Assessing Impacts of State Highway Stormwater Runoff on Stream Invertebrate Communities

Submitted by Earl Shaver (Aqua Terra International Ltd.) and Alastair Suren (National Institute of Water and Atmospheric Research Ltd.), Foreword prepared by Carl Reller, NZ Transport Agency.

www.nzta.govt.nz

October 2011

ISBN 978-0-478-38069-9

NZ TRANSPORT AGENCY
WAKA KOTAHI

New Zealand Government

Disclaimer

The NZ Transport Agency has endeavoured to ensure material in this document is technically accurate and reflects legal requirements. However, the document does not override governing legislation. The NZ Transport Agency does not accept liability for any consequences arising from the use of this document. If the user of this document is unsure whether the material is correct, they should refer directly to the relevant legislation and contact the NZ Transport Agency.

More information …

NZ Transport Agency
October 2011

ISBN 978-0-478-38069-9

If you have further queries, call our contact centre on 0800 699 000 or write to us:

NZ Transport Agency
Private Bag 6995
Wellington 6141

This document is available on the NZ Transport Agency's website at www.nzta.govt.nz.

Record of amendment

Amendment number	Description of change	Effective date	Updated by

Acknowledgements

- Funding for this study came from the NZ Transport Agency (NZTA) for which we are grateful.
- Thanks to Janine Wech and Braden Crocker for assistance in the field, and to Janine Wech, Paul Lambert, and Laura Drummond for processing all the invertebrate samples.
- The assistance of all landowners in allowing us access to the various streams throughout the country is also gratefully acknowledged.
- Thanks to Jennie Gadd (NIWA Auckland) and Helen Rouse (NIWA Christchurch) for review comments.

Abbreviations and acronyms

AADT	Annual average daily traffic
ANOSIM	Analysis of similarities
ANOVA	Analysis of variance
ANZECC	Australia and New Zealand Environment and Conservation Council
EPT	Ephemeroptera, Plecoptera, Trichoptera (i.e., mayflies, stoneflies, caddisflys)
EPT*	EPT (without Hydroptilidae caddisflys)
MCI	Macroinvertebrate community index
MOT	Ministry of Transport
NMDS	Non-metric multidimensional scaling
NZTA	New Zealand Transport Agency
PAHs	Polycyclic aromatic hydrocarbons
QMCI	Quantitative macroinvertebrates community index
SRE	Sensitive receiving environments
VEF	Vehicle emission factor
VKT	Vehicle kilometre travelled

Foreword

During the past 15 years the Ministry of Transport (MoT), NZ Transport Agency (NZTA) and its legacy agencies (Transit New Zealand, Transfund NZ and Land Transport New Zealand) and the Road Controlling Authorities Forum undertook a number of research projects on the environmental impacts of stormwater runoff of road transport. This work was initiated by MoT reports which indicated transport derived storm water runoff from roads could have adverse ecological impacts. Consequently, NZTA conducted a nation-wide risk assessment, researched international literature, conducted marine sediment, and fresh water sampling programs to evaluate magnitude of the effects. This report is on the fresh-water sampling programme. Below is a summary of work to date that led to this program.

Historical perspective on transport stormwater run off

1996 - MoT released a discussion paper on environmental externalities[1] as part of the Land Transport Pricing Study (MoT 1996). MoT reported the contribution of road runoff to total contaminant discharges to freshwater and marine environments was unclear, but was considered to account for 40-50% of urban metal contamination to aquatic ecosystems.

2002-2004 – MoT commissioned a series of investigations into the types and loads of contaminants derived from road transport, and their effects on freshwater and marine ecosystems.

2007 - In response to regulatory concerns, MoT's report and Transit's Strategic Plan 2004, a stormwater management retrofit program was established with the objective of achieving tangible improvements to the quality of stormwater discharged from critical parts of the state highway network that may be affecting sensitive water bodies. A GIS based screening tool was developed to identify regional water bodies at most risk based on catchments and VKT.[2]

2008 - Based on the above screening tool the five most heavily impacted sites with similar depositional area draining to sensitive receiving environments were thoroughly sampled and analysed[3] using a chemical fingerprinting technique which allowed differentiation of the contribution of road derived vs urban runoff. The study unexpectedly found road contaminant levels were well below ANZECC guidelines for marine sediment, possibly because of inter-tidal re-suspension and dispersion.

2009 - An international review of transport-derived storm water road runoff contaminant discharge models found two approaches. The US Department of Transportation model [4] (based on 993 events at 31sites) is mainly concerned with streams and lakes. The UK model[5] (based on 340 events at 30 sites) considers road runoff discharge from bridges and potential spills. Both models rely on water quality data not available in New Zealand. The UK model suggested treatment of discharges from sites with less than 30,000 AADT would not achieve measurable improvements. The US model suggests effects were not expected until 50,000 AADT was

[1] Land Transport Pricing Study: Environmental Externalities. Ministry of Transport, Wellington.

[2] Preliminary Screening of Sensitive Receiving Environments at Risk from State Highway Runoff. Transit NZ (2007)

[3] Reed et al (2008)
[4] Pollutant loading and impacts from highway stormwater runoff. Four volumes, publication No. FHWA-RD-88-006. (1990)

[5] Highway Agency water risk assessment tool. UK Highways Agency HA 216 (2008)

exceeded. Both models found poor correlation between AADT and contaminant discharge (< 40% variation explained.)

2011 - A monitoring network was proposed using the macro-invertebrate community index (MCI) as a cost effective measure to indicate water quality (This study). The use of the MCI is a similar approach to the use of nitrogen dioxide as an indicator for air quality in the successful NZTA Ambient Air Quality (Nitrogen Dioxide) Monitoring Network.

The results of this report will be used to make an informed decision as to whether an MCI monitoring network is a feasible tool to monitor the effects of state highway runoff on water quality.

Contents

Executive Summary

1. The NZ Transport Agency (NZTA) has done considerable work assessing the contaminant loadings from state highways. Although most of this work has focused on quantifying the nature and magnitude of contaminants being generated by highway runoff, considerably less work has focused on assessing the impact that these contaminants have on receiving systems. Indeed, in a review of the effects of road transport on freshwater and marine ecosystems, Kennedy (2003) noted that "*a search of the literature revealed that there has not been any published data describing the specific effects of road runoff (in contrast to urban runoff) on benthic [i.e., freshwater] invertebrates in New Zealand*." Freshwater invertebrates are a key ecosystem component that may be adversely affected by road runoff. The presence or absence of different invertebrates in a stream can illustrate its ecological condition. Given their ongoing and proven use in detecting changes in stream health as a result of pressures such as land-use changes, monitoring freshwater invertebrates may prove useful to assess impacts of state highway derived road runoff.

2. This study had two main aims. Firstly, to determine whether there was a demonstrable ecological effect of state highway derived road runoff on freshwater invertebrate communities. Secondly to develop protocols to detect potential ecological effects of road runoff on freshwater ecosystems for biological monitoring programmes throughout the country.

3. We examined the invertebrate communities above and below state highways in six streams potentially exposed to road runoff. Streams were selected based on seven criteria: 1) having enough potential contaminant load (an average annual daily traffic (AADT) loading greater than 10,000 vehicles per day); 2) having a small enough flow to minimise dilution; 3) being located in areas where the dominant contamination was from state highway runoff, and not from urban or agricultural sources; 4) having either natural substrates (gravel-bed streams) from which we could collect quantitative samples, or else in soft-bottomed streams being able to deploy artificial substrates in; 5) having similar habitats above and below the highway; 6) having similar riparian vegetation, and shade regime at sites above and below the state highway; 7) not flowing into other streams within 200 to 300 m below the road bridge in order to sample from 2 separate sites below the highway. Based on these criteria, two sites (Mangaone Stream and Smith Creek) were chosen on the Kapiti Coast that flowed under State Highway 1 (SH1), two sites (Pauatahanui Stream and a small unnamed stream) near Upper Hutt that flowed under State Highway 58 (SH58), and two near Warkworth (the North and South branches of the Mahurangi River) that flowed under SH1.

4. A central part of this work was to quantify potential differences to invertebrate density above and below the state highway. Because invertebrate communities vary greatly at small spatial scales, it was necessary to collect sufficient replicate samples to accurately estimate population densities at each sampling site. A power analysis was first done on boulders collected from the small stream on SH58 to determine how many replicate samples needed to be collected to accurately detect a 50% reduction in invertebrate density between different sites, with an 80% certainty. Results of this power analysis showed that 15 replicate samples were required in order to be able to detect differences with sufficient rigour. This number of samples also ensured the collection of the maximum number of different species of invertebrates at each site.

5. Quantitative samples were collected from natural substrates in three of the streams. It was not possible to adequately sample natural substrates from the unnamed stream flowing under SH58 as there were not enough boulders within the small areas we had identified as individual sites. Instead, small paving stones were deployed there as an artificial substrate and left for 10 weeks prior to sampling. After this time, the paving

stones were colonised by a variety of invertebrates that mimicked those on the real boulders that were collected for the power analysis. The two streams near Warkworth were characterised by their soft-bottomed substrates that precluded the use of normal quantitative sampling techniques. Artificial macrophytes were constructed and deployed at these sites and left in each stream for eight weeks prior to sampling. All invertebrate samples were processed to obtain data on species composition and abundance.

6. Physical habitat measurements such as depth, velocity, and substrate size were also recorded at each site, and stream temperature monitored over the summer by dataloggers placed immediately upstream and downstream of the state highway. Samples of periphyton (algae) and mosses were also collected from three of the streams in the Wellington region at sites upstream and downstream of the state highways to see whether these plants had accumulated metals arising from road runoff.

7. We tested two *a priori* hypotheses of expected changes to invertebrate communities from stormwater derived runoff. Firstly: densities of sensitive taxa would decline at the site immediately below the highway, and then increase with increasing distance downstream from the highway. Secondly: densities of tolerant taxa would increase at the site immediately below the highway, reflecting a reduction in densities of sensitive taxa. A number of biotic indices describing aspects of the invertebrate community were calculated from the data. These indices, plus the densities of the 10 most common taxa were examined using nested Analysis of Variance to see whether they differed upstream and downstream of the state highways. Ordination was also used to graphically represent differences between samples based on their invertebrate communities. Finally, Analysis of Similarity tested whether any significant differences between the ordination groups were evident.

8. A diverse invertebrate fauna was found in Mangaone Stream, Smith Creek, and Pauatahanui Stream. This fauna was dominated by caddisflies *Aoteapsyche* and *Pycnocentria,* the mayfly *Deleatidium*, midges (*Cricoctopus, Eukiefferiella, Naonella* and *Tanytarsus*), Elmid riffle beetles oligochaete worms, and the common snail *Potamopyrgus*. Few differences in invertebrate communities were evident upstream and downstream of the state highways in Mangaone Stream and Pauatahanui Stream. A number of biotic metrics, and densities of 4 taxa differed upstream and downstream of SH1 at Smith Creek, in ways that were consistent with the expected effects of stormwater derived runoff. Ordination of data from Smith Creek also showed large differences between samples collected above and below the state highway, but only moderate differences existed between sites immediately above and below the state highway, suggesting that any potential stormwater runoff entering the stream was having relatively minor effects on the overall invertebrate composition.

9. Analysis of periphyton and aquatic moss material from Mangaone Stream, Smith Creek, and Pauatahanui Stream showed some evidence of metal uptake at sites below the state highways, although in relative terms this increase was very small. The most noticeable differences in metal concentrations occurred in periphyton collected from Smith Creek, where samples collected below the road bridge displayed higher levels of copper, lead, nickel and zinc. However, these elevated levels were very low when compared to published levels found in periphyton exposed to road runoff in other streams, and considered unlikely to have a large effect on invertebrate communities as metal levels were well below ANZECC sediment guidelines. These results suggest that the changes to invertebrate composition at the site were unlikely to be caused by metal contamination. An alternative explanation for the observed changes to the invertebrate communities may instead reflect presence of a sheep pass under the road next to the stream that may have resulted in localised sediment or nutrient enrichment.

10. The paving stone substrates placed in the small stream running under SH 58 also supported a diverse fauna, dominated by orthoclad midges, five genera of mayflies (*Neozephlebia, Deleatidium, Coloburiscus, Zephlebia,* and *Nesameletus*), two genera of

caddisflies (*Oxyethira* and *Helicopsyche*), the snail *Potamopyrgus*, and the toe-biter *Archichauliodes diversus*. No significant differences in the invertebrate communities were found above and below the road bridge. The artificial macrophyte substrates placed in the North branch and South branch of the Mahurangi also supported a diverse fauna, dominated by the snail *Potamopyrgus,* oligochaete worms, the midges Orthocladiinae and *Tanytarsus*, the blackfly *Austrosimulium*, the amphipod *Paracalliope*, and the mayfly *Zephlebia*. No strong and consistent differences existed in invertebrate communities above and below the state highways, suggesting that any stormwater derived contamination was having little or no effect on invertebrate communities there.

11. The salient results of this study were that invertebrate communities in the five of the six streams monitored showed little evidence of being affected by runoff from state highways, despite high traffic densities, and despite the fact that all the streams sampled were in good ecological condition and unaffected by stresses associated with urban runoff. Changes to the invertebrate communities in the other stream (Smith Creek) as a result of road runoff were regarded as minor, at most. Likely reasons behind the lack of a strong consistent signal of road runoff most likely reflects a combination of the generally smooth-flowing vehicle behaviour resulting in lower emissions, and the presence of vegetated roadside drains that road runoff flowed into that minimised the direct conveyance to the streams and potentially reduced contaminant loads.

12. Aquatic invertebrates were successfully used to monitor effects of road runoff in a variety of streams, suggesting that such monitoring could be used elsewhere. Two flowcharts are presented to allow managers to firstly decide when biological monitoring of streams flowing under state highways is appropriate that deals with a number of issues that need to be considered prior to implementing a national invertebrate monitoring programme. Firstly is the need to sample streams which have not been impacted by other pressures such as those associated with intensive agriculture or urbanisation, as such activities are likely to mask any effects of road runoff. Secondly, monitoring is best done only in sites where a potential contamination source will be high. Thus, information is needed on either the AADT, or more preferably loadings expressed at vehicle kilometre travelled (VKT), or vehicle emission factors (VEFs). Thirdly, there is little point in monitoring receiving environments such as large, fast-flowing rivers that are not sensitive. Fourthly, our results suggest that adverse effects of road runoff are unlikely where the pathway of road runoff into the stream is diffuse.

13. A second flowchart is designed to help determine the most appropriate sampling methodology, commencing with an initial screening of periphyton or other aquatic plants above and below highway crossings. If metal concentrations at elevated below the road, an invertebrate monitoring program may be instigated. The influence of the nature of the stream bed in determining the resultant sampling methodology is highlighted . Although standard quantitative sampling can be used in gravel bed streams, the use of artificial substrates in soft bottomed streams is discussed and recommended.

1 Introduction

1.1 Purpose and objective

The NZ Transport Agency (NZTA) has done considerable work assessing the contaminant loadings from state highways. Most of this work has focused on quantifying the nature and magnitude of contaminants being generated by highway runoff (e.g., Macaskill and Williamson 1994; Kennedy and Gadd 2000; Moncrief and Kennedy 2002; Kouvelis and Armstrong 2004), but considerably less work has focused on assessing the impact that these contaminants have on receiving systems. Kennedy (2003) reviewed the effects of road transport on freshwater and marine ecosystems, and discussed the use of biological monitoring to evaluate changes in the environment that may have occurred as a result of road runoff. Kennedy highlighted the fact that the potential effects of road transport related pollutant sources on instream biota are poorly understood, with few tools available for freshwater managers for assessing specific effects of road runoff. This is in sharp contrast to the increasing number of studies undertaken to 1), quantify the nature of road runoff; 2) identify which parts of the road network that may require stormwater contaminant control; 3) quantify the effectiveness of existing stormwater contaminant control devices (e.g., Moores et al 2010). Although the review report of Kennedy (2003) highlighted many studies examining the effects of road runoff on freshwater ecosystems, many of these were undertaken in urban environments where streams are exposed to multiple pressures from a variety of different contaminant sources. Indeed, Kennedy noted that "*a search of the literature revealed that there has not been any published data describing the specific effects of road runoff (in contrast to urban runoff) on benthic invertebrates in New Zealand*."

Although the quantification of contaminant loadings into receiving systems is important, it does not necessarily indicate whether an ecological problem is caused by these contaminants, or what the magnitude of the impact is. As such it provides purely and indication of an effect, but gives no indication as to the magnitude of the effect. This project was therefore undertaken to determine firstly whether there was a demonstrable ecological effect of potential state highway derived road runoff on freshwater invertebrate communities. Secondly, it is aimed at developing a series of protocols to be used for other biological monitoring programs throughout the country aimed at detecting potential ecological effects of road runoff on freshwater ecosystems.

1.2 Background

Effects of stormwater runoff from state highways relate to the following impacts:

- Hydrological effects, and
- Water quality effects.

1.2.1 Hydrological effects

Hydrological effects relate to the quantity of stormwater runoff discharged as a result of increased impervious surface. Having impervious surfaces increases the total volume of stormwater being discharged, and consequently the frequency of water being discharged at specific volumes when compared to equivalent areas without impervious surfaces. These two factors increase the instability of a freshwater stream's physical structure, which may adversely impact stream biological health through an increase in stream disturbance via substrate movement and streambed scouring. Research on stream physical structure related to degree of catchment imperviousness (Herald 1989) indicates that stream cross-sectional area increases as catchment urbanisation increases as shown in Figure 1. Urban land cover is directly correlated to catchment imperviousness. The figure only looks at channel cross sectional area and thus is an indication of water quantity impacts on channel physical structure.

These close relationships between stream hydrology and imperviousness mean that monitoring flow (discharge and velocity) in streams subject to road runoff can provide information on the effects that increased stormwater discharge can have on the receiving system's physical structure. However, as mentioned above, the relationship between changes in stream hydrology and aquatic ecosystem health is indirect.

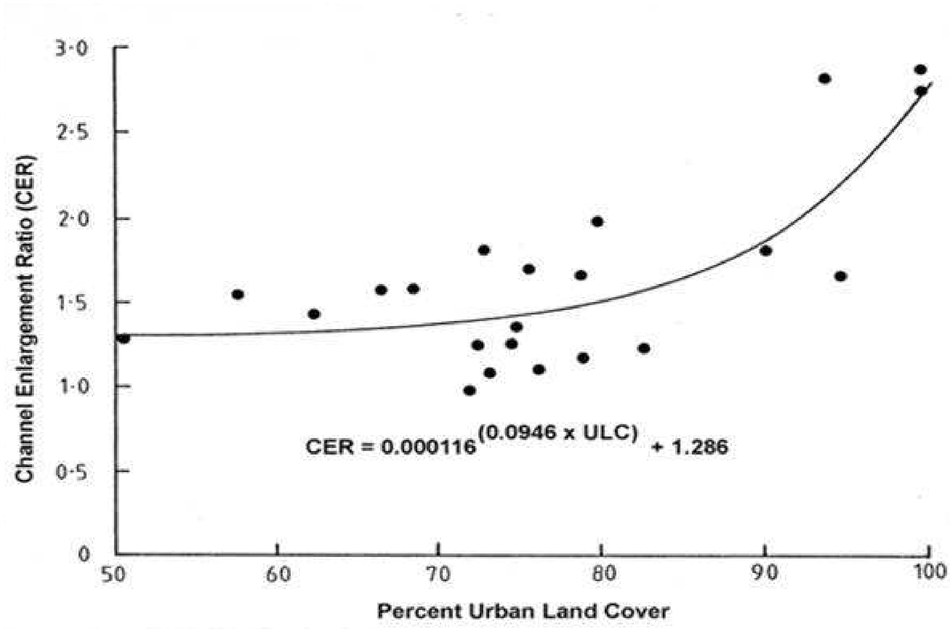


Figure 1Channel enlargement versus percent urban land cover

1.2.2 Water quality effects

Monitoring water quality from state highways is a common way to assess the level of contaminants that are being discharged to a receiving system, and many studies have documented the nature of road runoff contaminants and their impacts on water quality (e.g., Macaskill and Williams 1994; Kennedy 2003; O'Riley et al 2002; Moores et al 2010). Key contaminants of concern include metals such as copper and zinc (derived from brake linings and tyres, respectively), hydrocarbon compounds such as oil, grease and fuel (from leakages, spills and exhaust systems), and polycyclic aromatic hydrocarbons (PAHs), toxic compounds present in vehicle oils and fuels, and which are discharged from both exhaust and non-exhaust systems (Kennedy 2003).

Characterising the nature and quantity of runoff contamination is, however, only the first step in understanding its environmental effects. Little work has been done on identifying locations of contaminant deposition (if any), or on the impacts these contaminants have on ecosystem health. Given the fact that the road network traverses most of New Zealand, and that subsequent road runoff is such a fundamental feature of these roads which will discharge into potentially sensitive receiving environments, it is evident that attention be turned on assessing the ecological impacts of road runoff.

1.3 Measuring environmental effects

Despite the known adverse effects that road runoff can have on water chemistry, it is sometimes difficult to determine whether adverse ecological impacts of stormwater runoff from state highways are more related to changes in stream hydrology, stream physical structure, or water or sediment quality, either alone or in conjunction. There are three main components of New Zealand freshwater ecosystems that may be adversely affected by road runoff: algae (periphyton), fish, and invertebrates. Periphyton is the 'slime' that covers stones, wood, weeds or any other stable object in streams. Its main constituents are algae (e.g., diatoms, filamentous green algae) and cyanobacteria (blue-green algae), but it may also contain fungi and organic material. Its extent ranges from a barely visible film to extensive proliferations of green or brown filaments or mats. Periphyton communities are important for two main reasons. First, they are an energy source for higher levels in the food chain – invertebrates eat periphyton. Second, they can degrade the environment when they proliferate and this can have negative implications for the ecosystem and for human values such as visual aesthetics, recreation, stock-water and irrigation use.

Studies on the effects of road runoff on periphyton have shown mixed results. Johnson et al (2011) found that road runoff did not detrimentally affect periphyton growth or biomass, although Johnson et al found that community composition did change as a result of runoff. Boisson et al (2005 and 2006) also found little difference in periphyton growth exposed to road runoff, despite the fact that many substances present in road runoff could in theory stimulate periphyton growth. Lack of growth stimulation may have reflected amongst other things presence of heavy metals in the runoff which may reduce growth. Although the effects of road runoff on periphyton communities are equivocal, work by Suren and Elliot (2004) has shown that heavy metal contamination of road runoff may have other ecological consequences because it is taken up by periphyton. Indeed, levels of copper and zinc in periphyton collected from a stream draining an industrial estate in Christchurch exceeded ANZECC high trigger interim sediment quality guideline values, and were greater than metal concentrations in the sediment. A similar finding was reported by Davis and George (1987) whereby copper concentrations in the filamentous green algae *Cladocera conglomerata* were significantly higher below a discharge point from the M11 motorway in Essex, UK, than above. Thus, although road runoff may not adversely affect periphyton, contamination of periphyton food sources may have flow on effects to other parts of the ecosystem.

Although periphyton may be affected by road runoff, or may be contaminated by heavy metals, its use as an indicator to detect effects of road runoff may be somewhat limited. This reflects the fact that periphyton growth is strongly related to light regime, and will differ greatly in shaded and unshaded rivers. In addition, periphyton growth is strongly influenced by a river's flow regime, whereby even relatively small floods can remove periphyton from streams. Although periphyton growth often quickly recovers from flood events, it results in a highly variable community over time, which may compound the identification and interpretation of monitoring programs designed to detect effects of road runoff on periphyton communities. To date, no studies have assessed the impact of road runoff on periphyton communities in New Zealand, so we cannot categorically state the usefulness of this approach.

