0028–8330/05/3901–0017

Longitudinal patterns in benthic communities in an urban stream under restoration

TANYA J. BLAKELY
JON S. HARDING
School of Biological Sciences
University of Canterbury
Private Bag 4800
Christchurch, New Zealand
email: tanya.blakely@canterbury.ac.nz

Abstract Okeover Stream, on the University of Canterbury campus in Christchurch, New Zealand, has been the subject of restoration efforts since 1998. Our study focused on quantifying the response of this urban stream to current restoration efforts. Initially, physico-chemical conditions and biological communities at three sites along the Okeover Stream were compared with three physically similar sites on each of nearby Waimairi Stream and Avon River. General physical and chemical parameters were similar in all streams with circum-neutral pH, specific conductivity ranging from 167 to 173 µS/cm, dissolved oxygen ranging from 9.0 to 9.2 mg/litre, low turbidity, and similar hydrological conditions. However, analysis of heavy metals in the sediment showed mean lead (Pb) concentrations in Okeover and Waimairi Streams exceeded ANZECC ISQG-low trigger values (86.9 and 83.7 mg/kg, respectively), whereas Avon River sediment Pb levels (27.3 mg/kg) were below trigger values. Benthic taxonomic richness did not differ significantly among the three streams. However, Okeover Stream community was dominated by the amphipod *Paracalliope fluviatilis*, whereas in Waimairi Stream and Avon River the gastropod snails *Potamopyrgus antipodarum* and *Physella acuta* were the dominant benthic fauna. A further assessment made at six sites along 1200 m of Okeover Stream showed no distinct longitudinal patterns in physical or chemical conditions, but there was a strong pattern in benthic macroinvertebrate communities. Taxonomic richness and caddisfly diversity increased downstream, with twice as many taxa at the most downstream site than the uppermost sampling site. In upper reaches, copper (Cu), Pb, and zinc (Zn) concentrations in sediments all exceeded ANZECC ISQG-low trigger values. Despite ongoing restoration efforts in Okeover Stream, sedimentation, the presence of high heavy metal concentrations, intermittent flows in headwaters, and possible barriers to adult recolonisation seem to be having a continuing negative impact on benthic communities, especially in the headwaters.

Keywords urban streams; restoration; benthic invertebrates; water chemistry; heavy metals

M04048; Online publication date 18 February 2005
Received 13 February 2004; accepted 4 October 2004

INTRODUCTION

Stream ecosystems have been subjected to extensive modification through human development worldwide, and in contemporary lowland landscapes changes associated with urbanisation are particularly obvious. The effects of urbanisation on stream ecosystems are complex, and the study of urban stream ecology is still in its infancy (Campbell 1978; Hogg & Norris 1991; Grimm et al. 2000; Suren 2000). Nevertheless, it is well known that urban streams are frequently subjected to three general kinds of impacts: inputs of pollutants (e.g., chemicals and sediments), altered hydrological regimes (e.g., reduced water tables and exaggerated flooding), and changes in riparian and in-stream habitat (e.g., channelisation and reduced habitat complexity and diversity).

Urban run-off, either from stormwater drains or overland flow, often contains high levels of sediment as well as commercial and residential contaminants such as petrols, oils, pesticides and herbicides, heavy metals, polycyclic aromatic hydrocarbons (PAHs), nitrogen, phosphorus, and other pollutants, all of which markedly alter the water chemistry of receiving waterways (Whiting & Clifford 1983; Garie & McIntosh 1986; Beasley & Kneale 2002).

Although the mechanisms by which these pollutants affect individual aquatic taxa are not fully understood, their effects on stream communities as a whole are well documented (Paul & Meyer 2001).

An increase in the amount of impervious surfaces, such as roads, carparks, and buildings, is characteristic of urban catchments. Many of these surfaces are linked directly to stormwater systems and to urban streams. Ultimately, they affect a catchment's permeability to precipitation, as well as reducing groundwater recharge and drastically increasing run-off during rainfall events (Suren 2000; Paul & Meyer 2001). Furthermore, many urban stream channels have been straightened for flood control (Riley 1998), removing natural pool-riffle sequences and reducing in-stream habitat heterogeneity. The removal of in-stream and riparian vegetation is also common practice in developed catchments, and results in greater sediment run-off, loss of habitat associated with large woody debris, and altered water temperature regimes (LeBlanc et al. 1997; Timperley & Kuschel 1999). These changes can have significant effects on the ability of a stream to support benthic invertebrate diversity (Williams 1980), maintain high dissolved oxygen levels, and enable members of the biota to complete their life histories (Wallace & Anderson 1996; Scarsbrook 2000).

Generally, benthic invertebrate communities in urban streams have relatively few taxa, with pollution-sensitive insect orders, such as Ephemeroptera, Plecoptera, and Trichoptera (EPT taxa) either absent or poorly represented (Campbell 1978; Pratt et al. 1981; Hogg & Norris 1991; Roy et al. 2003).

Within New Zealand, much research on urban streams has focused on the influences of riparian vegetation, and to a lesser extent physico-chemical conditions (McConchie 1992; Collier et al. 1995; Hickey & Clements 1998, but see Maxted et al. 2003), whereas relatively few studies have investigated benthic invertebrate communities (but see Suren 1993; Suren et al. 1998).

Increasingly, local authorities and residents are recognising the degraded condition of urban streams and, as a result, greater emphasis is being placed on restoration and remediation of urban waterways. Nevertheless, surprisingly few studies have attempted to measure restoration success (but see Charbonneau & Resh 1992) and despite the large number of restoration projects being undertaken, post-restoration monitoring is rare (Kondolf & Micheli 1995; Riley 1998).

Within Christchurch City more than 27 km of waterways have been subject to some form of restoration (Rachel Barker, Christchurch City Council pers. comm.), and since 1999 over 4 km of artificial drains have been converted into "naturalised" waterways (Christchurch City Council 2003). In 1998, a partnership was formed between the Christchurch City Council and the University of Canterbury, to restore several streams on the University campus (O'Brien et al. 1998). Thus far, the main focus of these efforts has been Okeover Stream.

