Long-Term Monitoring of Cashmere Stream: 2008

Prepared for
Christchurch City Council

Prepared by
EOS Ecology

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CONTENTS

EXECUTIVE SUMMARY

On the 6th August 2004 a discharge of highly turbid water from a broken stopbank in the uncompleted Aidanfield detention basin entered Dunbars Drain. There were concerns at the time by Environment Canterbury that this discharge may have had a detrimental impact on the instream fauna of the receiving environment, namely Cashmere Stream. Consequently the Christchurch City Council commissioned EOS Ecology to undertake yearly monitoring of the instream fauna at sites along Cashmere Stream, both upstream and downstream of the discharge point into Cashmere Stream.

To date, monitoring has been undertaken in four consecutive years (2005, 2006, 2007, and 2008) and has been compared to 2004 data collected prior to the discharge event. The current monitoring programme incorporates five sites; the three original sites that were surveyed prior to the discharge event, an additional site that was included in 2005, and an additional site that was included in 2006. This report documents the results of the most recent survey, undertaken in 2008, with temporal comparisons to the previous four years of data used to determine any potential effects of ongoing discharges and maintenance practices on Cashmere Stream.

Invertebrates, fish, and habitat features at five sites along Cashmere Stream were surveyed on the 7th and the 12th March 2008. In addition, detailed channel profiles were recorded at the upstream end of each site. The invertebrate community was similar to that previously reported from Cashmere Stream and is typical of a silted lowland stream affected by urban and rural landuses. Forty invertebrate taxa were identified from the five sites, with the greatest representation of taxa within the two-winged fly (Diptera; 11 taxa) and caddisfly (Trichoptera; 10 taxa) orders. The invertebrate community is not dissimilar to that found in other New Zealand lowland urban and peri-urban streams. However, the relative diversity of caddisfly taxa, koura (freshwater crayfish), and kakahi (freshwater mussels) makes Cashmere Stream one of the healthier waterways in South West Christchurch.

Analysis of the invertebrate data over the five years showed few site specific changes between the last survey in 2007 and the current one (2008). However, a potential perturbation could have occurred in 2004 or 2005 with the percentage EPT taxa, particularly the caddisfly *Hudsonema amabile*, declining and then recovering over the following four years. This effect was unlikely to be related to the Aidenfield discharge via Dunbars Drain as it occurred along the length of the river. Many of the other site-specific fluctuations in stream health indices and taxon abundances are probably related to environmental stochasticity, rather than significant impacts of one-off stormwater or sediment input events. However, the input of sediment into Cashmere Stream continues, and the aquatic invertebrate community of this stream, particularly at the stony riffle sites, is likely to be sensitive to increases in soft sediment. Continued monitoring is needed to identify any insidious detrimental effects of sediment build-up on the invertebrate community.

The 2008 fish survey was characterised by a degree of recovery in fish numbers and diversity from the relatively low results reported in 2007. This recovery was consistent with the improvement in fish cover and slight physical changes in the monitored habitats, rather than trends consistent with sedimentation. Of interest was the identification of a single torrentfish from the most downstream site (Site 1), the first record of this species from a Christchurch river. The bluegill bully and freshwater crayfish (koura) are regarded as sensitive to increased substrate sedimentation and embeddedness. Yet, there is no indication of a decline in bluegill bullies, and koura numbers continue to remain consistent or at one site (Site 3) have actually increased in 2008. To date, we have yet to detect any indication that the Aidenfield discharge (via Dunbars Drain) in particular or any other specific event of sediment is causing deleterious effects to the fish fauna in Cashmere Stream. In contrast it is possible that riparian and instream vegetation management could be having a more significant impacts on fish and koura populations. Closer monitoring or experimental

studies pertaining to both of these factors is needed, if we are to determine their actual impacts on the aquatic fauna.

The future health of Cashmere Stream may be dependent on establishing a holistic approach to sediment control and vegetation management throughout the catchment, rather than concentrating on individual discharges on an ad hoc basis. The first course of action would be to control sediment inputs during low flow conditions, as these would be comparatively easy to source and control compared to sediment inputs during rainfall events, which encompass the entire catchment area.

1 INTRODUCTION

On the 6th August 2004 a discharge of highly turbid water from a broken stopbank in the uncompleted Aidanfield detention basin entered Dunbars Drain. There were concerns at the time by Environment Canterbury (ECan) that this discharge may have had a detrimental impact on the instream fauna of the receiving environment; mainly Cashmere Stream. Dunbars Drain continues to receive stormwater from the Aidanfield development, although at this stage the detention basin and much of the catchment has matured. The Christchurch City Council (CCC) commissioned EOS Ecology to undertake a survey of sites along Cashmere Stream on a yearly basis to elucidate any continuing effects of the Aidanfield discharge on the aquatic ecology of the stream.

A previous survey of three sites along Cashmere Stream was carried out during February 2004, prior to the Aidanfield discharge, as part of a wider Southwest Christchurch Integrated Catchment Management Plan survey (McMurtrie, 2005). These three sites were then re-surveyed in February 2005 (along with an additional site), approximately six months after the Aidanfield discharge event (EOS Ecology, 2005). The third and fourth surveys in 2006 and 2007 included a fifth site and detailed channel profile measurements to better determine changes to sediment depths along the river continuum. This current report, detailing the results from the March 2008 survey, therefore represents the fifth year of data collection, which repeats the surveys undertaken in 2006 and 2007.

2 METHODS

2.1 Site Selection and Study Design

Five sites were surveyed in the current study (Figure 1), including the three original sites from the (McMurtrie, 2005) survey (Sites 1, 2, and 5), plus additional sites from EOS Ecology (2005) (Site 3), and McMurtrie & Taylor (2006) (Site 4), which were primarily chosen to fulfil the consent requirements set by ECan.

Site selection was ultimately controlled by the sites that had been surveyed prior to the discharge event, as part of a wider Southwest Christchurch Integrated Catchment Management Plan survey (McMurtrie, 2005). While these sites were not specifically located to maximise upstream-downstream comparisons for the discharge event, their location, along with the addition of the two other sites, were sufficient to allow for some level of comparison regarding the ongoing effects of Dunbars Drain (e.g., Aidenfield) and other discharges on Cashmere Stream.

Three of the five survey sites were located upstream of Dunbars Drain confluence: one site in the headwaters (Site 5), one site upstream of Milnes Drain confluence (Site 4), and one site directly upstream of Dunbars Drain confluence (Site 3; Table 1). Two sites were located at increasing distances downstream of Dunbars Drain: one site just downstream of Dunbars Drain confluence (Site 2) and one site downstream of Cashmere Road (Site 1; Table 1).

Figure 1 Location of the five survey sites along Cashmere Stream, surveyed on the 7th and 12th March 2008. Further site details are provided in Appendix I (site photographs).

The five sites were surveyed between on the 7th and 12th March 2008, which was at the same time of year as in the previous surveys (EOS Ecology, 2005; McMurtrie, 2005; McMurtrie & Taylor, 2006; Burdon & Taylor, 2007)(Table 2). The sites surveyed were re-sampled at the same location and over the same reach. This was achieved firstly by guidance from GPS waypoints, then referral to site photographs and detailed location descriptions that indicated the location of end-of-site markers in respect to local features. Field surveys were undertaken by the same personnel as in previous years.

2.2 Field Methodology

2.2.1 Aquatic Invertebrates

Three replicate invertebrate samples were collected from each site by disturbing the substrate in an approximate 0.3 x 0.5 m area upstream of a kicknet (with a 0.5 mm mesh size). The samples were collected from the same location as in the previous four surveys. For Sites 1–4 these were situated within 1 m of the downstream end of the electrofishing site, and were collected from across the channel; one from the mid channel and one 0.7 m out from each bank. For Site 5 these were collected mid-channel within a 5 m section immediately downstream of the electrofishing site. The reason for the disparity of sampling at Site 5 was because the channel was too narrow (i.e., less than 1.2 m wide) to collect three samples across the channel.

Each invertebrate sample was kept in a separate container, preserved in the field in 60% isopropyl alcohol, and taken to the laboratory for identification. The contents of each sample were passed through a series of nested sieves (minimum mesh size of 500 µm). The contents of each sieve were then placed in a Bogorov sorting tray (Winterbourn *et al.*, 2006) and all invertebrates counted and identified to the lowest practical level, using a binocular microscope and an assortment of invertebrate identification keys (Winterbourn, 1973; Chapman & Lewis, 1976; Smith, 2001; Winterbourn *et al.*, 2006). Sub-sampling was utilised for particularly large samples and the unsorted fraction scanned for taxa not already identified. Invertebrate counts were converted to percentage abundance values for analysis.

A number of habitat variables were assessed from where each kicknet sample was collected, including substrate composition, water depth, silt depth, macrophyte percentage cover, water velocity, and stream width. Water velocity was not recorded at Site 2 as macrophytes precluded any gauging; consequently water velocity was not used in any data analysis. Substrate composition was determined by assigning the substrate at ten random points within each sample area to one of six substrate classes: silt (< 1 mm); sand (< 1–2 mm); gravel (2–16 mm); pebble (16–64 mm); small cobble (64–128 mm); and large cobble (128–256 mm).

Invertebrate samples were processed in the laboratory utilising the same personnel as in the McMurtrie (2005) study. However, in 2006, 2007, and 2008 sample processing procedures were modified to a more detailed technique. Taxa were counted and identified 'in situ' by processing the contents of each sieve under a microscope. In previous years, taxa were first separated from the contents of each sieve by placing the contents in a white tray and removing taxa by eye, prior to their identification and counting using a microscope. The change in processing technique for the current study should not have made any difference to counts of large invertebrate taxa (which are easily seen by the naked eye), but may have resulted in a more accurate count of smaller taxa such as micro-crustaceans and dipteran larvae, which can sometimes be missed when sorting by eye.

Table 1 General location and description of the five sites surveyed along Cashmere Stream in 2007 compared to previous years (e.g., EOS Ecology (2005), McMurtrie (2005), McMurtrie & Taylor (2006), Burdon and Taylor (2007)). Further site details are provided in Appendix I (site photographs).

Current Site No.	Previous Site No.[1]	Location Description	Habitat type
5[2]	22	Approximately 1.83 km upstream of the discharge point. Approximately 39 m upstream of Sutherlands Road	riffle
4[4]	n/a	Approximately 25 m upstream of the driveway bridge at Milns Drain confluence.	run
3[3]	27A	Approximately 35 m upstream of Dunbars Drain confluence.	run
2[2]	27	Approximately 30 m downstream of Dunbars Drain confluence. Running behind the property at 426 Cashmere Road.	run
1[2]	4	Approximately 0.43 km downstream of the discharge point, immediately downstream of Cashmere Road.	riffle

[1]Site numbering as it appears in EOS Ecology (2005) and McMurtrie (2005).
[2]Surveyed in 2004, 2005, 2006, 2007, and 2008.
[3]Surveyed in 2005, 2006, 2007, and 2008.
[4]Surveyed in 2006, 2007, and 2008

Table 2 Dates that each of the five sites were surveyed over the last five years (current study; EOS Ecology (2005), MCMurtrie (2005), McMurtrie & Taylor 2006, Burdon & Taylor (2007)).

