overhanging vegetation, and anthropogenic materials (fish hotels); and
- Multi-faceted approach to stormwater management, to reduce the inputs of sediments and contaminants into the river.

It's important to have realistic goals and expectations of ecological gains because of these rehabilitation works. It may be some time for certain taxa (e.g. some aquatic insects) to colonise the Avon River, due to dispersal constraints and / or barriers. Freshwater fishes may colonise more quickly than some aquatic insects, given the colonisation pathway for fishes via the Avon-Heathcote estuary. However, it's plausible that some taxa may need to be managed and even re-introduced into the river. This could be undertaken, but only once habitat conditions and food resources are deemed suitable for the species' survival.

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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

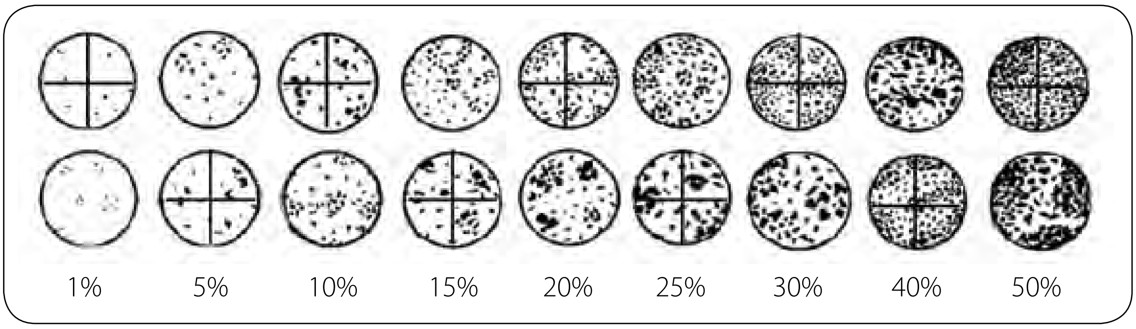
- Locate five random transects along the run.
- View the streambed at four randomly determined locations across each transect, starting at the downstream transect.
- Estimate the fine sediment cover in each quadrat of the underwater viewer in increments (1, 5, 10, 15, 20 …100%).
- Record results in the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

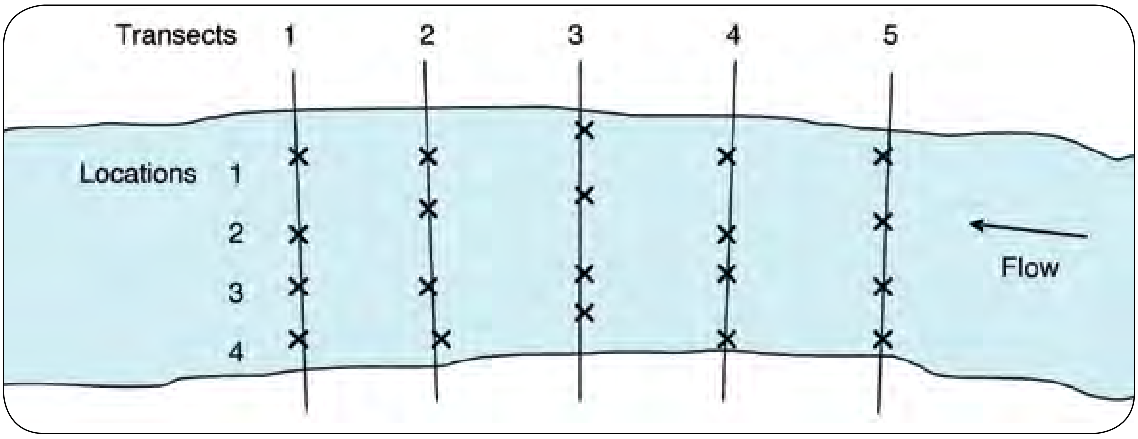
% sediment	Transect 1		Transect 2		Transect 3		Transect 4		Transect 5	
Location 1	Q1	Q2								
	Q3	Q4								
Location 2										
Location 3										
Location 4										

Useful images

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



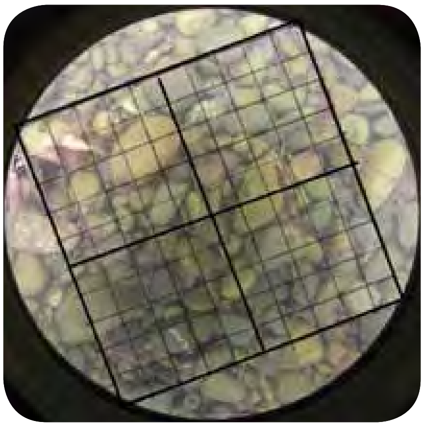
An example of viewer locations (x) for the in-stream visual assessment of sediment.



