Styx River Catchment Aquatic Ecology 2018

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EXECUTIVE SUMMARY

This report describes the current state and trends in aquatic ecology and sediment quality of the Styx River, following the most recent round of monitoring in 2018.

Monitoring data from 2018 indicate that riparian and instream habitat quality is unchanged compared to previous years at most of the 5-yearly monitoring sites. The greatest change in riparian habitat was observed at the Kā Pūtahi Creek monitoring site at Blakes Road. This section of waterway was recently realigned and enhanced to make way for the Northern Arterial Motorway. The new alignment includes a mix of run, riffle and pool habitat, wood and boulders for fish habitat, and extensive native plantings within a very wide riparian zone.

Sediment concentrations of common stormwater contaminants exceeded ANZECC (2000) guidelines at a number of sites in 2018, but there were no increasing trends at most of the sites. Zinc is the contaminant of most concern in sediments, as it is the only parameter to regularly exceed the ISQG-High guideline. Sediment zinc concentrations have been consistently elevated over time at all of the tributary sites, reflecting their generally more urbanised sub-catchments compared to the other mainstem Styx River sites.

Invertebrate community composition in 2018 was similar to previous years at the 5-yearly monitoring sites, being dominated by pollution-tolerant snails and crustaceans that are common to rural and urban Christchurch waterways. However, the abundance and diversity of pollution-sensitive EPT taxa also remains greater in the Styx River catchment than in the Avon, Heathcote, and Halswell Rivers. In addition, EPT taxa richness more than doubled from 2013 to 2018 at the realigned section of Kā Pūtahi Creek at Blakes Road. There were no significant correlations ($P>0.05$) between water quality, sediment quality and invertebrate community ordination axis scores for 2018 invertebrate samples.

The range of fish species caught in 2018 was also similar to previous years and was dominated by native species, particularly shortfin eels. A change in fish sampling methods in 2018 to standard CCC protocols saw lower numbers of fish caught from wadeable sites and more fish caught from non-wadeable sites, but no overall change in the total number of fish species caught compared to 2013. The presence of lamprey at two of the monitoring sites in Styx Mill Conservation Reserve is notable, because of their Nationally Vulnerable conservation status (Dunn et al. 2018).

The Styx SMP surface water quality objectives for total macrophyte cover, filamentous algae cover, and fine sediment cover have been consistently met at most sites over the last ten years at the 5-yearly monitoring sites. The SMP objective for invertebrate QMCI scores has been met by six of the nine monitoring sites for each year of monitoring. Although QMCI scores have varied within sites over the years, there has been no overall increasing or decreasing trend in QMCI scores evident across the sites monitored. This indicates that the overall ecological health of the Styx River is stable and that there is no indication of a declining trend that could be attributable to stormwater discharges or other landuse impacts.

Data from the annual monitoring site at Styx Mill Conservation Reserve indicate no significant increasing or decreasing trends in macroinvertebrate community health or habitat over the six years of monitoring from 2013 to 2018.

The recent scientific confirmation of a widespread and abundant population of kākahi (freshwater mussels) in the lower Styx River is an exciting development. The Styx River kākahi population is locally, if not regionally significant, due to its extent and density, and

because kākahi numbers are declining nationally. The recent discovery of a significant lamprey population in the Styx catchment is also of considerable ecological interest. The presence of large numbers of ammocoetes (juvenile lamprey) in Canal Reserve Drain indicates that the Styx River supports a viable breeding population. The Canal Reserve Drain lamprey population is unique by virtue of the large number of ammocoetes present, their far greater abundance compared to other fish species present, and the highly modified nature of the habitat present.

Recommendations include: lamprey pheromone trapping (high priority); ongoing kākahi monitoring; investigate increasing sediment contaminant levels at the Styx River Harbour Road and Redwood Springs sites; care taken during waterway enhancement/realignment of Wilsons and Horners Drains, due to the presence of contaminated sediments; and continue using the now-standard CCC ecology sampling methods used in this report.

1. INTRODUCTION

The Styx River is located on the northern fringe of Christchurch city and its catchment includes a mixture of urban and rural landuse. Christchurch City Council (CCC) monitors aquatic ecology of the Styx River, both to fulfil stormwater discharge consent requirements under the Styx Stormwater Management Plan (SMP) and as part of its long-term environmental monitoring programme. The first two rounds of regular monitoring were in 2008 and 2013, and this report presents the most recent results, from 2018.

The purpose of this report is to present the results of the most recent ecology and sediment quality monitoring, describe the state of the monitored waterways, and identify any trends over time. The following key components are included in this report:

- Current state and trends in the 5-yearly aquatic ecology and sediment quality monitoring programmes.
- Trends in the annual aquatic ecology monitoring site at Styx Mill Conservation Reserve.
- Comparison of all of the above to relevant SMP standards and guidelines.
- Discuss any environmental trends in relation to potential stormwater impacts.
- Summarise recent significant kākahi (freshwater mussel) and lamprey population discoveries.

This report does not include a detailed analysis of the monthly water quality monitoring undertaken by CCC at eight sites in the Styx catchment. Those data are summarised separately as part of an annual city-wide summary report (Margetts & Marshall 2018).

2. METHODS

2.1. Sampling Sites

There are a total of 12 sites included in the 5-yearly aquatic ecology monitoring programme. These comprise nine wadeable sites upstream of Marshlands Road and three non-wadeable sites from Marshlands Road down to Kainga Road, just upstream of the Styx River tide gates (Figure 1, Table 1). The 5-yearly ecology sites have been monitored on three occasions: 2008, 2013, and 2018. There are also 12 sites for the 5-yearly sediment monitoring programme (Figure 1). There are four years of sediment quality monitoring data: 1980, 2009, 2014, and 2018. The annual aquatic ecology monitoring site is within the Styx Mill Conservation Reserve (Site 14 on Figure 1) and there are six years of data, from 2013 to 2018.

Adjacent landuse varies amongst sites, and comprises a mix of rural and reserve land with limited residential development. With the exception of Site 11, landuse and monitoring site locations were unchanged from 2008 and 2013. Site 11 is located just downstream of Blakes Road and the monitoring site is now located within the newly enhanced section of Kā Pūtahi Creek, which was realigned to avoid the new Northern Arterial motorway. Note that CCC has recently adopted the traditional māori name for Kā Pūtahi, although its official gazetted name on maps, old reports, and the Christchurch District Plan is Kaputone Creek.

Ecology monitoring occurred from 12 March to 3 April 2018, under baseflow conditions.

Figure 1: CCC Styx River ecology and sediment quality monitoring sites.

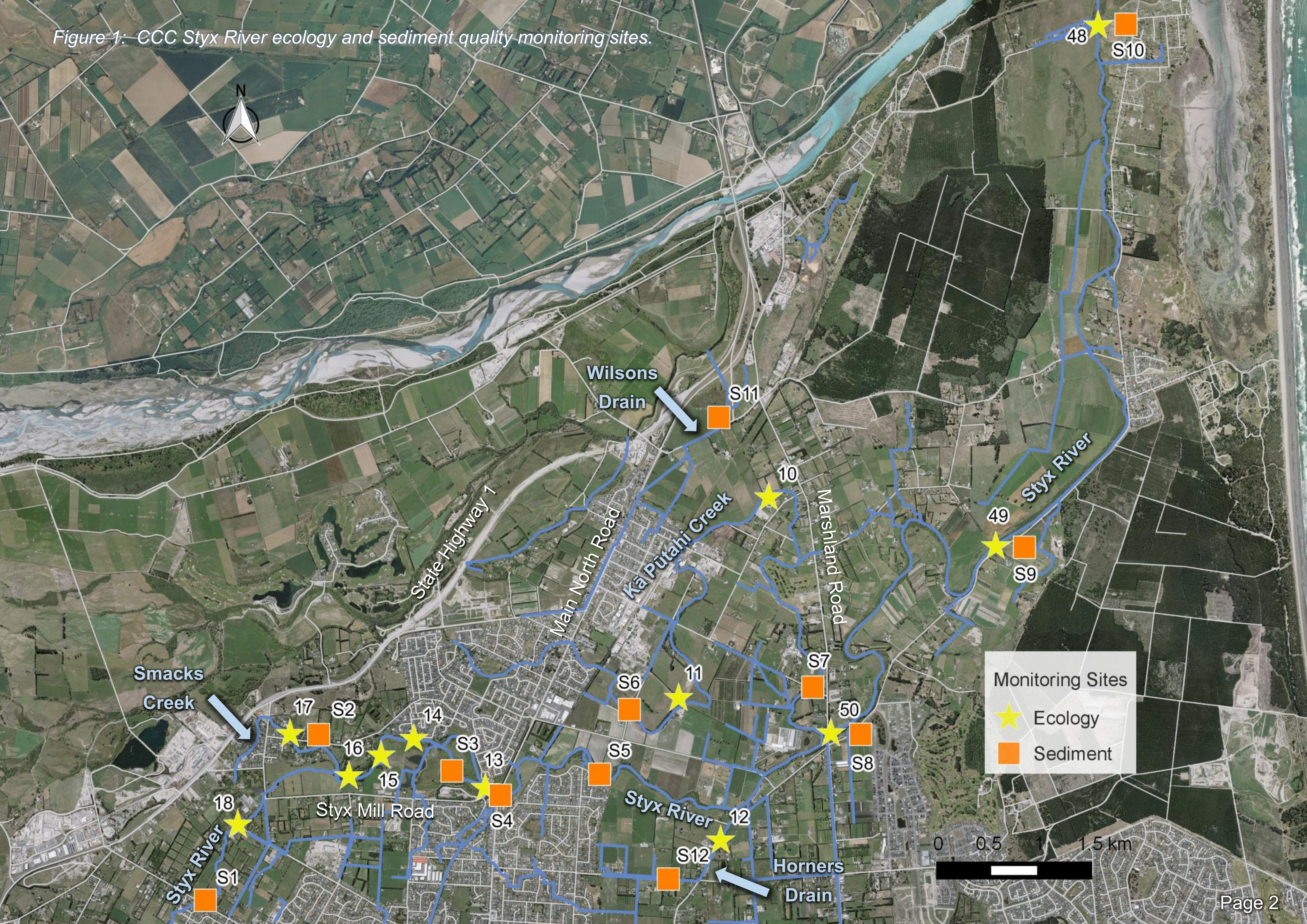


Table 1: Styx catchment ecology and sediment quality monitoring sites and their locations.

Site Code	Waterway	Site Name/Location	Easting	Northing
Ecology Monitoring Sites				
10	Kā Pūtahi Creek	Ouruhia Reserve	1571754	5190116
11	Kā Pūtahi Creek	Between Blakes and Belfast Roads	1570871	5188163
12	Horners Drain	Hawkins Road	1571292	5186781
13	Styx River	Main North Road	1568961	5187280
14	Styx River	Styx Mill Conservation Reserve	1568252	5187755
15	Styx River	Adjacent to Styx Mill Dog Area car park	1567927	5187591
16	Styx River	Styx at Dog Park	1567611	5187388
17	Smacks Creek	Hussey Road	1567026	5187795
18	Styx River	Claridges Road	1566513	5186913
48	Styx River	Kainga/Harbour Road	1575000	5194714
49	Styx River	Richards Bridge	1574005	5189650
50	Styx River	Marshlands Road	1572380	5187826
Sediment Monitoring Sites				
S1	Styx River	Sawyers Arms Road	1566195	5186178
S2	Smacks Creek	Husseys Road	1567073	5187748
S3	Styx River	Styx Mill Conservation Reserve	1568629	5187446
S4	Styx River	Main North Road	1569110	5187212
S5	Styx River	Redwood springs	1570092	5187420
S6	Kā Pūtahi Creek	Blakes Road	1570384	5188050
S7	Kā Pūtahi Creek	Belfast Road	1572199	5188275
S8	Styx River	Marshland Road	1572363	5187786
S9	Styx River	Richards Bridge	1574009	5189609
S10	Styx River	Kainga/Harbour Road	1575007	5194744
S11	Wilsons Drain	Otukaikino Memorial Reserve	1571258	5190892
S12	Horners Drain	Prestons Road	1570776	5186405

Note: Eastings and northings use the New Zealand Transverse Mercator 2000 (NZTM2000) projection.

2.2. New Sampling Methods for 2018

Previous monitoring involved sampling at 12 sites (9 wadeable, 3 non-wadeable), using more detailed methods than the now-standard CCC ecology sampling method (McMurtrie & Greenwood 2008, James 2013). As of the 2018 sampling round, the standard CCC ecology sampling methods are being used at the same 12 sites previously sampled, with the addition of sediment quality sampling at the same time. This section summarises similarities and differences between the methods, while the next sections detail the new standard methods.

Both new and old methods involve:

- Measuring habitat along a 20 m reach, with detailed measurements along 3 transects.
- Dissolved oxygen, temperature, pH, and conductivity measured once per site.

The major differences between the new and old methods are as follows:

- At each transect, detailed habitat measurements at:
  - 12 points (old method). Entire width sampled using kayak at non-wadeable sites.
  - 5 points (new method). Only edge habitat sampled at non-wadeable sites.
- At each transect, velocity measured at:
  - 10 points (old method).
  - 1 point (new method). Mid-channel for wadeable sites; approx. 1.5 m (safely wadable) from edge for non-wadeable sites.
- Invertebrate kicknet samples per site:
  - 3 (old method). Each sample is approx. 0.45 m² (1.5 x 0.3 m). Entire width sampled from kayak at non-wadeable sites.
  - 1 (new method). Each sample is approx. 1.5 m² (chosen to approximate previous 3 kicknets). Only edge habitat sampled at non-wadeable sites.
- Fish sampling effort for wadeable sites:
  - Old method: 2-pass electrofishing over 20 m reach
  - New method: 1-pass electrofishing over minimum 30 m reach
- Fish sampling effort for non-wadeable sites:
  - Old method: 5 unbaited fyke nets (15 mm stretched mesh) and 5 unbaited gee minnows
  - New method: 2 fyke nets (4 mm mesh, double-trap – per Joy et al. (2013)) baited with cat food, 5 gee minnows baited with marmite.

2.3. Habitat and Water Quality Sampling

At three representative transects located 10 metres apart, the following were collected:

- Bank and riparian habitat (for each bank for a 5 metre bank width): surrounding land use, bank material, bank height, bank erosion, bank slope, riparian vegetation, canopy cover, undercut banks, overhanging vegetation and ground cover vegetation

- Instream habitat (for five locations across each transect): wetted width, water depth, fine sediment depth, embeddedness and substrate composition using the following size classes: silt/sand (<2 mm); gravels (2-16 mm); pebbles (16-64 mm); small cobbles (64-128 mm), large cobbles (128-256 mm), boulders (256-4000 mm) and bedrock/concrete/artificial hard surfaces (>4000 mm) (modified from Harding et al., 2009).

Substrate composition data was converted to a substrate index to aid comparison of data amongst sites and over years. The substrate index was calculated using the following formula (modified from Harding et al. 2009):

Substrate index (SI) = (0.03 x %silt / sand) + (0.04 x %gravel) + (0.05 x %pebble) + (0.06 x (%small cobble + %large cobble)) + (0.07 x %boulder).

Water velocity was measured once per transect at the mid-channel using a Seba Mini velocity meter. At the reach scale, the relative percentage of riffle, run, and pool flow habitat was estimated visually.

Field measurements were taken of dissolved oxygen, water temperature, pH and conductivity in an area representative of the site (usually mid-channel). The water quality measurements were made using calibrated TPS water quality meters (model WP-82Y for dissolved oxygen and temperature; model WP-81 for pH and conductivity).

Macrophyte cover and composition, depth and type (emergent and total) was measured at five locations across each of the three transects. Periphyton cover and composition was also measured at the five locations across each of the three transects. Periphyton categories were be adapted from those outlined in Biggs & Kilroy (2000). These include thin mat forming algae, medium mat forming algae, thick mat forming algae, short filamentous algae and long filamentous algae. Percentage cover and description of organic matter was also recorded.

2.4. Sediment Quality

Sediment samples were collected by making multiple sweeps with a sampling container across the stream bed, with at least five subsamples composited into one sample, preferably of at least 1 kilogram. Sampling aimed to collect texturally similar sediment between sites, with the preferential collection of fine sediments (<2 mm) to ensure sufficient material for laboratory analysis. Samples were collected from the surface at a depth of no greater than 3 cm. Water was drained off directly from the jars.

After collection, samples were placed in a chilly bin containing ice-bricks and transported to Hill Laboratories (an International Accreditation New Zealand laboratory) within 24 hours. Samples stored overnight we kept chilled in a refrigerator.

Sediment samples were be analysed at all sites for the following using the most relevant US EPA methods and the <2 mm fraction (where relevant), with the detection limits for each parameter suitable to enable comparison of the results with relevant guideline levels and previous monitoring:

- Particle size distribution using the following size classes: silt and clay (<0.063 mm); fine sand (0.063-0.25 mm); medium sand (0.25-0.50 mm); coarse sand (0.5-2.0 mm); gravel and cobbles (>2 mm).
- Total recoverable copper, lead and zinc.
- Total organic carbon.
- Polycyclic aromatic hydrocarbons (PAHs).

Samples at six of the sites (Styx River at Sawyers Arms Road, Styx River at Main North Road, Kaputone Creek at Blakes Road, Styx River at Marshlands Road, Styx River at Kainga Road and Wilsons Drain at Ōtūkaikino Memorial Reserve) were also sampled for the following agricultural contaminants:

- Total recoverable arsenic
- Semi-volatile organic compounds (SVOCs) – including phenols, polycyclic aromatic hydrocarbons, organochlorine pesticides and plasticisers.

Sediment sampling fieldwork was undertaken during baseflow conditions on 18 March 2018.

2.5. Macroinvertebrates

Benthic macroinvertebrates were sampled at each site by collecting a single kicknet sample from the range of available habitats present, in proportion to the habitat types present, and covering a total area of approximately 1.5 m^2. Samples were preserved in the field using denatured ethanol and were sent to Biolive consultants for identification and enumeration. All invertebrates were counted and identified to species level where possible, using protocol P3 (full count with subsampling) of Stark et al (2001).

2.6. Fish

At the nine wadeable sites the fish community was sampled using backpack electric fishing, while a combination of fyke nets and Gee minnow traps were used to sample fish at the three non-wadeable sites. For the nine wadeable sites, the length of stream electric fished at each site was a minimum of 30 m and 30 m^2 in area. All habitat types within the reach were sampled without bias (e.g., pools, riffles, underhangs and backwaters). For the three non-wadeable sites, sampling involved deploying five Gee Minnow traps baited with marmite and two fyke nets (4 mm mesh and two internal traps, as per Joy et al. (2013)) baited with cat food. Fyke nets were set at a 15° – 30° angle to the bank, with the leader downstream. Nets and traps were left overnight and checked the following morning.

For both trapping and electric fishing, all fish caught were identified to species level where possible, counted, measured and released back into the waterway. Fish seen but not caught were recorded as missed fish (e.g. 'missed bully' or 'missed fish' if identification was uncertain), but not included in the total tally.

2.7. Data Analyses

2.7.1. Data Management

All ecology and sediment quality data collected in 2018 was collated into a single Excel spreadsheet. In addition, summary data from 2018 and all previous years of ecology and sediment monitoring (data provided by CCC) were combined into a single Microsoft Excel spreadsheet. Both spreadsheets were provided to CCC in electronic form at the time this report was submitted, and they are available from CCC on request.

2.7.2. Habitat and Water Quality Data

Field-measured water quality results were tabulated and compared against relevant freshwater outcomes and receiving water standards in the Canterbury Land and Water Regional Plan (LWRP).

Relevant habitat data that were chosen for statistical analyses included the following parameters: channel width, water depth, water velocity, substrate index, fine sediment (<2 mm diameter) depth, fine sediment cover, and bed cover with emergent macrophytes, total macrophytes, and long filamentous algae (>2 cm long). Of these parameters, stormwater consent water quality objectives are associated with fine sediment cover, emergent macrophytes, total macrophytes and long filamentous algae (Table 2).

Prior to 2018, there were single, site-wide estimates for emergent and total macrophyte cover, long filamentous algae cover and fine sediment cover (estimated by summing estimated cover of sediment <2 mm). In 2018, these parameters were estimated as per other transect data (i.e., the average of five measures per transect, and the site average obtained by the mean of three transects).

Habitat data were averaged for each transect (where relevant), plotted, compared with SMP water quality objectives, and inspected for evidence of any patterns over time or amongst sites.

Table 2: Styx Stormwater Management Plan (SMP) surface water quality objectives.

Parameter	Surface water quality objective
Minimum QMCI	4.5
Maximum fine sediment (<2 mm) cover	40%
Maximum total macrophyte cover	50%
Maximum filamentous algae cover	30%

Differences amongst sites over time were assessed using two-way analysis of variance (ANOVA) for the following parameters: width, depth, velocity, substrate index, and sediment depth. Tukey post-hoc tests were used to examine the statistical significance of any site x year interactions, particularly in terms of any increasing or decreasing trends in habitat quality over time.

It was not possible to use ANOVA to test for trends in fine sediment cover, emergent macrophyte cover, or total macrophyte cover over time, due to a lack of replication in previous years. Trend analysis using tests such as the Mann-Kendall trend test were also not possible, as they typically require more than three years of record. Therefore, these data were just examined visually for any indication of trends.

2.7.3. Sediment Quality Data

Particle size data from the laboratory was converted into a modified substrate index, to allow for easy comparison in particle size amongst sites and over time. Particle size categories common to all four years of monitoring were as follows: silt and clay (< 0.063 mm); fine sand (0.063-0.25 mm); medium sand (0.25-0.5 mm); coarse sand (0.5-2.0 mm). The modified substrate index (modified SI) was calculated as follows:

Modified SI = (0.01 x %silt and clay) + (0.02 x %fine sand) + (0.03 x %medium sand) + (0.04 x %coarse sand).

Total PAHs were calculated by summing the same 16 PAHs analysed in previous monitoring rounds, which include the PAHs listed as priority pollutants by the USEPA (1982). Total PAHs were normalised to 1% TOC, as recommended by ANZECC (2000), before comparison to the guidelines. Where one or more PAH compound was below the detection limit, half the detection limit was used in the calculation, which is consistent with previous reports (Whyte 2014, Boffa Miskell 2017).