Site No.	2004	2005	2006	2007	2008
5	4th February	24th February	2nd March	26th February	7th March
4	n/a	n/a	28th February	6th March	12th March
3	n/a	24th February	2nd March	6th March	12th March
2	20th February	23rd February	28th February	6th March	7th March
1	4th February	24th February	28th February	26th February	7th March

2.2.2 Fish and Freshwater Crayfish

As in previous years, fish and freshwater crayfish sampling followed invertebrate sampling and was carried out using a Kainga EFM 300 electro-fishing machine; the conventional and appropriate fishing technique for small streams. As in previous years, a setting of 200 Volts was used; the minimum level required to achieve an effective electric field with a current of 300-400 mA. Electro-fishing briefly (approximately 3 seconds) renders fish unconscious to facilitate their capture in nets for identification. A hand-held stopnet was used downstream to capture electro-narcotised fish. Overall conditions for fish capture using electro-fishing were good, because of intermediate water conductivity and high water clarity.

At each site in 2008, as in 2007 and 2006, an estimate of the nature and size of fish populations was approximated using a two-pass fish removal method. Thus, each site was fished twice, with fish captured from the first fishing exercise temporarily removed from the habitat by being retained in buckets. The site was then fished a second time, with all captured fish anaesthetised with 2-phenoxyethanol, before being identified and measured. Upon recovery from anaesthesia, fish were released (unharmed) into their resident habitat. Fishing time (the total amount of time that an electric current is passing through the water) was recorded for each pass.

A number of habitat parameters were visually assessed over each fished reach: substrate embeddedness, substrate composition, riparian bank composition, and the extent of fish cover. The methodology for these parameters is similar to that outlined in the field booklets for the New Zealand Freshwater Fish Database (National Institute of Water and Atmospheric Research), and some further detail is provided below.

- Substrate embeddedness is the degree (expressed as a percentage) to which large substrate particles are buried into fine substrate material (i.e. sand or silt). Thus, cobbles half-buried in surrounding silt are considered to be 50% embedded, while a silty stream bed is considered to be 100% embedded.
- The overall substrate composition (expressed as a percentage of stream bed area) was visually estimated using conventional substrate particle sizes (Jowett, 1993). The fished area of each site was based on the mean width (of three transects) multiplied by the length of the fished channel.
- Fish cover is the amount of refugia provided by aquatic macrophytes, instream debris, bank vegetation, undercut banks, or overhead shade. These components are estimated separately in the field as a percentage of the wetted habitat area, and summed to provide an estimate of total available fish cover.

2.2.3 Channel Profiles

Channel profiles were mapped at the upstream end of each electro-fished section. This was first incorporated into the 2006 study to provide for long-term comparisons in sediment build-up. For each transect, water, macrophyte, and sediment depths were quantified at ten equidistant points across the channel. Three depths were recorded; free-water depth (the depth of water free of vegetation or other material), macrophyte depth (the depth of water within which aquatic or terrestrial plants were growing), and soft sediment depth (the depth to which soft sediment such as silt and sand overlay a harder substrate). Total water depth was derived by summing free-water and macrophyte depths.

2.3 Data Analysis

2.3.1 Invertebrate data

Invertebrate data were summarised by taxa richness, Detrended Correspondence Analysis (DCA) axis scores, and biotic indices. Biotic indices calculated were the number of Ephemeroptera-Plecoptera-Trichoptera taxa (EPT richness), % EPT, and the Urban Community Index (UCI), the Macroinvertebrate Community Index (MCI) and their quantitative equivalents (QUCI and QMCI). Further details pertaining to these can be found in McMurtrie & Taylor (2006).

Data describing the substrate composition at the point of each invertebrate sample 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.

General patterns in invertebrate community composition across the sites in 2008 were investigated using DCA ordination. In addition, temporal trends at three sites (Sites 1, 2, and 5) were investigated over all five years (2004-2008). Two-way ANOVAs using site and year as predictors were performed, however only the temporal (i.e., yearly) changes were investigated for this longer-term data. More detailed site specific changes were investigated at all five sites over the three years that we had complete data sets (2006, 2007, and 2008). Two-way ANOVAs were again conducted and site effects, year effects and their interaction (site*year) were investigated. In these analyses the:

- 'Site' effect would determine if there were any differences in the invertebrate community along the stream continuum. A significant 'Site' effect would indicate a difference in communities at different sites, which would most likely be related to habitat variables.
- 'Year' effect would determine whether there were any differences in the invertebrate communities between the survey dates. A significant 'Year' effect would indicate a change in the communities over time, which could be due to natural fluctuations, or an alterations in habitat or water quality.
- 'Year*Site' effect would determine whether there were any significant differences in the invertebrate community at sites along the stream between the dates sampled. Potentially, a significant interaction effect (Year*Site) could indicate that there had been a change at a particular site reflecting either an increase or reduction in input of a pollutant (e.g., fine sediment).

Previously this data has been used to investigate impacts of the Aidenfield's discharge in 2004 using an upstream/downstream discharge approach (EOS Ecology, 2005; McMurtrie & Taylor, 2006), however these reports concluded no obvious significant effect of this particular discharge event. Given the multiple sources of sediment and stormwater inputs along the stream continuum it is more applicable to investigate changes at each site over time, rather than an upstream-downstream comparison of a specific discharge. To this end the data set from 2004-2008 was used in this current report to investigate general temporal changes in Cashmere Stream over the five years rather than focussing on a single discharge event in 2004.

Before undergoing statistical analysis data was first checked for taxonomic consistency between years. The differing levels of resolution for the identification of certain invertebrate taxa arising from a change in laboratory personnel and methodology from 2005 meant that comparisons between years would not be accurate unless discrepancies were rectified. Several taxa were therefore consolidated to ensure consistency across years in the analysis. The midge taxa *Chironomus* sp., *Chironoumus zealandicus*, and *Tanytarsus* were consolidated into the taxon Chironominae, the midge *Corynoneura* was consolidated into Orthocladiinae, and the pea clam taxa *Pisidium*, *Sphaerium*, and *Musculium* were consolidated into the taxon Shpaeriidae. The continuity of laboratory personnel in between 2006 and 2008 meant that the data used for ANOVAs testing changes between these years required little modification.

All data distributions were checked for normality (which is an assumption of the analysis undertaken) and square root or log transformed where needed to normalise data. Repeated measures analysis (a variant of ANOVA) was deemed to be unnecessary given the length of time between sampling dates.

2.3.2 Fish and freshwater crayfish data

Methods to estimate fish and crayfish populations, which involve multiple electro-fishing passes, were not employed during the earlier surveys (i.e. 2004 and 2005). Rather, the habitats were systematically electro-fished with one pass to assess the fish fauna composition and relative numbers. This method reflected the original objectives of the 2004 fieldwork, and was undertaken as part of the Southwest Christchurch Integrated Catchment Management Plan study (which focused on general distribution patterns). Fish-related fieldwork from 2006 onwards was therefore modified to allow fish populations to be monitored at the five sites over successive years, and involved a two-pass fishing ‘removal method’ (Carle & Strub, 1978). Thus 2006, 2007, and 2008 population estimates, based on the established removal method, are presented, although comparisons with 2004 and 2005 data are represented as a catch per unit effort (CPUE) index (numbers of fish captured per minute of fishing time).

Fishing time at each site was broadly consistent across the survey, although significantly more time was spent fishing at Sites 1 and 2 during the 2006 survey (Table 3). For this reason, representing CPUE as catch for fishing minute partially adjusts the CPUE against uneven fishing effort. Koura (freshwater crayfish) respond well to the electric field created by an electro-fishing machine, and are therefore easily captured. In contrast, koura are very rarely captured using conventional aquatic invertebrate sampling equipment. For this reason koura numbers are best represented on the CPUE plots with those of fish from each site.

In a confined habitat, and assuming consistent fishing pressure, the cumulative catch and the decline in fish numbers allows a statistical estimate to be made of the total population. The fished area and duration of electro-fishing was recorded, so that the population estimate can be expressed as population density estimate, either as fish numbers per square metre, or fish numbers per fishing minute (CPUE).

3 RESULTS

3.1 Invertebrates

3.1.1 Overview

A total of 40 invertebrate taxa were recorded from the five surveyed sites. The most diverse group was the two-winged flies (Diptera; 11 taxa), followed by caddisflies (Trichoptera; 10 taxa), snails and bivalves (Mollusca: 4 taxa), crustaceans (Crustacea: 4 taxa) and true-bugs (Hemiptera: 2 taxa). All other groups (mites, leeches, beetles, hydra, damselflies, nematodes, flatworms, rhabdoceols, and worms) were represented by one taxon.

The most numerically abundant taxa in the survey area were the native snail *P. antipodarum* (30.1% ± 3.7%) and the amphipod *P. fluviatilis* (28.0% ± 5.6%). Other common taxa included seed shrimps (Ostracoda; 8.0% ± 2.4%), orthoclad midges (Chironomidae; Orthocladiinae; 7.6% ± 1.2%) and oligochaete worms (Oligochaeta; 5.6% ± 2.9%). The most common EPT taxa were the caddisflies *Hudsonema amabile* (2.5% ± 0.7%) and *Triplectides obsoletus* (2.1% ± 0.9%). No mayflies (Ephemeroptera) or stoneflies (Plecoptera) were recorded from the samples collected.

The most widespread taxa were the snail *Potamopyrgus antipodarum* and the amphipod *Paracalliope fluviatilis* (found in all 15 samples). Other widespread taxa included the orthoclad midges, oligochaete worms, seed shrimps (14 samples), and the snail *Physella* (13 samples).

There were seven rare taxa which were found in only one kicknet sample each. These included three two-winged fly taxa (Ephydridae, Muscidae, and the tipulid *Hexatomini*), the caddisfly *Triplectides cephalotes*, and two marginal terrestrial species (adult staphylinid beetles and the water-skater *Microvelia*).

There was only one taxon recorded in this study that was absent in the previous four years. This was a two-winged fly (the tipulid *Hexatomini*), which was a rare taxon, found in only one sample during the current survey.

An ordination (DCA) of the invertebrate community data indicated that there was reasonable variation between sites in 2008 (Figure 2). The samples from Sites 4 were associated with communities composed of worms, *Oxyethira* caddisflies, tanypod midges, and pea clams (Sphaeridae). Sites 2 and 3 were similar to each other and more associated with invertebrate communities comprising amphipods (*P. fluviatilis*), the introduced snail *Physella*, and seed shrimps (Ostracoda). Site 5 was distinctly separated from the other sites and more associated with invertebrate communities containing diamesid midges, muscid flies, water-skaters (*Microvelia*), mites, and the ubiquitous snail *P. antipodarum*. Samples from Site 1 were variable, having an invertebrate composition either in common with Site 5 or Site 4. No habitat variables (water depth, macrophyte depth, sediment depth, substrate index) were significantly associated with either of the ordination axes.