In this study, we compared the response of Okeover Stream to 4 years of restoration with conditions in other nearby unimproved urban streams. We then conducted a more intensive assessment of physical, chemical, and biological conditions along Okeover Stream, to determine any longitudinal improvements.

METHODS

Study sites

When Christchurch City was first developed, the region consisted of a mosaic of wetlands and meandering spring-fed streams surrounded by indigenous wetland vegetation (Dendy 1900). In 1875, the Christchurch Drainage Board was empowered to manage city waterways, channelise streams, and pipe stormwater directly into the waterways to maximise floodwater drainage and reduce the risk of disease. Consequently, many of the smaller, spring-fed streams were piped underground or converted to box-culvert channels. As the city expanded, the water table lowered and numerous surface flowing streams disappeared.

Okeover Stream, one of two waterways that flow through the University of Canterbury campus, is a spring-fed, first order stream with headwaters now originating within the campus (some 3 km downstream from their origin 25 years ago; Robb 1980). It receives more than 40 stormwater and air-conditioning discharges, and both heavy rainfall and the operation of air-conditioning in university buildings can markedly alter flow conditions. Since 1998, restoration efforts within Okeover Stream have focused on riparian planting, in-stream habitat improvement, and sediment control (Table 1).

Fieldwork was undertaken at 12 sites along three spring-fed, first and second order tributaries of the Avon River, draining north-western Christchurch. The three tributaries, Okeover (sites O1–O6), Waimairi (W1–W3), and Avon (A1–A3), were within a 1.5 km radius of the University of Canterbury campus (Fig. 1). Where possible, sites were selected with similar morphological and flow

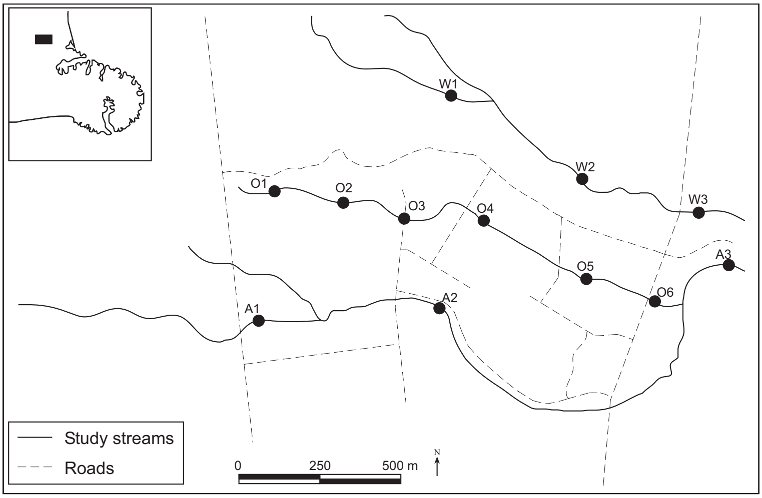

Fig. 1 Map showing the 12 study sites on Okeover (O1–6) and Waimairi Streams (W1–3) and the Avon River (A1–3), north-west Christchurch, New Zealand.

Table 1 Restoration projects implemented in Okeover Stream, Canterbury, New Zealand from 1998 to 2003.

Year	Activities	Length of stream	Effect
1998	Riparian plantings Ongoing—10 discrete planting events	Upper reaches (c. 400 m)	Reduced riparian run-off; increased in-stream shading
2000	In-stream habitat; boulder addition	Mid reaches (50 m)	Improved habitat and flow Substrate heterogeneity increased
2000	Wetland in-stream plantings	Upper and mid reaches (10 m)	Wetland established
2001	Sediment trap construction	Upper and mid reaches (100 m)	Reduced suspended and deposited sediments; riffles created
2003	Channel meandering; riparian and in-stream plantings; stony substrate added	Headwaters (150 m)	Riffle-pool sequences created; increased habitat heterogeneity

characteristics. Generally, they were shallow (0.05–0.15 m), narrow (<3.6 m wide channels), pebble-cobble dominated riffles, with similar water velocities (0.15–0.3 m s^{-1}). However, the two uppermost sites in Okeover Stream (O1 and O2) were subject to periodic drying, as a significant source of their flow was air-conditioning water discharges from university buildings, which were turned off during the University holidays. Furthermore, the streambed at site O1 was dominated by willow (*Salix* spp.) roots, rather than pebbles and cobbles.

To investigate the effects of restoration efforts in Okeover Stream, the three Okeover sites (O2, O4, O6) were compared with the Avon and Waimairi sites. Longitudinal patterns in Okeover Stream were also examined by sampling six sites along 1200 m of the stream (O1–O6).

Physico-chemical and biotic parameters

Basic water chemistry parameters were assessed in November and December 2002 (austral spring and summer), during baseflow conditions. Specific conductivity, pH and temperature (Oakton CON 10 Series), as well as turbidity (HACH 2100P Turbidimeter) and dissolved oxygen (DO) (YSI 550 DO) were recorded using standard meters. NO_3 and PO_4-P were estimated spectrophotometrically (using low range method, HACH 1990). However, because of errors in our NO_3 results we have excluded them from our analyses. Alkalinity was measured using an acid titration method (APHA 1980). DO and temperature were monitored more intensively at the six sites along Okeover Stream, on eight occasions between November 2002 and January 2003. At these times, measurements were taken at the same place, between 1100 and 1130 h.

Deposited inorganic sediment was collected randomly from riffles at 11 sites, in the three waterways, on one occasion and tested for copper (Cu), cadmium (Cd), lead (Pb), and zinc (Zn) using atomic absorption spectrophotometry (Hills Laboratory, Wellington, New Zealand). Average light levels, each calculated over a 15 s period, were recorded 2 m above the stream, along a 10 m section at each site using a Li-cor 250 meter, on two occasions. Substrate size was determined by estimating the percentage of each substrate size class, along a single transect across the stream channel, at each study site. Substrate was categorised as boulders (>256 mm), cobbles (64–256 mm), coarse gravel (10–63 mm), fine gravels (2–9 mm), or silt/sand (<2 mm). From this a substrate index (SI), ranging between 3 and 7 (where 3 indicates 100% silt/sand and 7 indicates 100% boulders) was calculated (Jowett & Richardson 1990).