Sediment quality data from the 12 sites sampled in 2018 were summarised and tabulated for comparison against ANZECC (2000) interim sediment quality guidelines (ISQG). Sediment quality data from 2018 were also compared against data collected in 1980, 2009, and 2014, using historic data provided by CCC. Historic Data were available for all sites except Smacks Creek.

Trends over time were examined statistically using the Mann-Kendall trend test in Time Trends statistical software (version 6.10). The Mann-Kendall test is appropriate when there is no seasonal variation in the observations. The method does not assume that the time series being tested is at regular intervals and missing values are allowed, making it appropriate to use in this situation. Because only four time periods are available for sediment monitoring (1980, 2008, 2013, and 2018), any trends are interpreted with caution.

2.7.4. Macroinvertebrates

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

Total Abundance: The total number of invertebrates per sample. Total abundance may be reduced by sedimentation, but is not a reliable metric for kicknet samples, due to variable sampling area.

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

%EPT: The percentage of all individuals collected made up of pollution-sensitive Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly) taxa. %EPT is typically reduced at polluted sites, and is particularly sensitive to sedimentation. This metric was calculated excluding pollution-tolerant hydroptilid caddisflies, which can skew %EPT results at sites where they are abundant.

EPT Taxa Richness: The number of different EPT taxa at a site. It is reduced at polluted sites. Calculated without hydroptilid caddisflies included.

MCI and QMCI: The Macroinvertebrate Community Index and the Quantitative MCI (Stark 1985). Invertebrate taxa are assigned scores from 1 to 10 based on their tolerance to organic pollution. Highest scoring taxa (e.g., many EPT taxa) are the least tolerant to organic pollution. The MCI is based on presence-absence data: scores are summed for each taxon in a sample, divided by the total number of taxa collected, then multiplied by a scaling factor of 20. The QMCI requires abundance data: MCI scores are multiplied by abundance for each taxon, summed for each sample, then divided by total invertebrate abundance for each sample. We calculated site MCI and QMCI scores using the tolerance scores for hard-bottomed streams for Sites 10 to 17 and soft-bottomed streams for Sites 18, 48, 49, and 50, to reflect the dominant substrate present (Stark & Maxted 2007). MCI and QMCI scores can be interpreted as per the quality classes of Stark & Maxted (2007), as summarised in Table 3.

Table 3: Interpretation of MCI and QMCI scores (from Stark & Maxted 2007).

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

As with reach-scale habitat data, it was not possible to conduct two-way ANOVA or trend analyses on the five-yearly macroinvertebrate data, due to a lack of replication in the new sampling method and because there are insufficient monitoring dates to conduct trend analysis. However, there is now six years of data for the annual Styx Mill monitoring site (2013-2018), so the Mann Kendall test was used to assess trends at this site. Statistical results from the Styx Mill monitoring site thus provide a reasonable indicator of changes over time.

One-way ANOVA was used to assess whether there were significant differences in macroinvertebrate indices amongst sampling years, using sites as replicates. The primary purpose of this was to determine whether there was any change in the total number of invertebrates collected using the new methods in 2018. This is of interest, because the total number of taxa collected increases with overall sample size. Hence, a change in the number of invertebrates collected could affect diversity-related indices, particularly overall taxa richness and EPT taxa richness.

Macroinvertebrate community composition was also compared amongst sites and over time using non-metric multi-dimensional scaling (NMDS), a form of ordination. The ordination was based on a Bray-Curtis dissimilarity matrix, using square-root transformed data and the Ecodist package in R. Spearman rank correlation was used to reveal which taxa most closely correlated with NMDS axis scores. Habitat data from the nine wadeable sites were also correlated with NMDS axis scores.

There were six sites for which sediment, ecology and monthly water quality monitoring sites are in close proximity. These are ecology sites 11, 13, 17, 48, 49, and 50. For these sites, NMDS axis scores for 2018 were correlated against sediment quality data (copper, lead and zinc), and key median water quality data from April 2017 to March 2018 (dissolved copper, dissolved lead, dissolved zinc, total suspended solids, turbidity, dissolved reactive phosphorus, and dissolved inorganic nitrogen). Median water quality data was based on monthly water quality samples and the data were provided by CCC.

QMCI scores were compared with the surface water quality objective of a minimum QMCI of 4.5 for the Styx SMP (Table 2).

3. RESULTS

3.1. Five-Yearly Monitoring Data

3.1.1. Habitat and Water Quality

Water temperatures were cool and pH near-neutral at all sites sampled in 2018 (Table 4). All sites had cooler temperatures than the LWRP freshwater outcome of 20 °C. Dissolved oxygen saturation exceeded (i.e., complied with) the LWRP freshwater outcome of 70% at all sites except for Smacks Creek, the Styx River at Claridges Road, the Styx River at Richards/Teapes, and the Styx River at Marshlands Road (Table 4). Lower oxygen levels at these sites likely reflects their proximity to headwater springs (for Smacks Creek and Styx River at Claridges Road), or the sluggish flow and high macrophyte cover (for the Styx River Richards/Teapes and Marshlands Road sites). All sites had pH values within the LWRP Receiving Water Standard of 6.5 to 8.5 (Table 4). Conductivity did not vary greatly amongst sites, but was highest at the Styx River Harbour Road site (196 µS/cm), which is the most downstream site and is tidal, and conductivity was lowest at the Styx River Claridges Road site (123 µS/cm), which is the most upstream site. Conductivity was generally higher in Kā

Pūtahi Creek, which likely reflects the impact of heavy industrial land in the upstream catchment on water quality. There are no SMP or LWRP guidelines for conductivity.

Table 4: Water quality measured at the 12 ecology monitoring sites.

Site number	Site name	Dissolved oxygen (%)	Temperature (°C)	pH	Conductivity (µS/cm)
10	Kā Pūtahi at Ouruhia reserve	88	15.1	7.6	157
11	Kā Pūtahi at Blakes	105	16.0	7.2	167
12	Horners Drain	83	14.2	7.4	148
13	Styx at Main North Road	94	14.6	7.5	133
14	Styx at Glen Oaks	86	13.8	7.0	131
15	Styx at Reserve	87	14.7	7.3	131
16	Styx at Dog Park	81	14.5	7.3	132
17	Smacks at Hussey Road	55	15.4	7.4	149
18	Styx at Claridges	56	13.4	7.3	123
48	Styx at Harbour Road	72	16.8	7.5	196
49	Styx at Richards/Teapes	55	15.8	7.3	137
50	Styx at Marshlands	62	14.8	7.4	144
LWRP Freshwater Outcome or Receiving Environment Standard		>80	<20	6.5-8.5	–

Adjacent landuse and riparian habitat remains largely unchanged in 2018 compared with 2013 at most sites (James 2013). All of the upper Styx River sites, the two Kā Pūtahi Creek sites and the Smacks Creek site are all located within reserve areas of varying sizes. Horners Drain (Site 12) has the most highly modified riparian and bank habitat, being a timber-lined waterway located within road reserve (Figure 2). Smacks Creek (Site 17) and Styx River at Claridges Road (Site 18) are the best shaded, with complete canopy cover along sections of the waterway (Figure 2). The character of the river changes with distance downstream, becoming broader, more open and slow-flowing, with a mixture of pasture and willows in the riparian zone (Figure 3). See Appendix 1 for photographs of all of the sites in 2018.

Figure 2: Contrasting riparian and bank habitat conditions in tributary waterways, with Horners Drain (Site 12) on the left and Smacks Creek (site 17) on the right.

Figure 3: Images from the upper and lower Styx River. Site 14 is within the Styx Mill Conservation Reserve and is the annual monitoring site (left), while Site 49 is in the lower, non-wadeable reaches of the river (right).

The only site with a marked habitat difference in riparian and instream habitat between 2013 and 2018 was Kā Pūtahi Creek at Blakes Road (Site 11). That is because this section of the waterway has been realigned and undergone a major restoration project, to avoid the new Northern Arterial Motorway. Riparian vegetation in the old alignment comprised a mix of exotic trees with some native sedges bordered by agricultural land with moderate shade. In contrast, the new alignment flows through extensive native plantings (particularly *Carex* spp.) and is situated within a new council reserve (Figure 4). There is currently relatively low to moderate channel shading over the new alignment, but numerous large tree specimens will grow over time to provide improved shading and riparian habitat.

Figure 4: Kā Pūtahi Creek at Blakes Road (Site 11) in 2013 (left) and in 2018 (right) after realignment and enhancement. The photograph on the left was taken by EOS Ecology for CCC.

The mainstem Styx River sites are generally wider and deeper than the tributaries, with the exception of Site 10 (Kā Pūtahi Creek at Ouruhia Reserve), which is relatively broad and deep (Figure 5). At the wadeable sites, mean width across all years ranges from 1.8 m at Site 11 (Kā Pūtahi Creek at Blakes Road) up to 6.1 m at Site 13 (Styx River at Main North Road), while mean depth ranges from 10 cm at Site 11 to 39 cm at Site 18 (Styx River at

Claridges Road; Figure 5). Two-way ANOVA revealed significant differences amongst sites for width (P<0.001) and depth (P<0.001), and a weak difference amongst years for width (P=0.04), but a large difference amongst years for depth (P<0.001), reflecting greater water depths in 2013 and 2018 (Figure 5). There was no significant site x year interaction for width (P>0.05), but there was for depth (P=0.001). Figure 5 shows increasing water depths over time at a number of sites (supported by numerous significant Tukey test comparisons); this either reflects different river flows in the different monitoring years or increasing depths over time. There is anecdotal evidence suggesting that the 2010/2011 Canterbury Earthquakes resulted in a deepening of the upper reaches of the Styx River (Belinda Margetts, CCC Waterways Ecologist, pers. comm.), however that would not explain the increase in water depth between 2013 and 2018 monitoring years. Therefore, differences in water depth between years are more likely simply due to flow differences between the sampling periods.

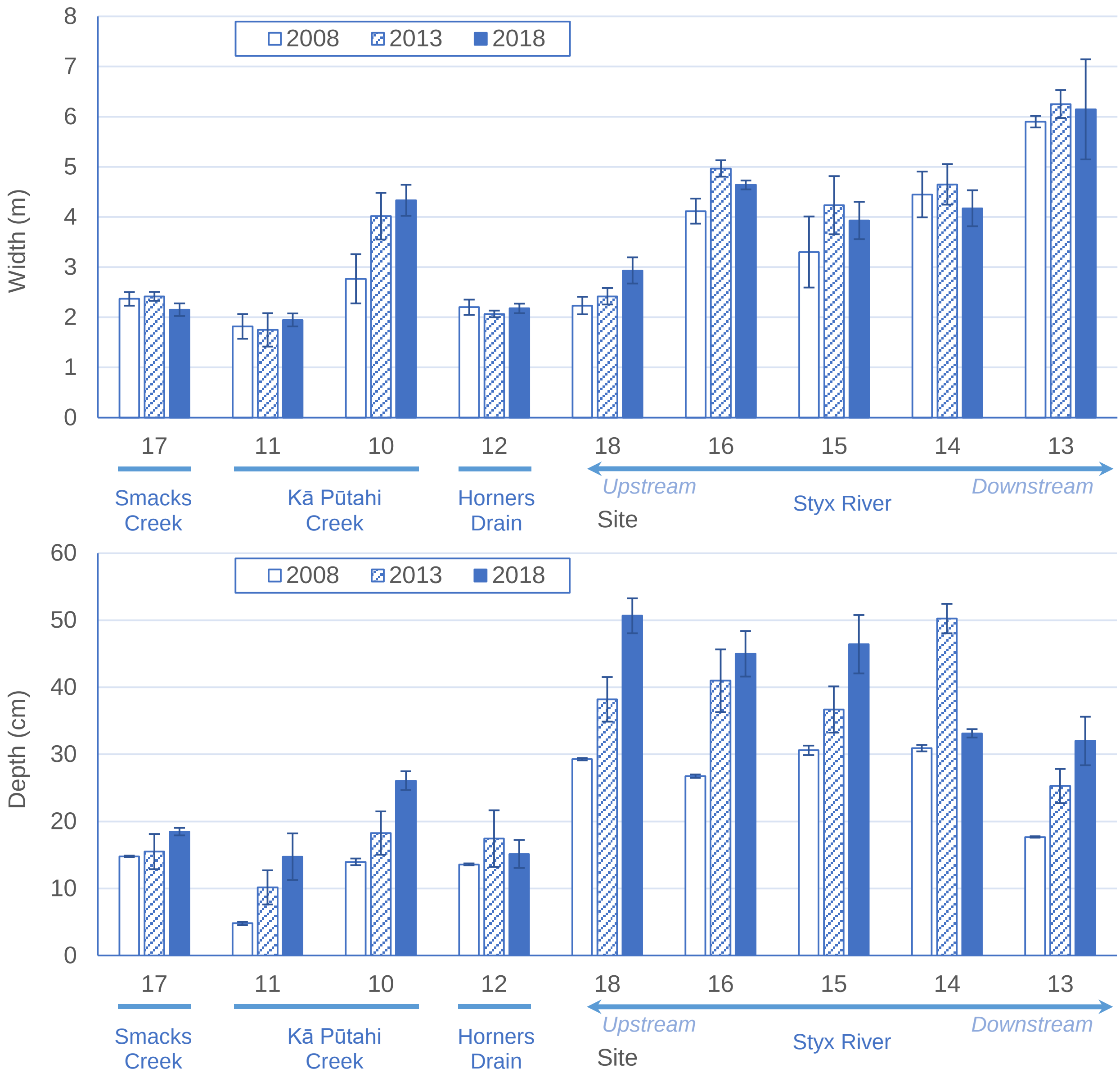


Figure 5: Mean (±1 SE) width (upper) and water depth (lower) at the nine wadeable sites.

Water velocity varies from site to site, with no strong pattern of differences between tributary and mainstem sites (Figure 6).Mean velocity at the wadeable sites across all years ranged from 0.25 m/s at Site 11 (Kā Pūtahi at Blakes Road) to 0.60 m/s at Site 13 (Styx River at Main North Road). Two-way ANOVA revealed significant differences amongst sites ($P<0.001$) and years ($P=0.002$), with a weak site x year interaction effect ($P=0.04$). Water velocities were generally greater in 2018, reflecting higher base flows during sampling compared to previous years.

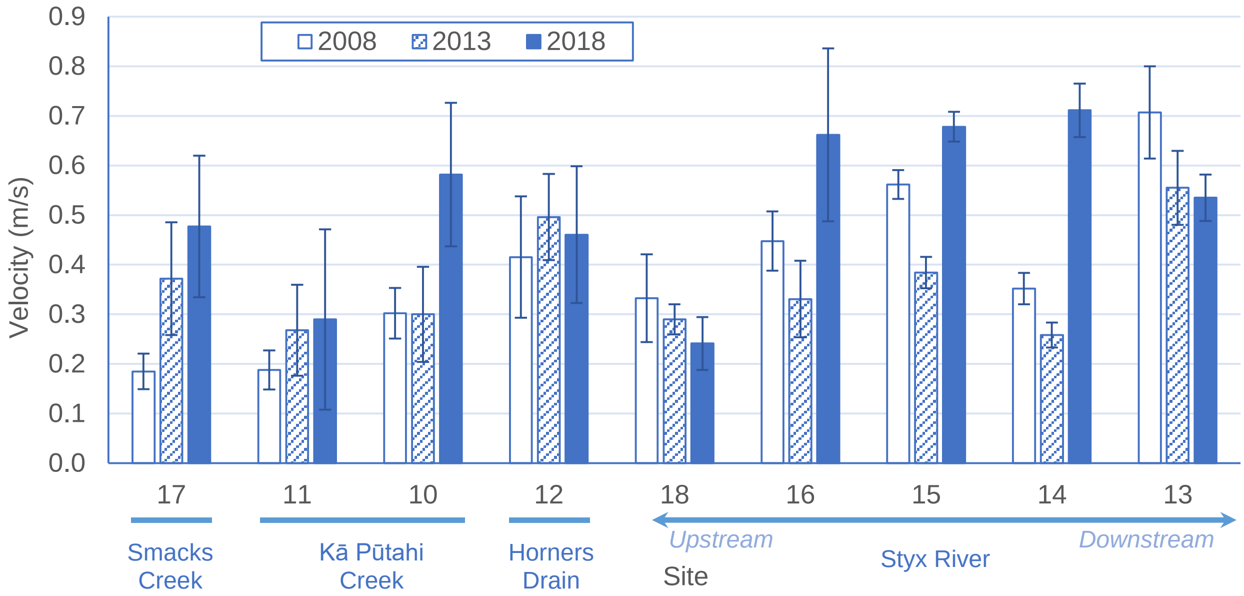


Figure 6: Mean (±1 SE) water velocity at the nine wadeable sites.

Bed sediments are dominated by coarse substrates (gravel and pebble) at all of the wadeable sites, with the exception of Site 18 (Styx at Claridges Road), which is dominated by fine sediments (<2 mm; Figure 7). Two-way ANOVA revealed significant differences amongst sites ($P<0.001$), no overall difference amongst years ($P>0.05$), and a weak site x year interaction ($P=0.047$). Apparent downward trends in substrate size for Sites 17, 10, and 18 shown in Figure 7 are not statistically significant (i.e., Tukey post-hoc comparisons across years for each site are all $P>0.05$). Site 11 (Kā Pūtahi at Blakes Road) was the only site with an increase in substrate index in 2018, indicating slightly coarser bed sediments, however this increase was not statistically significant (Tukey post-hoc $P>0.05$).

Fine sediment depth varies greatly amongst sites, ranging from a mean across years of approximately zero for Site 17 (Smacks Creek), Site 11 (Kā Pūtahi Creek at Blakes Road), and Site 12 (Horners Drain), up to 13 cm at Site 18 (Styx River at Claridges Road; Figure 8). Two-way ANOVA revealed significant differences amongst sites ($P<0.001$) and years ($P=0.008$), but no significant site x year interaction ($P>0.05$). Tukey post-hoc comparisons indicated greater fine sediment depth in 2013 than 2008 ($P=0.007$), but no significant difference between 2018 and either 2013 or 2008 ($P>0.05$); thus, there is no indication of an overall increasing fine sediment depths over time across all sites. There is an overall increasing trend in fine sediment depth apparent in Figure 8 for Site 18 (Styx River at

Claridges Road), but Tukey post-hoc comparisons revealed these differences were not statistically significant (P>0.05).

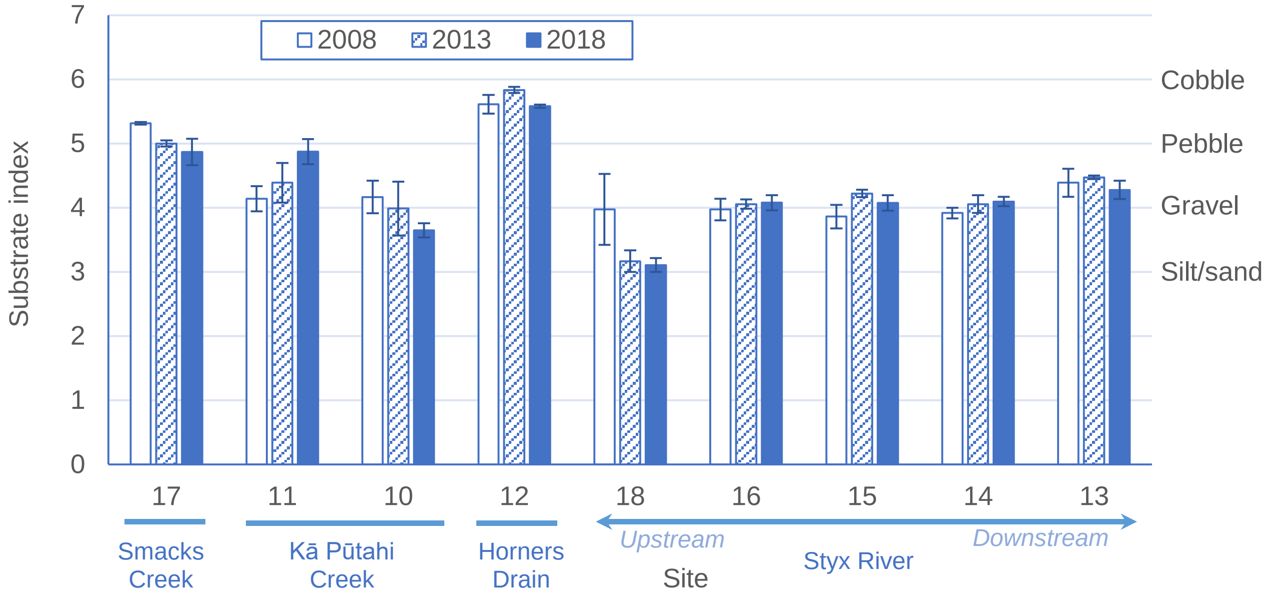


Figure 7: Mean (±1 SE) substrate index score at the nine wadeable sites.