3.1.2 Temporal changes over five years (2004–2008)

Temporal changes between 2004 and 2008 at Sites 1, 2, and 5 mainly occurred between 2004 and 2005 with little difference between 2006, 2007, and 2008 for most variables (Table 3, Figure 3). This included taxa richness, EPT richness, and the percentage abundance of *P. antipodarum*, all of which were low in 2004, increased in 2005 at remained at similar levels through to 2008 Table 3, Figure 3). Other variables fluctuated through time and were different in 2008 compared to previous years. MCI scores were low in 2008 and similar to 2005 levels, and QUCI scores in 2008 were similarly low as in 2006 (Figure 3). MCI-sb, QMCI and QMCI-sb scores were all also slightly lower in 2008 but not significantly different from the scores in 2007 (Figure 3). The percentage abundance of *H. amabile* continued to recover from a three-fold decrease in abundance between 2004 and 2005 (Figure 3).

3.1.3 Detailed site specific changes (2006–2008)

Two-way ANOVA examining the changes in the invertebrate fauna at all five sites sampled in the last three years (2006–2008) indicated that many of the invertebrate metrics differed significantly between year, site, and an interaction of both (year*site; Table 4). However, most of these differences were the same as those found in previous years, and in general, patterns in 2008 were relatively similar to 2007 (Table 4, Figure 4). Only four results altered in significance between the 2007 and 2008 data. Site differences for the MCI-sb score and the percentage of orthoclad midges that occurred in the 2007 data set had disappeared when 2008 data was added. In addition the significant year*site interaction for orthoclad midges in 2007 was gone with the 2008 data added, while the year*site interaction for *H. amabile* had changed from non-significant to a significant effect (Table 4). In this case *H. amabile* had increased again at Site 5 in 2008 from a low level in 2007, and at Site 3 had declined markedly in 2008 compared to high levels in 2007 (Figure 4). For both these sites the abundance of *H. amabile* in 2008 was similar to that in 2006.

In 2008 few significant changes from previous years occurred at Site 1. Percentage EPT abundance declined to 2006 levels following a large increase in 2007, but almost all other measured metrics showed no large changes at Site 1 between 2007 and 2008 (Figure 4).

In 2008, taxa richness increased at Site 2, however, this corresponded with a decrease in MCI, QMCI, QMCI-sb and QUCI indices, all indicating a decline in health from levels in 2006 and 2007 (Figure 4). The percentage EPT and EPT richness remained high in 2008, and at similar levels as in 2007.

At Site 3 EPT richness remained at the increased levels observed in 2007, however the percentage EPT declined markedly, returning to 2006 levels from a high level in 2007 (Figure 4). This is most probably driven by a dramatic decline in abundance of the caddisfly *H. amabile*, which also increased in 2007 at Site 3 but declined to low levels again in 2008 (Figure 4).

In contrast, percentage EPT increased at Site 4 in 2008, and this was not driven by changes in *H. amabile* abundance, which remained at similarly low levels as in previous years (Figure 4). In addition, EPT richness declined at this site in 2008 from a higher level in 2007. QMCI-sb scores increased at Site 4 in 2008 to levels similar to that found in 2006, following a decline in 2007.

At Site 5 EPT richness remained similar to levels in 2007 (which was lower than in 2006), while percentage EPT and abundance of the caddisfly *H. amabile* increased to 2006 levels following a decline in 2007. QMCI, QMCI-sb, and QUCI scores remained high at Site 5 in 2008, similar to 2007 levels (Figure 4).

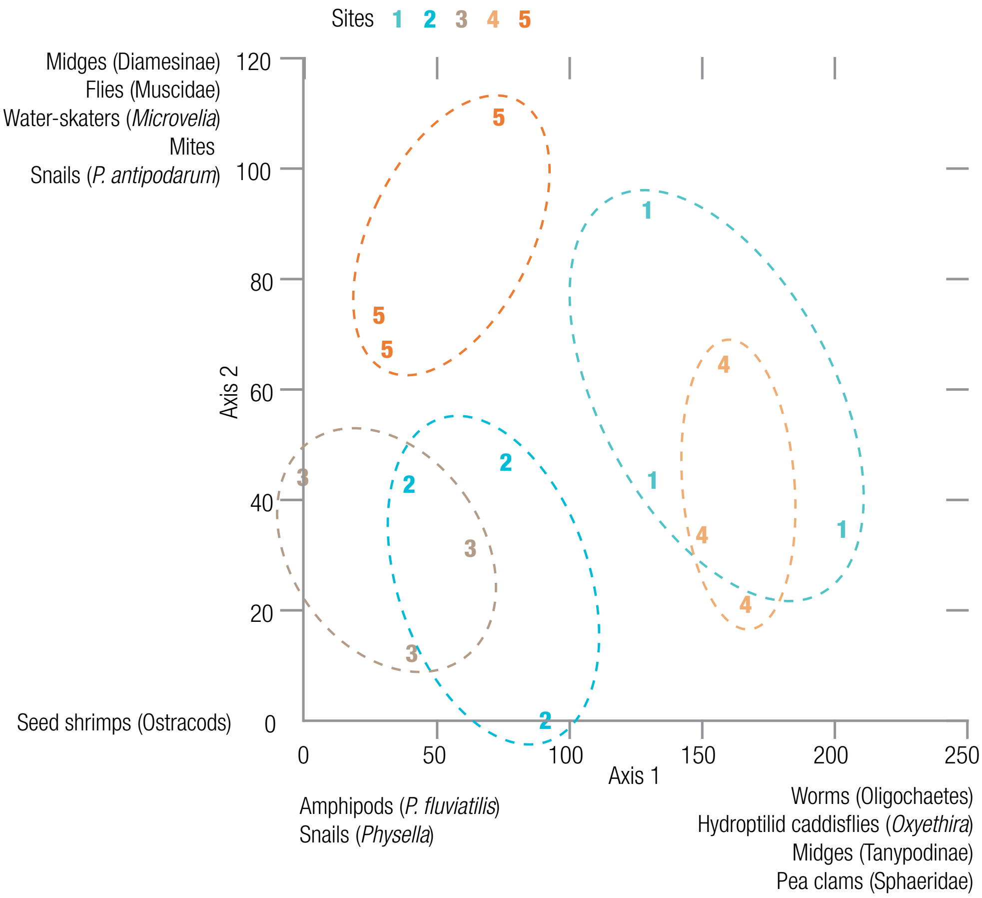

Figure 2 Detrended Correspondence Analysis ordination showing the similarity in invertebrate communities recorded from kicknet samples collected on the 7th and 12th March 2008. Invertebrates associated with each of the axes are shown. No habitat variables were significantly associated with the axes. Ellipses are shown to clarify site groupings.

Table 3 Results of two-way analysis of variance (ANOVA) testing for significant differences of various invertebrate metrics over time and between sites at three of the five sites (Sites 1, 2, and 5) on Cashmere Stream over five years (2004-2008). Only differences between years are presented here, as we want to investigate general temporal trends, in particular whether the 2008 survey is different from the previous years (2004-2007), which were analysed in Taylor & Burdon (2007).

Metric	Year Effect 2004–2007	Direction in 2007	Sig by 2008?	Direction by 2008
Taxa Richness	sig	Low in 2004, increasing	sig	same trend
EPT Richness	n/s	n/a	sig	Low 2004, increasing over time
Percentage EPT	sig	low 2005 and 2006	sig	same trend
MCI	sig	Low 2005	sig	Low 2008
MCI-sb	sig	Low 2005, increasing	sig	same trend (stabilising)
QMCI	sig	Low 2005	sig	same trend (fluctuating)
QMCI-sb	sig	High 2004, low 2006	sig	same trend (fluctuating)
UCI	sig	Low 2005	n/s	
QUCI	sig	Low 2005, 2006	sig	Low 2008
% *P. antipodarum*	sig	Low 2004, increasing	sig	same trend (stabilising)
% Amphipod	sig	Low 2006	sig	same trend
% Oligochaete worms	sig	High 2005	sig	same trend
% *H. amabile*	sig	High 2004, low 2005, increasing	sig	same trend

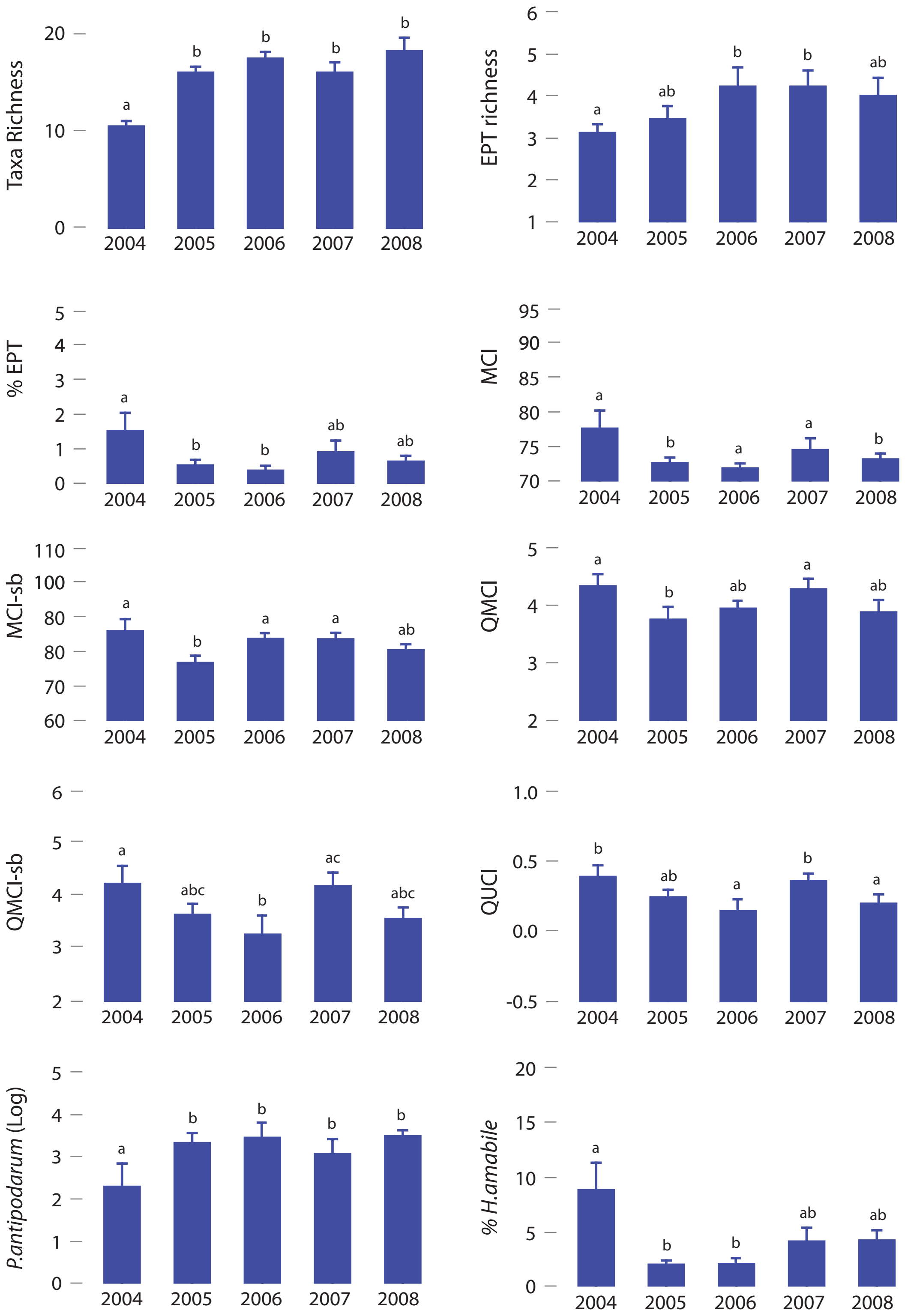

Figure 3 Temporal changes (mean ± SE) of selected variables from three sites (1, 2 and 5) along Cashmere Stream sampled from 2004 to 2008. Letters denote significant differences between years.