Biomass of deposited organic and inorganic sediments was measured on one occasion, in November 2002, by randomly collecting five cobbles from a pre-defined 10 m riffle, at each site. However, deposited sediments were not assessed at O1, as cobbles and pebbles were absent. All deposited organic and inorganic sediments were brushed from each cobble, vacuum filtered onto pre-weighed and pre-ashed Whatman glass microfibre filters (GF/C), dried at 50°C for 24 h and ashed at 500°C for 4 h. Dried and ashed filters were weighed (±0.001 g).

Chlorophyll *a* was estimated at each site, on one occasion, from a further five randomly selected cobbles. Pigments were extracted in 90% ethanol for 24 h at 4°C in the dark, and absorbances at 665 and 750 nm were measured by spectrophotometer. Chlorophyll *a* concentration was calculated as described by HACH (1990). Chlorophyll *a* was not measured at O1, as cobbles and pebbles were absent.

Benthic invertebrate sampling

Benthic invertebrate community composition at each of the 12 sites was examined using two methods. Two qualitative kick-net samples (500 µm mesh net) were collected from a pool and a riffle in a pre-defined 10 m reach at each site, where the substrata was disturbed for 45 s on two occasions, in November and December 2002. Quantitative benthic invertebrate sampling was conducted by taking three Surber samples (0.05 m^2, 500 µm mesh) at each of the 12 sites, in November 2002. Surber samples were randomly collected from shallow riffles and substrate was disturbed to an approximate depth of 5 cm. All samples were preserved in the field with 70% ethanol, and identified in the laboratory to the lowest practicable taxonomic level, usually species or genus (Winterbourn 1973; Winterbourn et al. 2000).

Statistical analyses

Inter-stream comparisons

To determine if physico-chemical and biotic variables differed between streams the parameters were analysed using one-way analyses of variance (ANOVAs), with the three sites along each stream as replicates. For variables measured on two sampling occasions mean values at each site were used as replicates of streams. Using a one-way ANOVA, inter-stream differences in cadmium (Cd), Cu, Pb, and Zn levels in deposited sediments were also tested, using the three sites in each stream as replicates.

To determine if invertebrate densities differed between streams, a nested ANOVA was used, where the three streams (each with three sites) were the main effect tested, whilst the three samples taken at each site were nested within these. To ascertain total species richness and number of EPT taxa at each site, kick-net samples taken from pools and riffles on the two occasions were combined, and any additional taxa present in Surber samples were included. Thus, differences in taxonomic richness and EPT taxa between streams were investigated using one-way ANOVAs, with sites as replicates of the three streams.

Longitudinal patterns along Okeover Stream

To determine if physico-chemical and biotic variables differed, one-way ANOVAs were used. We investigated if the variables that had been measured on two sampling occasions differed over time using *t* tests; no significant differences between the two sampling periods in any parameter were found (i.e., $P > 0.25$). Thus, each sample was treated as a replicate in subsequent ANOVAs as described by Underwood (1997).

Differences in invertebrate densities longitudinally along Okeover Stream were investigated using a one-way ANOVA, where samples were treated as replicates at each site. Taxonomic richness and the number of EPT taxa longitudinally along Okeover Stream were also compared graphically.

All response variables were log transformed (x+1 where necessary) to meet assumptions of homoscedasticity and normality (Zar 1999). Fisher's Least Significant Difference (LSD) post-hoc tests were made where applicable, to determine where pairwise differences lay.

RESULTS

Inter-stream comparisons

Comparisons of water chemistry, and physical conditions between the three streams indicated very few differences.

For example, across streams pH ranged from circum-neutral to slightly acidic and DO concentrations were similar, with means of 9.0–9.2 mg/litre (Table 2). Turbidity was consistently low in all streams under baseflow conditions, and neither temperature nor specific conductivity differed significantly among streams (Table 2). Alkalinity was significantly greater in Waimairi Stream (44.0 mg/litre $CaCO_3$) than Okeover Stream and Avon River (both 41.5 mg/litre $CaCO_3$), whereas PO_4-P concentrations were negligible in the three systems and did not differ significantly (Table 2). Furthermore, no significant inter-stream differences were found in light levels (Table 2). Similarly, substrate index values were comparable across the three streams, ranging from 5.2 to 5.4, indicating a predominance of cobble substrata.

Although amounts of deposited inorganic and organic sediments and periphyton concentrations varied across the three streams, no significant differences were detected (ANOVAs: $F_{2,6} = 0.303$, $P = 0.75$; $F_{2,6} = 0.259$, $P = 0.78$; and $F_{2,6} = 4.785$, $P = 0.057$, respectively) (Table 2).

Heavy metal concentrations in sediments were similar among streams, with no differences in levels of Cd, Pb, and Zn, among streams (Table 3). However, sediment Cu concentrations were highly variable among streams with Okeover Stream having the highest mean concentration (32.3 mg kg^{-1}) compared with Waimairi Stream and Avon River (16.8 and 9.5 mg kg^{-1}, respectively) (Table 3). Pb levels in Okeover and Waimairi Stream sediments

Table 2 Water chemistry and physical parameters measured in Okeover Stream (O) and the two reference streams, Waimairi Stream (W) and Avon River (A), New Zealand, on two occasions in November and December 2002. Note: deposited inorganic and organic sediments and periphyton were measured on one occasion in November 2002. pH is shown as a range, whereas other parameters are means ±1SE. (NS, not significant.)