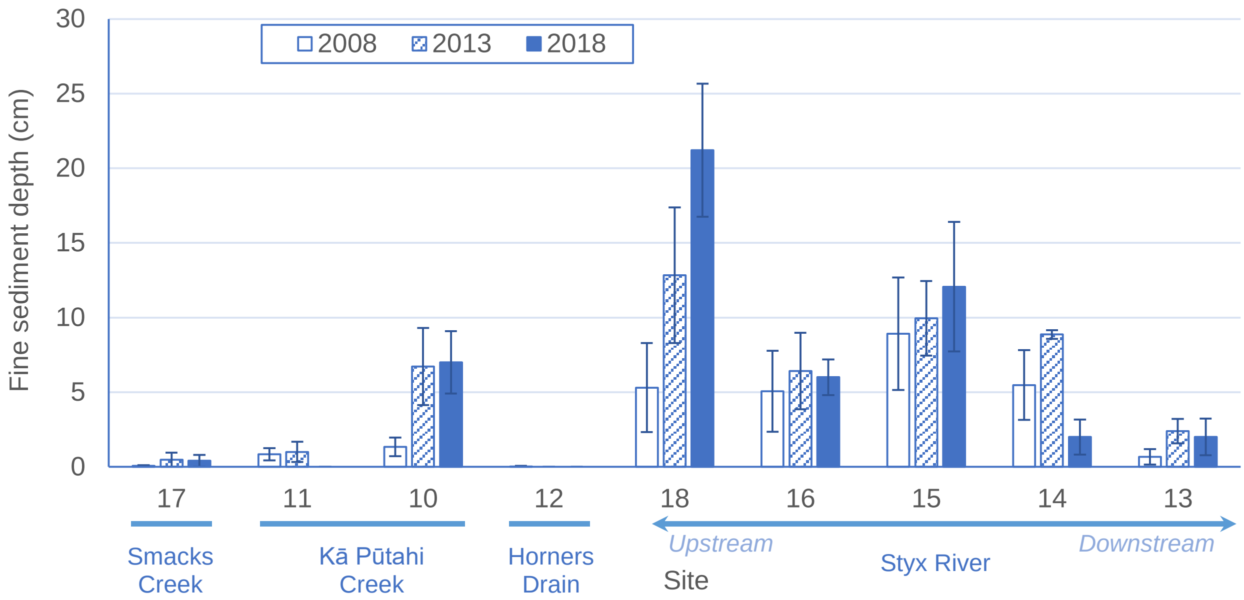


Figure 8: Mean (±1 SE) depth of fine sediment (<2 mm diameter) at the nine wadeabe sites.

Bed cover with fine sediment (<2 mm diameter) is generally lowest at the tributary sites and greatest at Site 18 (Styx River at Claridges Road). Site 18 has exceeded the SMP objective of 40% cover for all monitoring years, and had 95% cover in 2018 (Figure 9). The only other sites to exceed the SMP objective in 2018 were Site 10 (Kā Pūtahi Creek at Ouruhia Reserve), with 66% cover, and Site 16 (Styx River at Styx Mill Reserve Dog Park), with 40% cover. There is no obvious increasing or decreasing trend in fine sediment cover amongst the sites.

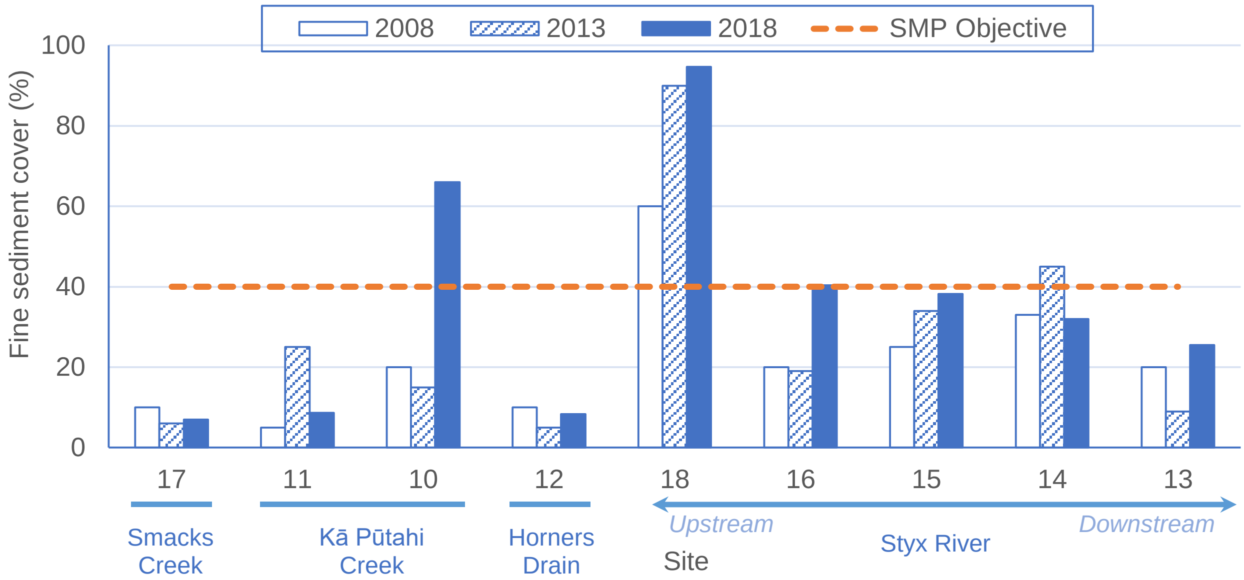


Figure 9: Bed cover with fine sediment (<2 mm diameter) at the nine wadeable sites in comparison to the SMP water quality objective of 40%.

Most of the wadeable sites have relatively low cover with macrophytes, reflecting the combination of reasonable shading and predominantly coarse bed sediments (Figure 10 and Figure 11). In 2018, the LWRP freshwater outcome of 30% cover of emergent macrophytes was only exceeded at Site 10 (Kā Pūtahi Creek at Ouruhia Reserve), which had 53% cover. Similarly, the SMP objective of 50% total macrophyte cover was only exceeded at Site 10, which had 63% cover. There is no obvious increasing or decreasing trend in emergent or total macrophyte cover amongst the sites (Figure 10 and Figure 11). However, total macrophyte cover at Site 11 (Kā Pūtahi Creek at Blakes Road) was higher in 2018 than in previous years. Increased macrophyte cover in 2018 reflects the waterway's recent realignment and lack of shade provided by the young native plantings. Presumably shading will increase and macrophyte cover will decrease as the native plants mature.

Bed cover with long filamentous algae (>2 cm) has been very low at all sites and 2018 was not different, with less than 10% cover at all sites. The SMP water quality objective of 30% was therefore complied with at all sites in 2018.

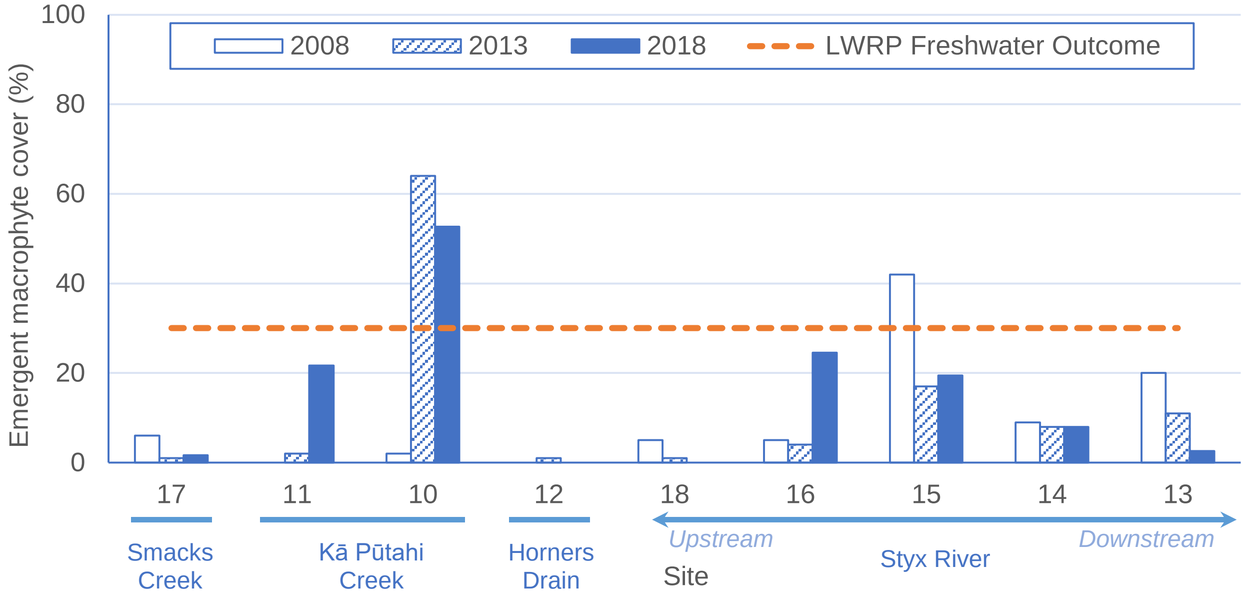


Figure 10: Bed cover with emergent macrophytes in comparison with the LWRP freshwater outcome of 30% for spring-fed streams.

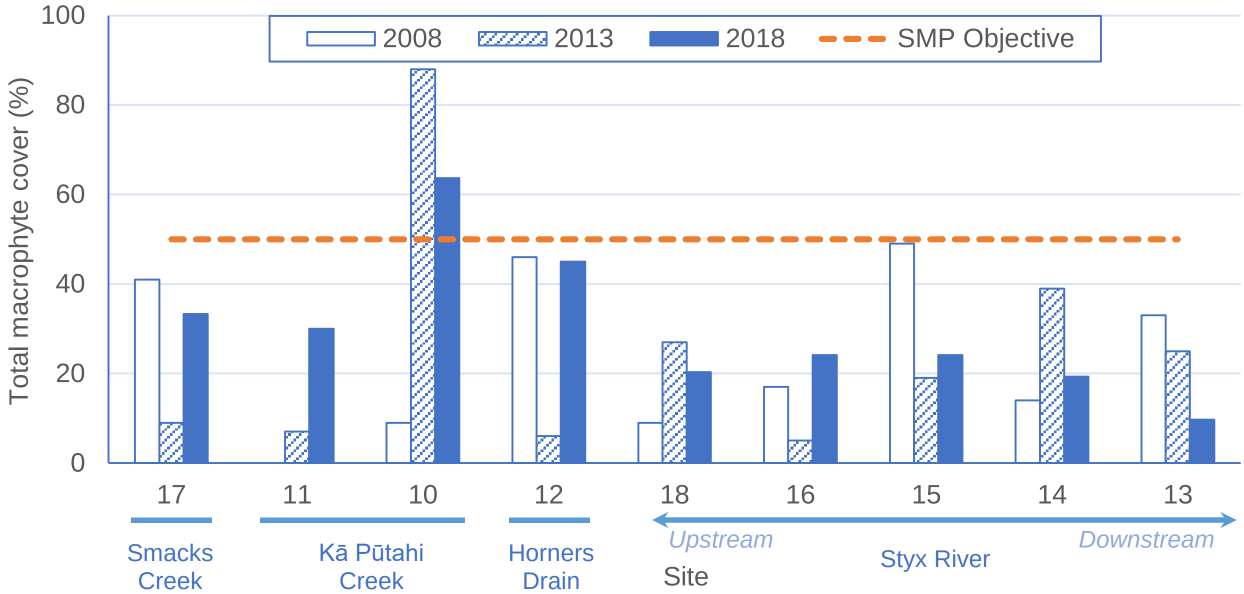


Figure 11: Bed cover with total macrophytes in comparison with the SMP water quality objective of 50%.

3.1.2. Sediment Quality

Sediment quality data from 2018 is summarised in Table 5 and all laboratory results are provided in Appendix 2. Laboratory-analysed sediments from all sites were dominated by particles in the range of fine sand, with corresponding modified substrate index (SI) values falling between 1 (silt/clay) to 2 (fine sand; Table 5). Total organic carbon (TOC) content varied amongst sites, ranging from from a low of 0.54 g/kg for the Styx at Marshlands Road site, up to 15.8 g/kg in Smacks Creek. These variations in TOC content likely reflect the influences of a combination of underlying geology, adjacent landuse, and local hydrology.

In 2018 zinc was the parameter that most frequently exceed guideline values, with the ISQG-Low trigger value exceeded at four sites and the ISQG-High guideline exceeded at two sites: Wilsons Drain and Horners Drain (Table 5). The ISQG-Low trigger value was exceeded at three sites for lead (Styx at Sawyers Arms Road, Wilsons Drain, and Horners Drain), one site for arsenic (Wilsons Drain), and one site for total PAHs (Styx at Marshlands Road). Copper concentrations were well below guideline values at all sites (Table 5).

Table 5: Sediment quality at monitoring sites in 2018. Units are mg/kg dry weight, except for total organic carbon (TOC), which is g/100 g dry weight, and substrate index (SI), which is unitless. Values exceeding the ANZECC (2000) Interim Sediment Quality (ISQG)-Low are in orange font and those exceeding ISQG-High are in red.

Site Code	Site	Arsenic	Copper	Lead	Zinc	TOC	SI	Total PAHs
S1	Styx at Sawyers Arms Rd	3.8	24	**99**	137	3.2	1.7	0.35
S2	Smacks Creek	-	44	28	161	15.8	2.0	0.03
S3	Styx at Styx Mill Reserve	-	21	27	**220**	14.3	1.7	0.01
S4	Styx at Main North Rd	2.3	4.4	11.9	61	1.48	1.6	0.66
S5	Styx at Redwood Springs	-	26	38	110	3.2	1.9	0.05
S6	Kā Pūtahi at Blakes Rd	9.8	6.3	17.3	**230**	1.68	2.0	0.97
S7	Kā Pūtahi at Belfast Rd	-	23	47	**280**	10.5	1.9	0.04
S8	Styx at Marshlands Rd	5.5	6.2	17.1	61	0.87	1.5	**19.38**
S9	Styx at Richards Bridge	-	5.9	9.1	50	0.54	1.9	ND
S10	Styx at Kainga Rd	12.9	24	29	**210**	4	2.0	ND
S11	Wilsons Drain	**30**	33	**51**	**430**	8.9	1.7	0.55
S12	Horners Drain	-	32	**61**	**790**	3.4	1.8	1.03
ISQG-Low		20	65	50	200	N/A	N/A	4
ISQG-High		70	270	220	410	N/A	N/A	45

Notes: Total PAHs are normalised to 1% TOC. ND indicates sites where all PAHs were less than detection limits, while "-" indicates sites where arsenic was not analysed for. N/A indicates no applicable guideline values.

For the majority of sites, the ISQG-High values are complied with for all parameters, indicating a relatively low risk of ecological effects caused by contaminated sediments. However, Wilsons Drain and Horners Drain both have sediment zinc concentrations exceeding the ISQG-High guideline, as well both sites exceeding the ISQG-Low for lead, and Wilsons Drain also exceeding the ISQC-Low for Arsenic (Table 5). This indicates that sediment contaminant concentrations at these sites are sufficiently high to be impact negatively on aquatic biota. Both Wilsons Drain and Horners Drain are scheduled for realignment and enhancement as part of CCC Outline Development Plans for the area. Care will therefore need to be taken during drain enhancement, in terms of managing the contaminated sediment present and minimising egress of contaminated sediment into any new channel alignments.

Comparison of sediment quality data from 1980 to 2018 shows that laboratory-measured sediments have typically been in the fine sand range for most sites (substrate index of 1-2), and sometimes in the medium sand range (substrate index of 2-3; Figure 12). No significant increasing or decreasing trend in substrate index was detected for any of the sites tested (P>0.05). The lack of trend may be expected, given that the field sampling method targets areas of fine sediment deposits, and is not intended to be representative of overall substrate composition at the reach scale.

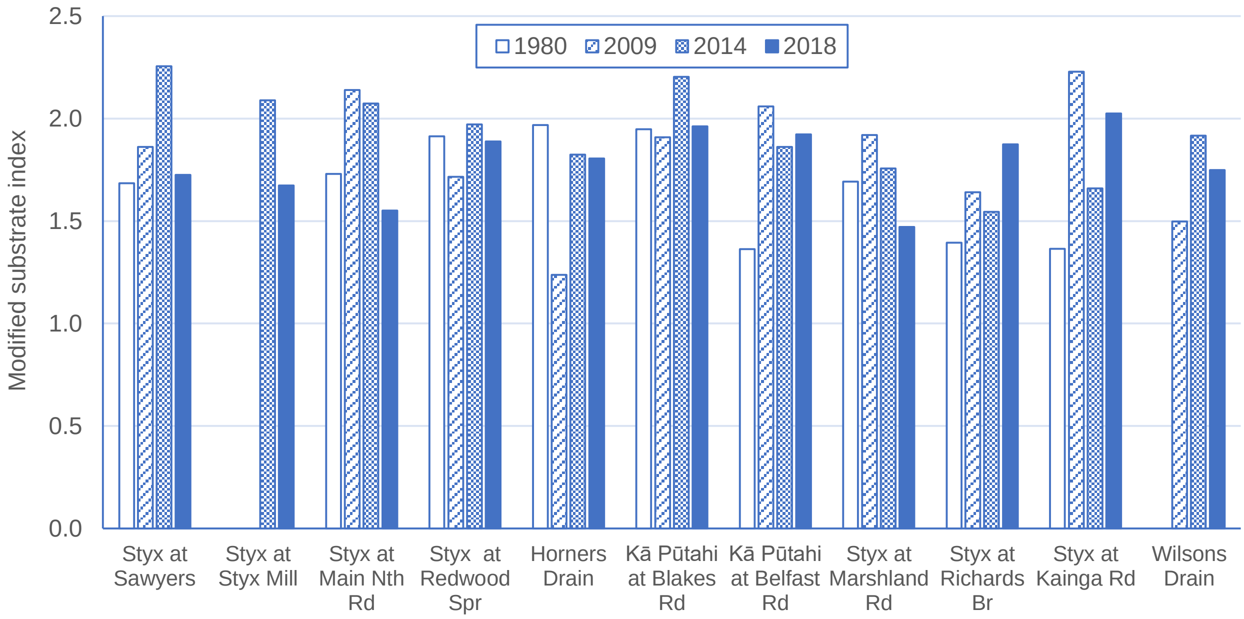


Figure 12: Modified substrate index for sediments analysed by the laboratory. An index score of 1 represents silt and clay-sized particles (<0.063 mm) and a score of 2 indicates fine sand (0.063-0.25 mm).

Copper concentrations in sediment have varied over time from 1980 to 2018, but they have almost always been well below the ISQG-Low trigger value (Figure 13). Trend analysis revealed weak positive (i.e., increasing) trends in copper concentration at the Styx at Redwood Springs site (P=0.042) and the Styx at Kainga/Harbour Road site (P=0.042). Copper concentrations remain well below the ISQG-Low trigger level at both of these sites, so adverse environmental effects due to elevated copper concentrations are unlikely. No significant increasing or decreasing trends were detected at any of the other sites (P>0.05).

Sediment lead concentrations have also varied considerably over time, with several sites exceeding the ISQG-Low trigger value, but no sites have exceeded the ISQG-High guideline on any occasion (Figure 14). Trend analysis revealed no significant increasing or decreasing trends in sediment lead concentrations (P>0.05) from 1980 to 2018.

Zinc concentrations in sediment have exceeded the ISQG-High guideline on at least two occasions in the past at the Horners Drain and the two Kā Pūtahi monitoring sites, and once at the Wilsons Drain site (Figure 15). The ISQG-Low trigger value has been exceeded on one occasion at four other sites (Figure 15). Trend analysis revealed a weak positive (i.e., increasing) trend in zinc concentration at the Styx at Kainga/Harbour Road site (P=0.042).

Zinc concentrations exceed the ISQG-Low trigger level first the first time in 2018 at this site; if this trend continues, adverse environmental effects may occur in the future. No significant increasing or decreasing trends were detected at any of the other sites (P>0.05).

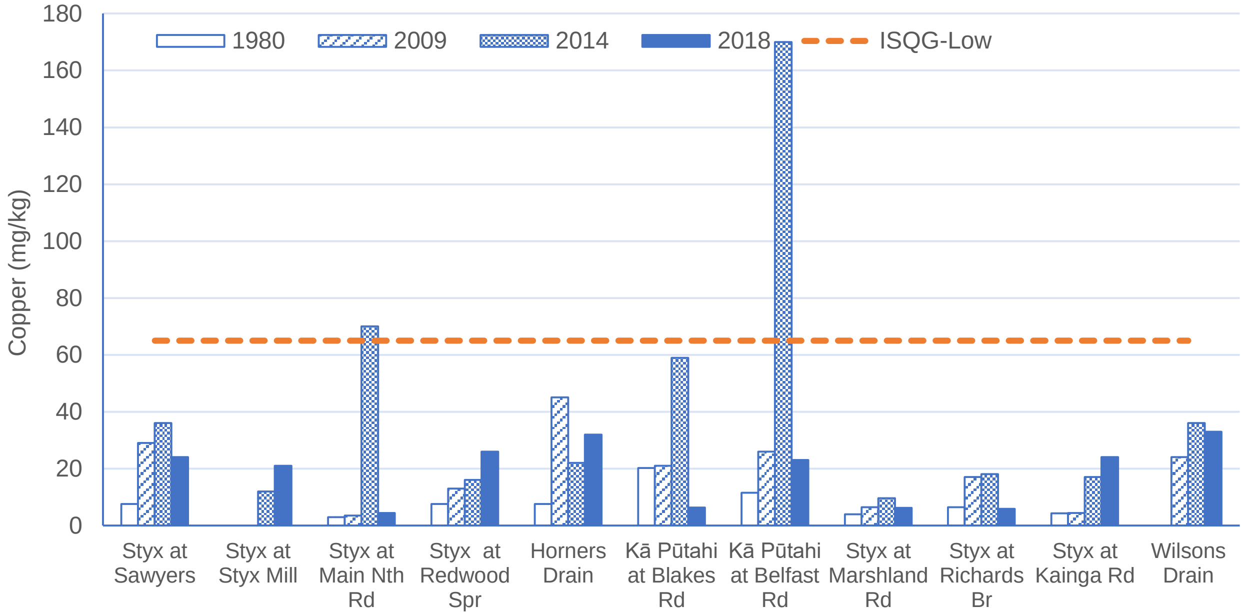


Figure 13: Sediment copper concentrations compared to the ISQG-Low trigger value from ANZECC (2000).