New Zealand rivers and streams are home to approximately 60 species of fish, of which 35 or so are native (McDowall 1990). Fish have important biodiversity, cultural and recreational values, and are found in most waterways throughout the country. Most fish are restricted to waterways in the foothills and lowland regions of the country: the same areas that contain the majority of the country's state highway network. Fish may be affected by road runoff, either directly or indirectly, and so can be used to assess aquatic system health, either by examining fish populations and abundance, or by doing assays on fish tissues to detect road derived contaminants. However, beyond evaluating changes in fish community composition as a result of urban runoff, little direct evidence exists on the effect of road derived runoff on fish communities (Paul and Meyer, 2001), although a recent study by Johnston et al (2011) showed that stormwater from a motorway with an AADT of approximate 61,000 in Western Michigan, USA, had little effect on survival or growth of pumpkinseed fish (*Lepomis gibbosus).*

Furthermore, the highly mobile nature of fish means that it may be difficult to detect an impact. This is especially so in New Zealand given that many of the native fish are diadromous; i.e., they move between the sea and the land as part of their life cycle. Thus, although they may be briefly exposed to road derived runoff, they are likely to spend a large part of their life in the upper, steep headwater reaches of catchments which are not subject to runoff from state highways. Their use to monitor and assess impacts of road runoff is thus likely to be somewhat limited.

The third ecosystem component that may be adversely affected by road runoff are the freshwater invertebrates. These animals generally comprise the larval (and occasionally adult) stages of insects, along with molluscs (snails, bivalves), crustaceans, and worms, and are a vital component of foodwebs in rivers and streams. Their importance lies in their role of transferring plant-based organic carbon (e.g., leaves, wood, periphyton) into animal-based organic carbon, which is then available to higher predators such as fish and birds. Unlike algae, freshwater invertebrates are relatively long-lived, and can spend months to years living in streams. They are also not washed away by small floods as easily as algae are. Finally, they are not as mobile as fish, making it possible to characterise their population densities in streams with a certain degree of accuracy. They are also relatively easy to identify, and the ecological tolerances of different invertebrates are relatively well known. Thus, the presence or absence of different invertebrates in a stream tells a lot about the ecological condition of that stream.

Because of these factors, many organisations throughout the world including New Zealand use freshwater invertebrates to determine the ecological condition of waterways (Barbour et al 1996; Rosenberg and Resh 1993; Stark 1985; Stark et al 2001). Furthermore, macroinvertebrate monitoring has been used extensively to determine effects of catchment development (Hall et al 2001; Lenat and Crawford 1994; Quinn and Cooper 1997), and to detect the effects of point source and diffuse such as those associated with road-runoff and urban development (e.g., Rosenberg and Resh 1993; Barbour et al 1996). Adverse effects of urban development on stream macroinvertebrates in particular are documented worldwide, and authors have consistently reported a reduction in biodiversity values of streams exposed to urban runoff (Hall et al 2001; Paul and Meyer 2001; Suren and Elliot 2004; Walsh 2000; Walsh et al 2005). Many studies have also illustrated negative correlations between percent impervious land cover and indices of biological health, (e.g., Hilsenhoff 1988; Jones and Clark 1987; Lenat and Crawford, 1994; Pitt, 2002; Pitt and Bozeman, 1983). An emerging paradigm is that adverse impacts of urbanisation on streams can occur with as little as 10 – 20% impervious surfaces (Paul and Meyer 2001; Beach 2002). This has been described as the "Urban Stream Syndrome" (Meyer et al 2005; Walsh et al 2005), which describes amongst other things, the loss of mayfly and stonefly taxa, and a shift in community to tolerant taxa such as oligochaetes, snails and midges. Within New Zealand, the snail *Potamopyrgus antipodarum*, Oligochaeta, midges, ostracods, the blackfly *Austrosimulium* and the hydroptilid caddisfly *Oxyethira* are often the only invertebrates commonly found in urban streams (Hall et al 2001; Maxted et al 2003; Suren 2000). Sensitive aquatic insect taxa such as mayflies, stoneflies and many caddisflies (i.e., Ephemeroptera, Plecoptera and Trichoptera (EPT)) are absent or scarce. For instance, Allibone et al (2001) recorded 11 EPT taxa at sites near Auckland with low (10%) catchment imperviousness, but < 2 EPT taxa at sites with > 30% imperviousness. Sites with > 40 % catchment imperviousness supported no EPT taxa (Figure 2).

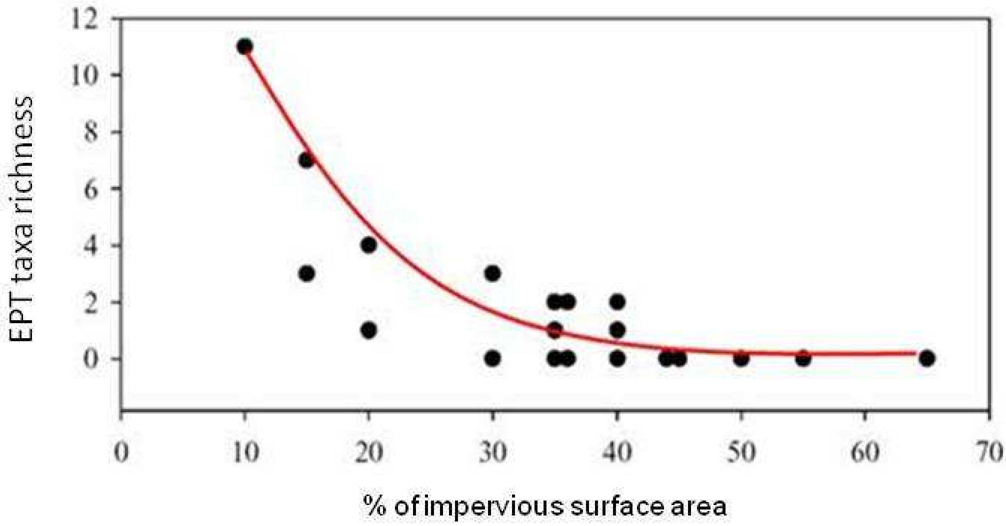


Figure 2 Plot of the response of sensitive invertebrate populations (as the number of Ephemeroptera, Plecoptera and Trichoptera taxa) related to the percentage of impervious surface area in a catchment

Although these studies do show strong effects of urban runoff on stream ecosystems, it must be remembered that urban runoff comes from a huge variety of sources, including state highways, local roads, parking lots, industrial areas, and both commercial and residential roof spaces. Whether such a strong effect is detected from runoff only from state highways in a non-urban environment is unclear. However, given their ongoing and proven use in being able to detect changes in stream health as a result of pressures such as land use changes, invertebrate communities may prove useful to assess the impacts of state highway derived road runoff on streams. We thus examined the macroinvertebrate communities in a number of streams potentially exposed to State Highway derived road runoff to see whether indeed this was having a demonstrable adverse effect.

1.4 Stream macroinvertebrate community monitoring

Aquatic invertebrates are widely used to monitor the ecological condition of waterways as outlined above, reflecting:

- their more or less ubiquitous nature;
- their relative ease of collection;
- the ability to identify these animals easily;
- their relatively long life stages, meaning they integrate antecedent conditions; and
- the fact that they are relatively sedentary, and do not move particularly great distances over time.

As mentioned, these organisms are widely used by many organisations to assess the ecological condition of waterways. Stark et al (2001) published New Zealand-based protocols for sampling and processing macroinvertebrates from streams, and this has further increased the attractiveness of using these organisms to assess the ecological condition of waterways.

A central assumption underlining the use of macroinvertebrates to monitor the ecological condition of streams is that the community present at a particular site is a product of its current and antecedent environmental conditions. Thus, it is assumed that significant variation in community composition will occur between, for example, polluted and unpolluted waters as a result of the changes in invertebrate community composition as a result of different tolerances or habitat preferences of each individual invertebrate species. A major problem and challenge for this method is a need to convey the somewhat complex community composition information into a series of simple biotic metrics and summarise certain attributes of the macroinvertebrates community. A number of such metrics have been developed in New Zealand to help simplify the often complex ecological data into simple numbers, which are more understandable by managers.

The first of these metrics is the macroinvertebrates community index (MCI), which is commonly used as an indicator of water quality in stony streams (Stark, 1993). This metric was developed by assigning tolerance values to a range of macroinvertebrates found in stony bottomed streams in the Taranaki ring plain exposed to organic enrichment from dairy sheds. Taxa which were intolerant of enriched conditions were assigned high tolerance scores, while taxa which were tolerant of such conditions were assigned low scores. Tolerance scores range from 1 to 10. The MCI and its quantitative variant (QMCI) were calculated as follows:

$$\text{MCI} = 20 \times [\Sigma\, a_i / S]$$

$$\text{QMCI} = \Sigma\, (n_i \times a_i) / N$$

where a_i = tolerance score for the i^{th} taxon, S = total number of taxa, n_i = the number of individuals in the i^{th} taxon, and N = the total number of individuals (Stark 1993). Calculated MCI scores can theoretically range from 20 (i.e., only one taxa with a tolerance score of one) to 200 (only one taxa with a tolerance score of 10). Calculated MCI scores above 120 are considered to represent streams in excellent condition, whereas scores below 60 are considered to indicate streams in highly degraded conditions. Similarly, calculated QMCI scores can range from 1 to 10, with streams having scores greater than seven representing streams in excellent condition, and streams having scores less than two representing highly degraded streams.

Stark developed the MCI tolerance scores in stony bottomed streams, and recognised that these scores do not work well in soft bottomed streams. In response to this, Stark and Maxted (2007) developed tolerance values for invertebrate taxa inhabiting soft bottomed streams. In doing so they created the soft bottomed versions of these metrics (MCI_sb and QMCI_sb).

Two other commonly used metrics to describe the invertebrate communities is the number and % of EPT (Ephemeroptera, Plecoptera, Trichoptera) taxa in a sample. These metrics convey useful information about overall invertebrate community composition and condition, as many species of these insect groups are sensitive to pollution, and show reductions in density in sites affected by contamination such as excess nutrient enrichment, or heavy metals. Furthermore, as sediment loads, or algal biomass, increase, the number or % of EPT taxa often declines. The exception to this is for the two Hydroptilidae caddisflys *Oxyethira* and *Paroxyethira*, which are common in streams dominated by high algal biomass, and are generally regarded as being highly tolerant of organic enrichment. As such, the number and percentage of EPT is often calculated without these animals (commonly referred to as EPT* and % EPT*).

Hickey and Clements (1998) investigated the effects of heavy metals from current and closed goldmines on Coromandel peninsula by sampling invertebrate communities above and below these discharges. They showed that the abundance and species richness of mayflies, the number of EPT taxa, and total taxonomic richness were the best indicators of heavy metal contamination in New Zealand streams. Of interest was the finding that the MCI and QMCI did not differ significantly in samples collected above and below areas receiving mine drainage. Hickey and Clements thus recommended that these metrics not be used, or be used with great caution when assessing the effects of metal contamination on streams.

In the current study, the ecological effects of road runoff arising from state highways were assessed by examining the macroinvertebrates communities in selected streams. A number of the metrics discussed above were used to investigate differences between macroinvertebrate communities upstream and downstream of state highways.

2 Methods

2.1 Introduction and hypotheses

Highway derived stormwater runoff contains a variety of toxic metals such as copper and zinc, as well as a number of other potentially toxic material such as hydrocarbons and polycyclic aromatic hydrocarbons (PAHs). These contaminants are expected to reduce densities of at least some sensitive taxa downstream of the state highway, whereas densities upstream would be higher, all other things being equal. We would also expect some form of recovery at the lowermost site if runoff was having a localised effect, and if metal or hydrocarbon contamination was decreasing from its source (the highway) at these lower sites. A fundamental part of this work was, therefore, the requirement to detect a difference in invertebrate density upstream and downstream of state highways. Invertebrate communities may also change naturally along a river, so to minimise this effect on our analysis, we sampled communities at two locations above, and two locations below selected state highway crossings to see if they differed at this spatial scale. In this way, any natural differences between pairs of sites above or below highway crossings would be accounted for in the analysis. We sampled sites as close as possible to the state highway, but within 50m of the highway bridge crossings. Additional sites were collected within 2-300m from these.

Studies by Shutes (1984) and Maltby et al (1995b) showed a reduction in invertebrate diversity and a loss of sensitive invertebrate taxa such as stoneflies, amphipods caddisflys and snails in streams below motorways in England. Moreover, work assessing the impact of heavy metals on invertebrate communities (e.g. Hickey and Clements1998), has shown that sensitive taxa such as mayflies, stoneflies and caddisflys are less common in streams exposed to metal contamination. These results allowed us to develop two *a priori* hypotheses of expected changes to invertebrate communities from stormwater derived runoff. The first hypothesis was that densities of sensitive taxa such as members of the EPT community would decline at the site immediately below the highway, and then increase with increasing distance downstream from the highway. This increase would reflect a reduction in the concentration of potential contaminants with increasing distance from the source. The second hypothesis was that densities of tolerant taxa would increase at the site immediately below the highway, reflecting a reduction in densities of sensitive taxa. Densities of tolerant taxa at the lowermost site would then decline to levels found at upstream sites as a result of the increase in numbers of sensitive taxa here (Figure 3).

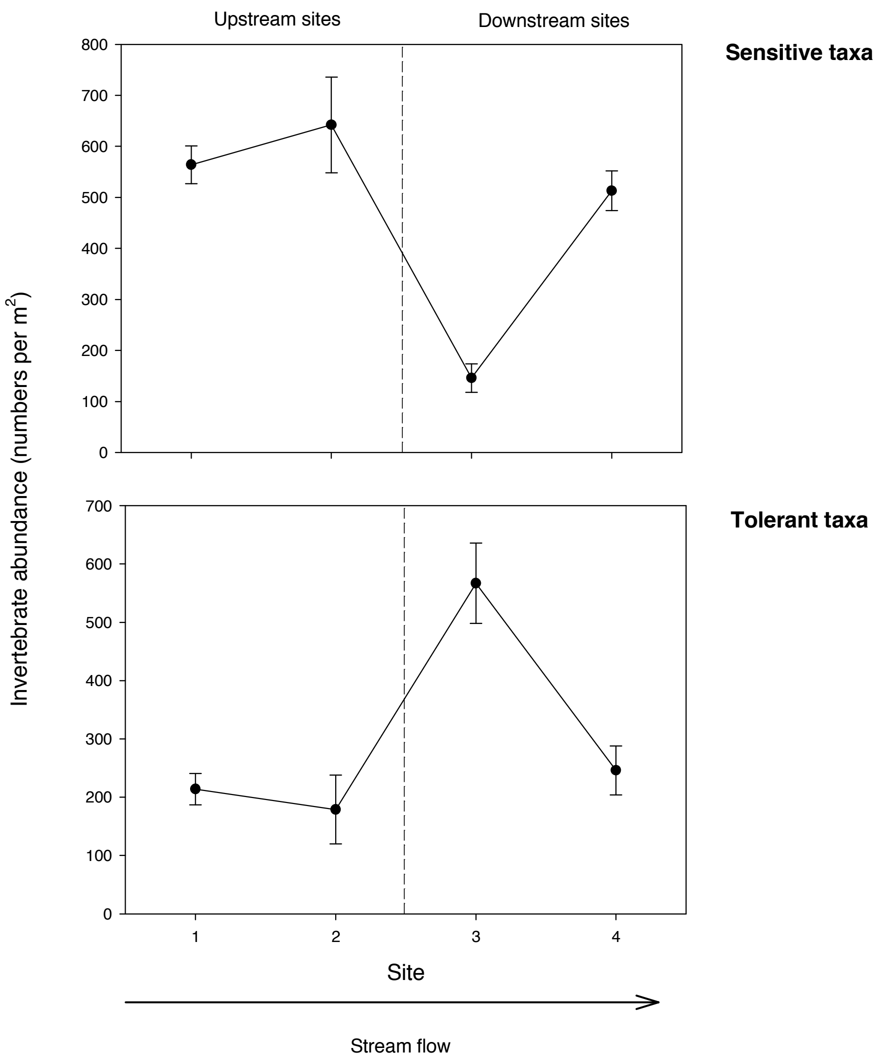


Figure 3. Theoretical changes to invertebrate density above and below state highway bridges showing the expected decrease in densities of sensitive taxa, or increases of tolerant taxa at the site immediately below the highway bridge. Note how this analysis assumes that densities do not change between sites above the highway, meaning that small-scale differences and habitat variables are not having a large effect on invertebrate communities within the study reach.

2.2 Stream selection and location

Gardiner and Armstrong (2007) and Gardiner et al (2008) developed a GIS based model that identifies potential sensitive receiving environments (SREs) at risk from state highway runoff. Most of these receiving environments included streams in the Auckland, Wellington and Canterbury regions, although lakes in the Bay of Plenty (e.g., Lake Rotorua and Taupo) were also identified. Sensitivity was defined as a combination of the physical nature of the receiving environment (i.e., depositional versus dispersive), ecological values, and human values. A central part of the GIS model developed by Gardiner and Armstrong (2007) was built around the *source - pathway - receptor* risk model. Here, the *source* is the road network that generates a contaminant load in runoff, the *receptor* is the water body (i.e. the receiving environment) and the *pathway* is the route taken by runoff from the point it leaves the road to the point that it reaches the receiving environment. Gardiner and Armstrong maintain that for a risk to be present to a water body, all three components must exist. The magnitude of the risk to the receiving environment was thought to depend on the strength of the source, the conductivity of the pathway, and the sensitivity of the receiving environment. Gardiner et al used AADT as a good surrogate for potential contaminant load. Maps created by the Gardiner et al GIS model

(Figure 4) were subsequently examined for suitable sites. A number of waterways highlighted in the Gardiner report (for example Lucas Creek, Otara Creek) were present in heavily urbanised areas, and as such would be affected by stormwater derived from areas other than the state highways. Potential sites were selected based on the following criteria:

- having enough potential contaminant load (from a state highway with an AADT loading greater than 10,000 vehicles per day);
- small enough not to cause substantial dilution or movement of contaminants away from the source;
- being in areas where the dominant contamination arises from state highway runoff, and not from urban or agricultural sources;
- Ideally having a stony streambed, as these samples will be easier to collect and process than would be the case for soft bottomed streams;
- similar habitats above and below the highway;
- similar riparian vegetation, and shade regime; and
- not flowing into other streams within 200 to 300m below the road bridge to allow multiple sampling sites below the state highway.

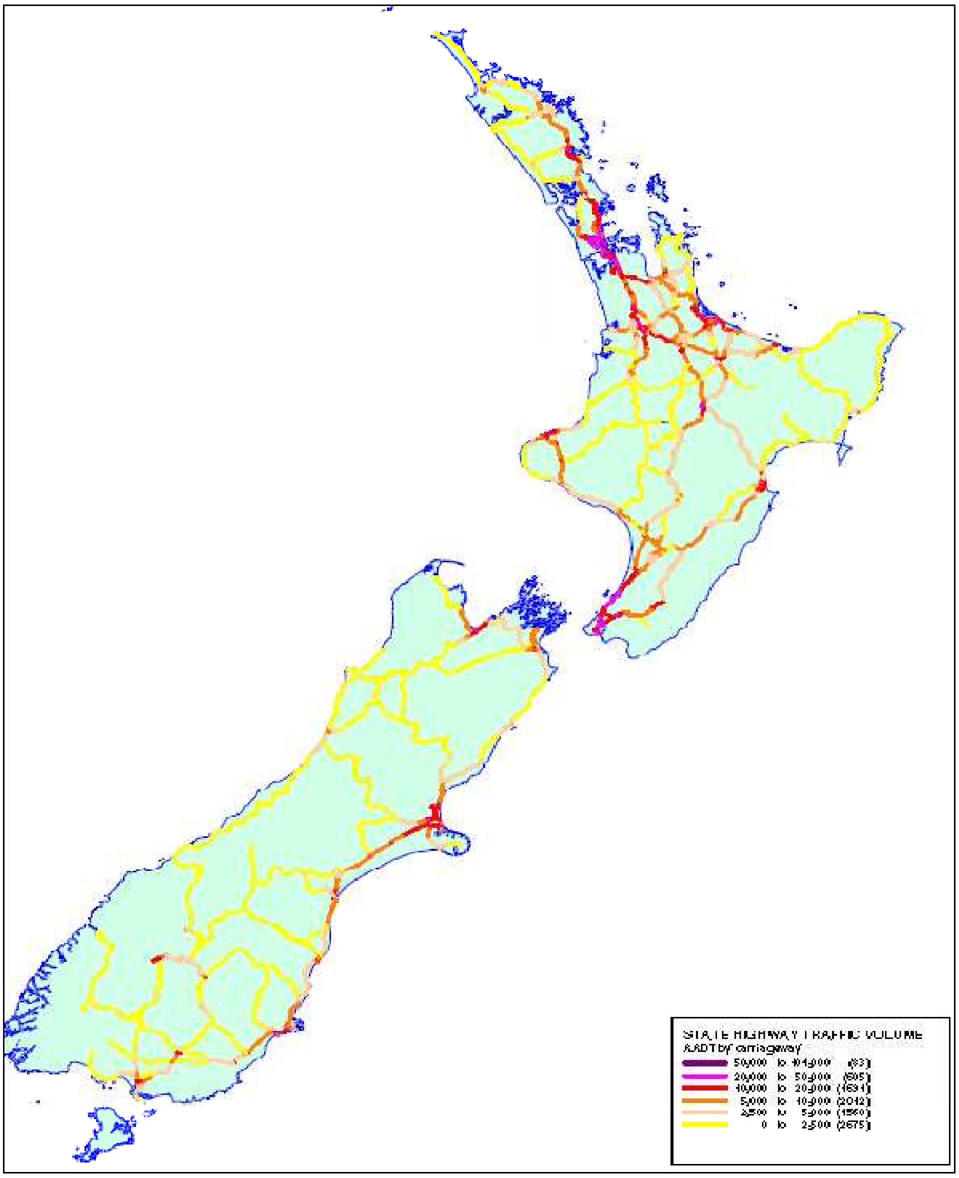


Figure 4 Map of modelled Average Annual Daily Traffic (AADT) showing the regions of greatest traffic use, from Gardiner et al (2008).

Examination of GIS maps showed the most likely locations were north of Auckland, on the Kapiti Coast, and near Upper Hutt. Field visits identified two suitable sites on the Kapiti Coast on SH1, two near Upper Hutt on SH58, and two near Warkworth on SH1 (Figures 5 and 6). The approximate location of each site, as well as a description of its streambed, riparian vegetation, and average channel width is shown in Table 1. With the exception of the Pauatahanui Stream on SH58, none of the roads had curbs or channels along them. This meant that any stormwater runoff simply flowed off the road and into gravel areas immediately adjacent to the road, and then into vegetated drainage channels that were present. At Pauatahanui Stream, a curb and channel ran to approximately 1km along the left-hand side of the river (heading north). Any stormwater flowing off the section of road appeared to be conveyed along this channel and into a stormwater pipe that discharged into the true left-hand branch of the stream below the road bridge.

Photos of all the sites, including roadside conditions, are presented in Appendix 1 of this report.