	O	W	A	*P*
pH	6.1–6.2	5.8–5.9	5.8–6.0	
Dissolved oxygen (mg/litre)	9.0±0.3	9.2±0.2	9.1±0.2	NS
Turbidity (NTU)	1.9±0.3	1.1±0.2	1.8±0.7	NS
Temperature (°C)	14.0±0.2	13.8±0.2	14.0±0.3	NS
Alkalinity (mg/litre $CaCO_3$)	41.5±0.4	44.0±0.1	41.5±0.1	$P < 0.01$
PO_4-P (mg/litre)	0.0±0.0	0.0±0.0	0.1±0.1	NS
Specific conductivity (µS cm^{-1})	167.3±4.1	171.0±2.0	173.0±3.2	NS
Light (µmol)	192±127	87±46	287±120	NS
Substrate index	5.2±0.3	5.4±0.1	5.2±0.4	NS
Inorganic sediment (mg cm^{-2})	1.82±0.9	2.89±0.4	3.39±1.8	NS
Organic sediment (mg cm^{-2})	1.38±0.6	1.82±0.7	1.39±0.7	NS
Periphyton (µg Chl. *a* cm^{-2})	0.48±0.2	0.83±0.3	3.32±1.2	NS

Table 3 Mean cadmium (Cd), copper (Cu), lead (Pb), and zinc (Zn) levels (mg kg^{-1} ±1SE, $n = 3$) in deposited sediments from the Okeover (O) and Waimairi (W) Streams and the Avon (A) River, New Zealand, sampled on one occasion in May 2003. ANZECC ISQG-low (Interim Sediment Quality Guideline) is equivalent to a "trigger value" whereas ISQG-high indicates high ecological impact (ANZECC 2000). Values in bold exceed (ISQG-low) trigger values.

	O	W	A	*P*	ISQG-low	ISQG-high
Cd	0.06±0.01	0.05±0.01	0.06±0.02	NS	1.5	10
Cu	32.3±7.2	16.8±4.0	9.5±1.7	$P < 0.05$	65	270
Pb	**86.9±46.6**	**83.7±21.9**	27.3±6.4	NS	50	220
Zn	109.8±9.4	102.4±11.8	113.5±19.8	NS	200	410

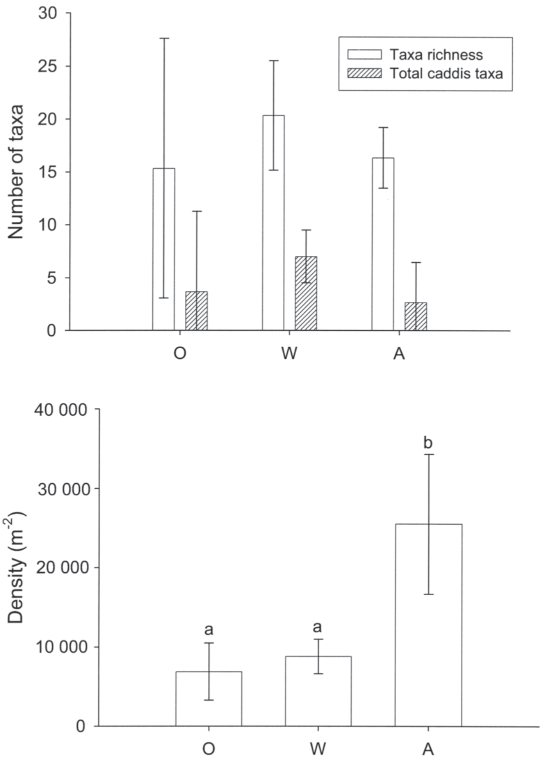

Fig. 2 Mean taxonomic richness (± 95% CI; n = 3) and total number of Trichoptera taxa (±95% CI; n = 3) (top), and benthic invertebrate densities (± 1SE; n = 3) in Okeover (O) and Waimairi Streams (W) and the Avon River (A), north-west Christchurch, New Zealand, between November and December 2002. Letters indicate significant (P < 0.05) inter-stream differences (Fisher's Least Significant Differences post-hoc test).

exceeded ANZECC trigger values (ISQG-low) (Table 3), whereas concentrations of Cd, Cu, and Zn, did not exceed ANZECC trigger values in any of the streams (Table 3).

Waimairi Stream had a greater total number of taxa (25) than Okeover Stream and Avon River (22 and 19, respectively). Mean taxonomic richness per stream ranged from 15 in Okeover to 20 in Waimairi Stream (Fig. 2). These differences were not statistically significant (ANOVA: $F_{2,6}$ = 2.12, P = 0.20). Furthermore, there were no statistical differences in mean numbers of caddisfly taxa among streams (ANOVA: $F_{2,6}$ = 2.11, P = 0.20; Fig. 2) and mayflies and stoneflies were absent.

Invertebrate densities were significantly higher in the Avon River (mean 25 000 per m^2) than in Okeover and Waimairi Streams (6900 and 8800

Table 4 Water chemistry and physical parameters at six sites along Okeover Stream, New Zealand sampled on two occasions in November and December 2002. Note: deposited inorganic and organic sediments and periphyton were measured on one occasion in November 2002. pH is shown as a range, whereas other parameters are means ±1SE. (NS, not significant.)

	O1	O2	O3	O4	O5	O6	P
pH	6.0–6.5	6.1–6.3	6.2–6.3	6.2–6.4	6.2–6.5	6.2–6.3	
Dissolved oxygen (mg/litre)	8.9±0.3	8.8±0.0	9.0±0.3	8.6±0.1	8.5±0.4	9.5±0.1	NS
Turbidity (NTU)	0.4±0.0	1.5±0.6	1.5±0.0	1.7±0.5	2.4±0.0	2.4±1.1	P < 0.05
Temperature (°C)	13.7±1.3	13.6±1.3	14.9±0.3	14.1±0.3	14.3±0.3	14.3±0.5	NS
Alkalinity (mg/litre $CaCO_3$)	42.3±1.3	42.3±1.3	40.8±0.3	41.3±0.3	40.8±0.3	41.0±0.5	NS
PO_4- P (mg/litre)	0.06±0.01	0.04±0.02	0.02±0.0	0.04±0.01	0.02±0.02	0.02±0.02	NS
Specific conductivity (µS cm^{-1})	158±3	160±6	154±3	174±15	178±20	168±9	NS
Light (µmol)	22±0	80±4	35±4	450±347	313±298	47±1	NS
Substrate index	3.4	4.6	4.2	5.2	5.5	5.7	
Inorganic sediment (mg cm^{-2})		1.15±0.4	0.9±0.1	3.72±1.8	1.46±0.5	0.6±0.1	NS
Organic sediment (mg cm^{-2})		1.14±0.3	0.49±0.1	2.54±1.1	1.04±0.6	0.45±0.1	P < 0.05
Periphyton (µg Chl. a cm^{-2})		0.83±0.2	0.25±0.1	0.49±0.2	0.16±0.0	0.12±0.0	P < 0.01