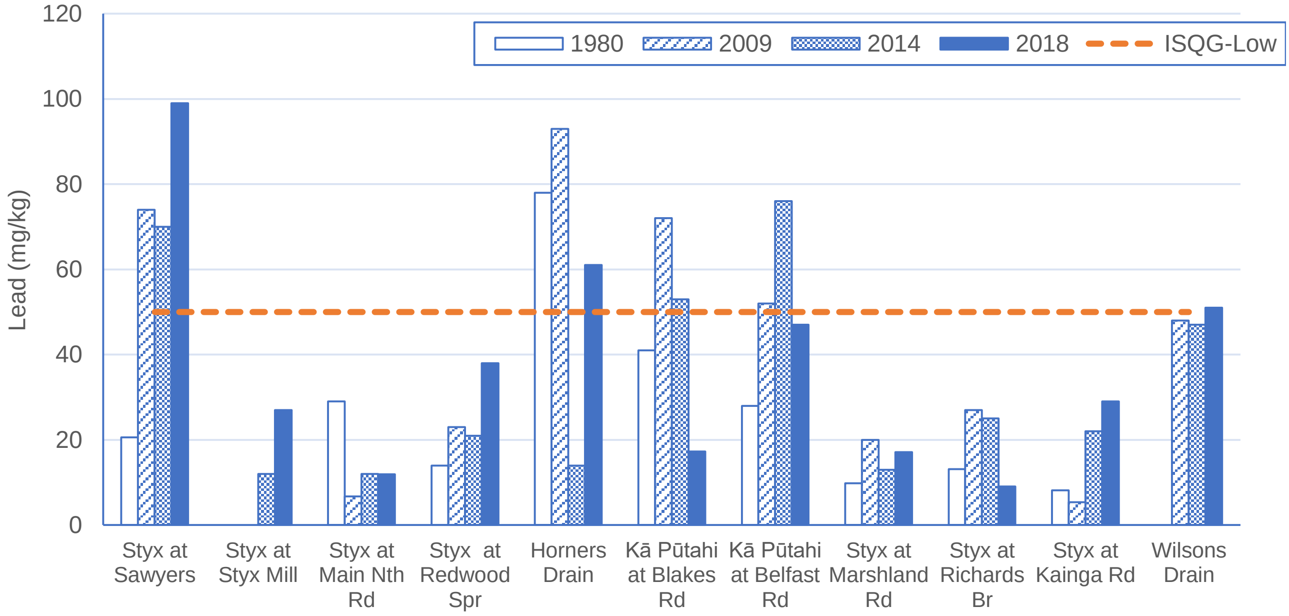


Figure 14: Sediment lead concentrations compared to the ISQG-Low trigger value from ANZECC (2000).

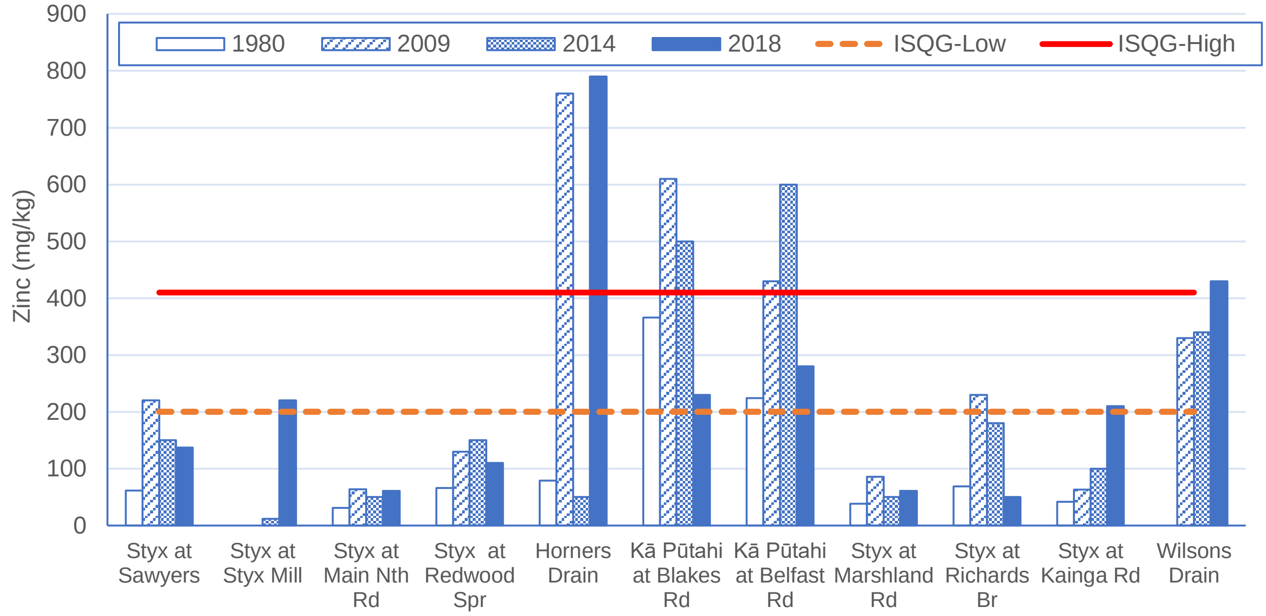


Figure 15: Sediment zinc concentrations compared to the ISQG-Low and ISQG-High guidelines from ANZECC (2000).

Total PAH sediment concentrations have remained very low at all sites from 1980 to 2018, with the exception of the Styx at Marshland Road site (Figure 16). Total PAHs exceeded the ISQG-Low trigger level in 2009 and 2018, but have been below the ISQG-High value of 45 on all occasions. No significant increasing or decreasing trend in PAHs was detected for any of the sites tested (P>0.05).

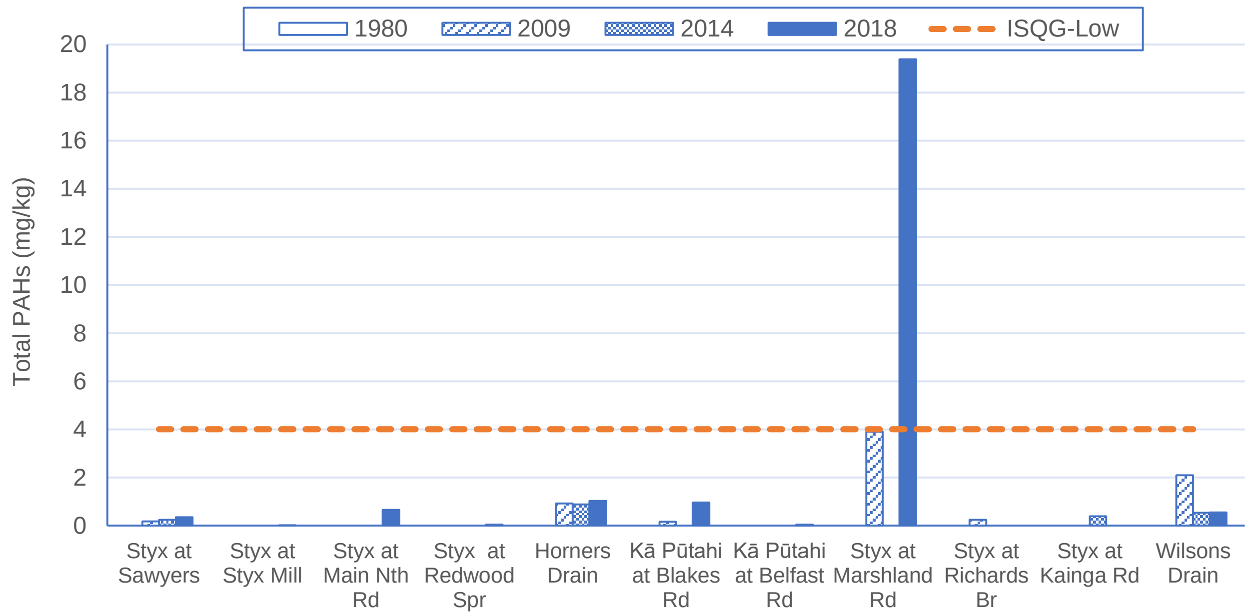


Figure 16: Sediment total PAH concentrations compared to the ISQG-Low trigger level from ANZECC (2000).

3.1.3. Macroinvertebrates

Total invertebrate abundance in 2018 varied from site to site with no particular pattern, as it has done in previous years (Figure 17). Total abundance was higher in 2018 than in previous years for 7 of the 12 sites monitored, and one-way ANOVA revealed a significant difference amongst years (P=0.022). Post-hoc tests indicated that total abundance was significantly greater in 2018 than in 2013 (P=0.020), but no significant difference between 2018 and 2008 (P=0.147). The ecological significance of these differences is discussed below, in terms of potential impacts on taxa richness.

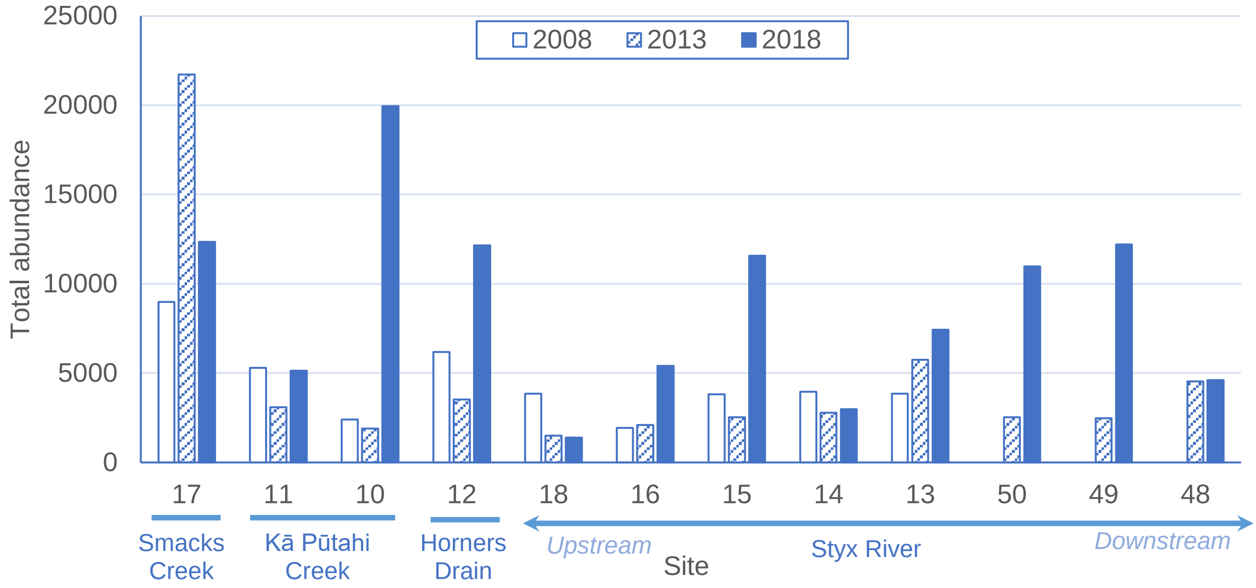


Figure 17: Total invertebrate abundance at each monitoring site.

As with total abundance, taxa richness in 2018 varied from site to site, with no strong pattern, as in previous years (Figure 18). There was no significant difference in taxa richness amongst sampling years (ANOVA P>0.05), with taxa richness higher in 2018 than in previous years for 6 sites and lower than at least one of the previous years at the other 6 sites. The lack of significant difference in taxa richness amongst sampling years suggests that the change in sampling method in 2018 has not affected the total number of taxa collected, despite an increase in the total number of individuals sampled compared with 2013. This indicates that it is valid to compare taxa-based indices such as total taxa richness and EPT taxa richness between 2018 and previous monitoring years.

In 2018 taxa richness ranged from a low of 21 taxa at Site 17 (Smacks Creek) and a high of 38 taxa at Site 11 (Kā Pūtahi Creek at Blakes Road). There was a large increase in taxa richness at Site 11 from 22 taxa in 2013 to 38 taxa in 2018, the greatest increase seen at any of the monitoring sites between any of the monitoring years (Figure 18). This increase is notable, as 2018 is the first year of monitoring Site 11 since it has been moved to the newly aligned and enhanced section of Kā Pūtahi Creek.

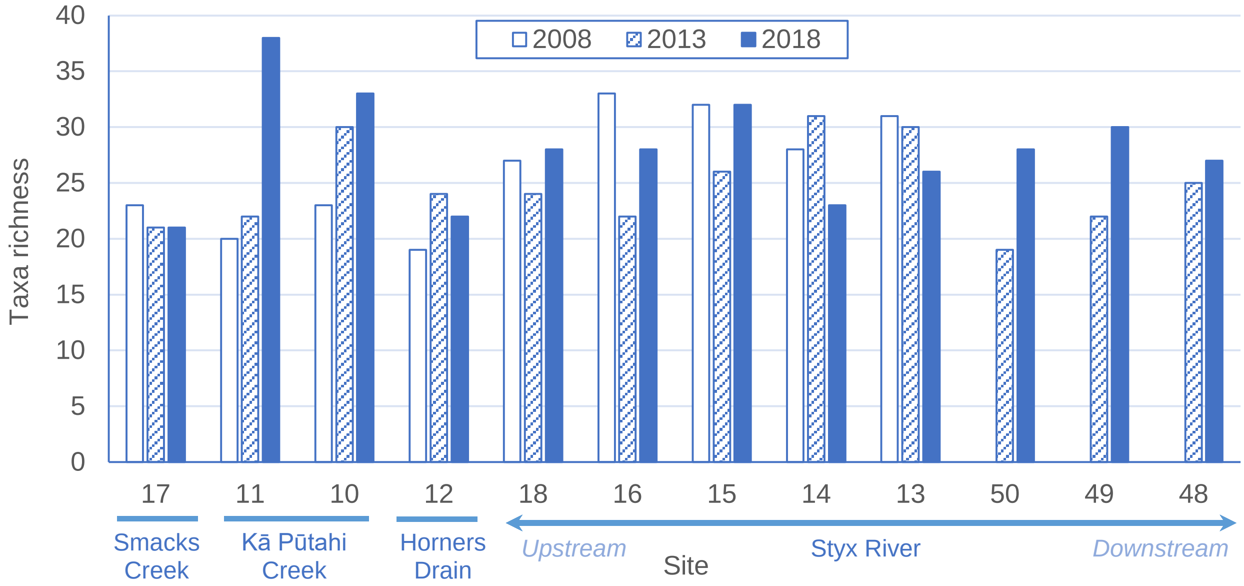


Figure 18: Invertebrate taxa richness at each monitoring site.

Invertebrate community composition was similar in 2018 to previous years, being dominated by the common mud snail *Potamopyrgus antipodarum* and the amphipod crustacean *Paracalliope fluviatilis* (Figure 19). These two pollution-tolerant taxa are very common in rural and urban Christchurch waterways, and they have dominated the invertebrate community every year. The next-most abundant taxon across all sites in 2018 was the cased caddisfly *Pycnocentria*. Although *Pycnocentria* have consistently been amongst the five most abundant taxa each year, 2018 is the first time it has ranked amongst the top three. This is noteworthy because *Pycnocentria* are relatively pollution-sensitive, with an MCI score of 7. In 2018 *Pycnocentria* were particularly abundant at Sites 13, 15 and 16, which are all stony, mid-catchment sites along the Styx River.

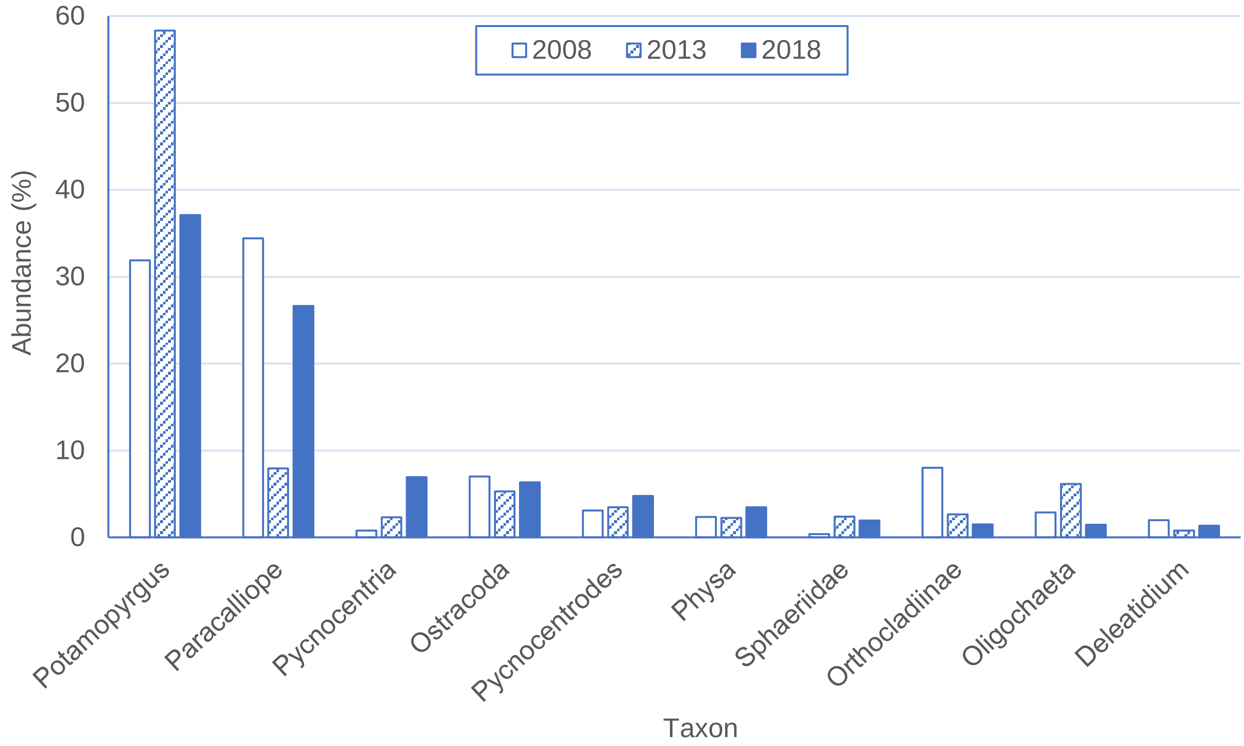


Figure 19: Abundance of the ten most common taxa across all sites in 2018 compared to previous years.

The fourth and fifth-most common taxa in 2018 were ostracod crustaceans and the cased caddis *Pycnocentrodes*, and these taxa were in the top five most abundant taxa in previous years too (Figure 19). The sixth to ninth-most abundant taxa in 2018 were Physa snails, sphaeriid bivalves, orthoclad midge larvae and oligochaete worms, and they are all relatively pollution tolerant and common. The common mayfly *Deleatidium* was the tenth-most common taxon in 2018, which is the second time this pollution-sensitve taxon has ranked in the top ten (it was ranked eighth in 2008). In 2018 *Deleatidium* were most abundant at Site 14 (Styx River in Styx Mill Reserve beside Glen Oakes), where they comprised 16% of total abundance, followed by Site 16 (Styx River in Styx Mill Reserve beside Dog Park), with 9% of total abundance. *Deleatidium* has also been most abundant at the Styx Mill Reserve sites in previous years.

A total of six pollution-sensitive taxa (MCI scores ≥7) were recorded from the Styx River catchment in 2018: the mayfly *Deleatidium* and the caddisflies *Oeconesus*, *Polyplectropus*, *Psilochorema bidens*, *P. tautoru*, and *Pycnocentria* (Table 6). All of these taxa were recorded in 2008 and 2013; we note that *Psilochorema* were not identified to species level in previous years, but the genus was recorded (James 2013).

Ten of the 12 monitoring sites recorded pollution-sensitive taxa in 2018, compared with eight or the 12 sites monitored in 2013 and nine of the ten sites monitored in 2008 (Table 6). The appearance of the caddisfly *Pycnocentria* (MCI=7) at Site 12 (Horners Drain) in 2018 is notable, because no pollution-sensitive taxa were recorded from that site in 2008 or 2013.

Table 6: Pollution-sensitive invertebrate taxa (MCI scores of ≥7) at monitoring sites from 2008 to 2018.

Waterway	Site	2008	2013	2018
Smacks Creek	17	*Psilochorema*	*Psilochorema*	*Deleatidium* *Polyplectropus* *Psilochorema bidens*
Kā Pūtahi Creek	11	*Deleatidium* *Psilochorema* *Pycnocentria*	*Psilochorema*	*Deleatidium* *P. bidens* *Pycnocentria*
	10	*Polyplectropus* *Pycnocentria*	*Polyplectropus* *Pycnocentria*	*Polyplectropus* *P. bidens*
Horners Drain	12	No taxa with MCI ≥7	No taxa with MCI ≥7	*Pycnocentria*
Styx River (upstream)	18	*Deleatidium* *Oeconesus* *Olinga feredayi* *Polyplectropus* *Psilochorema* *Pycnocentria*	*O. feredayi* *Polyplectropus* *Psilochorema* *Pycnocentria*	*Polyplectropus* *P. bidens* *Pycnocentria*
	16	*Deleatidium* *Oeconesus* *Polyplectropus* *Psilochorema* *Pycnocentria*	*Deleatidium* *Polyplectropus* *Psilochorema* *Pycnocentria*	*Deleatidium* *Oeconesus* *P. bidens* *Pycnocentria tautoru* *Pycnocentria*
	15	*Deleatidium* *Oeconesus* *Polyplectropus* *Psilochorema* *Pycnocentria*	*Deleatidium* *Oeconesus* *Polyplectropus* *Psilochorema* *Pycnocentria*	*Deleatidium* *Oeconesus* *Polyplectropus* *P. bidens* *Pycnocentria*
	14	*Deleatidium* *Polyplectropus* *Psilochorema* *Pycnocentria*	*Deleatidium* *Oeconesus* *Polyplectropus* *Psilochorema* *Pycnocentria*	*Deleatidium* *P. bidens* *Pycnocentria*
	13	*Deleatidium* *Psilochorema* *Pycnocentria*	*Deleatidium* *Oeconesus* *Psilochorema* *Pycnocentria*	*Deleatidium* *P. bidens* *Pycnocentria*
	50	*Deleatidium* *Psilochorema* *Pycnocentria*	No taxa with MCI ≥7	No taxa with MCI ≥7
Styx River (downstream)	49	No data	No taxa with MCI ≥7	No taxa with MCI ≥7
	48	No data	No taxa with MCI ≥7	*Polyplectropus*

The cased caddisfly *Olinga feredayi* was the only pollution-sensitive taxon not detected in 2018 that was found in 2008 and 2013. Examination of raw data sheets provided by CCC indicates that *O. feredayi* were found at Site 18 (Styx River at Claridges Road) in 2008 and

2013, but only 1 individual was collected in 2008 and 2 were collected in 2013. *O. feredayi* were more widespread in the Styx catchment in the 1970s and 1980s (James 2013), and their decline is of concern. That is because with an MCI score of 9, *O. feredayi* are the most pollution-sensitive taxon recorded in the catchment over the last 10 years. However, at such low densities, it is possible that they were simply not detected in 2018.