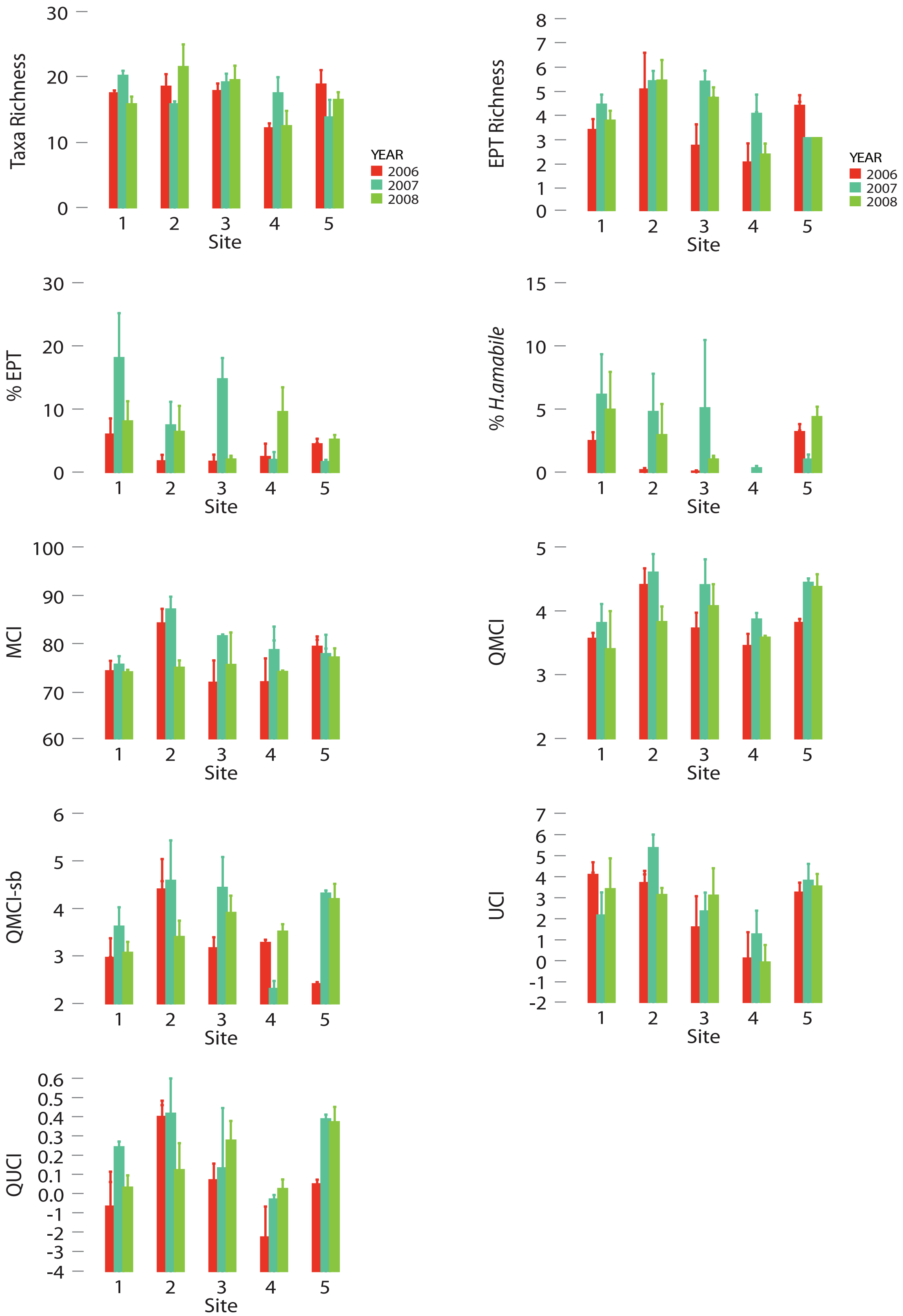

Figure 4 Bar graphs of selected variables (mean ± SE) from five sites along Cashmere Stream that showed significant differences in sites, years, or the interaction of sites (1–5) and years (2006–2008).

Table 4 Results of two-way analysis of variance (ANOVA) testing for significant differences between sites, years, or both sites and years (interaction effect) of various invertebrate metrics at all five sites on Cashmere Stream over three years (2006, 2007, and 2008). n/s = not significant, sig = significant ($p < 0.05$). Arrows indicate direction of change in significance between 2006–2007 data and 2006–2008 data.

Metric	Year	Yearly trend by 2008	Site	Year*Site
Taxa Richness	n/s		sig	sig
EPT Richness	sig	low 2006, 2007=2008	sig	sig
Percentage EPT	sig	low 2006, 2007=2008	sig	sig
MCI	sig	low in 2008	sig	n/s
MCI-sb	n/s		sig (2006–7) → n/s (2006–8)	n/s
UCI	n/s		sig	n/s
QMCI	sig	high in 2007, 2006=2008	sig	n/s
QMCI-sb	sig	low 2006, 2007=2008	sig	sig
QUCI	sig	low 2006, 2007=2008	sig	n/s
% *P. antipodarum*	n/s		sig	sig
% Amphipod	sig	2006<2007<2008	sig	sig
% Ostracods	sig	high 2006, 2007=2008	sig	n/s
% Orthoclad midges	n/s		sig (2006–7) → n/s (2006–8)	sig (2006–7) → n/s (2006–8)
% Oligochaete worms	n/s		sig	n/s
% *Hudsonema amabile*	sig	low 2006, 2007=2008	sig	n/s (2006–7) → sig (2006–8)

3.2 Fish and Freshwater Crayfish

3.2.1 Physical Habitat

Overview

In 2008, as well as in all previous years, Sites 1 and 5 were shallow reaches with a stony substrate and lacking fine sediment and significant amounts of aquatic macrophytes (Figure 5). Site 5 was narrower than the other four sites and, along with Site 1, showed much less variation in mean and maximum water depths (Figure 5). In contrast, Sites 2, 3, and 4 have abundant aquatic macrophyte growth. Sites 1–4 are subject to trimming of bank vegetation and aquatic weed clearance by Council staff in late summer/early autumn (Figure 5). The March 2008 survey occurred before this bank maintenance had been conducted, in contrast to sampling in 2007, which was conducted just after the bank maintenance schedule.

Temporal changes in habitat attributes

Since monitoring began in 2004, substrate fish cover at Site 1 has been abundant, but has increased since 2004 due to an apparent coarsening of the substrate. Visual assessment of substrate embeddedness at this site indicates a reduction in embeddedness, from 30% in 2004 and 2005, to 15-20% in 2006-2007. In addition, a permanent increase in overhead cover at Site 1 occurred in 2005, when a footbridge was constructed over the site (Figure 6). Some stable bank undercuts are now apparent along the true left bank at this site, after some initial erosion was noted post-construction. Plant debris has also accumulated around mid-channel boulders since 2004 at this site, which has the potential to provide refuge for fish and koura. This cover component has been present since monitoring began in 2004, although in recent years more plant debris has been left to accumulate.

In contrast to Site 1, Sites 2 and 3 have almost complete aquatic habitat coverage by instream macrophytes (Figure 6). They also have variable amounts of overhanging grass vegetation cover depending on the time of bank trimming activities compared to the time of the ecological survey. In 2008, our survey was conducted prior to channel maintenance works, and the length of overhanging vegetation was slightly longer than normal at Site 2 and similar to 2005 levels for Site 3 (Figure 6). Over the last two years, Site 3 has developed more floating macrophyte cover that is rooted in the banks. Mean sediment depths vary between 0.1m–0.2m at both Sites 2 and 3.

Monitoring of Site 4 commenced in 2006, and the site has always been heavily tree-shaded, with only patches of aquatic weed cover (Figure 6). After the 2007 survey, in which shading levels were suspected to have been reduced due to channel maintenance, 2008 shading levels were back up to levels originally assessed in 2006 (Figure 6). Sediment depths within Site 4 were less in 2007 than when first recorded in 2006, but then by 2008 were at similar levels as when first measured. This is in contrast to the sediment profile data, which indicates a gradual increase (albeit non-significant) in sediment depth over time (see Section 3.3.2).

Site 5, in the upper reaches of Cashmere Stream, has been monitored since the inception of regular surveying of Cashmere Stream in 2004. Its fish cover attributes appear to be little changed since first fully assessed in 2005 (Figure 6). Like Site 1, Site 5 has a coarse un-embedded substrate which confers value as cover for small fish. In addition, the site is also heavily shaded by an over-hanging shelter belt on the true right bank, although it appears to be trimmed to a variable extent from time to time. This site is subject to stock access on one bank, although bank damage does not appear severe. Mean depths (ca. 0.1 m) were similar to those recorded in 2007.

3.2.2 Fish and freshwater crayfish

2008 is the third consecutive year (since 2006) in which all five monitored sites have been fished with similar fishing effort. Overall, numbers of captured fish were intermediate between the low level in 2007 and the high fish numbers obtained in 2006. The 2006 fish numbers were inflated by high numbers of upland bullies captured that year from the weedy habitats. Nine fish species and one species of freshwater crayfish (koura) have now been identified from the five monitoring sites over the four years 2004-2008 (Figure 7). Of these, koura and eight fish species were identified in 2008. Ordered by abundance, these were the shortfin eel, upland bully, koura, bluegill bully, longfin eel, common bully, inanga, brown trout, and torrentfish. Site 1, the most downstream at the Hendersons Road/Cashmere Road intersection, has always had the highest diversity of any site, with eight species recorded in 2008 (Figure 7). Torrentfish have not previously been identified from the city rivers (Heathcote, Avon, and Styx catchments), thus it was exceptional to find one specimen in at Site 1 in 2008 (Figure 8). In contrast, giant bullies were not indentified from the monitoring sites in 2008, but in the past were recorded in low numbers from Site 3 in 2005 and 2007. Inanga were recorded from four of the five sites in 2008; the best recorded distribution in the monitoring programme for this species (Figure 7). Numbers of common bullies have declined slightly at Sites 1 and 2, both downstream of Dunbars Drain. While a single large brown trout has been caught at Site 2 in earlier years, we failed to do so in 2007 and 2008. Koura were caught at four of the five sites (Sites 1–4), with the highest numbers recorded in 2008.