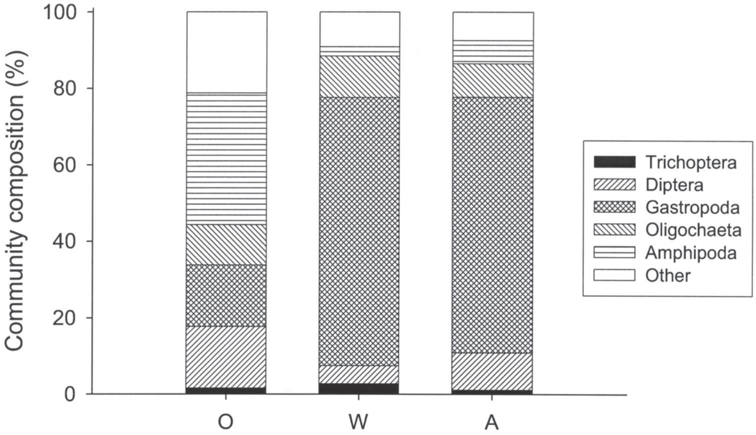

Fig. 3 Relative abundances (%) of six taxonomic groups in Okeover (O) and Waimairi Streams (W) and Avon River (A), north-west Christchurch, New Zealand, in November 2002.

Table 5 Cadmium (Cd), copper (Cu), lead (Pb), and zinc (Zn) levels (mg kg^{-1}) measured in deposited sediment at five study sites along Okeover Stream, New Zealand on one occasion in May 2003. ANZECC ISQG-low (Interim Sediment Quality Guideline) is equivalent to a "trigger value" whereas ISQG-high indicates high ecological impact (ANZECC 2000). Values in bold exceed (ISQG-low) trigger values.

	O2	O3	O4	O5	O6	ISQG-low	ISQG-high
Cd	0.05	0.26	0.04	0.03	0.08	1.5	10
Cu	23.9	**200**	46.7	23	26.2	65	270
Pb	42.9	**109**	37.8	**51.5**	**180**	50	220
Zn	105	**332**	128	113	96.4	200	410

per m^2, respectively) (Nested ANOVA: Stream $F_{2,6} = 14.007$, $P = 0.005$; Fig. 2). However, there was no significant difference between densities in each sample within sites (Nested ANOVA: Sample (Site) $F_{6,18} = 0.071$, $P = 0.79$). When whole invertebrate communities were compared, Waimairi Stream and Avon River had higher relative abundances of gastropods (*Potamopyrgus antipodarum* and *Physella acuta*), whereas Okeover Stream was dominated by the amphipod *Paracalliope fluviatilis* (34%) and several other taxa (21%), including flatworms, the bivalve *Musculium*, and freshwater mites, which were more poorly represented in the other two waterways (Fig. 3). In all the streams, caddisflies made up less than 2.7% of the total macroinvertebrate community (Fig. 3).

Longitudinal patterns along Okeover Stream

Water chemistry and physical conditions changed little along the 1200 m reach of Okeover Stream (Table 4). pH was circum-neutral, and DO remained high at all sites. Similarly, there was very little variation in water temperature, alkalinity, PO_4-P, or conductivity along Okeover Stream, and no significant differences between sites in these parameters. Turbidity was generally very low at baseflow, with the most downstream sites having the highest turbidity (ANOVA: $F_{5,6} = 5.37$, $P = 0.03$; O1 LSD $P < 0.05$; Table 4). Conductivity and light levels were consistently high across all sites (Table 4).

The substrate index increased downstream, as the streambed was composed of cobbles and fine silt upstream, but became dominated by cobbles downstream. There was no statistical difference in the amount of inorganic sediment on stone surfaces between the six sites (Table 4). In contrast, deposited organic sediment and periphyton biomass differed significantly between sites along Okeover Stream (ANOVAs: $F_{4,20} = 2.93$, $P = 0.046$ and $F_{4,20} = 7.52$, $P = 0.001$, respectively). Significantly more deposited organic sediment was found on cobbles at O4 than at other sites (LSD $P < 0.01$). O2 had the highest periphyton biomass (LSD $P < 0.01$) (Table 4).

Intensive sampling of DO and temperature indicated little variation between sites, except that O3 had water temperatures 1–1.5°C higher than at other sites. This could be attributed to air-conditioning

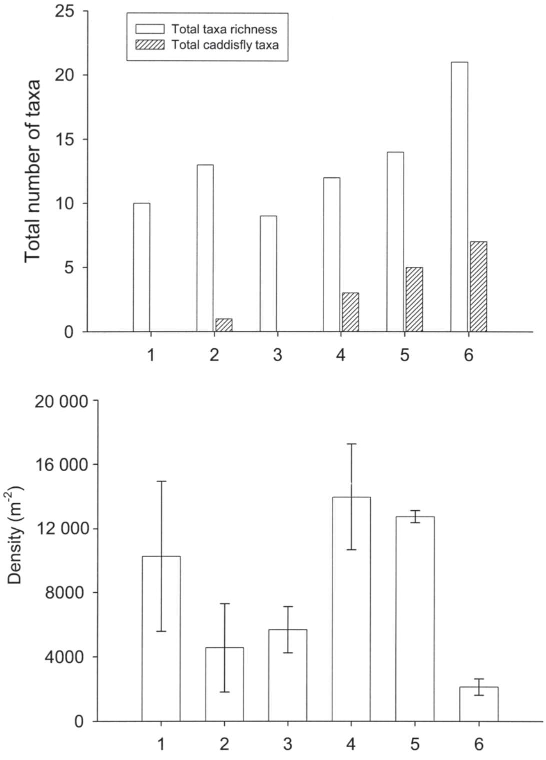

Fig. 4 Total taxonomic richness and total number of Trichoptera taxa (top), and benthic invertebrate densities (± 1SE; $n = 3$) (bottom) at six sites along Okeover Stream, north-west Christchurch, New Zealand, between November and December 2002.

inputs, piped into Okeover Stream directly upstream of the sampling site (O3). DO and temperature ranged from 8.5 to 9.5 mg/litre and 13.5–14.9°C, respectively along Okeover Stream.