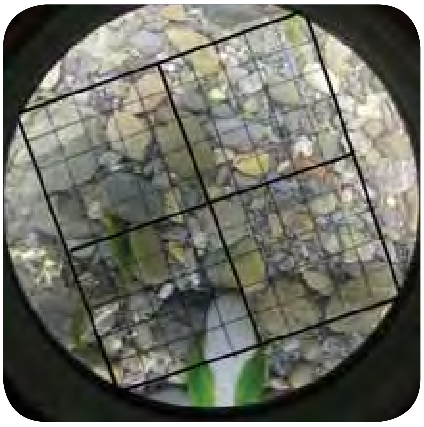
Real examples of percent cover of sediment on the streambed as seen through an underwater viewer.

1%

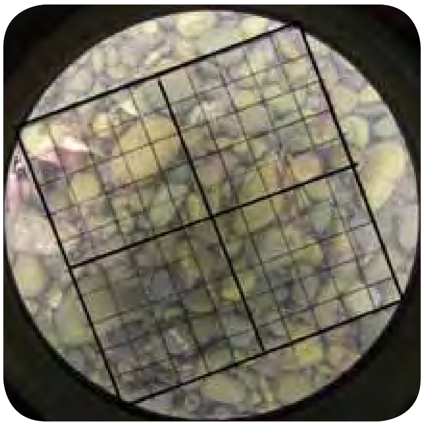
1%



5%



5%



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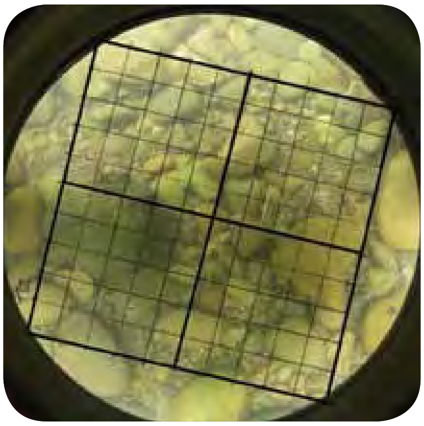

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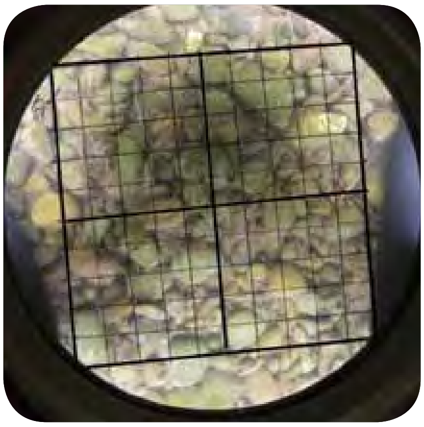

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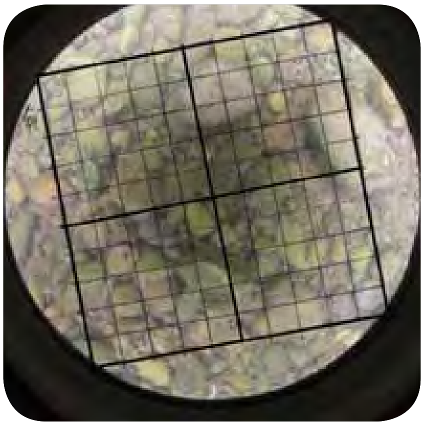

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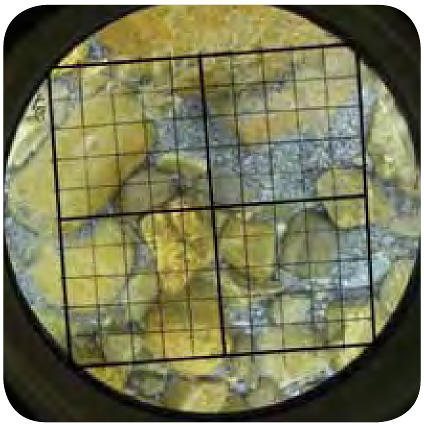

