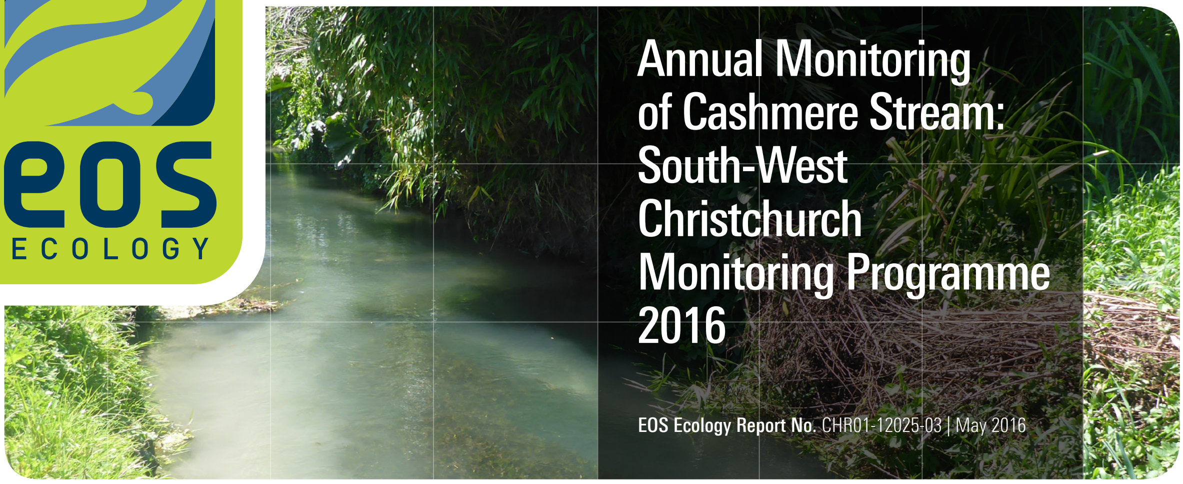

Annual Monitoring of Cashmere Stream: South-West Christchurch Monitoring Programme 2016

EOS Ecology Report No. CHR01-12025-03 | May 2016

AQUATIC SCIENCE &
VISUAL COMMUNICATION

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REPORT

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Christchurch City Council

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EXECUTIVE SUMMARY

Christchurch City Council (CCC) holds a stormwater discharge consent (CRC120223) from Environment Canterbury (ECan) that requires monitoring of aquatic invertebrates and habitat characteristics at three sites (downstream of Ballantines Drain (Site 1), downstream of Hendersons Rd Drain (Site 2), and downstream of Dunbars Drain (Site 3)) within Cashmere Stream. The primary aim is to determine whether stormwater discharges are having negative impacts on the streams' aquatic ecology (as measured by aquatic invertebrates and physical habitat) and determine if the surface water quality objectives of the consent are being met. This report represents the fourth year of monitoring (undertaken on the 10 February 2016), with the previous rounds having been undertaken in February 2013, 2014, and 2015.

The table below compares the relevant 2016 results with the surface water quality objectives from Consent CRC120223 (cells are shaded where the objectives were not met).

Parameter	Surface water quality objectives from Consent CRC120223	SITE 1: DS of Ballantines Drain	SITE 2: DS of Hendersons Rd Drain	SITE 3: DS of Dunbars Drain
		2016	2016	2016
Fine sediment cover	Maximum of 30%	1	5	100
Total macrophyte cover	Maximum of 30%	18	4	97
Filamentous algae cover (>20 mm long)	Maximum of 20%	1	1	1
Quantitative macroinvertebrate community index (QMCI)	Minimum score of 4–5	3.72	3.76	2.44

Since the current monitoring programme started in 2013, instream habitat at the three sites has changed little, with Sites 1 and 2 maintaining a stony, hard-bottomed stream bed and relatively modest macrophyte growth, while the silty, soft-bottomed Site 3 has remained as such and has also consistently had high macrophyte cover. Consequently Site 3 has exceeded the maximum fine sediment and total macrophyte cover water quality objectives of Consent CRC120233 every year since 2013 when the monitoring programme began.

The macroinvertebrate community has also remained relatively comparable over time, being comprised of taxa typical of New Zealand low gradient, lowland streams impacted by agricultural and/or urban development (i.e., dominated by the snail *Potamopyrgus antipodarum,* the amphipod crustacean *Paracalliope fluviatilis,* Ostracoda seed-shrimps, and oligochaete worms). The dominance of such taxa that are tolerant of degraded conditions mean the QMCI scores at all sites were low and in the 'poor' quality class. Consequently, all three sites failed to meet the surface quality objective of a minimum QMCI score of 4–5. Caddisflies were the only member of the more sensitive "cleanwater" EPT taxa present in Cashmere Stream, and then in small relative abundances (1.77% of total invertebrate abundance (including hydroptilids)).

The consent objectives, particularly the total macrophyte cover and QMCI are not necessarily useful for defining the ecological condition or "health" of Cashmere Stream. In a modified system like Cashmere Stream, macrophytes provide habitat and food for macroinvertebrates and provide cover and habitat for fish and freshwater crayfish/kōura. Hence a total macrophyte cover of greater than 30% does not necessarily have any bearing on ecological condition and greater cover may actually provide more habitat area for aquatic biota. Similarly the low QMCI scores at all three monitoring sites should not be taken to necessarily indicate ecological condition is poor given Cashmere Stream retains populations of freshwater crayfish/kōura and freshwater mussels/kākahi, and especially given that the sampling programme does not include any sites that do not receive urban stormwater to provide a reference.

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 2013, 2014, 2015, and 2016. The 2014 and 2015 results are presented in Drinan (2014) and James (2015) respectively. This report covers the 2016 results.

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, 2014, and 2015) 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 were the same as those surveyed on 8 February 2013, 3 February 2014, and 3 February 2015 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 main stem of Cashmere Stream, downstream (DS) of three tributaries: DS of Ballantines 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 the 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.

