URBAN STREAM ECOLOGY – IMPERVIOUSNESS AND STREAM INVERTEBRATE COMMUNITY STRUCTURE

by

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Chapter 1 General Introduction

Cities embody the environmental damage caused by modern civilisation; therefore it is not surprising that urbanisation has been a topic of great discussion and concern over the last decade (UNFPA 2007). Globally, cities are predicted to experience accelerated growth as more and more of the world's population migrate to urban areas. This alarm is reflected in the main theme of the 2007 United Nations' State of the World Population Report: Urbanisation (UNFPA 2007). Urban growth is also predicted to have the single largest influence on economic development (UNFPA 1996, UNFPA 2007). Apart from the socioeconomic issues of increased urbanisation (UNFPA 1996, Brueckner 2000, Paul and Meyer 2001), there are equal concerns on environmental grounds as agricultural or forested land is developed to house increasing populations.

Urbanisation is often associated with loss of aquatic biodiversity (Paul and Meyer 2001, Allan 2004). Natural and agricultural ecosystems are converted to urban landscapes that affect the functioning of aquatic ecosystems draining that land (Moore and Palmer 2005, Dow 2007) and the extent of impact depends on how urban locales expand (UNFPA 2007). In light of this, stream ecologists are faced with increasing pressure to understand the ecology of these impacted ecosystems. This includes: How is ecosystem functionality affected? How do communities respond and/or cope with this type of land use change? How does urbanisation affect trophic or species diversity? Even with a reduction in diversity, will impaired stream ecosystems be able to maintain ecosystem processes? Are there limits that should be set for urban development? Answers to these questions are necessary and urgent since urbanisation is an irreversible process with potentially severe impacts on ecosystems (Brabec *et al*. 2000, Booth 2005).

New Zealand's urbanisation trends are similar to those globally and similar concerns about the health of our aquatic systems apply. New Zealand has surpassed several countries in its rate of urbanisation and is ranked close to countries such as North America and Australia (UNPD 2008). Currently, more than 80 percent of New Zealand's population live in urban areas (UNPD 2008). Urban growth, independent of economic factors, is predicted to steadily increase (STATS

NZ, 2009). These statistics indicate that there are major disparities in the level of urbanisation in different regions of New Zealand. Some regions are faced with higher levels of urbanisation than others. The Greater Wellington Region of the lower North Island in New Zealand is one such region.

The main aim of the study was to investigate the effect of urbanisation on stream invertebrate community structure in the Greater Wellington Region by comparing communities in streams with differing levels of urbanisation.

The urban area of Wellington is approximately 8.13 km^2 with an estimated residential population of about 466,300 (STATS NZ, 2009). To investigate the effect of urbanisation it was necessary to identify a measure of urbanisation that: (i) could be easily defined, (ii) was practicable and (iii) captured the level of urbanisation at a particular site. There are several variables and methods already available in the literature to investigate the effects of urbanisation (Brabec *et al*. 2002), such include demographic variables: namely residential land class, population density, number of households and populations per acre (see Brabec *et al.* 2002). However, one of the unifying themes coming out of urban research is percent catchment 'imperviousness' (PCI) (Schueler 1994). PCI is simply the imperviousness or area of impervious surfaces, such as roads and rooftops that an area contains, and defining its percentage is usually done using aerial photographs or remote sensing. PCI was hence used as it met the three criteria above while preserving an important concept of urbanisation as defined by Schueler (1994) and Paul and Meyer (2001). While the urbanisation concept involves the phenomenon of migration to urban centres, one also essentially thinks of concrete structures for living and transport (Marzluff *et al*. 2008). PCI embodies this essence of urbanisation while circumventing the problems that one would face trying to define 'urban'. Some studies have used demographic variables such as residential land class, population density, and number of households or populations per acre while others have used political delineations to define 'urban' (Brabec *et al.* 2002). PCI is also an easily defined unit of measurement against which the ecological effect of urban development on aquatic systems can be measured (Schueler 1994, Brabec *et al*. 2002). Therefore, while several alternative variables could be used to quantify and define urbanisation, PCI was chosen as it assesses a possible mechanism of impact.

Imperviousness is particularly relevant to aquatic systems. Impermeable landscape cover is the most important feature of PCI (Schueler 1994) and this irreversible and sometimes rapid conversion of land to impermeable landscape cover is believed to be the precursor for stream impairment (Suren 2000, Paul and Meyer 2001). PCI causes impairment primarily through changes to hydrology (Leopold 1968) via decreased infiltration and increased surface runoff over impermeable surfaces. The construction of (i) large impervious surfaces for living and transport purposes (Schueler 1994, Chadwick *et al.* 2006, Marzluff *et al*. 2008) and (ii) associated stormwater drainage systems that serve to convey rainfall away from urban areas (Schueler 1994, Niemczynowicz 1999, Walsh, Fletcher and Ladson 2005) appear to be the primary mechanisms that alter stream hydrology.

With increased imperviousness and consequent reduction in the amount of vegetation cover, a high volume of water is moved off the land in a much shorter interval of time (termed flashiness by Poff *et al.* 1997). Thus, a high degree of flashiness is one of the most prominent characteristics of urban streams when compared to streams less urbanised (Leopold 1968). Flashy streams have rapid rates of change in flow while less urbanised streams have slower rates of flow change. Variables such as upstream slope can also affect flashiness (Kennen et al. 2008) as, theoretically, there is greater potential energy acting on per unit volume of water flowing from a higher sloped stream providing all other landscape variables stay constant. Hydrologically, flashiness is represented as higher peak and bankfull discharges, reduced lag time between peak rainfall and peak discharge, and decreased baseflows (Booth and Jackson 1997, Poff *et al.* 1997, Elliot *et al*. 2004, Dow 2007). Such flashy flows can be typified by the relatively linear design of former channelworks and the unabated flow of runoff from associated stormwater drainage systems (Booth 1991, Riley 1998, Walsh, Fletcher and Ladson 2005).

The sheer force of large volumes of stormwater being conveyed to streams and rivers (almost instantaneously) can drastically alter the stream's physical nature particularly if channel stability is low (Bledsoe and Watson 2001). Channel erosion, incision and widening drive changes in channel morphology and habitat structure as consequence of increased bankfull flows associated with high PCI (Hammer 1972, Booth and Jackson 1997). These major physical changes can significantly affect the biotic assemblages (Townsend and Hildrew 1994, Vinson and Hawkins 1998). In-stream physical structures such as culverts can also affect aquatic insect communities (Blakely and Harding 2005, Blakely *et al.* 2006).

Urban runoff does not only drastically affect the stream's physical nature but water quality is also impacted. In New Zealand's urban streams, nitrates and phosphates have been identified as one of the major sources of pollution due to broken or leaking sewer systems (MfE 2007). This type of enrichment as well as other sources of pollution such as heavy metals, sediment and petroleum based pollutants can affect feeding guilds, biotic diversity and trophic structure of instream biota (Lenat and Crawford 1994, Sutherland 2000, Walsh *et al.* 2001, Alberti 2007). While some types of heavy metals (Goodyear and McNeil 1999 Quinn *et al.* 2003, Demirak *et al*. 2005, Yang *et al.* 2006) and petroleum based pollutants (Barron *et al.* 2004), have been more intensely researched than others, the compounding effect of the triad of heavy metals and hydrocarbons sorbed to sediment (Hoffman *et al.* 1984, Sutherland 2000) can be traced to automobiles and their associated motorways and parking-lots (Maltby *et al.* 1995, Sutherland 2000, Mahler *et al.* 2005). Thus as the transport component of PCI increases, facilitating urban living purposes, the cumulative effects of the aforesaid factors becomes more obvious and pronounced but difficult to disentangle.

High PCI has been linked with stream degradation (Schueler 1994, Arnold and Gibbons 1996, Suren 2000). For example, studies looking at the effect of PCI on stream invertebrate communities have found decreased diversity at moderate intensities of PCI ranging from 10 to 25 percent impervious catchment (Jones and Clarke 1987, Schueler 1994, Maxted 1996). Others have found significant changes in communities at ranges as low as 5-10 PCI (Klein 1979, May *et al*. 1997, Morse *et al.* 2003). Generally, increasing PCI leads to the declines in trophic diversity and changes in community composition (Allibone *et al.* 2001, Hall *et al.* 2001, Walsh *et al.* 2001, Tabit and Johnson 2002). Suren (2000) has highlighted that within New Zealand there is a definite need to minimize future loss of stream biodiversity due to increasing trends in urbanisation. While there has been some focus on urban stream research, Blakely and Harding (2005) emphasize that within New Zealand there have not been many studies on benthic invertebrate communities of urban streams.

Therefore in addressing my overall aim:

To investigate the effect of urbanisation on stream invertebrate community structure in the Greater Wellington Region

The specific questions addressed are

(i) How do the potential drivers of PCI and average upstream slope (termed flashiness) interact to influence invertebrate community change?
   a. What are the relative contributions of these two potential drivers on invertebrate community change?
   b. Can PCI be used as a predictor variable for the state of invertebrate communities at a site?

(ii) What are the changes in species diversity to an artificial flashy event?
   a. If communities are affected, how long do they take to return to previous conditions?
   b. Does PCI influence communities' response to an artificial flashy event?

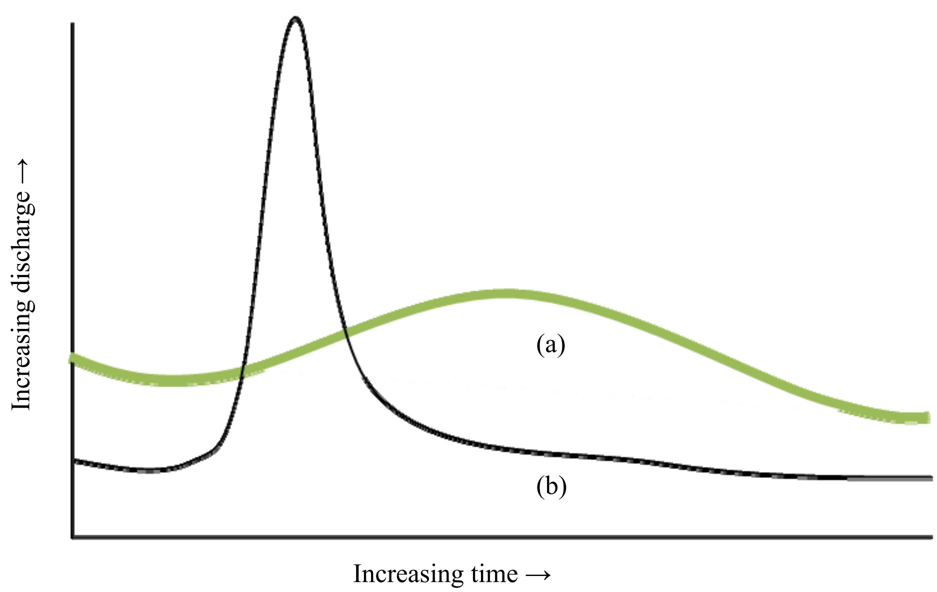
Chapter 2 The effect of Percent Catchment Imperviousness on stream invertebrate composition in Wellington, New Zealand.

Abstract

A measure of urbanisation is the percent of catchment imperviousness (PCI). Of the several associated habitat changes of PCI, stream hydrology is one of the most affected (Leopold 1968) with an increase in flashiness (Poff *et al*. 1997). Flashiness was also assumed to be influenced by average upstream slope (AvgUs) thus it is predicted that invertebrate communities in streams with steeper slope would be more affected by slope than less steep streams of similar PCI. Twenty one streams from the Greater Wellington region of the lower North Island were categorised as low (<5%), medium (5-20%) and high (>20%) PCI, and within each as having steep or shallow slope. Nine streams within the medium PCI group were examined for the potential effect of AvgUs (flashiness). No significant difference in invertebrate community composition was found between streams with high and low sloped sites. Examination of the effect of PCI on all streams indicated that there was a strong decline in biometrics such as species number, evenness, richness, the number of Ephemeroptera, Plecoptera and Trichoptera (EPT), Macroinvertebrate Community Index (MCI) and Quantitative Macroinvertebrate Community Index (QMCI) with increasing PCI. Invertebrate community composition also differed significantly between the low, medium and high PCI groups. Low PCI sites had communities dominated by Ephemeroptera taxa, namely *Deleatidium* and *Zephlebia*, with the caddisfly *Aoteapsyche* and *Archichauloides diversus* also numerically abundant. High PCI sites were dominated by gastropods, namely *Potamopyrgus antipodarum* with Orthocladiinae and Oligochatea also abundant. The presence of Ephemeroptera and Plecoptera was extremely rare in high PCI sites. The study suggests a threshold value of 15 to 20 PCI, at which a dramatic change in invertebrate community composition occurs.

Introduction

Increasing urbanisation is placing pressure on the ecological integrity of streams in many towns and cities (Arnold 1996, Paul and Meyer 2001, Gregory 2006). One of the most important driving forces of these pressures is the amount of the stream's catchment comprised of impervious materials (Brabec *et al*. 2002). Increased imperviousness or area of impervious surfaces, such as roads and rooftops, is one of the major consequences of increased urbanisation. This has been shown to lead to lower stream integrity: in that some ecosystem components are far from their natural condition, often with a reduction in pollution sensitive taxa such as Ephemeroptera, Plecoptera and Trichoptera (EPT) and a reduced diversity (Klein 1979, Lenat and Crawford 1994, Walsh *et al.* 2001, Stepenuck, Crunkilton and Wang 2002). Impervious surfaces are not intrinsically polluting but drive several changes as the percentage of catchment with impervious land use increases. The impacts associated with change in imperviousness may be roughly characterised into four parts (i) stream hydrology, (ii) habitat structure, (iii) water quality and (iv) biological components of streams (Schueler 1994, Paul and Meyer 2001)



<u>Figure 1</u>. Hydrograph showing change from pre-urbanisation (a) to post urbanisation (b). Increased peak flow, and more rapid, flashy flow is identified by a higher sharper peak while decreased groundwater and baseflow can be inferred from the reduced levels of discharge between both graphs. *Adapted from Leopold (1968)*

Hydrology determines several components of stream habitat (Clausen and Biggs 1997, Poff *et al*. 1997) and as a consequence, the structure of the biotic assemblages of streams (Biggs and Hickey 1994, Collier and Quinn 2003). Increasing imperviousness causes changes in hydrology (Leopold 1968) due to decreased infiltration, and increased surface runoff over impermeable surfaces. Overall, a high volume of water is moved off the land in a much shorter interval of time (i.e. flashiness). Poff *et al.* (1997) refers to flashiness as the rate of change in flow, that is, flashy streams have rapid rates of change in flow (Fig. 1). Hydrological changes often include higher peaks in discharge, a reduced lag time between peak rainfall and peak discharge, and decreased baseflows (Booth 1997, Poff *et al*. 1997, Elliot *et al*. 2004, Dow 2007). Change in stream hydrology is usually the precursor for consequent changes in channel morphology, stream habitat availability, water quality and thus potentially biotic integrity.

Changes in stream habitat structure and channel shape are more frequent and severe flooding occurs in more urban streams. Stream banks are eroded, and channel dimensions change which leads to habitat degradation (Dunne and Leopold 1978, Roberts 1989 as cited in Paul and Meyer 2001). Reduction in the extent of vegetation on land can increase sedimentation due to erosion which in turn leads to bed aggradation and overbank deposition. Channel erosion, incision and widening follow these events in order to accommodate the increased bankfull flows (Hammer 1972, Booth 1997). The physical heterogeneity of the streambed declines (Schueler, 1994) and there can be reductions in large woody debris (LWD) (Finkenbine *et al*. 2000), microform bed clusters (MBC) (Biggs *et al*. 1997), pool-riffle sequences (Gregory *et al*. 1994) and other refugia or habitats (Allibone *et al*. 2001). As stream habitats are changed biotic assemblages will also change (Southwood 1977, Townsend and Hildrew 1994, Vinson and Hawkins 1998).