Table 1. Summary description of the study sites selected for this work

Region	Site number	Approximate location	Streambed	Riparian vegetation	Pathway of runoff to stream	Site stream width (m)	Appendix Figures
Kapiti Coast	1	Mangaone Stream	Boulders and cobbles	Pasture grasses with some trees	Mix of vegetated and bare roadside verges	2–3	A1 – A4
	2	Smith Creek, near Paekakariki; SH1	Gravels and cobbles	Pasture and scrub u/s; pasture (sheep grazing) downstream	Mix of vegetated and bare roadside verges	1	A5 – A8
Upper Hutt	3	Pauatahanui Stream near Judgeford	Gravel and cobbles both u/s and d/s	Semi-shaded, pasture and exotic plants	Gutter along roadside discharging into the stream	2–4	A9 – A12
	4	Small stream off Hawards, SH58	Boulders cobbles, with smaller gravels in u/s reaches	u/s Semi-shaded, native bush and scrub; d/s some pasture, plantation forest	Bare roadside verge	1–2	A13 – A16
Warkworth	5	South Branch Mahurangi River	Silt and clay substrate and woody debris and some macrophytes	Semi shaded; native bush and willows in lower sites; willows and open pasture in upper sites	Mix of vegetated and bare roadside verges	2–3	A17 – A20
	6	North Branch Mahurangi River	Silt and clay overlying some cobbles in shallow places, some macrophytes	Semi shaded; native bush in upper site, lower sites semi shaded in exotic trees	Mix of vegetated and bare roadside verges	2–3	A21 – A22

Figure 5 Location of the 4 study sites in the Wellington Region showing the two sites on SH1, and two sites on SH58.

Figure 6 Location of the two study sites near Warkworth showing their locations on the Mahurangi River.

2.3 Sampling protocols

2.3.1 Power analysis

Invertebrate communities vary greatly at small spatial scales, reflecting a combination of both biological interactions such as predation and competition, and physical interactions with the surrounding environment. On a small scale, invertebrates are controlled by factors such as stream velocity and shear stress, presence of periphyton (algae), and the inherent stability of a particular substrate element (Biggs et al 2001; Fuller et al 1986; Statzner and Higler 1986; Vaughn 1986). For example, studies by Hart et al (1996) have shown that the micro distribution of filter feeding black flies is largely dependent upon the near-bed velocity and turbulent regime. Studies by Lancaster and Hildrew (1993) and Matthaei et al (2000) have also shown that stable boulders are colonised by a different suite of taxa than unstable boulders that move more frequently during periods of high flow. These small-scale factors can combine to produce a high degree of variability to invertebrate communities within a small area of streambed (Townsend, 1989).

To accurately detect a difference in density above and below the state highways, it is therefore necessary to collect a suitable number of replicate samples to obtain an accurate estimate of population densities at a particulate site. Most quantitative freshwater studies collect only 5 replicate samples per site (Elliot 1977; Stark et al 2001), resulting in considerable errors of estimating the true population mean at a site. To overcome this, it is necessary to collect more replicate samples. We used a power analysis to determine how many replicate samples needed to be collected to accurately detect a 50% reduction in invertebrate density between different sites, with an 80% certainty. This power analysis was based on data obtained from 20 replicate Hess samples collected at sites above SH1 at Smith Creek, and on 20 boulders collected from the small stream on SH58. Samples for the power analysis were collected as described below. For the power analysis, we examined densities of the nine most common taxa. The data for less common taxa was characterised by many zero fields (i.e., they were not present in as many samples) making analysis problematic.)

2.3.2 Field methods

At each field site, we sampled invertebrate communities at two locations above, and two locations below selected state highway crossings to see if they differed at this spatial scale. In this way, any natural differences between pairs of sites above or below highway crossings would be accounted for in the analysis. We sampled sites as close as possible to the state highway, but within 50m of the highway bridge crossings. Additional sites were collected within 2-300m from these. Sites were numbered from 1 to 4, starting from upstream to downstream.

The different substrate nature in each of the six streams meant that a variety of techniques were employed to quantitatively sample invertebrate communities. For the gravel and cobble-bottom streams (Smith Creek and Pauatahanui Stream) invertebrate communities were sampled using a circular Hess sampler (area=0.76 m^2; 300 μm mesh) which was driven slightly into the streambed and all gravels within the sampler were disturbed to a depth of 100 mm below the streambed surface (Figure 7). All dislodged organic material was subsequently collected in a downstream net, washed into a plastic collecting bottle and preserved with isopropyl alcohol. The large, boulder substrate found in Mangaone Stream precluded the use of the Hess sampler, so quantitative invertebrate samples were collected using the rock rolling technique. Here, a triangular mesh net (aperture size = 0.3 mm) was placed downstream of a large cobble or boulder, which was quickly rolled into the net which was then raised from the streambed. All organisms attached to the boulder or dislodged from the streambed were collected in the net. These were washed into a plastic collecting bottle at the apex of the net and preserved. The X, Y, and Z dimensions of each boulder sampled were recorded, and the surface area calculated using the formula given by Biggs and Kilroy (2000). The small bouldery stream flowing under SH58 was too coarse to sample with the Hess sampler. Furthermore, the rock rolling technique could not be used here as suitable boulders could not be found within a small enough location at each site. To overcome this, paving stones (180 mm × 115 mm × 25 mm) were deployed at the four sampling locations (Figure 8), and left for 10 weeks. This is a sufficiently long enough time to allow the stones to be colonised by a variety of periphyton and invertebrate communities. Upon retrieval, a small triangular net (300 μm mesh) was placed downstream from each paving stone (Figure 9), which was then quickly lifted into the net, and invertebrates attached to the stones were removed by scrubbing and collected into a plastic sample bottle.

Figure 7. Quantitative samples collectde from gravel-bed streams using a circular Hess sampler. This is driven into the streambed which is then disturbed inside the sampler. All dislodged material (invertebrates, algae and leaf litter) floats into a small collecting container at the end of a downstream net.

Figure 8 Paving stones placed in SH58 Site and left for 10 weeks for algae and invertebrates to colonise them

Figure 9. Sampling the paving stones in the SH58 site after a 10 week colonization period

The two northern streams were characterised by their soft-bottomed substrates, and absence of larger gravels and cobbles, precluding the use of techniques such as the Hess sampler. Studies by Maxted, Evans and Scarsbrook (2003) and Collier et al (1999) have shown that woody debris and macrophytes represent important habitat patches in such streams. In particular, Maxted et al (2003) recommend sampling woody debris in soft bottomed streams as part of monitoring programmes using aquatic invertebrates. However, collecting quantitative samples from woody debris or macrophytes can be problematic, owing to the large scale variation in habitat complexity between different samples. To overcome this problem, we developed artificial macrophyte analogues consisting of triple stranded nylon rope (14mm wide x 350mm long) threaded into a small paving block (180 mm × 115 mm × 25 mm). A knot was tied in the base of the rope, and the major strands unwound to mimic individual leaves of a macrophyte. These nylon macrophytes acted as a surrogate for real plants, in a similar way that Suren (1991a) used nylon string woven into a nylon base to act as a surrogate for aquatic mosses. Each artificial macrophyte was placed in selected locations in these soft bottomed streams, and the paving stone pressed firmly into the clay streambed (Figure 10). The artificial macrophytes were left in each stream for eight weeks. This timeframe is sufficient for algal and invertebrate colonisation to occur (Biggs 1988; Lamberti and Resh 1985; Suren and Winterbourn 1992). Invertebrate communities associated with these artificial macrophytes were sampled by placing a net behind each plant, and quickly lifting the plant into the net. The nylon rope was separated from the brick, and placed into a plastic collecting bottle. Any invertebrates adhering to the paving stone were removed by brushing, and collected in the net. All material was preserved with isopropyl alcohol.

Invertebrates were also collected from natural macrophytes (*Egeria densa* or *Elodea canadensis*) in each stream so that comparisons could be made between the communities found in these habitats to those found on the artificial substrates. Samples were collected using a kick net that was vigourously swept through macrophyte stands for approximately 2 minutes per sample.

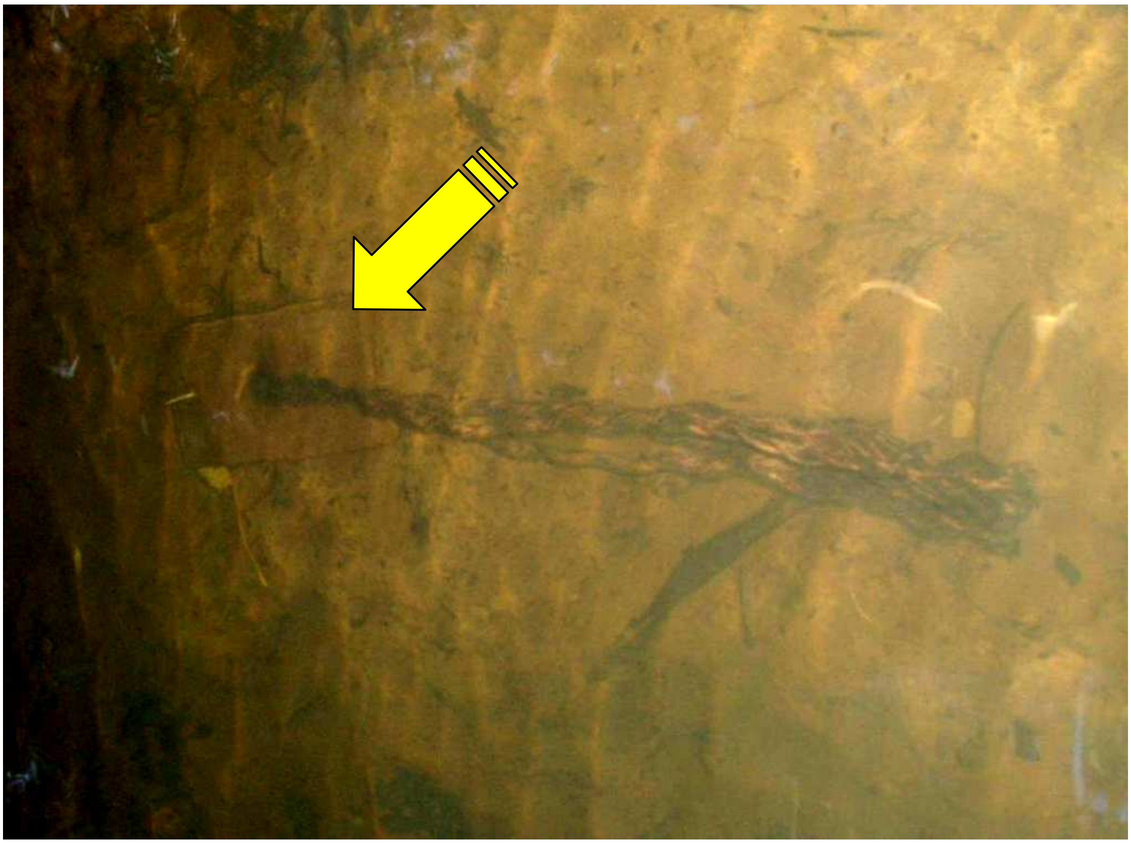


Figure 10 Artificial macrophyte mimicking substrates placed in Mahurangi Stream showing the untwined nylon rope trailing downstream from the paving stone anchor (arrowed).

Stream depth and velocity were also measured at each sampling site using a small Ott meter. Stream temperature was recorded at each site over the summer period by deploying a number of TidBit ® temperature loggers at the site immediately above and below the state highway. Each logger was housed within a metal cylinder, which was attached to a stake hammered into the streambed. These were programmed to record water temperature every 15 minutes. These were deployed at all sites in early December 2010, and were retrieved in April 2011. The dominant substrate nature in each site was also characterised by measuring the size of 100 randomly selected substrate particles (Wolman, 1954). This was done at both the Pauatahanui Stream and the SH58 stream. The Wolman technique was not used at Mangaone Stream, where instead the X, Y and Z dimensions of each boulder sampled were recorded, and surface area calculated as per the method of Biggs and Kilroy (2000). No size assessment was made of the two Mahurangi Stream sites reflecting the use of artificial macrophytes here. Instead, the length of each nylon rope was measured, and the surface area of this and the brick to which it was attached to was calculated.

The Greater Wellington Regional Council and Auckland Council operate hydrological monitoring sites at the Mangaone Stream and Mahurangi Stream respectively. Hydrological information from these two sites was obtained from each council and the data examined to characterise flow conditions at each site during the study. No gauging stations were present in the other streams we sampled so there flow conditions were described by examination of hydrographs from the nearest stream.

2.4 Invertebrate sorting methods

All invertebrate samples were processed using a modification of Protocol P3: full count (Stark et al 2001). Here, all material collected in the field was rinsed through a series of nested sieves (2mm, 1mm, 0.3mm). The contents of each sieve were examined for invertebrates under a dissecting microscope, where invertebrates were counted and identified to as low a taxonomic resolution as possible; usually to Family, Order, or Genera, and counted. Some of the larger insects (e.g., Trichoptera) could be identified to species, while other insects were either too small to identify to species, or could not be identified due to lack of suitable identification keys. Abundance data from all samples with the exception of those collected from natural macrophytes, woody debris and leaf litter in the north and south branches of the Mahurangi Stream were converted to density data by dividing the abundance values by the area sampled. This equated to the area of the Hess sampler (Smith Creek, and Pauatahanui Stream), the area of individual rocks sampled from Mangaone Stream, the area of individual paving stones at the unnamed SH58 stream, and the combined area of the artificial macrophytes thus paving stones at the two Maharangi sites. Data from the natural macrophytes and woody debris at these latter sites were converted to percentages as we had not sampled a consistent surface area.

2.5 Metal analysis of plant material

As mentioned (section 1.3), periphyton can accumulate significant amounts of heavy metals (Davis and George 1987; Suren and Elliot 2004) presumably as a result of uptake of dissolved metals from road derived runoff. Another common component of small streams, especially on stable substrates, are aquatic bryophytes (mosses and liverworts). These small statured plants represent important stable habitats within streams, and often support very high invertebrate densities (Bowden et al 1997; Percival and Whitehead 1929; Suren 1991b). Aquatic bryophytes have particularly strong affinities for dissolved metals (Kelly et al 1987; Wehr et al 1981; Whitton et al 1982), and so may also show elevated metal concentrations in sites subject to road runoff. As with aquatic invertebrates, both algae and aquatic bryophytes act as integrators of the antecedent conditions they have been exposed to. In this way, analysis of the metal content in periphyton and bryophytes may sometimes show up evidence of metal contamination whereas short-term water quality monitoring may not, given the intermittent nature of road runoff discharges.

Periphyton samples were collected from the Mangaone, Pauatahanui and Smith Creek sites by scraping material from selected cobbles using a stainless steel scalpel blade at sites above and below the State highways. Because it was necessary to obtain sufficient material to do the analysis, up to 5 cobbles at each site was selected, and all material combined into one sample. Samples of aquatic moss (*Amblystegium* sp) were also collected from a large stable boulder at the Pauatahanui site below the road bridge. No periphyton was collected from the SH58 site reflecting its shaded nature and absence of thick periphyton material. All collected material was placed in acid-washed plastic bottles and frozen prior to analysis. Samples were then sent to RJ Hill laboratories for analysis. Here, they were thawed, and homogenised by mincing and chopping. Samples were then digested using nitric and hydrochloric acid (85°C for one hour) and a range of heavy metals (arsenic, cadium, chromium, copper, lead, nickel and zinc) were analysed by inductively coupled plasma mass spectroscopy (ICP-MS). All results are reported as milligrams per dry weight kilogram.

2.6 Data analysis

2.6.1 Power analysis

To determine the minimum sample size required to detect a known amount of change with a specific degree of certainty, we used the statistical package "R" and applied the "n.paired.t.test" function taken from the "Samplesize" package. This function computes the overall power of an analysis with increasing sample size in detecting a mean difference between samples for a given probability level. For this case, we set the desired power at 80%, and the given probability level of 0.95. The mean difference between samples we wished to detect was 80%. We analysed densities of the top nine taxa to see how density estimates varied with increasing sample size. We also examined the cumulative species richness curve as more replicates were added to see if any plateaus were reached.

2.6.2 Invertebrate data analysis

The following biological indices were calculated from the invertebrate data: total density; taxonomic richness; the number of EPT and EPT* taxa; % EPT and %EPT*, and the MCI and QMCI (or, for the Maharangi streams, the MCI-sb and QMCI-sb, reflecting the predominantly soft substrate at these sites). All these biotic indices, plus the densities of the 10 most common taxa, were examined using a nested analysis of variance (ANOVA) to see whether these differed above and below the state highways. For this analysis, the main treatment effect was location (i.e. above or below), with individual sites (2 at each location) nested within each location. This design allowed us to distinguish differences in biotic indices upstream or downstream of the state highways, and differences between sites within a location either upstream or downstream of the state highways. Effects of road runoff would be evident with a significant difference in location, as well as between sites downstream of the state highway. Ideally, no differences would be apparent between sites upstream of the highway: i.e., invertebrate communities would not be responding to natural small-scale habitat differences within the study section, as this would complicate the interpretation of the results. Thus, densities of sensitive taxa would be significantly lower at sites below the highway than above (i.e., a significant location effect), and significantly lower at the site immediately below the highway than at the site further downstream (i.e., a significant "site nested within location" effect). Densities of these taxa would be similar in both upstream sites. The reverse would be true for densities of tolerant taxa: viz, significantly higher at sites below the highway, and significantly higher at the site immediately below the highway than further downstream. Individual ANOVAs were done on data collected at each site, as there was no *a priori* reason in examining any interactions between location and river. Furthermore, invertebrate communities from each stream were very different to each other (unpublished data), so the value of including all the data in the one analysis was questionable. All data was examined for normality, and transformed where necessary by fourth root transformation, which appeared the most powerful.

We next calculated the similarity of invertebrate communities collected at all sites in each stream using the Bray-Curtis similarity score. Bray-Curtis similarity is used to quantify the compositional similarity between two different sites. It is bound between 0 and 1, where 0 means the two sites have no species in common, while 1 means the two sites have the same composition (that is they share all the species). We calculated similarity within an individual site, and similarity between all the possible pairwise combinations of sites. An ordination (NMDS) was then performed on the data. This statistical technique graphically represents the location of samples based on their invertebrate communities, such that samples with similar communities appear close together on a graph, and samples with very different communities appear far apart from each other. Samples are plotted in two dimensions with arbitrary sample scores. Thus, if road runoff was having a demonstrable impact on invertebrate communities, then samples collected from these different locations would be well separated on the ordination graph.

A nested Analysis of Similarity (ANOSIM) was then used to test whether there were any significant differences between two or more groups of sampling units. In this case, the groups represented samples collected from different locations (above or below the state highways) or between sites nested within a location. ANOSIM produces a statistic, *R*, which indicates the magnitude of difference among groups of samples. An *R* of 1 indicates that the communities completely differ among defined groups, and an *R* of 0 indicates no difference among groups. The statistical significance of *R* was tested by Monte Carlo randomization. We followed the protocol of Clarke and Gorley (2001) and Clarke (1993) in using nested ANOSIM whereby two discrete hypotheses were tested:

- the null hypothesis of no difference between sites within each treatment (i.e., between sites 1 and 2 above the state highway, or 3 and 4 below the state highway);
- the null hypothesis of no difference between treatments (i.e., all sites above or below the state highway)

The first hypothesis was answered by calculating the average R statistic for sites both above and below the state highway, and testing its significance using permutation procedures ($n = 999$). If this hypothesis was rejected, then each site within each treatment could be regarded as a true replicate, and so the difference between locations above and below the highway could subsequently be tested. If this hypothesis was, however, true, then effectively there have been no “site” replicates in each location.

Following the first hypothesis, we tested the more relevant hypothesis, that of whether the different treatments differed. This was tested using a normal one-way ANOSIM, which is analogous to an ANOVA where each site was allocated to a specific treatment effect of either “above” or “below” the state highway, and where each site acted as a replicate. When a significant treatment effect was detected, a final pairwise ANOSIM was done on all sites to determine the pairwise differences between them. If stormwater runoff was having a large impact, we would expect the greatest difference between sites (as measured by the R statistic) to be between sites 2 and 3 (i.e. the site immediately above and below the highway). Both NMDS and ANOSIM were run using the Primer package (Clarke and Gorley, 2001)

3 Monitoring results

3.1 Power analysis

For the Hess samples collected from Mangaone Stream, the power analysis showed that we could detect a 70% difference in densities of the nine most common taxa with 80% certainty if 15 samples were collected. Variability was higher in the rock samples collected from the SH58 stream, as 24 samples would need to be collected to achieve the same degree of statistical power. Examination of the species-area curves from the Mangaone Stream showed a gradual increase in the total number of species encountered with increasing sample size. A slight plateau was apparent between 10 and 15 samples (Figure 11) after which time the cumulative richness continued to slowly increase. There was a much greater increase in cumulative taxa richness at the SH58 stream, especially for the first 8 samples. After this number, cumulative richness tended to plateau between 8 and 15 replicates, before increasing again.

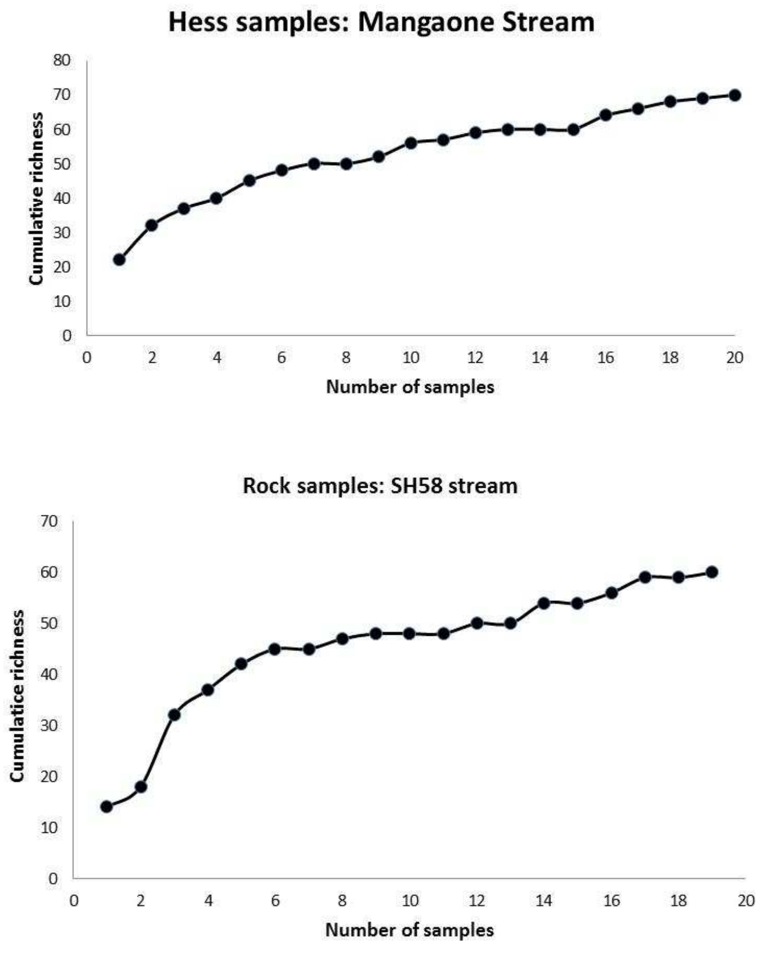


Figure 11 Cumulative taxonomic richness in samples collected from Hess samples in the Mangaone Stream, and from natural cobbles in a small stream flowing under SH58.

Given the trade-off between statistical accuracy, and the costs inherent with processing samples, it was decided to collect 15 replicate samples from all streams. Such a relatively high number reflects the power analysis results at the Mangaone Stream, as well as the cumulative richness plots. This is generally three times the number of samples collected for many quantitative studies (Stark et al 2001).

3.2 Physical conditions

Examination of flow data from the Mangaone Stream (obtained from the Greater Wellington Regional Council) showed that two relatively large floods (close to or equivalent to the 1 in 2 year the event) occurred in September 2010 and early October 2010 after which time flows remained mostly low and stable (Figure 12). Sampling was conducted in early December 2010, giving at least an eight week recovery period following the last flood. This would have been sufficient for any algal or invertebrate communities to have recovered following the flood. A very small flood (commonly termed a "fresh") occurred in early November 2010, which would have done little to directly affect invertebrates, but may have washed stormwater runoff into the stream.