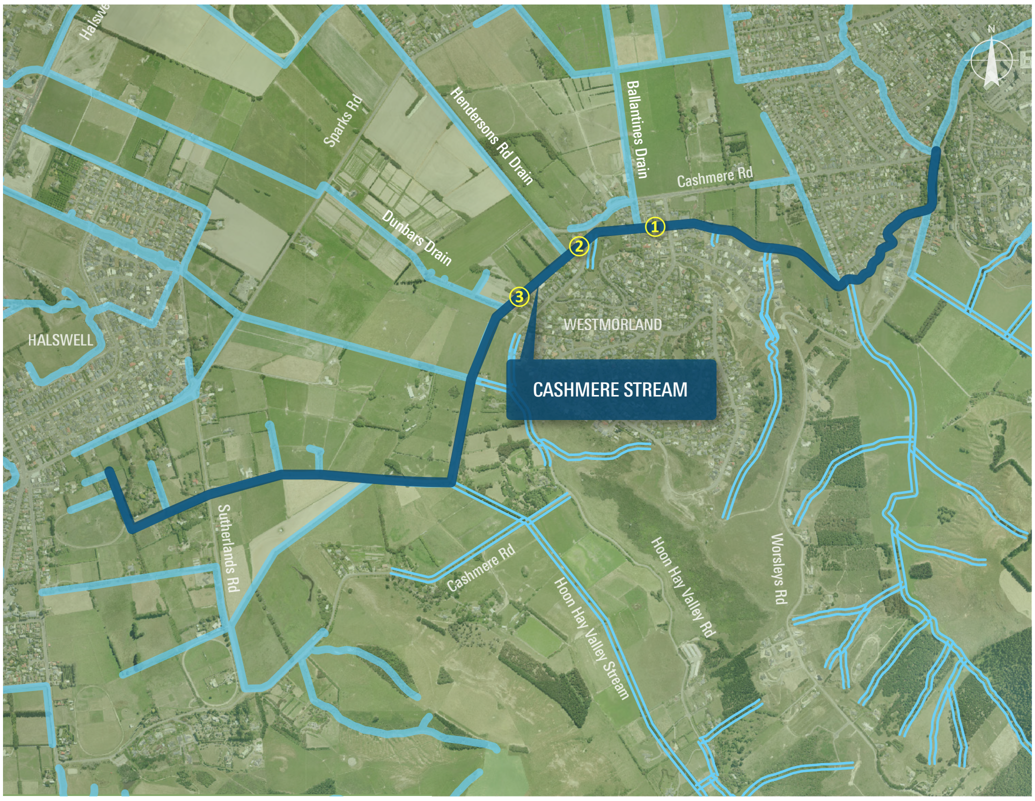

FIGURE 1 Location of the three monitoring sites on Cashmere Stream. Site photographs are provided in the Appendix (Section 8.1).
SITE 1: DS of Ballantines Drain
SITE 2: DS of Hendersons Rd Drain
SITE 3: DS of Dunbars Drain

2.2 Sampling

Following fine weather conditions, EOS Ecology undertook habitat and aquatic invertebrate surveys at each of the three monitoring sites on 10 February 2016. At each site aspects of the instream habitat and aquatic invertebrate community were quantified along three transects across the stream, spaced at 10 m intervals (i.e. at 0, 10 and 20 m).

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/manmade 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 the 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, looking at only 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 percentage 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 stream (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 percentage 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 stream (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 percentage 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 percentage 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/manmade 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 & 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 on either side of the bank within the 20 m site. 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 kick net (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 *et al.*, 2006; Winterbourn, 1973; 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, 2015, and 2016 compared to 2013. These methodological differences were:

- The macrophyte cover assessment was altered in 2014 and subsequent years, compared with 2013. In 2013, macrophytes were assessed over the whole site, while in 2014 and subsequent years 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 and subsequent years, compared with 2013. In 2013, only the 'algal mats' and 'filamentous algae' categories were used, while in 2014 and subsequent years 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 is of 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, and non-metric multidimensional scaling ordination (NMS). Biotic indices calculated were 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.

- NMS is an ordination of data that is often used to examine how communities composed of many different taxa differ between sites. It can graphically describe communities by representing each site as a point (an ordination score) on an x–y plot. The location of each point/site reflects its community composition, as well as its similarity to communities in other sites/points. Thus points situated close together indicate sites with similar macroinvertebrate communities, whereas points with little similarity are situated further away. Habitat variables can also be associated with the different axes, indicating whether the macroinvertebrate communities are responding to habitat differences.

- 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 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 stream systems, although there is a small set 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 & Maxted (2007a) 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, 2007b). 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.
- 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 2016. Data transformations were used (e.g. log10), 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* Holm-Sidak test was used to identify site means that were significantly different. Where the requirements of the parametric tests (i.e. equal variance and normality) could not be achieved with data transformation, the non-parametric Kruskal-Wallis test was used along with the *post hoc* Dunn's test where significantly different site means were observed.

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, 2015, and 2016). For the purposes of considering temporal change, only significant year and site×year interactions are discussed within the text. Although significant site results are 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, thus the average for each transect was used. 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, three data points for channel width, substrate index, and all the invertebrate community indices).

3 RESULTS

3.1 Habitat

3.1.1 Overview of 2016 Results

Overall the habitat of the three Cashmere Stream sites was similar to previous years. Adjacent land use has not changed since 2013, being a mix of rural (farming) and residential use. Riparian vegetation composition was typically comprised of a grass/herb mix at all three sites, with various native and exotic shrubs and trees (Table 1). Site 1 was well shaded, while Site 3 was relatively open. The bridge overhead at Site 2 provided substantial permanent shading of the stream. Site 3 had different habitat to Sites 1 and 2 being 100% run habitat, 100% silt bottomed, having lower water velocities, deeper water, and greater macrophyte cover (Table 1, Figure 2, Table 2). Sites 1 and 2 had greater habitat variability (50% run, 50% riffle) and a coarser bed substrate made up of a mix of gravel, pebble, and cobble sized particles compared to Site 3 (Table 1).

There were statistically significance differences for all the six analysed instream habitat variables in 2016 (Figure 2; Table 2). Apart from channel width, these differences simply highlight the contrasting habitat of Site 3 from that of Sites 1 and 2. Water depth, fine sediment depth, and macrophyte depth were significantly greater at Site 3 while water velocity and substrate index were greater at Sites 1 and 2 (Table 2).

Macrophyte cover was greatest at Site 3 (total cover 97%) and much lower at Site 1 (18%) and Site 2 (4%) (Table 3). This is likely a result of the physical habitat at Site 3 being particularly amenable to macrophyte growth (i.e., open canopy, soft sediments, low water velocities). Apart for the ubiquitous native *Lemna minor* (duckweed) all other macrophytes identified were exotic, with *Elodea canadensis* (Canadian pondweed; 50% cover at Site 3) and *Potamogeton crispus* (curly pondweed; 15% and 45% cover at Site 1 and 3, respectively) the dominant species (Table 3). Algal mats were prominent at Sites 1 and 2, while filamentous algae were found at all sites in small amounts (Table 3). Site 1 was notably different from the other two sites in having a high cover (50%) of bryophytes (mosses/liverworts) attached to the coarse substrate, implying the bed is rarely disturbed by water velocities that cause scouring and movement of gravels.