Oeconesus is the only other taxon with an MCI score of 9 recorded from the Styx catchment from 2008 to 2018. *Oeconesus* were recorded from three sites in 2008 and 2013 and two sites in 2018. *Oeconesus* have always been found at densities of fewer than 5 per sample (actually per combined three samples for 2008 and 2018). It is therefore possible that *Oeconesus* were simply not detected at one of the sites this year, or that their absence is indicative of a further gradual decline in the Styx catchment.

Overall EPT taxa richness in 2018 followed the same general pattern as previous years, with taxa richness highest at the Styx River sites in Styx Mill Reserve (Sites 15 and 16), and lowest at the non-wadeable sites downstream of Marshland (Sites 48, 49, and 50; Figure 20). There was no significant difference in EPT richness between monitoring years (ANOVA P>0.05). However, a large increase in EPT richness was observed at Site 11, the Kā Pūtahi Creek site that has recently been realigned (Figure 20). EPT richness at Site 11 more than doubled from 5 EPT taxa in 2013 to 11 EPT taxa in 2018.

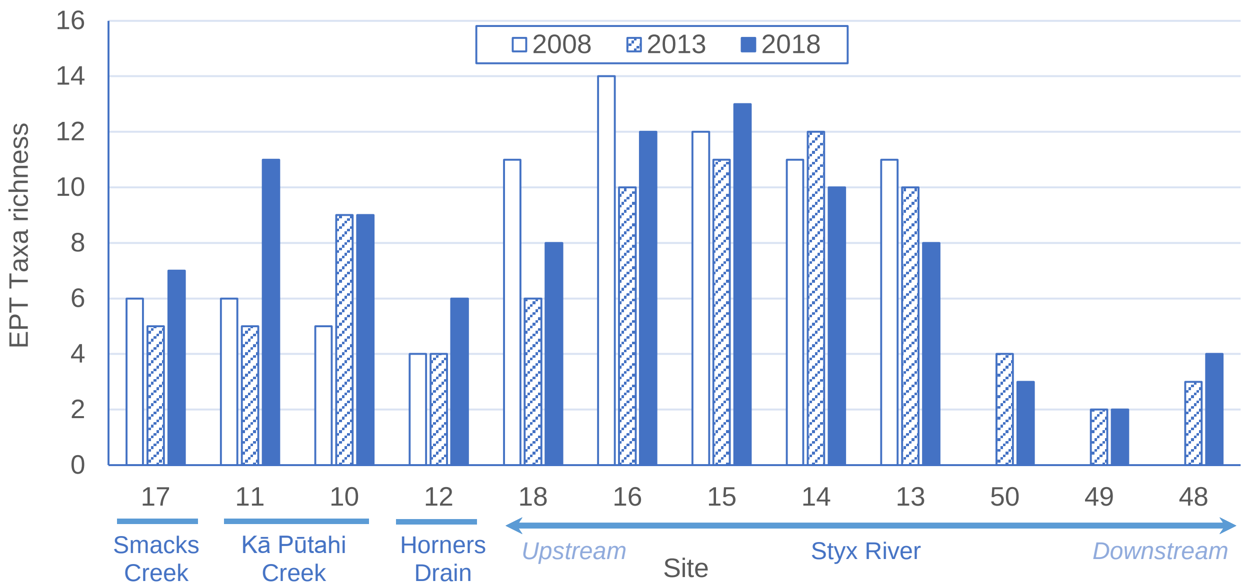


Figure 20: EPT taxa richness at each monitoring site.

Percent EPT abundance was greatest at Sites 13 to 16 on the Styx River in 2018, as it has been in previous years (Figure 21). EPT abundance in 2018 was 67% at Site 15 and 66% at Site 16, which is well above the maximum of 44% previously recorded at Site 16. Large numbers of *Pycnocentria* and *Pycnocentrodes* caddisflies were mainly responsible for the high percent EPT abundance at Sites 15 and 16 in 2018. There was no significant difference in EPT abundance between years (ANOVA P>0.05).

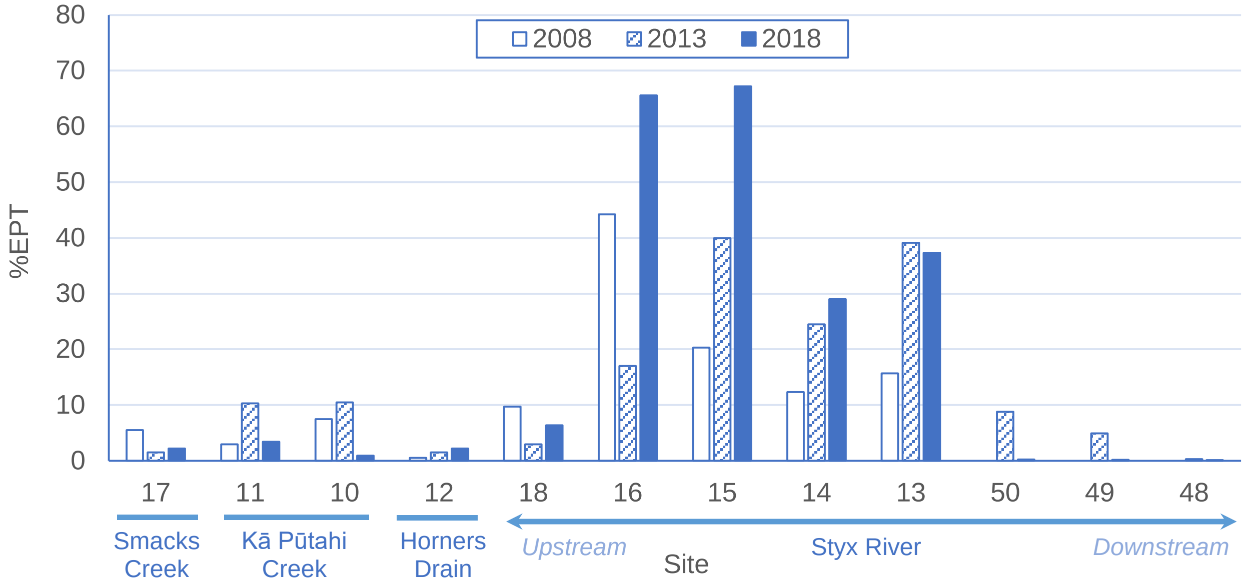


Figure 21: Percent EPT abundance at each monitoring site.

MCI scores in 2018 ranged from a low of 64 at Site 50 (Styx River at Marshland Road) to a high of 95 at Site 18 (Styx River at Claridges Road; Figure 22). MCI scores in 2018 followed a similar pattern to previous years, with highest scores from the upper Styx River sites (Sites 15, 16, and 18) and the lowest scores at the three non-wadeable sites in the lower Styx River Sites 48, 49, and 50). Overall, MCI scores in 2018 were indicative of fair quality (MCI scores of 80 to 100) at most sites, with the exception of Horners Drain (Site 12) and the three non-wadeable sites, which all had MCI scores indicative of poor quality (MCI scores below 80). However, it should be noted that MCI scores are typically lower at non-wadeable sites due to the dominance of finer bed sediments, but this does not necessarily mean that the overall water quality is degraded.

QMCI scores in 2018 met or exceeded the SMP objective of 4.5 at three of the 12 monitoring sites: Site 16 (Styx River at Styx Reserve Dog Park), Site 15 (Styx River at Styx Mill Reserve), and Site 13 (Styx River at Main North Road; Figure 22). QMCI scores in 2018 were indicative of good quality (scores between 5 and 6) at two sites, fair quality (scores between 4 and 5) at seven sites, and indicative of poor quality (scores below 4) at the remaining three sites. This was the first year that QMCI scores have exceeded 5 and therefore been within the good quality category, with QMCI scores of 5.6 and 5.5 at Sites 16 and 15, respectively (Figure 22). However, a QMCI score of 2.3 was recorded in 2018 from the most downstream Styx River site, and this is the lowest QMCI score recorded over the ten year monitoring period. Despite variations of QMCI scores within sites, there was no significant difference in QMCI scores between years (ANOVA P>0.05).

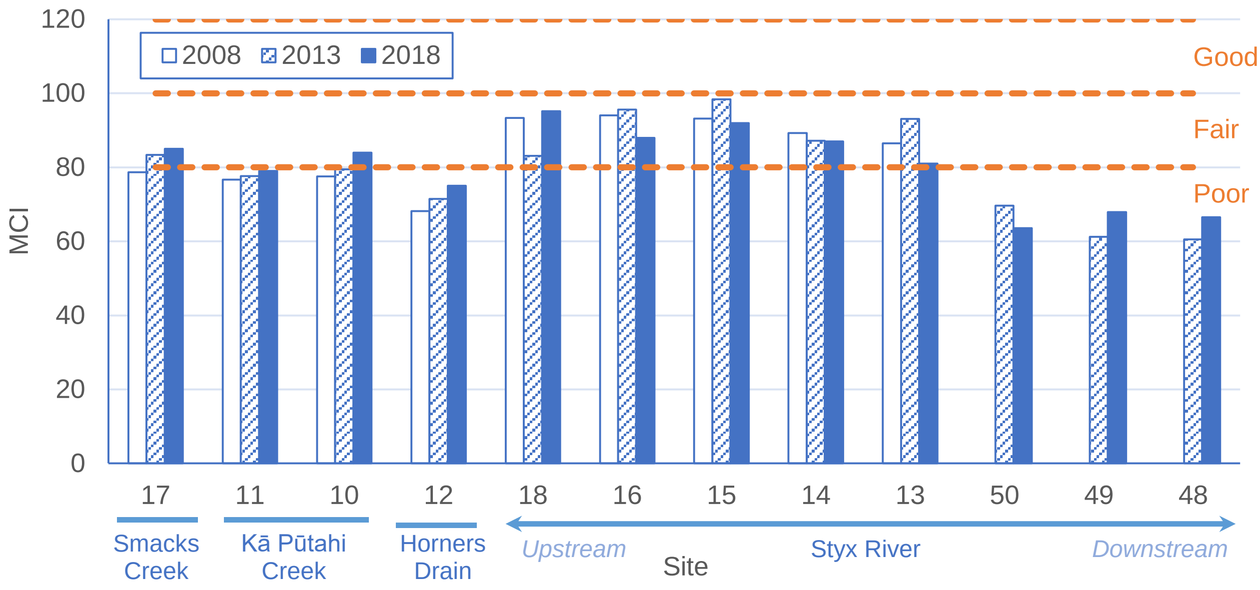


Figure 22: MCI scores at each monitoring site.

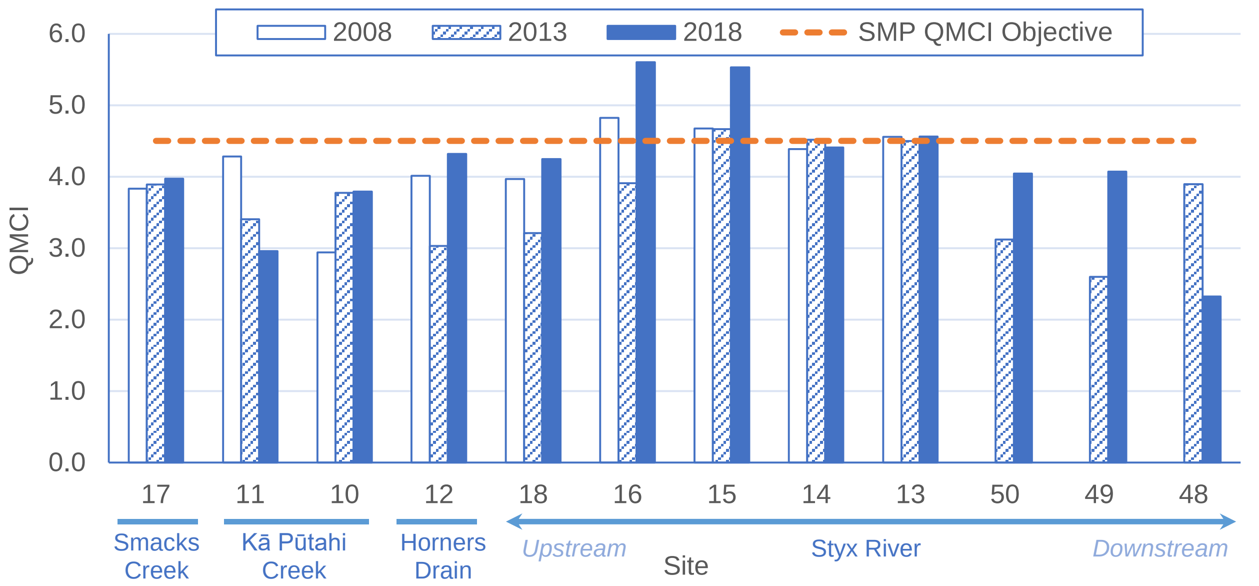


Figure 23: QMCI scores at each monitoring site.

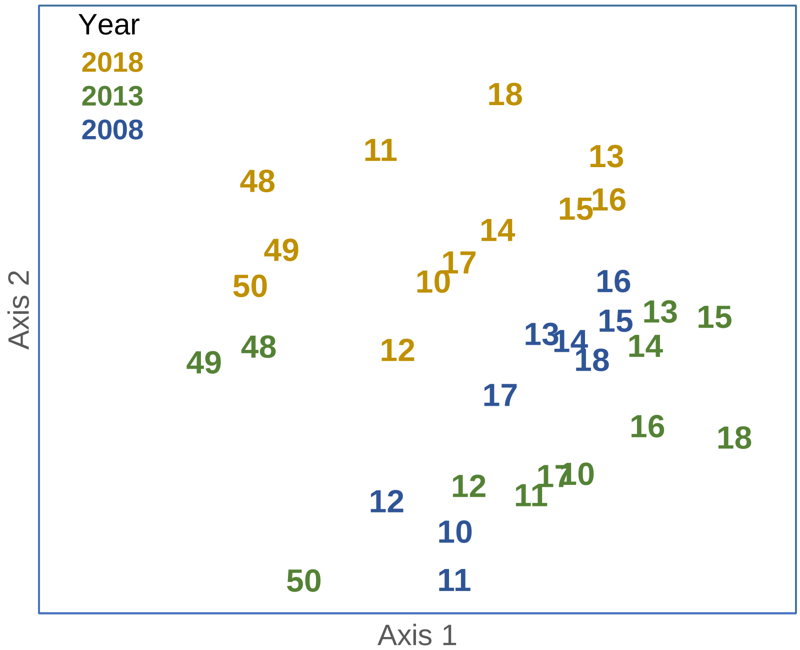
Freshwater crayfish (*Paranephrops zealandicus*), also known as kōura or kēkēwai were collected during electric fishing in 2018 at Sites 14 and 15, within Styx Mill Reserve. Kōura are valued both culturally and from a conservation perspective, because of their At Risk – Declining threat status (Grainger et al. 2014). It was therefore also encouraging to collect a very small juvenile (occipital carapace length = 9 mm) in a kicknet sample from Site 14 (it was returned to the stream live; Figure 24). Kōura were caught at one additional site in 2013, Site 18 (Styx River at Claridges Road).

In 2013, a single freshwater mussel (*Echyridella menziesii*), also known as kākahi, was collected from Site 49, in the lower Styx River (James 2013), but none were recorded during the 2018 survey. Kākahi are mahinga kai and are also of conservation interest, with an At Risk – Declining threat status (Grainger et al. 2014). A dive survey of the lower Styx River in 2017 revealed that the Styx River kākahi population is large and widespread throughout the lower river. Results of the dive survey and their significance are detailed in Section 4.1 below.

Figure 24: A juvenile kōura or kēkēwai (freshwater crayfish) from the Styx River in Styx Mill Reserve (Site 14).

The NMDS ordination yielded a two-dimensional solution with a stress value of 0.18 for all sites and 0.20 for an ordination on just the wadeable sites (Figure 25). Stress values of 0.18 to 0.20 indicate a fair relationship with the underlying similarity matrix (Clarke 1993). The non-wadeable sites (Sites 48, 49, and 50) are grouped towards the left of Axis 1, indicating their community composition is distinct from that of the wadeable sites (Figure 25). Water depth, channel width, fine sediment depth and cover, and macrophyte cover were all significantly ($P<0.01$) and negatively correlated with Axis 1 scores.

The other clear pattern in the ordination plots is that samples from 2018 sit higher on Axis 2 than the two previous years (Figure 25), suggesting differences in community composition between 2018 and previous years. However, this likely reflects subtle differences in taxonomic resolution amongst the sampling years, rather than habitat differences, as there were no significant habitat correlations with Axis 2 ($P>0.01$). This is corroborated by the significant positive correlation ($P<0.01$) of *Triplectides cephalotes* and *Psilochorema bidens* with Axis 2 scores, when *T. cephalotes* was rarely reported prior to 2018 and *P. bidens* was not recorded at all prior to 2018.



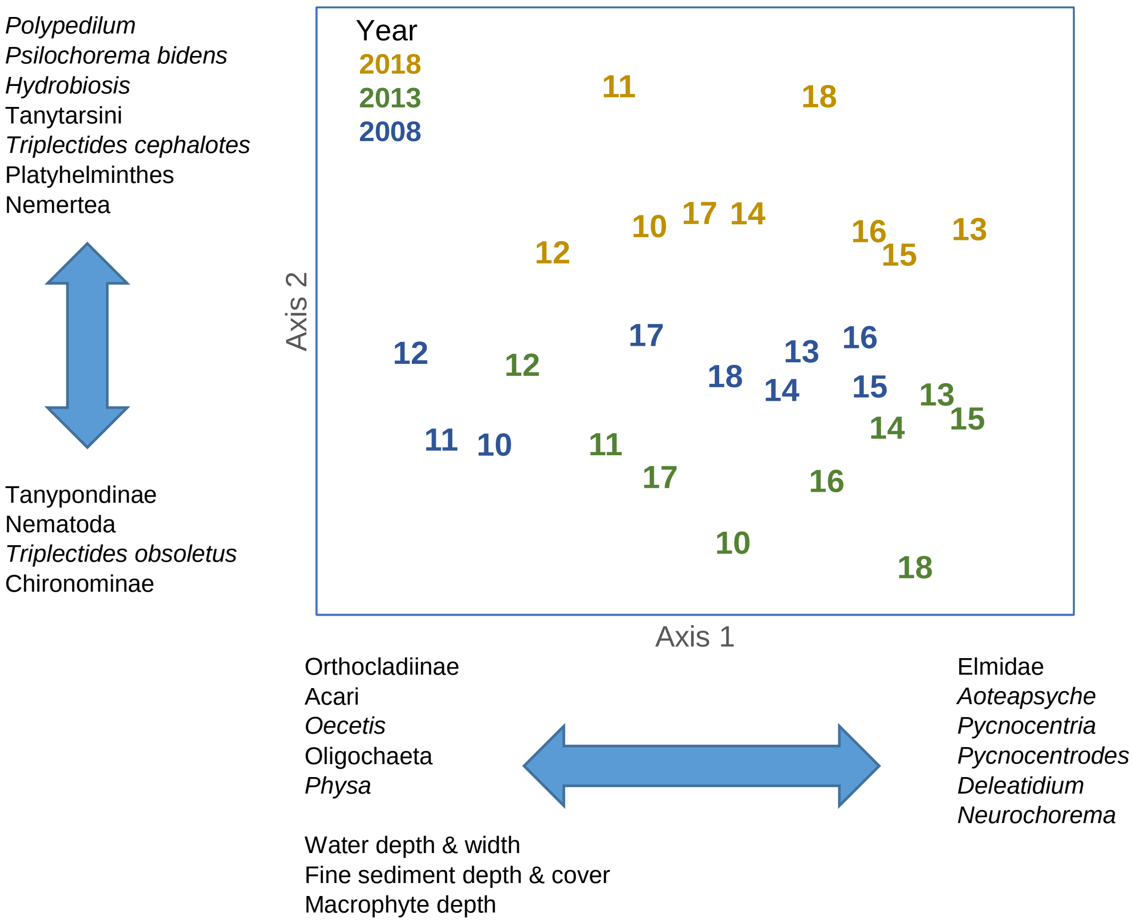


Figure 25: NMDS plot of invertebrate communities for all sites (top) and wadeable sites only (bottom). Coloured numbers site codes and colours refer to sampling years. Habitat parameters and species most strongly correlated with wadeable site axis scores are shown. Plot stress is 0.18 (all sites) and 0.20 (wadeable sites only).

When the non-wadeable sites are removed from the ordination, there is a more clear separation between Styx River mainstem and tributary sites. Thus, Sites 10 and 11 from Kā Pūtahi Creek, Site 12 from Horners Drain, and Site 17 from Smacks Creek are all grouped towards the left of Axis 1 (Figure 25). These sites generally had greater cover with fine sediment, greater macrophyte depth and greater widths and depths, and tended to be more associated with pollution-tolerant invertebrate taxa such as oligochaete worms, *Physa* snails, and orthoclad midge larvae.

There were no significant correlations ($P>0.05$) between water quality, sediment quality and axis scores for 2018 invertebrate samples.

3.1.4. Fish

A total of nine fish species were caught in 2018, comprising eight native species and one introduced species, brown trout (Table 7). Shortfin eel were the most widespread species and they were found at all 12 sites. Longfin eel were found at seven sites, but were less abundant at each site. Common bully were found at five sites and they were particularly abundant at the two most downstream Styx River sites, as were juvenile bullies (Table 7). Inanga were only recorded from the three non-wadeable Styx River sites in the lower river. The lack of inanga at sites further upstream likely reflects the different sampling methods at wadeable and non-wadeable sites, with inanga more readily caught by the combination of fyke nets and minnow traps used at the non-wadeable sites than via electric fishing (used at the wadeable sites).

The total number of fish caught from all wadeable sites was 347 in 2013, compared with 103 in 2018, while at non-wadeable sites the total was 81 in 2013 and 368 in 2018. These differences between years remained, even when fishing effort was standardised to area fished for the electric fishing sites (Figure 26) and fish caught per net for the non-wadeable sites (Figure 27).