A similar period of flow stability following a large flood in early October 2010 was observed in other gauged stations in the vicinity of the Pauatahanui Stream and the small stream running under SH58. However, there was much less evidence of freshes occurring following this period (Figure 13).

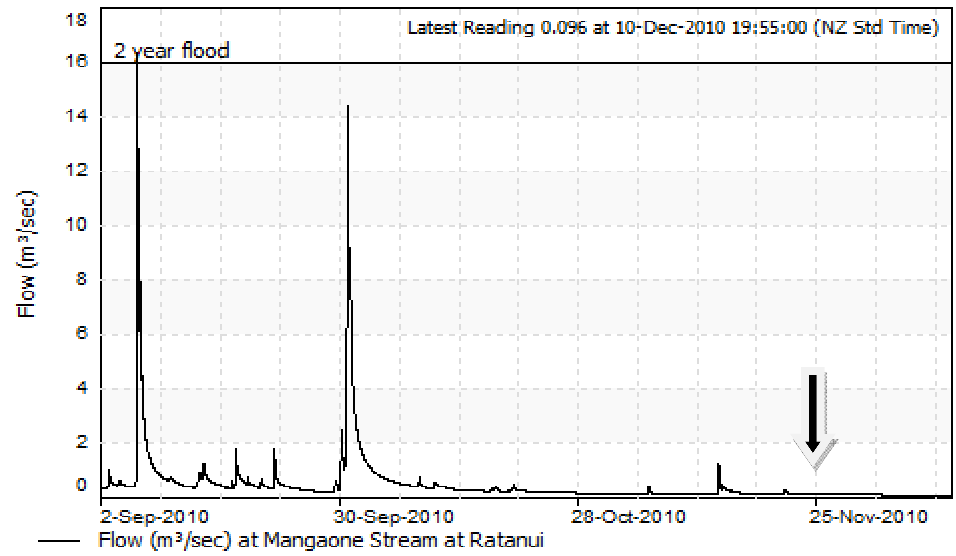


Figure 12. Flow hydrograph at Mangaone Stream show the antecedent flow conditions three months prior to the invertebrate sampling (3 December 2010 - arrowed).

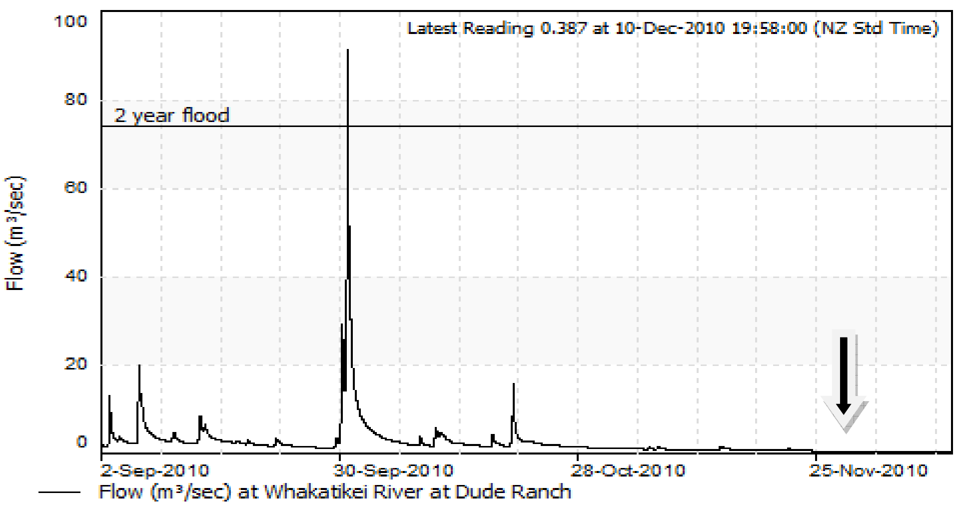


Figure 13 Flow hydrograph at Whakatikei River showing the antecedent flow conditions three months prior to the invertebrate sampling (3 December 2010 - arrowed). This site was located to the north-east of the Pauatahanui Stream and the small stream under SH58, but likely to have reflected flow conditions at these sites as well.

Flows in the Mahurangi River were much more variable than the Wellington sites (Figure 14). Artificial macrophytes were initially deployed on 10 December 2010, but a very large flood (with a 25 year return period) in late January 2011 washed all trace of these away. Following this, new artificial macrophytes were replaced in the stream on 5 April 2011, some four months later. These were sampled seven weeks later on 23 May 2011. In this period, five small freshes had occurred, but all were less than the mean annual flood and will thus thought not to have a large direct effect on the invertebrate communities, especially given the relatively flood prone nature of these rivers.

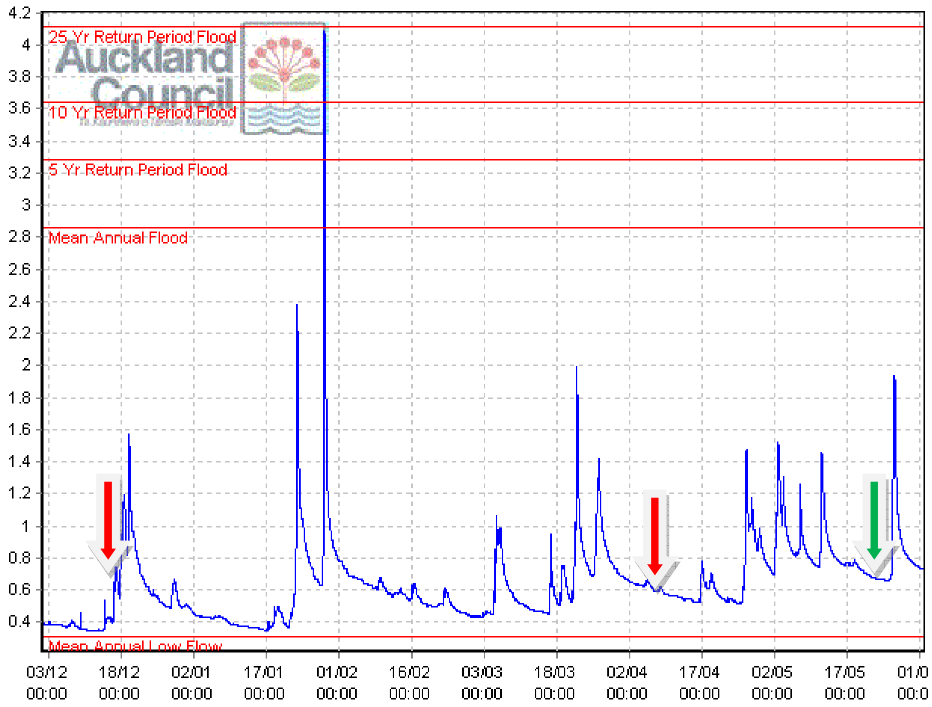


Figure 14 Flow hydrograph of the Mahurangi River showing the antecedent flow conditions prior to the final invertebrate sampling of the artificial macrophytes (23 May 2011 – green arrow). Artificial macrophytes were first deployed in early December, and then again in early April (red arrows).

Measured stream depth was similar at sites above and below the state highway at both Mangaone and Pauatahanui streams, but was greater above the highway (mean depth = 130 mm) than below (mean depth = 94 mm) at Smith Creek. There was no difference in the measured stream velocity in any of the sites above or below the state highways. Velocity was quickest in Pauatahanui Stream (mean = 43 cm per second), and slowest in Smith Creek (mean = 13 cm per second). There was no difference in measured temperature regimes above or below the state highways at either Mangaone Stream or Smith Creek, although temperature range was slightly cooler at Smith Creek (min = 9°C, max = 22 °C) than Mangaone Stream (min = 12°C, max = 25°C). Unfortunately, the downstream temperature logger at the Pauatahanui site was lost. Furthermore, all temperature loggers place that both Mahurangi sites were also lost following a large flood in early February 2011.

The calculated surface area of large cobbles collected from Mangaone Stream during the Hess sampling protocol here did not differ between sites above or below the state highway. Wolman sampling also showed no differences in substrate size at the Pauatahanui Stream. At Smith Creek, Wolman sampling showed a slightly coarser substrate was found at the site immediately below the state highway. Substrate sizes at the other sites in Smith Creek appeared similar to each other and were slightly smaller.

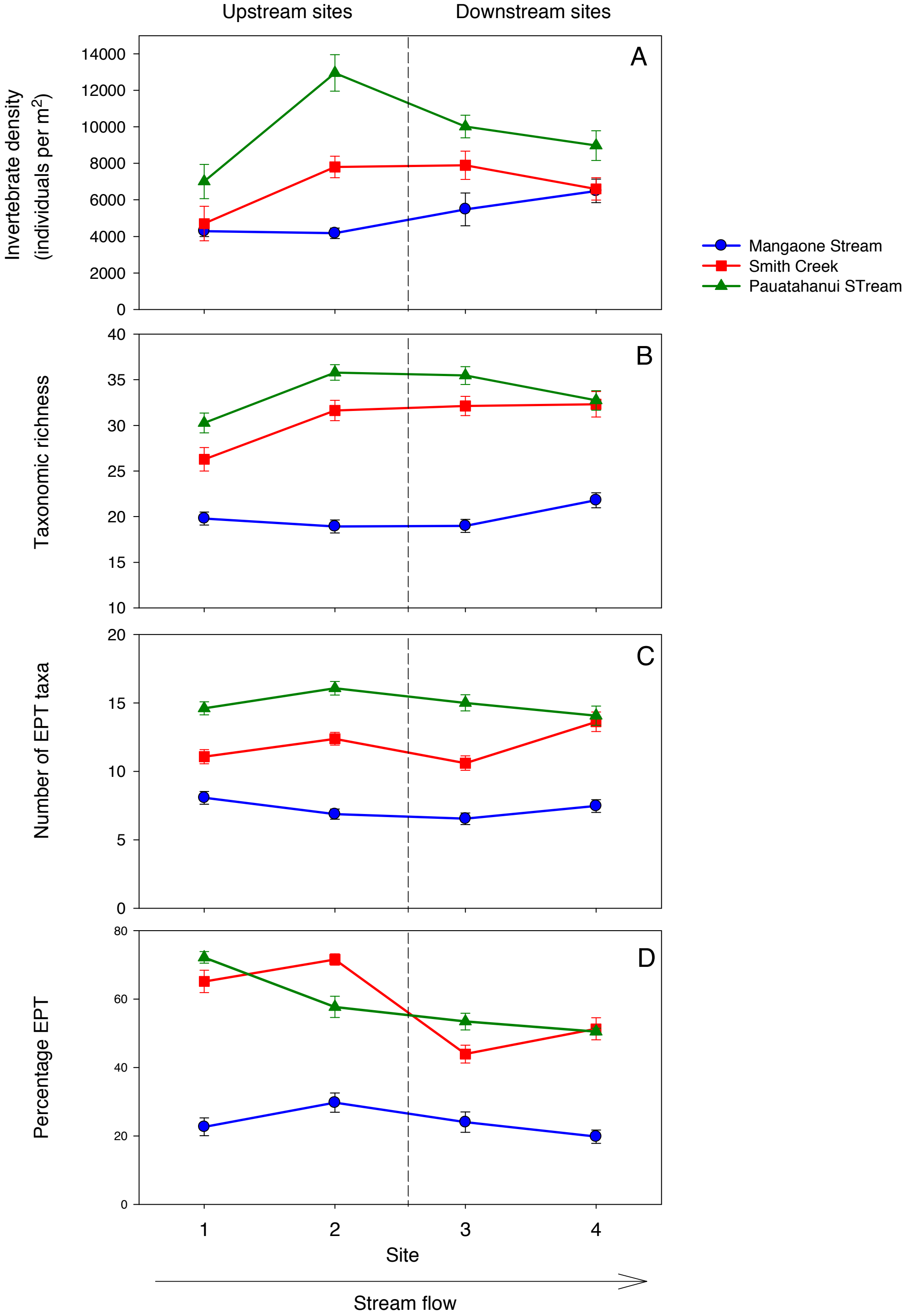
3.3 Wellington region – natural substrates

3.3.1 The invertebrate fauna

A total of 94 invertebrate taxa were collected from natural substrates in the three streams in the Wellington region. This fauna was dominated by the caddisflies *Aoteapsyche* and *Pycnocentria*, the mayfly *Deleatidium*, four genera of midges (*Cricoctopus, Eukiefferiella, Naonella* and *Tanytarsus*), Elmid riffle beetles oligochaete worms, and the common snail *Potamopyrgus*. Nested ANOVA showed no significant difference in invertebrate abundance above or below the state highways at either Smith Creek or Pauatahanui Stream, whereas abundance was significantly higher below the state highway at Mangaone Stream (Figure 14A). There were no differences between sites either upstream or downstream in Mangaone Stream, suggesting a true effect of location. Taxonomic richness was also similar above and below the state highways at both Mangaone and Pauatahanui streams, but was higher at Smiths Creek below the highway (Figure 14B). However, taxonomic richness in Smith Creek differs significantly between the two upstream sites, and was highest at the site immediately upstream of the highway. This site had similar richness to both sites downstream of the highway. This was counter to expectations if road runoff was having an adverse effect on invertebrate communities.

The number of EPT and EPT* taxa at all three sites was similar above and below the state highways (Figure 15C and E), suggesting no effect of runoff on these metrics. In contrast, however, the percentage of EPT and EPT* was higher above the state highway than below at both the Smith Creek, and Pauatahanui Stream (Figure 15D and F). However, significant differences in sites within each location were evident only at the Smith Creek site below the state highway. Here, the percentage of EPT and EPT* were lower at the site immediately below the road, and increased further downstream. Such a result is consistent with our initial hypothesis of a potential effect of road runoff.

Calculated MCI scores were lowest at the Mangaone Stream site (average = 90), and higher at the other two sites (average = 103). MCI scores were significantly higher above the state highways than below for all three streams (Figure 15 G), although examination of site differences showed significant differences only at Smiths Creek. Here, MCI scores were lower at the site immediately below the highway and increased somewhat at the site further downstream. This behaviour matched our predicted hypothesis about the behaviour of metrics in the presence of an impact. Similar patterns were observed for the QMCI scores (Figure 15H).


Upstream sites
Downstream sites
A
B
C
D
Invertebrate density (individuals per m2)
Taxonomic richness
Number of EPT taxa
Percentage EPT
Site
Stream flow
Mangaone Stream
Smith Creek
Pauatahanui STream

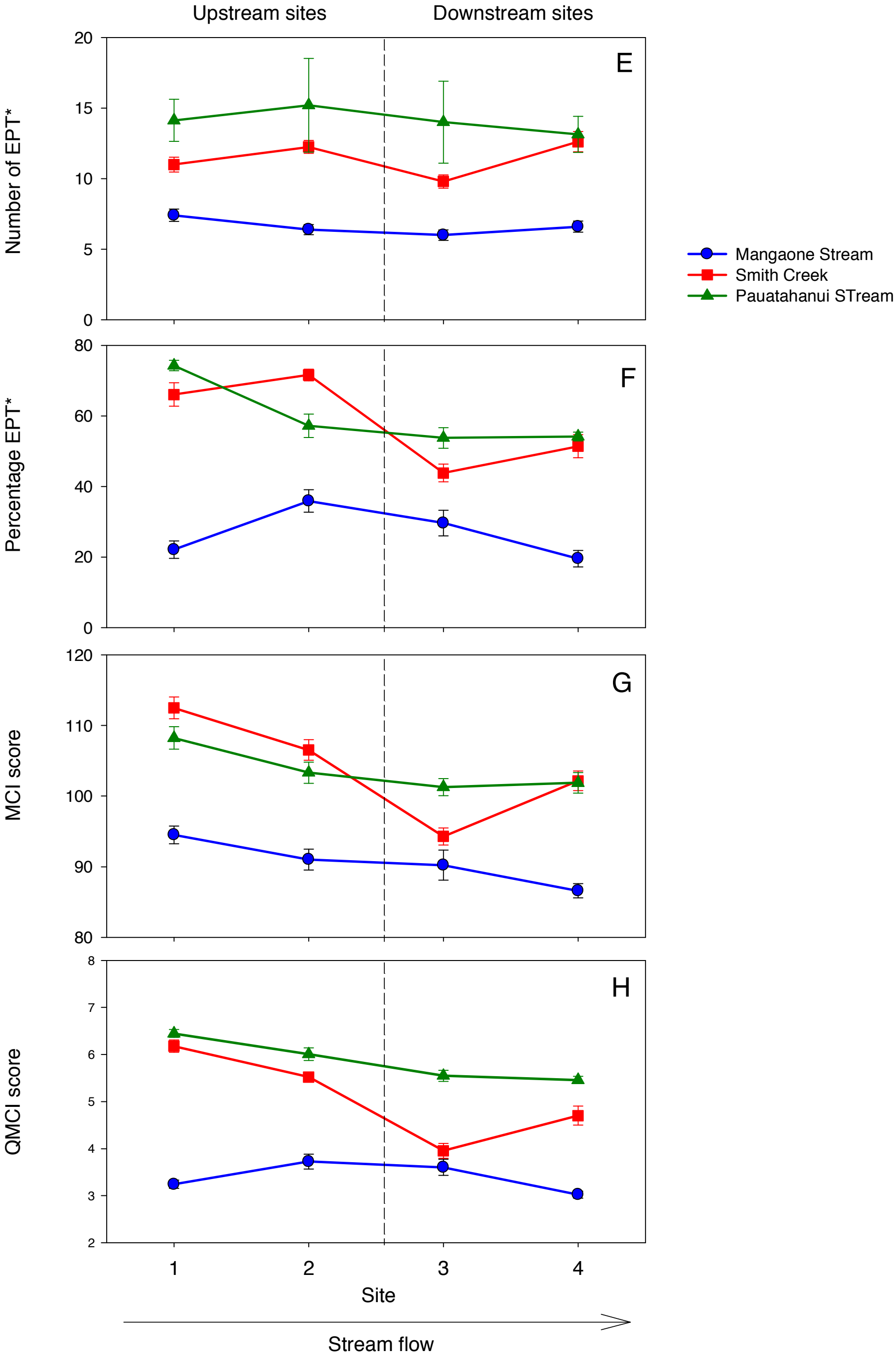


Figure 15 A-H. Invertebrate abundance, taxonomic richness, the number of EPT and EPT* taxa, the % of EPT and EPT* taxa in samples collected from the four sites in the three streams sampled with natural substrates in the Wellington Region. Also shown are the calculated MCI and QMCI scores (x ± 1se, *n* = 15).

These results show that invertebrate abundance, richness, and the number of EPT and EPT* taxa were either similar above and below the state highways, or were higher below. In contrast, we found a consistent reduction in the percentage of EPT and EPT* taxa, and in the behaviour of calculated MCI and QMCI scores at Smith Creek which may have suggested a potential effect of stormwater runoff from the state highway on these metrics at this site only. These results suggest that potential stormwater runoff at the three streams examined had little or no demonstrable impact on most of the calculated biotic metrics.

Of the 10 most common taxa examined, densities of 6 showed no consistent effect of potential stormwater runoff. These taxa either did not differ in abundance above or below the state highways, or else their densities were higher below the highways than above. Densities of only four taxa, but only at Smith Creek, showed possible consistent effects of runoff from the state highway (Figure 16). Here, densities of two midges (*Cricoctopus* and *Naonella*) were highest at the site immediately below the state highway, then declined at the furthest downstream site to densities similar to that found above the state highway. Densities of the mayfly *Deleatidium* were also significantly lower in this stream immediately below the state highway, but increased at the furthest downstream site. Finally, densities of oligochaetes in Smith Creek were higher at both downstream sites.

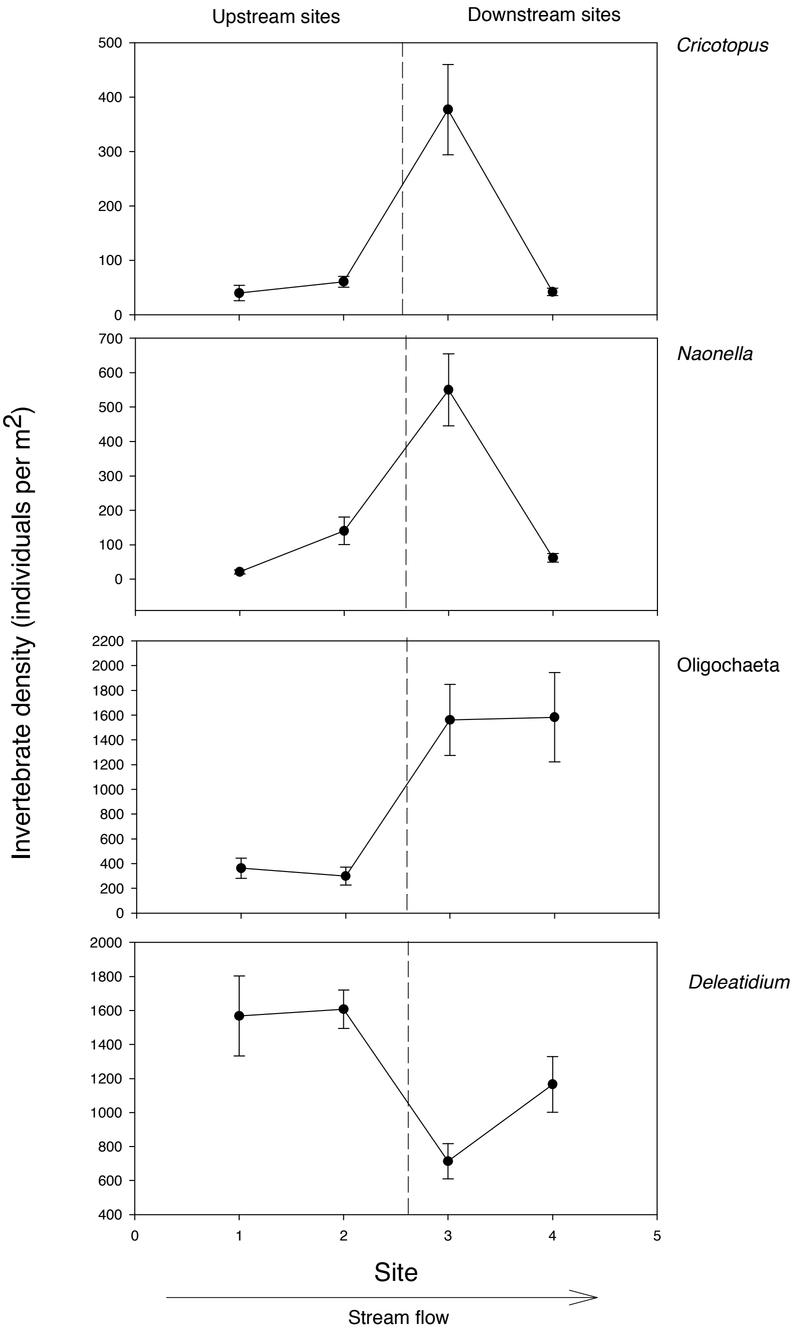


Figure 16. Densities of the four at Smith Creek that showed possible consistent effects of runoff from the state highway (x ± 1se, *n* = 15).

These results suggest that any stormwater derived contamination of the lower sites sampled for the study was having at worst only a limited effect on densities of the most common taxa. No consistent trends were evident for most of the taxa above and below the state highways, with the exception of the four taxa at Smith Creek. Here, differences in densities appeared to follow those predicted by our *a priori* hypotheses suggesting that road runoff here may have been affecting the invertebrate communities.