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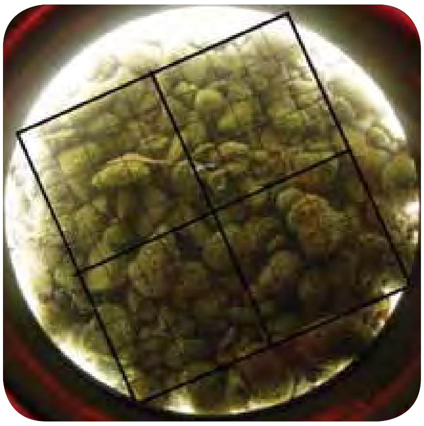

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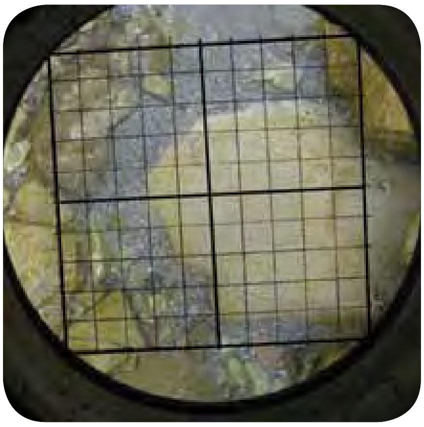

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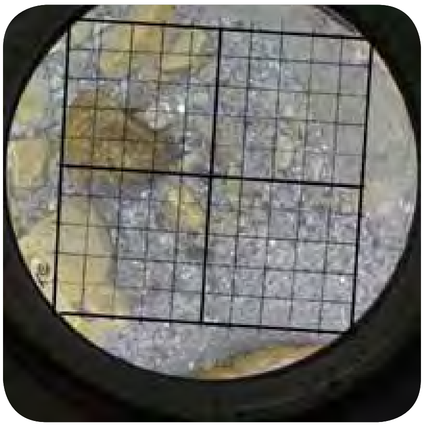

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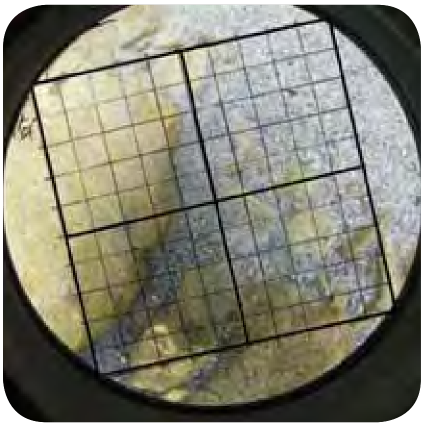

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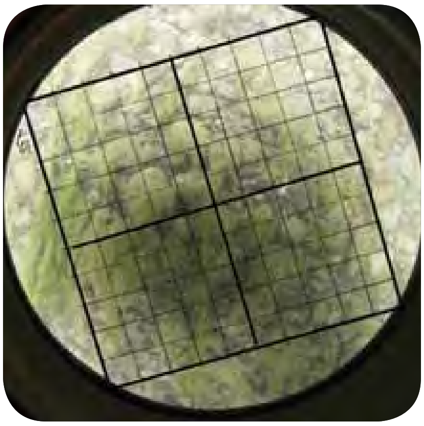

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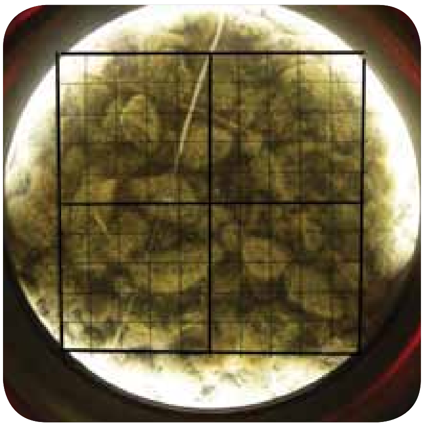

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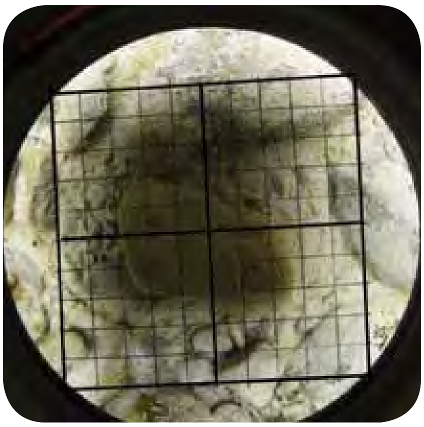

Appendix 3: Sediment Assessment Method 6 (SAM6), Clapcott et al. (2011)

