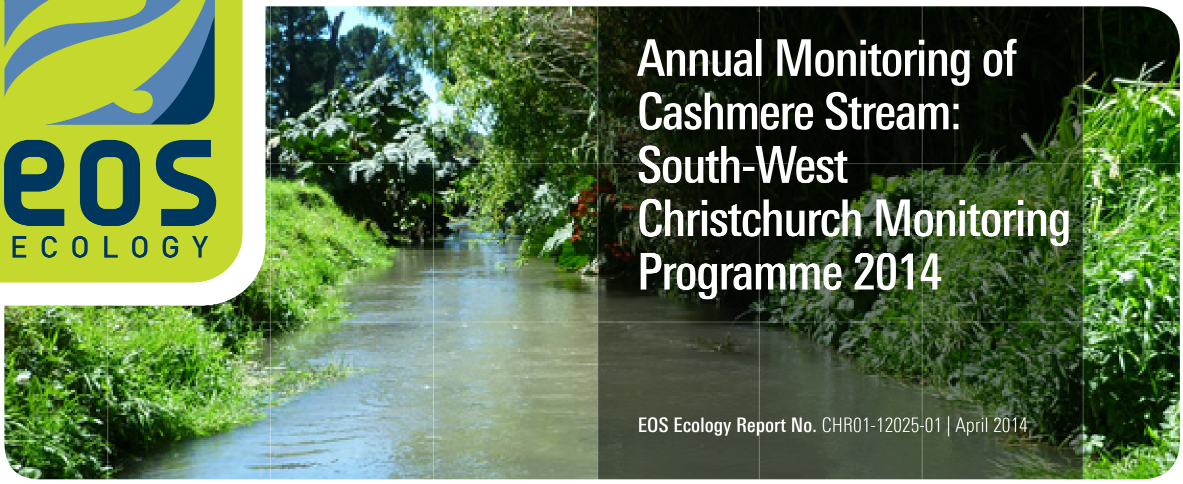

Annual Monitoring of Cashmere Stream: South-West Christchurch Monitoring Programme 2014

EOS Ecology Report No. CHR01-12025-01 | April 2014

AQUATIC RESEARCH & SCIENCE COMMUNICATION CONSULTANTS

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REPORT

Prepared for
Christchurch City Council

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EXECUTIVE SUMMARY

Christchurch City Council (CCC) holds a stormwater discharge Consent (SMP; CRC120223) from Environment Canterbury (ECan) that requires annual ecological monitoring of Cashmere Stream, with the primary aim to determine whether stormwater discharges are negatively affecting the system's aquatic ecology. The CCC established three sites along Cashmere Stream to monitor habitat, aquatic plants (algae, macrophytes) and aquatic invertebrates. Sampling for this has been carried out once previously, in February 2013. On the 3 February 2014, EOS Ecology undertook the second survey with the results presented in this report.

There was a large difference in the habitat of the three sites, with Site 3 consisting of a soft-bottomed run habitat, in comparison with a coarser substrate, faster-flowing habitat at Sites 1 and 2. In general Site 3 was more reflective of the wider Cashmere Stream environment, which is predominantly soft-bottomed and supports abundant macrophyte growth. With respect to temporal change between years, water velocity significantly increased, while total water depth and macrophyte depth significantly decreased. Similarly, fine sediment depth decreased (albeit not significantly), and macrophyte cover decreased at all three sites between years, considerably so at Sites 1 and 2. Given that this section of Cashmere Stream undergoes regular channel maintenance (macrophyte and sometimes sediment removal), it is probable that the majority of the habitat differences between years (i.e., a decrease in water and macrophyte depth, and an increase in water velocity) were due to macrophyte removal in 2014 being one month closer to the sampling date than in 2013.

The invertebrate communities were dominated at the three sites by the snail *Potamopyrgus antipodarum*, which accounted for over 40% total abundance at each site, followed by either *Paracalliope* or *Oxyethira albiceps*. Species evenness was low at all sites, with the three most abundant taxa at each site accounting for over 70% of total abundance. A total of seven caddisfly taxa were recorded from the three survey sites; however, they only represented a total of 16% of overall invertebrate abundance, with most of this (12%) being accounted for by the algal-piercing hydrotilid *O. albiceps*.

There were few differences in invertebrate community indices between sites in 2014. Percentage EPT abundance (both including and excluding Hydroptilidae) was significantly greater at Site 1, Site 2 recorded a significantly greater QMCI score, and Site 3 recorded a significantly lower UCI score. With respect to temporal change between years, there was a decreased relative abundance of the amphipod crustacean *Paracalliope* at all sites from 2013 to 2014. This could potentially be a result of macrophyte removal as this amphipod has a known affinity with macrophyte beds. With respect to community indices, taxa richness at all three sites was significantly greater in 2014 in comparison with 2013. While this was statistically significant it may be an artefact of sampling picking up taxa that are rare or have patchy distribution patterns, rather than an improvement in invertebrate community condition *per se*. The % EPT abundance (both including and excluding Hydroptilidae) at all three sites significantly increased between years, with this increase being greatest at Site 1. This difference, for % EPT abundance including hydroptilids at least, was largely a result of greater abundance of the more pollution-tolerant, algal-piercing hydroptilid *O. albiceps* at Site 1 (0.2% in 2013 compared with 20% in 2014), and as such does not indicate an improvement in the health of the aquatic invertebrate communities. The increase in the relative abundance of non-hydroptilid EPT taxa was much smaller in comparison (all sites <5.4% abundance).

All three sites recorded MCI scores that were within the 'poor' water quality category, although Site 2 had undergone a downgrade in quality ratings, dropping from a 'fair' water quality rating in 2013. QMCI and QUCI scores decreased at all sites between the two years, with this resulting in Site 1 and 2 being downgraded from 'fair' to 'poor' with respect to the QMCI water quality ratings. While the year comparison and site*year interaction for MCI/QMCI scores were not statistically significant, the drop from one water quality category to another is biologically relevant and has ramifications for meeting the stormwater discharge consent conditions. Longer-term data on QMCI scores for Site 2 and 3 indicate that Site 2 has remained stable, while Site 3 has been trending downward over time, dropping almost two QMCI points over the last decade – decreasing from a score that was almost within the 'good' water quality category in 2004, to a score that is well within the 'poor' category in 2014. Such a large reduction is more likely due to a specific environmental factor (either natural or anthropogenic) rather than just natural temporal variation.

When comparing both the habitat and aquatic invertebrate results with the surface water quality objectives (from Consent CRC120223), the only site to fail any of the habitat-based objectives in 2014 was Site 3. Given that this site is soft-bottomed, and dominated by macrophytes, it was not surprising that it exceeded the consent objective of a maximum fine sediment cover of 30% and a maximum total macrophyte cover of 30%. Macrophyte cover at both of the other sites had dropped sufficiently in 2014 such that they met the receiving environment target value. This is likely a result of macrophyte removal being undertaken one month closer to the sampling period than in the previous year. In light of the fact that the macroinvertebrate communities of all three sites were categorised as 'poor' with respect to organic pollution in 2014, all three monitoring sites failed to meet the surface water quality objective for QMCI (minimum 4–5).

It is not possible to determine if stormwater discharges are having an impact on the receiving environment due to a number of limitations in the study design. What can be taken from the monitoring results is that the macroinvertebrate communities of the three monitoring sites continue to be dominated by non-insect taxa (e.g., snails and amphipods) that are considered more tolerant of degraded systems. The 'quality' of the macroinvertebrate communities at the three monitoring sites (as inferred from the QMCI scores) has decreased from 2013 to 2014, with this reduction being associated with an increased relative abundance of pollution-tolerant taxa such as the hydroptilid caddisfly *O. albiceps,* oligochaetes and Sphaeriidae, and a decreased relative abundance of *Paracalliope*. Given that macrophtyes represent the main stable habitat in this primarily soft-bottomed system, it is likely that the effects of periodic macrophyte removal on the macroinvertebrate communities overrides the more subtle effects of stormwater discharges; however, the current study design does not allow their respective influences to be differentiated.

The main limitations of the study design are the lack of a control/reference site (to provide a baseline against which to compare habitat or community change at the 'impact' sites), and (in relation to determining stormwater discharge effects) the lack of a complementary water quality monitoring programme. As there was no site sampled that was not influenced by either macrophyte removal or stormwater discharges, it is simply not possible to determine the exact cause(s) of the overall change in the invertebrate communities between years, which could be attributable to differences in the timing of macrophyte removal, stormwater discharge impacts, or natural temporal variation. It is also difficult to discuss the potential effects of stormwater discharges in the absence of any water quality data (base flow vs rain events) within the duration of the monitoring period and within the vicinity of the habitat/

invertebrate sites. In addition, given the within-site variability of some of the invertebrate metrics, relying purely on the results of the analytical statistics on the invertebrate data should be done with caution, given the likelihood of the lack of statistical power increasing the chance of a Type II error (a false negative).

A number of recommendations are made to improve the current study design, including timing invertebrate sampling at the same time after macrophyte removal each year to avoid the complicating factor of invertebrate communities being at a different stage of recovery. Other considerations, such as first establishing the level of contamination in the environment derived from stormwater contaminants, and revising the indices that are reported on are also provided. Finally, the possibility of establishing a more conservative consent condition level regarding maximum macrophyte cover (especially for the soft-bottomed Site 3) is raised, given the wider benefits that macrophytes provide (in terms of the provision of stable habitat and cover) in soft-bottomed systems such as this one.

1 INTRODUCTION

Christchurch City Council (CCC) holds a stormwater discharge consent from Environment Canterbury (ECan) that requires annual ecological monitoring of Cashmere Stream. This Consent, for the South-West Christchurch Stormwater Management Plan (SMP; CRC120223), requires monitoring of aquatic invertebrates and habitat characteristics at three sites within Cashmere Stream. This monitoring programme, including the selection of sampling sites and sampling methodology, was established by the CCC and first carried out in February 2013. The CCC then commissioned EOS Ecology to undertake the aquatic surveys in 2014 and report on the results, as presented in this report.

The aim of this report, based on the objectives of the CCC stormwater discharge consent monitoring programme, is to (i) compare the results with the receiving environment objectives (both habitat characteristics and invertebrate community indices) included as part of the resource consent conditions for Consent CRC120223, (ii) compare the results with the previous year's (2013) monitoring results to investigate if any trends/patterns are evident, and (iii) to assess whether stormwater discharges are negatively affecting the aquatic ecology of Cashmere Stream.

2 METHODS

2.1 Site Selection

The three monitoring sites on Cashmere Stream for 2014 were the same as those surveyed on 8 February 2013, which represent the yearly monitoring programme for the South-West Christchurch Stormwater Management Plan. Each of the three survey sites (Sites 1–3) are located on the mainstem of Cashmere Stream, downstream (DS) of three tributaries: DS of Ballintines Drain (Site 1) [E1567915 N5175095], DS of Hendersons Rd Drain (Site 2) [E1567664 N5175040] and DS of Dunbars Drain (Site 3) [E1567370 N5174795] (Figure 1). According to CCC, these sites were selected to represent a waterway with high ecological values where it would be useful to observe trends over time because of the level of development planned within the catchment.