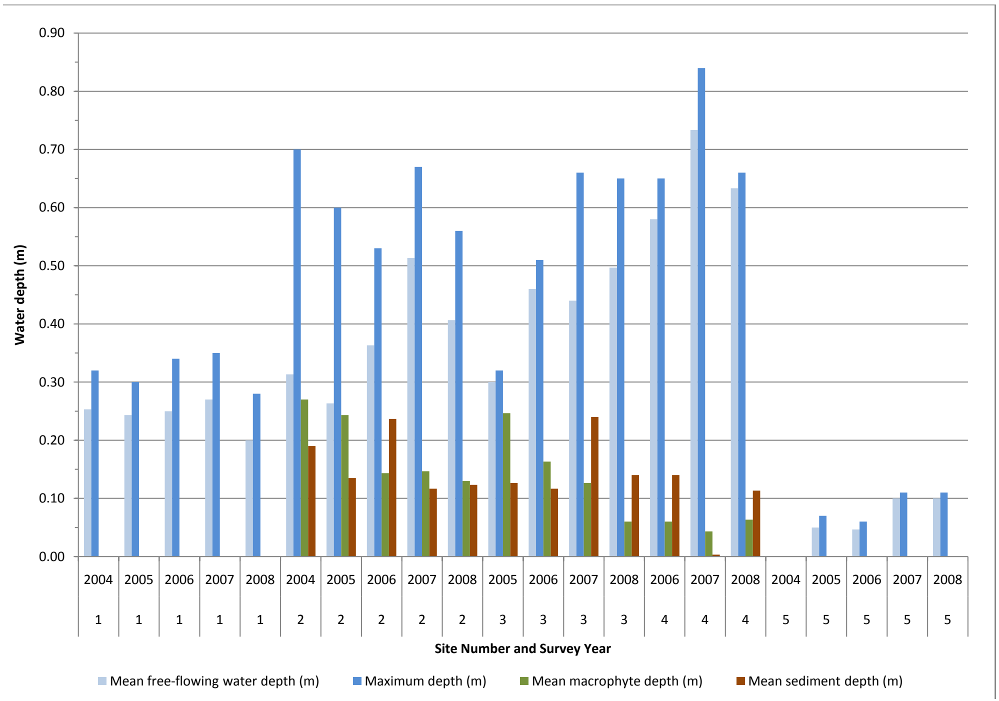

Figure 5 Temporal changes in aquatic habitat (free-flowing water depth, maximum water depth, average macrophyte depth, and average sediment depth) at five sites surveyed along Cashmere Stream between 2004 and 2008.

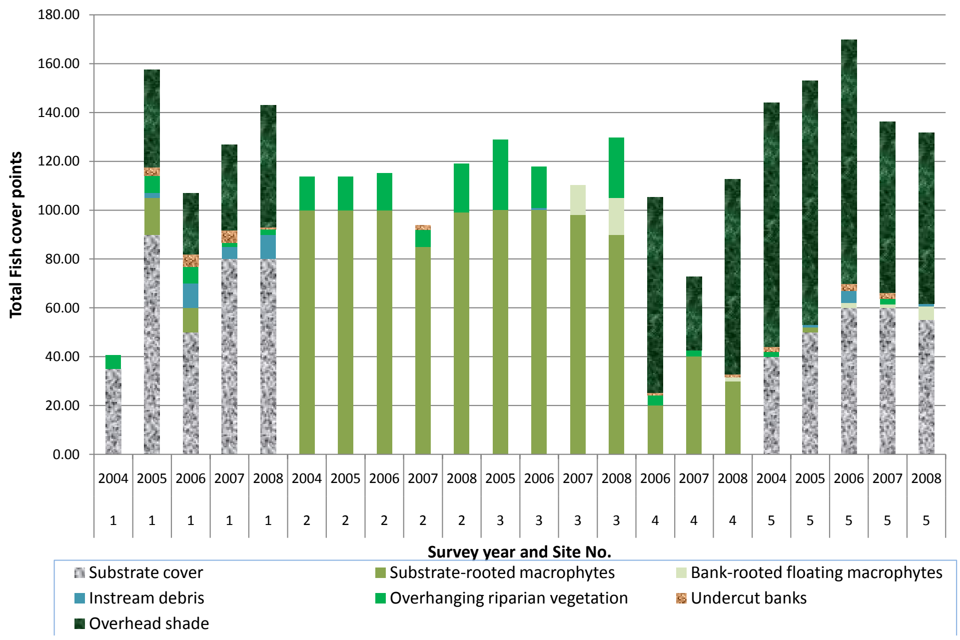

Figure 6 Temporal changes in fish cover ariables at five sites along Cashmere Stream surveyed between 2004 and 2008.

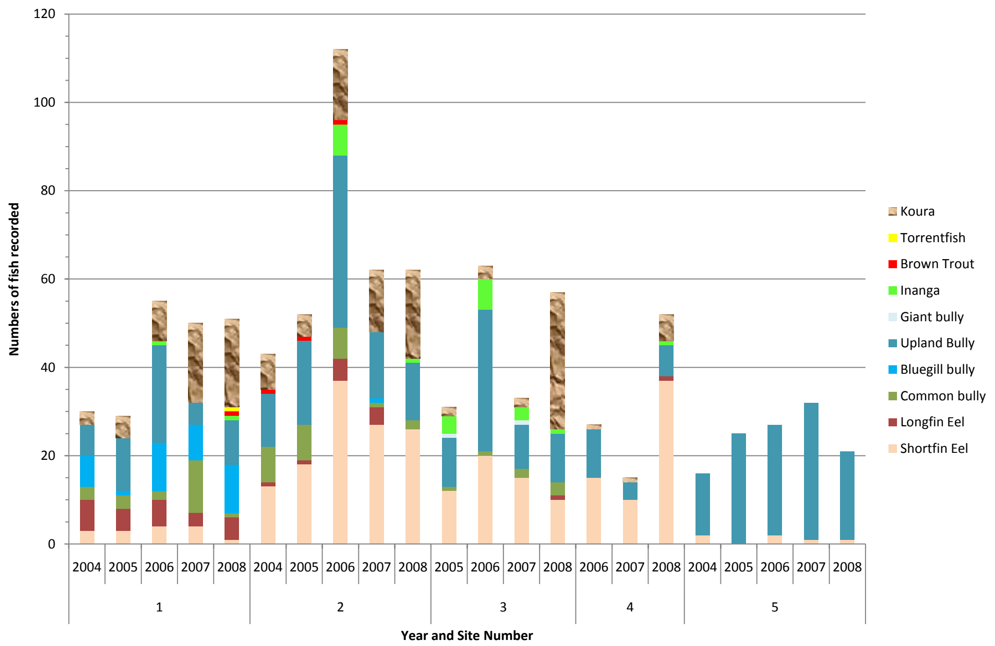

Figure 7. Total number of fish and koura recorded from each of the five sites along Cashmere Stream from 2004–2008

Figure 8. Torrentfish have a body shape adapted for staying in place in fast flowing water and are generally found in braided rivers. This is the first record of a torrent fish within a Christchurch waterway. Photos © Shelley McMurtie.

3.2.3 Temporal Changes in fish and koura

In 2008, the fish catch obtained during the first electro-fishing sweep generally improved markedly from the low levels in 2007 (Figure 9), and was comparable with those obtained prior to the commissioning of the Aidenfield stormwater outfall. Koura numbers caught during the first sweep had also increased substantially at all sites where previously present (Sites 1-4, Figure 9).

Removal method techniques were used to obtain population estimates for the more common species, and the restoration of low fish numbers from the 2007 survey was apparent at Sites 2, 3, and 4 (Figure 10). However at Site 1 total fish population was lower in 2008 than it had been previously, with a decline in the number of upland bullies, common bullies, and shortfin eels. In contrast, numbers of the bluegill bullies (rare in Christchurch Rivers) and koura were increasing at Site 1 (Figure 10).

The fish population at Site 2 was at similar levels to 2007, and was therefore lower than the 2006 levels, but the reduction is attributable to a decline in the number of upland and common bullies compared to 2006 levels (Figure 10). In contrast, numbers of koura and shortfin eels are being maintained or increasing at this site. No longfin eels were caught from this site in 2008, and only one inanga. Given the low abundance of these two species the results are probably inconclusive. A very large longfin eel was known to reside at this site in the past, but has not been seen in recent years. The loss of this large individual has caused a reduction in the mean size of the measured longfin eels, but the loss of this one fish from the dataset has not caused a statistically significant decrease in mean length over all sites, because the variation in eel size in any one year is quite substantial (Table 5).

Upstream of Dunbars Drain, numbers of koura and shortfin eels increased at both of the two deep-water sites (Sites 3 and 4 Figure 10). In contrast, there has been a decrease in the population of upland bullies. Low numbers of inanga are occasionally identified from these sites, and 2008 specimens were mature, containing either eggs or milt, as inanga spawn at this time of year. Site 5 has a fish fauna composing just two species, upland bullies and low numbers of shortfin eels. There has been little change in the estimated fish population over time at this site (Figure 10).

In 2008, the mean length of measured shortfin eels was slightly greater than in 2007, although the length increase (4 mm) was not significant (Table 5). There would still appear to be a marked decrease in shortfin eel size since monitoring began in 2004, and the reduction in shortfin eel size is restricted to the loss of small number of adult eels from the two most upstream sites, specifically Sites 4 and 5 in the last two years. However, the trend is based on a low number of fish from these sites, and there was no significant reduction in shortfin eel length from sites downstream of Dunbars Drain. Within each of the five monitoring sites, there is no significant overall temporal shift in fish lengths over the survey years from 2004-2008 (Table 5).