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 results - macroinvertebrates

SIMPER
Similarity Percentages - species contributions

Two-Way Analysis

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

Parameters
Resemblance: S17 Bray Curtis similarity
Cut off for low contributions: 90.00%

Factor Groups

Sample	Year	Treatment
Ref1	2017	Ref
Ref2	2017	Ref
Ref3	2017	Ref
Rehab1	2017	Rehab
Rehab2	2017	Rehab
Rehab3	2017	Rehab
Rehab4	2017	Rehab
Rehab5	2017	Rehab
Ref 1	2015	Ref
Ref 2	2015	Ref
Ref 3	2015	Ref
Rehab 1	2015	Rehab
Rehab 2	2015	Rehab
Rehab 3	2015	Rehab
Rehab 4	2015	Rehab
Rehab 5	2015	Rehab
Ref 1	2013	Ref
Ref 2	2013	Ref
Ref 3	2013	Ref
Rehab 1	2013	Rehab
Rehab 2	2013	Rehab
Rehab 3	2013	Rehab
Rehab 4	2013	Rehab
Rehab 5	2013	Rehab

Examines Year groups
(across all Treatment groups)
Group 2017
Average similarity: 56.25

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Paracalliope	399.53	34.49	3.04	61.32	61.32
Potamopyrgus	35.58	5.08	2.16	9.02	70.34
Oligochaeta	36.08	2.72	2.05	4.84	75.19
Pycnocentrodes	59.10	2.51	0.54	4.46	79.64
Oxyethira	16.25	2.36	1.80	4.19	83.83
Physella (Physa)	20.75	1.63	2.11	2.90	86.73
Hudsonema	32.03	1.38	1.01	2.46	89.19
Orthocladiinae	12.80	1.30	1.09	2.31	91.50

Group 2015
Average similarity: 44.01

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Pycnocentrodes	112.48	10.23	0.70	23.24	23.24
Potamopyrgus	72.53	6.88	1.03	15.63	38.87
Paracalliope	99.38	6.09	0.86	13.84	52.70
Hudsonema	62.98	5.99	1.02	13.60	66.31
Ostracoda	60.28	3.71	1.37	8.43	74.74
Orthocladiinae	39.20	2.22	0.91	5.04	79.77
Oligochaeta	22.13	1.96	0.83	4.46	84.24
Physella (Physa)	22.53	1.90	1.73	4.31	88.55
Tanytarsini	72.38	1.39	1.09	3.17	91.72

Group 2013
Average similarity: 56.51

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Orthocladiinae	84.68	14.05	1.57	24.86	24.86
Oligochaeta	65.95	12.87	2.98	22.78	47.64
Paracalliope	110.68	11.56	1.49	20.47	68.11
Potamopyrgus	54.13	11.44	2.35	20.24	88.34
Pycnocentrodes	23.24	1.37	0.50	2.42	90.76

Groups 2017 & 2015
Average dissimilarity = 59.66

	Group 2017	Group 2015				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Paracalliope	399.53	99.38	18.38	1.26	30.80	30.80
Pycnocentrodes	59.10	112.48	10.50	1.04	17.61	48.41
Tanytarsini	14.48	72.38	5.80	0.52	9.72	58.13
Potamopyrgus	35.58	72.53	5.39	1.30	9.04	67.17
Ostracoda	19.83	60.28	4.31	0.93	7.23	74.39
Hudsonema	32.03	62.98	3.43	0.94	5.75	80.15
Orthocladiinae	12.80	39.20	3.12	0.80	5.22	85.37
Physella (Physa)	20.75	22.53	1.89	0.92	3.17	88.54
Oligochaeta	36.08	22.13	1.56	1.11	2.62	91.16

Groups 2017 & 2013
Average dissimilarity = 58.01

	Group 2017	Group 2013				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Paracalliope	399.53	110.68	21.84	1.44	37.65	37.65
Orthocladiinae	12.80	84.68	9.55	1.02	16.47	54.12
Pycnocentrodes	59.10	23.24	5.98	0.95	10.31	64.44
Oligochaeta	36.08	65.95	4.71	1.36	8.12	72.56
Potamopyrgus	35.58	54.13	4.19	0.99	7.22	79.78
Oxyethira	16.25	0.81	1.86	1.43	3.20	82.98
Hudsonema	32.03	3.58	1.80	1.10	3.10	86.08
Ostracoda	19.83	9.69	1.48	1.03	2.54	88.63
Physella (Physa)	20.75	3.06	1.18	1.15	2.03	90.66