FIGURE 1 Location of the three monitoring sites on Cashmere Stream. Site photographs are provided in the Appendix (Section 9.1). Site 1: DS of Ballintines Drain; Site 2: DS of Hendersons Rd Drain; and Site 3: DS of Dunbars Drain. DS = downstream.

2.2 Sampling

Following fine weather conditions, habitat and aquatic invertebrate surveys were undertaken by EOS Ecology at each of the three monitoring sites on 3 February 2014. At each site, three equally spaced transects were placed across the stream at 10 m intervals (i.e., at 0, 10 and 20 m), and aspects of the instream habitat and aquatic invertebrate community quantified along them.

Instream habitat variables were quantified at 12 equidistant points across each of the three transects, with the first and last measurements across each transect at the water's edge. Habitat variables measured at each of these 12 points on each of the three transects (i.e., 36 points per site) included substrate composition (mud/silt/clay: <0.06 mm; sand: 0.06–2 mm; gravel: 2–16 mm; pebble: 16–64 mm; small cobble: 64–128 mm; large cobble: 128–256 mm; boulder: >256mm; bedrock/man-made concrete), presence and type of organic material (submerged and emergent macrophytes, filamentous algae and algal mats, moss/liverworts, fine/coarse detritus, and terrestrial vegetation), depths (water, macrophyte and sediment). Water velocity was measured (using a Sontek ADV meter) at 10 of the 12 points across each of the three transects (points 1 and 12 along each transect were excluded as these points were at the water's edge). As per standard convention, water velocity was measured at 0.4 x water depth, and was measured at each sampling point over a 30-second interval. General bank attributes, including lower and upper bank height and angles, lower bank undercut, and lower bank vegetative overhang were measured for each bank at each transect. Bank material composition and stability were also recorded.

A visual qualitative assessment of macrophyte cover was also assessed across each of the three transects. This involved qualitatively assessing macrophyte cover within a 1 m band along each of the three transects with the following variables recorded: visual estimation of streambed cover (%), identification of the dominant species present, and identification of the type present (emergent or submerged). Because macrophyte cover is often patchy at the site scale, only looking at three transects does not necessarily give a good estimate of cover or composition. Therefore, a visual qualitative assessment of macrophyte cover was also undertaken over the entire site (see below).

A visual qualitative assessment of a number of habitat parameters was also carried out over the entire site (i.e., site-wide assessments). The parameters measured at the site-scale included the following:

- Habitat type (% riffle/run/pool, and maximum pool depth).
- Visible sky was assessed as one of five percent cover categories (<5%, 5–25%, 25–50%, 50–75%, >75%), as per the Christchurch River Environment Assessment Survey (CREAS) criteria (McMurtrie & Suren, 2008). As per CREAS, measurements were taken in each half of the river (by splitting the channel down the centreline) and categorised as for the true right bank (TRB) or true left bank (TLB). Visible sky is a measure of how much sky is visible from the centre of the stream, and so takes into account steep banks, buildings and other objects that may be situated back from the channel but still block the sky in some way.
- Canopy tree cover was assessed as one of five percent cover categories (<5%, 5–25%, 25–50%, 50–75%, >75%), as per the CREAS criteria. As per CREAS, measurements were taken in each half of the river (by splitting the channel down the centreline) and categorised as for the true right bank (TRB) or true left bank (TLB). This is also a measure of channel shading as it is an estimate of how much of the channel is shaded by tree cover within the site.
- Substrate embeddedness (the percentage of fine sediment surrounding large particles within the streambed) was assessed as one of five percent cover categories (<5%, 5–25%, 25–50%, 50–75%, >75%), as per the CREAS criteria.

- Bank attributes (bank erosion and bank vegetation cover), were assessed as one of five percent cover categories (< 5%, 5–25%, 25–50%, 50–75%, > 75%), as per the CREAS criteria.
- Lower bank material was categorised into one of seven categories: earth (includes soil, sand, and gravel), wood, brick, rock, concrete, iron and tyres.
- Substrate composition – the percentage cover of the following particle size categories: mud/silt/clay: < 0.06 mm; sand: 0.06–2 mm; gravel: 2–16 mm; pebble: 16–64 mm; small cobble: 64–128 mm; large cobble: 128–256 mm; boulder: > 256mm; bedrock/man-made concrete, as per the CREAS criteria. Percentage fine sediment cover was calculated as the combined coverage of mud/silt/clay and sand particle size categories.
- Bryophyte (moss, liverworts) coverage.
- Macrophyte coverage and composition. Macrophytes were identified to the lowest practicable level (either to genus or species), including whether it was a submerged or emergent growth form.
- Periphyton (including algae) coverage and composition. The periphyton types recorded were classified using the groups outlined in Biggs and Kilroy (2000): thin mat/film (< 0.5 mm thick); medium mat (0.5–3 mm thick); thick mat (< 3 mm thick); filaments, short (< 2 cm long); and filaments, long (> 2 cm long).

The riparian zone condition was assessed within a 5 m band along the 20 m site on either side of the bank. The cover of 15 different vegetation types was estimated on a ranking scale of present (< 10%), common (10–50%), and abundant (> 50%). The vegetation was assessed three dimensionally so included ground, shrub, and canopy cover levels. The vegetation categories were taken from the CREAS criteria (McMurtrie & Suren, 2008).

Aquatic benthic invertebrates were collected at each transect by disturbing the substrate across an approximate 1.5 m width and within a 0.3 m band immediately upstream of a conventional kicknet (500 µm mesh size). The full range of habitat types were surveyed across each transect, including mid-channel and margin areas, inorganic substrate (e.g., the streambed), and macrophytes (aquatic plants). Each invertebrate sample was kept in a separate container, preserved in 70% isopropyl alcohol, and taken to the laboratory for identification. The contents of each sample were passed through a series of nested sieves (2 mm, 1 mm, and 500 µm) and placed in a Bogorov sorting tray. All invertebrates were counted and identified to the lowest practical level using a binocular microscope and several identification keys (Winterbourn, 1973; Winterbourn *et al.*, 2006; Chapman *et al.*, 2011). Sub-sampling was utilised for particularly large samples and the unsorted fraction scanned for taxa not already identified. The lowest sub-sampling level used for any particular size fraction of a sample collected was 12.5% (i.e., one eighth of the sample).

There were two aspects of habitat sampling that was slightly different in 2014 compared with 2013. These methodological differences were:

- The macrophyte cover assessment was altered in 2014, compared with 2013. In 2013, macrophytes were assessed over the whole site, while in 2014 they were assessed over the entire site as well as across each transect. We have chosen to present the site wide percentage cover assessment as this allows comparison with 2013 and earlier data. Additionally, site wide percentage cover provides a better indication of macrophyte cover than only looking at three transects, as macrophytes often have a patchy distribution at the site scale.

» The algal cover assessment (both site-wide and across each transect) was altered in 2014, compared with 2013. In 2013, only the 'algal mats' and 'filamentous algae' categories were used, while in 2014 the categories of Biggs & Kilroy (2000) were recorded: (thin mat/film (< 0.5 mm thick); medium mat (0.5–3 mm thick); thick mat (< 3 mm thick); filaments, short (< 2 cm long); and filaments, long (> 2 cm long)). Filamentous algae were not recorded at any of the three sites in 2013, so this change of is no consequence for inter-year comparisons.

2.3 Data Analysis

The data describing the substrate composition was simplified by creating a substrate index, such that:

Substrate index = [(0.7 x % boulders) + (0.6 x % large cobbles) + (0.5 x % small cobbles) + (0.4 x % pebbles) + (0.3 x % gravels) + (0.2 x % sand) + (0.1 x % silt) + (0.1 x % concrete/bedrock)] / 10

Where derived values for the substrate index range from 1 (i.e., a substrate of 100% silt) to 7 (i.e., a substrate of 100% boulder); the larger the index, the coarser the overall substrate. In general, coarser substrate (up to cobbles) represents better instream habitat than finer substrate. The same low coefficients for silt and concrete/bedrock reflect their uniform nature and lack of spatial heterogeneity, and in the case of silt, instability during high flow.

Invertebrate data were summarised by taxa richness, total abundance, abundance of the five most common taxa. Biotic indices calculated included the number of Ephemeroptera-Plecoptera-Trichoptera taxa (EPT taxa richness), % EPT abundance, the Macroinvertebrate Community Index (MCI), Urban Community Index (UCI), and their quantitative equivalents (QMCI and QUCI, respectively). The points below provide brief clarification of these metrics.

» Taxa richness is the number of different taxa identified in each sample. Taxa is generally a term for taxonomic groups, and in this case refers to the lowest level of classification that was obtained during the study. Taxa richness can be used as an indication of stream health or habitat type, where sites with greater taxa richness are usually healthier and/or have a more diverse habitat.

» EPT refers to three Orders of invertebrates that are generally regarded as 'cleanwater' taxa. These Orders are Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies); forming the acronym EPT. These taxa are relatively intolerant of organic enrichment or other pollutants and habitat degradation. The exception to this are the hydroptilid caddisflies (e.g., Trichoptera: Hydroptilidae: *Oxyethira, Paroxyethira*), which are algal piercers and often found in high numbers in nutrient-enriched waters and degraded with high algal content. For this reason, EPT metrics are presented with and without these taxa. EPT taxa richness and % EPT abundance can provide a good indication as to the health of a particular site. The disappearance and reappearance of EPT taxa also provides evidence of whether a site is impacted or recovering from a disturbance. EPT taxa are generally diverse in non-impacted, non-urbanised streams, although there is a small subset of EPT taxa that are also found in urbanised waterways.

» In the mid-1980s, the MCI was developed as an index of community integrity for use in stony riffles in New Zealand streams and rivers, and can be used to determine the level of organic enrichment for these types of streams (Stark, 1985). Although developed to assess nutrient enrichment, the MCI will respond to any disturbance that alters macroinvertebrate community composition (Boothroyd & Stark, 2000), and as such is used widely to evaluate the general health of waterways in New Zealand. Recently a variant for use in streams with a streambed of sand/silt/mud (i.e., soft-bottomed) was developed by Stark and Maxted (2007b) and is referred to as the MCI-sb. Both the hard-bottomed (MCI-hb) and soft-bottomed (MCI-sb) versions calculate an overall score for each sample, which is based on pollution-tolerance values for each invertebrate taxon that range from 1 (very pollution tolerant) to 10 (pollution-sensitive). MCI-hb and MCI-sb are calculated using presence/absence data and a quantitative version has been developed that incorporates abundance data and so gives a more accurate result by differentiating rare taxa from abundant taxa (QMCI-hb, QMCI-sb). MCI (QMCI) scores of ≥120 (≥6.00) are interpreted as 'excellent', 100–119 (5.00–5.99) as 'good', 80–99 (4.00–4.99) as 'fair', and <80 (<4.00) as 'poor' (Stark & Maxted, 2007a). As mud/silt/clay (<0.06 mm) was the dominant substrate size class at Site 3 (DS of Dunbars Drain), only the soft-bottomed variants (MCI-sb and QMCI-sb) were used at this site. The hard-bottomed variants were used at the remaining two sites (Sites 1 & 2) as these sites were dominated by stony substrata (e.g., pebble).