Heavy metal concentrations in the sediment varied between sites, with no obvious trends being found. Cd concentrations were generally low, although O3 had 3–9 times higher levels than the other sites (Table 5). Cu, Pb, and Zn concentrations at O3 consistently exceeded ISQG-low trigger values, whereas at O5 and O6, ANZECC values were exceeded for Pb only (Table 5).

Despite few differences in physical and chemical conditions being found, taxonomic richness generally increased downstream. This trend was broken at site O3, which had fewer taxa than either its neighbouring upstream or downstream sites (Fig. 4). A similar trend was shown by caddisflies, which were present in very low numbers at O2, absent from O3, and gradually increased to seven taxa at O6 (Fig. 4). Invertebrate densities were highly variable among sites and were lowest at O6, however this variability was not statistically significant (ANOVA: $F_{5, 12} = 2.45$; $P = 0.09$). When entire macroinvertebrate communities were examined, site O1 was almost entirely dominated by oligochaete worms (Fig. 5), but a more diverse community, including amphipods, snails, dipterans, and caddisflies, was present downstream (Fig. 5). At O6, caddisflies made up 14.5% of the benthic invertebrate community, compared with <0.5% elsewhere (Fig. 5).

DISCUSSION

Restoration efforts in Okeover Stream have centred on improved riparian planting and aquatic macrophyte management, with in-stream habitat modification undertaken in two separated reaches c. 50–150 m long. Although efforts to date may have improved in-stream and riparian conditions in limited reaches of the stream (such as increased water flows and improved habitat heterogeneity, as well as enhanced stream shading and sediment traps), they have not resulted in significant improvements along the 1200 m reach within the university campus. Interestingly, the results of our longitudinal study indicate that within Okeover Stream there is a distinct increase in species richness downstream, and that several factors may be limiting the recovery of benthic communities further upstream.

Urban streams are frequently exposed to three main modifiers: altered hydrological regimes, extensive channel and riparian modification, and inputs of pollutants and sediment. The urban waterways in this study were subjected to all three impacts. Okeover Stream receives at least 40 stormwater and air-conditioning inputs and because of this, the headwaters of this stream are subject to highly variable discharge. Although we have no specific data for either Waimairi Stream or Avon River, both of these systems receive stormwater from surrounding roads. All three waterways have box-culvert and channelised reaches, as well as some natural meandering sections. Thus, it is probably not surprising that we found few differences in the physical and chemical conditions of these three Christchurch waterways. In general, pH was circum-neutral, and specific conductivity was higher than levels recorded in 100 similar sized streams in a variety of rural catchments throughout the South Island (Harding et al. 1997). Although alkalinity differed between

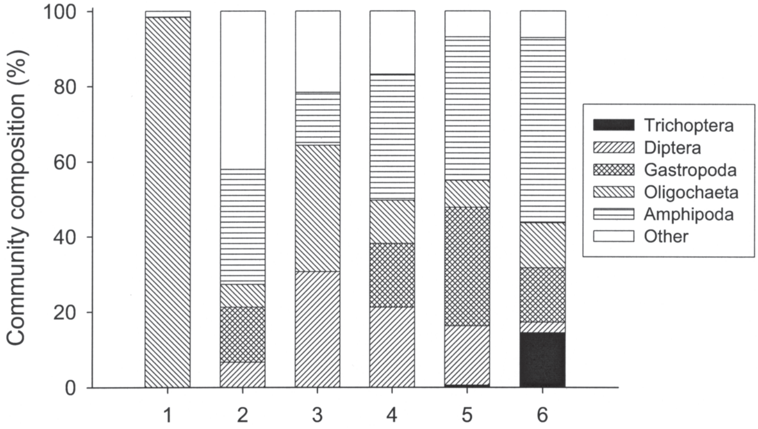

Fig. 5 Relative abundance (%) of six taxonomic groups at six sites along Okeover Stream, north-west Christchurch, New Zealand, in November 2002.

streams, we suspect the differences were so small as to have little ecological significance.

Thus, despite marked differences in the restoration efforts undertaken in Okeover Stream, compared with Waimairi Stream and the upper reaches of the Avon River, there were no significant differences in either taxonomic richness or the numbers and kinds of EPT taxa. There were, however, obvious differences in the composition of benthic communities. Okeover Stream was dominated by amphipods, whereas the other two streams were dominated by gastropod snails. We do not have an adequate explanation for this compositional difference between the streams, but it may in part be a result of the presence of isolated reaches within Okeover Stream, which are dominated by seasonal macrophytes, possibly favouring amphipods, as found elsewhere in Canterbury (Stout 1968; Marshall 1973).

Sedimentation is potentially a major limiting factor in the recovery of the benthic community in Okeover Stream. The upper reaches of Okeover Stream have several pools that act as sediment repositories and, as a result, downstream riffles and runs are exposed to intermittent clogging from sediment entrained during high flows. Furthermore, flow is negligible at times in the upper reaches of Okeover Stream. Thus, any sediment that has accumulated is unlikely to be flushed downstream. Urban streams are frequently characterised by high sediment levels (Suren 2000; Paul & Meyer 2001), and fine inorganic sediments have been shown to reduce stream habitat quality (Timperley & Kuschel 1999) as well as negatively effect benthic invertebrates in New Zealand (Winterbourn et al. 1971; Quinn & Hickey 1990; Jowett et al. 1991).