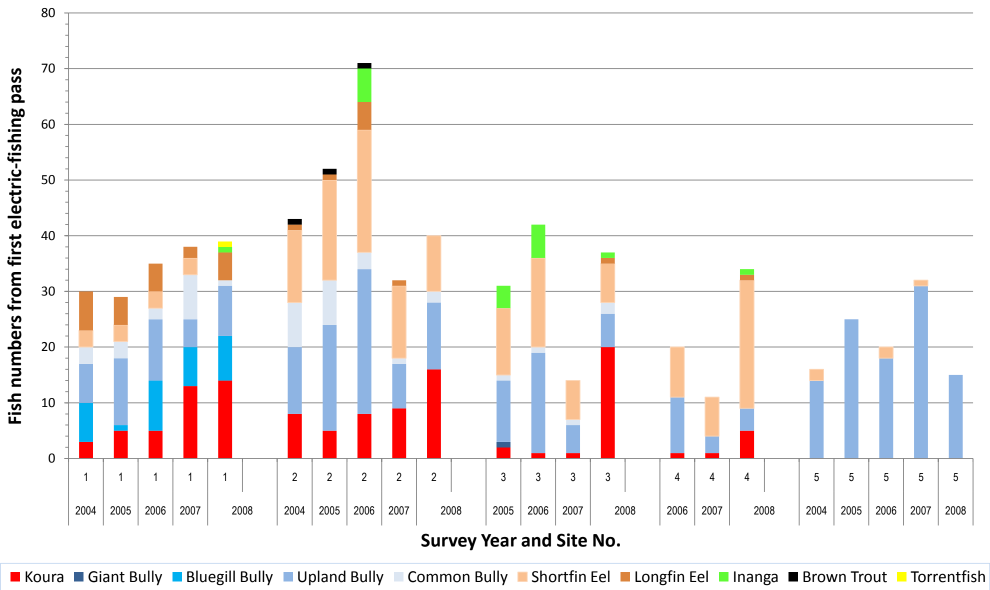

Figure 9 First sweep electro-fishing numbers from the five sites along Cashmere Stream surveyed between 2004 and 2008.

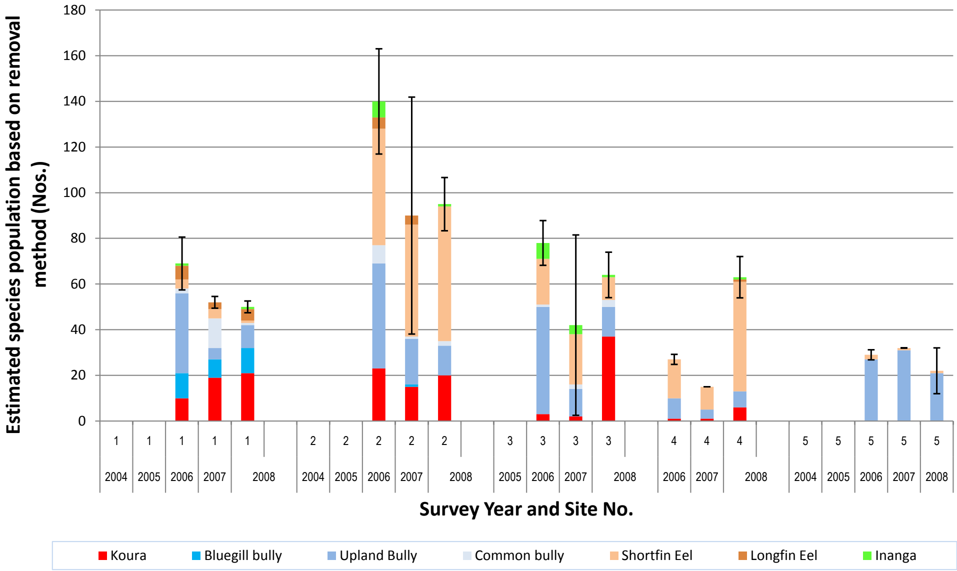

Figure 10 Fish population estimates from each of the five sites along Cashmere Stream surveyed between 2004 and 2008. No estimates are provided for rare species: giant bully, brown trout, and torrentfish. Data is not available for 2004 and 2005, as electro-fishing only involved a 'single pass', meaning that population estimates could not be calculated. Error bars equals one standard error based on the total fish population estimate.

Table 5 Length statistics for koura and fish caught in electro-fishing surveys at five sites along Cashmere Stream between 2004 and 2008. SD = standard deviation. Koura length is measured as the OCC (orbital carapace length). Koura length data from 2004 is not included in the table as it was not recorded.

Species	Mean Length (mm)	Minimum (mm)	Maximum (mm)	Standard Deviation (mm)	n
Shortfin Eel	**221.63**	**92**	**644**	**78.67**	**261**
2004	233.83	125	455	83.94	18
2005	230.52	106	508	77.97	33
2006	241.45	92	644	96.95	78
2007	203.77	119	376	56.17	57
2008	207.76	105	538	66.37	75
Longfin Eel	**269.10**	**123**	**666**	**131.42**	**39**
2004	252.88	143	439	113.18	8
2005	248.33	161	398	94.35	6
2006	294.73	123	583	166.62	11
2007	313.86	146	666	174.29	7
2008	220.43	126	316	57.33	7
Upland bully	**48.72**	**17**	**99**	**10.76**	**355**
2004	48.55	22	81	11.43	33
2005	47.94	25	65	8.20	67
2006	48.09	24	76	10.64	129
2007	51.35	22	99	13.74	65
2008	48.20	17	66	9.36	61
Common bully	**91.81**	**41**	**186**	**23.96**	**54**
2004	99.00	77	186	29.80	11
2005	83.58	41	111	23.33	12
2006	90.80	57	121	25.94	10
2007	87.60	44	105	19.24	15
2008	107.33	84	129	15.59	6

Bluegill bully	**56.59**	**42**	**81**	**7.86**	**39**
2004	56.14	42	60	6.74	7
2005	62.00	62	62	-	1
2006	55.45	46	65	5.79	11
2007	57.11	42	66	7.04	9
2008	57.09	42	81	11.37	11
Inanga	**71.88**	**55**	**104**	**11.59**	**26**
2005	68.25	63	74	5.56	4
2006	73.20	55	104	14.08	15
2007	73.67	71	77	3.06	3
2008	69.25	59	81	10.90	4
Brown Trout	**283.75**	**121**	**400**	**122.18**	**4**
2004	264.00	264	264	-	1
2005	400.00	400	400	-	1
2006	350.00	350	350	-	1
2008	121.00	121	121	-	1
Giant bully	**105.50**	**103**	**108**	**3.54**	**2**
2005	103.00	103	103		1
2007	108.00	108	108		1
Koura	**17.14**	**7**	**37**	**15.41**	**164**
2005	6.25	1.5	11	3.22	12
2006	9.28	4	17	4.22	29
2007	18.21	5	41	10.00	35
2008	15.48	7	37	8.08	77
Torrentfish	**63.00**	**63**	**63**	**-**	**1**
2008	63.00	63	63	-	1
Grand Total	**105.30**	**7**	**63**	**101.33**	

3.3 Channel Profiles

3.3.1 Overview

The channel profile varied along the length of Cashmere Stream, from a small channel with little soft sediment, to a wider, deeper channel with variable soft sediment depths (Table 6). Not surprisingly, sediment depth was negligible at Sites 1 and 5, reflecting the fast flow and coarse substrate of these two riffle sections. The sediment depth was greatest at Site 3 (upstream of the Dunbars Drain confluence), but was also elevated at the other two run sections (Sites 2 and 4; Table 6).

3.3.2 Temporal Changes

There were no significant temporal changes between 2006, 2007, and 2008 in the substrate index, free-water depth, or macrophyte depth (Table 7). However, sediment depth showed a significant change over time at Site 3, increasing in depth significantly in 2007, and remaining high in 2008 (Table 7; Figure 11). Free-water depth was lowest at Site 5 in all years, while it increased at Site 2 in 2007, before declining again in 2008. The substrate index showed no changes over time at the different sites, with large substrate present in Sites 1 and 5, and remaining constant over the years (Table 7; Figure 11). Macrophyte depth showed the same pattern, with the only significant difference for this variable being greater macrophyte depths at Site 3 than the other sites (Table 7; Figure 11). Note that for Site 2, while macrophytes dominated the fish and invertebrate survey sites, the channel profile transect was located upstream under trees, which stunted or precluded macrophyte growth.

Table 6 Summary statistics of width, and depths of free-water, macrophytes, and sediment recorded from detailed transects at the upstream end of each survey site. Total water depth is a summation of the free-water and macrophyte depths.

		Site 1	Site 2	Site 3	Site 4	Site 5
Width (m)		4.9	4.3	2.3	3.17	1.15
Free-water depth (m)	Mean	0.21	0.28	0.24	0.49	0.06
	Max	0.29	0.41	0.48	0.71	0.09
	Min	0.15	0.05	0	0.20	0.02
Macrophyte depth (m)	Mean	0	0.09	0.40	0.14	0.01
	Max	0	0.22	0.74	0.37	0.07
	Min	0	0	0.08	0.02	0
Sediment depth	Mean	0	0.18	0.40	0.14	0.01
	Max	0	0.27	0.74	0.37	0.07
	Min	0	0.06	0.08	0.02	0

Table 7. Results of two-way Analysis of Variance (ANOVA) testing for significant differences between sites, years, or the interaction of them both (year*site) for habitat variables recorded from channel profiles at five sites on Cashmere Stream in 2006, 2007, and 2008. n/s = significant, sig = significant ($p < 0.05$).

Habitat variable	Year	Site	Year*Site
Substrate Index	n/s	sig	n/s
Free-water depth (m)	n/s	sig	sig
Macrophyte depth (m)	n/s	sig	n/s
Sediment depth (m)	sig	sig	sig

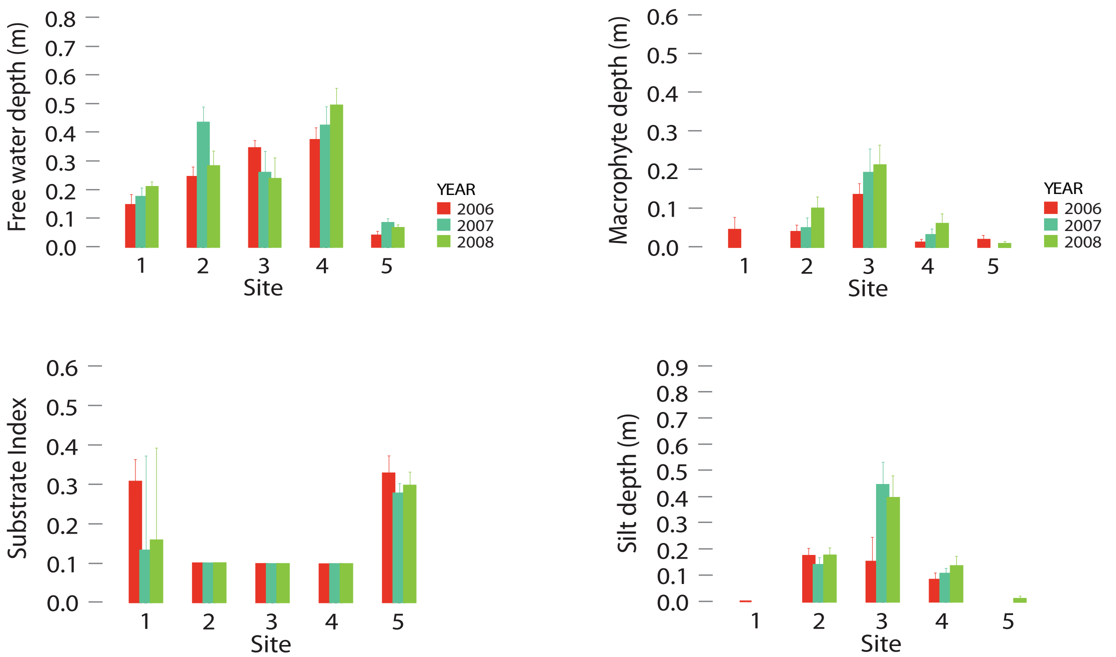

Figure 11 Temporal and site-specific changes in aquatic habitat variables at five sites along Cashmere Stream in 2006, 2007, 2008. Sites 1 and 2 are downstream of Dunsbar Drain.