TABLE 1 Habitat attributes from each of the three monitoring sites on Cashmere Stream for 2016. These attributes were measured over the entire site (i.e. a single site-wide value). TLB = true left bank, TRB = true right bank.

Habitat attributes		SITE 1: DS of Ballantines Drain	SITE 2: DS of Hendersons Rd Drain	SITE 3: DS of Dunbars Drain
Substrate composition (dominant substrate is emboldened)	Man-made (concrete)	1%	1%	0%
	Boulder	10%	1%	0%
	Large cobble	20%	10%	0%
	Small cobble	**39%**	**45%**	0%
	Pebble	29%	18%	0%
	Gravel	0%	20%	0%
	Sand	0%	5%	0%
	Mud/silt/clay	1%	0%	**100%**
Surrounding land use	TLB	70% residential (new) & 30% park/reserve	50% rural with stock (unfenced) & 50% residential (old)	100% rural with stock (fenced)
	TRB	50% residential (new) & 50% park/reserve	50% rural with stock (unfenced) & 50% residential (old)	100% residential (old)
Habitat type (% riffle:run:pool)		50:50:0	50:50:0	0:100:0
Bank material composition		Earth and rock with some concrete on TLB	Earth, rock & concrete (with minor wood)	Earth (with minor rock)
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% (25–50% when including bridges)	<5%
	TRB	>75%	<5% (25–50% when including bridges)	5–25%
Substrate embeddedness		25–50%	25–50%	>75%

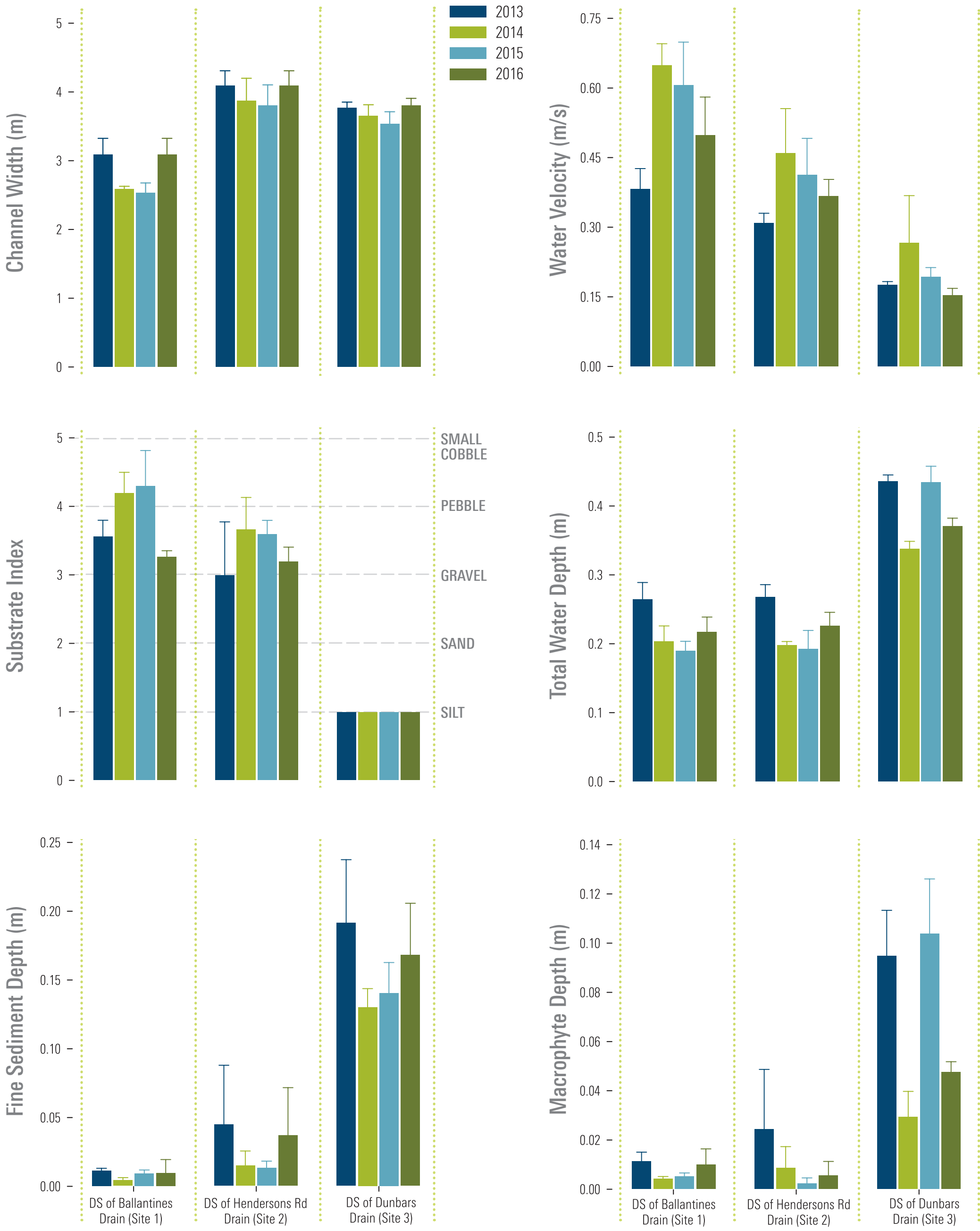

FIGURE 2 Mean (+ 1 standard error) habitat attribute values at each of the three monitoring sites on Cashmere Stream for 2013–2016. Aquatic invertebrate and habitat surveys were undertaken on 8 February 2013, 3 February 2014, 3 February 2015, and 10 February 2016 by EOS Ecology.

TABLE 2 Results of the one-way analysis of variance (ANOVA) or Kruskal-Wallis test on aquatic habitat attributes from 2016 data. The Holm-Sidak *post-hoc* test (ANOVA) or Dunn's test (Kruskal-Wallis) was used to find which site means were significantly different.

Habitat parameter	ANOVA result	Significant site differences
Channel width	$F_{2,8}$= 6.80, p=0.029	2>1. 3=1, 2=3
Water velocity	H= 21.21, p<0.001	1=2>3
Substrate index	$F_{2,8}$= 97.65, p<0.001	1=2>3
Total water depth	$F_{2,8}$= 22.62, p=0.002	3>2=1
Fine sediment depth	$F_{2,8}$= 7.97, p=0.020	3>2=1
Macrophyte depth	$F_{2,8}$= 18.37, p=0.003	3>2=1

TABLE 3 Macrophyte and periphyton attributes from each of the three monitoring sites on Cashmere Stream for 2016. These attributes were measured over the entire site (i.e. a single site-wide value). Total macrophyte cover includes both emergent and submerged macrophytes.