Another factor limiting macroinvertebrate recovery in Okeover Stream may be the presence of relatively high concentrations of heavy metals in deposited sediments. Heavy metals are known to be toxic to some aquatic organisms (Hickey & Clements 1998; Beeson et al. 1999; Hickey 2000), and we detected relatively high concentrations of Cu, Pb, and Zn in stream sediments, with levels at O3 exceeding the ISQG trigger values (ANZECC 2000). These guidelines are predominantly based on international data, as there has been relatively little research on ecological impacts of heavy metals in New Zealand (but see Hickey & Clements 1998). The ISQG-low and high values indicate the likelihood (statistical probability) of potential ecological effects if the contaminants exceed this value. Therefore, there is a 10% probability that heavy metal concentrations in the sediment at O3 will cause ecological damage. The presence of high metal concentrations at O3 corresponded with a decline in species richness, including an absence of caddisflies (EPT taxa). High levels of heavy metals have been found to have lethal effects on several freshwater invertebrates in New Zealand (Hickey & Clements 1998), including the cased caddis *Olinga feredayi*, the common mayfly *Deleatidium* spp., and the amphipod *Paracalliope fluviatilis* (Hickey 2000). The increase in total and caddis taxonomic richness downstream corresponded with a drop in heavy metal concentrations in the sediment. However, the presence of heavy metals alone does not explain the longitudinal change in richness in Okeover Stream, as there is only a very subtle change in heavy metal concentrations in sediments along Okeover, with the exception of O3.

The physical nature of urban streams and the surrounding catchments may in themselves present barriers to recolonisation by aquatic invertebrates.

We believe that the presence of patchy accumulations of sediment, and toxic heavy metals in Okeover Stream does not fully explain the higher overall diversity and the presence of more caddisfly taxa downstream. Some other variable or condition not assessed in this study seems to be at work. For example, roads dissect many urban streams and within north-west Christchurch most streams are piped through constricted culverts beneath these roads. We suggest that these culverts may act as physical barriers to upstream flight by adult aquatic insects, which consequently may fail to lay eggs in upstream reaches. If so, this could explain the absence of larvae whose upstream movements will also be hindered by the presence of culverts. Malaise trapping above and below road culverts lends some support to this hypothesis (authors' unpubl. data).

In summary, the results of our study indicate that restoration efforts in urban streams may not necessarily result in measurable improvements to stream ecosystems unless a sound understanding of the factors limiting ecological recovery are obtained.

ACKNOWLEDGMENTS

We thank Angus McIntosh, Hamish Greig, Leanne O'Brien, and others involved with the Freshwater Ecology Research Group, University of Canterbury, for their helpful discussions. Thanks also to Mike Winterbourn and the anonymous referees whose comments greatly improved an early draft of this manuscript. T. J. Blakely received a summer scholarship, provided by Rachel Barker and the Christchurch City Council.

REFERENCES

ANZECC 2000: National water quality management strategy—Australian and New Zealand guidelines for fresh and marine water quality. Australian and New Zealand Environment and Conservation Council, and Agricultural & Resource Management Council of Australia and New Zealand.

APHA 1980: Standard methods for the examination of water and wastewater. Washington D.C., American Public Health Association.

Beasley, G.; Kneale, P. 2002: Reviewing the impact of metals and PAHs on macroinvertebrates in urban watercourses. *Progress in Physical Geography 26*: 236–270.

Beeson, D. R.; Powell, J. M.; Lewis, M. C. 1999: Effects of pollution on freshwater organisms. *Water Environment Research 71*: 1092–1100.

Campbell, I. C. 1978: A biological investigation of an organically polluted urban stream in Victoria. *Australian Journal of Marine and Freshwater Research 29*: 275–291.

Charbonneau, R.; Resh, V. H. 1992: Strawberry Creek on the University of California, Berkeley campus: a case history of urban stream restoration. *Aquatic Conservation 2*: 293–307

Christchurch City Council 2003: Waterways, Wetlands and Drainage Guide. Part A: Visions. Christchurch, New Zealand, Christchurch City Council.

Collier, K. J.; Cooper, A. B.; Davies-Colley, R. J.; Rutherford, J. C.; Smith, C. M.; Williamson, R. B. 1995: Managing riparian zones: a contribution to protecting New Zealand's rivers and streams. Concepts. Wellington, Department of Conservation.

Dendy, A. 1900: The plants and animals of Canterbury. *In*: Association, N. Z. N. *ed*. Canterbury—old and new, 1850–1900: a souvenir of the jubilee. Christchurch, New Zealand, Whitcombe & Tombs. Pp. 178–197.

Garie, H. L.; McIntosh, A. 1986: Distribution of benthic macroinvertebrates in a stream exposed to urban runoff. *Water Resources Bulletin 22*: 447–455.

Grimm, N. B.; Grove, J. M.; Pickett, S. T. A.; Redman, C. L. 2000: Integrated approaches to long-term studies of urban ecological systems. *Bioscience 50*: 571–584.

HACH 1990: Procedures manual DR/200 Spectrophotometer. Loveland, United States, HACH.

Harding, J. S.; Winterbourn, M. J.; McDiffett, W. F. 1997: Stream faunas and ecoregions in South Island, New Zealand: do they correspond? *Archiv für Hydrobiologie 140*: 289–307.

Hickey, C. W. 2000: Ecotoxicology: laboratory and field approaches. *In*: Collier, K. J.; Winterbourn, M. J. *ed*. New Zealand stream invertebrates: ecology and implications for management. Christchurch, New Zealand Limnological Society. Pp. 313–343.

Hickey, C. W.; Clements, W. H. 1998: Effects of heavy metals on benthic macroinvertebrate communities in New Zealand streams. *Environmental Toxicology and Chemistry 17*: 2338–2346.

Hogg, I. D.; Norris, R. H. 1991: Effects of runoff from land clearing and urban development on the distribution and abundance of macroinvertebrates in pool areas of a river. *Australian Journal of Marine and Freshwater Research 42*: 507–518.