Groups 2015 & 2013
Average dissimilarity = 62.28

Species	Group 2015 Av.Abund	Group 2013 Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Pycnocentrodes	112.48	23.24	11.44	1.15	18.37	18.37
Paracalliope	99.38	110.68	10.21	1.22	16.40	34.77
Orthocladiinae	39.20	84.68	6.45	1.10	10.36	45.12
Tanytarsini	72.38	5.24	6.38	0.55	10.24	55.36
Hudsonema	62.98	3.58	5.90	0.92	9.48	64.84
Ostracoda	60.28	9.69	5.37	0.74	8.63	73.47
Potamopyrgus	72.53	54.13	4.68	1.41	7.52	80.98
Oligochaeta	22.13	65.95	4.37	1.83	7.01	88.00
Physella (Physa)	22.53	3.06	1.99	0.91	3.20	91.20

Examines Treatment groups
(across all Year groups)
Group Ref
Average similarity: 45.96

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Paracalliope	280.22	14.18	1.08	30.85	30.85
Oligochaeta	56.33	9.14	1.15	19.89	50.74
Hudsonema	58.47	5.56	0.74	12.09	62.83
Orthocladiinae	19.89	3.52	1.11	7.66	70.49
Potamopyrgus	28.71	2.54	1.68	5.52	76.01
Pycnocentrodes	34.71	2.23	0.71	4.86	80.87
Tanytarsini	24.07	2.06	1.61	4.49	85.36
Ostracoda	40.34	1.83	1.06	3.99	89.35
Playthelminthes	9.49	1.09	1.22	2.37	91.71

Group Rehab
Average similarity: 54.14

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Paracalliope	156.97	18.34	1.16	33.88	33.88
Potamopyrgus	69.29	9.37	1.76	17.31	51.19
Orthocladiinae	60.96	6.55	0.75	12.10	63.29
Pycnocentrodes	83.08	5.44	0.50	10.05	73.34
Oligochaeta	32.41	4.87	1.03	8.99	82.34
Ostracoda	23.68	1.79	0.79	3.31	85.65
Hudsonema	17.49	1.68	0.84	3.11	88.75
Physella (Physa)	16.05	1.53	1.51	2.83	91.58

Groups Ref & Rehab
Average dissimilarity = 53.98

Species	Group Ref Av.Abund	Group Rehab Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Paracalliope	280.22	156.97	15.89	1.13	29.43	29.43
Pycnocentrodes	34.71	83.08	7.68	1.02	14.22	43.65
Orthocladiinae	19.89	60.96	5.60	0.71	10.38	54.03
Potamopyrgus	28.71	69.29	5.41	1.16	10.02	64.05
Hudsonema	58.47	17.49	3.54	0.94	6.56	70.61
Tanytarsini	24.07	34.67	3.47	0.48	6.43	77.04
Ostracoda	40.34	23.68	3.16	0.73	5.86	82.90
Oligochaeta	56.33	32.41	2.98	1.26	5.52	88.42
Physella (Physa)	14.44	16.05	1.42	0.88	2.63	91.05

Appendix 5: SIMPER results - fish

SIMPER
Similarity Percentages - species contributions

Two-Way Analysis

Data worksheet
Name: Fish 2
Data type: Abundance
Sample selection: All
Variable selection: All

Parameters
Resemblance: S17 Bray Curtis similarity
Cut off for low contributions: 90.00%

Factor Groups

Sample	Year	Treatment
Reference 1	2017	Reference
Reference 2	2017	Reference
Reference 3	2017	Reference
Rehab 1	2017	Rehab
Rehab 2	2017	Rehab
Rehab 3	2017	Rehab
Rehab 4	2017	Rehab
Rehab 5	2017	Rehab
Ref 1	2015	Reference
Ref 2	2015	Reference
Ref 3	2015	Reference
Rehab 1	2015	Rehab
Rehab 2	2015	Rehab
Rehab 3	2015	Rehab
Rehab 4	2015	Rehab
Rehab 5	2015	Rehab
Ref1	2013	Reference
Ref2	2013	Reference
Ref3	2013	Reference
Rehab1	2013	Rehab
Rehab2	2013	Rehab
Rehab3	2013	Rehab
Rehab4	2013	Rehab
Rehab5	2013	Rehab

Examines Year groups
(across all Treatment groups)
Group 2017
Average similarity: 56.57