Increased imperviousness can lead to stream warming (Galli 1991 as cited in Schueler 1994) as runoff flowing over heated impervious surfaces into streams causes an increase in temperature. This, coupled with the reduction or loss of riparian vegetation, increases the overall temperature regime of the stream. With increasing PCI, increases in nutrient load, heavy metal concentrations, suspended solids, sedimentation and pesticides have also been recorded as contributory factors that seem to have negative impacts on stream communities (Lenat and Crawford 1994, May *et al*. 1997, Suren 2000, Blakey and Harding 2005). Harmful contaminants such as polychlorinated biphenyls (PCBs), polycyclic aromatic hydrocarbons (PAHs), and petroleum-based aliphatic hydrocarbons (A.P.H.A. 2005) are also frequently detected in impacted streams or stream reaches with high PCI (Maltby *et al.* 1995, Van Metre 2000, Scoggins, McClintock and Gosselink 2007). Of all water quality parameters, higher concentrations of total phosphorous and nitrates is one of the most widespread effects of increasing urbanisation (Winger and Duthie 2000, Sandin 2005, MfE 2007).

It is not surprising that all of these changes lead to a change in the biotic communities of urbanised watersheds. Diversity metrics decline as impervious surface cover increases, 'Sensitive' taxa (e.g., Ephemeroptera, Plecoptera and Trichoptera taxa) decline in richness and density while 'Pollution-tolerant' taxa such as chironomids, snails, and oligochates increase (Morse *et al*. 2003). However, it is never clear from these studies which one or combinations of variables mentioned above have contributed to the change in the invertebrate communities. To mitigate or reduce the effects of urbanisation it is therefore preferable to know which of the potential drivers of community change are responsible. This study attempts to partition the effects of two potential drivers of invertebrate community change, percent impervious land use and increased flashiness and to investigate their relative contribution to potential changes in the biota.

Study sites

The Wellington urban area of North Island, New Zealand covers approximately 8.13 km^2 with an estimated residential population of about 466,300 (STATS NZ, 2009). Of this population, less than 5% live in rural areas. The study sites are within major catchments within the Greater Wellington Region: Kapiti Coast, Porirua, Hutt Valley and Wellington (Fig. 2). Dominant soil types and rainfall data at nearby gauged sites are shown in Appendix 2.2.

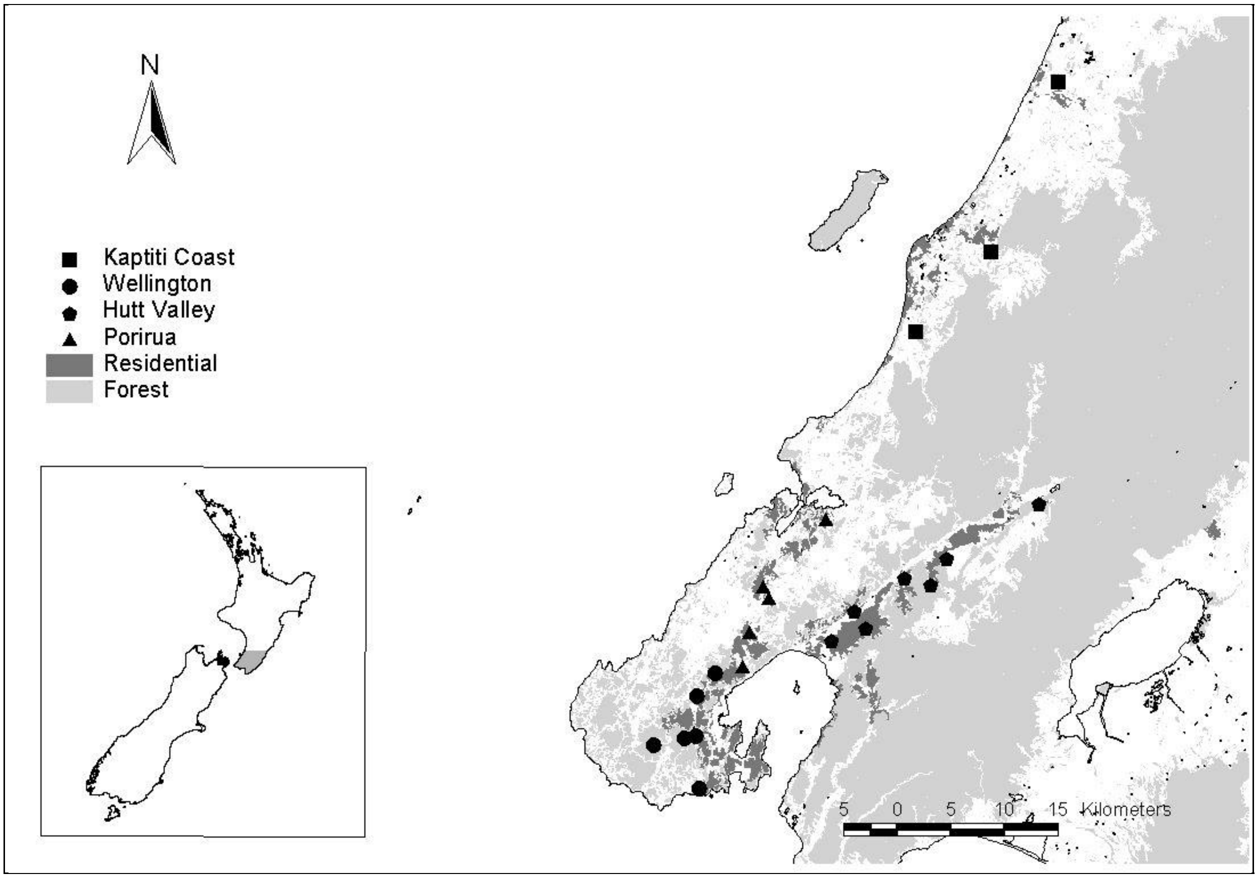


Figure 2. Map showing the 21 streams sampled in the Greater Wellington Region, New Zealand.

In land-use studies such as this, there are three possible methodological approaches: (i) the single catchment approach, where a gradient of increasing imperviousness is examined, (ii) the reference catchment approach that compares an urbanised catchment, to one with a desired condition, and (iii) the multiple catchment approach that considers gradients of imperviousness in several catchments (Paul and Meyer 2001). I used the latter approach as it helps avoid psuedoreplication considering the spatial distribution of streams I intended to study.

Usually metrics for describing streamflow characteristics (such as flashiness, duration, magnitude, and frequency) are calculated using flow data, (Poff and Ward 1989, Poff and Allan 1995, Richter *et al.,* 1996, Clausen and Biggs 1997). However, many of the proposed study streams chosen were relatively small and not gauged. Therefore, average upstream slope data (AvgUs) from the Freshwater Environments, New Zealand (FWENZ) dataset (Leathwick *et al.*

2008) was used as a surrogate of flashiness. The normalized values for upstream slope were extracted from within the three groups of Percent Catchment Imperviousness resulting in a design of PCI: low (<5%), medium (5-20%), and high (>20%) and Slope: low (<14.3) and high (>14.3). This resulted in streams being grouped into low PCI-low slope (K5, BLK), low PCI-high slope (K7, K9, SP1, DCL), med PCI-low slope (KW3, SSU, POT, CONV), med PCI-high slope (OWH2, PHU, TAK, WAT, WHU), high PCI-low slope (Train1, BM, W6, P6) and high PCI-high slope (SVL, N1). Accessibility to the site and availability of prior biomonitoring data limited the choice of sites that could fit into the proposed design. The effect of slope on species composition was only examined within the medium PCI group as this PCI group caused the greatest change in species composition and it was believed to be better to look at the effect of slope in one group rather than across all sites. See Appendix 2.1 for further site specific data.

Methods

Physicochemical characteristics

Physicochemical variables were measured at each site concurrently with macroinvertebrate sampling in the Austral summer between January and March, 2008. Conductivity, pH and temperature were measured with the respective ECScan™ handheld meters. One water sample was taken from each site and stored in a dark ice box for less than 48 hours before analysis for ammonia, nitrate, nitrite and phosphorus. These analyses were conducted using a colorimetric method on a flow injection analyser (APHA 2005). At five equidistant positions along the study reach of each stream, depth, width and flow velocity was measured at the thalweg using a ruler, tape measure and velocity head rod, respectively. The Wolman's pebble count (Wolman 1954) was used to quantify mean and median particle size from 100 randomly selected bed-particles at each of the sites. These 100 bed-particles were also used to calculate Substrate Index (SI) by categorising particles into groups: bedrock (>>300mm), boulders (>300 mm), large cobbles (300-128 mm), small cobbles^1 (128-90.5 mm), small cobbles^2 (90.5-64 mm), pebbles^1 (64-45.3mm), pebbles^2 (45.3mm-32 mm), pebbles^3 (32-22.6 mm), gravel^1 (22.6-16 mm), gravel^2 (16-11.3 mm), sand/silt^1 (11.3-8mm) or silt/sand^2 (<8 mm). The bedrock group was given the highest score. Scores declined exponentially until the sand/silt group. Frequencies of particles within each particle-size group were then multiplied by the group score and all products were summed to attain the SI for that stream, where a higher SI suggests greater frequencies of larger sized particles. Streambed stability was assessed using the bottom component of the Pfankuch's Index (Pfankuch 1975, Death and Winterbourn 1994). This involves summing the scores of 6 environmental variables related to the stream bottom (scored according to the perceived importance); lower scores conferring more stability for a stream. Substrate embeddedness was subjectively assessed at each site after moving the substrate, where 1= loosely packed and 4= tightly packed (see Death and Joy 2004). Riparian cover was assessed visually as percentage shaded area over the sampling reach. Upstream forest was extracted from the FWENZ dataset. Algae form (Periphyton type) and streambed sediment was also visually assessed as percentage coverage.

Biological Sampling

Macroinvertebrates

Benthic invertebrates were sampled during late summer (January-March 2008) using a Surber sampler ($0.1m^2$) fitted with a 250 μm mesh net. Five samples were taken from randomly selected locations within riffle habitats along a 60-90 m reach and material preserved in 70% ethanol. Macroinvertebrates were later removed by hand from sediment and debris in the laboratory and identified to the lowest practical taxonomic level using Winterbourn, Gregson and Dolphin (2006), McFarlane (1951) and NIWA guides (NIWA, 2009). Most insect taxa were identified to genus, with most non-insect taxa identified to order or class. Chironomidae (Diptera) were identified to subfamily or tribe. Subsampling with species of more than 200 individuals was conducted using a gridded tray or sample splitter box.

Periphyton

Biomass of periphyton was assessed by collecting five randomly selected stones at each macroinvertebrate sampling point. The stones were kept in a dark icebox before being frozen at the laboratory. Pigments were extracted by soaking the stones in 90% acetone for 24 hours at

4°C in a dark chiller. Absorbency was then measured with a Cary 50 Conc UV-Visible spectrophotometer™ and chlorophyll *a* and phaeophytin were calculated using the method of Steinman and Lamberti (1996).

Data Analysis

Several metrics were used to assess the effects of PCI and AvgUs, including Margalef's index (D = (S-1)/ln N where S = number of species and N = the total number of individuals collected) as a measure of species richness and the Shannon-Weiner index (H' = $-\sum_{i=1}^{S} \mathrm{p}_{\mathrm{i}} \ln \mathrm{p}_{\mathrm{i}} - \left[\frac{\mathrm{S}-1}{2\mathrm{N}}\right]$, where S = species richness, N = species abundance and p_i = relative abundance of each species, that is to say, the proportion of S made up of the ith species) as a measure of species evenness. Rarefied species richness was also calculated (see McCabe and Gotelli 2000). Relationships between biological indices and environmental variables were examined with PCA using Primer (Clarke and Gorley, 2006). A Spearman rank correlation was also done to identify the descriptors that correlated with the major axis (Legendre and Legendre 1998). This PCA major axis as well as biological indices of community structure, including species richness, evenness, MCI and QMCI (Stark 1993, Stark *et al.* 2001), were regressed against percent catchment imperviousness using Statistix 9. While the MCI and QMCI are not explicity structured for use in urban streams (Stark 1985, Stark 1993) the response of both were examined and included in the study (Table 2). MCI $= \frac{\text{site score}}{\text{no.of scoring taxa}}$ x $20.$ Where site score = the sum of individual taxon scores for all taxa present in a sample. QMCI = $\sum_{i=1}^{i=S} \frac{(n^i \; x \; a)}{N}$. Where S = the total number of taxa in the sample, n_i, is the number of individuals in the i-th scoring taxon, a, is the score for the i-th taxon, and N is the total number of individuals collected in the sample. Guidance on taxa scores can be found in Winterbourn, Gregson and Dolphin (2000). Non-metric multidimensional scaling analysis (MDS) using Bray-Curtis similarity was conducted on log-transformed biological data using Primer (Clarke 1993, Clarke and Warwick 1994, Clarke and Gorley 2006). A One Way Analysis of Similarities (ANOSIM) was also conducted on communities based on their groupings of imperviousness.

Results

Benthic invertebrate community

The invertebrate community of low (<5) PCI groups was dominated by Ephemeroptera, while medium (5-20) and high (>20) PCI groups were dominated by Diptera and Mollusca, respectively (Fig. 3a). In the sites with low PCI, benthic macroinvertebrate communities were dominated by the mayfly genera *Deleatidium* (34%) and *Zephlebia* (17%). The other common animals were dipterans (mainly Orthocladinae (6%)), the trichopteran *Aoteapsyche* (6%) and the megalopteran *Archichauloides diversus* (6%). The medium PCI group was dominated almost equally by molluscs (mainly the gastropod *Potamopyrgus antipodarum* (21%) and dipterans (mainly Orthocladiinae (25%)) while Oligochata (8%), *Deleatidium* (8%), amphipods (5%) and *Aoteapsyche* (5%) were a less dominant part of the community. In high PCI sites, more than half of the community comprised gastropods (*Potamopyrgus* (49%), *Physa* (4%) and *Gyraulus* (4%)) with Orthocladinae and Oligocheatea being the next most abundant (23% and 9% respectively).

In at least 19 of the 21 sites amphipods, chironomids (Orthocladinae), simulids (*Austrosimulium*), gastropods (*Potamopyrgus*) and oligochaetes were present. There were significant differences among the three PCI groups in the relative abundance of Coleoptera (H = 6.79, p = 0.034), Ephemeroptera (H = 12.6, p = 0.002), Mollusca (H = 5.38, p = 0.068), Oligochates (H = 8.99, p = 0.011), Plecoptera (H = 8.61, p = 0.014) and Trichoptera (H = 5.27, p = 0.072) (Fig. 3a). Sites grouped as having low PCI had highest Coleoptera and EPT abundances but lowest Crustacea, Diptera and Mollusca, while the converse trend was observed with high PCI sites: having highest Mollusca and Oligocheata abundances but lowest EPT and Coleoptera.

A two way nested ANOSIM test showed no significant difference between higher sloped and lower sloped sites within each of the three impervious groupings (R = 0.006, p = 0.44) but impervious groups did differ (R = 0.45, p = 0.001). In the medium PCI group there were no significant differences between high slope and low slope sites in invertebrate composition (Fig. 3b).