4 DISCUSSION

4.1 Invertebrate Values

4.1.1 Overview

The invertebrate fauna found at the five surveyed sites typified that recorded from previous sampling rounds, and reflected the characteristics of Cashmere Stream as a silted lowland stream affected by the combined stresses of agricultural and urban landuses. The invertebrate fauna was indicative of its impacted status, with a community characterised by low diversity and dominated by the ubiquitous freshwater snail *P. antipodarum* and crustaceans such as the amphipod *P. fluviatilis*. Of the three cleanwater taxa groups (e.g., EPT taxa), which consists of mayflies, stoneflies, and caddisflies, only caddisflies were recorded.

The absence of such clean-water taxa as the mayfly *Deleatidium* is most likely linked to the effects of siltation on the stream biota, and serves to distinguish this waterway from the freshwater invertebrate diversity “hot-spots” found in Christchurch, specifically the upper Otukaikino catchment and the upper reaches of the Styx River (McMurtrie & Greenwood, 2008). However, the presence of certain caddisfly taxa and freshwater crayfish has led to Sites 1, 2, and 5 being recognised for their ecological values (EOS Ecology *et al.*, 2005), and the past (EOS Ecology, 2005; McMurtrie & Taylor, 2006; Burdon & Taylor, 2007) and present monitoring surveys have identified a total of ten caddisfly taxa present in Cashmere Stream. Similarly, recent survey work indicates that there is a relatively extensive population of freshwater mussels in the lower reaches of Cashmere Stream, and that freshwater crayfish may also be relatively well distributed in the upper reaches (EOS Ecology’s unpublished data).

To reiterate the views of McMurtrie & Taylor (2006) and Burdon & Taylor (2007), although Cashmere Stream is partially degraded, it still has relatively high ecological values when compared to other waterways located in the Southwest Christchurch area. The presence of ten caddisfly taxa, koura (freshwater crayfish) and kakahi (freshwater mussels) support the efforts to protect and restore this peri-urban waterway.

4.1.2 Temporal and Spatial Changes 2004–2008

Given that we have changed the emphasis of the analysis to focus more on long-term monitoring of temporal changes in stream ecology since 2004 and of site-specific effects since 2006, we have not included a section on discharge effects in this report (similar to the 2007 report). This is a reflection of the fact that there are a multitude of discharge points along the stream continuum which vary in the magnitude, duration, and timing of discharges, and in their water quality (particularly levels of suspended sediment).

There were relatively few significant temporal changes at three sites (Sites 1, 2, and 5) surveyed yearly between 2004 and 2008, although 2004 did have several consistent trends with lower taxa richness and EPT richness than the following years. In addition, *H. amabile* abundance was high in 2004, dropped markedly in 2005 and seems to be continuing a slow recovery over the years since. These points are discussed in detail in (Burdon & Taylor, 2007) but it does appear that there may have been a perturbation around 2004–2005 and subsequent recovery in the more pollution sensitive EPT taxa in Cashmere Stream over the last four years. It seems unlikely that this was an impact of the Aidenfield discharge as the effects were seen at both upstream and downstream sites. In general, most of the invertebrate indices fluctuate over time since the initial decline in 2005, with no consistent trend across the indices. This function may be related to environmental stochasticity, reflecting natural variation in communities in response to weather, disease, competition, predation, and/or other factors external to the biological interactions within communities, rather than a definite decline in stream health. Further monitoring will elucidate this difference.

No significant changes in taxa richness occur after 2004, indicating that the different sample processing techniques employed in 2004–2005 and 2006–2008 had little effect on taxa richness at least. In addition, there appeared to be no obvious effects of the bank vegetation trimming maintenance that had occurred just prior to the 2007 sampling at Sites 1–4 with no decreases in stream health indices or in abundances of sensitive EPT taxa in that year. The only significant temporal differences occurring between the last survey (2007) and this current survey were a decline in MCI and QUCI index scores in 2008. However, over this time the abundance of sensitive EPT taxa has remained similar and the abundance of the caddisfly H. amabile has continued its increasing trend since 2005. Thus the decline in stream health scores were not necessarily related to a decline in sensitive EPT taxa.

Unsurprisingly, there were differences between sites reflecting both their position along the stream continuum and their respective physical habitat types (i.e. riffle and run) in the more detailed site specific analysis conducted on the 2006–2008 data. There were no obvious and consistent differences at any particular site within a year that would indicate a large impact of a stormwater discharge event. In particular, the marked increase in sediment at Site 3 in 2007 seemed to have little impact on the invertebrate community with an actual increase in EPT richness and percentage EPT in this year. This may be related to aquatic macrophytes being the dominant habitat at this site; as long as sufficient macrophytes are present the depth of the sediment below them may not affect the invertebrate community. However, although percentage EPT increased in 2007 2008 it showed a marked decline at Site 3 in 2008, driven by the abundance of the caddisfly H. amabile. It is possible that this is a delayed response to the increase in sediment depth in 2007 as deeper sediment leads to a more unstable substrate that has a greater chance of being resuspended during floods or macrophyte maintenance practices. However it is also likely that some unknown factor caused H. amabile populations to increase during 2007, driving up percentage EPT as well. Further monitoring will investigate whether H. amabile abundance and percentage EPT continue to fluctuate naturally or remain low, which could potentially indicate an effect of the increased sediment levels at this site.

While most of these results may point to some level of resilience in the invertebrate community, this could be due to the fact that the invertebrates most sensitive to sediment pollution (e.g., mayflies and more sensitive caddisfly taxa) have already been lost from the catchment. Despite this, there is still concern that the effects of continual sediment discharges may be slowly and insidiously changing the fauna of Cashmere Stream (Eos Ecology, 2005), particularly in the sensitive riffle sections (Burdon & Taylor, 2007). The adverse effects of silt on stream biota have been well-documented in review (see Ryan, 1991) and the potential impacts on Cashmere Stream are discussed in depth in (Burdon & Taylor, 2007). Undoubtedly this is one of the key problems to be addressed in the Cashmere Stream catchment.

4.2 Fish Values

4.2.1 Overview

In 2008, the fish fauna of Cashmere Stream was similar to that recorded in previous years, although brown trout were not recorded at Site 2, and fish numbers were intermediate between the high numbers caught in 2006 and the much lower numbers recorded in 2007. The high number of fish in 2006 was due to the large numbers of juvenile upland bullies, approximately twice as many as normally found. In addition, in 2007 the lower fish numbers followed patterns of habitat modification caused by bank-vegetation trimming and channel macrophyte removal, which was undertaken by Council staff just prior to the surveys.

The discovery of one torrentfish at Site 2 in the current (2008) study was unusual as these fish are generally found in fast-flowing riffle sections of large braided rivers. They are very rare within Christchurch City streams (i.e., have never been previously recorded), and possibly this fish was an immigrant from the Waimakariri River, where the fish are quite frequently encountered.

No trout were caught at Site 2 in 2008, where previously relatively large individuals have been captured. However, this does not mean that the trout are no longer present here as larger fish are faster and more difficult to catch with an electro-fishing machine. In addition, the landowner reports regular sightings of trout at this site. Electro-fishing works well for sampling populations of small or slow swimming fish species however it is not a suitable method for monitoring populations of large fast-swimming fish like trout.

4.2.2 Time effects

Physical fish habitat features continue to change slightly over time, although the general physical characteristics of the five sites remain broadly the same. Substrate embeddedness at Site 1 (Cashmere Road Bridge) may be reducing because of a change in the local hydraulics caused by construction of a foot bridge in 2005. Often bridges confine the flow slightly, and the banks under the bridge lack vegetation. These factors may be causing a reduction in substrate embeddedness, and this would be beneficial to resident bluegill bullies which are thought to feed on invertebrates dwelling on the underside of the stones (McDowall, 1990). In contrast, the torrentfish, first identified from this site in this (2008) survey, is a fast-water specialist, that is likely to feed on the top side of the rocks and has a body shape (Figure 8) that is an adaption for creating down-force in fast water.

Torrentfish have not been recorded from Christchurch's city waterways previously, but little can be inferred from the identification on one specimen from Cashmere Stream. Given the immigration into Cashmere Stream by the torrentfish it is not surprising that it remained at Site 1; as this is the only location of a riffle in Cashmere Stream and of a water velocity that would come close to the preferred conditions for this fish. Another example of a disorientated sea migrant is Chinook salmon. These large fish, which normally spawn in the Waimakariri and Rakaia Rivers, do occasionally enter the Heathcote River, and spawn there (Taylor, 2003). A passerby this year reported finding a dead salmon carcass at Site 1 on Cashmere Stream, and this would be consistent with our perceptions from the 2003 spawning survey.

The CPUE plot (Figure 10) indicates that numbers of bluegill bullies are indeed being maintained at Site 1, and the coarse, un-embedded nature of the substrate could be a major reason of the sustainability of this species at this site. There has also been always a large boulder mid-channel which has collected time-variable quantities of in-channel vegetation (Figure 12). This small but important microhabitat has proved to be an excellent sanctuary for koura and for inanga, a weak-swimming fish which could not otherwise tolerate the swift flows at Site 1. Indeed, at Site 1 the increasingly stable true left (north) bank that occurred with an increase in rooted macrophytes and the retained instream detrital matter that has been allowed to accumulate in the 2007 and 2008 surveys (Figure 12) may be of benefit to koura, by providing both habitat and food.

In 2008, when our survey preceded in-channel and bank maintenance work, inanga were recorded from four of the five sites. This is the best distribution for this species in the monitoring programme, given that in 2007, after recent bank-trimming, inanga were only recorded from one site (Site 3). As mentioned in the

2007 report (Burdon & Taylor, 2007), inanga are pelagic fish which are sensitive to loss of instream and overhanging vegetation cover, and this species is likely to be quite sensitive to the types of channel works we observed (Richardson & Taylor, 2002).

The lack of preceding channel clearing in 2008 had allowed the total fish cover at Sites 2 and 3 to revert from the low levels recorded in 2007 to levels similar to that recorded in 2006. Fish numbers had also recovered from the low levels recorded in 2007. Of particular note was the large increase in koura numbers at Site 3, compared to the levels recorded from previous years. It is unsure what has caused the substantial increase in numbers at this site, but it is possible that in-channel works (macrophyte removal) and/or bank vegetation removal may have lowered the numbers in previous years, although this cannot be substantiated. Alternatively, the higher numbers in the 2008 survey may merely reflect better survivability in the koura population during the preceding twelve months. No clear conclusions can be made at this stage regarding the increase in koura numbers at this site. Further monitoring may elucidate a long-term trend, while a study concentrating on different bank and in-channel maintenance practices may help to determine whether or not this has any lasting effect on koura, which live in the banks and amongst the macrophytes. Numbers of koura appear to have remained consistent at Site 1 and 2 over the last three years (2006–2008), with the slightly lower numbers in 2004 and 2005 possibly an artefact of less fishing pressure during these earlier surveys.