Macrophyte & periphyton attribute	SITE 1: DS of Ballantines 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): 80%	Algae – filaments long (>20 mm long): 1%
	Algae – filaments long (>20 mm long): 1%	Algae – medium mat (0.5–3 mm thick): 30%	*E. canadensis* (Canadian pondweed): 50%
	Algae – filaments short (<20 mm long): 1%	Algae – filaments short (<20 mm long): 1%	*P. crispus* (curly pondweed): 45%
	Moss/liverworts: 50%	Algae – filaments long (>20 mm long): 1%	*Lemna minor* (duckweed): 1%
	Potamogeton crispus (curly pondweed): 15%	Moss/liverworts: 1%	*Glyceria* (sweetgrass): 1%
	Callitriche: 1%	*P. crispus* (curly pondweed): 1%	Woody debris: 1%
	Glyceria (sweetgrass): 1%	*E. canadensis* (Canadian pondweed): 1%	
	Elodea canadensis (Canadian pondweed): 1%	*Callitriche*: 1%	
	Terrestrial roots/vegetation: 10%	*Glyceria* (sweetgrass): 1%	
	Woody debris: 2%	Terrestrial roots/vegetation: 1%	
		Fine detritus: 1%	
		Woody debris: 1%	
Emergent macrophyte cover	0%	1%	0%
Total macrophyte cover†	18%	4%	97%

* 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.

3.1.2 Temporal Change (2013–2016)

Two of the six analysed instream habitat variables were significantly different over the three years (Figure 2; Table 4). Water velocity overall was significantly greater in 2014 in comparison with 2013 and 2016 and in 2015 compared to 2013 (Figure 2; Table 4). Total water depth was significantly greater in 2013 compared to the subsequent three years (Figure 2; Table 4). There was a significant site × year interaction for macrophyte depth with Site 3 having greater macrophyte depth than the other two sites, but only in 2013 and 2015 (Figure 2; Table 4).

TABLE 4 Results of the two-way analysis of variance (ANOVA) (with site and year as main factors) on aquatic habitat attributes from 2013–2016. The Holm-Sidak post-hoc test was used to find which site means were significantly different. n/s = not significant; n/a = not applicable. Note the water velocity data could not meet the normality or equal variance assumptions even after transformation.

Habitat parameter	Site	Year	Site × Year	Comparisons between years
Channel width	$F_{2, 24}$= 33.39, p<0.001	n/s	n/s	n/a
Water velocity	$F_{2, 350}$= 50.51, p<0.001	$F_{3, 350}$= 7.55, p<0.001	n/s	2014>2013=2016, 2015>2013, 2015=2016, 2014=2015
Substrate index	$F_{2, 24}$= 83.50, p<0.001	n/s	n/s	n/a
Total water depth	$F_{2, 24}$= 119.36, p<0.001	$F_{3, 24}$= 8.98, p<0.001	n/s	2013>2014=2015 =2016
Fine sediment depth	$F_{2, 24}$= 42.07, p<0.001	n/s	n/s	n/a
Macrophyte depth	$F_{2, 24}$= 33.33, p<0.001	$F_{2, 24}$= 3.89, p=0.021	$F_{6, 24}$= 2.96, p=0.026	Site 3 > than other sites in 2013 & 2015

3.2 Aquatic Invertebrates

3.2.1 Overview of 2016 Results

A total of 29 invertebrate taxa were recorded from the three aquatic invertebrate and habitat monitoring sites in 2016, with taxa richness per site ranging from 16 to 21. The most diverse groups were caddisflies (Trichoptera: 9 taxa), followed by true flies (Diptera: 8 taxa), molluscs (Mollusca: 4 taxa) and crustaceans (Crustacea: 2 taxa). Mites (Arachnida: Acari), Coleoptera (beetles), roundworms (Nematoda), damselflies (Odonata), worms (Oligochaeta), and flatworms (Platyhelminthes) were each represented by a single taxon.

The three most abundant taxa were molluscs, with the snails *Potamopyrgus antipodarum* (68%) and *Physa acuta* (6%) along with Sphaeriidae pea-clams (5%) together accounting for 79% of all invertebrates captured. These 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 cased caddis *Hudsonema amabile* (1% of total invertebrate abundance). The remaining eight caddisfly taxa included the pollution-tolerant hydroptilid *O. albiceps* (0.25%) and early instar Hydroptilidae (0.06%), as well as the 'cleanwater' caddisfly taxa *Triplectides* (0.63%), *Oecetis* (0.43%), *Hydrobiosis* (0.22%), *Psilochorema* (0.08%), *Polyplectropus* (0.07%) and *Hudsonema alienum* (0.03%) – which combined accounted for 1.77%(including hydroptilids) or 1.46% (excluding hydroptilids) of total invertebrate abundance.

In terms of the five most abundant taxa, the communities of all three sites in 2016 were broadly similar and dominated by mostly non-insect taxa that are common in lowland Canterbury waterways (e.g., the snails *P. antipodarum* and *P. acuta,* Sphaeriidae pea-clams, ostracod crustaceans, oligachaete worms, and *Paracalliope fluviatilis* amphipods (Figure 3)). Species evenness was low at all sites, with the five most abundant taxa at each site accounting for over 89% of total abundance.

In 2016, total abundance (i.e. total number of invertebrate individuals per sample), taxa richness, EPT richness (both including and excluding hydroptilids), percentage EPT abundance (both including and excluding hydroptilids), UCI, and QUCI were statistically similar between the three sites (Figure 4; Table 5). MCI and QMCI showed significant site differences with Site 3 being greater and lower than the other two sites for MCI and QMCI respectively (Figure 4; Table 5). For MCI, Site 3 was in the "fair" 'quality class' of Stark & Maxted (2007b), while the other two sites were "poor" (Figure 4). QMCI indicated all three sites were in the 'poor' quality class (Figure 4). These differences between MCI and QMCI interpretation classes at Site 3 were curious and warrant further explanation. Site 3 has a soft sediment streambed; hence the soft-bottomed MCI-sb was used at this site. Samples from this site had some taxa with relatively high MCI-sb scores (e.g., *Polyplectropus*, *Oecetis* and *Hudsomena* caddisflies and Tanypodinae midge larvae) and low taxa richness, hence the MCI-sb scores for Site 3 samples were higher than the MCI-hb scores from the other two sites. However, once the abundance of each taxon were taken into account with the QMCI, taxa with low MCI-sb scores dominate (especially *P. antipodarum* snails and ostracod crustaceans), while those with relatively high MCI-sb scores were present in low numbers, hence the QMCI score is low. Such results highlight the pitfalls of only relying on a single metric, as in this case referring to only MCI would give an inaccurate assessment of conditions.