Figure 3 (a) Relative mean abundance of invertebrate Orders observed in the three PCI groupings: low (<5), medium (5-20), high (>20) in the Greater Wellington Region (where, n = 3, n = 9, n = 9, respectively. Errors bars show ±SE)

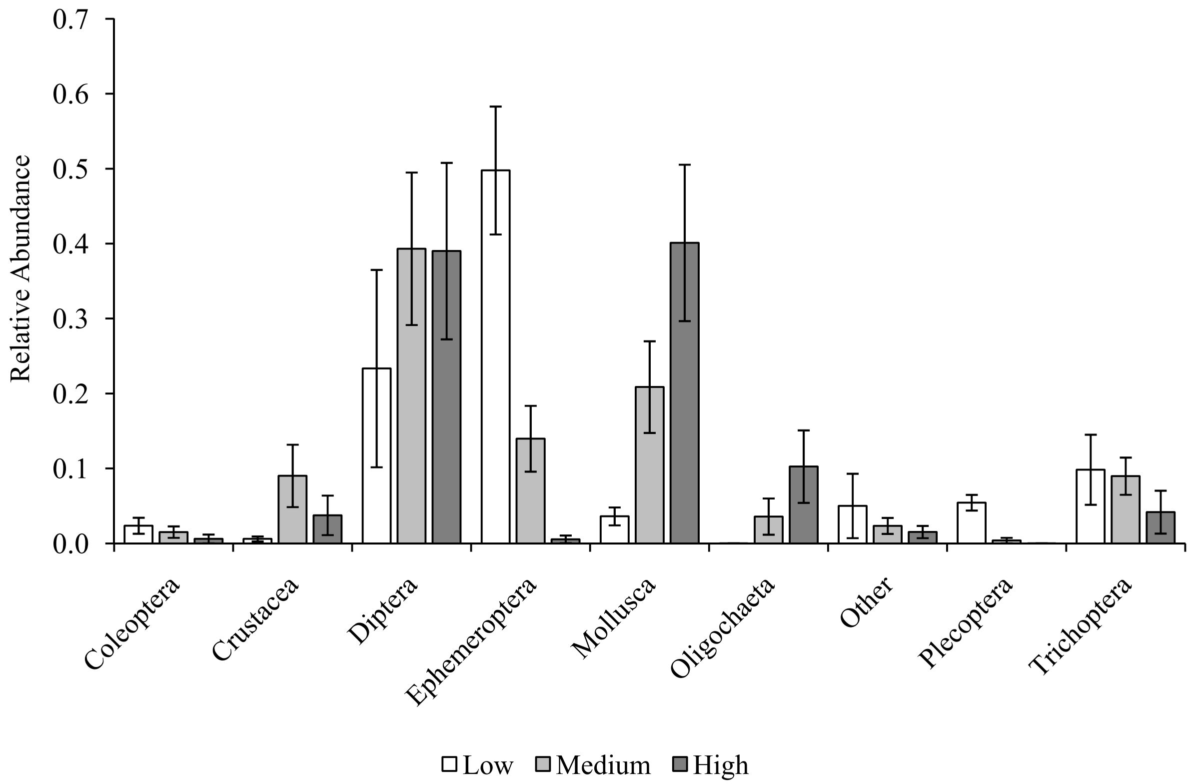


Figure 3 (b) Relative mean abundance of invertebrate Orders observed in high (n = 5) and low (n = 4) sloped sites within the medium PCI group in the Greater Wellington Region. (Error bars show ±SE)

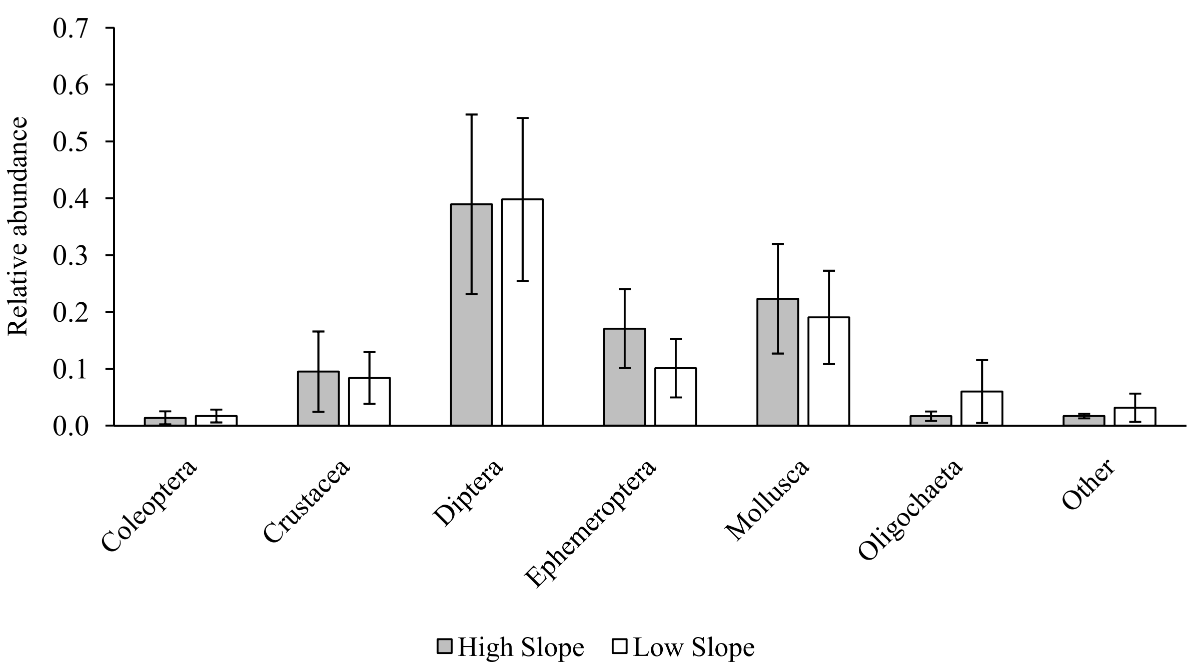
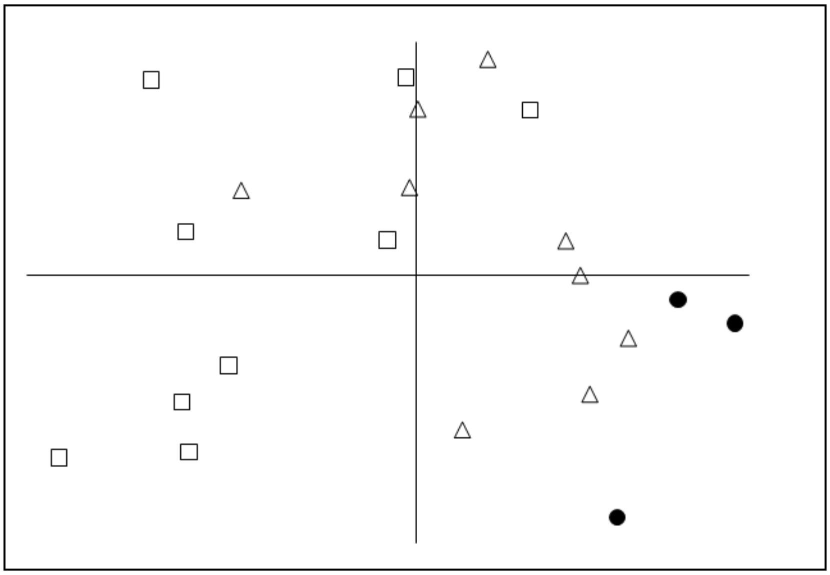


Figure 4. Non-metric Multidimensional Scaling (MDS) ordination of invertebrate abundance at sites with low (solid circles), medium (open triangles) or high (open squares) percent catchment imperviousness (PCI) groups. The cross is a scaled representation of the maximum score range of both axes in the analysis. The stress value is 16%.



Effect of Imperviousness (PCI) and Slope

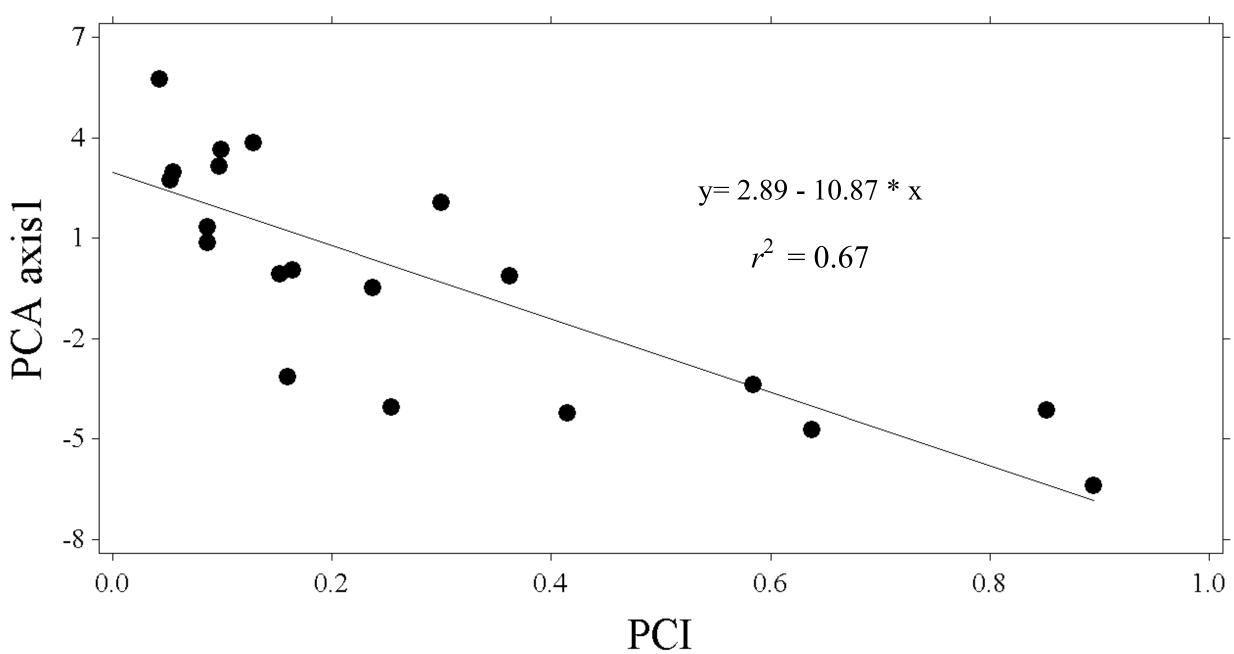
Most site physicochemical measures were similar between the three imperviousness classes (Table 1). MDS ordination grouped sites with higher PCI to the right of sites with low PCI (Fig. 4). There was a significant difference between invertebrate communities in the three Impervious groups (R = 0.45, p = 0.001). The largest difference was between the low and high PCI groups (R = 0.89, p = 0.005) while medium and high groups had the least difference (R = 0.27, p = 0.007)

<u>Table 1</u>. Range of environmental variables recorded at study sites in each of the three PCI groups.

	Low PCI	Medium PCI	High PCI
Chlorophyll-a (µg/cm)	4 – 5.7	2.2 – 16.8	2.4 – 21.1
Upstream slope (°)	15 - 25	9 – 24	0.4 – 19
Percent Natural Cover (%)	47 -93	15 – 76	1 – 63
Mean Depth (cm)	3-7	3 – 23	6 – 20
Mean Width (m)	0.8 – 1.1	0.5 – 3.1	0.7 – 3.1
Mean Velocity (m/s)	0.4 – 0.7	0.3 – 0.7	0.1 – 0.6
Temperature (°C)	15 – 15.6	15.5 – 19.2	15.5 – 20
pH	5.2 – 7.9	4.7 – 8.6	5.6 – 9.4
Conductivity (µS)	262 – 365	186 – 423	163 – 350
DIN (mg/l)	0.113– 0.2	0.01 – 2.23	0.03 – 1.49
Nitrite (mg/l)	<0.01 – 0.01	0.01 – 0.16	0.01– 0.06
Nitrate (mg/l)	0.113– 0.19	<0.01 – 2.22	0.02 – 1.67
Dissolved reactive Phosphorous (mg/l)	0.02 – 0.02	0.01 – 0.08	0.02 – 0.21
Pfankuch's Index	29 – 35	24.5 – 54	29 – 49
Means substrate size (mm)	45 – 64	32 – 90	8 – 128
Substrate Index	6.36 – 8.69	3.63 – 13.23	3.17 – 14.17
Site code (low upstream slope)	K5, BLK	KW3, SSU, POT, CONV	Train1, BM, W6, P6
Site code (high upstream slope)	K7, K9, SP1, DCL	OWH2, PHU, TAK, WAT, WHU	SVL, N1

The Principal components analysis identified three components that accounted for 64% of the variation in the habitat data. The first axis accounted for 39% of the variation and was positively linked with most biological indices. It was also negatively associated with chemistry variables, including pH, phosphate, nitrates and nitrites. Axis two only accounted for 12% of the variation and was positively associated with habitat measures such as depth, velocity, width, mean substrate size and substrate size index. Linear regression of PCA axis 1 revealed a strong inverse relationship ($r^2 = 0.67$) (Fig. 5). This also reflected a decline in biological metrics as PCA axis 1 was highly correlated with EPT number, QMCI, MCI, species number (S), species evenness (Shannon-Weiner Index (H')) and species richness (Margalef's Index (d) and rarified species richness) (Table 2). The Spearman rank correlation showed a strong correlation of PCA axis 1 with most biological metrics while PCI correlated most strongly with the nutrient measures such as phosphate, nitrates and nitrites (Table 2). Other habitat variables such as shade, Pfankuch Stability Index and Periphyton type (%Diatom, %Filamentous, % Bare) were not significantly correlated. Upstream slope exhibited a positive correlation with percent EPT (Table 2;).

Figure 5. Relationship between Percent Catchment Imperviousness and PCA axis 1 scores for habitat data collected at 24 Wellington urban streams (where $p < 0.005$).



<u>Table 2</u>. Spearman's Rank Correlation of biological and habitat measures and PCI, PCA 1 and Upstream slope. .

	PCA axis1	PCI	Upstream slope
PCI	-0.78****	---	---
Upstream slope (°)	0.51*	-0.51*	---
EPT abundance	0.88 ****	-0.70 ***	0.4 *
QMCI	0.84 ****	-0.68 ***	0.31
MCI	0.91 ****	-0.71 ***	0.38 *
Species number (S)	0.83 ****	-0.69 ***	0.33
Shannon-Weiner (H')	0.92 ****	-0.66 **	0.36
Margalef's index (d)	0.9 ****	-0.64 **	0.43 *
Rarefaction ES(20)	0.91****	-0.67**	0.36
Nitrite (mg/l)	-0.83****	0.74***	-0.42*
Nitrate (mg/l)	-0.4*	0.4*	-0.17
DIN (mg/l)	-0.41*	0.41*	-0.17
Dissolved reactive Phoshorus (mg/l)	-0.41*	0.66**	-0.48*
Upstream Forest	0.47*	-0.45*	0.69***
Shade	0.26	-0.05	-0.02
Temperature (°C)	-0.41*	0.27	-0.21
Mean velocity (cm/sec)	0.19	-0.43*	0.34
Pfankuch's Index	-0.16	0.17	-0.05

* $p < 0.05$, ** $p < 0.01$, *** $p < 0.001$, **** $p < 0.0001$. Upstream forest, Upstream Slope and PCI were extracted from the FWENZ dataset.

Discussion

Slope

In this study Percent EPT, EPT number, MCI and species richness all increased with slope (AvgUs). Assuming that upstream slope is a reach-scale surrogate of flashiness, the initial hypothesis was that upstream slope would more strongly influence benthic macroinvertebrate community structure particularly at lower PCI. However this was not the case. Flashiness has been identified in other studies as an important, ecologically relevant hydrological characteristic with considerable effect on aquatic assemblage structure (Kennen *et al*. 2008). Therefore, it was hypothesised that if upstream slope is indeed an appropriate surrogate of flashiness then its impact should be more apparent on assemblage structure (i) in streams with higher upstream slope and (ii) with lower PCI. The reasoning behind this hypothesis is that the pressures associated with increasing PCI should not be as profound within groups of low to medium PCI compared to high PCI groups (Schueler 1994, Morse *et al*. 2003), therefore it would be easier to observe upstream slope effects. However, upstream slope did not have a different effect in the medium PCI target group. It is likely that AvgUs may not be an appropriate surrogate of flashiness. Thus, the influence of upstream slope on macroinvertebrate community structure would be better investigated at low PCI by examining a wider cross section of sites. However, finding sites with low PCI and low upstream slope proved difficult in my initial endeavours and may be a bit of a problem as I observed that most modern cities and the phenomenon of urban sprawl are more likely to occur at lower altitudes and relatively flat topology: Hence the observance that upstream slope was less in more urban streams.