Differences in sampling methods between years most likely accounts for the differences in total abundance observed between years for the non-wadeable sites. Thus, the switch from unbaited coarse-meshed fyke nets in 2013 to baited fine mesh fyke nets with double traps in 2018 was associated with a large increase in the number of small fish caught (Figure 27).

There was reduced fishing effort in 2018 compared to 2013 at the electric-fished wadeable sites, which partially accounts for the reduced fish numbers at those sites. Thus, greater fishing effort was expended at the wadeable sites in 2013, with two passes of a 20 m reach with an electric fishing machine, compared to a single pass of a 30 m reach in 2018. However, this alone does not explain the differences between years, as standardising the catch per unit area should have more closely aligned the total catch between years. Other possible reasons for reduced fish numbers in 2018 are impacts of high flows earlier in the season on fish numbers, and effects of higher baseflows on catch efficiency in 2018.

Despite differences in the total numbers of individual species caught, the total range and number of fish species recorded in 2018 was similar to 2013. The only difference in the species caught between years was that a single rainbow trout was caught in 2013 at Site 16 and none were caught in 2018, while a single black flounder was caught in 2018 at Site 48 and none were previously caught in 2013 (Table 7).

Table 7: Total number of fish and kōura caught per site in 2018. Size range (mm) is in brackets.

Waterway	Site	Shortfin eel	Longfin eel	Elver	Common Bully	Upland Bully	Giant Bully	Juvenile Bully	Inanga	Lamprey	Black flounder	Brown Trout	Kōura
Smacks Creek	17	4 (224 - 379)	6 (328 - 579)			1 (73)							
Kā Pūtahi Creek	11	4 (179 - 622)		1 (92)		3 (41 - 58)		1 (38)					
	10	19 (149 - 682)		1 (100)	2 (41 - 53)			2 (34)					
Horners Drain	12	1 (289)	11 (176 - 509)		2 (63 - 90)	1 (56)							
Styx River (upstream)	18	1 (325)										1 (600)	
	16	4 (206 - 369)	1 (396)										
	15	2 (281 - 371)	1 (470)							1 (102)			1 (29)
	14	15 (157 - 403)	1 (563)	2 (97 - 117)						1 (102)			4 (11 - 31)
	13	1 (182)	3 (274 - 553)	5 (114 - 154)									
	50	3 (510 - 640)	2 (426 - 569)		12 (32 - 124)	2 (51 - 64)			3 (70 - 92)				
	49	2 (294 - 362)			21 (46 - 90)			144 (22 - 38)	1 (48)				
Styx River (downstream)	48	20 (344 - 702)			56 (39 - 104)		6 (116 - 135)	78 (22 - 39)	17 (48 - 117)		1 (65)		

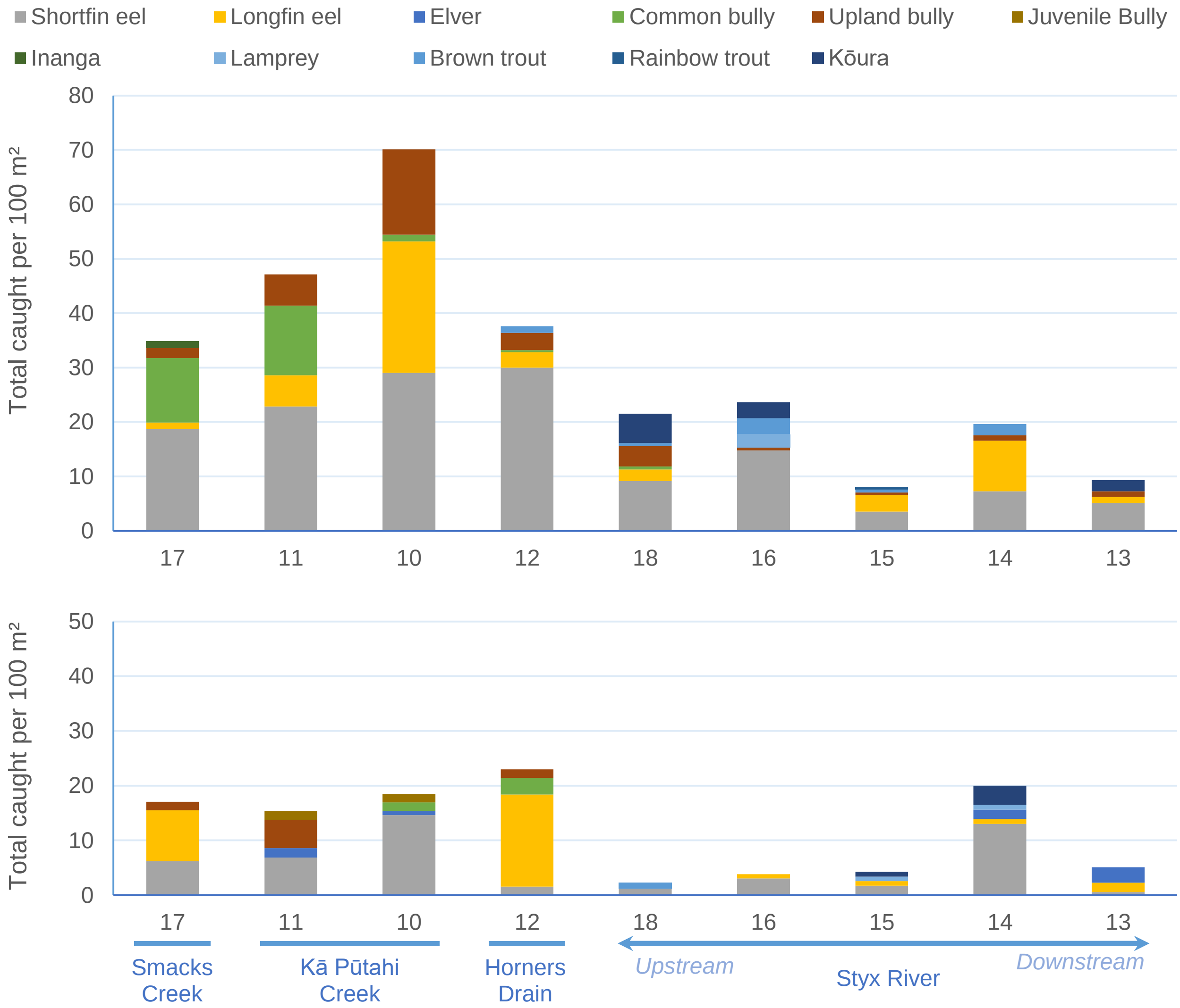


Figure 26: Comparison of electric fishing results at wadeable sites from 2013 (top) and 2018 (bottom).

The same four native species with a conservation status caught in 2013 were caught again in 2018. These species are longfin eel, inanga, and giant bully which all have an At Risk – Declining threat status, and lamprey, which have a Nationally Vulnerable status (Dunn et al. 2018).

A noteworthy observation was collecting a single whitebait (juvenile inanga) in 2018 from Site 49 in the lower Styx River. This catch was notable because the whitebait was caught in early autumn, when the peak of the upstream whitebait migration occurs in spring. This observation highlights the fact that while many native fish have predictable periods of peak migration, migratory juveniles and adults may be found at any time of the year.

Another notable find in 2018 was the presence of juvenile lamprey at Styx River Sites 14 and 15 in Styx Mill Reserve (Figure 28), when they were only recorded at Site 15 in 2013. Lamprey are the most threatened native fish recorded in the catchment, so their discovery at another monitoring site is encouraging. In addition, a very significant population of lamprey was discovered in Canal Reserve Drain, a tributary of the Styx River, in 2015 (Taylor and

Marshall 2015). The Canal Reserve Drain lamprey population is discussed further in Section 4.2 below.

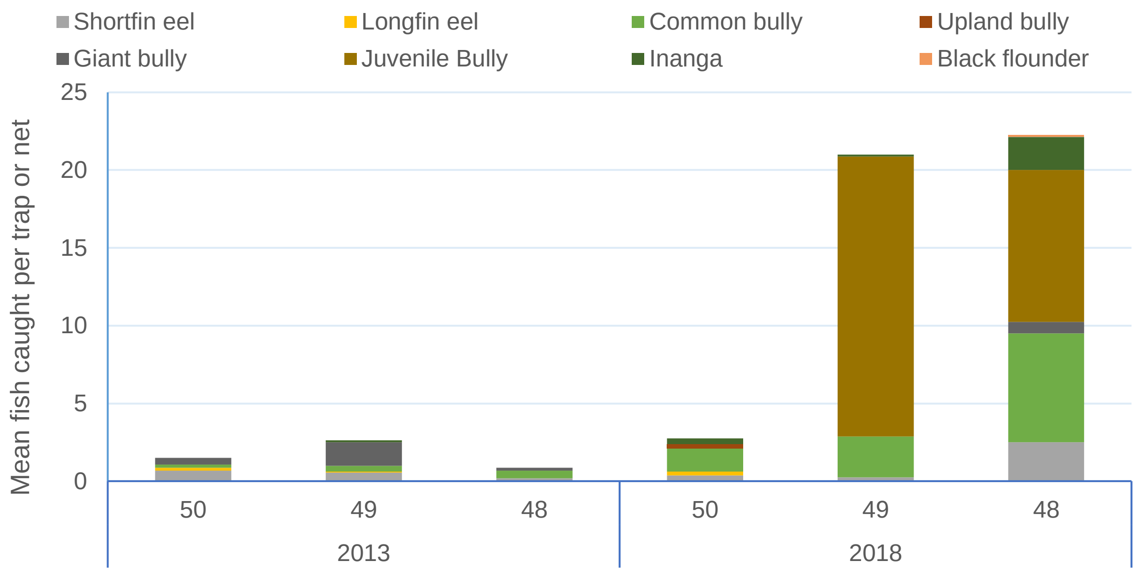


Figure 27: Comparison of fish caught at the three non-wadeables sites in 2013 (left) and 2018 (right).

Overall, the majority of Styx River fish community is similar to that present in other Christchurch waterways. However, the presence of lamprey elevates the overall conservation value of the catchment, particularly given the presence of a spawning population in the Canal Drain Reserve tributary.

Figure 28: A juvenile lamprey caught from the Styx River within Styx Mill Conservation Reserve in 2018.

3.2. Annual Monitoring Results from Styx Mill Conservation Reserve

Annual invertebrate monitoring of the Styx River has been undertaken on six occasions from 2013 to 2018, at Site 14 in Styx Mill Conservation Reserve. The primary purpose of the annual monitoring site is to pick up any trends in aquatic habitat and invertebrate community health that might otherwise be missed by the 5-yearly monitoring programme at this site of relatively high ecological value.

Bed cover with fine sediment (<2 mm diameter) at the annual monitoring site was 32% in 2018, so complied with the SMP water quality objective of 40% (Figure 29). Total macrophyte cover was 19% in 2018 and therefore complied with the SMP water quality objective of 50% (Figure 29). Fine sediment cover and total macrophyte cover have varied over time, but there was no significant increasing or decreasing trend detected for either parameter (P>0.05).

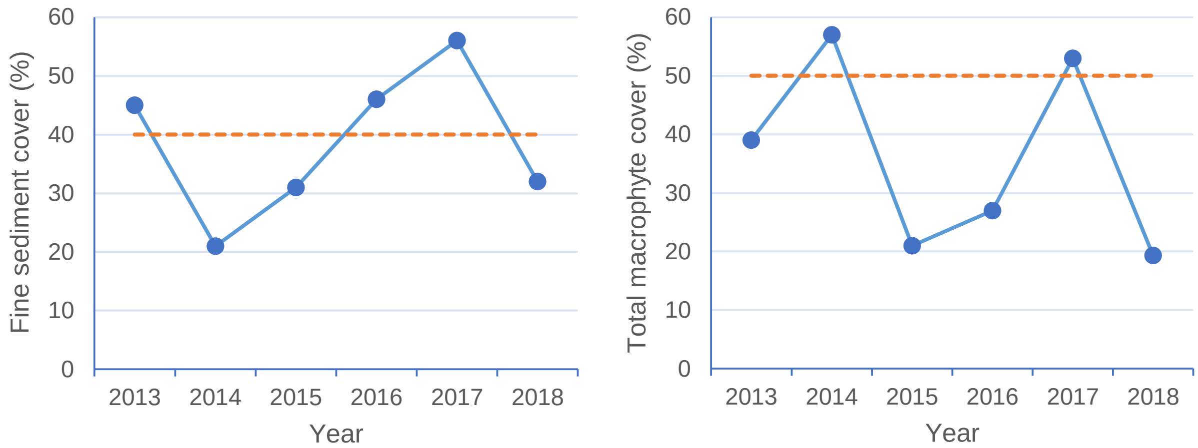


Figure 29: Fine sediment cover (left) and total macrophyte cover (right) at the Styx Mill Reserve annual monitoring site. The dashed horizontal line indicates the SMP water quality objective, which is a maximum of 40% for fine sediment and 50% for total macrophyte cover.

A total of 23 taxa and 10 EPT taxa were collected from the annual monitoring site in 2018, which is intermediate to values recorded in previous years (Figure 30). EPT abundance was 29% in 2018, which was also within the range of values recorded in previous years (Figure 30). In 2018, the annual monitoring site recorded a QMCI score of 4.4, which is just below (i.e., does not comply with) the SMP water quality objective of 4.5, but is within the range of values recorded in previous years (Figure 30). None of the invertebrate community indices had a significant increasing or decreasing trend over the six year monitoring period (P>0.05).

In summary, there were no significant trends (P>0.05) for any of the macroinvertebrate indices or habitat parameters over the six years of monitoring from 2013 to 2018.

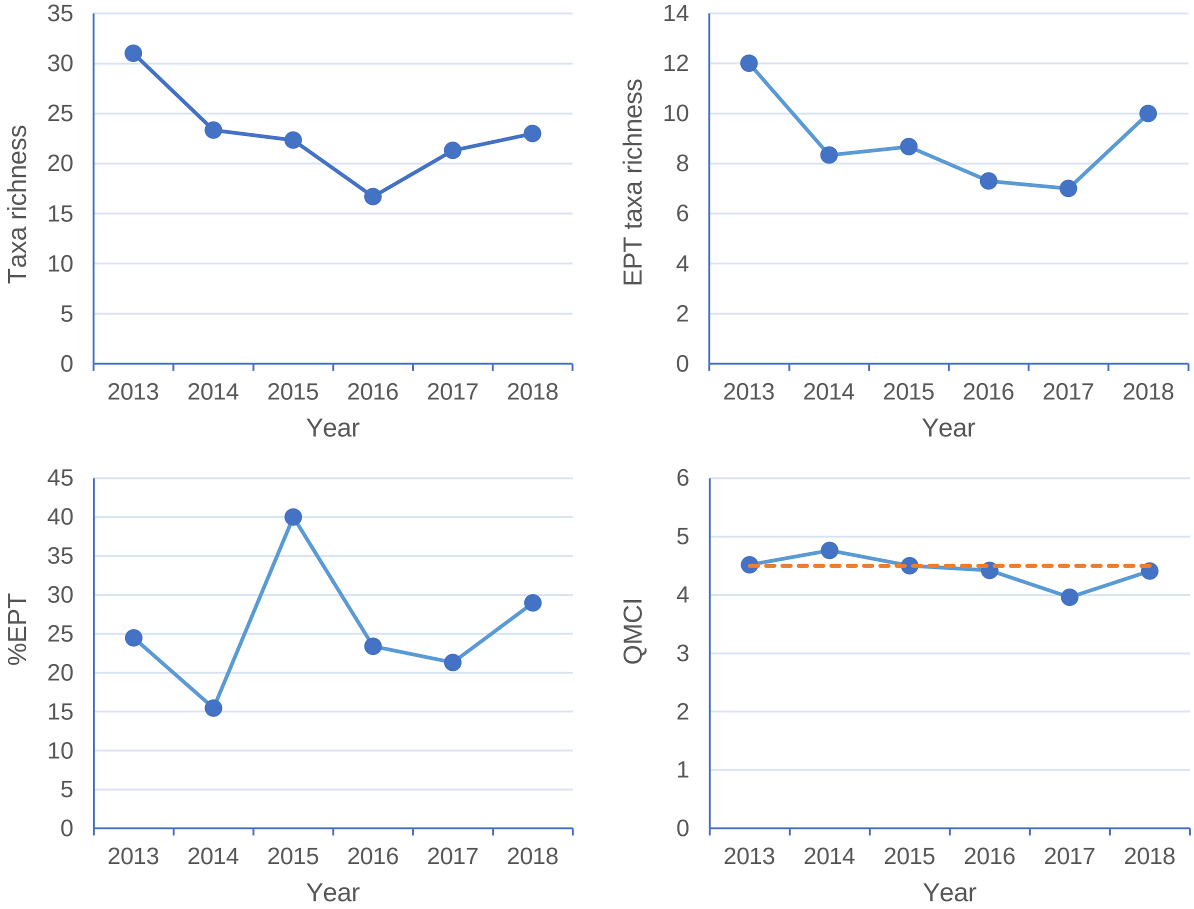


Figure 30: Invertebrate taxa richness, EPT taxa richness, percent EPT abundance, and QMCI scores at the Styx Mill Reserve annual monitoring site. The dashed horizontal line on the QMCI plot indicates the SMP water quality objective, which is a minimum of 4.5.

4. NEW ECOLOGICAL DISCOVERIES

4.1. Kākahi in the Lower Styx River

Previous monitoring reports have noted reports from weed cutting contractors that kākahi are present in the lower reaches of the Styx River (James 2013). This is of ecological interest, because kākahi have an “At Risk” threat status (Grainger et al. 2014) and they are uncommon in Christchurch waterways. However, monitoring of the Styx River population to date has not occurred, because the kākahi observations have been from the deeper sections of river downstream of Marshlands Road, which is not easily sampled.

Following discussion with CityCare contractors, Duncan Gray and Greg Burrell went for a recreational snorkel in the lower Styx River in August 2017, looking for kākahi in their personal time. They found kākahi at all five locations they visited, from Marshlands Road to Earlham Street, and kākahi were common at all sites except Marshlands Road (Figure 31).

Mussels were found both mid-channel and at the edges, and in both soft and hard sediments, although they appeared to be less common amongst the finest, least compacted sediments. This recreational snorkel therefore confirmed observations of the weed cutter contractors that freshwater mussels are widespread throughout the lower Styx River.

Figure 31: Kākahi on the bed of the Styx River.

The observations of Burrell and Gray suggest that the Styx River supports the most extensive population of kākahi known from the city's spring-fed rivers. The only other comparable kākahi population in the city is in Cashmere Stream and it covers approximately 2.3 km of river length (Burdon & McMurtrie 2009)). By comparison, the Styx River population covers at least 9 km and is likely more extensive, as mussels were found at every site visited. Given the extent and size of the kākahi population, and their At Risk conservation status, the Styx River kākahi are a locally, and likely regionally, significant population.

The two major pressures faced by Styx River kākahi are urbanisation and river dredging. Kākahi are very uncommon in urban areas, due to a combination of development impacts on water quality and hydrology. The relatively undeveloped nature of the Styx River catchment compared to other Christchurch urban waterways is the likely reason for kākahi still being abundant in the lower river. The Styx SMP should help protect water quality and hydrology – and the kākahi population – through the requirement for stormwater detention and treatment for all new residential, commercial, and industrial developments.

Dredging was historically undertaken regularly in the Styx River and other Christchurch rivers, in response to public concerns about flooding. Dredging of Christchurch rivers has reduced in more recent years, for environmental reasons and because dredging often has a short-lived value to flood mitigation. However, following the Canterbury earthquakes of 2010 and 2011, there has been pressure to dredge several Christchurch waterways, due to liquefaction and lateral spread reducing channel capacity

Impacts of dredging on freshwater mussels have been well studied overseas, but nothing has been published on dredging impacts on New Zealand mussel species. Therefore, when the CCC Land Drainage team indicated they were to dredge a section of the Styx River where kākahi are abundant, they also funded a kākahi monitoring and kākahi salvage operation. Because of the depth of the river, monitoring was undertaken by SCUBA divers from NIWA (Figure 32).

Figure 32: NIWA divers monitoring the Styx River kākahi population (left) and a catch-bag full of kākahi (right).

Prior to dredging, the divers found mussel densities of 40 to 55 mussels per m² at four sites upstream of Spencerville Road bridge (CCC data). The dive team then removed all visible kākahi from the dredging reach and relocated them upstream of the dredging operation. A total of 17,164 kākahi were collected by the divers from an approximately 500 m length of river that was to be dredged and relocated upstream (CCC data). To the best of our knowledge, this is the largest kākahi recovery operation undertaken in New Zealand.

Follow-up monitoring of the kākahi population has been undertaken and will be ongoing, to assess how quickly it recovers from the translocation. A report detailing all of the methods and results will be prepared by CCC later in 2018 or early 2019, depending on when the final monitoring round occurs. These kākahi monitoring data will be extremely useful, both for understanding this significant population in a Christchurch waterway, and for predicting impacts of any future dredging operations.

4.2. Canal Reserve Drain Lamprey

A significant lamprey population was discovered in Canal Reserve Drain in 2015, during a routine ecological assessment prior to a road intersection upgrade and culvert realignment (Taylor & Marshall 2015). Canal Reserve Drain is a narrow and primarily a timber-lined waterway that flows alongside Marshland Road, and flows both north into the Styx River (at Marshland Road Bridge) and south into tributaries of the Avon River. The section of drain sampled by Taylor & Marshall (2015) was immediately upstream of the Styx River confluence and the culvert in question runs under Hawkins Road. A total of 358 ammocoete (an early life stage) lamprey were caught, and they were the dominant fish species present in the drain (Taylor & Marshall 2015). Figure 33 shows a photograph of lamprey ammocoetes from Canal Reserve Drain.

Figure 33: Juvenile lamprey from Canal Reserve Drain, just upstream of the Styx River.