FIGURE 3 Photographs of the five most abundant taxa (% relative abundance per site indicated) from the three monitoring sites for 2013–2016. Those taxa designated as 'widespread' were found at all three monitoring sites in that particular survey year.

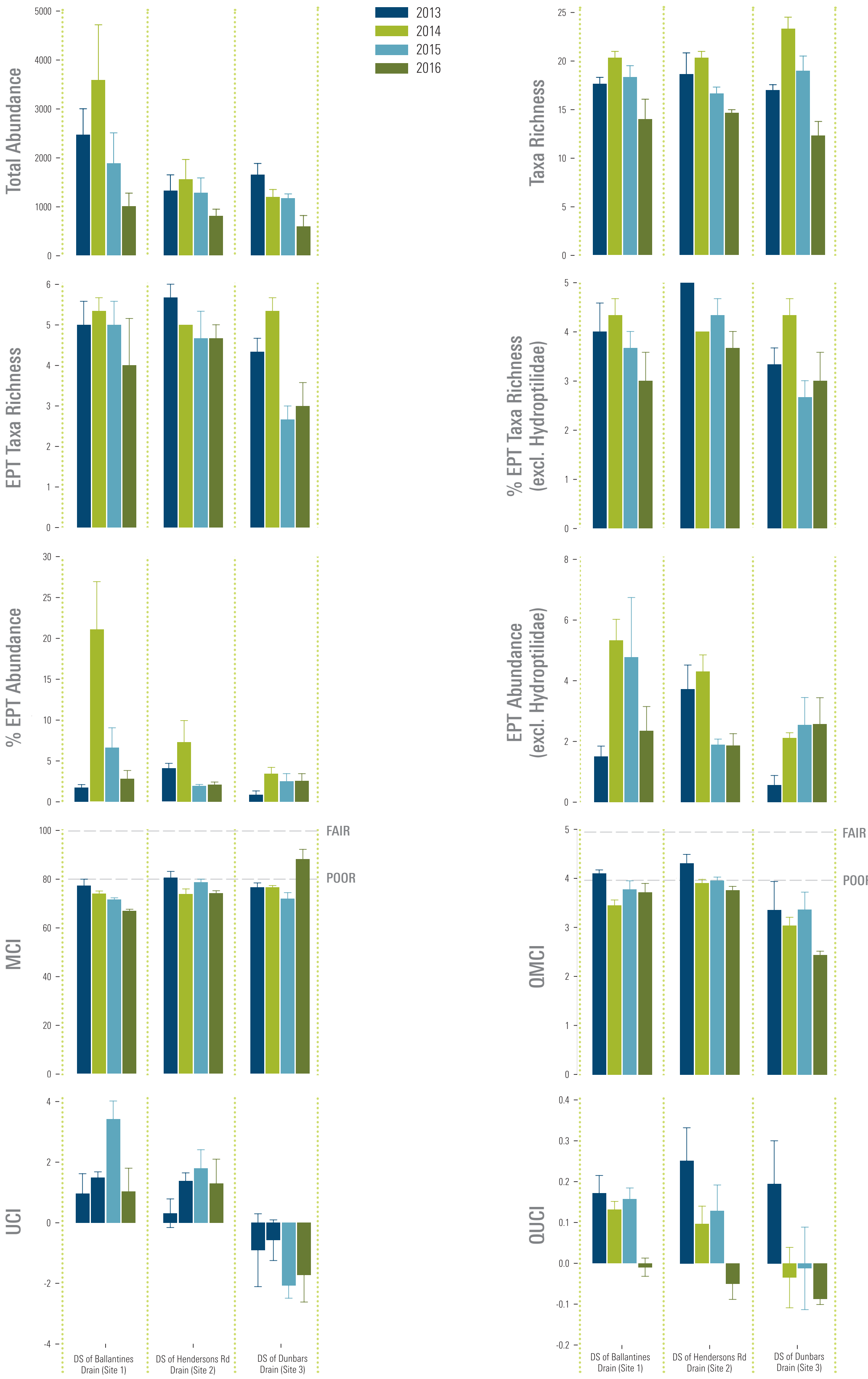

FIGURE 4 Mean (± 1 standard error) community indices at each of the three monitoring sites on Cashmere Stream for 2013–2016. EPT metrics are presented with and without Hydroptilidae, as hydroptilid trichopterans (*Oxyethira* and *Paroxyethira*) are algal piercers that are often abundant in polluted waterways. 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 & Maxted (2007b).

TABLE 5 Results of the one-way analysis of variance (ANOVA) on community indices from 2016. Where the normality and equal variance assumptions of ANOVA could not be met, the Kruskal-Wallis AVOVA on ranks was used. The Holm-Sidak *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	Signifcant site differences
Total abundance	n/s	n/a
Taxa richness	n/s	n/a
EPT taxa richness	n/s	n/a
% EPT abundance	n/s	n/a
EPT taxa richness (excl. Hydroptilidae)*	n/s	n/a
% EPT abundance (excl. Hydroptilidae)*	n/s	n/a
MCI	$F_{2,6}$= 18.8, p=0.003	3>2=1
QMCI	$F_{2,6}$= 38.2, p<0.001	1=2 >3
UCI	n/s	n/a
QUCI	n/s	n/a

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

3.2.2 Temporal Change (2013–2016)

In terms of the five most abundant taxa, the communities of all three sites in 2016 were broadly similar to previous years with the same core taxa dominating (Figure 3). While the order of the most abundant taxa shifts a little between years, *P. antipodarum* typically dominates numerically, being the most abundant taxon for 10 of the 12-site/year combinations (Figure 3). Ostracods are particularly prevalent at the soft-bottomed Site 3, being the second most abundant taxon for the last two years there.

Total abundance and taxa richness were significantly lower in 2016 compared to previous years (Figure 4; Table 6). Taxa richness was also significantly higher in 2014 than the other years. EPT taxa richness (both including and excluding hydroptilid caddisflies) was significantly higher in 2014 compared to 2016, although the other years were not significantly different to either year (apart from 2013 also being greater than 2016 for the 'excluding hydroptilid' data) (Figure 4; Table 6).