Streams that have been classified as stressed previously have been found to have PCI values >6 (Klein 1979, Schueler 1994, May *et al*. 1997, Morse *et al*. 2003), therefore the influence of PCI may overwhelm that of upstream slope even at these relatively low levels of PCI. As there were no hydrographs from these streams, this study did not directly investigate the efficacy of using upstream slope as a surrogate for flashiness. Nonetheless, investigating this variable within PCI groups of <5 PCI, and comparing upstream slope with hydrologically based metrics of flashiness is a promising area of future research. It would add to the advancement of methods for conducting synoptic studies about the stream ecology effects of urbanisation (McMahon *et al.* 2003).

Invertebrate community response

PCI was strongly linked with several biometrics of Wellington's urban stream macroinvertebrate community. The strong inverse relationship between most metrics and PCI illustrates the usefulness of impervious cover data for predicting urban stream integrity. It appears that there may be an inflection point of stream integrity at approximately 15-20 PCI (Fig. 5). This is similar to conclusions from other studies (Schueler 1994, May *et al.* 1997, Walsh *et al.* 2001, Wang *et al.* 2001, Stepenuck, Crunkilton and Wang 2002, Morse *et al.* 2003, Wang and Kanehl 2003). These studies show that there is an accumulation of biological effects once a certain proportion of the watershed becomes urbanised. Convergence on a single "threshold" value that is universally applicable is less realistic because of observed variability, from one locale to another. This can be due to differences in economic, social and more so environmental conditions (Brown *et al.* 2005, Potapova *et al.* 2005).

Nonetheless, this inverse relationship between invertebrate community structure and PCI, as well as the apparent threshold response of the invertebrate community to increasing PCI can be expected. There is an increase in more tolerant taxa with increasingly urbanised streams. This study highlights these trends and one of the main findings was the loss of EPT taxa in the more urbanised streams. These streams tended to support more tolerant taxa. *Deleatidium, Zephlebia* and *Aoteapsyche* were more common in streams with <20 PCI while those with >20 PCI were dominated by gastropods and chironomids (mainly Orthocladiinae) which are more commonly associated with polluted streams (Stark 1985, Stark 1998, Joy and Death 2003). The causal factors and processes underlying this change is not clear but may relate to increasing, nitrates, nitrites, phosphates and deteriorated water chemistry. While these were only measured once, a number of studies also relate water chemistry to increasing urbanisation and reduced biological metrics (Lenat and Crawford 1994, Ismail 1997, Van Metre *et al.* 2000, Bledsoe and Watson 2001). At the same time, absence of 'sensitive' species, such as *Deleatidium* may be more a result of change in physical instream habitat conditions (Suren *et al.* 2005) rather than water chemistry. It is likely that this is because hydrological changes are more important to invertebrate community structure in New Zealand streams (Clausen and Biggs 1997, Townsend and Scarsbrook 1997, Collier and Quinn 2003). The highest conductivity value (365µs) was found in Karori yet the pristine catchment vegetation possibly supported the high EPT number.

Deleatidium and *Potamopyrgus* are two contrasting examples of stream invertebrates belonging to the different PCI groups. A definite change in species traits occurred from streams with less to higher PCI. Both species share similar species traits (Doledec *et al*. 2006), for example, medium potential size (>10 to 20mm), weak dietary preferences, plurivoltine and scaper feeding but differ in body form and flexibility. *Deleatidium* seems to be more displaced in higher PCI sites than *Potamopyrgus* possibly due to an unfavourable flow regime and less suitable body form and method of respiration. Although body form or morphology has been criticized as a common and weak form of evidence linked to flow regime (Lytle and Poff 2004, Statzner 2008), this stark contrast in morphological species traits exhibited by different invertebrates in PCI groups may imply a strong influence of flow regime. Life history traits that are sensitive to varying degrees or aspects of urbanisation offer potential use in developing targeted management actions (Doledec *et al*. 2006) by highlighting potential causal mechanisms.

In conclusion PCI has been shown to have a very strong negative relationship with invertebrate community structure in streams in the Greater Wellington region of New Zealand and it is a useful index for predicting the response of stream invertebrates to urbanisation. Urban sprawl is likely to continue and the percent of urbanised catchment to increase therefore it seems more sensible that further research be directed to understand the functioning of urban stream ecosystems in relation to the degree of impervious catchment the stream drains.

<u>**Chapter 3 The effect of urbanisation on stream Macroinvertebrate recovery following an artificial disturbance**</u>

Abstract

Urban land use is an irreversible change and a disturbance that acts over an extended period, and is therefore a press disturbance (Lake 2000). Floods in contrast are natural pulse disturbances that often occur in streams regularly and are crucial for proper ecological functioning. It is postulated that changes in land use may affect the intensity of those natural flood events. To test the interaction between urbanisation and flood events an artificial hydrological disturbance was applied to streams with differing degrees of catchment urbanisation. Three groups of stream were characterised using percent catchment imperviousness (PCI) as none (0), low (<5) or medium (5-20) levels of urbanisation. The disturbance and degree of urbanisation strongly influenced benthic invertebrate community structure. Species abundance and richness were reduced immediately (1 hour) after the disturbance while species evenness was unchanged. At one week post-disturbance, species evenness and richness remained lower than the samples collected predisturbance. Species abundance was not affected by disturbance. Three weeks following the hydrological disturbance, all three diversity indices were similar to those predisturbance. The study also suggests that the disturbance acted differently over the three PCI groups in terms of species abundance, richness and evenness. Resistance (sample similarity immediately before and 1 hour after) was significantly different among the PCI groups. Resistance to the disturbance effect was higher at sites with higher levels of PCI. While resilience (sample similarity immediately before and 3 weeks after) was not significantly different, it tended to show similar trends as with the resistance measure. If sites with higher PCI are to be thought of as being more unstable then it is suggested that communities at less stable sites have much greater resistance than resilience.

Introduction

Urbanisation can affect streams in four main areas: (i) stream hydrology, (ii) habitat structure, (iii) water quality and (iv) biological components of streams (Schueler 1994, Arnold 1996, May et al. 1997). Of these, the hydrological flow regime is most important as it affects several components of stream habitat (Clausen and Biggs 1997, Poff *et al.* 1997) and consequently the structure of the biotic assemblages of streams (Biggs and Hickey 1994, Collier and Quinn 2003). Hydrology can be greatly affected by an increase in urbanisation (Leopold 1968). This is typified by increased percent catchment imperviousness (PCI) or area of impervious surfaces, such as roads and rooftops. As PCI increases there is decreased infiltration and increased surface runoff over impermeable surfaces. Therefore, high volumes of water are conveyed over land to streams in a shorter period of time (termed flashiness) than in the absence of urban developments.

The term disturbance is defined here as any relative discrete event in time that disrupts the ecosystem, community or population structure, and that changes resources, the availability of substratum or the physical environment (White and Pickett 1985). It is challenging to capture all major attributes of disturbance in stream ecology: frequency, extent, intensity and duration (Shea, Roxburgh and Rauschert 2004). However, temporal patterns and the intensity of impact can be categorized as pulse (short timeframe) and press (long timeframe) disturbances (Lake 2000, Collier and Quinn 2003). Recently the concept of ramp disturbances (steady increase in intensity over time) was also introduced (Lake 2000). Urban land use is an irreversible change and can be considered a press disturbance on hydrology and habitat (Elliot *et al.* 2004, Dow 2007). Overlaid on this disturbance regime can be the impacts of flood events that may interact to influence postdisturbance invertebrate recovery (see Collier *et al*. 2003).

Distrubance from flood events have been extensively investigated in stream systems (Poff *et al.* 1997, Poff 1997) and are thought to be one of the main factors structuring invertebrate communities (Statzner and Higler 1986, Townsend 1989, Townsend, Scarsbrook and Doledec 1997, Death and Zimmerman 2005). Hydrological disturbances from floods and spates (high flows that do not overtop banks) are known to decrease benthic invertebrate abundance, density and diversity (Matthaei, Uehlinger and Frutiger 1997, Palmer *et al.* 1996, Robinson, Uehlinger and Monaghan 2004). Water levels rise, velocity increases and the increase in stream bed shear stress dislodge invertebrates from substratum. Invertebrates are washed away or smothered and crushed. However, most, but not all high discharge events are deleterious since many invertebrates have life cycles and behaviours that enable them to cope with such disturbance events (Death 2008). Usually, recovery from spates or floods is fast and a return to predisturbance levels is attained in months or even days (Melo *et al.* 2003) as individuals persist and begin recolonisation from nearby patches or refuges (Townsend 1989, Melo *et al.* 2003). Investigating recovery patterns can be done by examining phenomenological events (Collier and Quinn 2003, Boulton *et al.* 1992), comparative studies in streams with differing disturbance regimes (Death and Winterbourn 1995) or by experimental manipulations (Matthaei, Uehlinger and Frutiger 1997, Melo *et al.* 2003, Robinson, Uehlinger and Monaghan 2004). After the stream is subjected to a natural or artficial flood event the pattern of recovery can be described in many ways and the community response subsequently identified.

Recovery from hydrological disturbances can be achieved through (i) the resistance to the disturbing force imposed and (ii) resilience, which refers to the return to predisturbance conditions. Resilience itself has two components namely, elasticity: how quickly a community can return to the point of equilibrium, and amplitude: how much disturbance it can return from (Stiling 1999). Therefore resistance and resilience can be described using components of diversity such as species number, richness or evenness. Both terms can be translated to describe how a community responds to an imposed disturbing force(s). In the context of this study, resistance and resilience are viewed with respect to the diversity components of species abundance, richness and evenness, as well as community similarity in ordination space. It is hypothesized that streams with higher percent catchment imperviousness (PCI) will be more resistant and resilient than streams with less PCI because higher PCI sites are less 'stable' from the underlying press disturbance of land use. Sites with higher PCI can be thought of as being more unstable in view of their physical (particularly increased flashiness (Poff *et al.* 1997, Dow 2007), chemical (Lenat and Crawford 1994) and stability characteristics (Schueler 1994). I also hypothesize that sites with higher PCI would have higher resistance and resilience, reflecting a pool of species that posses traits that confer with such due to the more disturbed nature of their habitat (such traits include- the choice of oviposition site, generation time, body form, attachment and feeding habits (but see Dolec et al. 2006, Verberka, Seipel and Esselink 2008)) .

Study Sites

The study was carried out in the Wellington Region of Lower North Island, New Zealand. The sites spanned approximately 8 km^2 located within two regions or major catchments of the Wellington Region: Porirua and Wellington (Figure 1). Five streams were chosen from three *a priori* groupings of Percent Catchment Imperviousness (PCI) of none (0), low (<5), and medium (5-20). One stream belonged to the first PCI group (none (0)) while two streams were chosen from each of the other two PCI groups. Replication was kept low due to time and equipment constraints. There was also the case of malfunctioning equipment.

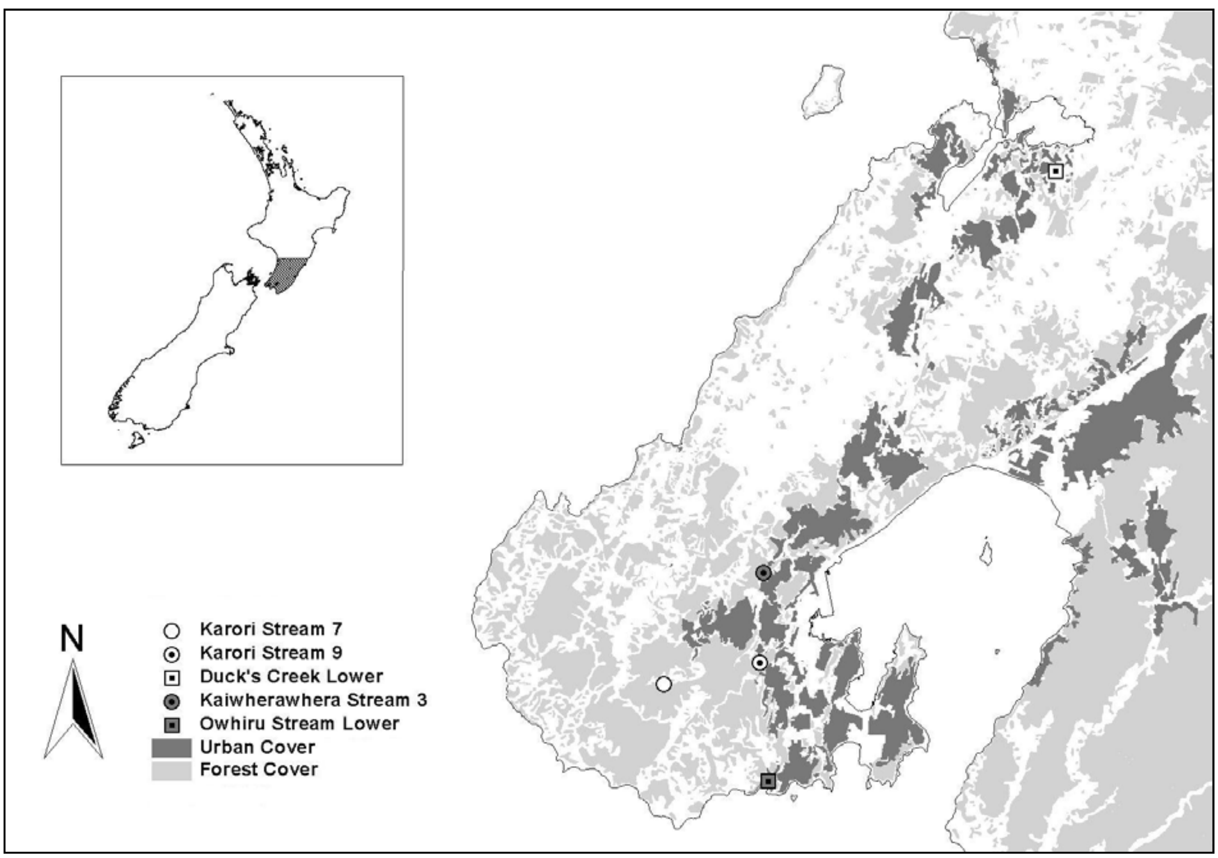


Figure 1. Map showing stream locations in the Greater Wellington Region, New Zealand.

Previous work in the Wellington region, illustrated an inflection point around 20 PCI after various biological indicators including species evenness, species richness, EPT number and MCI was regressed against PCI (Chapter 1). Several other studies (Schueler 1994, May *et al.* 1997, Morse *et al.* 2003) have found similar levels of urbanisation to cause biological change. The change in community structure seemed to be more obvious around 20% and too drastic beyond 20% impervious cover hence the choice of these *a priori* groupings.

Background site characteristics

Physicochemical variables were measured at each site during the Austral summer period in 2008 and 2009. The only water samples were taken once in 2008. Water samples were taken from each site and stored on ice in the darkness for less than 48 hours prior to ammonia, nitrate, and nitrite and phosphorus analyses. Analyses were determined by colorimetry on a flow injection analyser (APHA 2005). Also a Wolman's pebble count (Wolman, 1954) was used to quantify mean and median particle size from 100 randomly selected bed-particles at each of the sites. Streambed stability was assessed using the bottom component of the Pfankuch's Index (Pfankuch 1975, Death and Winterbourn 1994). This involves summing the scores of six environmental variables related to the stream bottom (scored according to the perceived importance); lower scores conferring more stability of a stream channel. Substrate embeddedness was subjectively assessed at each site after moving the substrate, where 1= loosely packed and 4= tightly packed (Death and Joy 2004). Riparian cover was assessed visually as percentage shaded area over the sampling reach. Algae form (periphyton type) and streambed sediment were also visually assessed as a percentage of the patch observed. Prior to biological sampling, conductivity, pH and temperature were measured with an ECScan™ handheld meter. After collecting the first biological sample, before the hydrological disturbance was applied, physical measurements were made each of the five macroinvertebrate sampling patches along the study reach of each stream: depth, width and velocity were measured at the thalweg using a ruler, tape measure and velocity head rod, respectively (Table 1).