» The UCI/QUCI score can be used to determine the health of urban and peri-urban streams by combining tolerance values for invertebrates with presence/absence or abundance invertebrate data (Suren *et al.*, 1998). This biotic index is indicative of habitat relationships, and to some degree incorporates urban impacts. Negative scores are indicative of invertebrate communities tolerant of slow-flowing water conditions associated with soft-bottomed streams (and often with a high biomass of macrophytes), whereas positive scores are indicative of communities present in fast-flowing streams with coarse substrates (Suren *et al.*, 1998).

One-way analysis of variance (ANOVA) was used to investigate differences in habitat attributes and aquatic invertebrate community indices between sites (Sites 1–3) in 2014. Data transformations were used (e.g., square root and fourth root), where necessary, to fulfil the requirements of the parametric tests (i.e., equal variance and normality). The level of significance was set at $p = 0.05$. Where significant differences were observed, the *post-hoc* Tukey test was used to find site means that were significantly different.

In addition, two-way ANOVAs – with site and time as main factors – were used to investigate differences in aquatic invertebrate community indices and habitat attributes between sites (Sites 1–3) and years (2013 & 2014). For the purposes of considering temporal change, only significant year comparisons and site*year interactions were discussed within the text. Although significant site results were also included in the tables for completeness, they were not relevant to discuss further as site-based differences are better interpreted on the current year's data only.

For the ANOVAs on invertebrate community indices, tests were all based on a single value per transect (i.e., three values per site). With respect to the ANOVAs on habitat attributes, tests were based on a single value per transect for channel width, substrate index, total water depth, fine sediment depth and macrophyte depth. Although total water depth, fine sediment depth and macrophyte depth are measured across each of the 12 equidistant points on each transect, normality could not be achieved by including all 36 data points per transect due to the high level of variation between transect points. For water velocity, all 10 data points per transect were used.

With respect to figures, the mean and standard error (SE) values presented on the graphs were calculated from the full set of data points recorded for each attribute at each site (e.g., 36 data points for total water depth, fine sediment depth, and macrophyte depth; 30 data points for water velocity, 3 data points for channel width, substrate index, and all the invertebrate community indices).

3 RESULTS

3.1 Habitat

3.1.1 Overview of 2014 Results

The adjacent land use for Cashmere Stream near the three monitoring sites is of rural (farming) and residential use. The banks of Site 1 (DS of Ballintines Drain) and Site 3 (DS of Dunbars Drain) were comprised mainly of natural earth (Table 1). As a road bridge and footbridge were present at Site 2 (DS of Hendersons Rd Drain), the banks were comprised of a mixture of both natural earth and brick/concrete, with some corrugated iron also present (<5% coverage) (Table 1). Riparian vegetation composition was typically comprised of a grass/herb mix at all three sites, with Site 1 having a canopy of mainly native trees and shrubs, while Site 3 had a canopy comprising mostly of exotic trees and shrubs (Table 1). Canopy shade was greatest at Site 1 (up to 50–75% on the TRB), but only reached 5–25% at Site 3. While canopy shade was low at Site 2 (<5% on both banks), site-wide shading reached >75% when the shade from the overhead road bridge and footbridge was also taken into account. Sites 2 and 3 were predominantly run habitat, while Site 1 contained an even proportion of riffle (50%) and run (50%) habitat (Table 1). Sites 1 and 2 were dominated by a coarser substrate (pebble: 16–64 mm), while Site 3 was dominated by mud/silt/clay (<0.06 mm) (Table 1). Due to the predominance of fine substrate at Site 3, substrate embeddedness was also greatest at this site (Table 1).

Five of the six analysed instream habitat variables were significantly different between sites (Figure 2; Table 2). The wetted channel width at Site 1 was significantly narrower than the other two sites (Figure 2; Table 2), while water velocity at Sites 1 and 2 was significantly faster than that at Site 3 (Figure 2; Table 2). The highest mean water velocity (0.65 m/s) was recorded at Site 1 (Figure 2). In terms of substrate, Site 3 had a significantly lower substrate index value (made up of fine silt) than the other two sites that had a greater proportion of pebble/cobble substrate (Figure 2; Table 2). As a corollary to the finer sediment of Site 3, this site also had a significantly greater fine sediment depth than the other two sites (Figure 2; Table 2). Site 3 was also significantly deeper than the other two sites (Figure 2; Table 2). Although macrophyte depth was greater at Site 3 in comparison with the other two sites (Figure 2), this difference was not statistically significant due to a large within-site variation at Site 2 and 3, which would indicate a patchy distribution of macrophytes across the transects.

Despite the lack of a statistically significant difference in macrophyte depth, it was clear that Site 3 had a much greater amount of macrophyte cover than the other two sites (Table 3). In terms of macrophyte cover, Site 3 had the greatest total cover (at 65%), which was primarily made up of the exotic macrophytes *Elodea canadensis* (40%) and *Potamogeton crispus* (20%) (Table 3). Total macrophyte cover was low at Site 1 and 2 (8% and 15%, respectively), and was made up mostly of *P. crispus* (5 and 10% cover, respectively) (Table 3). With the exception of *Azolla* (native free-floating fern) and *Lemna minor* (duckweed), all macrophyte taxa recorded were exotic (i.e., introduced) species. In terms of algal cover, only one of the five Biggs & Kilroy (2000) algal types (algal mats (thin)) were recorded, and this was at 40% and 30% cover at Sites 1 and 2, respectively. At 10% cover, bryophytes (mosses/liverworts) had a greater coverage at Site 1 (where they would have been attached to the coarse substrate) than at the other sites (Table 3).

3.1.2 Temporal Change (2013–2014)

Three of the six analysed instream habitat variables were significantly different between 2013 and 2014 (Figure 2; Table 4). Water velocity at all three sites was significantly greater in 2014 in comparison with 2013 (Figure 2; Table 4). Total water depth and macrophyte depth reduced significantly from 2013 to 2014 (Figure 2; Table 4). There was no significant site*year interaction for any of the six analysed habitat variables (Table 4).

TABLE 1 Habitat attributes from each of the three monitoring sites on Cashmere Stream for 2014. These attributes were measured over the entire site (i.e., a single site-wide value). TLB = true left bank, TRB = true right bank. DS = downstream.

Habitat attributes		Site 1: DS of Ballintines Drain	Site 2: DS of Hendersons Rd Drain	Site 3: DS of Dunbars Drain
Substrate composition (dominant substrate is in blue)	Man-made (concrete)	1%	5%	None
	Boulder	2%	2%	None
	Large cobble	3%	1%	None
	Small cobble	4%	2%	None
	Pebble	**70%**	**50%**	None
	Gravel	5%	25%	None
	Sand	5%	5%	None
	Mud/silt/clay	10%	10%	**100%**
Surrounding land use	TLB	100% residential (new)	50% rural with stock (unfenced) & 50% residential (old)	100% rural with stock (fenced)
	TRB	100% residential (new)	50% rural with stock (unfenced) & 50% residential (old)	100% residential (old)
Habitat type (% riffle:run:pool)		50:50:0	5:90:5	0:100:0
Bank material composition		Earth (with minor wood on TLB/TRB & minor concrete on TLB)	Earth & concrete (with minor corrugated iron on TLB)	Earth (with minor concrete on TLB & minor wood on TRB)
Riparian vegetation		Grass/herb mix, some low ground cover, ferns, rushes, native shrubs, native trees and exotic deciduous trees	Grass/herb mix, some low ground cover, ferns and native trees	Grass/herb mix, some low ground cover, exotic shrubs, native trees and exotic deciduous trees
Canopy cover (% stream shade)	TLB	25–50%	<5% (>75 % when including bridges)	<5%
	TRB	50–75%	<5% (>75 % when including bridges)	5–25%
Substrate embeddedness		25–50%	50–75%	>75%

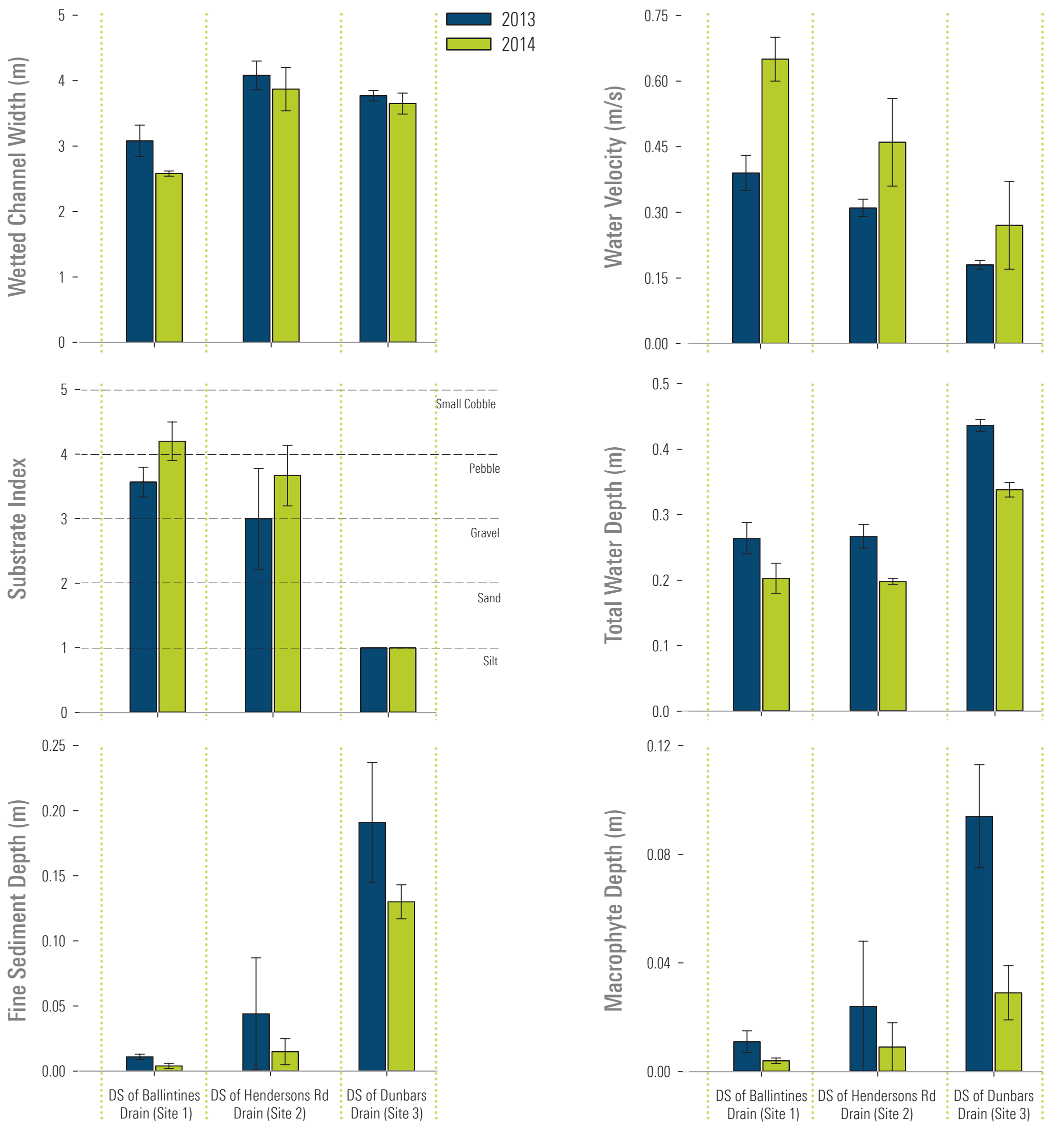

FIGURE 2 Mean (± 1 standard error) habitat attribute values at each of the three monitoring sites on Cashmere Stream for both 2013 and 2014. Aquatic invertebrate and habitat surveys were undertaken on 8 February 2013 and 3 February 2014 by EOS Ecology. n = 3 (per individual bar) for channel width and substrate index; n = 30 (per individual bar) for water velocity; n = 36 (per individual bar) for total water depth, fine sediment depth and macrophyte depth.