Percentage EPT (both including and excluding hydroptilid caddisflies) had a significant site×year interaction. For percentage EPT (including hydroptilids) Site 1 was greater than other sites in 2014; a relationship that was not evident in other years (Figure 4; Table 6). With hydroptilids excluded, 2014 was greater than 2013 at Site 1, although statistically significant differences in this metric are not particularly informative as EPT (excluding hydroptilids) have at all times been a very small percentage of all invertebrates captured at any of the three sites (i.e., <5% combined relative abundance in any one year (Figure 4; Table 6)). MCI was the only other metric with a significant site×year interaction, which was the result of MCI at Site 3 being higher in 2016 than other years (Figure 4; Table 6). UCI was not significantly different between years with Sites 1 and 2 consistently having positive values and Site 3 negative values (Figure 4; Table 6). Both QMCI and QUCI had significant year differences. For QMCI, 2013 was greater than 2016, although each of these years was not significantly different to 2014 and 2015 (Figure 4; Table 6). QUCI was significantly lower in 2016 compared to

2013 and 2015, but not 2014 (Figure 4; Table 6). However, the most notable aspect about QUCI is that 2016 saw Sites 1 and 2 have slightly negative values for the first time since monitoring began in 2013 (Figure 4).

The NMS ordination showed samples from Site 1 and 2 to be separated from those of Site 3 most strongly along Axis 2, and were associated with higher water velocities and a coarser streambed substrate, on both Axis 1 and 2 (Figure 5). Along Axis 2, Site 3 samples were associated with taxa such as acarina (mites), ostracods, and tanypod and *Chironomus* midge larvae, while Sites 1 and 2 were associated with taxa such as orthoclad midge larvae, oligochaete worms, and the caddisflies *Hudsomema, Psilochorema, Hydrobiosis*, and *Oxyethira* (Figure 5). Samples towards the right of Axis 1 (which includes the majority of those from Site 1 and 2) were associated with the snails *P. antipodarum* and *Physa*, while those to the left were associated with ostracods and *Paracalliope* (Figure 5).

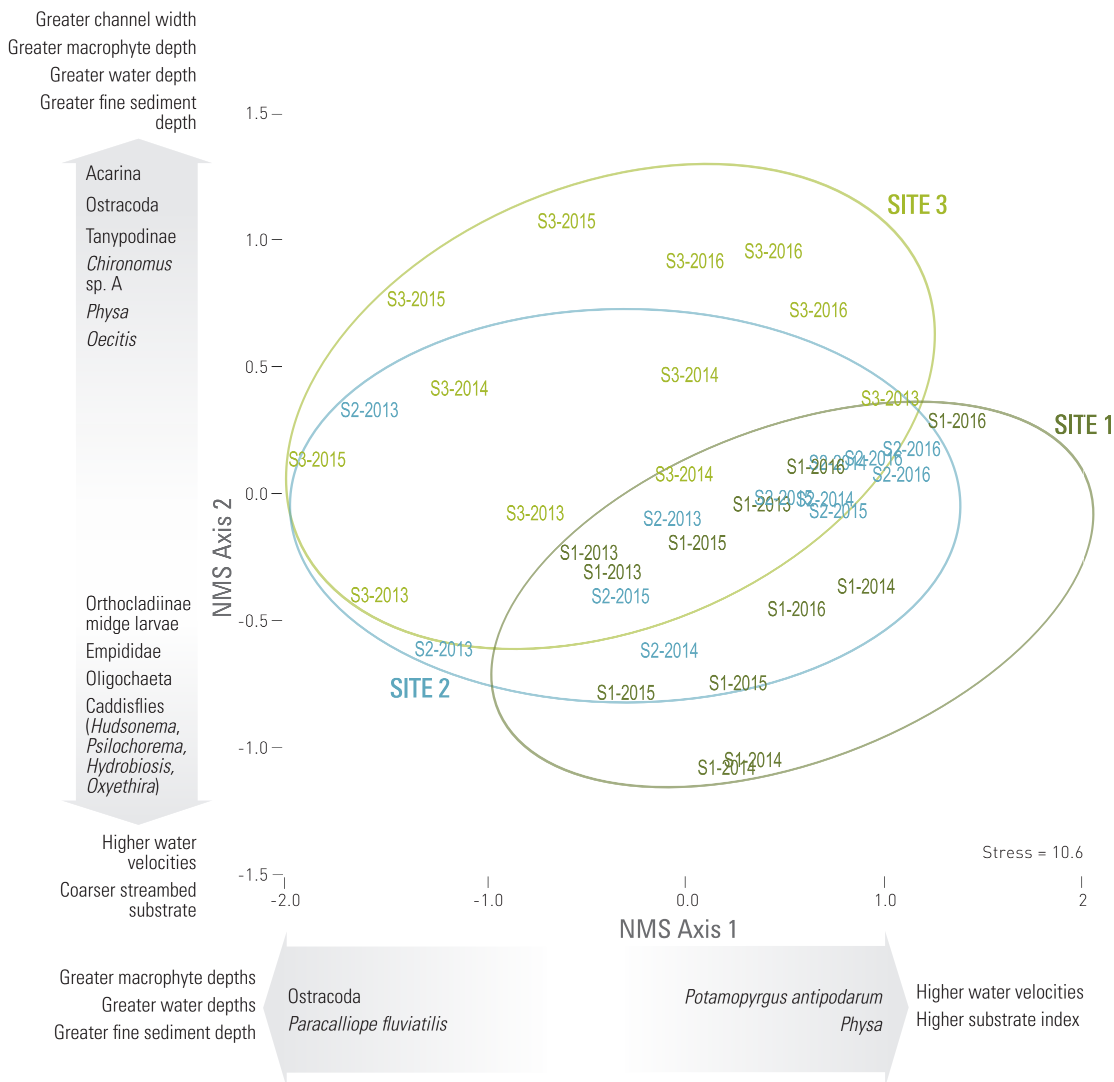

FIGURE 5 **Non-metric multidimensional scaling ordination of benthic macroinvertebrate samples collected at the three sites along Cashmere Stream in 2013–2016:**
S1 = Site 1 (downstream of Ballantines Drain)
S2 = Site 2 (downstream of Hendersons Rd Drain)
S3 = Site 3 (downstream of Dunbars Drain)
Macroinvertebrate taxa and habitat variables that were correlated with each axis are shown. A stress value of 10.6 is indicative of a fair ordination that is useable.