Hydrological Disturbance Event

A Kärcher™ (model: G 2500 OH) pressure washer was used to simulate the experimental hydrological disturbance, in a similar fashion to that of Melo *et al*. 2003. This model was portable with an operating pressure of 2500 PSI, and a maximum water volume of 2.4 gallons per minute. This disturbing force was applied for about 20 minutes to each stream site, slowly and systematically "blasting" the entire breadth of streambed and about 0.5-1m distance inland on the lowerstream bank from the streams' edge. The length of stream disturbed varied between 15-45m as some riffle patches were further away from each other in some streams. The disturbance easily dislodged and moved gravel particles. Water became turbid, and pieces of moss were detached in the stream with no PCI.

Biological Sampling

Macroinvertebrates

Biological sampling was carried out immediately before, 1 hour, 1 week and 3 weeks after the disturbance. Benthic invertebrates were sampled using a Surber sampler ($0.1m^2$) fitted with a 250 μm mesh net. Five samples were taken from randomly selected locations within riffle habitats along a 10-30m reach and material preserved in 70% ethanol in the field. Macroinvertebrates were later removed by hand from sediment and debris in the laboratory and identified to the lowest practical taxonomic level using Winterbourn, Gregson and Dolphin (2006), McFarlane (1951) and online guides from NIWA (NIWA 2009). Most insect taxa were identified to genus and many non-insect taxa were only identified to order or class. Chironomidae (Diptera) were identified to subfamily or tribe. Subsampling of numerically abundant species with more than 200 individuals was conducted using a gridded tray or sample splitter box .

Periphyton

Biomass of periphyton was assessed by collecting five randomly selected stones at each macroinvertebrate sampling point. Periphyton sampling was carried out immediately before, 1 hour, 1 week and 3 weeks after the disturbance. The stones were kept in a dark icebox before being frozen at the laboratory. Pigments were extracted by soaking the stones in 90% acetone for 24 hours at 4°C in a dark chiller. Absorbancy was then measured with a Cary 50 Conc UV-Visible spectrophotometer™ and chlorophyll *a* and phaeophytin were calculated using the method of Steinman and Lamberti (1996).

Data Analysis

Biological metrics were assessed using a nested two-way repeated-measures analysis of variance (ANOVA), with PCI grouping (none, low, medium) as the between-subjects factor and time (before, 1 hour after 1 week after and 3 weeks after) as the within-subject factor. Data from sampling units (Surbers) were nested within streams as the subject of the analysis. Subsequent analysis of the various treatment levels in the analysis was done by comparing means to identify any true significant differences using Tukey's HSD All-Pairwise Comparison. These analyses were done using Statistixs9™ software.

Three components of diversity were examined, including Margalef's index (D), as a measure of species richness

$$D = (S-1)/\ln N$$

where S = number of species and N = the total number of individuals collected. Also and the Shannon-Weiner index (H') as a measure of species evenness

$$H' = -\sum_{i=1}^{s} p_i \ln p_i - \left[\frac{S-1}{2N}\right],$$

where S = species richness, N = species abundance and p_i = relative abundance of each species, that is to say, the proportion of S made up of the ith species).

Metrics were log ($x + 1$) transformed prior to analysis to reduce heterogeneity of variances. Non-metric multidimensional scaling analysis (MDS) using Bray-Curtis similarity was conducted on log-transformed biological data using Primer™ (Clarke and Warwick 1994, Clarke 1993, Clarke and Gorley 2006). Results are also illustrated using an index of resistance and resilience (after Melo *et al*. 2003). The index is simply an average of Bray Curtis similarity before and after the disturbance. The resemblance matrix created between samples during the analysis stage of the MDS test was used to create this index of resistance (sample similarity immediately before and 1 hour after) and resilience (sample similarity immediately before and 3 weeks after). These indices were then regressed against PCI using Statistix 9™.

Results

Environmental variables

Streams with low, or no PCI had the highest percentage of shade while medium PCI streams had the least. The Karori 7 stream had highest upstream slope and second largest substrate size but was also lowest in percent catchment imperviousness and stability score. Conversely, Kaiwherawhera stream (KW3) had the highest mean substrate size, percent catchment imperviousness but lowest upstream slope (Table 1). Average velocities ranged from $0.34ms^{-1}$ to $0.56ms^{-1}$. Stream width ranged between 0.72m to 3.07m with medium PCI streams having greater widths and higher temperatures and pH (Table 1). In general, medium PCI streams also had higher nitrate levels than lower PCI streams.

Periphyton

Generally, there was a definite reduction in periphyton biomass (measured as total pigment concentration) recorded on stones at each sampling patch and among all PCI groups (Fig. 3). The only true significant difference between the predisturbance period compared to any instance within the postdisturbance period was within the medium PCI group (Fig. 3): even after three weeks, Tukey HSD within group comparison showed that levels had not recovered to the predisturbance state ($p < 0.05$). Among the three PCI groups, Tukey HSD highlighted a significant difference in periphyton biomass: medium sites were significantly higher than sites with low or no PCI (Fig. 3; $p < 0.05$). Consequently, medium PCI sites were observed to be more affected by disturbance being reduced in biomass by a factor of about 3 and were unable to recover to similar levels of the predisturbance stage. Periphyton biomass generally showed a constant increase as the time from disturbance event increased however these trends were not significant.

Benthic Invertebrate Community

Generally, there was a clear distinction in invertebrate assemblages among the various PCI groups in the predisturbance stage (Fig. 2). Streams with medium PCI were dominated by Dipteran (mainly Chironomids: Orthocladiinae) and Trichopteran (*Oxyethira* and *Aoteapsyche* spp.) fauna while low PCI groups tended to have a greater number of Emphemeroptrans (mostly *Deleatidium* spp. and *Zephlebia* spp.). Karori 5 (no PCI) had the highest species number (37) in the predisturbance period and was dominated by *Deleatidium* spp. Duck's Creek (low PCI) had the highest species number (53) just one more observed taxa than Karori 5.

Figure 2. Non-metric Multidimensional Scaling (MDS) of mean invertebrate abundance at the five streams prior to disturbance: Karori Stream 7 (open circle), Karori Stream 9 (open square), Duck's Creek (open triangle), Kaiwherawhera Stream (solid triangle) and Owhiro Stream (solid square), (where n = 5 for each stream). The cross is a scaled representation of the maximum score range of both axes in the analysis. The best 2-D configuration stress value is 0%.

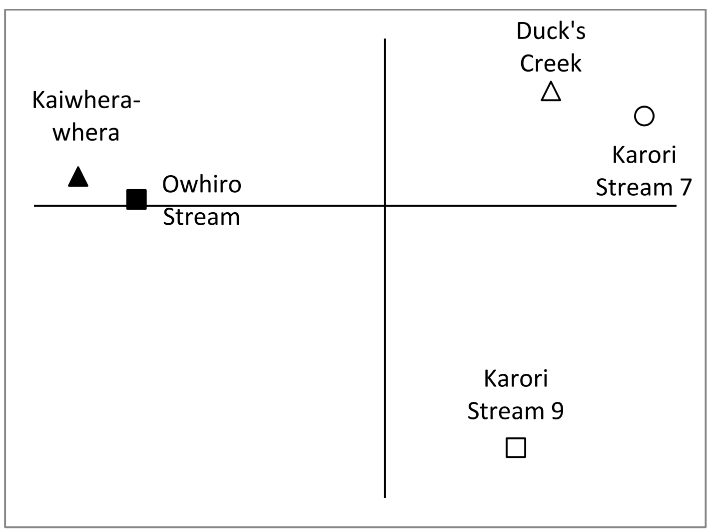


Table 1. Environmental variables for sampled streams in each of the three PCI groups (*one-off measurement taken in 2008)

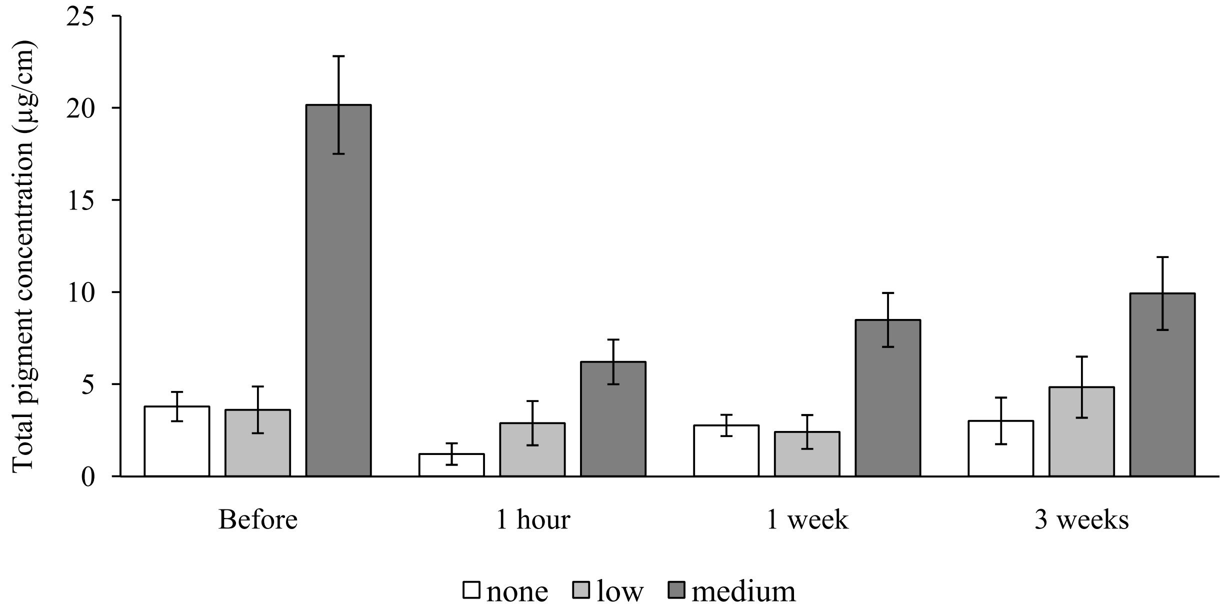
	None	Low		Medium	
	Karori Stream 7	Karori Stream 9	Duck's Creek	Owhiro Stream	Kaiwhera-whera
PCI (%)	0	1	5.5	15.9	16.4
Upstream Slope (°)	24.7	18.2	15	19.8	14.2
Mean Depth (cm)	7	3	12	12	11
Mean Width (m)	1.3	0.8	2.0	2.1	2.8
Mean Velocity (cm/sec)	0.41	0.33	0.47	0.41	0.56
Temperature (°C)	15.6	15	16.5	17	16
Conductivity (µs)	262	350	237	423	312
pH	5.2	5.6	7.6	8.4	8.6
DIN* (mg/l)	0.196	0.205	0.021	0.834	2.23
Nitrite* (mg/l)	0.008	0.008	0.008	0.009	0.016
Nitrate* (mg/l)	0.188	0.198	0.013	0.825	2.22
Dissolved inorganic phosphorus* (mg/l)	0.023	0.034	0.011	0.013	0.025
Pfankuch's Index	29	35	34.5	43	34
Mean substrate size (mm)	45	32	45	45	90
Substrate Index	8.69	3.81	6.77	6.64	11.19

Table 2. Repeated-measures analysis of variance (ANOVA) of the effects of Percent Catchment Imperviousness (none, low, medium) on components of diversity: (*a*) log10 abundance, (*b*) Marglef's index, and (*c*) Shannon-Weiner's index. The results for the Shapiro-Wilk test for normality are also shown under each diversity component.

	df	MS	*F*	*p*
(a) Log_{10} abundance				
($W = 0.99$, $p = 0.75$)				
Between subject factor				
PCIg	2	0.795	12.36	0.001
Within subject factor				
T	3	1.221	41.25	0.000
T * PCIg	6	0.033	1.12	0.373
(b) Marglef's Index				
($W = 0.99$, $p = 0.46$)				
Between subject factor				
PCIg	2	7.182	16.43	0.000
Within subject factor				
T	3	2.511	14.03	0.000
T * PCIg	6	0.934	5.22	0.001
(c) Shannon's Index				
($W = 0.99$, $p = 0.51$)				
Between subject factor	2			
PCIg		0.727	7.17	0.009
Within subject factor	3			
T	6	0.586	13.35	0.000
T * PCIg		0.194	4.41	0.002

NOTE: T, time from disturbance event; PCIg, Percent Catchment Imperviousness group, S, Streams. The error term for the within-subject *F* tests was the interaction of T * Stream * PCIg. Symbols (not shown). .*indicates crossing relationships among variables.

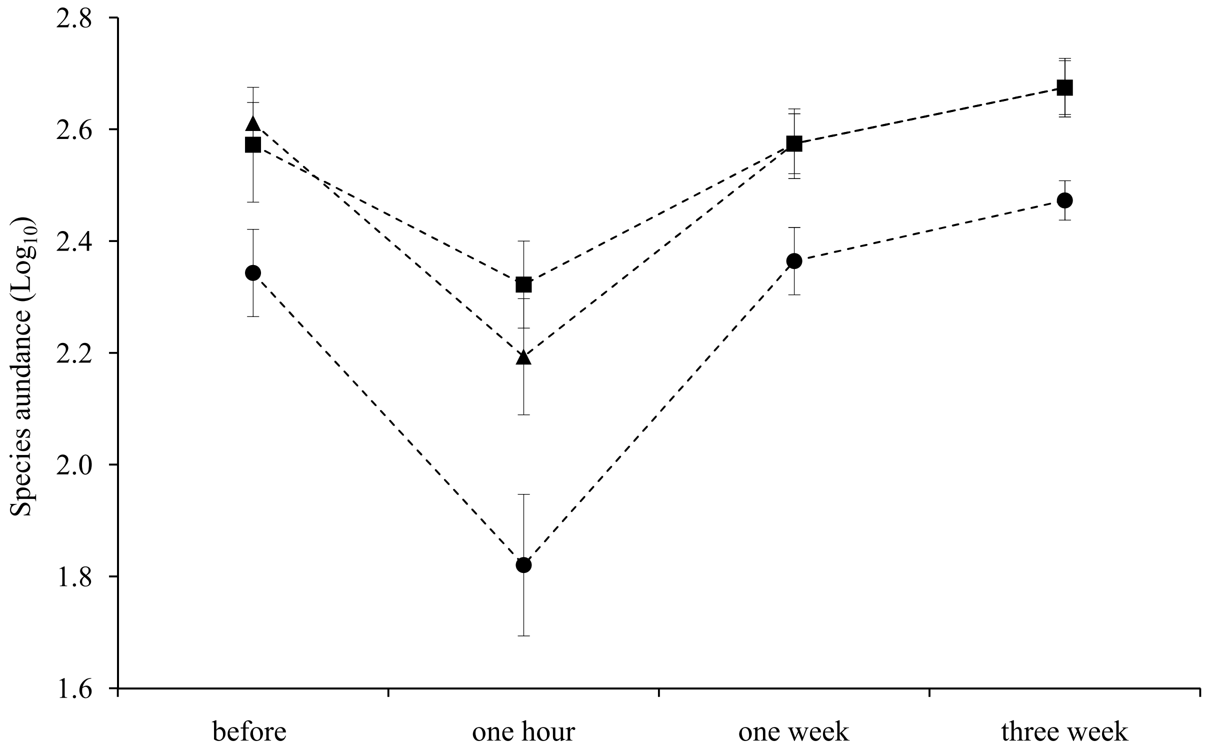
Figure 3. Periphyton biomass as mean total pigment concentration (chlorophyll *a* and phaeopigment) (±1 SE) on stones in patches from streams with no (n = 5), low (n = 10) and medium (n = 10) PCI immediately before the disturbance, 1 hour, 1 week and 3 weeks after the hydrological disturbance.