TABLE 2 Results of the one-way analysis of variance (ANOVA) on aquatic habitat attributes from 2014 data. The Tukey *post-hoc* test was used to find which site means were significantly different. n/s = not significant; n/a = not applicable.

Habitat parameter	ANOVA result	Significant site differences
Channel width	$F_{2,8}$= 10.4, p<0.05	2=3>1
Water velocity	$F_{2,89}$= 8.6, p<0.001	1=2>3
Substrate index	$F_{2,8}$= 28.7, p<0.001	1=2>3
Total water depth	$F_{2,8}$= 28.8, p<0.001	3>2 =1
Fine sediment depth	$F_{2,8}$= 49.8, p<0.001	3>2 =1
Macrophyte depth	n/s	n/a

TABLE 3 Macrophyte and periphyton attributes from each of the three monitoring sites on Cashmere Stream for 2014. These attributes were measured over the entire site (i.e., a single site-wide value). Total macrophyte cover includes both emergent and submerged macrophytes. DS = downstream.

Macrophyte & periphyton attribute	Site 1: DS of Ballintines Drain	Site 2: DS of Hendersons Rd Drain	Site 3: DS of Dunbars Drain
Aquatic vegetation & organic material cover*	Algae – thin mat/film (<0.5 mm thick): 40%	Algae – thin mat/film (<0.5 mm thick): 30%	*Elodea canadensis* (Canadian pondweed): 40%
	Moss/liverworts: 10%	*Potamogeton crispus* (curly pondweed): 10%	*Potamogeton crispus* (curly pondweed): 20%
	Potamogeton crispus (curly pondweed): 5%	Moss/liverworts: 1%	*Nitella*: 2%
	Terrestrial roots/vegetation: 4%	*Ranunculus trichophyllus* (water buttercup): 1%	Terrestrial roots/vegetation: 2%
	Ranunculus trichophyllus (water buttercup): 1%	*Glyceria* (sweetgrass): 1%	Moss/liverworts: 1%
	Glyceria (sweetgrass): 1%	*Elodea canadensis* (Canadian pondweed): 1%	*Ranunculus trichophyllus* (water buttercup): 1%
	Elodea canadensis (Canadian pondweed): 1%	*Lemna minor* (duckweed): 1%	*Lemna minor* (duckweed): 1%
	Fine detritus: 1%	*Azolla*: 1%	*Azolla*: 1%
	Woody debris: 1%	Terrestrial roots/vegetation: 1%	Fine detritus: 1%
		Fine detritus: 1%	Woody debris: 1%
		Woody debris: 1%	
Emergent macrophyte cover	1%	4%	3%
Total macrophyte cover†	8%	15%	65%

* Only those aquatic vegetation and organic material cover categories that were present are shown (i.e., all other macrophyte and periphyton attributes had zero values).

† Total macrophyte cover only includes those macrophyte species from the 'aquatic vegetation and organic material cover' category, and so excludes algae, moss/liverworts, terrestrial roots/vegetation, fine detritus and woody debris.

TABLE 4 Results of the two-way analysis of variance (ANOVA) (with site and year as main factors) on aquatic habitat attributes from 2013 and 2014. The Tukey *post-hoc* test was used to find which site means were significantly different between years. n/s = not significant; n/a = not applicable.

Habitat parameter	Site	Year	Site*Year	Comparisons between years
Wetted channel width	$F_{2, 17} = 17.2$, $p<0.001$	n/s	n/s	n/a
Water velocity	$F_{2, 178} = 17.0$, $p<0.001$	$F_{1, 178} = 16.6$, $p<0.001$	n/s	2014>2013
Substrate index	$F_{2, 17} = 29.1$, $p<0.001$	n/s	n/s	n/a
Total water depth	$F_{2, 17} = 57.1$, $p<0.001$	$F_{1, 17} = 31.5$, $p<0.001$	n/s	2013>2014
Fine sediment depth	$F_{2, 17} = 19.2$, $p<0.001$	n/s	n/s	n/a
Macrophyte depth	$F_{2, 17} = 8.9$, $p<0.01$	$F_{1, 17} = 6.8$, $p<0.05$	n/s	2013>2014

3.2 Aquatic Invertebrates

3.2.1 Overview of 2014 Results

A total of 33 invertebrate taxa were recorded from the three aquatic invertebrate and habitat monitoring sites in 2014, with taxa richness per site ranging from 24 to 29. The most diverse groups were the true flies (Diptera: 11 taxa), followed by caddisflies (Trichoptera: 7 taxa), crustaceans (Crustacea: 4 taxa) and molluscs (Mollusca: 4 taxa) and water bugs (Hemiptera: 2 taxa). Damselflies (Odonata), mites (Arachnida: Acari), leeches (Hirudinea), roundworms (Nematoda), and worms (Oligochaeta) were each represented by a single taxon.

The snail *Potamopyrgus antipodarum* was the dominant species, accounting for 51% of all invertebrates captured. This was followed by the algal-piercing hydroptilid caddisfly *Oxyethira albiceps* and the amphipod crustacean *Paracalliope, which* each accounted for 12% of all invertebrates captured. These three taxa were widespread, being recorded from all three sites. 'Cleanwater' EPT taxa were uncommon across all sites, with no mayflies (Ephemeroptera) or stoneflies (Plecoptera) recorded. Of the caddisflies (Trichoptera), the most abundant and widespread taxon recorded was the pollution-tolerant hydroptilid *O. albiceps* (12% of total invertebrate abundance). The remaining six caddisfly taxa (which are considered 'cleanwater' species) – *Hudsonema amabile* (1.8%), *Hydrobiosis parumbripennis* (0.9%), *Triplectides* (0.9%), *Psilochorema* (0.6%), *Oecetis unicolor* (0.1%) and *Polyplectropus* (0.02%) – combined accounted for 4.4% of total invertebrate abundance.

In terms of the five most abundant taxa, the communities of all three sites in 2014 were broadly similar in that *P. antipodarum* was numerically dominant (>40%) at all sites (Figure 3), followed by either *Paracalliope* or *O. albiceps* (Figure 3). These three taxa accounted for the top three taxa at Site 1, while *P. antipodarum, Paracalliope*, and either Oligochaete worms or pea clams (Sphaeriidae) accounted for the top three taxa at Site 2 and 3, respectively (Figure 3). Species evenness was low at all sites, with the three most abundant taxa at each site accounting for over 70% of total invertebrate abundance.

With respect to community indices, total abundance (i.e., total number of invertebrate individuals per sample), taxa richness and EPT taxa richness were statistically similar between the three sites (Figure 4; Table 5). Similarly, as *O. albiceps* was the only hydroptilid caddisfly recorded from any of the three sites, EPT richness excluding hydroptilids also did not significantly vary between sites (Figure 4; Table 5). Percentage EPT abundance (both including and excluding hydroptilids), was significantly greater at Site 1. This difference was most dramatic in the '% EPT including hydroptilids', and was driven by the large numbers of the hydroptilid caddisfly *O. albiceps* at this site (Figure 3 and 4; Table 5). While there was no significant difference in MCI scores between sites, the quantitative variant QMCI was significantly greater at Site 2 compared with Site 3 (Figure 4; Table 5). Both the MCI and QMCI scores for the three sites indicated that all sites were in the 'poor' category with regards to organic pollution in 2014 (Figure 4). The UCI scores were significantly different between sites, with Site 3 having a significantly lower score than the other two sites (Figure 4; Table 5). Site 3 also recorded the lowest QUCI score of the three sites; however, the difference between sites was not statistically significant due to a large within-site variation (Figure 4; Table 5).

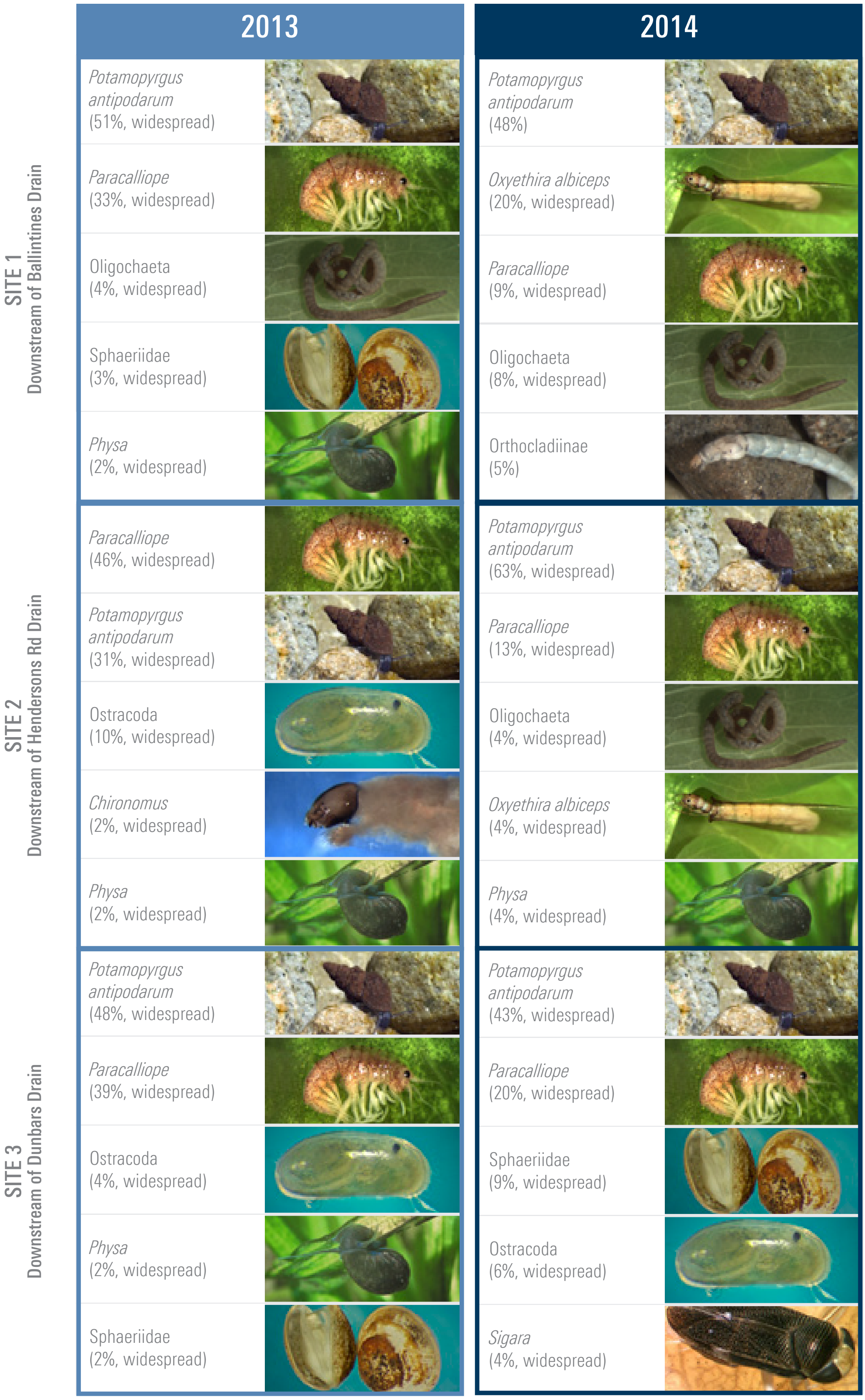

FIGURE 3 Photographs of the five most abundant taxa (% relative abundance per site indicated) from the three monitoring sites on Cashmere Stream for both 2013 and 2014. Those taxa designated as 'widespread' were found at all three monitoring sites in that particular survey year.