TABLE 6 Results of the two-way analysis of variance (ANOVA) (with site and year as main factors) on community indices from 2013–2016. The Holm-Sidak *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	$F_{2, 24} = 5.1$, $p=0.014$	$F_{3, 24} = 6.6$, $p=0.002$	n/s	2013=2014=2015>2016
Taxa richness	n/s	$F_{3, 24} = 19.0$, $p<0.001$	n/s	2014>2013=2015>2016
EPT taxa richness	$F_{2, 24} = 5.7$, $p=0.010$	$F_{3, 24} = 4.7$, $p=0.010$	n/s	2014>2016, 2013=2014=2015, 2013=2015=2016
% EPT abundance	$F_{2, 24} = 7.5$, $p=0.003$	$F_{3, 24} = 10.3$, $p<0.001$	$F_{6, 24} = 2.9$, $p=0.030$	Site 1 > other sites in 2014
EPT taxa richness (excl. hydrops)*	$F_{2, 24} = 5.7$, $p=0.010$	$F_{3, 24} = 4.5$, $p=0.012$	n/s	2013=2014>2016, 2013=2014=2015, 2015=2016
% EPT abundance (excl. hydrops)*	$F_{2, 24} = 3.8$, $p=0.037$	$F_{3, 24} = 3.5$, $p=0.031$	$F_{6, 24} = 2.6$, $p=0.042$	Site 1: 2014>2013, 2013=2015=2016, 2014=2015=2016
MCI	$F_{2, 24} = 8.7$, $p=0.001$	n/s	$F_{6, 24} = 7.9$, $p<0.001$	Site 3 > other sites in 2016
QMCI	$F_{2, 24} = 18.3$, $p<0.001$	$F_{3, 24} = 3.4$, $p=0.033$	n/s	2013>2016, 2014=2015=2016, 2013=2014=2015
UCI	$F_{2, 24} = 23.0$, $p<0.001$	n/s	n/s	n/a
QUCI	n/s	$F_{3, 24} = 8.9$, $p<0.001$	n/s	2013=2015>2016, 2013>2014, 2014=2016, 2014=2015

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

3.3 Receiving Environment Objectives

Just as they did in 2014 and 2015, in 2016 Sites 1 and 2 meet the surface water quality objectives from Consent CRC120223 with the exception of QMCI (Table 7). Unsurprisingly, the soft-bottomed Site 3 has consistently not met the fine sediment cover objective in any year, nor those for total macrophyte cover or QMCI.

A comparison with selected 'Freshwater Outcomes for Canterbury Rivers' for Banks Peninsula rivers from the Canterbury Land and Water Regional Plan (LWRP), indicates all sites consistently fail to meet the minimum QMCI score from 2013–2016, while Site 3 would also have exceeded the 20% maximum cover of fine sediment for all years (Table 8).

TABLE 7 Comparison of 2013–2016 results with the surface water quality objectives from Consent CRC120223. Parameters that breach the objectives are shaded. Total macrophyte cover includes both emergent and submerged macrophytes

Parameter	Surface water quality objectives from Consent CRC120223	SITE 1: DS of Ballantines Drain				SITE 2: DS of Hendersons Rd Drain				SITE 3: DS of Dunbars Drain			
		2013	2014	2015	2016	2013	2014	2015	2016	2013	2014	2015	2016
Fine sediment cover	Maximum of 30%	15	15	7	1	14	15	8	5	100	100	100	100
Total macrophyte cover	Maximum of 30%	55	8	23	18	31	15	6	4	79	65	97	97
Filamentous algae cover (>20 mm long)	Maximum of 20%	0	0	1	1	0	0	0	1	0	0	0	1
Quantitative macro-invertebrate community index (QMCI)	Minimum score of 4–5	4.10	3.45	3.77	3.72	4.31	3.90	3.95	3.76	3.35	3.03	3.36	2.44

TABLE 8 Comparison of 2013–2016 results with selected 'Freshwater Outcomes for Canterbury Rivers' from Table 1a of the Canterbury Land and Water Regional Plan (Environment Canterbury, 2015) for "Banks Peninsula" class waterways. Parameters that would breach the limits are shaded.

Parameter	Proposed Canterbury Land & Water Regional Plan – Decisions Version (18 January 2014)	SITE 1: DS of Ballantines Drain				SITE 2: DS of Hendersons Rd Drain				SITE 3: DS of Dunbars Drain			
		2013	2014	2015	2016	2013	2014	2015	2016	2013	2014	2015	2016
Fine sediment (<2 mm diameter)	Maximum cover of 20%	15	15	1	1	14	15	5	5	100	100	100	100
Filamentous algae (>20 mm long)	Maximum cover of 20%	0	0	1	1	0	0	0	1	0	0	0	1
Quantitative macro-invertebrate community index (QMCI)	Minimum score of 5	4.10	3.45	3.77	3.72	4.31	3.90	3.95	3.76	3.35	3.03	3.36	2.44

4 DISCUSSION

4.1 Habitat and Macrophytes

There have been no dramatic changes in Cashmere Stream sites over the four years this monitoring programme has been active. The general habitat attributes of the three sites are unchanged with the hard-bottomed Site 1 and Site 2 and the soft-bottomed Site 3 remaining as such. McMurtrie & James (2013) provides a good summary of the overall state and pressures being faced by the Cashmere Stream catchment. As with the majority of Christchurch's waterways, Cashmere Stream has been significantly modified by land use change since European settlement and suspended and deposited sediment are major ecological stressors. Most of the Cashmere Stream main stem has a fine sediment streambed; hence the coarser cobble-pebble-gravel stony beds at Site 1 and 2 are atypical of the majority of Cashmere Stream.

The higher water velocities at Sites 1 and 2, compared to Site 3 are indicative of a higher bed slope creating conditions where fine sediment (either suspended or as bed load) does not settle, hence maintaining a predominantly hard rocky streambed. Site 3 in contrast is a depositional zone, with lower velocities and deeper water creating an environment where macrophytes flourish and trap still more fine sediment within their roots and stems. Consequentially, Site 3 is likely to constantly breach the fine sediment cover objective of Consent CRC120223 (maximum fine sediment cover of 30%). Site 3 was also the only site to breach the total macrophyte cover objective (maximum cover of 30%). The macrophyte community at the monitoring sites was dominated by the exotic *P. crispus* (curly pondweed) and *E. canadensis* (Canadian pondweed), that despite being introduced weedy species, nevertheless provide habitat and food for aquatic invertebrates (including kōura) and cover for fish. Because dense macrophyte growth reduces channel flood capacity, the Cashmere Stream is subjected to a regular macrophyte removal, an activity that will influence macrophyte cover and the macroinvertebrate community (see James, 2011). Over the years of monitoring the timing of macrophyte removal relative to sampling has varied (Table 9).