The Canal Reserve Drain lamprey population is locally – if not nationally – significant, for a number of reasons. Firstly, the number of ammocoetes caught is remarkably high, and finding them in such numbers is very uncommon. Second, this is the only known site where lamprey were the most abundant fish species recorded (pers comm, Cindy Baker, NIWA fish ecologist). Third, the large number of juvenile lamprey, including those of various sizes, indicates that spawning occurs somewhere within the Canal Reserve Drain catchment, and there are only a handful of known lamprey spawning sites in the country. Fourth and lastly, all other known lamprey spawning locations within New Zealand have previously been within forested streams with cobble-boulder habitats (pers comm. Cindy Baker, NIWA fish ecologist), whereas Canal Reserve Drain is a highly modified urban waterway, with wood lining, sandy bed sediments, and no boulders.

The Canal Reserve Drain lamprey population spurned an intensive study by NIWA fishery scientists to try and find the spawning location and to avoid impacts of the road realignment on the population (Baker et al. 2017). In 2016, researchers caught 12 adult migratory lamprey entering the drain and tagged them with Passive Integrated Transponder (PIT) tags, to enable them to track the whereabouts of individual fish (Figure 34). Tagged adults were recorded up to 800 m up the drain from the Styx River confluence, and at least one was believe to be residing behind the wooden lining of the drain. However, no spawning sites were located, despite a concerted tracking effort and use of an endoscope to probe behind breaks in the wood lining (Baker et al. 2017).

The lamprey discovery resulted in a significant redesign of the intersection upgrade, to avoid impacts on the adjacent lamprey habitat. Ongoing threats to the lamprey population include cars, which regularly crash into the drain, and the poor state of repair of the drain lining, which is rotting and collapsing in a number of locations.

Figure 34: Canal Reserve Drain habitat downstream of the Hawkins Road culvert (left). NIWA staff using a modified PIT tag reader to search for tagged adult lamprey upstream of the culvert (right).

Because the drain lining is in need of repair and it also harbours a significant lamprey population, CCC is commencing research into options for relining Canal Reserve Drain. In addition to commencing engineering investigations, CCC is collaborating with ecologists from Environment Canterbury, NIWA, and the Department of Conservation. The ecology investigations involve further study of the lamprey population, including use of PIT tags, radio tags, electric fishing, and trialling the use of boulders added to the channel for additional spawning habitat. The overall aim of the research is to better understand where in the drain lamprey are spawning, what their spawning habitat is, and how best to protect and enhance spawning habitat. This information will then be used to inform the drain relining strategy.

5. DISCUSSION

5.1. Current State and Trends in Aquatic Ecology

Monitoring data from 2018 indicate that riparian and instream habitat quality remains largely unchanged compared to previous years at most of the monitoring sites. Compared with the more heavily-urbanised Avon and Heathcote River catchments, the Styx River has relatively broad and vegetated riparian zones at many sites. These riparian strips provide habitat, shade, and organic inputs to waterways and help buffer them from overland flow by filtering out contaminants. Despite riparian zones being dominated by grass and exotic plants at some locations, they still provide an important ecological function. Overall, the current state of riparian and instream health of the Styx River is high relative to the Avon and Heathcote Rivers, and lower than the Otukaikino River. A key reason for the relatively good state of riparian condition at the Styx River monitoring sites is that most of them are within council-owned reserves. The prevalence of extensive waterway reserves along the Styx River is a testament to the Styx Vision, which is a long term plan to enhance the waterway and provide a "source to sea" experience (CCC 2000).

The greatest change in riparian habitat was observed at the Kā Pūtahi Creek monitoring site at Blakes Road. This section of waterway was recently realigned and enhanced to make way for the Northern Arterial Motorway. Although the monitoring site focussed on a 20 m reach of riffle and run habitat, the broader realignment includes a mix of run, riffle and pool habitat, and extensive native plantings within a very wide riparian zone. Habitat enhancements have included the use of wood in the bank to provide fish habitat (Figure 35), along with boulders positioned along and within the channel.

Figure 35: New native plantings and logs in the bank to provide fish habitat in the recently-realigned and enhanced section of Kā Pūtahi Creek (Site 11).

Sediment concentrations of common stormwater contaminants exceeded ANZECC (2000) guidelines at a number of sites in 2018, but there were no increasing trends at most of the sites. Zinc is the contaminant of most concern in sediments, as it is the only parameter to regularly exceed the ISQG-High guideline. Sediment zinc concentrations have been consistently elevated over time at all of the tributary sites, reflecting their generally more urbanised sub-catchments compared to the other mainstem Styx River sites. Weak, but statistically significant, increasing trends were detected for zinc at the Styx River at Harbour/Kainga Road site and copper at the Styx at Redwood Springs sites. Although zinc and copper concentrations remained well below the ISQG-High guidelines at these sites, an ongoing trend would be of ecological concern. This highlights the need for ongoing implementation of stormwater treatment devices in new developments, as provided for in the Styx SMP, as well as retro-fitting stormwater treatments in older urban areas whenever the opportunity arises.

Invertebrate community composition in 2018 was similar to previous years, being dominated by pollution-tolerant snails and crustaceans that are common to rural and urban Christchurch waterways. However, the abundance and diversity of pollution-sensitive EPT taxa also remains greater in the Styx River catchment than in the Avon, Heathcote, and Halswell Rivers. Thus, a total of 18 EPT taxa, comprising 17 caddisfly taxa and one mayfly taxon, were recorded from the 12 Styx monitoring sites in 2018. This compares with a total of 10 EPT taxa recorded from 29 Avon catchment sites (Boffa Miskell 2014), 9 EPT taxa from 15 Heathcote catchment sites (Boffa Miskell 2015), and 9 EPT taxa from 5 Halswell catchment sites (Boffa Miskell 2016). The Styx River catchment therefore still remains only second behind the Otukaikino River catchment in terms of diversity of pollution-sensitive invertebrate taxa for Christchurch waterways. A total of 15 EPT taxa were recorded from 9 Otukaikino catchment sites in 2017, but unlike the Styx catchment, EPT taxa dominated total abundance overall (Boffa Miskell 2017).

Annual invertebrate monitoring of the Styx River has been undertaken on six occasions from 2013 to 2018 at Site 14 in Styx Mill Conservation Reserve. In 2018, the annual monitoring site recorded a QMCI score of 4.4, which is just below the SMP water quality objective of 4.5, but was within the narrow range of values recorded in previous years. In addition, no increasing or decreasing trends were detected in habitat parameters or invertebrate community health over the six years of record. This indicates that although invertebrate community health may be regarded as only "fair" (as indicated by QMCI scores), it is stable and there is no declining trend that is of concern.

The range of fish species caught in 2018 was also similar to previous years and was dominated by native species, particularly shortfin eels. A change in fish sampling methods in 2018 to standard CCC protocols was associated with an overall reduction in the total number of fish caught at wadeable electric-fishing sites, and an increase at the wadeable sites, but no overall change in the total number of fish species caught compared to 2013. The Styx River fish community is similar to that present in other Christchurch waterways, with a dominance of native species and few introduced species (brown trout was the only introduced species caught in 2018). However, the presence of lamprey at two of the monitoring sites in Styx Mill Conservation Reserve is notable, because of their Nationally Vulnerable conservation status (Dunn et al. 2018).

Ecological monitoring of Kā Pūtahi Creek approximately 10 months after realignment and habitat enhancement revealed no positive impact on the invertebrate and fish community to date (James 2017). The lack of a positive ecological response to the habitat enhancements likely reflected both the short time since the new channel was constructed, but it may also reflect the pervasive effects of upstream rural and urban landuse (James 2017). Irrespective of the current state of aquatic ecology, the extensive native plantings have significantly increased the local native plant diversity and the value of the plantings will increase as the plants grow over time. In addition, the large increase in EPT taxa in 2018 at the realignment site compared to 2013 (pre-realignment), suggests that there is a "seed" population of sensitive taxa that could increase as the young native plantings mature and associated habitat quality improves.

5.2. Comparison to Styx SMP Surface Water Quality Objectives

The Styx SMP includes surface water quality objectives for QMCI scores, and maximum bed cover with fine sediment, macrophytes, and filamentous algae. The SMP objectives for total macrophyte cover and filamentous algae cover have been consistently met at most sites over the last ten years (Table 8). This likely reflects the combination of moderating shading present, the dominance of coarse substrates, and regular macrophyte removal by CCC contractors (although it is noted that ecological monitoring is undertaken prior to macrophyte removal). The fine sediment SMP objective of <40% cover has also been complied with at most sites (Table 8), although Site 18 (Styx River at Claridges Road) has always exceeded this value (Figure 9).

Table 8: Compliance with Styx SMP water quality objectives at each of the nine wadeable sites over time..

Parameter	SMP objective	Complying sites each year (total out of 9 sites)		
		2008	2013	2018
Minimum QMCI	4.5	6	6	6
Maximum fine sediment (<2 mm) cover	40%	8	7	6
Maximum total macrophyte cover	50%	8	9	8
Maximum filamentous algae cover	30%	9	8	9

Of particular interest is the SMP objective for QMCI, because the QMCI is an indicator of invertebrate community health, and invertebrates are influenced by both water quality and habitat. The SMP objective for QMCI is a minimum of 4.5, and this objective has been met by six of the nine monitoring sites for each year of monitoring (Table 8). Although QMCI scores have varied within sites over the years, there has been no overall increasing or decreasing trend in QMCI scores evident across all of the sites monitored every five years, or at Site 14 in Styx Mill Conservation Reserve that is monitored annually. This indicates that the overall ecological health of the Styx River is stable and that there is no indication of a declining trend that could be attributable to stormwater discharges or other landuse impacts.

5.3. Recent Discoveries

The recent scientific confirmation of a widespread and abundant population of kākahi in the lower Styx River is an exciting development. The Styx River kākahi population is locally, if not regionally significant, due to its extent and density, and because kākahi numbers are declining nationally (Grainger et al. 2014). The fact that previous routine invertebrate and fish monitoring only detected a single kākahi highlights the fact that they are not readily detected using standard sampling methods. Snorkelling provided a rapid way of establishing the presence and relative abundance of kākahi at different sites, while SCUBA divers were required to estimate densities. The Styx River kākahi population highlights the fact that the non-wadeable reaches of lowland rivers can provide habitat for ecologically-significant species that may be otherwise overlooked by conventional sampling techniques. Ongoing monitoring of the Styx kākahi population will provide useful insight into the impacts and recovery of kākahi from dredging operations.

The recent discovery of a significant lamprey population in the Styx catchment is also of considerable ecological interest. Although the status of lamprey within the catchment were once uncertain (Golder Associates 2009), the presence of large numbers of ammocoetes in Canal Reserve Drain indicates that the Styx River supports a viable breeding population. The Canal Reserve Drain lamprey population is unique by virtue of the large number of ammocoetes present, their far greater abundance compared to other fish species present, and the highly modified nature of the habitat present. Lamprey are fairly cryptic species, which means they are usually only detected in low numbers, making them a problematic species to manage from a conservation perspective. However, researchers have had considerable success using pheromone traps to detect the presence of lamprey within multiple tributaries of a catchment (pers. comm., Cindy Baker, NIWA fish ecologist). Pheromone trapping would be an affordable and efficient way of determining the presence of lamprey within the Styx River catchment tributaries.

6. RECOMMENDATIONS

Based on the results and discussion presented above, we recommend the following:

- Undertake pheromone trapping throughout the Styx River catchment, to better understand where lamprey are found and how best to manage the wider population. This recommendation should be given a high priority, given the amount of residential development and waterway realignment occurring in the catchment, and lamprey's threatened species status.
- Continue monitoring kākahi populations in the Styx River and use the monitoring results to inform weed removal and dredging activities in the Styx River and other waterways.
- Investigate increasing trends of sediment copper and zinc at the Styx River Harbour Road site and increasing sediment copper at the Styx River at Redwood Springs site.
- Care will need to be taken managing contaminated sediments during any enhancement or realignment of Wilsons and Horners Drains, given the elevated zinc and lead concentrations present in both waterways, and elevated arsenic levels in Wilsons Drain.
- Continue using the now-standard CCC ecology sampling methods. That is because results in this report indicate that the standard CCC methods yield similar results to more labour-intensive methods used in previous years.

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APPENDIX 1: SITE PHOTOGRAPHS FROM 2018

Figure 1: Site 10 (Kā Pūtahi Creek at Ouruhia Reserve) - downstream end of reach looking upstream.

Figure 2: Site 10 (Kā Pūtahi Creek at Ouruhia Reserve) - upstream end of reach looking downstream.

Figure 3: Site 11 (Kā Pūtahi Creek at Blakes Road) - downstream end of reach looking upstream.

Figure 4: Site 11 (Kā Pūtahi Creek at Blakes Road) - upstream end of reach looking downstream.

Figure 5: Site 12 (Horners Drain) - downstream end of reach looking upstream.

Figure 6: Site 12 (Horners Drain) - upstream end of reach looking downstream.

Figure 7: Site 13 (Styx River at Main North Road) - downstream end of reach looking upstream.

Figure 8: Site 13 (Styx River at Main North Road) - upstream end of reach looking downstream.

Figure 9: Site 14 (Styx River in Styx Mill Conservation Reserve by Glen Oakes) - downstream end of reach looking upstream.

Figure 10: Site 14 (Styx River in Styx Mill Conservation Reserve by Glen Oakes) - upstream end of reach looking downstream.

Figure 11: Site 15 (Styx River in Styx Mill Conservation Reserve) - downstream end of reach looking upstream.

Figure 12: Site 15 (Styx River in Styx Mill Conservation Reserve) - upstream end of reach looking downstream.

Figure 13: Site 16 (Styx River in Styx Mill Conservation Reserve at Dog Park) - downstream end of reach looking upstream.

Figure 14: Site 16 (Styx River in Styx Mill Conservation Reserve at Dog Park) - upstream end of reach looking downstream.

Figure 15: Site 17 (Smacks Creek) - downstream end of reach looking upstream.

Figure 16: Site 17 (Smacks Creek) - upstream end of reach looking downstream.

Figure 17: Site 18 (Styx River at Claridges Road) - downstream end of reach looking upstream.

Figure 18: Site 18 (Styx River at Claridges Road) - upstream end of reach looking downstream.

Figure 19: Site 48 (Styx River at Harbour/Kainga Road) - downstream end of reach looking upstream.

Figure 20: Site 48 (Styx River at Harbour/Kainga Road) - upstream end of reach looking downstream.

Figure 21: Site 49 (Styx River at Richards Bridge) - downstream end of reach looking upstream.

Figure 22: Site 49 (Styx River at Richards Bridge) - upstream end of reach looking downstream.

Figure 23: Site 50 (Styx River at Marshland Road) - downstream end of reach looking upstream.

Figure 24: Site 50 (Styx River at Marshland Road) - upstream end of reach looking downstream.

APPENDIX 2: SEDIMENT QUALITY LABORATORY RESULTS

Hill Laboratories
TRIED, TESTED AND TRUSTED

R J Hill Laboratories Limited
28 Duke Street Frankton 3204
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Certificate of Analysis

SPv1

Client:	Instream Consulting Limited	**Lab No:**	1946214
Contact:	G Burrell	**Date Received:**	19-Mar-2018
	C/- Instream Consulting Limited	**Date Reported:**	15-May-2018
	PO Box 28173	**Quote No:**	89934
	Christchurch 8242	**Order No:**	
		Client Reference:	Styx River Sediment Samples
		Submitted By:	G Burrell

Sample Type: Sediment

Sample Name:		Smacks @ Husseys 18-Mar-2018 9:20 am	Styx @ Styx Mill 18-Mar-2018 9:40 am	Styx @ Redwood Springs 18-Mar-2018 11:00 am	Styx @ Richards 18-Mar-2018 12:15 pm	Kaputone @ Belfast 18-Mar-2018 11:35 am
Lab Number:		1946214.1	1946214.2	1946214.3	1946214.4	1946214.5
Individual Tests						
Dry Matter	g/100g as rcvd	10.8	14.6	28	53	13.9
Total Recoverable Copper	mg/kg dry wt	44	21	26	5.9	23
Total Recoverable Lead	mg/kg dry wt	28	27	38	9.1	47
Total Recoverable Zinc	mg/kg dry wt	161	220	110	50	280
Total Organic Carbon*	g/100g dry wt	15.8	14.3	3.2	0.54	10.5
7 Grain Sizes Profile						
Dry Matter of Sieved Sample	g/100g as rcvd	12.3	11.8	40	65	17.1
Fraction >/= 2 mm*	g/100g dry wt	24.5	22.8	1.8	0.3	15.0
Fraction < 2 mm, >/= 1 mm*	g/100g dry wt	10.7	6.0	3.0	0.4	6.8
Fraction < 1 mm, >/= 500 µm*	g/100g dry wt	10.5	8.4	2.5	0.3	7.6
Fraction < 500 µm, >/= 250 µm*	g/100g dry wt	12.0	8.7	9.5	5.1	15.1
Fraction < 250 µm, >/= 125 µm*	g/100g dry wt	17.7	10.8	23.6	50.5	19.3
Fraction < 125 µm, >/= 63 µm*	g/100g dry wt	15.8	18.8	31.7	24.9	14.7
Fraction < 63 µm*	g/100g dry wt	8.8	24.5	27.8	18.5	21.5
Polycyclic Aromatic Hydrocarbons Trace in Soil						
1-Methylnaphthalene	mg/kg dry wt	0.018	< 0.010	< 0.005	< 0.003	0.015
2-Methylnaphthalene	mg/kg dry wt	0.013	< 0.010	< 0.005	< 0.003	0.014
Acenaphthene	mg/kg dry wt	< 0.013	< 0.010	< 0.005	< 0.003	< 0.010
Acenaphthylene	mg/kg dry wt	< 0.013	< 0.010	< 0.005	< 0.003	< 0.010
Anthracene	mg/kg dry wt	< 0.013	< 0.010	< 0.005	< 0.003	< 0.010
Benzo[a]anthracene	mg/kg dry wt	0.031	< 0.010	0.008	< 0.003	0.022
Benzo[a]pyrene (BAP)	mg/kg dry wt	0.051	< 0.010	0.014	< 0.003	0.039
Benzo[b]fluoranthene + Benzo[j] fluoranthene	mg/kg dry wt	0.043	< 0.010	0.014	< 0.003	0.036
Benzo[e]pyrene	mg/kg dry wt	0.022	< 0.010	0.007	< 0.003	0.017
Benzo[g,h,i]perylene	mg/kg dry wt	0.030	< 0.010	0.011	< 0.003	0.027
Benzo[k]fluoranthene	mg/kg dry wt	0.013	< 0.010	< 0.005	< 0.003	0.012
Chrysene	mg/kg dry wt	0.032	< 0.010	0.008	< 0.003	0.023
Dibenzo[a,h]anthracene	mg/kg dry wt	< 0.013	< 0.010	< 0.005	< 0.003	0.014
Fluoranthene	mg/kg dry wt	0.094	0.016	0.022	< 0.003	0.060
Fluorene	mg/kg dry wt	< 0.013	< 0.010	< 0.005	< 0.003	< 0.010
Indeno(1,2,3-c,d)pyrene	mg/kg dry wt	0.031	< 0.010	0.010	< 0.003	0.029
Naphthalene	mg/kg dry wt	< 0.07	< 0.05	< 0.03	< 0.013	< 0.05
Perylene	mg/kg dry wt	< 0.013	< 0.010	0.010	< 0.003	0.011
Phenanthrene	mg/kg dry wt	0.064	0.013	0.011	< 0.003	0.033
Benzo[a]pyrene Potency Equivalency Factor (PEF) NES	mg/kg dry wt	0.06	< 0.03	0.018	< 0.007	0.06

Sample Type: Sediment						
Sample Name:		Smacks @ Husseys 18-Mar-2018 9:20 am	Styx @ Styx Mill 18-Mar-2018 9:40 am	Styx @ Redwood Springs 18-Mar-2018 11:00 am	Styx @ Richards 18-Mar-2018 12:15 pm	Kaputone @ Belfast 18-Mar-2018 11:35 am
Lab Number:		1946214.1	1946214.2	1946214.3	1946214.4	1946214.5
Polycyclic Aromatic Hydrocarbons Trace in Soil						
Benzo[a]pyrene Toxic Equivalence (TEF)	mg/kg dry wt	0.06	< 0.03	0.018	< 0.007	0.06
Pyrene	mg/kg dry wt	0.088	0.014	0.020	< 0.003	0.056

Sample Name:		Horners 18-Mar-2018 1:05 pm	Styx @ Sawyers 18-Mar-2018 9:00 am	Styx @ Main Nth Rd 18-Mar-2018 10:40 am	Styx @ Marshlands 18-Mar-2018 12:55 pm	Styx @ Kainga 18-Mar-2018 11:55 am
Lab Number:		1946214.6	1946214.7	1946214.8	1946214.9	1946214.10
Individual Tests						
Dry Matter	g/100g as rcvd	31	36	34	47	32
Total Recoverable Arsenic	mg/kg dry wt	-	3.8	2.3	5.5	12.9
Total Recoverable Copper	mg/kg dry wt	32	24	4.4	6.2	24
Total Recoverable Lead	mg/kg dry wt	61	99	11.9	17.1	29
Total Recoverable Zinc	mg/kg dry wt	790	137	61	61	210
Total Organic Carbon*	g/100g dry wt	3.4	3.2	1.48	0.87	4.0
7 Grain Sizes Profile						
Dry Matter of Sieved Sample	g/100g as rcvd	40	40	69	70	34
Fraction >/= 2 mm*	g/100g dry wt	0.5	5.4	31.1	26.2	0.5
Fraction < 2 mm, >/= 1 mm*	g/100g dry wt	4.8	2.8	1.7	2.5	5.6
Fraction < 1 mm, >/= 500 µm*	g/100g dry wt	4.8	2.0	1.7	1.2	5.5
Fraction < 500 µm, >/= 250 µm*	g/100g dry wt	7.5	4.2	16.3	7.0	10.0
Fraction < 250 µm, >/= 125 µm*	g/100g dry wt	10.6	16.9	30.9	35.3	32.9
Fraction < 125 µm, >/= 63 µm*	g/100g dry wt	26.8	38.2	12.5	13.0	16.8
Fraction < 63 µm*	g/100g dry wt	44.9	30.5	5.9	14.8	28.7
Polycyclic Aromatic Hydrocarbons Trace in Soil						
1-Methylnaphthalene	mg/kg dry wt	0.015	-	-	-	-
2-Methylnaphthalene	mg/kg dry wt	0.012	-	-	-	-
Acenaphthene	mg/kg dry wt	0.022	-	-	-	-
Acenaphthylene	mg/kg dry wt	0.042	-	-	-	-
Anthracene	mg/kg dry wt	0.098	-	-	-	-
Benzo[a]anthracene	mg/kg dry wt	0.26	-	-	-	-
Benzo[a]pyrene (BAP)	mg/kg dry wt	0.31	-	-	-	-
Benzo[b]fluoranthene + Benzo[j] fluoranthene	mg/kg dry wt	0.32	-	-	-	-
Benzo[e]pyrene	mg/kg dry wt	0.161	-	-	-	-
Benzo[g,h,i]perylene	mg/kg dry wt	0.20	-	-	-	-
Benzo[k]fluoranthene	mg/kg dry wt	0.116	-	-	-	-
Chrysene	mg/kg dry wt	0.24	-	-	-	-
Dibenzo[a,h]anthracene	mg/kg dry wt	0.049	-	-	-	-
Fluoranthene	mg/kg dry wt	0.60	-	-	-	-
Fluorene	mg/kg dry wt	0.052	-	-	-	-
Indeno(1,2,3-c,d)pyrene	mg/kg dry wt	0.21	-	-	-	-
Naphthalene	mg/kg dry wt	< 0.03	-	-	-	-
Perylene	mg/kg dry wt	0.083	-	-	-	-
Phenanthrene	mg/kg dry wt	0.43	-	-	-	-
Benzo[a]pyrene Potency Equivalency Factor (PEF) NES	mg/kg dry wt	0.46	-	-	-	-
Benzo[a]pyrene Toxic Equivalence (TEF)	mg/kg dry wt	0.46	-	-	-	-
Pyrene	mg/kg dry wt	0.54	-	-	-	-
Haloethers Trace in SVOC Soil Samples by GC-MS						
Bis(2-chloroethoxy) methane	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
Bis(2-chloroethyl)ether	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
Bis(2-chloroisopropyl)ether	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
4-Bromophenyl phenyl ether	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17