Resistance and Resilience

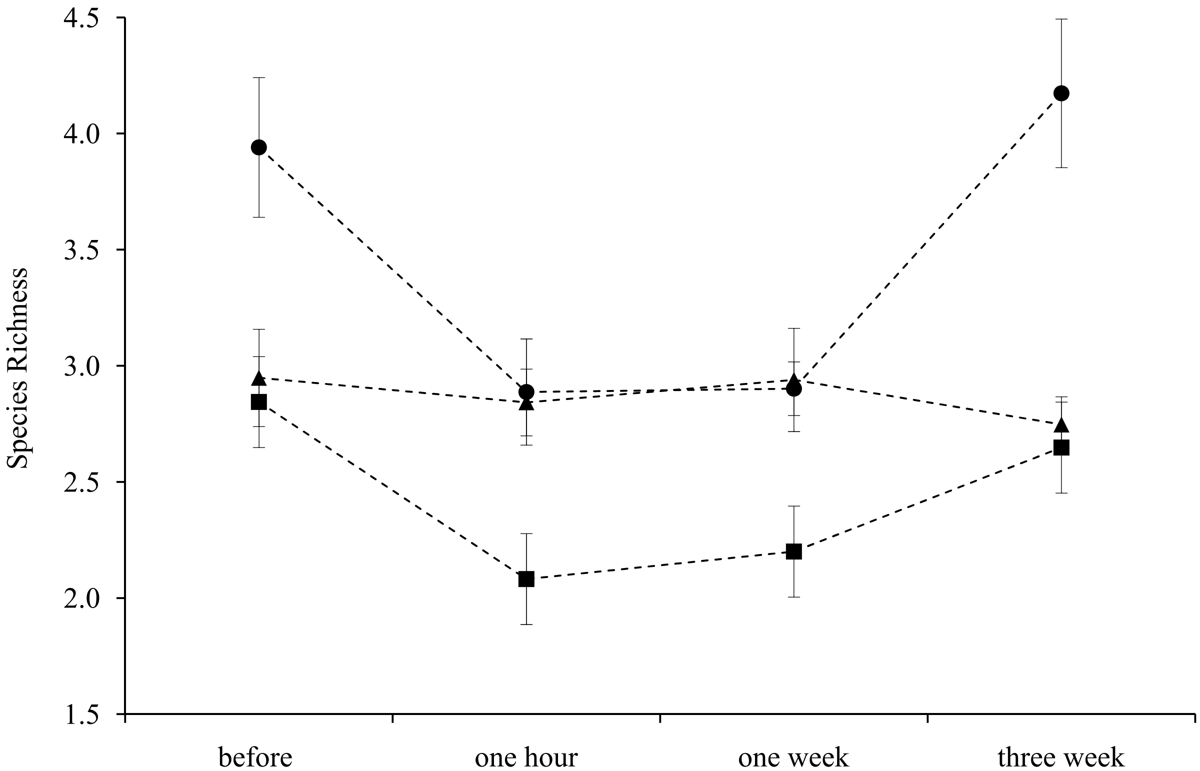
The hydrological disturbance had a significant effect on species abundance in all PCI groups. There was a significant reduction in mean species abundance immediately after the disturbance in all groups except streams with medium PCI (Figure 4 (a); Table 2). From 1 to 3 weeks after the disturbance, mean abundance levels in all PCI groups recovered to levels at predisturbance. Late postdisturbance abundance was also significantly different from levels immediatey after the disturbance. (Fig. 4 (a), Table 2): a Tukey HSD procedure revealed that these pairwise differences at the same level of PCI groups were significant, p <0.05. One notable observation was the slight increase in abundance beyond the predisturbance level. At three weeks postdisturbance abundance was higher than predisturbance levels in all PCI groups. While there was no significant interaction between PCI group and time since disturbance (p = 0.373; Table 2) streams with low or medium PCI were less affected by disturbance as the difference between mean postdisturbance abundance versus abundance immediately after disturbance was greatest in the none-PCI group.

Figure 4 (a) Mean invertebrate abundance (Log_{10} transformed) in the three PCI groups: none (solid circle), low (solid triangles) and medium (solid squares) for four sampling times. Standard error of means is shown. For the three PCI groups of none, low and medium n = 5, n = 10, n = 10, respectively.



The response of Marglef's index was most peculiar within the low PCI group (Figure 4 (b), not conforming to the general trend observed with the medium and no PCI groups. There were significant interactions between PCI group and time (p = 0.001; Table 2). Namely, the species richness depended on the PCI group that the site belonged to and time from disturbance (Fig. 4 (b): where PCI groups of none and medium were more sensitive to the disturbance than the low PCI group. In the predisturbance, and three weeks postdisturbance the interaction between land use and time and disparity between none and medium PCI groups are easily observed. None and medium PCI groups show significant differences between their species evenness levels before the disturbance. Declines were greater in sites with no PCI compared to medium PCI and Tukey HSD revealed significant reductions in species richness immediately after the disturbance within both PCI groups (p < 0.05). Also, within their respective groups they both exhibit a significant decrease of species richness immediately after the disturbance while low PCI sites show little effect (Figure 4 (b)). A distinct recovery and significant increase in species richness was only evident within the no PCI group, in this case reaching marginally above predisturbance levels. The effect of the disturbance or time from disturbance ($p < 0.001$; Table 2) generally causes a decline in species richness and a more definite increase or recovery after one week into postdisturbance period.

Figure 4 (b) Mean species richness (Marglef's index) in the three PCI groups: none (solid circle), low (solid triangles) and medium (solid squares) for four sampling times. Standard error of means is shown. For the three PCI groups of none, low and medium $n = 5$, $n = 10$, $n = 10$, respectively.



Likewise, Shannon-Weiner's species evenness displayed inconsistent trends for the low PCI group of (Figure 4 (c). The disturbance or time from disturbance does have a significant effect ($p < 0.001$; Table 2), generally causing a decline in species evenness up to one week into postdisturbance. This trend of decline is not particularly evident within the low PCI group as an increase is first observed immediately after the disturbance, followed by a significant decline from this peak at three weeks postdisturbance: Tukey HSD revealed significant reduction between times ($p < 0.05$). PCI group appears to also affect species evenness ($p < 0.009$; Table 2). The difference among PCI groups arises particularly between the low and medium PCI grouping where Tukey HSD highlights low PCI groups as having significantly greater species eveness than medium PCI sites ($p < 0.05$). There were significant interactions between land use and time ($p = 0.001$; Table 2). More specifically, species evenness depended on the PCI group that the site belonged to and time from disturbance (Fig. 4 (c). For example, immediately after the disturbance, inverse trends occur particularly between low PCI and medium PCI sites.

Figure 4 (c) Mean invertebrate species evenness (Shannon-Weiner index) in the three PCI groups: none (solid circle), low (solid triangles) and medium (solid squares) for four sampling times. Standard error of means is shown. For the three PCI groups of none, low and medium n = 5, n = 10, n = 10, respectively.

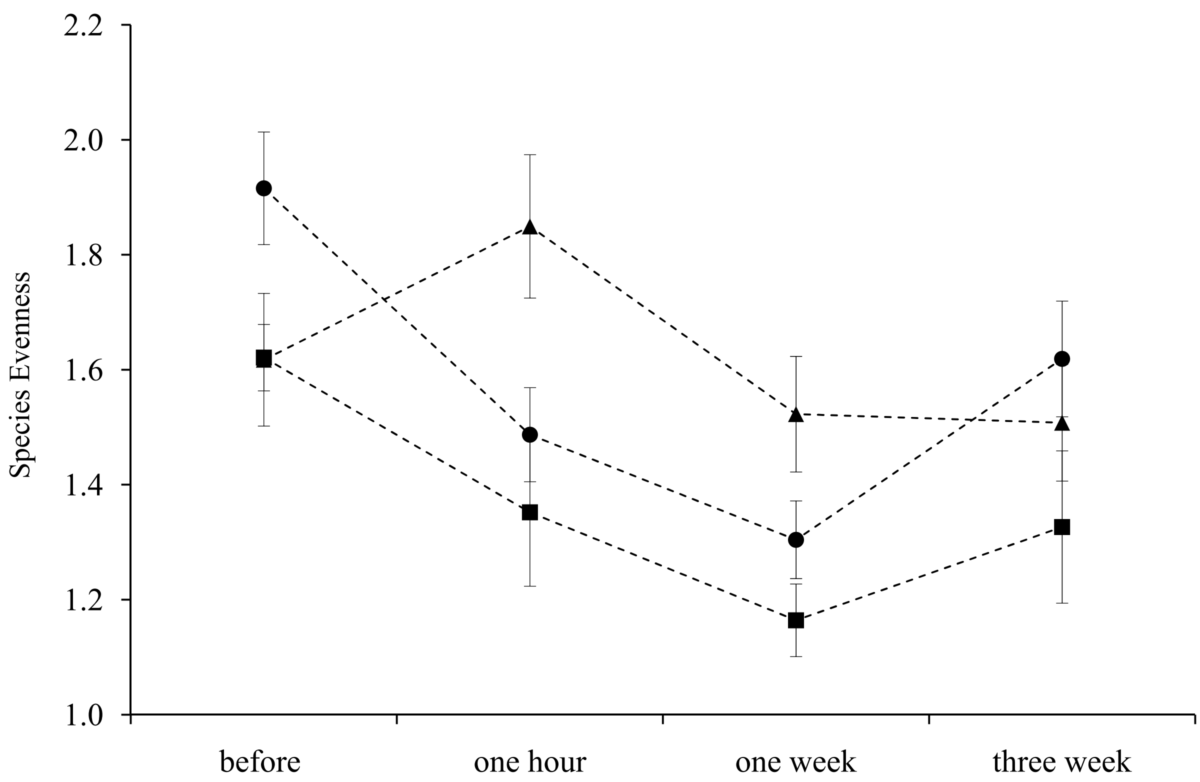


Figure 5(a) – (e). Nonmetric multidimensional scaling (MDS) plots of invertebrate collections, averaged for each of four time periods: immediately before (solid circles), 1 hour after (open triangles), 1 week after (open squares) and 3 weeks after (open circles). Each stream was plotted separately. The cross is a scaled representation of the maximum score range of both axes for each analysis. (a) Karori 7, (b) Karori 9, (c) Duck's Creek Lower, (d) Owhiro Lower, (e) Kaiwherawhera. Stress value is approximately 0% for all ordinations.

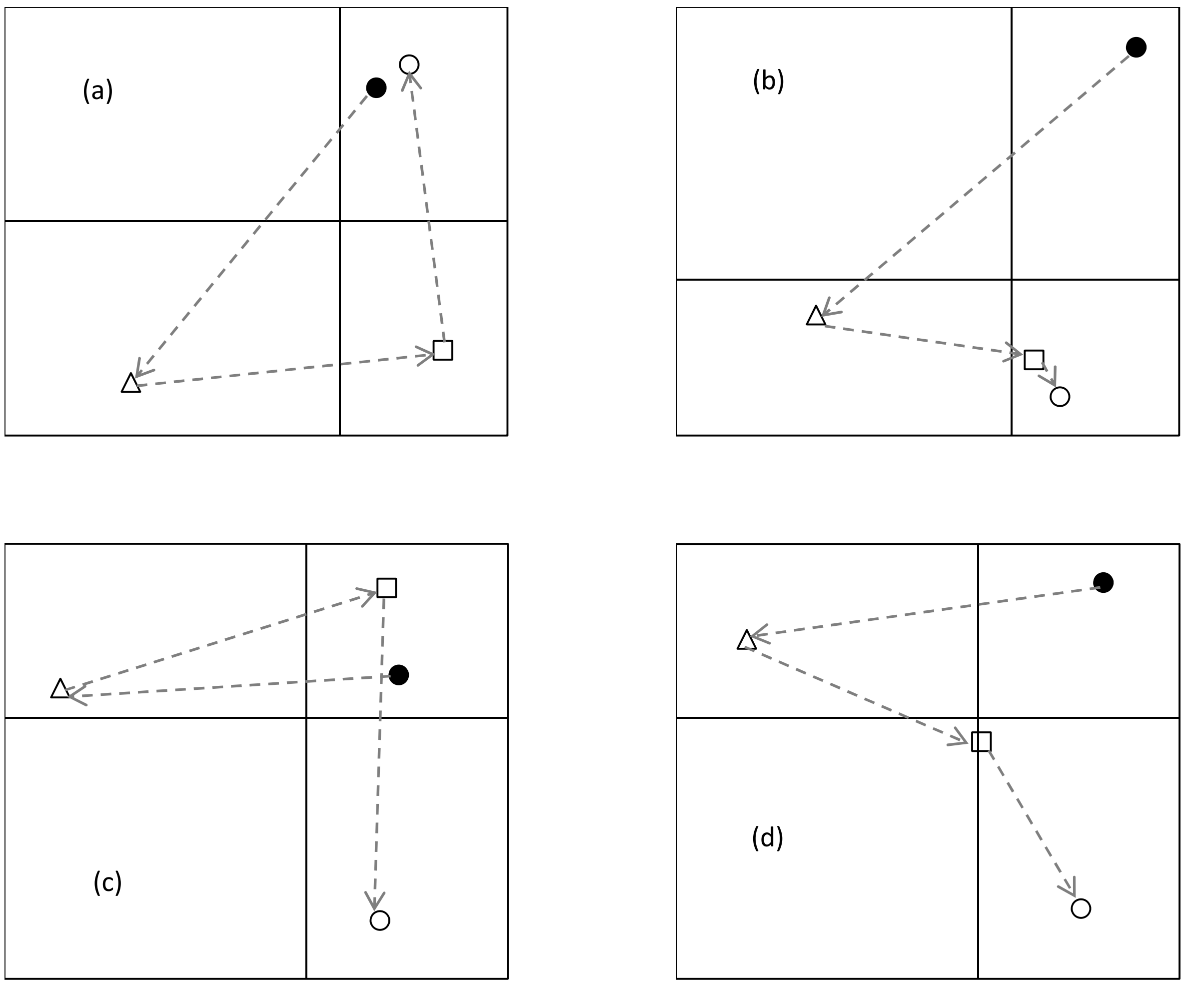
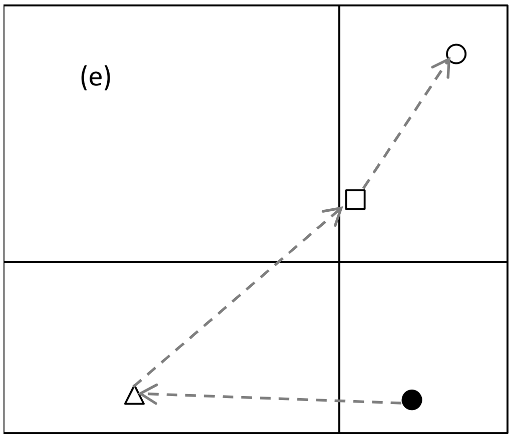


Figure 5(a) – (e) (Continued.)



Resistance Resilience Index

Results of the resistance index (obtained using Bray-Curtis similarity) showed significant differences between PCI groups (Fig. 6; F = 13.2, p = 0.0002). No significant difference in resilience was observed among PCI groups (Fig. 6; F = 0.34, p = 0.71). Resistance was highest in the medium PCI group and tended to increase with increasing PCI (Fig. 6, t = 0.6943, $p <$ 0.001). Sites with no PCI had lowest resistance scores. The ratio of resilience to resistance in the no PCI site was greatest due to the high index of resilience, relative to resistance (1:1.13). This ratio decreased in the low PCI group (1:1.1). Low PCI sites had greatest resilience but greater resistance vales than the no PCI group. As PCI increased the ratio tended toward a marginally higher resistance than resilience as in the medium PCI group (1: 0.97). To illustrate the relationship of resistance, a nonlinear regression of mean resistance (Bray-Curtis similarity) showed a reasonably strong fit to an asymptotic model (Fig. 7, R^2 = 0.78).