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Upland bully	42.50	27.02	3.16	47.77	47.77
Bluegill bully	17.88	9.37	1.04	16.55	64.32
Common bully	21.00	8.11	0.97	14.34	78.66
Shortfin eel	9.50	5.80	1.15	10.25	88.91
Longfin eel	7.75	4.02	0.72	7.10	96.01

Group 2015
Average similarity: 56.63

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Common bully	17.25	24.56	1.53	43.37	43.37
Longfin eel	4.38	8.30	2.22	14.66	58.03
Upland bully	7.00	7.44	0.92	13.13	71.16
Shortfin eel	4.38	6.23	0.84	11.00	82.16
Bluegill bully	4.00	3.11	0.74	5.50	87.66
Eel sp	4.00	2.97	0.73	5.25	92.91

Group 2013
Average similarity: 48.77

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Shortfin eel	11.88	20.31	2.07	41.66	41.66
Common bully	9.50	8.91	1.05	18.26	59.92
Eel sp	4.75	7.07	1.73	14.50	74.42
Longfin eel	4.50	5.82	1.23	11.93	86.34
Upland bully	1.63	2.28	0.56	4.68	91.02

Groups 2017 & 2015
Average dissimilarity = 59.58

	Group 2017	Group 2015				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Upland bully	42.50	7.00	22.35	1.69	37.52	37.52
Common bully	21.00	17.25	12.36	1.25	20.74	58.26
Bluegill bully	17.88	4.00	9.34	1.01	15.68	73.94
Shortfin eel	9.50	4.38	4.55	1.36	7.64	81.57
Longfin eel	7.75	4.38	4.31	0.99	7.24	88.81
Eel sp	1.63	4.00	3.32	0.98	5.56	94.38

Groups 2017 & 2013
Average dissimilarity = 67.39

	Group 2017	Group 2013				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Upland bully	42.50	1.63	26.98	2.43	40.03	40.03
Common bully	21.00	9.50	11.74	1.24	17.42	57.45
Bluegill bully	17.88	5.13	10.50	1.09	15.58	73.03
Shortfin eel	9.50	11.88	5.31	1.85	7.88	80.91
Longfin eel	7.75	4.50	4.87	1.03	7.22	88.13
Eel sp	1.63	4.75	3.31	1.06	4.91	93.05

Groups 2015 & 2013
Average dissimilarity = 52.42

	Group 2015	Group 2013				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Common bully	17.25	9.50	14.29	1.37	27.26	27.26
Shortfin eel	4.38	11.88	10.66	1.68	20.33	47.59
Upland bully	7.00	1.63	7.77	1.26	14.82	62.41
Bluegill bully	4.00	5.13	6.03	1.15	11.50	73.91
Eel sp	4.00	4.75	5.00	1.14	9.54	83.45
Longfin eel	4.38	4.50	3.37	1.15	6.43	89.88
Brown trout	1.63	2.00	2.54	0.83	4.84	94.72

Examines Treatment groups

(across all Year groups)
Group Reference
Average similarity: 52.95

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Upland bully	16.89	18.26	1.03	34.48	34.48
Shortfin eel	7.67	13.86	1.28	26.17	60.65
Longfin eel	3.44	5.69	1.03	10.74	71.40
Common bully	10.56	5.11	0.68	9.66	81.05
Brown trout	1.89	4.45	1.30	8.40	89.45
Eel sp	2.00	2.85	0.91	5.39	94.84

Group Rehab
Average similarity: 54.30

Species	Av.Abund	Av.Sim	Sim/SD	Contrib%	Cum.%
Common bully	19.13	16.48	1.18	30.35	30.35
Upland bully	17.13	10.44	0.97	19.23	49.59
Shortfin eel	9.13	9.86	1.00	18.15	67.74
Bluegill bully	14.07	6.36	0.80	11.72	79.46
Longfin eel	6.80	6.15	1.26	11.33	90.78

Groups Reference & Rehab
Average dissimilarity = 57.12

	Group Reference	Group Rehab				
Species	Av.Abund	Av.Abund	Av.Diss	Diss/SD	Contrib%	Cum.%
Common bully	10.56	19.13	18.19	1.30	31.85	31.85
Bluegill bully	0.56	14.07	9.77	1.15	17.11	48.97
Upland bully	16.89	17.13	7.85	1.10	13.74	62.71
Shortfin eel	7.67	9.13	6.74	0.91	11.79	74.50
Eel sp	2.00	4.33	5.35	0.80	9.37	83.88
Longfin eel	3.44	6.80	5.13	1.23	8.98	92.86