3.3.2 Changes to community composition

NMDS ordination of the invertebrate data from the Mangaone Stream showed a considerable degree of overlap between samples collected above and below the state highway (Figure 17). Despite this overlap, significant differences existed between locations above and below the state highway ($R = 0.12$, $P < 0.001$). Significant differences also existed between the different sites within each location (average R-statistic across both locations = 0.255, $P < 0.001$). A pairwise ANOSIM of all sites showed that the largest between-site differences were found between site 1 (the uppermost site) and site 4 (the lowermost site), while the least difference occurred between sites 2 and 3 (i.e. the site immediately above and below the highway: Table 2). The fact that sites 2 and 3 differed the least between all the possible pairwise comparisons suggested that there was no evidence of an effect of state highway derived runoff on the overall community composition in Mangaone Stream.

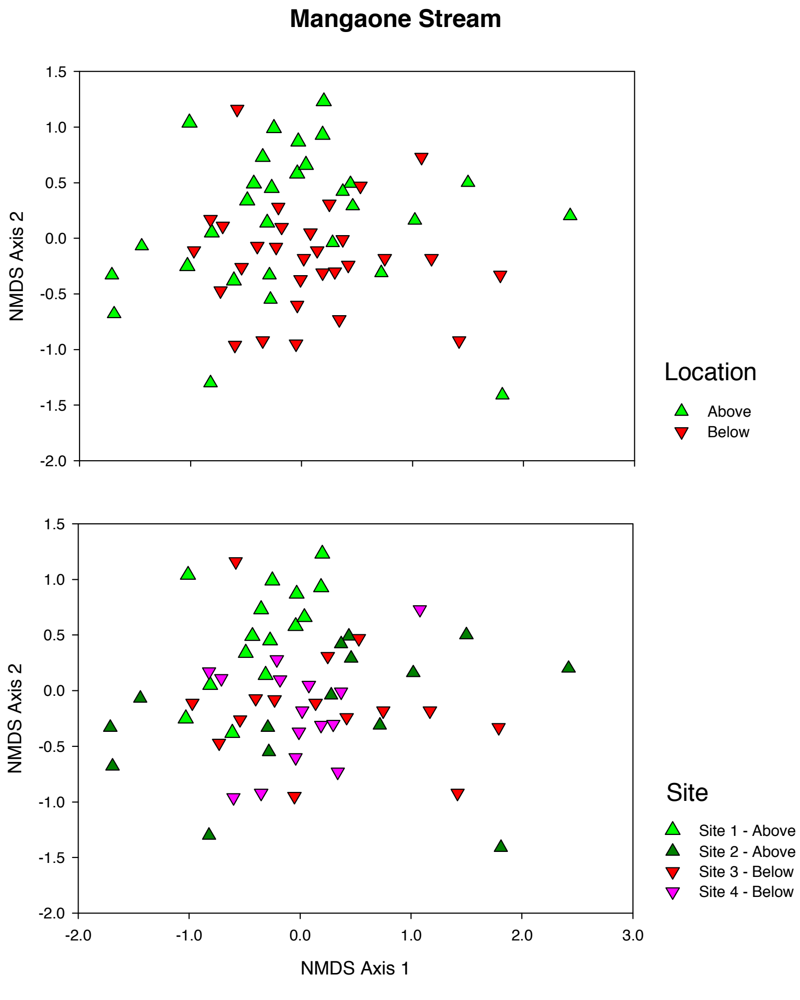


Figure 17 Results of NMDS ordination of invertebrate data collected from the Mangaone Stream showing sites coded by location (upper graph) or sites number (lower graph).

Site	R statistic	Significance Level%	Possible Permutations	Actual Permutations	Number>= Observed
4,3	0.295	0.1	77558760	999	0
4,2	0.335	0.1	77558760	999	0
4,1	0.462	0.1	77558760	999	0
3,2	0.038	14	77558760	999	139
3,1	0.236	0.1	77558760	999	0
2,1	0.215	0.1	77558760	999	0

Table 2. Results of pairwise ANOSIM to determine the differences in community composition in the 4 sites samples in the Managone Stream, (Site 1 – most upstream site, Site 4 – most downstream site).

Ordination of data from Smith Creek at Paikakariki showed large differences between samples collected above and below the state highway (Figure 18). Results of the nested ANOSIM showed significant differences between locations above and below the state highway ($R = 0.479$, $P < 0.001$), and also between sites within locations above and below the state highway (average R-statistic across both locations = 0.622, $P < 0.001$). All pairwise comparisons between sites were significantly different, suggesting that invertebrate communities varied between all sites along this stream. Examination of the pairwise differences (Table 3) showed the largest difference ($R = 0.884$) occurred between site 3 (immediately below the road bridge) and site 1 (the uppermost site). The smallest pairwise differences occurred between sites 3 and 4 (i.e. both lower sites), and between sites 1 and 2 (i.e. both upper sites). Only moderate differences existed between sites 2 and 3 (i.e. sites immediately above and below the state highway). If stormwater runoff was having an effect on the invertebrate communities, the greatest between sites difference should have occurred between these two sites. These results suggest that invertebrate communities instead were responding to other environmental conditions within each site, that appear independent of road runoff.

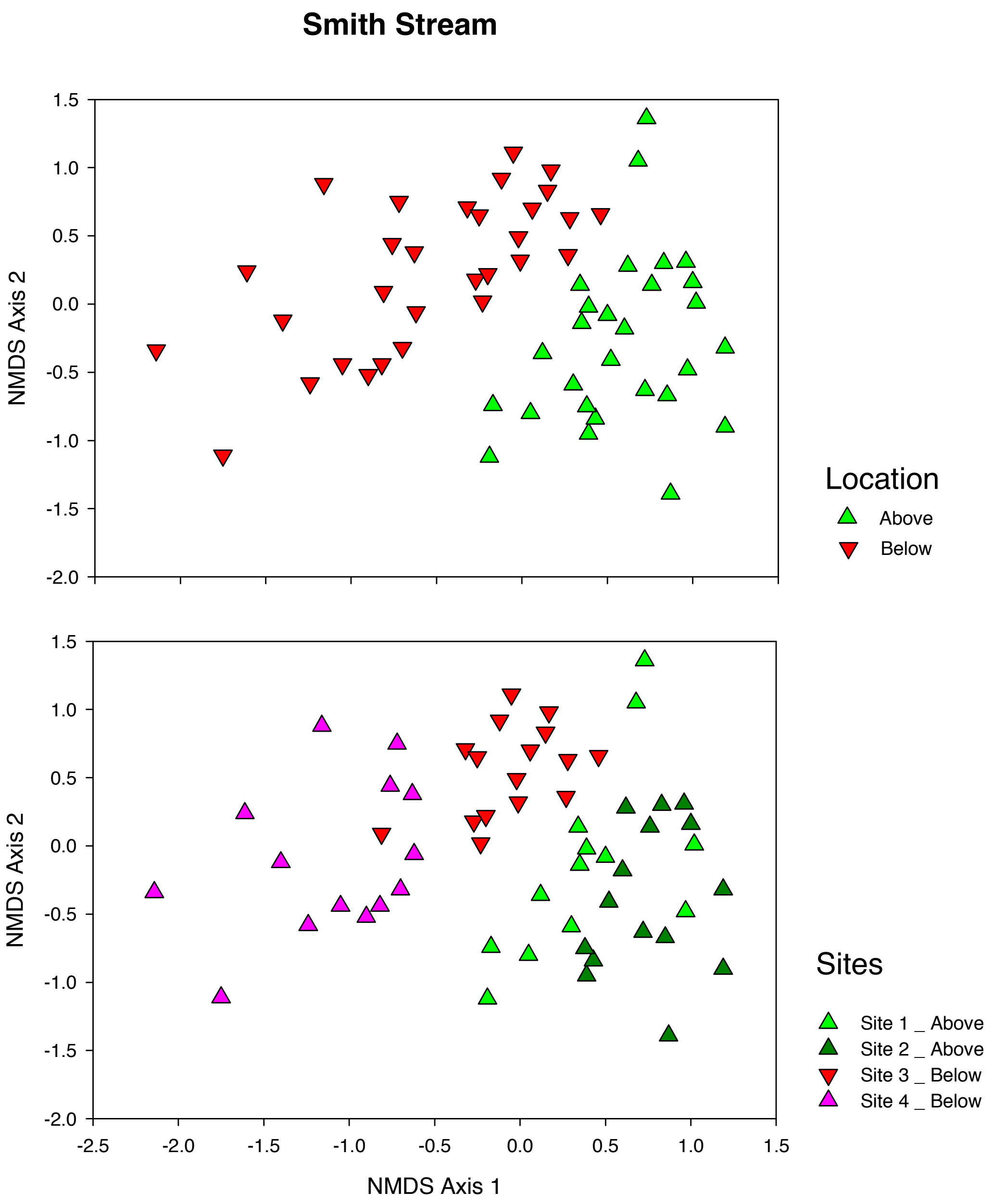


Figure 18 Results of NMDS ordination of invertebrate data collected from Smith Creek showing sites coded by location (upper graph) or sites number (lower graph).

Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
4,3	0.619	0.1	37442160	999	0
4,2	0.741	0.1	67863915	999	0
4,1	0.826	0.1	20058300	999	0
3,2	0.739	0.1	300540195	999	0
3,1	0.884	0.1	77558760	999	0
2,1	0.625	0.1	145422675	999	0

Table 3 Results of pairwise ANOSIM to determine the differences in community composition in the four sites sampled in Smith Creek.

Similar patterns were found at the Pauatahanui Stream, where the NMDS showed relatively large differences between samples above and below the state highway (Figure 19) but slightly more overlap than in Smith Stream. Nested ANOSIM showed significant differences between sites within locations above and below the state highway ($R = 0.451$, $P < 0.001$) and between locations ($R = 0.276$, $P < 0.001$). Examination of pairwise differences (Table 4) showed the largest difference ($R = 0.629$) occurred between site 4 (the lowermost site) and site 2 (immediately upstream from the road bridge). The smallest pairwise differences occurred between sites 2 and 3 (i.e. at sites immediately above and below the state highway). This was contrary to our expectations if state highway derived runoff was having an effect on the invertebrate communities, as we would have expected the biggest difference in community composition to be at the site immediately above and below the source of potential contaminants.

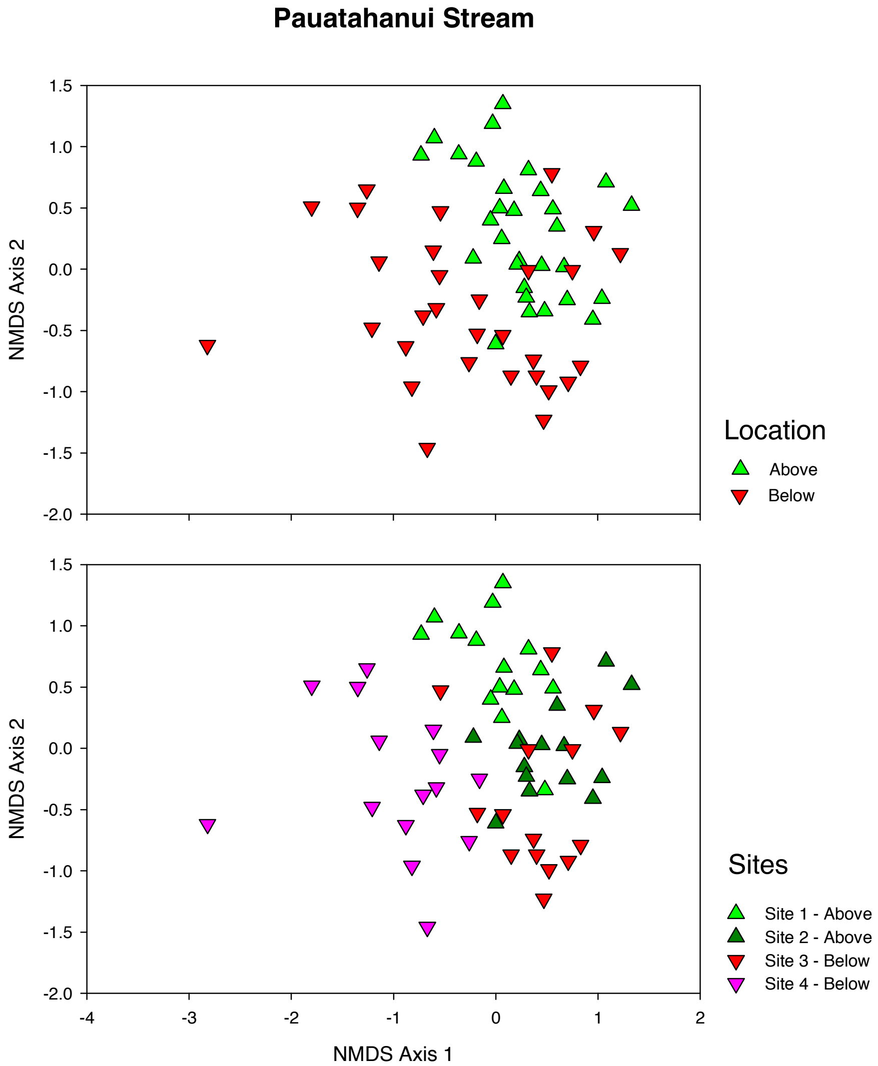


Figure 19 Results of NMDS ordination of invertebrate data collected from Pauatahanui stream sites coded by location (upper graph) or sites number (lower graph).

Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
4,3	0.483	0.1	77558760	999	0
4,2	0.629	0.1	77558760	999	0
4,1	0.545	0.1	77558760	999	0
3,2	0.306	0.1	77558760	999	0
3,1	0.538	0.1	77558760	999	0
2,1	0.419	0.1	77558760	999	0

Table 4. Results of pairwise ANOSIM to determine the differences in community composition in the 4 sites sampled in Pauatahanui Stream.

3.3.3 Metal contamination of plant material

Analysis of periphyton (and aquatic moss) material from the three streams showed some evidence of metal uptake at sites below the state highway (Table 6). At the Pauatahanui site, the aquatic moss showed slightly elevated concentrations of zinc, however concentrations within periphyton at the same site were lower than that of periphyton collected in the upstream sites. No mosses were found in the upstream sites, so we cannot determine whether the increased zinc concentration reflected increased zinc loads below the highway bridge, or reflected the increased ability of the aquatic moss to absorb dissolved zinc across its cell walls and into its tissue when compared to periphyton. At Mangaone Stream, periphyton collected at site 3 (immediately below the road bridge) exhibited increased levels of copper, lead, nickel and zinc when compared to samples collected at site 2 above the road bridge (Table 6). Concentrations of zinc at the lowermost site 4 in this stream were lower than at site 3, but still higher than those observed above the road bridge, suggesting a potentially large area of contamination.

The most noticeable differences in metal concentrations occurred in periphyton collected from Smith Creek, where samples collected below the road bridge display higher levels of copper, lead, nickel and zinc. Levels of zinc were particularly high in periphyton collected in the downstream site (Table 6).

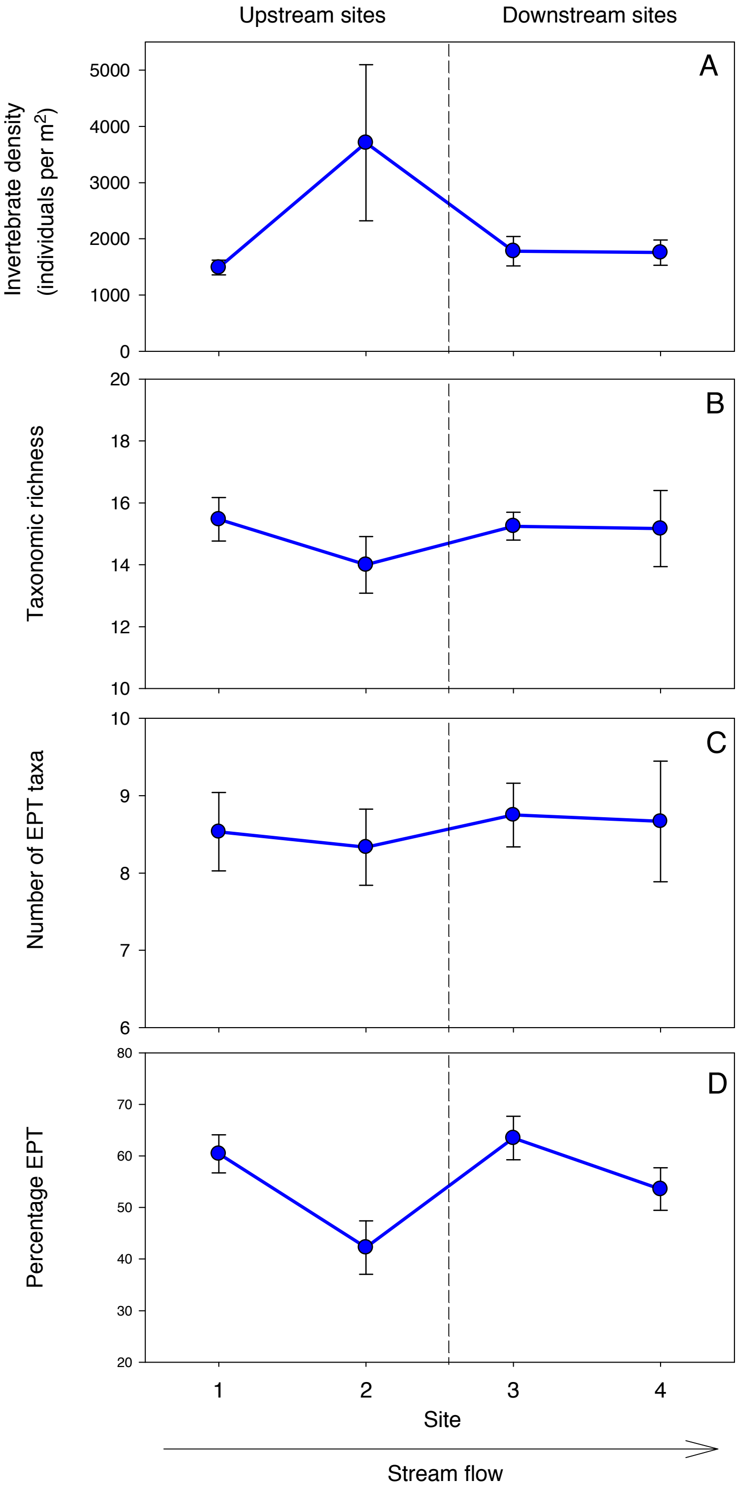
River	Location	Notes	Arsenic	Chromium	Copper	Lead	Nickel	Zinc
Pauatahanui	Above	Site 2	0.9	2.2	1.64	1.94	1.27	11.2
	Below	Site 3	0.66	0.97	1.54	1.55	0.61	9.2
	Below	Site 3 _ Moss	0.92	1.67	2.2	2.1	1.34	**15**
Mangaone	Above	Site 2	0.41	6.2	1.07	1.18	1.31	7
	Below	Site 3	0.73	7.8	2	**2.2**	**2.5**	**13.4**
	Below	Site 4	0.57	6.9	1.48	1.59	1.8	**10**
Smith	Above	Site 2	0.7	8.3	1.4	1.36	2.4	6.4
	Below	Site 3	1.78	8.1	**6.1**	**6.2**	**6.7**	**29**

Table 6 Heavy metal concentrations (in mg/kg dry weight) in periphyton (or moss) collected from sites above and below state highway road bridges in three streams. Concentrations which appeared higher than found in periphyton samples collected from above the road are highlighted (tan).

3.4 Wellington region – artificial substrates

3.4.1 The invertebrate fauna

Of the 15 paving stones deployed in the small unnamed stream on SH58, three could not be recovered from the uppermost site (Site 1), and seven from Site 2. Some of the missing stones at Site 2 were found washed onto the banks of the stream where they were only partially wet, and so were not sampled. All the stones deployed at the lower sites were, however, recovered. A total of 47 taxa were collected from artificial substrates placed in this stream. This fauna was dominated by orthoclad midges, five genera of mayflies (*Neozephlebia, Deleatidium, Coloburiscus, Zephlebia,* and *Nesameletus*), two genera of caddisflies (*Oxyethira* and *Helicopsyche*), the snail *Potamopyrgus*, and the toe-biter *Archichauliodes diversus*. Nested ANOVA showed no significant difference in invertebrate abundance, taxonomic richness, or the number or percentage of EPT taxa above and below the state highway (Figure 20 A-D). Indeed, the percentage of EPT was highest at the site immediately below the state highway (63%) and lowest immediately above (42%: Figure 20 D) – contrary to our initial expectations of these sensitive taxa being adversely affected by road runoff.



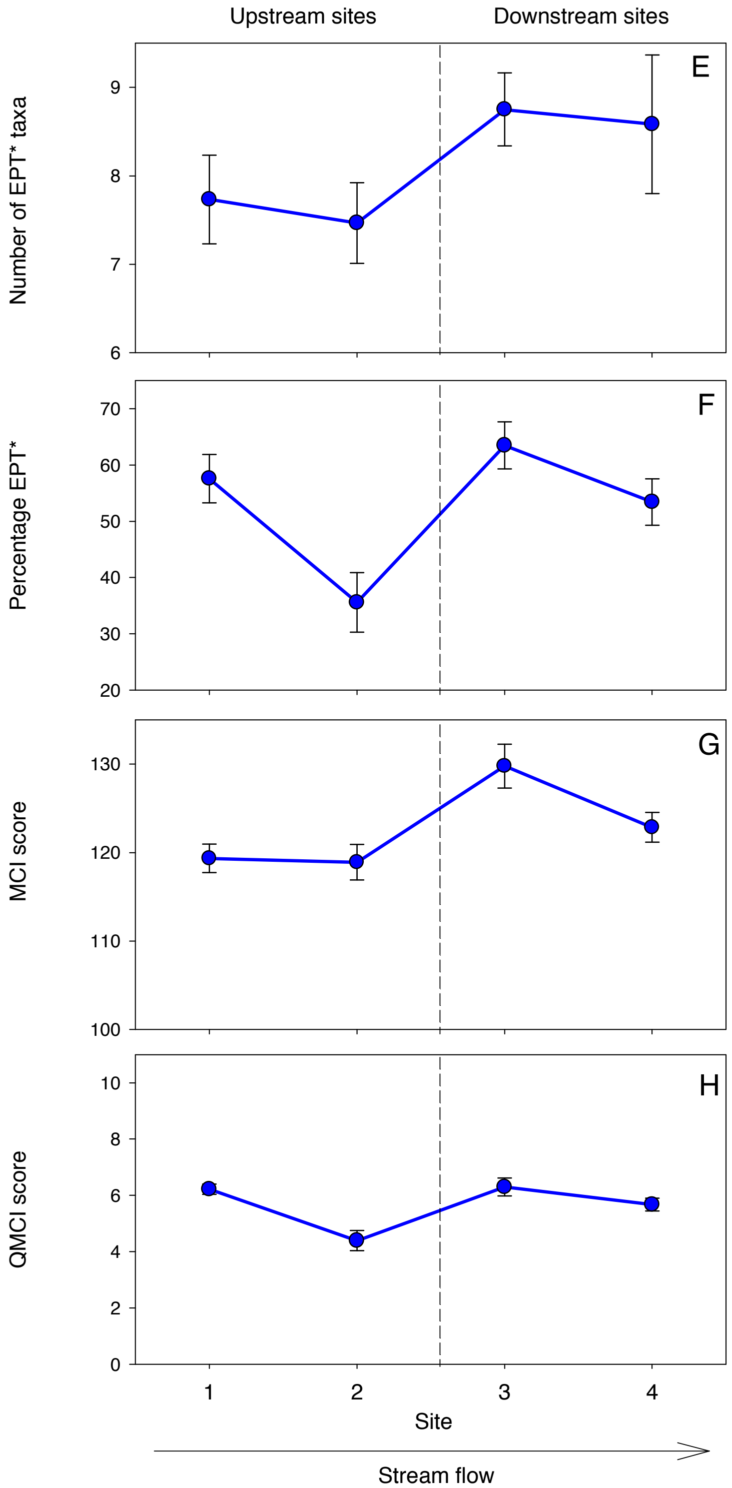


Figure 20 A-H. Invertebrate abundance, taxonomic richness, the number of EPT and EPT* taxa and the % of EPT and EPT* taxa in samples collected from artificial substrates (paving stones) in a small unnamed stream flowing under SH58 (x ± 1se, *n* = 12 (site 1), n = 8 (site 2), *n* = 15 (sites 3 and 4)).