Figure 6. Average resistance and resilience of invertebrates is shown; following an experimental hydrological disturbance event in streams with no, low and medium PCI. Resistance was obtained as macroinvertebrate community resemblance (similarity) immediately after the disturbance divided by the similarity value before it. Resilience was obtained by dividing the similarity value three weeks after the disturbance by similarity immediately before the disturbance. Standard errors are shown; for the three PCI groups of none, low and medium $n = 5$, $n = 10$, $n = 10$, respectively.

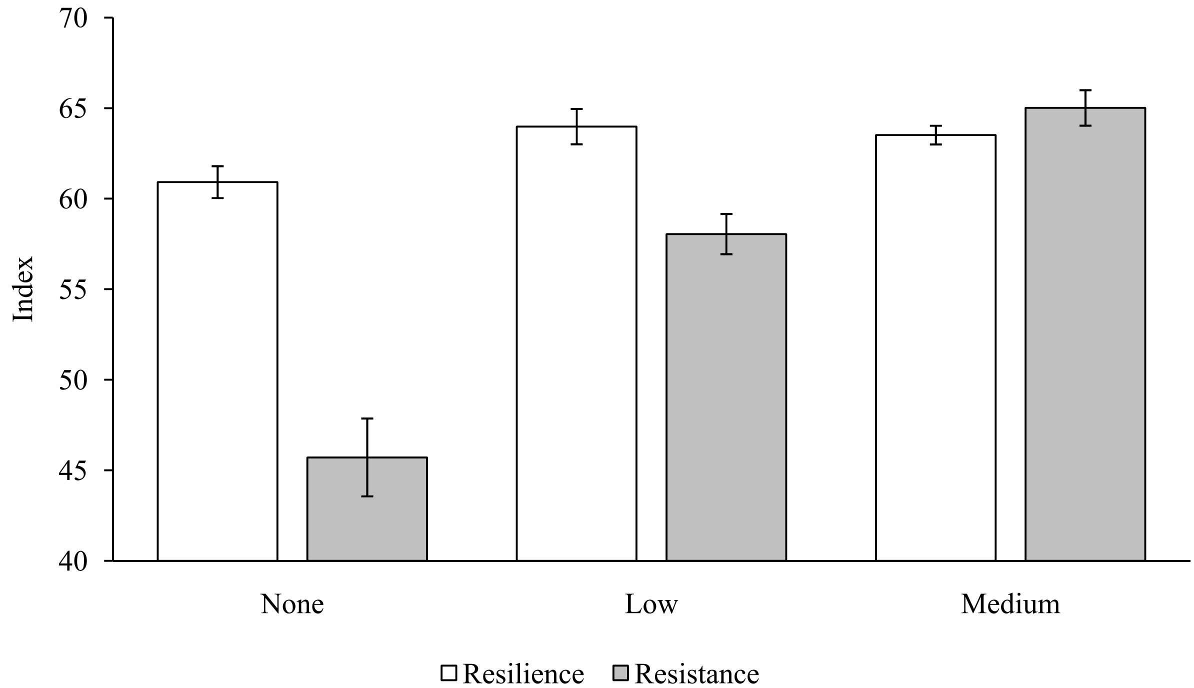
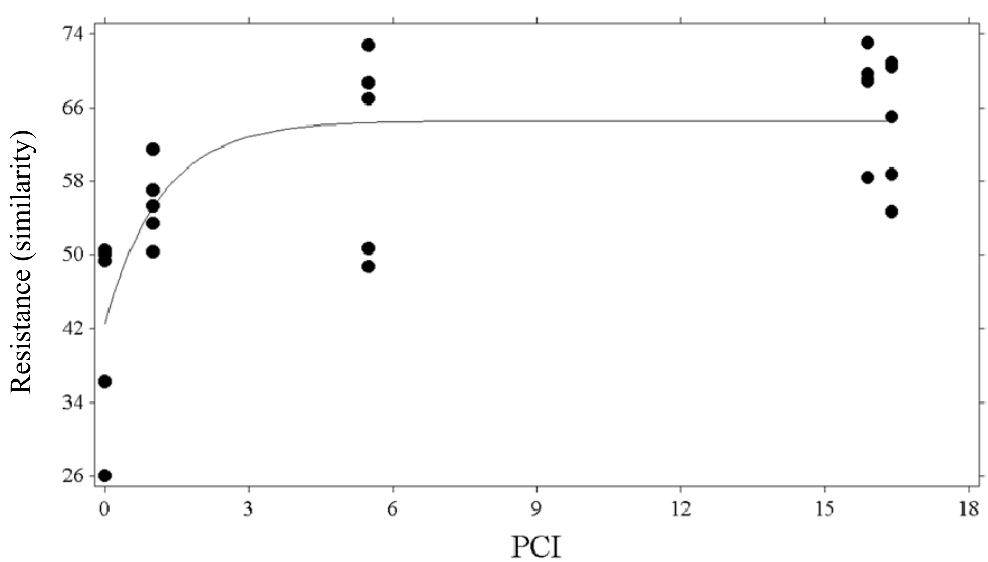


Figure 7. Non-linear asymptotic regression of Resistance of invertebrate communities against percent catchment imperviousness (PCI). Pseudo R-Squared = 0.78 (n= 25, model: y = 64 – 22(0.4)^x).



Discussion

Both investigatory factors of (i) the level of hydrological disturbance imposed and (ii) the underlying impact of catchment imperviousness (PCI), influenced benthic invertebrate community structure. The main effect of the hydrological disturbance is reduced diversity in terms of species abundance, richness and evenness which confers with the similar studies (Collier and Quinn, 2003, Robinson, Uehlinger and Monaghan, 2004). Indeed, disturbance leads to reduced diversity (Death, 1996b, Matthaei, Uehlinger and Frutiger 1997, McCabe and Gotelli, 2000). Of the 12 studies that looked at disturbance effects listed in McCabe and Gotelli (2000), 11 showed a reduction in species abundance, eight of which showed reductions in species density (number of species per sample). Species richness and abundance are reduced immediately after the disturbance. It appears that the main mechanism of decline is a combination of the catastrophic moving of substrate and dislodgement from substratum due to the turbulent flow generated by the machine (personal observation).

At the same time it is highly likely that the hydrological disturbance caused detachment of moss (personal observation) and the reduction in periphyton biomass that lead to the reduced diversity (Death, 1996a, Death and Zimmerman, 2005). Karori 5 stream (no PCI), Kaiwherawhera and Owhiru streams (medium PCI) all experienced a significant and drastic reduction in periphyton biomass immediately after the hydrological disturbance. Also, marked reductions in species abundance, richness and evenness were noted within these streams. However, diversity components of species evenness and richness seemed to be less affected in the low PCI streams that did not suffer a drastic reduction in periphyton biomass. Death and Zimmerman (2005) suggested that diversity is a function of time since last disturbance, mediated by the recovery of the periphyton food base: and this aptly explains the trends noted in streams that suffered a drastic reduction in periphyton biomass. The other two streams that did not suffer such a loss exhibited similar trends to that of Melo et al. (2003), reporting an increase in rarefied species richness and no change of species density (called species richness in this study). Although rarefied species richness (McCabe and Gotelli, 2000) were not examined directly actual trends were similar to that of species evenness (data not shown). Nonetheless, the reason these streams had unchanged species density and increased rarefied species richness is quite possibly due to unchanged periphyton biomass of postdisturbance. While this study compares to that of Melo *et al.* (2003), they did not include the reduction of the food base in their study as a factor for diversity patterns. Therefore, although evidence from empirical studies suggest that a reduction of species richness can be attributed mainly to disturbance and productivity (Death and Winterbourn, 1995, Death and Zimmerman, 2005, Cardinale, Hillebrand and Charles, 2006) it is difficult to say whether the reduced invertebrate densities of this study was primarily as a result of the hydrological disturbance or whether they were more affected by the reduction of the algal food base.

To gauge biological recovery, the state of the postdisturbance community should tend toward that of the predisturbance state irrespective of the community metric used. For a relatively small artificial flood as in this study, diversity metrics should generally follow a relatively predictable trend until the rate of change tapers off to about zero indicating recovery (or local stability (May, 1972, Death 1996a, Death 1996b)). The recovery time for species abundance seems to be within a one week period in this study which is the shortest recovery interval for any diversity metric. Likewise, Melo *et al*. (2003), noted a marked recovery in species abundance within 8 days of a similar artificial hydrological disturbance. In a more natural scenario, Flecker and Feifarek (1994) showed a strong positive relationship between insect abundances and time elapsed since the last storm event. At these scales of disturbance species abundance seem to share a similar trend among studies (Chase and Leibold, 2002). On the other hand, Death (1996a) found an increase in absolute abundance after disturbing individual cobble baskets in two stable and two unstable streams but this may be because of the smaller spatial scale of the applied disturbance (Chase and Leibold, 2002) (but see discussion in Death, (1996a)). Species evenness and richness appear to take three or more weeks to recover to predisturbance levels. The increase in taxon richness usually follows a smooth trajectory to an asymptote after rapid species abundance recovery (Lake, 2000). This recovery pattern is clearest in the three streams that suffered a drastic reduction in periphyton biomass. In Death's (2009) discussion of the disturbance-diversity relationship, disturbance is shown to remove organisms and lower taxon richness: with recovery or subsequent increases in diversity and taxa recolonization continuing until a subsequent disturbance or until the maximum productivity capacity of the site is achieved. If we assume that the predisturbance state was near the carrying capacity of habitat productivity then this may also be the case at three weeks postdisturbance sites since could be nearing such a point (Fig. 3).

Apart from the level of hydrological disturbance imposed, the underlying impact of catchment imperviousness also influenced benthic invertebrate community structure. The former chapter showed that PCI has a very strong negative relationship with invertebrate community structure. While that chapter showed a change in community type in streams with high PCI (>20 percent imperviousness) it seems that even at PCI levels of 5-20, community shifts may be occurring. Increasing PCI appears to amplify the general effect of hydrological disturbances, that is, it further reduces diversity. Sites that were under a higher land-use disturbance treatment or higher PCI were the ones that experienced less change in invertebrate community structure as they would have been more disturbed in the first place. Sites with higher PCI can be thought of as being more disturbed or unstable in view of their physical (particularly increased flashiness (Poff *et al.* 1997, Dow 2007)), chemical (Lenat and Crawford 1994) and stability characteristics (Schueler 1994). Thus, PCI may be thought of a disturbance in itself. For example, Owhiro had the highest concentrations of nitrates (Arnold 1996, Morse *et al.* 2003). Also, its streambed was least stable. On the other hand, Duck's Creek had the lowest values of nitrogen oxides, nitrates and dissolved reactive phosphorus while its streambed stability was in the median range. By comparing these two streams with respect to their water chemistry and streambed stability, Owhiro can be thought of as a less stable site while Duck's Creek may represent a more stable site in terms of these simple measures. Thus, if these sites are considered to represent a range from more stable (no PCI) to less stable (medium PCI) then the results suggest that communities at less stable sites (high PCI) have greater resistance rather than resilience. The ratio of resilience to resistance tends to decrease with increasing PCI (Fig. 6). This greater resistance is due to the nature of the persisting species pool. These species may possess traits that confer higher resistance and resilience due to the more disturbed nature of their habitat (Lancaster and Belyea, 1997, Doledec *et al.*, 2006, Verberka, Siepel and Esselink 2008). However, there may be a smaller pool of available colonists at more disturbed, higher PCI sites (Klein, 1979, Lenat and Crawford, 1994, Stepenuck, Crunkilton and Wang, 2002, Walsh *et al.*, 2001). Sites with no to low PCI had mutual species such as *Hydrochorema crassicaudatum* and Tanytarsini that became extinct following the disturbance. Lower PCI sites, although having greater taxa richness, suffered more extinctions. This particular community composition added to a higher resilience to resistance ratio. The community in higher PCI sites was different: higher PCI sites had lower tax richness and less extinctions however, the few species that it did have such as *Deleatidium* spp., *Aoteapsyche*, *Oxyethira* and *Archichauliodes diversus* were very resistant to the hydrological disturbance while being able to persist in higher PCI conditions.

In conclusion urban streams appear to recover from floods more quickly that directly results from the increased resistance of a smaller species pool, possessing traits that confer higher resistance. While it would be very difficult to stop the increase of PCI (Brueckner and Fansler 1983), it is possible to predict invertebrate responses to disturbances in urban streams.

Chapter 4 Synthesis

Increasing urbanisation in many towns and cities is placing pressure on the ecological integrity of streams (Arnold and Gibbons 1996, Paul and Meyer 2001, Gregory 2006, McKinney 2006). These towns and cities are simply habitats constructed almost exclusively to meet the demands of humans (McKinney 2006). They are typified by devegetated, paved plots with high-rises and residential buildings corresponding to increased percent catchment imperviousness (PCI) - one of the consequences of increased urbanisation. A corollary of the human demand for urbanisation is the compromised integrity of stream ecosystems (Schueler 1994, Arnold 1996, Paul and Meyer 2001).

What happens to the stream's invertebrate community is that as the percentage of the catchment becomes increasingly urban, more pollution tolerant taxa are present (Suren 2000, Paul and Meyer 2001) and the biota is more homogenized (McKinney 2006). Generally, community changes are quite evident, namely lower biological metric scores for species number, evenness, richness and the number of pollutant intolerant taxa such as Emphemeroptera, Plecoptera and Tricoptera (EPT). These biometrics clearly exhibit an inverse relationship with urbanisation (May *et al*. 1997, Walsh *et al.* 2001, Morse *et al*. 2003). Evidence from this study suggests that if the urbanisation intensity within the Greater Wellington Region surpasses 15-20 PCI there is an accumulation of biological effects, namely the loss of EPT taxa. At low PCI, EPT taxa and species richness are well maintained. At the same time, the study suggests that lower PCI streams (with greater species richness) give rise to a greater proportion of their invertebrate communities exhibiting high resilience and low resistance to streambed disturbances. Conversely, the relative abundance of EPT taxa of the species pool follows a declining trend as urbanisation increases leading to the eventual absence of EPT in streams with high PCI and a remnant species pool that exhibits high resistance and resilience to streambed disturbances. Essentially, the stream communities become very simplified and have lower biometric values indicating that stream integrity has become compromised (Arnold 1996, May *et al*. 1997, Walsh *et al*. 2001, Morse *et al*. 2003).

What is likely is that the resultant community in sites with high PCI are made up of species that posses a combination of traits that enable them to persist despite the highly disturbed and unstable nature of their habitat (Lancaster and Belyea 1997, Doledec *et al.* 2006, Verberka, Siepel and Esselink 2008). Although high PCI streams are depauperate, the species pools are able to 'resist' streambed disturbances and other possible deleterious forces. Effects of imperviousness such as altered stream hydrology, habitat structure and water quality contribute to this highly disturbed regime. Thus, as streams become increasingly urban with consequent increases in disturbance frequency and/or intensity, species traits that enable population persistence become more prevalent (Lytle and Poff 2004, Doledec *et al*. 2006).