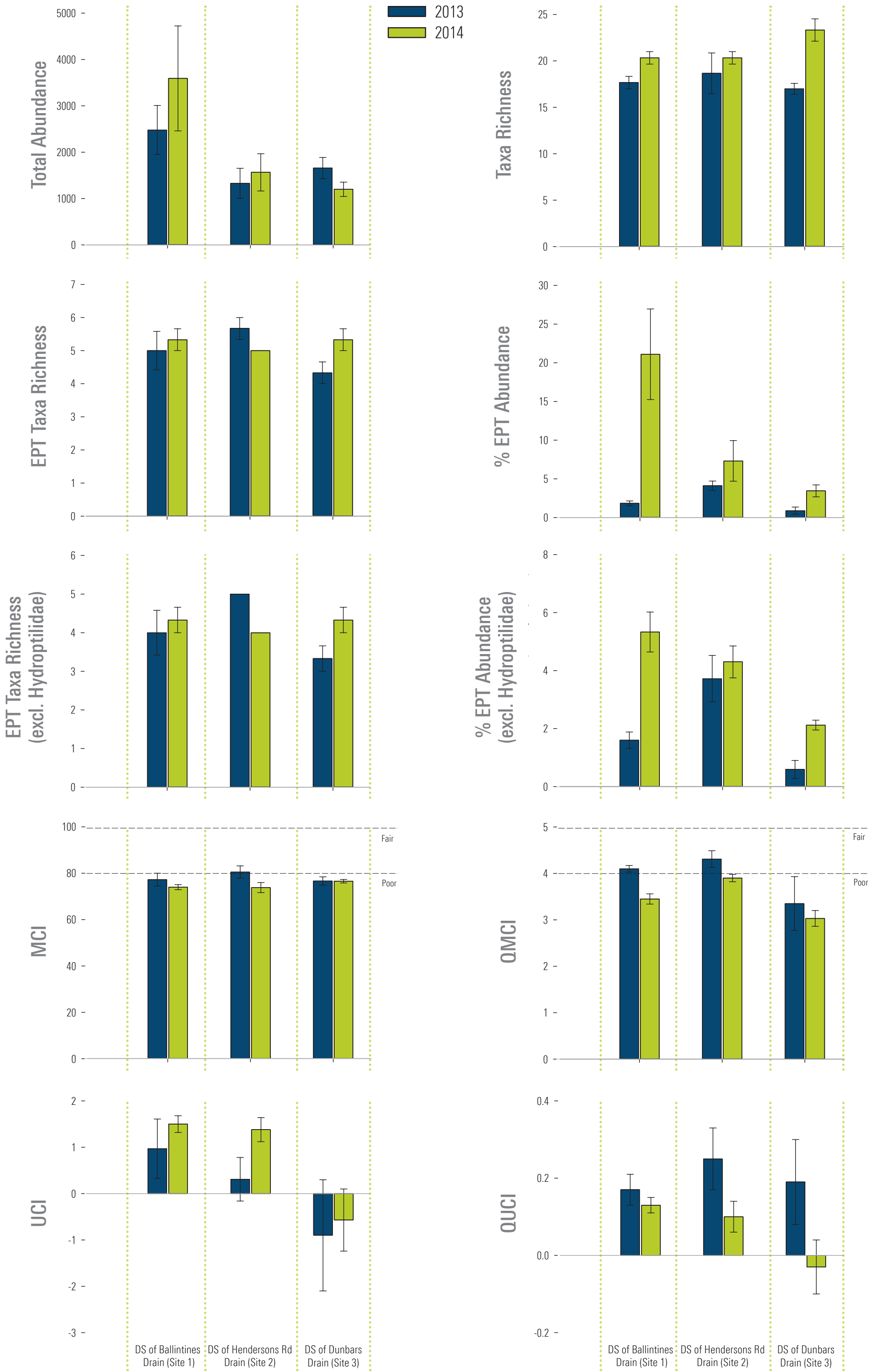

FIGURE 4 Mean (± 1 standard error) community indices at each of the three monitoring sites on Cashmere Stream for both 2013 and 2014. EPT metrics are presented with and without Hydroptilidae, as hydroptilid trichopterans (*Oxyethira* spp. and *Paroxyethira* spp.) are algal piercers that are often abundant in nutrient-enriched waterways. n = 3 (per individual bar) for all indices. The dashed lines on the MCI and QMCI graphs show the 'quality class' interpretation categories of Stark and Maxted (2007a). DS = downstream.

3.2.2 Temporal Change (2013–2014)

In terms of the most abundant taxa, Site 3 remained dominated by the same two taxa (*P. antipodarum* and *Paracalliope*), although the overall abundance of *Paracalliope* did drop marginally (Figure 3). At Site 2, there was a reversal in order of the two most abundant taxa from 2013 to 2014, with *P. antipodarum* dominating over *Paracalliope* in 2014, in contrast with 2013 (Figure 3). At Site 1, *O. albiceps* markedly increased in relative abundance (0.2% in 2013, to 20% in 2014), while *Paracalliope* decreased in relative abundance between 2013 and 2014 (33% in 2013, to 9% in 2014)(Figure 3).

With respect to community indices, taxa richness at all three sites was significantly greater in 2014 in comparison with 2013 (Figure 4; Table 6). There were no significant differences in EPT taxa richness,

TABLE 5 Results of the one-way analysis of variance (ANOVA) on community indices from 2014. The Tukey *post-hoc* test was used to find which site means were significantly different. n/s = not significant; n/a = not applicable.

Community indices	ANOVA result	Signifacnt site differences
Total abundance	n/s	n/a
Taxa richness	n/s	n/a
EPT taxa richness	n/s	n/a
% EPT abundance	$F_{2,8}$= 6.2, p<0.05	1>3, 1=2, 2=3
EPT taxa richness (excl. Hydroptilidae)*	n/s	n/a
% EPT abundance (excl. Hydroptilidae)*	$F_{2,8}$= 9.9, p<0.05	1>3, 1=2, 2=3
MCI	n/s	n/a
QMCI	$F_{2,8}$= 11.7, p<0.01	2>3, 1=2, 1=3
UCI	$F_{2,8}$= 7.4, p<0.05	1=2>3
QUCI	n/s	n/a

* Hydroptilidae trichopterans (*Oxyethira* spp. and *Paroxyethira* spp.) are excluded as they are algal piercers that are often abundant in nutrient-enriched waterways.

TABLE 6 Results of the two-way analysis of variance (ANOVA) (with site and year as main factors) on community indices from 2013 and 2014. The Tukey *post-hoc* test was used to find which site means were significantly different. n/s = not significant; n/a = not applicable.

Community indices	Site	Year	Site*Year	Comparisons between years
Total abundance	$F_{2,17}$= 5.4, p<0.05	n/s	n/s	n/a
Taxa richness	n/s	$F_{1,17}$= 14.4, p<0.01	n/s	2014>2013
EPT taxa richness	n/s	n/s	n/s	n/a
% EPT abundance	$F_{2,17}$= 8.8, p<0.01	$F_{1,17}$= 26.4, p<0.001	$F_{2,17}$= 6.8, p<0.05	2014>2013
EPT taxa richness (excl. Hydroptilidae)*	n/s	n/s	$F_{2,17}$= 4.5, p<0.05	n/a
% EPT abundance (excl. Hydroptilidae)*	$F_{2,17}$= 14.8, p<0.001	$F_{1,17}$= 21.0, p<0.001	$F_{2,17}$= 4.8, p<0.05	2014>2013
MCI	n/s	n/s	n/s	n/a
QMCI	$F_{2,17}$= 5.8, p<0.05	n/s	n/s	n/a
UCI	$F_{2,17}$= 5.0, p<0.05	n/s	n/s	n/a
QUCI	n/s	$F_{1,17}$= 6.5, p<0.05	n/s	2013>2014

* Hydroptilidae trichopterans (*Oxyethira* spp. and *Paroxyethira* spp.) are excluded as they are algal piercers that are often abundant in nutrient-enriched waterways.

but there was a significant site*year interaction for EPT taxa richness excluding hydroptilids, which was due to an increase at Site 3 compared with no change or a decrease at Sites 1 and 2 between years (Figure 4; Table 6). Percentage EPT abundance, both including and excluding hydroptilids, at all three sites were significantly greater in 2014 in comparison with 2013 (Figure 4; Table 6). A significant site*year interaction was also evident, caused by a much greater increase at Site 1 between 2013 and 2014 compared with the other two sites (Figure 4; Table 6). This difference was largely driven by an increase in the abundance of the algal-peircing hydroptilid *O. albiceps*, which increased dramatically between years at this site (from 0.2% of total abundance in 2013, to 20% of total abundance in 2014).

The greatest change in MCI scores from 2013 to 2014 occurred at Site 2, where the site was downgraded from 'fair' to 'poor' with regards to the Stark & Maxted (2007a) water quality categories – although this difference was not statistically significant (Figure 4; Table 6). The MCI scores at Sites 1 and 3 remained largely unchanged between years, and remained within the 'poor' category (Figure 4). While there were no statistically significant changes in QMCI score over time or significant site*year interactions (Table 6), QMCI scores did decrease at both Sites 1 and 2 from 2013 to 2014 such that they were downgraded from 'fair' to 'poor' with regards to the Stark & Maxted (2007a) water quality categories (Figure 4). Similarly, the QMCI score at Site 3 decreased from 2013 to 2014, but still remained within the 'poor' quality rating (Figure 4). UCI scores increased at all sites between years; however, these differences were not significant (Figure 4; Table 6). In contrast, the QUCI scores at all sites decreased significantly between 2013 and 2014 (Figure 4; Table 6).