Sample Type: Sediment						
Sample Name:		Horners 18-Mar-2018 1:05 pm	Styx @ Sawyers 18-Mar-2018 9:00 am	Styx @ Main Nth Rd 18-Mar-2018 10:40 am	Styx @ Marshlands 18-Mar-2018 12:55 pm	Styx @ Kainga 18-Mar-2018 11:55 am
Lab Number:		1946214.6	1946214.7	1946214.8	1946214.9	1946214.10
Haloethers Trace in SVOC Soil Samples by GC-MS						
4-Chlorophenyl phenyl ether	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
Nitrogen containing compounds Trace in SVOC Soil Samples, GC-MS						
N-Nitrosodiphenylamine + Diphenylamine	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
2,4-Dinitrotoluene	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
2,6-Dinitrotoluene	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
Nitrobenzene	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
N-Nitrosodi-n-propylamine	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
Organochlorine Pesticides Trace in SVOC Soil Samples by GC-MS						
Aldrin	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
alpha-BHC	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
beta-BHC	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
delta-BHC	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
gamma-BHC (Lindane)	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
4,4'-DDD	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
4,4'-DDE	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
4,4'-DDT	mg/kg dry wt	-	< 0.3	< 0.7	< 0.3	< 0.4
Dieldrin	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
Endosulfan I	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
Endosulfan II	mg/kg dry wt	-	< 0.5	< 0.5	< 0.5	< 0.5
Endosulfan sulphate	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
Endrin	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
Endrin ketone	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
Heptachlor	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
Heptachlor epoxide	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
Hexachlorobenzene	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
Polycyclic Aromatic Hydrocarbons Trace in SVOC Soil Samples						
Acenaphthene	mg/kg dry wt	-	< 0.10	< 0.10	< 0.10	< 0.10
Acenaphthylene	mg/kg dry wt	-	< 0.10	< 0.10	0.35	< 0.10
Anthracene	mg/kg dry wt	-	< 0.10	< 0.10	0.59	< 0.10
Benzo[a]anthracene	mg/kg dry wt	-	< 0.10	< 0.10	1.21	< 0.10
Benzo[a]pyrene (BAP)	mg/kg dry wt	-	< 0.15	< 0.16	1.18	< 0.17
Benzo[b]fluoranthene + Benzo[j] fluoranthene	mg/kg dry wt	-	< 0.15	< 0.16	1.23	< 0.17
Benzo[g,h,i]perylene	mg/kg dry wt	-	< 0.15	< 0.16	0.79	< 0.17
Benzo[k]fluoranthene	mg/kg dry wt	-	< 0.15	< 0.16	0.45	< 0.17
1&2-Chloronaphthalene	mg/kg dry wt	-	< 0.11	< 0.12	< 0.10	< 0.12
Chrysene	mg/kg dry wt	-	< 0.10	< 0.10	1.17	< 0.10
Dibenzo[a,h]anthracene	mg/kg dry wt	-	< 0.15	< 0.16	0.12	< 0.17
Fluoranthene	mg/kg dry wt	-	0.18	< 0.10	3.1	< 0.10
Fluorene	mg/kg dry wt	-	< 0.10	< 0.10	0.21	< 0.10
Indeno(1,2,3-c,d)pyrene	mg/kg dry wt	-	< 0.15	< 0.16	0.86	< 0.17
2-Methylnaphthalene	mg/kg dry wt	-	< 0.10	< 0.10	< 0.10	< 0.10
Naphthalene	mg/kg dry wt	-	< 0.10	< 0.10	< 0.10	< 0.10
Phenanthrene	mg/kg dry wt	-	< 0.10	< 0.10	2.6	< 0.10
Pyrene	mg/kg dry wt	-	0.19	< 0.10	2.9	< 0.10
Benzo[a]pyrene Potency Equivalency Factor (PEF) NES	mg/kg dry wt	-	< 0.4	< 0.4	1.7	< 0.5
Benzo[a]pyrene Toxic Equivalence (TEF)	mg/kg dry wt	-	< 0.4	< 0.4	1.7	< 0.5
Phenols Trace in SVOC Soil Samples by GC-MS						
4-Chloro-3-methylphenol	mg/kg dry wt	-	< 0.5	< 0.5	< 0.5	< 0.5
2-Chlorophenol	mg/kg dry wt	-	< 0.2	< 0.2	< 0.2	< 0.2
2,4-Dichlorophenol	mg/kg dry wt	-	< 0.2	< 0.2	< 0.2	< 0.2

Sample Type: Sediment						
Sample Name:		Horners 18-Mar-2018 1:05 pm	Styx @ Sawyers 18-Mar-2018 9:00 am	Styx @ Main Nth Rd 18-Mar-2018 10:40 am	Styx @ Marshlands 18-Mar-2018 12:55 pm	Styx @ Kainga 18-Mar-2018 11:55 am
Lab Number:		1946214.6	1946214.7	1946214.8	1946214.9	1946214.10
Phenols Trace in SVOC Soil Samples by GC-MS						
2,4-Dimethylphenol	mg/kg dry wt	-	< 0.4	< 0.4	< 0.4	< 0.4
3 & 4-Methylphenol (m- + p-cresol)	mg/kg dry wt	-	< 0.4	< 0.4	< 0.4	< 0.4
2-Methylphenol (o-Cresol)	mg/kg dry wt	-	< 0.2	< 0.2	< 0.2	< 0.2
2-Nitrophenol	mg/kg dry wt	-	< 0.4	< 0.4	< 0.4	< 0.4
Pentachlorophenol (PCP)	mg/kg dry wt	-	< 6	< 6	< 6	< 6
Phenol	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
2,4,5-Trichlorophenol	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
2,4,6-Trichlorophenol	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
Plasticisers Trace in SVOC Soil Samples by GC-MS						
Bis(2-ethylhexyl)phthalate	mg/kg dry wt	-	1.6	< 0.7	< 0.5	< 0.7
Butylbenzylphthalate	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
Di(2-ethylhexyl)adipate	mg/kg dry wt	-	< 0.2	< 0.2	< 0.2	< 0.2
Diethylphthalate	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
Dimethylphthalate	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
Di-n-butylphthalate	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
Di-n-octylphthalate	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
Other Halogenated compounds Trace in SVOC Soil Samples by GC-MS						
1,2-Dichlorobenzene	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
1,3-Dichlorobenzene	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
1,4-Dichlorobenzene	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
Hexachlorobutadiene	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
Hexachloroethane	mg/kg dry wt	-	< 0.3	< 0.4	< 0.3	< 0.4
1,2,4-Trichlorobenzene	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
Other SVOC Trace in SVOC Soil Samples by GC-MS						
Benzyl alcohol	mg/kg dry wt	-	< 1.5	< 1.6	< 1.2	< 1.7
Carbazole	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17
Dibenzofuran	mg/kg dry wt	-	< 0.15	< 0.16	0.18	< 0.17
Isophorone	mg/kg dry wt	-	< 0.15	< 0.16	< 0.12	< 0.17

Sample Name:		Kaputone @ Blakes 18-Mar-2018 11:20 am	Wilsons 18-Mar-2018 12:30 pm			
Lab Number:		1946214.11	1946214.12			
Individual Tests						
Dry Matter	g/100g as rcvd	21	16.5	-	-	-
Total Recoverable Arsenic	mg/kg dry wt	9.8	30	-	-	-
Total Recoverable Copper	mg/kg dry wt	6.3	33	-	-	-
Total Recoverable Lead	mg/kg dry wt	17.3	51	-	-	-
Total Recoverable Zinc	mg/kg dry wt	230	430	-	-	-
Total Organic Carbon*	g/100g dry wt	1.68	8.9	-	-	-
7 Grain Sizes Profile						
Dry Matter of Sieved Sample	g/100g as rcvd	32	21	-	-	-
Fraction >/= 2 mm*	g/100g dry wt	6.7	5.1	-	-	-
Fraction < 2 mm, >/= 1 mm*	g/100g dry wt	3.5	5.7	-	-	-
Fraction < 1 mm, >/= 500 µm*	g/100g dry wt	4.0	6.2	-	-	-
Fraction < 500 µm, >/= 250 µm*	g/100g dry wt	13.0	8.2	-	-	-
Fraction < 250 µm, >/= 125 µm*	g/100g dry wt	27.3	11.1	-	-	-
Fraction < 125 µm, >/= 63 µm*	g/100g dry wt	27.2	16.7	-	-	-
Fraction < 63 µm*	g/100g dry wt	18.3	47.1	-	-	-
Haloethers Trace in SVOC Soil Samples by GC-MS						
Bis(2-chloroethoxy) methane	mg/kg dry wt	< 0.3	< 0.4	-	-	-
Bis(2-chloroethyl)ether	mg/kg dry wt	< 0.3	< 0.4	-	-	-
Bis(2-chloroisopropyl)ether	mg/kg dry wt	< 0.3	< 0.4	-	-	-

Sample Type: Sediment						
Sample Name:		Kaputone @ Blakes 18-Mar-2018 11:20 am	Wilsons 18-Mar-2018 12:30 pm			
Lab Number:		1946214.11	1946214.12			
Haloethers Trace in SVOC Soil Samples by GC-MS						
4-Bromophenyl phenyl ether	mg/kg dry wt	< 0.3	< 0.4	-	-	-
4-Chlorophenyl phenyl ether	mg/kg dry wt	< 0.3	< 0.4	-	-	-
Nitrogen containing compounds Trace in SVOC Soil Samples, GC-MS						
N-Nitrosodiphenylamine + Diphenylamine	mg/kg dry wt	< 0.6	< 0.7	-	-	-
2,4-Dinitrotoluene	mg/kg dry wt	< 0.6	< 0.7	-	-	-
2,6-Dinitrotoluene	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Nitrobenzene	mg/kg dry wt	< 0.3	< 0.4	-	-	-
N-Nitrosodi-n-propylamine	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Organochlorine Pesticides Trace in SVOC Soil Samples by GC-MS						
Aldrin	mg/kg dry wt	< 0.3	< 0.4	-	-	-
alpha-BHC	mg/kg dry wt	< 0.3	< 0.4	-	-	-
beta-BHC	mg/kg dry wt	< 0.3	< 0.4	-	-	-
delta-BHC	mg/kg dry wt	< 0.3	< 0.4	-	-	-
gamma-BHC (Lindane)	mg/kg dry wt	< 0.3	< 0.4	-	-	-
4,4'-DDD	mg/kg dry wt	< 0.3	< 0.4	-	-	-
4,4'-DDE	mg/kg dry wt	< 0.3	< 0.4	-	-	-
4,4'-DDT	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Dieldrin	mg/kg dry wt	< 0.3	< 0.4	-	-	-
Endosulfan I	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Endosulfan II	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Endosulfan sulphate	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Endrin	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Endrin ketone	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Heptachlor	mg/kg dry wt	< 0.3	< 0.4	-	-	-
Heptachlor epoxide	mg/kg dry wt	< 0.3	< 0.4	-	-	-
Hexachlorobenzene	mg/kg dry wt	< 0.3	< 0.4	-	-	-
Polycyclic Aromatic Hydrocarbons Trace in SVOC Soil Samples						
Acenaphthene	mg/kg dry wt	< 0.13	< 0.17	-	-	-
Acenaphthylene	mg/kg dry wt	< 0.13	< 0.17	-	-	-
Anthracene	mg/kg dry wt	< 0.13	< 0.17	-	-	-
Benzo[a]anthracene	mg/kg dry wt	< 0.13	0.30	-	-	-
Benzo[a]pyrene (BAP)	mg/kg dry wt	< 0.3	< 0.4	-	-	-
Benzo[b]fluoranthene + Benzo[j] fluoranthene	mg/kg dry wt	< 0.3	0.5	-	-	-
Benzo[g,h,i]perylene	mg/kg dry wt	< 0.3	0.4	-	-	-
Benzo[k]fluoranthene	mg/kg dry wt	< 0.3	< 0.4	-	-	-
1&2-Chloronaphthalene	mg/kg dry wt	< 0.18	< 0.3	-	-	-
Chrysene	mg/kg dry wt	< 0.13	0.37	-	-	-
Dibenzo[a,h]anthracene	mg/kg dry wt	< 0.3	< 0.4	-	-	-
Fluoranthene	mg/kg dry wt	< 0.13	0.64	-	-	-
Fluorene	mg/kg dry wt	< 0.13	< 0.17	-	-	-
Indeno(1,2,3-c,d)pyrene	mg/kg dry wt	< 0.3	0.5	-	-	-
2-Methylnaphthalene	mg/kg dry wt	< 0.13	< 0.17	-	-	-
Naphthalene	mg/kg dry wt	< 0.13	< 0.17	-	-	-
Phenanthrene	mg/kg dry wt	< 0.13	0.41	-	-	-
Pyrene	mg/kg dry wt	0.14	0.74	-	-	-
Benzo[a]pyrene Potency Equivalency Factor (PEF) NES	mg/kg dry wt	< 0.6	< 0.8	-	-	-
Benzo[a]pyrene Toxic Equivalence (TEF)	mg/kg dry wt	< 0.7	< 0.9	-	-	-
Phenols Trace in SVOC Soil Samples by GC-MS						
4-Chloro-3-methylphenol	mg/kg dry wt	< 0.6	< 0.7	-	-	-
2-Chlorophenol	mg/kg dry wt	< 0.3	< 0.4	-	-	-

Sample Type: Sediment						
Sample Name:		Kaputone @ Blakes 18-Mar-2018 11:20 am	Wilsons 18-Mar-2018 12:30 pm			
Lab Number:		1946214.11	1946214.12			
Phenols Trace in SVOC Soil Samples by GC-MS						
2,4-Dichlorophenol	mg/kg dry wt	< 0.3	< 0.4	-	-	-
2,4-Dimethylphenol	mg/kg dry wt	< 0.4	< 0.4	-	-	-
3 & 4-Methylphenol (m- + p-cresol)	mg/kg dry wt	< 0.6	< 0.7	-	-	-
2-Methylphenol (o-Cresol)	mg/kg dry wt	< 0.3	< 0.4	-	-	-
2-Nitrophenol	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Pentachlorophenol (PCP)	mg/kg dry wt	< 6	< 7	-	-	-
Phenol	mg/kg dry wt	< 0.6	< 0.7	-	-	-
2,4,5-Trichlorophenol	mg/kg dry wt	< 0.6	< 0.7	-	-	-
2,4,6-Trichlorophenol	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Plasticisers Trace in SVOC Soil Samples by GC-MS						
Bis(2-ethylhexyl)phthalate	mg/kg dry wt	< 1.1	7.6	-	-	-
Butylbenzylphthalate	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Di(2-ethylhexyl)adipate	mg/kg dry wt	< 0.3	< 0.4	-	-	-
Diethylphthalate	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Dimethylphthalate	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Di-n-butylphthalate	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Di-n-octylphthalate	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Other Halogenated compounds Trace in SVOC Soil Samples by GC-MS						
1,2-Dichlorobenzene	mg/kg dry wt	< 0.6	< 0.7	-	-	-
1,3-Dichlorobenzene	mg/kg dry wt	< 0.6	< 0.7	-	-	-
1,4-Dichlorobenzene	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Hexachlorobutadiene	mg/kg dry wt	< 0.6	< 0.7	-	-	-
Hexachloroethane	mg/kg dry wt	< 0.6	< 0.7	-	-	-
1,2,4-Trichlorobenzene	mg/kg dry wt	< 0.3	< 0.4	-	-	-
Other SVOC Trace in SVOC Soil Samples by GC-MS						
Benzyl alcohol	mg/kg dry wt	< 3	< 4	-	-	-
Carbazole	mg/kg dry wt	< 0.3	< 0.4	-	-	-
Dibenzofuran	mg/kg dry wt	< 0.3	< 0.4	-	-	-
Isophorone	mg/kg dry wt	< 0.3	< 0.4	-	-	-

Summary of Methods

The following table(s) gives a brief description of the methods used to conduct the analyses for this job. The detection limits given below are those attainable in a relatively clean matrix. Detection limits may be higher for individual samples should insufficient sample be available, or if the matrix requires that dilutions be performed during analysis.

Sample Type: Sediment			
Test	**Method Description**	**Default Detection Limit**	**Sample No**
Individual Tests			
Environmental Solids Sample Preparation	Air dried at 35°C and sieved, <2mm fraction. Used for sample preparation. May contain a residual moisture content of 2-5%.	-	1-12
Dry Matter (Env)	Dried at 103°C for 4-22hr (removes 3-5% more water than air dry) , gravimetry. (Free water removed before analysis, non-soil objects such as sticks, leaves, grass and stones also removed). US EPA 3550.	0.10 g/100g as rcvd	1-12
Total Recoverable digestion	Nitric / hydrochloric acid digestion. US EPA 200.2.	-	1-12
Total Recoverable Arsenic	Dried sample, sieved as specified (if required). Nitric/Hydrochloric acid digestion, ICP-MS, trace level. US EPA 200.2.	0.2 mg/kg dry wt	7-12
Total Recoverable Copper	Dried sample, sieved as specified (if required). Nitric/Hydrochloric acid digestion, ICP-MS, trace level. US EPA 200.2.	0.2 mg/kg dry wt	1-12
Total Recoverable Lead	Dried sample, sieved as specified (if required). Nitric/Hydrochloric acid digestion, ICP-MS, trace level. US EPA 200.2.	0.04 mg/kg dry wt	1-12
Total Recoverable Zinc	Dried sample, sieved as specified (if required). Nitric/Hydrochloric acid digestion, ICP-MS, trace level. US EPA 200.2.	0.4 mg/kg dry wt	1-12

Sample Type: Sediment			
Test	**Method Description**	**Default Detection Limit**	**Sample No**
Total Organic Carbon*	Acid pretreatment to remove carbonates present followed by Catalytic Combustion (900°C, O2), separation, Thermal Conductivity Detector [Elementar Analyser].	0.05 g/100g dry wt	1-12
7 Grain Sizes Profile*		-	1-12
Polycyclic Aromatic Hydrocarbons Trace in Soil	Sonication extraction, SPE cleanup, GC-MS SIM analysis US EPA 8270C. Tested on as received sample [KBIs:5784,4273,2695]	0.002 - 0.010 mg/kg dry wt	1-6
Semivolatile Organic Compounds Trace in Soil by GC-MS	Sonication extraction, GPC cleanup, GC-MS FS analysis. Tested on as received sample	0.002 - 6 mg/kg dry wt	7-12
7 Grain Sizes Profile			
Dry Matter for Grainsize samples	Drying for 16 hours at 103°C, gravimetry (Free water removed before analysis).	0.10 g/100g as rcvd	1-12
Fraction >/= 2 mm*	Wet sieving with dispersant, 2.00 mm sieve, gravimetry.	0.1 g/100g dry wt	1-12
Fraction < 2 mm, >/= 1 mm*	Wet sieving using dispersant, 2.00 mm and 1.00 mm sieves, gravimetry (calculation by difference).	0.1 g/100g dry wt	1-12
Fraction < 1 mm, >/= 500 µm*	Wet sieving using dispersant, 1.00 mm and 500 µm sieves, gravimetry (calculation by difference).	0.1 g/100g dry wt	1-12
Fraction < 500 µm, >/= 250 µm*	Wet sieving using dispersant, 500 µm and 250 µm sieves, gravimetry (calculation by difference).	0.1 g/100g dry wt	1-12
Fraction < 250 µm, >/= 125 µm*	Wet sieving using dispersant, 250 µm and 125 µm sieves, gravimetry (calculation by difference).	0.1 g/100g dry wt	1-12
Fraction < 125 µm, >/= 63 µm*	Wet sieving using dispersant, 125 µm and 63 µm sieves, gravimetry (calculation by difference).	0.1 g/100g dry wt	1-12
Fraction < 63 µm*	Wet sieving with dispersant, 63 µm sieve, gravimetry (calculation by difference).	0.1 g/100g dry wt	1-12

These samples were collected by yourselves (or your agent) and analysed as received at the laboratory.

Samples are held at the laboratory after reporting for a length of time depending on the preservation used and the stability of the analytes being tested. Once the storage period is completed the samples are discarded unless otherwise advised by the client.

This certificate of analysis must not be reproduced, except in full, without the written consent of the signatory.

Martin Cowell - BSc
Client Services Manager - Environmental