From 2013–2015 macrophyte removal typically happened one to two months prior to macroinvertebrate sampling, however in 2016 sampling occurred prior to macrophyte removal. Therefore it would be expected that 2016 samples would be the least affected by the disturbance of macrophyte removal. However, it is notable that total macrophyte cover is very similar between 2015 and 2016. The almost complete cover of the streambed by macrophytes at Site 3 in both years would imply macrophytes grow back quickly.

TABLE 9 Macrophyte removal activities in Cashmere Stream relative to macroinvertebrate sampling 2013–2016.

Year	Macrophyte removal	Sampling Date	Approximate Recovery Period
2013	November 2012	8 February 2013	Two months
2014	December 2013	3 February 2014	One month
2015	Late November–Early December 2014	3 February 2015	Two months
2016	Late February 2016	10 February 2016	12+ months

4.2 Aquatic Macroinvertebrates

The aquatic macroinvertebrate communities of the three sites were dominated by taxa typical of sluggish, soft-bottomed streams with abundant macrophyte growth in agricultural and urban catchments in New Zealand (i.e., the snail *P. antipodarum,* the amphipod crustacean *P. fluviatilis,* Ostracoda seed-shrimps, and oligochaete worms). As in previous years, of the cleanwater EPT taxa (mayflies, stoneflies and caddisflies), only caddisflies were recorded from the three monitoring sites. In 2016 nine caddisfly taxa were recorded, with seven of these taxa actually considered 'cleanwater' species: *Hudsonema amabile*, *H. alienum, Hydrobiosis*, *Triplectides*, *Psilochorema*, *Oecetis* and *Polyplectropus* (*O. albiceps* and early instar Hydroptilidae are pollution-tolerant caddisfly taxa). All these taxa are previously known from Cashmere Stream and other Christchurch urban streams, thus these 'cleanwater' caddisfly taxa are able to persist in urban waterways. In 2016, as in previous years, these caddisflies accounted for a relatively minor component of the macroinvertebrate community, contributing no more than 4% of total invertebrate abundance. While Site 3 had an invertebrate community that was separated from that of Sites 1 and 2 in ordination space, this was mostly the result of subtle differences in the relative abundance of dominant taxa rather than any major differences in macroinvertebrate community structure and probably the result of key habitat differences (i.e., the cobble-pebble substratum, faster velocities, and fewer macrophytes at Site 1 and 2, and the fine sediment substratum and abundant macrophytes at Site 3).

Of the macroinvertebrate community indices, invertebrate abundance and taxa richness were lower at all sites in 2016 compared to previous years. It is unclear whether this is natural variation or the result of some event in Cashmere Stream. The 2017 sampling will indicate if these metrics are in fact showing a downward trend of if it is just natural variation. Given EPT taxa are numerically a minor part of the macroinvertebrate community EPT-based metrics are unsurprisingly relatively low, with EPT richness being in the 3–5 taxa range and % EPT abundance generally being less than 5% for all sites and sampling years. Apart from Site 3 being "fair" in 2016, all sites fall within the "poor" quality class for MCI in 2014–2016. For QMCI, apart for Sites 1 and 2 being "fair" in 2013, all sites and years have been within the "poor" quality class. Additionally, the soft-bottomed Site 3 has consistently had lower QMCI scores in all years. The low QMCI scores have meant all sites do not meet the QMCI surface water quality objective of a minimum score of 4–5 (from Consent CRC120223) in 2014, 2015, or 2016. With only four years of data it is not possible to determine any trends. There is older invertebrate data from Cashmere Stream (James, 2010; James & Taylor, 2010), which is covered in Drinan (2014). The distinct difference in MCI and QMCI interpretation categories at Site 3 in 2016 was notable and indicates the importance of not relying on a single biotic index to assessing impacts on macroinvertebrate communities.

It is important to remember that being categorised as 'poor' by the QMCI does not have a bearing on the ecological value of Cashmere Stream. The macroinvertebrate fauna is dominated by endemic species in a highly modified landscape and Cashmere Stream retains populations of freshwater crayfish/kōura and freshwater mussels/kākahi - two notable mega-invertebrate species that are rare in urban or peri-urban waterways in Christchurch - and 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). Hence it is considered the best quality sub-catchment of the Heathcote River (James, 2010).

4.3 Assessment of Stormwater Effects

The comments regarding study design in James (2015) are still relevant and will not be repeated in full here. In summary the survey design lacks any control or reference sites, hence it is impossible to determine if stormwater discharges are having any impact on Cashmere Stream. As stated in the previous report all these results really indicate are that habitat conditions and macroinvertebrate communities at the three monitoring sites have changed little since 2013.

5 RECOMMENDATIONS

The recommendations given in Drinan (2014) and James (2015) are still relevant and all of these will not be repeated here. However, there are some key recommendations that relate directly to the aims and management outcomes of undertaking such resource consent monitoring. These are outlined below.

1. 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. Alteration to the study design is required if there is a desire to isolate the effects of stormwater discharges from other temporal variability.
2. The site selection of the current monitoring of Cashmere Stream fails to take into account hillside urban developments, which disturb and mobilise erosion-prone loess soils.
3. Some of the surface water quality objectives from Consent CRC120223 are not necessarily in alignment with maintaining ecological health, or directly related to the effects of stormwater discharges. Macrophyte cover in Cashmere Stream is related to maintenance practices and lack of canopy cover rather than stormwater discharges. Additionally, as there is currently little physical habitat diversity within Cashmere Stream, macrophytes provide a major habitat and food source for macroinvertebrates including kōura, provide cover for fish, and trap sediment that is otherwise continuously transported along the stream. Thus keeping macrophyte cover below 30% could be counter to the actual benefits that macrophytes provide this system. I would therefore regard macrophyte cover of greater than 30% to be of no ecological concern, and indeed may be better for the ecological health of this stream.
4. The QMCI surface water quality objective from Consent CRC120223, which indicates all three monitoring sites are in "poor" condition, should not be the only parameter used to indicate ecological "health" or value. Cashmere Stream retains populations of kākahi and kōura as well as nine species of fish, while the macroinvertebrate community is comprised of mostly endemic and native species in a heavily modified landscape dominated by exotic biota. Hence the ecological condition of Cashmere Stream is arguably not "poor".

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8 APPENDICES

8.1 Site Photographs

SITE 1	SITE 2	SITE 3
Downstream of Ballantines Drain, looking downstream	Downstream of Hendersons Rd Drain, looking upstream	Downstream of Dunbars Drain, looking downstream

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