3.3 Comparison of Results with Receiving Environment Objectives and Other Guidelines

When comparing the habitat-based surface water quality objectives from Consent CRC120223 (Table 7), fine sediment cover and macrophyte cover were exceeded in both 2013 and 2014 at Site 3. Given that this site is a soft-bottomed site dominated by macrophytes it was not surprising that it exceeded these Consent objectives. Overall, macrophyte cover was less in 2014 than in 2013 for all sites. For Site 1 and

TABLE 7 Comparison of 2013 and 2014 results with the surface water quality objectives from Consent CRC120223. Parameters that breach the objectives are coloured black (2013) and red (2014). Total macrophyte cover includes both emergent and submerged macrophytes.

Parameter	Surface water quality objectives from Consent CRC120223	Site 1: DS of Ballintines Drain 2013	Site 1: DS of Ballintines Drain 2014	Site 2: DS of Hendersons Rd Drain 2013	Site 2: DS of Hendersons Rd Drain 2014	Site 3: DS of Dunbars Drain 2013	Site 3: DS of Dunbars Drain 2014
Fine sediment cover	Maximum of 30%	15	15	14	15	**100**	**100**
Total macrophyte cover	Maximum of 30%	**55**	8	**31**	15	**79**	**65**
Filamentous algae cover (>20 mm long)	Maximum of 20%	0	0	0	0	0	0
Quantitative macroinvertebrate community index (QMCI)	Minimum score of 4–5	4.10	**3.45**	4.31	**3.90**	**3.35**	**3.03**

2, this reduction in macrophyte cover was sufficient to drop them below the threshold level of 30%. No filamentous alga (> 20 mm long) was recorded from any of the sites in either 2013 or 2014 (Table 7).

The only invertebrate-based metric in the surface water quality objectives from Consent CRC120223 is the QMCI score. As with the 2013 monitoring results, Site 3 failed to meet the receiving environment target value for QMCI (minimum 4–5) in 2014 (Table 7). The QMCI scores of both Sites 1 and 2 had dropped in 2014 such that they also did not meet the receiving environment target value of a minimum of 4–5 (Table 7).

When comparing each year's results (2013 & 2014) with the latest version of selected 'Freshwater Outcomes for Canterbury Rivers' for Banks Peninsula rivers from the proposed Canterbury Land and Water Regional Plan (pLWRP) (Environment Canterbury, 2014), it is apparent that all three sites would have failed to meet the minimum QMCI score of 5 in both years (Table 8). Site 3 would also have exceeded the 20% maximum cover of fine sediment, in both 2013 and 2014 (Table 8). As no filamentous alga was recorded from any site in either year, all sites would have been below the 20% maximum cover target value for filamentous algae (> 20 mm long) (Table 8).

TABLE 8 Comparison of 2013 and 2014 results with selected 'Freshwater Outcomes for Canterbury Rivers' for Banks Peninsula rivers from the proposed Canterbury Land and Water Regional Plan – Decisions version (18 January 2014) (Environment Canterbury, 2014). Parameters that would breach the proposed limits are coloured black (2013) and red (2014).

Parameter	Proposed Canterbury Land & Water Regional Plan – Decisions Version (18 January 2014)	Site 1: DS of Ballintines Drain 2013	Site 1: DS of Ballintines Drain 2014	Site 2: DS of Hendersons Rd Drain 2013	Site 2: DS of Hendersons Rd Drain 2014	Site 3: DS of Dunbars Drain 2013	Site 3: DS of Dunbars Drain 2014
Fine sediment (<2 mm diameter)	Maximum cover of 20%	15	15	14	15	**100**	**100**
Filamentous algae (>20 mm long)	Maximum cover of 20%	0	0	0	0	0	0
Quantitative macroinvertebrate community index (QMCI)	Minimum score of 5	**4.10**	**3.45**	**4.31**	**3.90**	**3.35**	**3.03**

4 DISCUSSION

4.1 Habitat

The heavily silted, run habitat present at Site 3 is typical of that found throughout the majority of Cashmere Stream. Although Cashmere Stream's upper reaches were historically dug out as a drainage channel, years of sediment inputs to the river as a result of historic forest clearance, hill erosion and land development means that there is now more fine sediment present within the system than what would have occurred naturally. For example, it is estimated that almost 30% of Cashmere Stream has layers of fine sediment of between 0.1–0.3 m deep (McMurtrie & James, 2013). This same study also identified many of the channelised tributaries (e.g., Dunbars Drain, Hendersons Rd Drain, Milns Drain) as having high levels of benthic sediment (McMurtrie & James, 2013). In contrast, the larger mineral substrate-dominated benthos at the other two sites (Sites 1 and 2) is atypical of the majority of Cashmere Stream (McMurtrie & James, 2013). The source of these pebble/gravel substrates is likely to be remnants of the substrate that was naturally present within the system; however, such substrates may also have been added at Site 2 during the construction of the road bridge and footbridge.

The catchment contains a high proportion of rural land use: horticulture and agriculture (sheep, horses, cattle) on the plains, and agriculture (low-density sheep grazing) and conifer forestry on the Port Hills (McMurtrie & James, 2013). As a result of the land clearance needed for the establishment of agriculture within the catchment, there are large stretches of the river (especially the middle reaches) with relatively little stream shading. Consequently, the low levels of canopy shading, coupled with the increased nutrients being received as a result of both organic and inorganic farm fertilisers, yields high macrophyte growth within the channel (McMurtrie & James, 2013). These macrophyte communities are usually dominated by the exotic Canadian pond weed (*Elodea canadensis*); however, patches of native large water milfoil (*Myriophyllum propinquum*) and *Nitella*/*Chara* are also present (McMurtrie & James, 2013). In contrast, some of the highest levels of shading found along the waterway are located in residential areas, with some being the result of native plantings (shrubs and trees) in reserve areas. The majority of shading along the waterway, however, is provided by exotic tree species such as poplars, macrocarpas and/or willows (McMurtrie & James, 2013).

Despite a considerable proportion of the catchment being urban land use, the water quality along the length of Cashmere Stream is considered better than that of more fully urbanised nearby catchments such as the Heathcote and Avon River catchments (McMurtrie & James, 2013). Concentrations of typically urban-derived heavy metals such as copper and zinc are lower in Cashmere Stream than in the nearby Heathcote River during base flow conditions (McMurtrie & James, 2013). As with other spring-fed lowland Canterbury waterways, nutrient levels (nitrogen and phosphorus) are relatively high in Cashmere Stream, although ammonia and phosphorus are still generally lower than that found in the Heathcote River during baseflow conditions (McMurtrie & James, 2013). Suspended sediment, however, has been identified as one of the major stressors on the biological communities of this system (McMurtrie & James, 2013).

Given the sediment-related issues for Cashmere Stream, it was encouraging to see that fine sediment depths at all sites had reduced since 2013, albeit not significantly so. This reduction in fine sediment depth between years at all sites is likely to be associated with the macropyhte removal carried out along the stream. Macrophytes act as a natural filter for fine sediment, trapping it within their roots and stems. Therefore, it is likely that fine sediment is being removed with the macrophytes themselves during stream maintenance. The finding that the substrate index values at the two stony-substrate dominated sites (Sites 1 and 2) had not significantly declined between years is also encouraging, as the maintenance of this habitat type is crucial for the regionally uncommon bluegill bully *Gobiomorphus hubbsi*, which has previously been recorded from Cashmere Stream at Site 2 (James & Taylor, 2010).

The only site to fail any of the habitat-based surface water quality objectives (from Consent CRC120223) in 2014 was Site 3. This site was completely dominated by fine sediment, as per large reaches of Cashmere Stream; therefore, it is unsurprising that it recorded a percentage fine sediment cover value far greater than the Consent objective (maximum fine sediment cover of 30%). Nor is it surprising that this site also exceeded the receiving environment target value for macrophyte cover (maximum total macrophyte cover of 30%), given the predominance of macrophytes along Cashmere Stream between Cashmere Road and Hoon Hay Valley Stream. Although both Sites 1 and 2 exceeded the receiving environment target value for macrophyte cover in 2013, macrophyte cover at both sites had dropped sufficiently in 2014 such that they met the receiving environment target value. This reduction in macrophyte cover at all sites is most likely a result of macrophyte removal practices, rather than any other variable. For example, a reduction in percentrage macrophyte cover from 55% (in 2013) to 8% (in 2014) for Site 1 is far greater than what would be expected through temporal stochastic change. There was a considerable difference in timing of macrophyte removal between 2013 and 2014: in 2012 it was completed in November, while in 2013 it was completed a month later in December (Dale Wilheim, City Care, pers. comm.), meaning there was less time between the completion of macrophyte removal and sampling in 2014 in comparison with 2013. At this time of year (late spring, summer), a loss of one month's macropyhte growth can result in a considerable reduction in macrophyte biomass at a given site.

The reduction in fine sediment depth and total water depth, the decrease in macrophyte cover and depth, and the increase in water velocity at all sites can be all also related back to stream maintenance (i.e., removal of macrophytes and associated sediment). As mentioned above, the reduced fine sediment depth could be as a result of the sediment being removed with the macrophytes themselves, while the higher water velocity at all sites in 2014 was likely a result of the reduced macrophyte cover and depth.

4.2 Aquatic Invertebrates

The overall health of the three monitoring sites on Cashmere Stream in 2014, as categorised by their MCI and QMCI scores, were within the 'poor' category. The invertebrate communities of the three sites were dominated by taxa such as the snail *P. antipodarum* and the amphipod crustacean *Paracalliope* that prefer, or are tolerant of, sluggish and soft-bottomed streams that are impacted by agricultural and urban land uses (Suren, 2000). Of the EPT taxa that are associated with clean water (mayflies, stoneflies and caddisflies), only caddisflies were recorded from the three monitoring sites. Seven caddisfly taxa were recorded, with six of these taxa actually considered 'cleanwater' species: *Hudsonema amabile*, *Hydrobiosis parumbripennis*, *Triplectides*, *Psilochorema*, *Oecetis unicolor* and *Polyplectropus* (*Oxyethira albiceps* is a pollution-tolerant caddisfly taxa). These taxa are a subset of the 12 caddisfly taxa that have been recorded from Cashmere Stream since the mid-2000s: *Hudsonema alienum*, *Hudsonema amabile*, *Hydrobiosis parumbripennis*, *Hydrobiosis umbripennis*, *Oecetis unicolor*, *Oeconesus*, *Polyplectropus*, *Psilochorema*, *Triplectides cephalotes*, *Triplectides obsoletus*, *Paraoxyethira* and *Oxyethira albiceps* (McMurtrie & James, 2013). Of the caddisfly taxa that were recorded in 2014, the 'cleanwater' taxa can, nevertheless, tolerate some suspended sediment and can live in soft-bottomed streams provided that there is some suitable habitat to live on (such as submerged woody debris, or macrophytes) (Winterbourn *et al.*, 2006; McMurtrie & James, 2013).