Pollution tolerance is also equally important in this particular case. Thus, highly urbanised sites would have some species that are able to rebound relatively quickly after a disturbance but moreso, species that are able to withstand the disturbance and pollution tolerant. In moderately urban sites species such as *Deleatidium, Oxyethira* and some simulids exhibit good resilience, resistance and pollution tolerance. However, highly urbanised streams favour species like *Potamopyrgus antipodarum* and certain Chironomid species with traits such as short generation times and multiple reproductive cycles per individual (as with the former of the two species). As a result, the species pool that persists in highly urban streams has greater resistance and resilience to the altered physical, chemical and stability characteristics of such sites and equally important is their high pollution tolerance. Although such species end up dominating highly urbanised streams, this study highlights that there is a smaller pool of available colonists at the more disturbed and more urban sites (also see Klein 1979, Lenat and Crawford 1994, Stepenuck, Crunkilton and Wang 2002, Walsh *et al.* 2001) as only the highly resilient and resistant, pollution-tolerant taxa are able to persist in the face of the disturbances.

At the same time, the ability to rebound from the in-stream and riparian disturbances associated with urbanisation PCI also depends on the success of the adult terrestrial stages of stream invertebrates (Collier and Scarsbrook 2000, Winterbourn and Crowe, 2001). Of the stream invertebrates that exhibit adult terrestrial stages, the ability to either laterally penetrate to nearby streams or complete their life cycles within the same stream adds to the overall resilience to disturbance. Results show that generally, aquatic invertebrate stages show an increase in population resistance and resilience with increasing urbanisation. On the other hand, any increase in PCI amounts to a decrease in forest or vegetation cover that affects terrestrial stream invertebrate stages (Collier and Scarsbrook 2000, Peterson *et al*. 2004) and ultimately leads to less 'out-of-stream' resilience. Adults of some species (EPT taxa in particular) use the ambient vegetation or riparian zone for completing their life cycles and sometimes as food source (Peterson *et al*. 1999, Collier and Scarsbrook 2000). In this light, intact riparian vegetation and low PCI helps provide supportive habitat that helps maintain the species pool of a particular stream. If a flashy event occurs in a stream, adult terrestrial stages serve as an *ex-situ* source for recolonisation. This typified idea of resilience is observed in streams with no PCI and intact catchment vegetation (eg. Karori 9 site) such that community composition returns quite closely to that of the pre-disturbance stage. The invertebrate community of streams with high PCI are not only depauperate but may take longer to return to pre-disturbance following the streambed disturbance. Therefore, it is likely that the few riparian corridors and stands of intact vegetation that are left standing within a particular catchment of a high PCI stream may be insufficient habitat for adult terrestrial stages to persist.

Thus, knowing the PCI of a site can indirectly highlight these probable changes that occur within the urban stream community, and can be useful management tool in predicting the state of urban stream invertebrate communities (Arnold and Gibbons 1996, May *et al*. 1997, Morse *et al*. 2003). Scientists such as Derek Booth have long been a proponent of setting watershed limits, particularly for the extent of urbanisation within watersheds. It is plausible that an initial limit of 15-20 PCI could be set for any future development in Greater Wellington, New Zealand, however this limit in itself may not be as protective to stream values due to future confounding pressures such as increased flashiness or more severe low flows with climate change (Meyer *et al.* 1999, Milly, Dunne and Vecchia 2005, Nelson *et al.* 2009). Even with an urbanisation limit of 15-20 PCI, artificial streambed disturbance caused extirpation of species in the streams around 15% PCI. Therefore, watershed limits need to work alongside remedies to protect streams that have not yet reached this limit. It would be more practical to invest into simple, economical remedies to protect streams nearing this limit while those that have surpassed this will need intensive investment, particularly into stormwater management to reduce peak flows (Nelson *et al.* 2009) Hypothesised remedies for restoring urban streams primarily entail re-establishing natural flow regimes (Poff et al. 1997, Poff et al. 2003, Arthington et al. 2006) and restoring physical elements such as channel morphology and riparian habitat and removing sources of pollution (Booth 1997, Finkenbine et al. 2000, Booth 2005).

While setting PCI limits is a powerful semi-protective tool for a manager, scientists still need to provide conclusive evidence to inform managers more about important mechanisms that will enable effective management and restoration of urban streams. This will involve understanding and documenting the disturbance regime of urban streams. The process of isolating potential drivers of invertebrate community change or identifying to what extent changes in hydrology (such as increased flashiness), habitat or water quality drive invertebrate community change is a step in this direction. Apart from the need to understand mechanisms, the evidence that more urban streams are homogenized and have species that posses traits that confer greater resistance and resilience, raises several difficult ecological questions (Lytle and Poff 2004), such as the persistence and rate of evolution of organisms in face of altered regimes. Within a conservation biology context, the irreversible press-type disturbance of urbanisation (Lake 2000) poses questions to researchers of whether populations can keep evolutionary pace with current rates of alteration. Urban land transformation can occur relatively quickly, severely or even instantaneously to facilitate urban population growth and societal demand. As a consequence, this provides little or no time for stream populations to adapt gradually to these conditions (Lytle and Poff 2004) and can lead to possible losses of species in areas with extensive urbanisation. Also, with the loss of species, it is uncertain whether stream ecosystems will still have efficient functioning and provide essential ecosystem services (Harwell *et al*. 1999). An understanding and consideration of this is particularly important in setting limits. There are several arguments that may ensue this particularly when it comes to sacrificing stream services and value to facilitate human demands.

In-stream diversity is always encouraged as it reflects good stream health (Scrimgeour and Wicklum 1996, Rapport, Costanza and McMichael 1998, Boulton 1999) conferring with the provision of ecosystem services (eg water purification and recreation value) but it seems that as the need for urban sprawl continues, managers may have to decide whether to slightly compromise a healthy stream and permit loss of a few species or further sacrifice an unhealthy depauperate one. Some managers may follow the idea that different species may have redundant functions in ecosystems (McGrady-Steed, Harris and Morin 1997). Therefore in the healthy stream, increased diversity increases the odds that the streams will have functional redundancy by containing species that are capable of functionally replacing important species (Lawton and Brown 1994, McGrady-Steed, Harris and Morin 1997). Although it is probable that even with a reduction of species in slightly impaired streams, some degree of ecosystem processes may be preserved: I am not suggesting to any manager that one should compromise an ecosystem to facilitate urban land use. Good ecosystem insurance (Yachi and Loreau 1999) is what is needed and reduction in species richness does not provide this. Therefore, these considerations need to be carefully examined within the urban context for managing already urbanised streams so that scientific ideals and political policy may find even grounds. Decisions will have to be made to reflect socioeconomic values imposed on urban streams but it is also important that autonomous ecological values be known and available to guide decisions.

Without ecological values being conveyed and patiently understood, urban streams will be rife with murky reflections of misapprehensive socioeconomic principles that will fail to maximize aggregate societal and economical well-being (Brueckner and Fansler 1983, Brueckner 2000). Urban streams will continue to be also rich with opportunity for researchers to investigate the intimately woven concepts of disturbance and stability (McNaughton 1978, Death and Winterbourn 1995, Meyer, Paul and Taulbee 2005), weakened trophic interactions as with more urban streams (McCann, Hastings and Huxel 1998) and concepts of coexistence and competitive exclusion (McCann, Hastings and Huxel 1998). As there continues to be ground-truthing of these ecological questions and concepts, watershed management will have several effective solutions for the preservation and restoration of urban streams.

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APPENDIX 2.1. Photographs of stream sites sampled with values for PCI, AvgUs, species abundance, species number, EPT number and co-ordinates (Northing/Eastings).

PORIRUA

<table>
<tr><td rowspan="9"></td><td colspan="2"><u>Duck's Creek Lower</u></td></tr>
<tr><td>Site Code</td><td>DCL</td></tr>
<tr><td>PCI</td><td>5.5</td></tr>
<tr><td>AvgUs</td><td>14.99</td></tr>
<tr><td>Species Abundance</td><td>1471</td></tr>
<tr><td>Species Number</td><td>26</td></tr>
<tr><td>EPT number</td><td>13</td></tr>
<tr><td>Northing</td><td>2669117</td></tr>
<tr><td>Easting</td><td>6008455</td></tr>
<tr><td rowspan="9"></td><td colspan="2"><u>Porirua-Tawa</u></td></tr>
<tr><td>Site Code</td><td>POT</td></tr>
<tr><td>PCI</td><td>30</td></tr>
<tr><td>AvgUs</td><td>12.08</td></tr>
<tr><td>Species Abundance</td><td>535</td></tr>
<tr><td>Species Number</td><td>23</td></tr>
<tr><td>EPT number</td><td>8</td></tr>
<tr><td>Northing</td><td>2663268</td></tr>
<tr><td>Easting</td><td>6002175</td></tr>
</table>

	Tekapu Stream Site Code PCI AvgUs Species Abundance Species Number EPT number Northing Easting	 TAK 15.3 14.18 2883 22 8 2663793 6001113
	Mid Porirua Stream Site Code PCI AvgUs Species Abundance Species Number EPT number Northing Easting	 P6 85.3 8.19 1350 16 4 2662000 5997900
No picture available	Ngauranga Stream Site Code PCI AvgUs Species Abundance Species Number EPT number Northing Easting	 N1 63.8 15.49 432 11 1 2661300 5994700

WELLINGTON

	Karori Stream 5 Site Code PCI AvgUs Species Abundance Species Number EPT number Northing Easting	 K5 4 15.41 432 27 16 2655908 5987978
	Karori Stream 7 Site Code PCI AvgUs Species Abundance Species Number EPT number Northing Easting	 K7 0 24.68 344 20 12 2653000 5987309

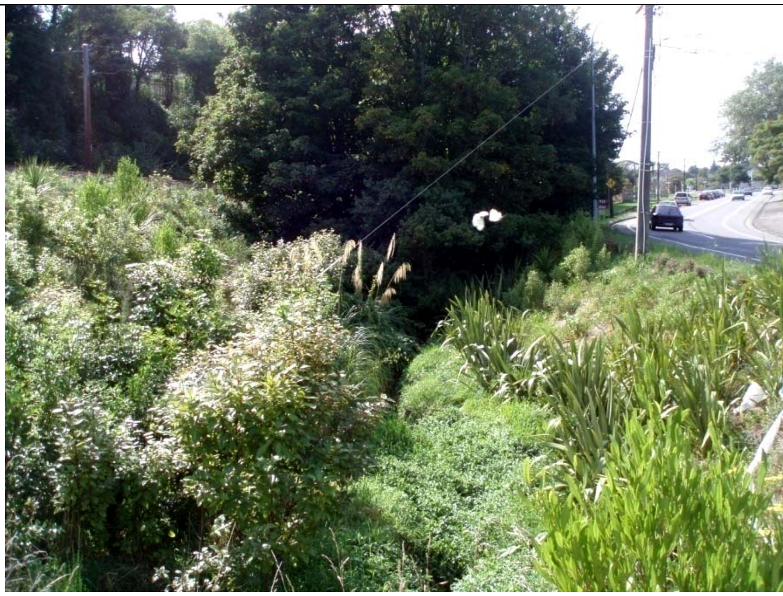
<table>
<tr><td></td><td><u>Kaiwherawhera Stream 3</u>

Site Code
PCI
AvgUs
Species Abundance
Species Number
EPT number
Northing
Easting</td><td>

KW3
16.4
14.23
688
18
7
2657100
5991900</td></tr>
<tr><td></td><td><u>Owhiro Stream Lower</u>

Site Code
PCI
AvgUs
Species Abundance
Species Number
EPT number
Northing
Easting</td><td>

OWH2
15.9
19.83
315
9
1
2657300
5983300</td></tr>
</table>

	<u>Kaiwherawhera Stream 1</u> Site Code PCI AvgUs Species Abundance Species Number EPT number Northing Easting	 TRAIN 58.4 10.51 289 10 1 2658800 5994100
	<u>Karori Stream 9</u> Site Code PCI AvgUs Species Abundance Species Number EPT number Northing Easting	 K9 38.6 18.22 331 19 10 2656948 5988188

HUTT VALLEY

<table>
<tr><td rowspan="9"></td><td><u>Hutt Recreation Field</u></td><td></td></tr>
<tr><td>Site Code</td><td>BM</td></tr>
<tr><td>PCI</td><td>89.5</td></tr>
<tr><td>AvgUs</td><td>0.4</td></tr>
<tr><td>Species Abundance</td><td>3508</td></tr>
<tr><td>Species Number</td><td>9</td></tr>
<tr><td>EPT number</td><td>1</td></tr>
<tr><td>Northing</td><td>2669663</td></tr>
<tr><td>Easting</td><td>5997101</td></tr>
<tr><td rowspan="9"></td><td><u>Pine Haven Upper</u></td><td></td></tr>
<tr><td>Site Code</td><td>PHU</td></tr>
<tr><td>PCI</td><td>5.3</td></tr>
<tr><td>AvgUs</td><td>17.73</td></tr>
<tr><td>Species Abundance</td><td>2182</td></tr>
<tr><td>Species Number</td><td>25</td></tr>
<tr><td>EPT number</td><td>14</td></tr>
<tr><td>Northing</td><td>2678900</td></tr>
<tr><td>Easting</td><td>6002300</td></tr>
</table>

	<u>Speedy's Stream</u> Site Code PCI AvgUs Species Abundance Species Number EPT number Northing Easting	 SP1 8.8 14.61 912 20 6 2671700 5999900
	<u>Silverstream Upper</u> Site Code PCI AvgUs Species Abundance Species Number EPT number Northing Easting	 SSU 23.8 11.26 3376 22 7 2680400 6004800
No picture available	<u>Blaike's Stream</u> Site Code PCI AvgUs Species Abundance Species Number EPT number Northing Easting	 BLK 13 9.7 507 25 13 2688952 6009849

	Stokes Valley Lower	
	Site Code	SVL
	PCI	36.3
	AvgUs	15.74
	Species Abundance	1287
	Species Number	18
	EPT number	5
	Northing	2676400
	Easting	6003000
	Waiwhetu Upper	
	Site Code	W6
	PCI	41.5
	AvgUs	13.37
	Species Abundance	1808
	Species Number	10
	EPT number	1
	Northing	2672800
	Easting	5998300

KAPITI COAST

	<u>Convent Road</u> Site Code PCI AvgUs Species Abundance Species Number EPT number Northing Easting	 CONV 25.4 1.4 1026 8 1 2690771 6049236
	<u>Waikanae Tributary</u> Site Code PCI AvgUs Species Abundance Species Number EPT number Northing Easting	 WAT 9.8 23.47 627 20 8 2684550 6033450

	Whareroa Upper	
	Site Code	WHU
	PCI	8.75
	AvgUs	14.4
	Species Abundance	465
	Species Number	12
	EPT number	3
	Northing	2677527
	Easting	6025951

Appendix 2.2 Table showing soil type and rainfall data at automatic rain gauge stations near to study sites (from Watts and Gordon 2008, Keenan and Gordon 2009)

	Major Catchments						
	Kapiti Coast		Wellington		Upper/ Lower Hutt		Porirua
Major Soil types	Brown stony		Brown Stony		Ultic & Recent soils (within valley) and Brown Stony (beyond valley)		Brown stony & Pallic soils
Nearby rain guage	Waikanae	QE Park Whareroa	Kaiwhera-whera	Regional Council Centre	Birch Lane Hutt	Mang-aroa	Seton Nossiter Park
Easting	2684600	2676258	2656100	2658900	2671000	2679000	2662300
Northing	6033300	6024009	5988400	5989200	5997600	5999600	5998100
Rainfall totals (mm) and long-term mean monthly totals in parenthesis							
Dec '08	141 (104.6)	134.5 (n/a)	147.6 (88.6)	92 (66.1)	106.5 (115.5)	144 (117.3)	131.5 (95.6)
Jan '08	109.5 (79.7)	84.5 (n/a)	34.2 (80.5)	23 (67.9)	37 (94)	84.5 (92.6)	46.5 (75.9)
Feb '08	22.5 (78.5)	17.5 (n/a)	47.2 (76.4)	43 (60.2)	44 (116.5)	55.5 (82)	45 (81.3)
Mar '08	108 (88.6)	115.5 (n/a)	166.6 (85.5)	123.6 (55.7)	104.5 (63.1)	104.5 (95.9)	133.5 (74.2)