Jowett, I. G.; Richardson, D. M. 1990: Microhabitat preferences of benthic invertebrates and the development of in-stream flow-habitat models for *Deleatidium* spp. *New Zealand Journal of Marine and Freshwater Research 24*: 19–30.

Jowett, I. G.; Richardson, J.; Biggs, B. J.; Hickey, C. W.; Quinn, J. M. 1991: Microhabitat preferences of benthic invertebrates and the development of generalised *Deleatidium* spp. habitat suitability curves, applied to four New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research 25*: 187–200.

Kondolf, G. M.; Micheli, E. R. 1995: Evaluating stream restoration projects. *Environmental Management 19*: 1–15.

LeBlanc, R. T.; Brown, R. D.; FitzGibbon, J. E. 1997: Modelling the effects of land use change on the water temperature in unregulated urban streams. *Journal of Environmental Management 49*: 445–469.

Lenat, D. R.; Crawford, J. K. 1994: Effects of land use on water quality and aquatic biota of three North Carolina piedmont streams. *Hydrobiologia 294*: 185–199.

Marshall, J. W. 1973: A benthic study of the Avon Spring Stream, Christchurch. *Mauri Ora 1*: 79–90.

Maxted, J. R.; Evans, B. F.; Scarsbrook, M. R. 2003: Development of standard protocols for macroinvertebrate assessment of soft-bottomed streams in New Zealand. *New Zealand Journal of Marine and Freshwater Research 37*: 793–808.

McConchie, J. A. 1992: Urban hydrology. *In*: Mosley, M. P. *ed*. Waters of New Zealand. Wellington, New Zealand Hydrological Society Inc. Pp. 335–364.

O'Brien, L.; Barker, R. M.; Weston, J. 1998: Restoring the waterways within the University of Canterbury. A partnership plan: analysis and preliminary proposals. Christchurch, New Zealand, Christchurch City Council.

Paul, M. J.; Meyer, J. L. 2001: Streams in the urban landscape. *Annual Review of Ecology and Systematics 32*: 333–365.

Pedersen, E. R.; Perkins, M. A. 1986: The use of benthic invertebrate data for evaluating impacts of urban runoff. *Hydrobiologia 139*: 13–22.

Pratt, J. M.; Coler, R. A.; Godfrey, P. J. 1981: Ecological effects of urban stormwater runoff on benthic macroinvertebrates inhabiting the Green River, Massachusetts. *Hydrobiologia 83*: 29–42.

Quinn, J. M.; Hickey, C. W. 1990: Characterisation and classification of benthic invertebrate communities in 88 New Zealand rivers in relation to environmental factors. *New Zealand Journal of Marine and Freshwater Research 24*: 387–409.

Riley, A. L. 1998: Restoring streams in cities. A guide for city planners, policy makers, and citizens. Washington D.C., Island Press.

Robb, J. A. 1980: A biological survey of rivers in the metropolitan Christchurch areas and outlying districts. The Avon, Heathcote and Styx Rivers and their tributaries. Christchurch, New Zealand, Christchurch Drainage Board.

Roy, A. H.; Rosemond, A. D.; Paul, M. J.; Leigh, D. S.; Wallace, J. B. 2003: Stream macroinvertebrate response to catchment urbanisation (Georgia, U.S.A.). *Freshwater Biology 48*: 329–346.

Scarsbrook, M. R. 2000: Life-histories. *In*: Collier, K. J.; Winterbourn, M. J. *ed*. New Zealand stream invertebrates: ecology and implications for management. Christchurch, New Zealand Limnological Society. Pp. 76–99.

Stout, V. M. 1968: The invertebrate fauna of rivers and streams. *In*: Knox, G. A. *ed*. The natural history of Canterbury, Royal Society of New Zealand. Wellington, Reed. Pp. 471–97.

Suren, A. M. 1993: The effects of urbanisation on the invertebrate fauna of the Avon and Heathcote Rivers. *New Zealand Freshwater Miscellaneous Report 36*. Christchurch, New Zealand, National Institute of Water and Atmospheric Research.

Suren, A. M. 2000: Effects of Urbanisation. *In*: Collier, K. J.; Winterbourn, M. J. *ed*. New Zealand stream invertebrates: ecology and implications for management. Christchurch, New Zealand Limnological Society. Pp. 260–88.

Suren, A. M.; Snelder, T. H.; Scarsbrook, M. R. 1998: Urban stream habitat assessment method (USHA). Client report CHC98/60, NIWA. Christchurch, New Zealand.

Timperley, M.; Kushcel, G. 1999: SWAT's up Doc? The effects of storm water and transport on urban streams and estuaries. *Water & Atmosphere 7*: 22–5.

Underwood, A. J. 1997: Experiments in ecology: their local design and interpretation using analysis of variance. Cambridge, United Kingdom, Cambridge University Press. Pp. 376–7.

Wallace, J. B.; Anderson, N. H. 1996: Habitat, life-history, and behavioral adaptations of aquatic insects. *In*: Merritt, R. W.; Cummins, K. W. *ed*. An introduction to the aquatic insects of North America. Dubuque, Iowa, Kendall/Hunt. Pp. 41–73.

Whiting, E. R.; Clifford, H. F. 1983: Invertebrates and urban runoff in a small northern stream, Edmonton, Alberta, Canada. *Hydrobiologia 102*: 73–80.

Williams, D. D. 1980: Some relationships between stream benthos and substrate heterogeneity. *Limnology and Oceanography 25*: 166–172.

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

Winterbourn, M. J.; Alderton, P.; Hunter, G. G. 1971: A biological evaluation of organic pollution in the lower Waimakariri River system 1970–71. *New Zealand Marine Department, Fisheries Technical Report No. 67.* Wellington, New Zealand.

Winterbourn, M. J.; Gregson, K. L. D.; Dolphin, C. H. 2000: Guide to the aquatic insects of New Zealand. 3rd ed. Auckland, Entomological Society of New Zealand.

Zar, J. H. 1999: Biostatistical analysis. 4th ed. New Jersey, Prentice-Hall Inc.