Despite the health of the sites in the study area being categorised as 'poor', Cashmere Stream, in general, is considered the best quality sub-catchment of the Heathchote River (James, 2010). Two notable invertebrate species that are rare in urban or peri-urban waterways in Christchurch but have good populations in Cashmere Stream include freshwater crayfish/kōura and freshwater mussels/kākahi. Cashmere Stream is a hotspot for freshwater crayfish, especially in the middle reaches where the tall earth banks and macrophytes provide suitable habitat (McMurtrie & James, 2013). Freshwater mussels are quite rare in Christchurch's waterways and are declining nationally and globally as a result of land use intensification, pollution, habitat alteration, sedimentation and non-native species introductions (McDowall, 2002; McMurtrie & James, 2013); therefore, this species is considered a noteworthy inhabitant of a peri-urban waterway like Cashmere Stream. In addition, Cashmere Stream has a good diversity of fish species (nine species), with most widely distributed and some limited to specific habitats (e.g., bluegill bully) (McMurtrie & James, 2013).

The macroinvertebrate communities of the three sites in 2014 were all dominated by the snail *P. antipodarum* (at least 40% total abundance at each site). The amphipod crustacean *Paracalliope* was also quite abundant at most sites. One of the biggest among-site differences in communities in 2014 was the significantly greater relative abundance (%) of EPT taxa at Site 1, relative to Site 3. This was largely due to the greater relative abundance of the hydroptilid caddisfly *O. albiceps* at Site 1. The greater relative abundance of *O. albiceps* at this site may potentially be due, at least partially, to the greater cover of algal mats present at this site, as this species is an algal piercer with a well-known preference for nutrient-enriched and disturbed sites (Quinn, 2000). However, it may also be due to natural spatio-temporal patchiness in this species' distribution. Further information is required to elucidate the exact underlying reason(s) for this difference. Site 1 also contained a significantly greater relative abundance of non-hydroptilid EPT taxa, relative to Site 3; this was largely due to the greater relative abundance of *H. umbripennis* and *Psilochorema* at this site. Although this difference was statistically significant, the site differences were considerably less than that recorded for % EPT abundance including hydroptilids (i.e., all three sites contained low relative abundance (<5.4%) of non-hydroptilid EPT taxa).

There was also a significant difference in QMCI scores between sites in 2014, with Site 2 having a significantly greater score than Site 3, despite all three sites being categorised as 'poor' with respect to organic pollution. This difference is likely attributable to the greater relative abundance of both Sphaeriidae and Ostracoda at the soft-bottomed Site 3, as both these taxa have low MCI tolerance values (thereby indicating that both taxa are quite tolerant of organic pollution). The fact that the snail *P. antipodarum* has a considerably lower MCI tolerance value in soft-bottomed streams (SB: 2.1), in comparison with hard-bottomed streams (HB: 4), is also likely to have had an influence on this site difference, as Site 2 (hard-bottomed) would have received higher QMCI scores for this same species than the soft-bottomed Site 3. The significantly lower UCI score at Site 3, in comparison with the other two sites, reflects that this site contains a macroinvertebrate community tolerant of slow-flowing water conditions associated with soft-bottomed streams, with a high biomass of macrophytes (Suren *et al.*, 1998).

Notwithstanding that this was only the second year of the current monitoring programme, there were some changes in the invertebrate communities at each of the three sites between 2013 and 2014. Taxa richness at all sites increased significantly between years, as did % EPT abundance (both including and excluding hydroptilids). Albeit not significant, there was a reduction in MCI scores at Sites 1 and 2 in 2014 compared with 2013, and a consistent reduction in QMCI scores at all sites from 2013 to 2014, with this latter reduction being associated with an increased relative abundance of pollution-tolerant taxa such as the hydroptilid caddisfly *O. albiceps,* oligochaetes and Sphaeriidae, and a decreased relative abundance of *Paracalliope*. The decreased percentage abundance of *Paracalliope* at all sites between years is likely related to macrophyte removal (i.e., decreased macrophyte depth and coverage at all sites between years), as this taxa is well known to display an affinity for macrophytes (Chapman *et al.*, 2011). The increased percentage abundance of *O. albiceps* at Sites 1 and 2 between years (a significant site*year interaction for Site 1) may be due to the increased coverage of algal mats at these sites between years. However, as mentioned above, it may also be due to other stochastic effects, and without further information it is impossible to accurately determine the exact cause(s).

The finding of a significant site*year interaction for non-hydroptilid EPT taxa richness, due to a single taxa increase at Site 3 between years, highlights the caution that is needed when interpreting the results of analytical statistical comparisons. Notwithstanding that there was a statistically significant increase in non-hydroptilid EPT taxa richness at Site 3, this difference would most likely be considered biologically ambivalent, due to it being a result of such a small number of taxa (i.e., a mean difference of 1 taxa), which would be an artefact of sampling (picking up rare species or those with patchy distributions in the current year) rather than any real change in EPT taxa richness. A similar argument can be applied to the finding of a significant site*year interaction in the relative abundance of non-hydroptilid EPT taxa at Site 1. Despite having significantly increased between years at this site, overall, the three sites contained a low relative abundance of non-hydroptilid EPT taxa in both years (< 5.4%) and fluctuations below that level are just as likely to be an artefact of sampling as opposed to a real community change.

The significant decrease in QUCI scores at all sites from 2013 to 2014, contrasts with the increased trend shown by the UCI scores. This inter-year difference in QUCI scores is likely attributable to the decreased relative abundance of *Paracalliope* at all sites between years, as *Paracalliope* has a much higher UCI tolerance value (0.65) than both *P. antipodarum* (0.023) and *O. albiceps* (0.248). Notwithstanding this temporal change in communities between years, overall, the dominant macroinvertebrate taxa present at each site continues to be non-insect taxa that are largely insensitive to physico-chemical changes over time.

Unsurprisingly, considering the change in the macroinvertebrate communities and the reduction in QMCI scores at all sites between years (as mentioned above, all three sites were categorised as 'poor' with respect to organic pollution in 2014), all three monitoring sites failed to meet the surface water quality objective (from Consent CRC120223) for QMCI (minimum 4–5) in 2014.

There was no control site sampled that is not influenced by either macrophyte removal or stormwater discharges; therefore, it is not possible to determine the exact cause of the overall change in the invertebrate communities between years as it could be attributable to differences in the timing of macrophyte removal, stormwater discharges impacts, or natural temporal variation. The study design does not allow for the clear elucidation of the influence of any of the aforementioned drivers, and as a result, it is impossible to draw firm conclusions from the data. It is possible, however, that macrophyte removal at Site 3 may be having a disproportionate influence (relative to the other two hard-bottomed sites) on the macroinvertebrate communities, as this is the only stable habitat present at this site for invertebrates, while invertebrates at the other two sites still have a stable pebble/gravel habitat to utilise in the absence of macrophytes. Macrophyte removal is known to have a major impact on aquatic invertebrate communities, via the removal of habitat/food for many taxa, the release of suspended sediment and alteration of diurnal oxygen ranges (James, 2011). Given the macrophyte removal was completed one month later in 2014 than in 2013, this represents one month less for the aquatic invertebrate communities to recover in comparison with the 2013 survey. It is also difficult to discuss the potential effects of stormwater discharges in the absence of any water quality (base flow vs rain events) data within the duration of the monitoring period and within the vicinity of the habitat/invertebrate sites.

In addition to this Consent-related monitoring of Cashmere Stream, aquatic ecology surveys of Cashmere Stream have also been previously undertaken as part of the CCC's long-term monitoring programme (James, 2010), and as part of ecological monitoring related to the Aidanfield development (James & Taylor, 2010). Comparison of this year's results (2014) with older invertebrate monitoring data for Cashmere Stream[1] reveals that Site 1 has changed little since 2010, as this year's QMCI score (3.45) was comparable to those recorded in 2010 (QMCI – 3.52) by James (2010). Both scores, in 2010 and 2014, categorise the site as being in the 'poor' quality category. For both Sites 2 and 3, the QMCI scores decreased in 2005 (Table 9). Although the QMCI scores at both sites began to recover from 2005 to 2007, there was another marked reduction in 2008 and 2009. Although there is no data available for these sites from 2010 to 2012 inclusive, the trend from 2009 to 2014 at both sites is contrasting: Site 2 has generally increased since 2009 (with its highest score in 2013), whilst Site 3 has continued to decrease each year (Table 9). Analytical statistics are not needed to show that the Site 3 QMCI scores have dropped by almost two QMCI points over the last decade – decreasing from a score that was almost within the 'good' water quality category in 2004, to a score that is well within the 'poor' category in 2014. Notwithstanding natural temporal variation, this large reduction in QMCI scores at Site 3 since 2004 is more likely due to a specific environmental factor (either natural or anthropogenic).

TABLE 9 Comparison of the current monitoring programme QMCI scores (for 2013 and 2014) at Sites 2 and 3 with scores from previous years' invertebrate monitoring data from or near these sites on Cashmere Stream. Data from 2004–2009 obtained from James & Taylor (2010). The Stark & Maxted (2007a) quality categories are indicated in parenthesis for each site for each year.

Site	Indice	2004	2005	2006	2007	2008	2009	2013	2014
2	QMCI	3.90 (Poor)	3.08 (Poor)	3.56 (Poor)	3.81 (Poor)	3.39 (Poor)	3.25 (Poor)	4.31 (Fair)	3.90 (Poor)
3	QMCI	4.92 (Fair)	4.26 (Fair)	4.39 (Fair)	4.58 (Fair)	3.40 (Poor)	3.57 (Poor)	3.35 (Poor)	3.03 (Poor)

1 Site 1 was surveyed in 2010, using comparable survey methods, as part of the long-term monitoring of the Heathcote River (James, 2010). Two sites on Cashmere Stream situated in close proximity to Sites 2 and 3 were surveyed from 2004–2009, as part of the long-term monitoring of Cashmere Stream for the Aidanfield development, albeit using less intensive survey methods (James & Taylor, 2010).

5 ASSESSMENT OF STORMWATER EFFECTS

Due to the limitations of the current study design, it is impossible to determine if stormwater discharges are having an impact on the receiving environment. As the three sites are essentially 'impact' sites, there is no control/reference site (i.e., a site that is not influenced by stormwater discharges) to compare against to determine if there are any stormwater-mediated temporal or spatial trends. With the limitations of the study design, all that can be taken from the monitoring results is that the 'quality' of the macroinvertebrate communities at the three monitoring sites (as inferred from the QMCI scores) has decreased from 2013 to 2014 (Sites 1 and 2 have each dropped a quality class from 'fair' to 'poor', while Site 3 has further declined within the 'poor' quality category). This reduction is associated with an increased relative abundance of pollution-tolerant taxa such as the hydroptilid caddisfly *O. albiceps*, oligochaetes and Sphaeriidae, and a decreased relative abundance of *Paracalliope*. The macroinvertebrate communities of the three monitoring sites continue to be dominated by non-insect taxa (e.g., snails and amphipods) that are considered tolerant of degraded systems.

The reduction in macrophyte cover at all sites between years is likely to have little to do with stormwater effects. Given that this part of Cashmere Stream undergoes regular channel maintenance (macrophyte and sometimes sediment removal), it is probable that the reduction in macrophyte cover and the related changes to habitat attributes (i.e., a decrease in macrophyte depth, water and soft sediment depth and an increase in water velocity) were due to the fact that macrophyte removal in 2014 was one month closer to the sampling date than in 2013. In turn, given that marophtyes represent the only stable habitat in this primarily soft-bottomed system, it is also likely that the periodic macrophyte removal is having an impact on the macroinvertebrate communities, and these effects could potentially override any of the more subtle effects of stormwater discharges under the current study design.