No significant differences were observed for the number of EPT* taxa above or below the highway. The % EPT* was significantly higher at sites below the highway (Figure 20 F), again counter to our initial hypothesis if indeed road runoff was having an adverse effect on the invertebrate community at this site. Calculated MCI and QMCI scores were also significantly higher at sites below the highway (Figure 20 G and H), although the QMCI score was similarly high in the uppermost site.

Of the ten most common taxa examined, densities of seven did not differ between locations upstream or downstream of the state highway. Densities of two taxa (the mayfly *Nesameletus*, and the caddisfly *Oxyethira*) were significantly higher at the two sites upstream of the road bridge than downstream. Densities of *Nesamaletus* were similar within sites 1 and 2 above the road, and within sites 3 and 4 below the road, while densities of *Oxyethira* were higher at the site immediately above the road bridge than at the uppermost site, but similar between the two sites located below the road bridge. Densities of the filter-feeding mayfly *Coloburiscus* were significantly higher at the two downstream sites, contrary to our expectation of a reduction in densities of mayflies below the road bridge. These results suggested that any stormwater derived contamination of the lower sites in this stream was having little or no effect on the densities of the most common taxa at this site.

3.4.2 Changes to community composition

NMDS ordination of the invertebrate data from artificial substrates placed at the SH58 stream showed a considerable degree of separation between samples collected above and below the state highway (Figure 21). Nested ANOSIM showed significant difference between sites within locations above or below the state highway ($R = 0.232$, $P < 0.001$), as well as significant differences between locations ($R = 0.237$, $P < 0.001$). Pairwise ANOSIM of the different site combinations (Table 5) showed that the biggest difference was observed between site 2 (immediately above the highway) and site 4 (the lowermost site), and the least difference between the two downstream sites (sites 3 and 4). Differences between sites 2 and 3 were also relatively small ($R = 0.275$), suggesting that any state highway derived runoff was having negligible effects on the overall community composition in this stream.

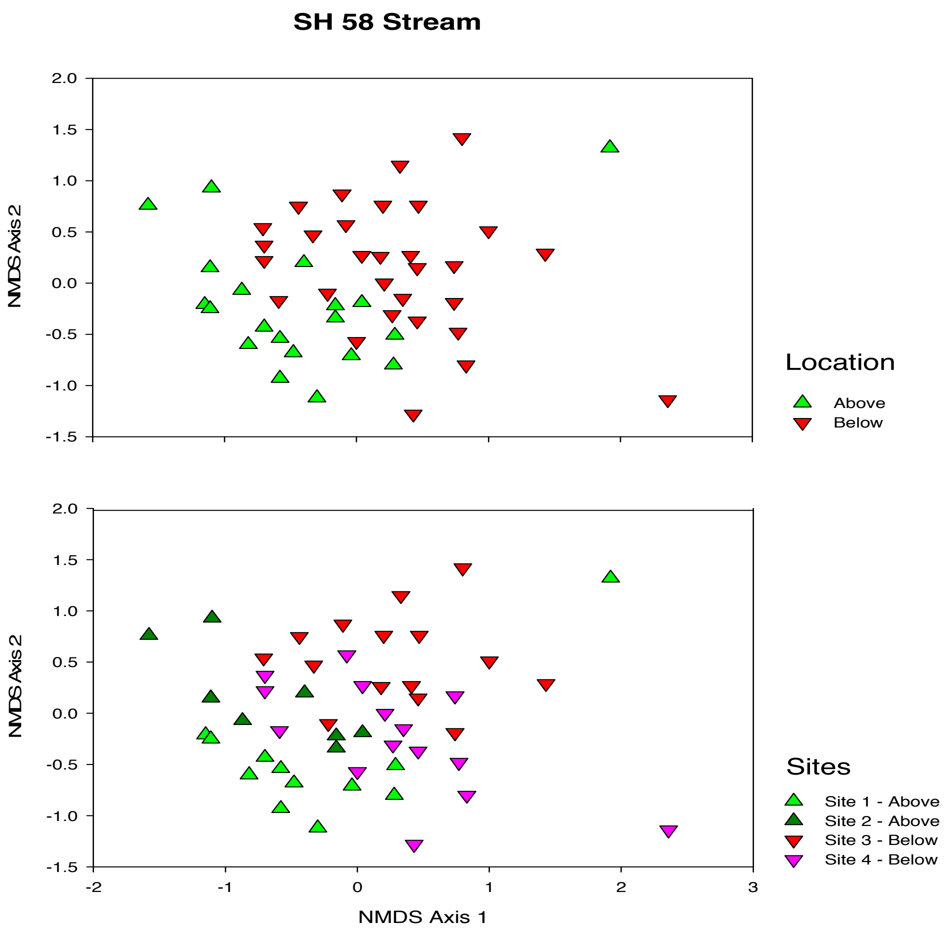


Figure 21. Results of NMDS ordination of invertebrate data collected from the SH 58 stream sites coded by location (upper graph) or sites number (lower graph).

Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
4,3	0.216	1.6	125970	999	15
4,2	0.482	0.1	17383860	999	0
4,1	0.316	0.1	17383860	999	0
3,2	0.275	0.3	490314	999	2
3,1	0.289	1.2	490314	999	11
2,1	0.239	0.1	77558760	999	0

Table 5. Results of pairwise ANOSIM to determine the differences in community composition in the 4 sites sampled in the small stream flowing under SH58.

3.5 Warkworth (Auckland) region – artificial substrates

3.5.1 Comparisons between natural and artificial substrates

Taxonomic richness was higher in samples collected from artificial macrophyte samples in both the North Branch and South Branch of the Mahurangi River than in samples collected from natural macrophytes in these same sites (Table 7). The number and percentage of EPT taxa was also higher in the artificial macrophytes than real ones in both sites (Table 7). Within the North Branch, densities of common taxa such as *Potamopyrgus*, and *Paracalliope* were very similar between artificial substrates and macrophytes, while densities of other common taxa (e.g., Oligochaeta and Orthocladinae) were much higher on artificial macrophytes. Densities of *Austrosimulium* and the midge *Polypedilum* were higher on natural macrophytes (Table 7). Within the South Branch, densities of the blackfly *Austrosimulium* were very similar between artificial and natural macrophytes, while densities of other common taxa differ greatly between these two habitat types (Table 7). NMDS ordination showed little grouping was evident between natural and artificial macrophytes in the North Branch, although artificial macrophytes from the South Branch were grouped separately to samples collected from real macrophytes here (Figure 22).

	North branch Artificial macrophytes	North branch Natural macrophytes	South branch Artificial macrophytes	South branch Natural macrophytes
Taxonomic Richness	49	25	45	38
EPT	14	10	19	17
Percentage EPT	2.0	1.3	5.3	3.8
Potamopyrgus	65.1	70.5	26.0	75.3
Oligochaeta	14.2	3.6	20.9	1.1
Austrosimulium	4.6	13.1	6.3	6.2
Orthocladiinae	2.8	0.9	15.6	1.9
Tanytarsus	0.2	0.06	19.7	0.4
Paracalliope	5.0	4.6	0.8	6.0
Polypedilum	0.1	2.0	0.6	3.8
Acarina	2.7	1.4	1.9	1.7

Table 7 Calculated biotic metrics, and densities of the most common taxa collected from artificial substrates and natural macrophytes in the North and South branches of the Mahurangi River.

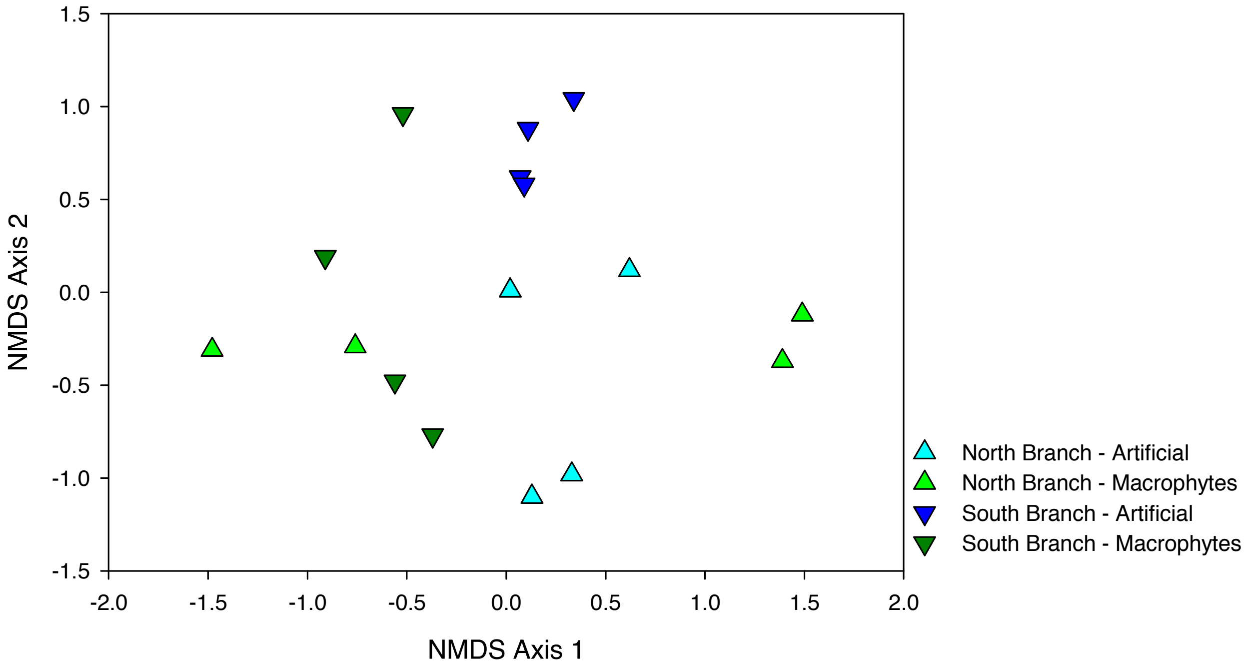
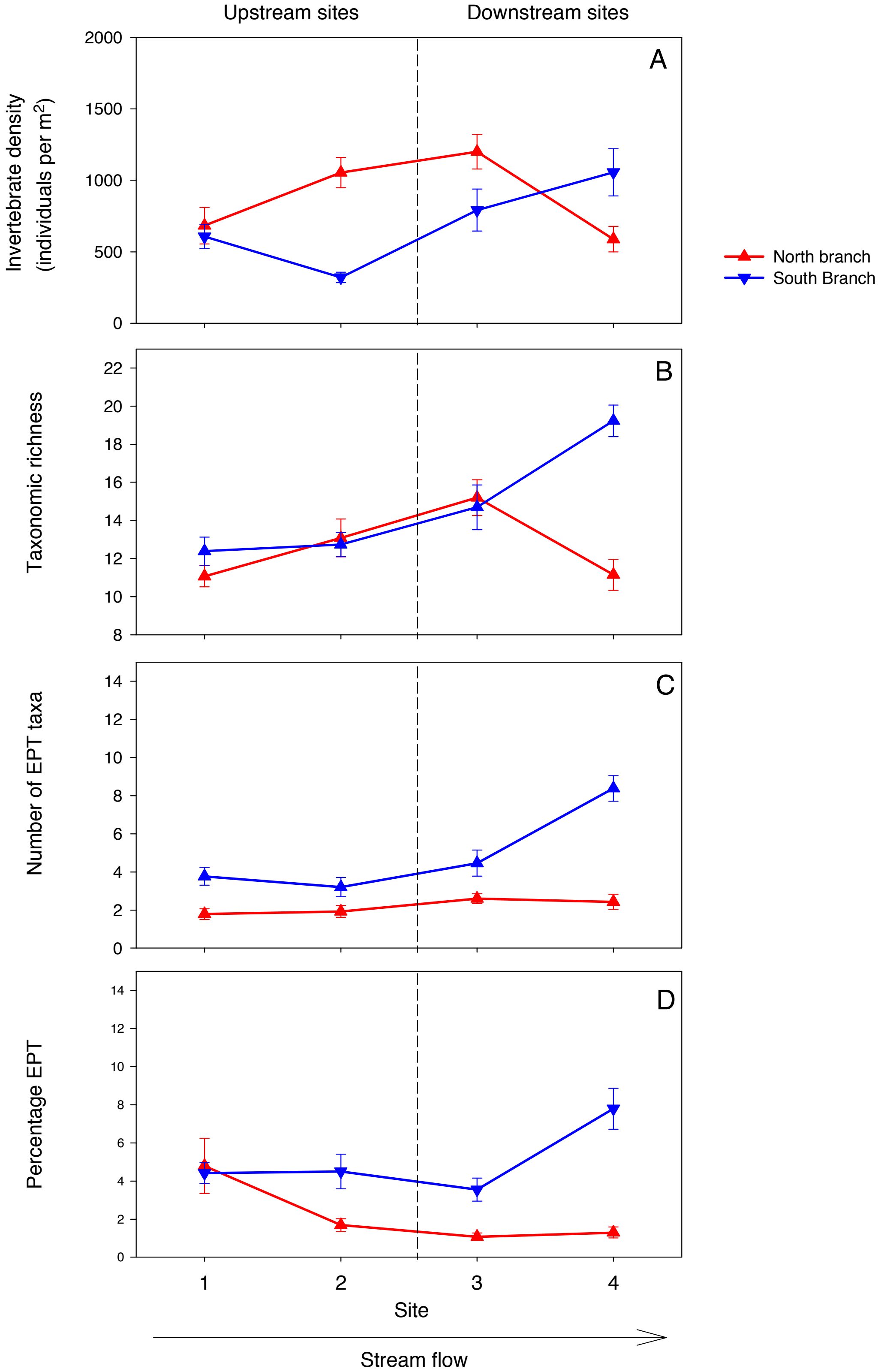


Figure 22 Results of NMDS ordination of invertebrate data collected from natural and artificial macrophytes collected from the north and south branches of Mahurangi River. For clarity, the graph shows only the mean of all replicate samples from artificial substrates at each site.

Despite these small differences in ordination scores for samples collected from the South Branch, nested ANOSIM showed that there was no significant difference to the invertebrate communities found on artificial substrates and macrophytes in both streams ($R = 0.25$, $P = 0.66$). There were however significant differences in the invertebrate communities found in the North or South branches ($R = 0.417$, $P = 0.05$). Absence of a large difference in invertebrate community composition between artificial substrates and macrophytes gave us confidence in being able to use the artificial macrophytes as good analogues for the invertebrate communities found in the Mahurangi River.

3.5.2 The invertebrate fauna

A total of 63 taxa were collected from artificial substrates placed in the North and South branches of the Mahurangi River. This fauna was dominated by the snail *Potamopyrgus,* oligochaete worms, the midges Orthocladiinae and Tanytarsus, the blackfly *Austrosimulium*, the amphipod *Paracalliope*, and the mayfly *Zephlebia*. Nested ANOVA of samples collected from the North Branch showed no significant difference in invertebrate abundance, taxonomic richness, or the number or percentage of EPT and EPT* taxa, and the MCI-sb above and below the state highway (Figure 23 A-H). The only significant location differences were for the number of EPT*, which were higher below the state highway than above, and calculated QMCI-sb scores which were higher above than below (Figure 23E and H). None of these metrics displayed consistent differences that could be attributed to the effects of road runoff.



Upstream sites
Downstream sites
A
B
C
D
Invertebrate density (individuals per m2)
Taxonomic richness
Number of EPT taxa
Percentage EPT
North branch
South Branch
Site
Stream flow

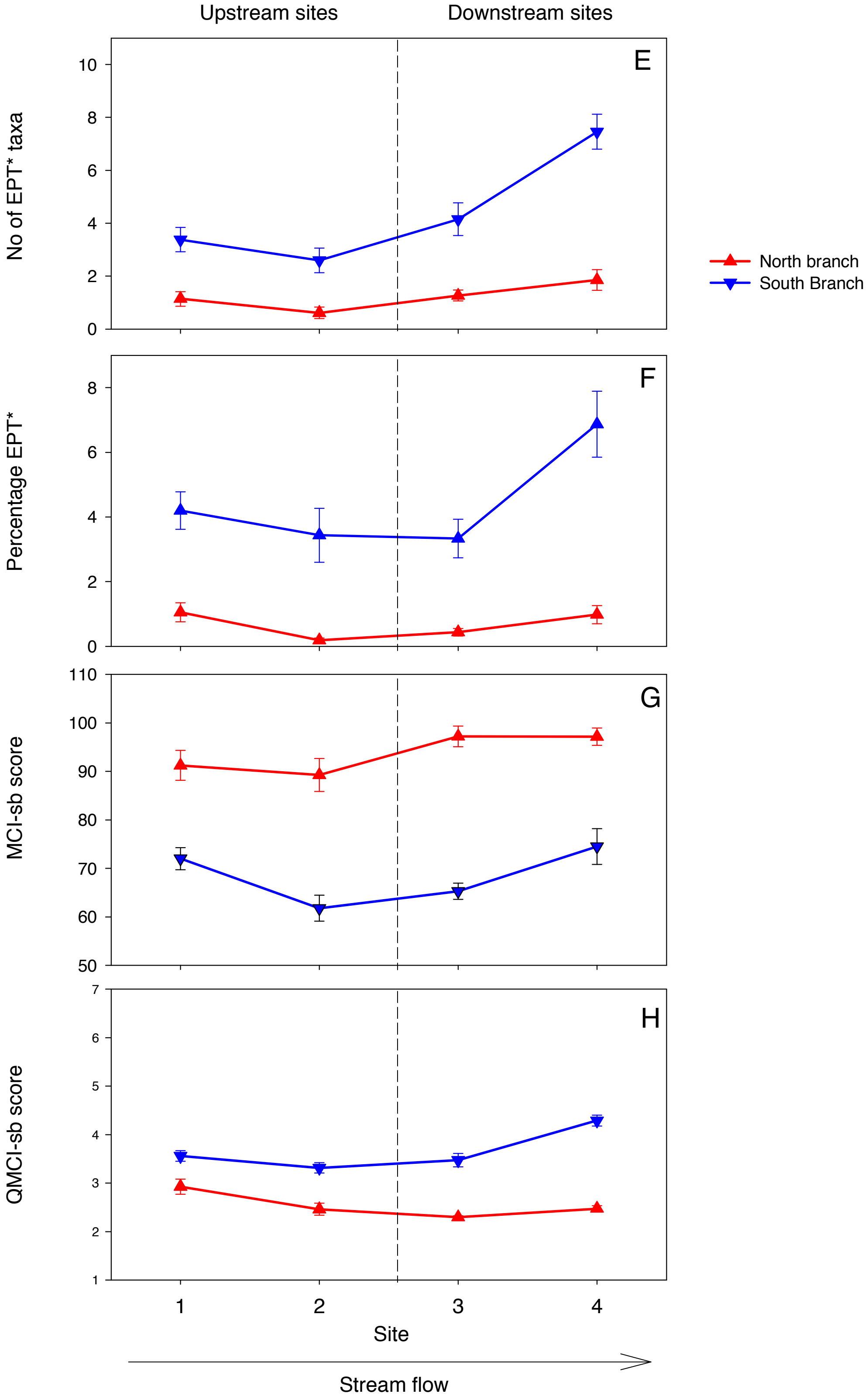


Figure 23 A-H. Invertebrate abundance, taxonomic richness, the number of EPT and EPT* taxa and the % of EPT and EPT* taxa in samples collected from artificial macrophyte mimics placed in the North and South branches of the Maharangi Stream flowing under SH1(x ± 1se, *n* = 15).

Of the 10 most common taxa collected at the North Branch, densities of 3 did not differ between sites above or below the state highway. Densities of oligochaetes and Orthoclad midges were significantly higher at sites above the road bridge than below, while densities of the snail *Potmopyrgus* were significantly higher at the downstream sites. Absence of strong and consistent trends to density differences above and below the road bridge suggest that any stormwater derived contamination of the lower sites in the North Branch of the Mahurangi was having little or no effect on invertebrate communities there.

For the South Branch, nested ANOVA showed that total invertebrate abundance, taxonomic richness, and the number of EPT and EPT*, the MCI-sb and QMCI-sb were significantly higher at sites below the state highway than above (Figure 23 A-H). Differences within sites were apparent only below the state highway, where these metrics had higher values in the lowermost sites than the site closest to the state highway. These had similar values as to the sites above the state highway. Densities of orthoclad and *Tanytarsus* midges were also significantly higher at sites below the road bridge than above. However, further examination of the data showed that these density differences were due mainly to low densities of these taxa at the site immediately above the road bridge: densities at the upper most site were similar to densities at the lower sites. Lack of strong and consistent trends as hypothesised also suggested that any stormwater derived contamination of the lower sites in the South Branch of the Mahurangi was having little or no effect on invertebrate communities here.

3.5.3 Changes to community composition

NMDS ordination of the invertebrate data from artificial substrates placed in the North Branch of the Mahurangi showed little separation between samples collected above and below the state highway (Figure 24), but clear separation of samples according to the site from where they were collected from (Figure 24). Nested ANOSIM showed significant differences between sites within locations above or below the state highway ($R = 0.681$, $P < 0.001$), but no significant difference between locations ($R = 0.175$, $P > 0.05$). Pairwise ANOSIM of the different site combinations showed that the biggest difference was observed between site 1 (the most upstream site above the highway) and site 3 (the site immediately below the highway: Table 8), and the least difference between the sites 2 and 3 immediately above and below the highway ($R = 0.143$), suggesting that any state highway derived runoff was having negligible effects on the overall community composition in this stream.

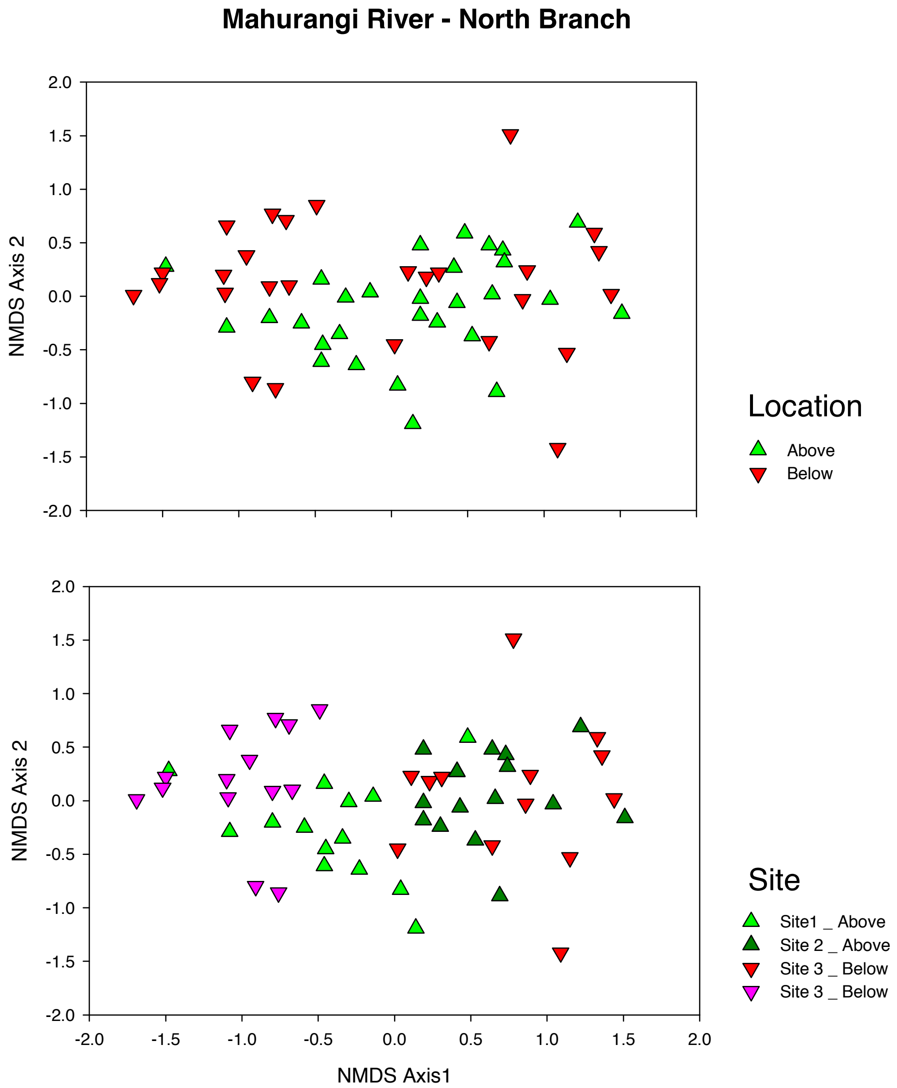


Figure 24. Results of NMDS ordination of invertebrate data collected from artificial substrates placed in the Mahuranghi River North Branch, coded by location (upper graph) or sites number (lower graph).

Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
4,3	0.599	0.1	77558760	999	0
4,2	0.594	0.1	20058300	999	0
4,1	0.537	0.1	20058300	999	0
3,2	0.143	0.6	37442160	999	5
3,1	0.873	0.1	77558760	999	0
2,1	0.776	0.1	20058300	999	0

Table 8 Results of pairwise ANOSIM to determine the differences in invertebrate community composition collected from artificial macrophytes in the 4 sites sampled in the North Branch of the Mahurangi River.

A different pattern was observed from artificial substrates placed in the South Branch of the Mahurangi River. Here, the NMDS showed some separation between samples collected above and below the state highway (Figure 25), as well as separation of samples according to the site where they were collected from (Figure 25). Nested ANOSIM showed significant differences between sites within locations above or below the state highway ($R = 0.245$, $P < 0.001$), as well as a significant difference between locations ($R = 0.195$, $P <0.01$). Pairwise ANOSIM of the different site combinations showed that the biggest difference was observed between site 4 (the lowermost site below the highway) and site 2 (the site immediately above the highway), and the smallest difference (R = 0.143) between the sites 1 (the upper most site above the highway) and site 3 (immediately below the highway: Table 9), suggesting that any state highway derived runoff was having negligible effects on the overall community composition in this stream. ANOSIM also showed only very small differences in community composition between sites 2 and 3 (R = 0.156), suggesting that any road derived runoff was having little effect on the invertebrate community at this stream.

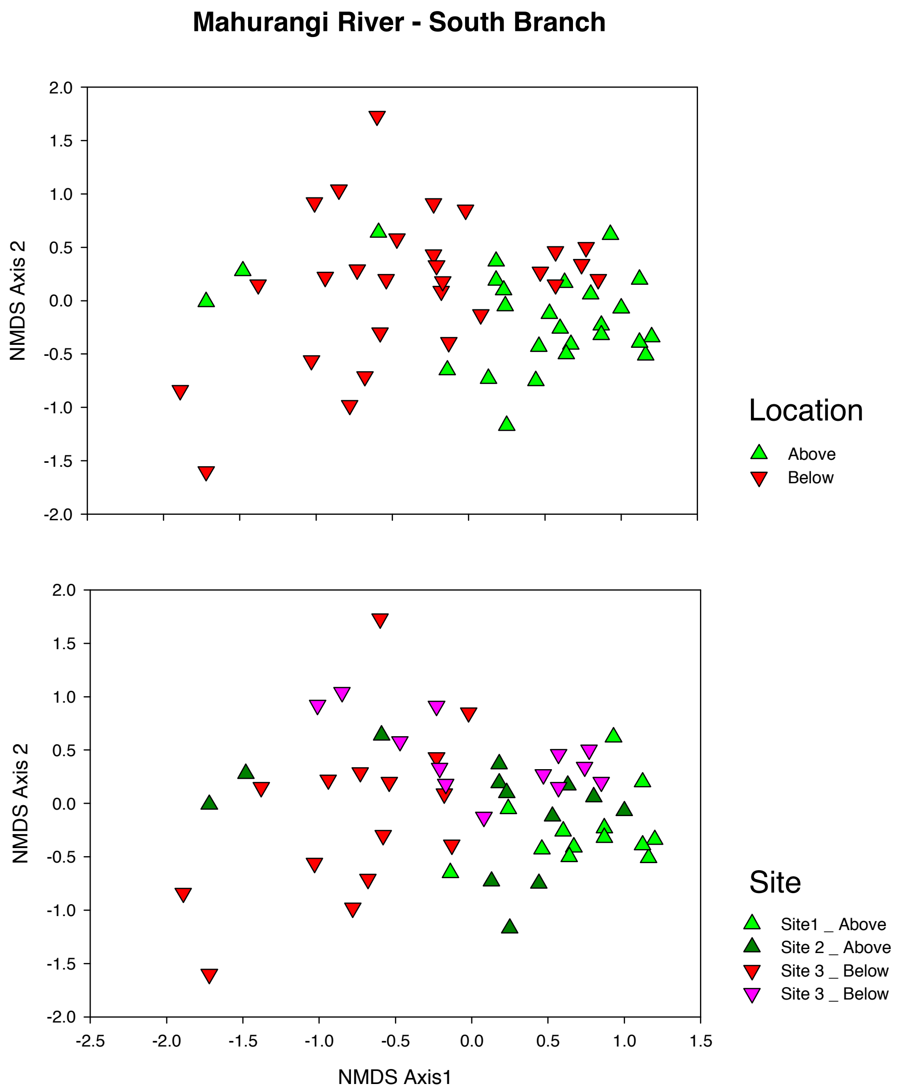


Figure 25. Results of NMDS ordination of invertebrate data collected from artificial substrates placed in the Mahuranghi River South Branch, coded by location (upper graph) or sites number (lower graph).

Groups	R Statistic	Significance Level %	Possible Permutations	Actual Permutations	Number >= Observed
4,3	0.268	0.1	5200300	999	0
4,2	0.567	0.1	37442160	999	0
4,1	0.437	0.1	5200300	999	0
3,2	0.156	1.2	37442160	999	11
3,1	0.103	3.6	5200300	999	35
2,1	0.225	0.1	37442160	999	0

Table 9 Results of pairwise ANOSIM to determine the differences in invertebrate community composition collected from artificial macrophytes in the 4 sites sampled in the South Branch of the Mahurangi River.

4 Discussion

This study aimed to quantify changes to the freshwater invertebrate communities above and below state highways that may have arisen due to the input of contaminated stormwater from these roads. It was based on the assumption that the invertebrate communities present in the streams would respond in a predictable manner to any stormwater discharges, which often contain chemicals that may have adverse ecological effects. Only streams in good ecological condition and not flowing through either urban or intensively farmed areas were selected. In this way, any adverse effects of land-use changes which may have resulted in a loss of sensitive tax such as mayflies, caddisflys, or stoneflys (Quinn and Cooper, 1997; Scott et al 1994; Winterbourn, 1986) were minimised. Furthermore, by restricting our selection to only small streams, the effects of dilution of any stormwater would have been reduced. When rated according to the system developed by Gardiner and Armstrong (2007) for identifying sensitive receiving environments (see Section 2.2), four streams were deemed to be of high sensitivity and two streams to be of medium sensitivity. Despite this sensitivity, we detected possible changes in the invertebrate communities only at Smith Creek that may have reflected the effects of road runoff.

At Smith Creek we detected changes to the invertebrate community that were consistent with our hypothesised changes that would occur to invertebrates as a result of stormwater runoff. Densities of four common taxa (*Cricotopus*, *Naonella*, Oligochaeta and *Deleatidium*) behaved in ways consistent with our hypothesis, as did four of the calculated biotic metrics (percentage EPT and EPT*, and calculated MCI and QMCI scores). These differences were thought unlikely to reflect differences in the physical habitat in the stream, as no consistent differences existed in measured depth or velocity between the different sites and locations (unpublished data). Substrate size was slightly larger in the site immediately below the state highway, but this was also unlikely to have greatly influenced the results, and in particular the percentage EPT, and calculated MCI and QMCI values, and density of the mayfly *Deleatidium*. Taxa with high scores for these metrics, and *Deleatidium* itself, in general favour coarser substrates, which were characteristic of the downstream site. However, the values of these metrics were lower at this site, counter to what would have been expected if the greater substrate size here was affecting the invertebrate communities. It is likely, therefore, that the observed differences in the metrics and densities of the four taxa were in fact responding to water, periphyton or sediment quality.

Although the ANOSIM analysis did not detect a large change in the invertebrate community composition in the site immediately above and below the state highway, this may have reflected the fact that a measurement of overall community similarity takes into consideration all the taxa in the samples. This would include a mixture of taxa potentially tolerant to metals (such as midges and snails), as well as more sensitive taxa such as *Deleatidium* (Hickey and Vickers 1992). This may help explain the apparently anomalous result of detecting a potential effect of road runoff on individual biotic metrics and densities of specific taxa, and not detecting a strong effect on the entire community composition based on pairwise differences.

We also observed a consistent, albeit slight, increase in concentrations of heavy metals in the periphyton below the state highway at this site. Such an increase is also consistent with the hypothesis of metal contaminated runoff being able to enter the stream via runoff, where the dissolved fraction is taken up by periphyton. However, the magnitude of this increase was very small, and concentrations only five times those of upstream periphyton were observed (and only for zinc). These observed concentrations were all below ANZECC interim sediment contaminant guidelines, and were all at least an order of magnitude less than the metal concentrations observed in algae by Hickey and Clements (1998), Suren and Elliott (2004), or Davis and George (1987): indeed Suren and Elliott found higher metal concentrations in periphyton collected from a rural stream not exposed to road runoff (for example, zinc concentration of 43 mg/kg). As such, it is considered unlikely that the increased metal concentrations observed in periphyton were having a large effect on the invertebrates here. However, the observed changes to some aspects of the invertebrate community, combined with elevated metal concentrations in periphyton collected from cobbles below the state highway are consistent with our hypothesised effects associated with road derived runoff.

Despite this evidence of an effect of road runoff at Smith Creek, there may have been another plausible reason for the reduction in some of the biotic metrics at the site immediately below the state highway. A sheep underpass had been installed next to the stream above this site (Figure A8 in Appendix 1), and use of this pass may have resulted in some localised sediment or nutrient enrichment at this site. This too, would have reduced the calculated biotic metrics, and help explain increased densities of the midges *Naonella* and *Cricotopus* and of Oligochaeta, as these taxa are tolerant of enriched and relatively silty conditions. Densities of *Deleatidium* would have declined as well, reflecting its preference for generally non-silty substrates that are not thickly covered with algae (Suren 2005). Whether any sediment inputs would also cause an increase in metal concentration in the periphyton at this site is however unknown. Given the presence of this potentially small-scale disturbance at the highway bridge, we cannot therefore unequivocally state that the observed differences were the result of road runoff.

Despite the apparent sensitivity of the other five streams studied as ranked by the Gardiner and Armstrong (2007) method, we detected no significant differences in the invertebrate communities upstream and downstream of the state highways. Lack of a stronger positive result was surprising, especially when considering the large number of studies that have documented the nature of stormwater runoff from roads (see Kennedy and Gadd 2003; Moores et al 2010 for New Zealand reviews), and which have highlighted the potentially toxic nature of this material (e.g., Maltby et al. 1995a; Wheeler et al 2005). It was also surprising in the light of a review of the impacts of highways and subsequent landscape urbanisation on stream habitat and biota (Wheeler et al. 2005) where highways were regarded as "*pervasive, pernicious threats to stream ecosystems because of their short and long-term physical, chemical, and biological impacts.*" However, our results generally did not support this contention, nor are they without precedent. Both Perdikaki and Mason (1999) in East Anglia in the UK, and Smith and Kaster (1983) in Milwaukee, Wisconsin, USA found little differences in stream macroinvertebrate communities above and below rural highways. The study in East Anglia looked at nine rivers above and below crossings of the A12 and A14 trunk roads that had an average annual daily traffic (AADT) load of 67,000 and 31,560 vehicles respectively. The highway in the Wisconsin study was less busy, having an AADT of only 8000 vehicles. Estimated AADTs of the state highways in our study were intermediate between these two studies, and so the observed lack of a response was unlikely to reflect a low traffic load. However, road runoff from the highways examined by Perdikaki and Mason, by Smith and Kaster, and indeed in this study travelled through unpaved ditches or vegetative filter drains.

There are several possible reasons for an absence of effect in five of the six streams examined. Firstly, the contaminant loads generated on the road may be lower than expected. Secondly, there may have been treatment of the road runoff prior to entering the streams. Thirdly, hydrological factors may have confounded the results at some sites. These issues are discussed further below.

Vehicle generated contaminant loads are not always correlated to the number of vehicles. In a study of runoff from four road sites of differing traffic characteristics, (two highly congested roads and two relatively uncongested roads) Moores et al (2010) found that vehicle emission factors were highest on the more congested roads and lower on roads with free flowing traffic. Previously, heavy metal contamination has been reported higher at sites where braking and acceleration are common, such as motorway off-ramps, or intersections (Drapper et al 2000; Kennedy, 2003; Kennedy and Gadd, 2003; Muschack, 1990). The road at most of the sites in the current study was either straight, or had only relatively gentle bends, and as such vehicle traffic would not be expected to be constantly braking or accelerating. As such, the degree of potential contaminant load may not be as high as would be expected from examination of traffic volume (as AADT) alone. This assertion is supported by other studies which have shown traffic volumes are not necessarily good indicators of contaminant concentrations in the road runoff (see references in Moores et al 2010).

The loads generated may have been further reduced by pre-treatment of road runoff prior to entering the streams. Vegetated roadside drainage channels were the pathway for transport of the road runoff into five of the streams studied. Although these channels are not designed for stormwater treatment, they can reduce contaminant concentrations and loads. Moores et al (2010) quantified and documented runoff from four road sites of differing traffic characteristics and runoff treatment devices. Four runoff treatment types were also investigated: 1) none; 2) discharge to swales, or 3) drainage channels, and 4) discharge to a stormwater pond. They also found that the four treatment devices differed greatly in their efficiency of contaminant removal. The roadside drainage channel was the most effective at removing total solids, copper and zinc, despite not being constructed specifically for this task. For example, one roadside drainage channel (at Huapai) removed approximately 96% of total suspended solids, as well as particulate and total copper and zinc, while a vegetated swale (at Northcote) also removed more than 90% of copper and zinc. This contrasted sharply with the observed poor performance of the specially constructed stormwater treatment pond, which was ineffective at removing dissolved metals (removing only 71% of total suspended solids, 40% of total copper, and 67% of total zinc (Moores et al 2010)). The good performance of the drainage channel was thought to reflect the high degree to which runoff was able to infiltrate into the soil, which is where most of the treatment to absorb heavy metal contamination is thought to occur (Moores et al 2010). It is likely that the vegetated roadside drainage channels at five of the streams investigated in the current study reduced the loads of contaminants entering the streams. Vegetated roadside drainage channels are common on New Zealand State Highways. As such, the results of this study lend strength to the notion that road runoff from the many hundreds of kilometres of state highway in New Zealand may be unlikely to have a demonstrable impact on aquatic ecosystems due in part to the effectiveness of vegetated roadside channels in trapping runoff. Although we obviously cannot extend the results of this limited study to encompass all streams that flow under state highways throughout the country, it must be remembered that we had already confined our sampling to roads with an AADT exceeding 10,000 vehicles. Lack of a strong effect from the state highways we selected suggest that adverse effects would be even more unlikely in parts of the highway network that have an AADT of less than 10,000 vehicles. This would include highways throughout much of the South Island, in the central west of North Island, and around East Cape of the North Island (see Figure 4).

Significant variation in storm related stream flows may have impacted the invertebrate results especially for the Warkworth sites. This is mainly from a flow depth and velocity perspective rather than from a water quality aspect. There was significant evidence of stream cross-sectional scour as is evident by the loss of the initial set of artificial substrates deployed, and of the temperature data loggers. However, the second series of artificial substrates was not affected by the five small freshes that occurred during their deployment, but we were unable to detect any effect of road runoff on the invertebrate communities colonising these structures. The implication here is that incised sites subject to frequent flood events, where instream velocities and shear stress can be high, may not be affected by road derived runoff as much as more stable sites, due to the overarching hydrological constraints on the invertebrate communities.

Gardiner and Armstrong (2007) present a *source - pathway - receptor* model for examining the effects of road runoff and maintain that for a risk to be present to a water body, all three components of the source-pathway-receptor model must exist. The overall pollution risk factor for a particular waterway is calculated from the source strength of contaminants, the nature of the pathway that the contaminant takes as it leaves the road to the where it is discharged, and the sensitivity of the receiving environment. The strength of the source term is approximated by a term called vehicle kilometres travelled (VKT), which is the AADT multiplied by the road length contributing runoff into the waterway. Gardiner and Armstrong recognised three pathway types that runoff takes between the source and the receiving environment:

1. Direct - storm runoff pipes directly to the sensitive receiving environment (SRE)
2. Indirect - urban highway with sump and culvert discharging to fast stream leading to the SRE
3. Diffuse - rural highway with stormwater allowed to run off carriageway onto the verge (i.e., no curb and channel collection system)

The pathway type for five of the streams investigated was diffuse, as all runoff would have entered vegetated drains along both sides of the road. The Pauatahanui Stream site, however, was characterised by the presence of curbs and channels on one side of the road, which conveyed stormwater to the stream via a pipe. This site thus had a direct pathway from source to the stream.

Calculation of the risk factors for the six streams examined showed that the Pauatahanui Stream at the biggest risk, whereas the small stream near Haywards has the least risk. The other four streams were all intermediate to these (Table 11). The high risk at the Pauatahanui Stream reflects a combination of the high pathway factor score, and high VKT. The high VKT reflected the approximately 1km long catchment with a curb and channel system that collected road runoff into a pipe and discharged it into the stream. Note that this stream had a lower SRE score, reflecting its higher gradient and faster velocities than the other sites. The risk rating results emphasise the importance of minimising the strength of the pathway factor, and allowing road runoff to simply percolate into vegetated swales and roadside ditches. In this way, the total load and the speed at which contaminated runoff can enter sensitive receiving environments is greatly reduced, which has a large effect on minimising the overall contamination at a site.

The results from the invertebrate monitoring do not reflect the pollution risk rating calculated above. The risk rating for Smith Creek is relatively low, whereas this was the only stream where a potential effect on invertebrate communities was observed. The Pauatahanui Stream had the highest risk rating yet showed no demonstrable effects of road runoff. This was unexpected given the direct entry of runoff into the stream at this site and the road conditions. The road here was at the base of a hill on a bend – conditions that may have favoured the production of contaminants as vehicles either braked downhill, or accelerated uphill. Despite this, we detected little change to the invertebrate community, and no increase in the concentration of zinc in periphyton there. Although VKT is regarded as a better indicator of potential contaminant source, it is recognised that this simple measure does not consider other factors that influence contaminant deposition such as long periods of dry weather prior to a rainfall events, vehicle braking and acceleration patterns, road design, adjacent land use and the composition of the vehicle fleet (Moores et al 2010). The straight roads at the sites investigated in this study may result in a lower risk than calculated by the risk rating system.

Stream	AADT	Length (m)	VKT	Pathway factor (Pf)f	Sensitivity Rating factor (SRf)a	Pollution risk rating (R (b)
SH58 Haywards	15000	270	4050	0.05	2.6	526
Pauatahanui	15000	1200	18000	0.8	2.9	41760
Smith Ck	35000	406	14210	0.05	3.5	2486
Mangaone Stream	35000	407	14245	0.05	4.6	3276
South Branch Mahurangi	15000	800	12000	0.05	3.5	2100
North Branch Mahurangi	15000	400	6000	0.05	3.5	1050

[a] SRE score divided by 12

[b] R = (S) × (Pf) × (SRf), and are arbitrary units.

Table 11 Calculation of pollution risk rating (R, in arbitrary units) for road runoff from the six sites examined in this study. The highest risk was found in the Pauatahanui stream, reflecting the high strength of the source (as VKT) and high pathway factor, despite the low sensitivity of this stream.

Overall, the lack of any large and consistent effect of road runoff on the macroinvertebrates communities in even the Pauatahanui site (which had the highest pollution risk rating) most likely reflected a combination of low contaminant inputs into the streams due to the presence of vegetated ditches, and the fact that most of the vehicles using these roadways were doing so at a constant speed and were not engaged in braking, thus further minimising the amounts of heavy metals released on to the road surface.

It should also be noted that the NZTA has recently implemented a Stormwater Treatment Standard for State Highway Infrastructure (NZTA May, 2010) whose implementation on new state highway projects will further reduce the impact that highways have on freshwater streams. This Standard addresses water quantity and quality issues related to state highways.

5 Protocols for use on a wider basis

This study was undertaken to firstly assess the impact of roadside runoff from state highways on the invertebrate communities in streams, and secondly to help NZTA make informed decisions as to whether an invertebrate monitoring approach using such metrics as the MCI etc. is feasible to monitor the effects of state highway runoff. The salient results were that invertebrate communities in the streams studied showed little evidence of being affected by road runoff, despite potentially relatively high contaminant loads and despite the fact that all the streams sampled were in relatively good ecological condition and unaffected by stresses associated with urban runoff. Although we detected subtle increases in metal concentrations in periphyton in some of the sites below the state highways, the magnitude of this increase was considered low when compared to values in the published literature.

The likely reasons behind these results reflect a combination of the generally smooth-flowing vehicle behaviour, and the presence of vegetated roadside drains that road runoff flowed into. In five of the six streams, road runoff simply drained to the side of the state highway, where it presumably slowly percolated into the soil. Under such a scenario, the likelihood of direct runoff into the streams was minimal. Even at the Pauatahanui Stream site, where runoff was collected in a curb and channel system and transported to the stream via a stormwater pipe, no demonstrable differences in the invertebrate communities were observed. This may have reflected a combination of the fast flowing water there lowering the sensitivity of this receiving environment, and possibly a lower than expected contaminant load due to the generally free flowing traffic at this location.

The use of invertebrates to assess changes in stream health as a result of road runoff has not been trialled before within New Zealand, despite the preponderance of biological monitoring programs throughout the country for a wide range of different purposes. We successfully sampled invertebrates in a variety of streams, and despite not finding a consistent impact, suggest that such monitoring could be used in other freshwater environments throughout the country. However, a number of significant issues and challenges exist in using freshwater invertebrates to assess stream health. First is the need to sample streams which have not been impacted by other pressures such as those associated with intensive agriculture or urbanisation. Such activities will alter the invertebrate communities to such an extent that sensitive animals will not be found. As such, the remaining tolerant fauna may not respond to any further pressures arising from road runoff from state highways. The second issue is the fact that the effects of road runoff may only be manifested by changes to the abundance of selective taxa, or subtle changes to the percentage composition of the community. There may, however, be no changes to the overall species composition. Because of this, it is necessary to implement a quantitative sampling program to detect subtle potential differences. The first challenge with this requirement is to make sure any sampling protocol is robust enough to be able to detect and differentiate differences in invertebrate densities arising from the effects of road runoff from differences caused by natural variations to invertebrate densities along a stream continuum. This means that a high number of replicate samples may be required, as shown by the power analysis undertaken in this study. The second challenge with the requirement of quantitative sampling is to undertake this successfully in soft-bottomed streams. Although we successfully constructed and deployed artificial substrates in the two Mahurangi sites, these were washed away due to an unpredictable and unseasonable flood event. This danger will always be present in streams, especially in areas where rainfall events are common. The only way to mitigate against this is to ensure that artificial substrates are well anchored to the stream bed.