The number of shortfin eels has declined to negligible levels at Site 1, and this may be related to the changes at this site over time. However, numbers of upland bullies have declined at most sites from 2006 levels, when they numerically dominated the catch. For four of the five sites, upland bully numbers were at their highest recorded level in 2006. Our data from other lowland waterways around Christchurch has shown than upland bullies can reach high numbers when they are spawning locally (Taylor & McMurtrie, 2002; McMurtrie *et al.*, 2005), but these could be quite variable over time.

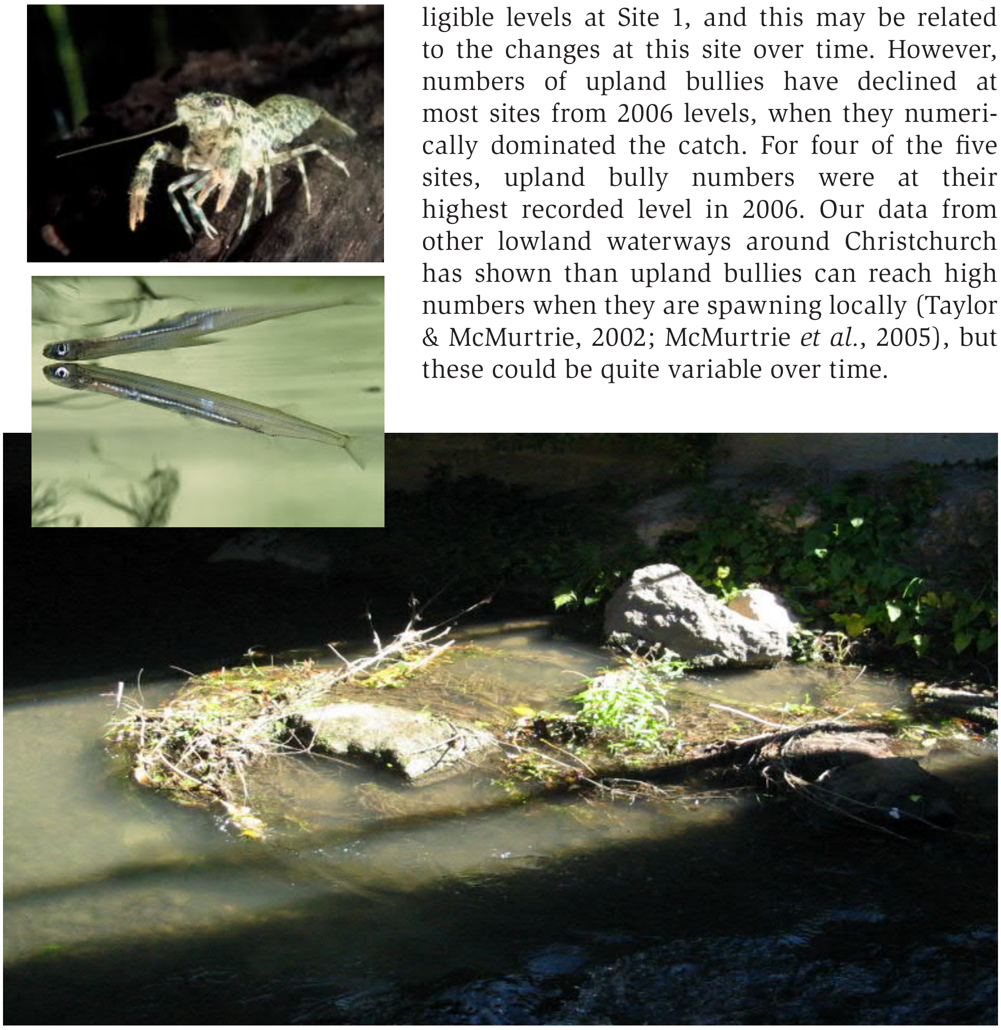

Figure 12 Accumulated debris amongst stony rubble at Site 1 in 2008 (above). Inanga (middle) and koura (top) were found among slow-water habitat within this debris cluster.

The failure to capture one particular large longfin eel from Site 2 in 2008 was probably the main reason for an approximate 10 cm decline in the mean length of longfin eels. This large fish (666 mm T.L.) was only recorded from this site in 2006 and 2007, and is known by the landowner to reside in a pipe in a concrete wall at this site. For this reason, it may not have been disturbed by bank works in the vicinity, and it is possible that this large fish could have migrated to sea to spawn, from which it will never return.

The giant bully has been twice recorded from Site 3, once in 2005, and again during the second pass at this site in 2007. It could be the same fish, but it is not possible to draw much from its absence in 2008.

5 CONCLUSION

The fluctuations in the invertebrate community composition and stream health metrics during the years 2004–2008 do not appear to be linked to any large individual stormwater or sediment inputs; certainly none large enough to cause significant effects on the invertebrate community at a particular site. It is possible that a natural or man-made perturbation occurred in 2004–2005 causing a decline and subsequent recovery by key EPT taxa, such as the cased caddisfly *H. amabile* but this was not linked to the Aidenfield discharge via Dunbars Drain. Environmental stochasticity and processing methodology (only in the 2004 survey) may therefore be causing many of the small fluctuations in stream health measures.

Although it remains equivocal as to whether there was an adverse effect of the Aidenfields discharge via Dunbars Drain, the rapid build-up of silt at Site 3 (upstream of Dunbars Drain) over the past three years demonstrates that there are other sources of sediment entering Cashmere Stream. It does not appear that the accumulation of soft sediment between 2006 and 2007 at this site is detrimentally affecting the invertebrate community, although a decline in the caddisfly H. amabile and the % EPT taxa at this site in 2008 may have been a delayed response. The macrophytes present across the channel at this site may help to buffer the effects of soft-sediment build-up by providing a stable habitat above the silted substrate.

The effects of sedimentation are likely to be the most conspicuous in riffle habitats (e.g. Sites 1 and 5) where the accumulation of fine sediment can smother invertebrates, their periphyton food source and the interstitial spaces between the substrate which they use as habitat. Thus, sediment entering the stream remains to be the most pertinent issue to be addressed regarding the management of Cashmere Stream. The continued scrutiny of major earthworks and associated stormwater run-off from subdivisions such as Aidenfields is essential.

However, although the sediment is undoubtedly affecting invertebrate communities there is no compelling evidence of direct sedimentation of fish habitats, or a change in the fish fauna to one more tolerant of habitat siltation. In contrast, in sites downstream of the Aidenfield outfall, the fish fauna is trending towards one less tolerant of sedimentation with increasing numbers of koura, bluegill bully, and inanga. Rather, shifts in fish fauna were reflective of riparian and instream maintenance works and not a gradual systemic deterioration that would be expected from chronic sedimentation or eco-toxicity problems. Thus, riparian vegetation management and stream-bed dredging as a management practice to enhance drainage values need to be investigated as they seem to be impacting fish populations, and the large plumes of sediment that are freed to drift downstream during this practice may begin to build up in slower flowing areas downstream and eventually affect invertebrate populations as well.

There is need for an integrated catchment management plan which tackles all sources of sediment entering Cashmere Stream (i.e. both urban and rural) and riparian and instream vegetation management practices. EOS Ecology is currently investigating the main sources of sediment input to Cashmere Stream during high and normal flows and there is a catchment management plan for South West Christchurch in development. It is anticipated that this plan will address the above issues by better integrating landuse planning with catchment management, but also by highlighting the responsibility of environmental stewardship to all stakeholders.

6 RECOMMENDATIONS

The continued monitoring of the five selected sites over time should be the focus of future ecological survey work. However, specific experimental studies would be required to determine whether there is any actual impact of in-channel and bank maintenance practices on fish and koura. Continued co-operative liaisons between stream maintenance crews and environmental monitoring staff will aid in the continuity of sampling conditions at the monitoring sites over time. All sites should be assessed along the continuum of increasing sediment and stormwater inputs with distance downstream. EOS Ecology's new water quality monitoring programme should be used in conjunction with ecological work and should serve to identify some of the more significant contributors of sediment to the stream. This will help give direction and impetus to management efforts aimed at mitigating and removing the causes of these inputs.

7 ACKNOWLEDGEMENTS

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9 APPENDICES

9.1 Appendix I: Site photographs

Site 1: 2004 Survey: Immediately downstream of Cashmere Road bridge, looking downstream from road.

Site 1: 2005 Survey: Immediately downstream of Cashmere Road bridge, looking downstream from foot bridge. (Note the new footbridge to the left of the image).

Site 1: 2006 Survey: Immediately downstream of Cashmere Road bridge, looking downstream from foot bridge.

Site 1: 2007 Survey: Immediately downstream of Cashmere Road bridge, looking downstream from foot bridge.

Site 1: 2008 Survey: Immediately downstream of Cashmere Road bridge, looking downstream.

Site 2: 2004 Survey: Downstream of Dunbar's Drain, looking downstream.

Site 2: 2005 Survey: Downstream of Dunbar's Drain, looking downstream.

Site 2: 2006 Survey: Downstream of Dunbar's Drain, looking downstream.

Site 2: 2007 Survey: Downstream of Dunbar's Drain, looking downstream.

Site 2: 2008 Survey: Downstream of Dunbar's Drain, looking downstream

Site 3: 2005 Survey: Upstream of Dunbar's Drain, looking downstream.

Site 3: 2006 Survey: Upstream of Dunbar's Drain, looking downstream.

Site 3: 2007 Survey: Upstream of Dunbar's Drain, looking downstream.

Site 3: 2008 Survey: Upstream of Dunbar's Drain, looking downstream.

Site 4: 2006 Survey: Upstream of Milns Drain, looking upstream.

Site 4: 2007 Survey: Upstream of Milns Drain, looking upstream.

Site 4: 2008 Survey: Upstream of Milns Drain, looking upstream.

Site 5: 2004 Survey: Upstream of Sutherland Road, looking upstream.

Site 5: 2005 Survey: Upstream of Sutherland Road, looking upstream.

Site 5: 2006 Survey: Upstream of Sutherland Road, looking upstream.

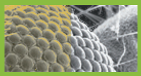
Site 5: 2007 Survey: Upstream of Sutherland Road, looking upstream.

Site 5: 2008 Survey: Upstream of Sutherland Road, looking upstream.

ASSESSMENT AND MONITORING

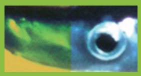
RESTORATION DESIGN

LABORATORY SERVICES



INFORMATION DESIGN

PHOTOGRAPHIC RESOURCES



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