6 RECOMMENDATIONS

The following are a number of recommendations that may help to define future monitoring. They are not in any specific order of preference.

- To reduce natural temporal variation as much as possible, it is recommended that future sampling be carried out at the same time of year (early February). However, to minimise the 'noise' caused by yearly differences in the timing of invertebrate sampling relative to the completion of macrophyte removal, it is recommended that macrophyte removal be completed by a set date every year, if possible. This will help reduce macrophyte removal-mediated temporal 'noise' between years in future monitoring results. Otherwise, comparisons between years are meaningless if communities are potentially at a different stage of recovery at the time of sampling each year.
- Notwithstanding that EPT taxa are generally regarded as 'clean-water' taxa and provide a good indication as to the health of a particular site, the exception to this are the hydroptilid caddisflies (e.g., Trichoptera: Hydroptilidae: *Oxyethira, Paroxyethira*), which are algal piercers and often found in high numbers in nutrient-enriched waters and degraded with high algal content. EPT metrics in this report are presented with and without these taxa; however, last year's report only presented these metrics with these taxa included. It is recommended that in future monitoring reports, EPT richness and % EPT abundance should be presented with and without these taxa. This will ensure that readers do not incorrectly interpret the EPT metrics with respect to stream health.
- While total abundance, UCI and QUCI were required to be included and analysed in this report, it is recommended that they are not included in future analysis and reporting (notwithstanding the

recommendations below regarding improvements to the study design). Abundance (i.e., number of individuals per sample) is not a relevant metric to statistically compare for semi-quantiative sampling such as kicknetting, as it cannot be sufficiently standardised. While UCI and QUCI are useful biotic metrics they appear to more closely reflect habitat-mediated changes and so are unlikely to reflect water quality-based changes.

» The greatest limitation of this study (in relation to achieving its reporting objectives) is its design, including site selection, sample replication, and lack of supporting water quality data.
  - Without adequate control sites, future monitoring rounds will, like this year's data, also fail to elucidate whether or not stormwater discharges (or any other activities) are having an affect on the aquatic ecology of the receiving environment. For any monitoring programme that is wanting to compare change between sites over time, control sites are needed to quantify the habitat and invertebrate community data (and their natural temporal variation) in the absence of key anthropogenic drivers (e.g., macrophyte removal or stormwater discharges), so as to provide a reference/baseline against which to compare community change at the 'impact' sites (i.e., downstream of the stormwater discharges). Without such a design (i.e., isolating individual factors that are influencing the communities), both descriptive and analytical statistics will be unable to determine if the spatial and temporal changes in the data are due to physico-chemical-mediated changes or natural stochasticity.
  - While a number of control reaches have been summarised below as options, the pros and cons of each of these reaches would need to be carefully considered against the goals of the monitoring programme in order to select the most suitable control sites.
    - **Control reach 1:** From Milns Drain confluence upstream to the confluence of Hoon Hay Valley Stream with Cashmere Stream – this reach undergoes periodic macrophyte removal and is subject to the same high inputs of sediment derived mainly from the upstream hill tributaries as the three current monitoring sites (James & McMurtrie, 2009), but is located upstream of the main stormwater discharges from the southwest Christchurch area (McMurtrie & James, 2013).
    - **Control reach 2**: Located from the confluence of Hoon Hay Valley Stream/Farm Drain upstream to Bunz/Bowis Drain. This section also undergoes periodic macrophyte removal; however, it is not subject to the same high inputs of sediment as the first reach and does not have any appreciable stormwater discharges from either flat or hilly urban catchments.
    - **Control reach 3**: Location from the confluence of Bunz/Bowis Drain upstream. This section does not undergo regular macrophyte removal and does not have any appreciable stormwater discharges from either flat or hilly urban catchments.
  - If there is a desire to fully elucidate, through statistical analysis, significant differences of invertebrate-based indices between sites over time, then an increase in site sample replication (which is currently only three) may be required to increase statistical power and, thereby, decrease the chance of getting a false result (i.e., a type II error or a false negative).
  - To fully understand whether stormwater discharges are having an impact on the receiving environment, it is important to also have water/sediment quality data (base flow and rain event) to refer to. In the absence of any data, a change in the biotic metrics at an impact site cannot be specifically linked back to stormwater discharges *per se*, if there is no water/sediment quality data to at least gain an appreciation of what the key stormwater constituents are (i.e., heavy metal toxicity, nutrient enrichment or suspended solids).

- As the aquatic invertebrate communities at each of the three monitoring sites are already dominated by taxa that are relatively insensitive to the impacts of stormwater discharges, it will be difficult to detect stormwater discharge effects by solely relying on broad-scale metrics as QMCI. More targeted quantitative sampling to determine any trends in the densities of the more sensitive taxa present (e.g., non-hydroptilid caddisfly taxa) could be more relevant. In addition, given that Cashmere Stream is currently only partly urbanised it may be pertinent to first determine the level of contamination in the environment that is derived from stormwater-related contaminants, via monitoring of heavy metal concentrations (e.g., zinc, lead, copper) in algal, macrophyte, and macroinvertebrate herbivore (e.g., *Potamopyrgus*) tissue. This latter component may be more relevant to undertake as a monitoring programme for this system, given that the invertebrate communites are also affected by other larger-scale issues (suspended sediment from the hill catchments, regular macrophyte removal).
- A more detailed look (i.e., more specific monitoring) at Site 3 is recommended to determine the cause of the reduction in ecological quality at this site over the last ten years.
- As Site 3 was the only one of the three sites that had a substrate completely dominated by silt, it is likely that the invertebrate community at this site is more susceptible to the loss of the only stable substrate present (macrophytes). However, to elucidate this would require a separate (and wider) study assessing the affect of macrophyte removal in Cashmere Stream on aquatic invertebrates, which is presumably outside of the remit of monitoring relating to Consent CRC120223.
- As an aside, considering the historical pressures that Cashmere Stream has been subject to, there is currently very little physical habitat diversity within the majority of the channel. Apart from macrophytes, there are few areas of large mineral substrates available for the invertebrates to inhabit. Similarly, as a consequence of historical land use change in the upper catchment, large woody debris (LWD) is also quite sparse within the system. Essentially, the majority of invertebrates are forced to use macrophytes as physical habitat. Therefore, in this system at least, it appears that the surface water quality objective (from Consent CRC120223) for keeping macrophyte cover below 30% is counter to the actual benefits that macrophytes provide. Not only do macrophytes provide suitable habitat for the invertebrate communities present, they also provide a biological filter by taking up nutrients and trapping sediment before it reaches the Heathcote River. Macrophytes are also a vital component to the life cycle of freshwater crayfish/kōura within the waterway. Considering these benefits, one would have to question whether a more conservative macrophyte cover estimate (i.e., much greater than 30%) would be more relevant for this waterway.

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8 REFERENCES

Biggs, B.J.F. & Kilroy, C. 2000. Stream Periphyton Monitoring Manual. NIWA, Christchurch, New Zealand. 246 p.

Boothroyd, I. & Stark, J.D. 2000. Use of invertebrates in monitoring. In: Winterbourn, M.J. & Collier, K.J. (ed). *New Zealand Stream Invertebrates: Ecology and Implications for Management*. New Zealand Limnological Society, Christchurch. Pp. 344–373.

Chapman, M.A., Lewis, M.H. & Winterbourn, M.J. 2011. *Guide to the Freshwater Crustacea of New Zealand*. New Zealand Freshwater Sciences Society, Christchurch. 188 p.

Environment Canterbury. 2014. Proposed Canterbury land and water regional plan – decisions version (18 January 2014) [http://ecan.govt.nz/our-responsibilities/regional-plans/regional-plans-under-development/lwrp/Pages/plan-decisions-version.aspx – accessed 10 March 2014].

James, A. 2010. Long-term monitoring of aquatic invertebrates in Christchurch's waterways: Heathcote River catchment 2010. EOS Ecology, Christchurch, New Zealand. EOS Ecology Report No: 06064-CCC02-03. 18 p.

James, A. 2011. A review of the ecological effects of macrophyte management in soft-bottomed waterways. EOS Ecology, Christchurch. EOS Ecology Report No. 11019-WRC02-01. 34 p.

James, A. & McMurtrie, S. 2009. Sources of sediment input into Cashmere Stream. EOS Ecology, Christchurch. EOS Ecology Report No. 08031-ENV01-01. 54 p.

James, A. & Taylor, M. 2010. Long-term monitoring of Cashmere Stream: 2009. EOS Ecology, Christchurch. EOS Ecology Report No. 09004-CCC01-01. 33 p.

McDowall, B. 2002 Decline of the Kakahi – identifying cause and effect. Water & Atmosphere 10: 8–9.

McMurtrie, S. & Suren, A. 2008. Field methodology for the Christchurch River Environment Assessment Survey (CREAS). EOS Ecology, Christchurch, New Zealand. EOS Ecology Report No. 05007-CCC02-01. 18 p.

McMurtrie, S. & James, A. 2013. Cashmere Stream – reducing the pressures to improve the state. EOS Ecology, Christchurch. EOS Ecology Report No: 10049-ENV01-01. Environment Canterbury Report No. R13/20. 50 p.

Quinn, J.M. 2000. Effects of pastoral development. In: Winterbourn, M.J. & Collier, K.J. (ed). *New Zealand Stream Invertebrates: Ecology and Implications for Management*. New Zealand Limnological Society, Christchurch. Pp. 208–229.

Stark, J.D. 1985. A macroinvertebrate community index of water quality for stony streams. Taranaki Catchment Commission, Wellington. Water & Soil Miscellaneous Publication No. 87. 53 p.

Stark, J.D. & Maxted, J.R. 2007a. A User Guide for the Macroinvertebrate Community Index. Cawthron Institute, Nelson. Report No. 1166. 66 p.

Stark, J.D. & Maxted, J.R. 2007b. A biotic index for New Zealand's soft-bottomed streams. *New Zealand Journal of Marine and Freshwater Research* 41: 43–61.

Suren, A. 2000. Effects of urbanisation. In: Winterbourn, M.J. & Collier, K.J. (ed). *New Zealand Stream Invertebrates: Ecology and Implications for Management*. New Zealand Limnological Society, Christchurch. Pp. 260–288.

Suren, A., Snelder, T. & Scarsbrook, M. 1998. Urban Stream Habitat Assessment method (USHA). National Institute of Water and Atmospheric Research (NIWA), Christchurch. NIWA Client Report No. CHCH98/60.

Winterbourn, M.J. 1973. A guide to the freshwater mollusca of New Zealand. *Tuatara* 20(3): 141–159.

Winterbourn, M.J., Gregson, K.L.D. & Dolphin, C.H. 2006. *Guide to the Aquatic Insects of New Zealand*. 4th (ed). Entomological Society of New Zealand, Auckland.

9 APPENDICES

9.1 Site Photographs

Site photographs of each of the three monitoring sites, in both 2013 and 2014.