An advantage of using artificial substrates is, however, the reduced time it takes to process the sample. A normal sample collected from a soft bottomed stream would consist of a mixture of the leaf litter, woody debris, and fine substrate particles such as sand etc. The small invertebrates would be found amongst this material. It can take between two and three hours per sample to process, depending on the amount of organic matter that has to be searched through for invertebrates. However, using artificial substrates minimises the amount of organic material in a sample, which reduces sample processing time accordingly.

Based on the above, we recommend the following flowchart be implemented to determine whether biological monitoring using stream invertebrates should be considered for assessing the effects of road runoff (Figure 26). The first issue to consider is the need to restrict monitoring sites where road derived runoff is the major source of contamination – as discussed above, there is little to be gained in trying to detect a biological impact above and below a state highway in the catchment that is already impacted by land use changes such as urbanisation. Secondly, following the Gardiner and Armstrong (2007) model, monitoring is best done in only sites where a potential contamination source is likely to be high (Figure 24). Thus, information is needed on AADT, or more preferably VKT, as well as variables such as VEFs. Because this can be significantly influenced by traffic behaviour, we suggest that little may be gained by sampling state highways not subject to a high VKT, or where VEFs are low due to traffic behaviour. In addition, our results combined with the model developed by Gardiner and Armstrong (2007) suggest that there is little point in monitoring an impact in receiving environments that are not sensitive. Thus, any large fast flowing rivers are unlikely to display any effects whereas such effects may be more apparent in depositional environments such as slow flowing streams. Finally, we contend that the nature of the runoff pathway is important in determining whether an adverse effect of road runoff will be detected. Where the pathway factor is diffuse, we suggest that there is unlikely to be a demonstrable ecological effect as a result of binding of heavy metals and other contaminants with vegetation and the soil. In this way, little direct runoff of contaminated material into a stream will occur. However, where the pathways of runoff are direct through curb and channel and pipes, we suggest that biological monitoring could be used to determine whether such contaminated runoff is having an ecological effect.

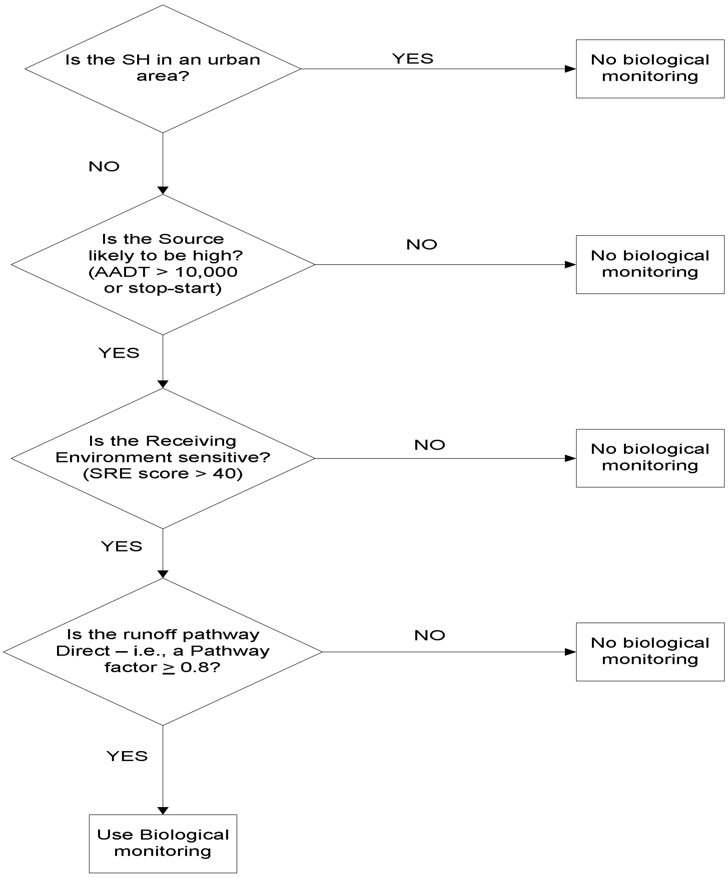


Figure 26 Decision support tree showing managers a series of questions allowing them to determine whether biological monitoring of streams above and below the state highways could be used to detect potential impacts of road runoff. (AADT = Average annual daily traffic; SRE score and Pathway Factor calculated as per Gardiner and Armstrong 2007)

If a biological monitoring protocol has been deemed appropriate, a logical initial step for this would be to undertake an initial screening assay of aquatic plants such as periphyton or aquatic mosses collected at sites above and below the state highway. This suggestion is based on results from a recent study by Ancion (2010) investigating the use of biofilm microbial communities (bacteria and fungi) as potential biomonitoring tools to detect metal contamination. He found that metals associated with such biofilms could indeed provide highly relevant indicators of the presence of metals in freshwater ecosystems at concentrations detrimental to aquatic biota. Although changes to periphyton communities above and below roads have yielded only equivocal results overseas (Boisson et al 2005 and 2006; Johnson et al 2011), periphyton does take up metals from the surrounding environment, and so become contaminated. This means that both periphyton and biofilms may act as good sentinel indicators for the presence of heavy metal contamination.

If these samples show evidence of bioaccumulation of metals (or PAHs), then quantitative invertebrate sampling should be considered to establish whether there are any ecological consequences of such contamination. Quantitative sampling is regarded as the most rigorous biomonitoring approach, as it is conceivable that potentially toxic effects of road runoff may have little or no effect on the overall community composition, but may instead reduce invertebrate densities. As discussed earlier, the natural variability of invertebrates in streams means that potentially large numbers of replicate samples need to be collected in order to accurately detect changes in densities over and above natural variation. A power analysis is a useful way to determine the number of replicate samples required to obtain a specific degree of certainty in being able to detect a particular difference in densities between upstream and downstream sites. If this is not done, and a relatively low number of samples (e.g., 5) are subsequently collected (as is commonly done) there is a risk that erroneous conclusions could be made in, for example, avoiding to detect an effect when indeed there was one.

The nature of the stream bed can also have a profound influence as to the type of sampling that is undertaken (Figure 27). As done in this study, artificial substrates should be used for soft bottomed streams where quantitative sampling is otherwise problematic. We developed a simple, relatively cost effective design using nylon rope threaded through a paving stone, which in turn was anchored to the clay streambed by a u-shaped bent wire. Similar designs could be utilised in other soft-bottomed streams. Where streams have a predominantly gravel and small cobble substrate, standard quantitative techniques (Stark et al 2001) can be used. Where the substrate is dominated by coarser large cobbles or boulders, quantitative samples can still be collected using the rock-rolling technique used in this study. In the case of small streams where enough suitable boulders cannot be sampled in a small enough area, artificial substrates such as paving stones should be deployed for invertebrate communities to colonise. These are easily bought at hardware or landscaping suppliers.

It is also important to realise that the creation of biotic metrics such as EPT and MCI is independent to the process of sampling. These metrics purely describe aspects of the invertebrate community that has been collected in a particular river by a particular method. Once the samples have been collected, these metrics are easily calculated, and can be analysed statistically to see whether they differ at sites above and below potential road runoff.

Note that we have only focused on biological monitoring using invertebrates, yet acknowledge that similar monitoring can be done using periphyton. However, results of detecting changes to periphyton communities above and below roads have yielded only equivocal results (Boisson et al 2005 and 2006; Johnson et al 2011), and the use of periphyton as an indicator of road runoff contamination within New Zealand has not yet been trialled. Of interest is a recent study by Ancion (2010) investigating the use of biofilm microbial communities (bacteria and fungi) as potential biomonitoring tools to detect metal contamination. He found that metals associated with such biofilms could indeed provide highly relevant indicators of the presence of metals in freshwater ecosystems at concentrations detrimental to aquatic biota. This means that both periphyton and biofilms may act as good sentinel indicators for the presence of heavy metal contamination.

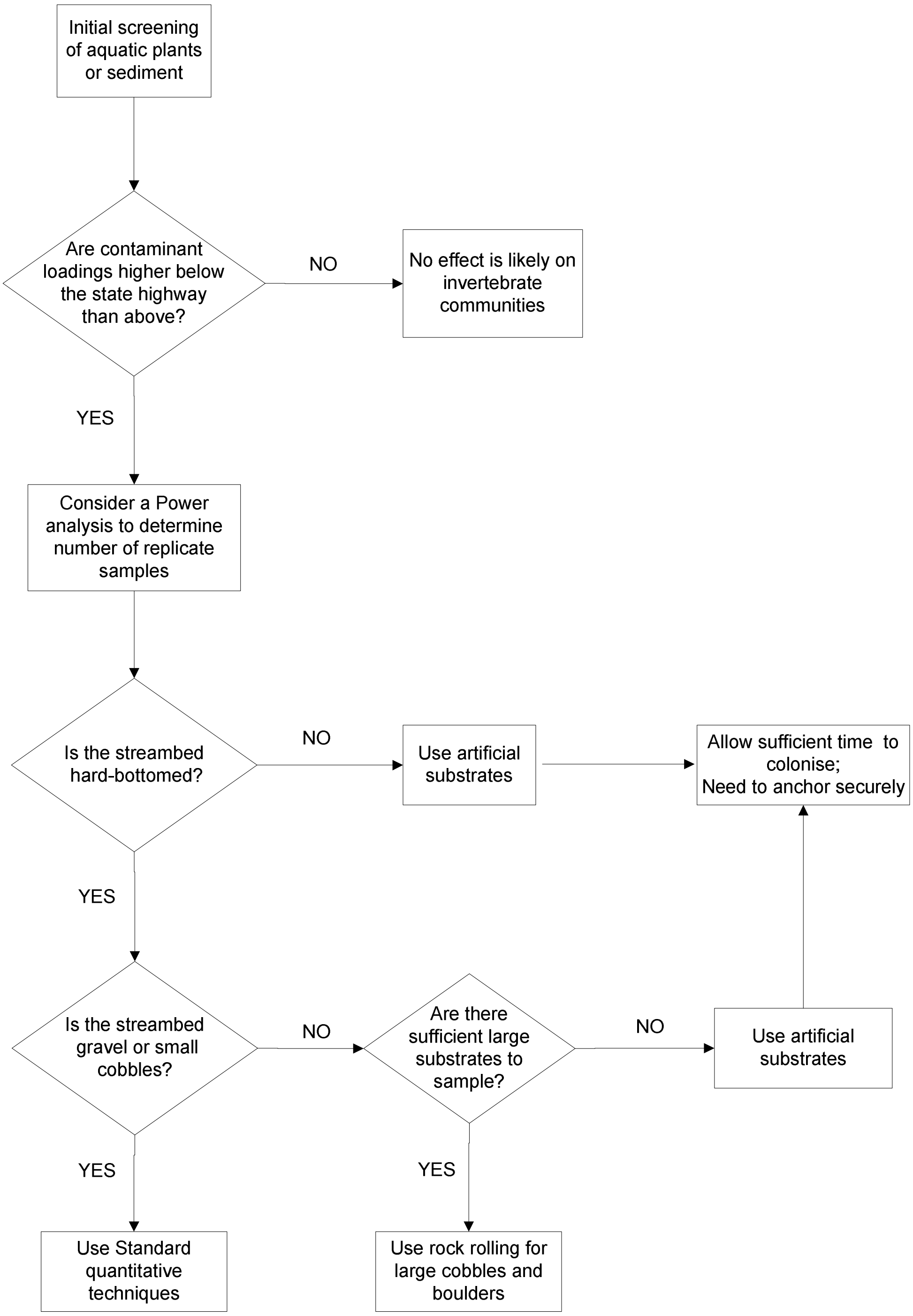


Figure 27 Decision support tree showing the different types of invertebrate sampling that can be undertaken depending on the type of biological sampling chosen, and the nature of the stream bed.

6 Conclusions

This study indicates that state highways with daily traffic volumes of approximately 10,000 vehicles/day in rural areas, having vegetated pathways and little variation in traffic speed, have minor impact on freshwater environments. This conclusion does not relate to culvert crossings where the culvert itself may impact aquatic life but rather relates to state highway generated stormwater runoff.

It could be argued that the lack of a strong and consistent response of the invertebrate communities to potential road runoff from the state highways may reflect the fact that the sites we selected received very little stormwater, as they were predominantly in rural locations. However, as mentioned earlier, the decision to sample such streams reflects the fact that more urban streams or streams in areas of higher intensity agricultural activity may have lost many of the sensitive invertebrates as a result of these land-use changes. As such, any extra pressure arising from road runoff was likely to be minimal in comparison to the pressures from urban development or intensive agriculture. The other reason we detected very little response reflected the fact that runoff from most of the roads examined simply flowed from the road into vegetated side drains or ditches, from where the water soaked into the ground. This appears to be the default treatment of runoff on many of the state highways we examined as part of our site selection, although we did not expressly quantify this. An obvious exception to the use of vegetated drains was some of the curb and channels observed on some of the larger bridges that crossed large rivers. However, adverse effects of road runoff from these sites would be extremely unlikely given the huge degree of dilution that any stormwater would be exposed to.

If the bulk of New Zealand's state highways do drain to ground, then the results of this study suggest that any adverse effects are likely to be, at worst, only minor. If, however, significant portions of curb and channel are used to convey stormwater into receiving waters, then potential adverse effects may still occur. This may be particularly true in areas of high AADT, such as on the new Northern Gateway Toll Road. Another aspect to consider is that we only examined the effects of potential road runoff on small streams. Although some of these were regarded as sensitive receiving environments, the fact that they were not depositional means that the effects of any runoff contamination may have been further reduced. There may indeed be an effect of road runoff in depositional environments such as those associated with freshwater wetlands, or non-urban estuaries. Although the contaminant loadings of estuaries have recently been quantified (Reed et al 2008), many of these estuaries were located within Auckland, and as such were exposed to runoff from both state highways, and from the urban environment in general. Moreover, this study did not attempt to sample the invertebrate communities within these environments. We thus feel that there is further scope to determine whether potential road runoff from state highways has an effect on such depositional environments in non-urban catchments.

7 References

Allibone R., Horrox J. and Parkyn S.M. (2001). Stream classification and instream objectives for Aucklands urban streams. *NIWA Client Report,* ARC00257, 1-79.
Ancion P.-Y. (2010). Use of stream biofilm microbial communities and associated metals as indicators of urban runoff impact on freshwater ecosystems. PhD Thesis. The University of Auckland. 188 p.
Barbour M.T., Gerritsen J., Griffith G.E., Frydenborg R., McGarron E., White J.S. and Bastian M.J. (1996). A framework for biological criteria for Florida streams using benthic macroinvertebrates. *Journal of the North American Benthological Society,* 15, 185-211.
Beach D. (2002). Coastal sprawl: The effects of urban design on aquatic ecosystems in the United States. *Pew Oceans Commission.* 1-33 p.
Biggs B.J.F. (1988). Artificial substrate exposure times for periphyton biomass estimates in rivers. *New Zealand Journal of Marine and Freshwater Research,* 22, 507-515.
Biggs B.J.F. and Kilroy C. (2000). *Stream Periphyton Monitoring Manual.* Ministry for the Environment, Wellington, New Zealand.
Biggs B.J.F., Duncan M.J., Suren A.M. and Holomuzki J.R. (2001). The importance of bed stability to benthic ecosystems. In: *Gravel-Bed Rivers V.* ((Eds) M.P. Mosley.), pp. 423-449. New Zealand Hydrological Society, Christchurch.
Boisson J.C., Boisson C. & Perrodin Y. (2005). The impacts of road runoff on periphyton in a small upland stream during summer. *Aquatic Ecosystem Health and Management,* 8,
Boisson, J. C. and Y. Perrodin (2006). "Effects of road runoff on biomass and metabolic activity of periphyton in experimental streams." *Journal of hazardous materials* A132: 148-154.
Bowden W.B., Arscott D., Pappathanasi D., Finlay J., Glime J.M., LaCroix J., Laio C.-L., Hershey A., Lampella T., Peterson B., Wolheim W., Slavik K., Shelley B., Chesterton M.B., Lachance J.A., LeBlanc R.N., Steinman A. and Suren A.M. (1997). Roles of bryophytes in stream ecosystems. *Journal of the North American Benthological Society,* 18, 151-184.
Clarke K. and Gorley R. (2001). *PRIMER v5: User Manual / Tutorial.* PRIMER-E, Plymouth.
Clarke K.R. (1993). Non-parametric multivariate analyses of change in community structure. *Australian Journal of Ecology,* 18, 117-143.
Collier K.J., Champion P.D. and Croker G.F. (1999). Patch - and reach-scale dynamics of a macrophyte - invertebrate system in a New Zealand Lowland stream. *Hydrobiologia,* 392, 89-97.
Davis J.B. and George J.J. (1987). Benthic invertebrates as indicators of urban and motorway discharges. *The science of the total environment,* 59, 291-302.
Death R.G. and Winterbourn M.J. (1995). Diversity patterns in stream benthic invertebrate communities: the influence of habitat stability. *Ecology,* 76, 1446-1460.
Drapper D.R., Tomlinson R. and Williams P. (2000). Pollutant concentrations in road runoff: South East Queensland case study. *Journal of Environmental engineering,* 126, 313-320.
Elliot J.M. (1977). *Some methods for the statistical analysis of samples of benthic invertebrates.* 2nd edn. Freshwater Biological Association, Ambleside.
Fuller R.L., Roelofs J.L. and Fry T.J. (1986). The importance of algae to stream invertebates. *Journal of the North American Benthological Society,* 5, 290-296.
Gardiner L. and Armstrong B. (2007). Identifying sensitive receiving environment at risk from road runoff. *Land transport New Zealand research report,* No 315, 1-68.
Gardiner L., Osborne A. and Reller C. (2008). Preliminary assessment of sensitive receiving environments at risk from state highway run off. New Zealand Water and Waste Association Conference, 20p.
Hall M.J., Closs G.P. and Riley R.H. (2001). Relationships between land use and stream invertebrate community structure in a South Island, New Zealand, coastal stream catchment. *New Zealand Journal of Marine and Freshwater Research,* 35, 591-603.
Hart D.D., Clark B.D. and Jasentuliyana A. (1996). Fine-scale field measurement of benthic flow environments inhabited by stream invertebrates. *Limnology and Oceanography,* 41, 297-308.
Herald J.R. (1989). Hydrological impact of urban development in the Albany Basin, Auckland. Ph. D Thesis. University of Auckland

Hickey C.W. and Clements W.H. (1998). Effects of heavy metals on benthic macroinvertebrate communities in New Zealand streams. *Environmental toxicology and chemistry,* 17, 2338-2346.
Hickey C.W. & Vickers M.L. (1992). Comparison of the sensitivity to heavy metals and pentachlorophenol of the mayflies *Deleatidium* spp. and the Cladoceran *Daphnia magna. New Zealand Journal of Marine and Freshwater Research,* 26, 87-93.
Hilsenhoff W.L. (1988). Seasonal correction factors for the biotic index. *The Great Lakes Entomologist,* 21, 9-13.
Johnson K.A., Steinman A.D., Keiper W.D. and Ruetz C.R., III. (2011). Biotic responses to low-concentration urban road runoff. *Journal of the North American Benthological Society,* 30, 710-727.
Jones R.C. and Clark C.C. (1987). Impact of watershed urbanization on stream insect communities. *Water Resources Bulletin,* 23, 1047-1055.
Kelly M.G., Girton C. and Whitton B.A. (1987). Use of moss-bags for monitoring heavy metals in rivers. *Water Research,* 21, 1429-1435.
Kennedy P. (2003). The effects of road transport on the freshwater and marine ecosystems. 1-142 p.
Kennedy P. and Gadd J. (2003). Evaluation of road surface contaminant loadings in Waitakere City for the development of the vehicle fleet emission model - water. 1-34 p.
Kouvelis B. and Armstrong B. (2004). Integrated stormwater management guidelines for the New Zealand roading network. *Transfund New Zealand research report,* 260, 1-166.
Lamberti G.A. and Resh V.H. (1985). Comparability of introduced tiles and natural substrates for sampling lotic bacteria, algae and macroinvertebrates. *Freshwater Biology,* 15, 21-30.
Lancaster J. and Hildrew A.G. (1993). Flow refugia and the microdistribution of lotic macroinvertebrates. *Journal of the North American Benthological Society,* 12, 385-393.
Lenat D.R. and Crawford J.K. (1994). Effects of land use on water quality and aquatic biota of three North Carolina piedmont streams. *Hydrobiologia,* 294, 185-199.
Macaskill J.B. and Williamson B. (1994). Water quality impacts of road runoff. *NIWA consultancy report,* MTR001, 1-42.
McDowall R.M. (1990). *New Zealand Freshwater Fish - a natural histroy and guide*. Heinemann Reed, Auckland.
Maltby L., Boxall A.B.A., Forrow D.M., Calow P. and Betton C. (1995a). The effects of motorway runoff on freshwater ecosystems: 2. Identifying major toxicants. *Environmental Toxicology and Chemistry,* 14, 1093-1101.
Maltby L., Forrow D.M., Boxall A.B.A., Calow P. and Betton C. (1995b). The effects of motorway runoff on freshwater ecosystems: 1. Field study. *Environmental Toxicology and Chemistry,* 14, 1079-1092.
Matthaei C.D., Arbuckle C.J. and Townsend C.R. (2000). Stable surface stones as refugia for invertebrates during disturbance in a New Zealand stream. *Journal of the North American Benthological Society,* 19, 82-93.

Maxted J.R., Evans B.F. and Scarsbrook M.R. (2003). Development of standard protocols for macroinvertebrate assessment of soft-bottomed streams in New Zealand. *New Zealand Journal of Marine and Freshwater Research,* 37, 793-807.
Meyer J.L., Paul M.J. and Taulbee W.K. (2005). Stream ecosystem function in urbanizing landscapes. *Journal of the North American Benthological Society,* 24, 602-612.
Moncrief I. and Kennedy P. (2002). *Road transport effects on aquatic ecosystems, issues and context for policy development.* Fuels and Energy Ltd, and Kingett Mitchell Ltd.
Moores J., Pattinson P. and Hyde C. (2010). Enhancing the control of contaminants from New Zealand's roads: results of a road runoff sampling program. *New Zealand transport agency research report,* No 395, 1-121.
Muschack W. (1990). Pollution of street traffic runoff by traffic and local conditions. *The Science of the Total Environment,* 93, 419-431.
NZ Transport Agency (2010) Stormwater Treatment Standard for State Highway Infrastructure, May 2010.
O'Riley A., Pandey S., Langdon A. and Wilkins A. (2002). Characterisation of runoff contaminants from New Zealand roads, and effects of rainfall events. *Transfund New Zealand Research Report,* 228

Paul M.J. and Meyer J.L. (2001). Streams in the Urban Landscape. *Annual Review of Ecology and Systematics,* 32, 333-365.

Percival E. and Whitehead H. (1929). A quantitative study of the fauna of some types of stream-bed. *Journal of Ecology,* 17, 282-314.

Perdikaki K. and Mason C.F. (1999). Impact of road runoff on receiving streams in eastern England. *Water Research,* 33, 1627-1633.

Pitt R. (2002). Receiving water impacts associated with urban wet weather flows. In: *Handbook of Ecotoxicology.* ((Eds) D. Hoffman; B. Rattner; G.A. Burton&J.J. Cairns.), CRC_Lweis, Boca Raton.

Pitt R. and Bozeman M. (1983). Sources of urban runoff pollution and its effect on an urban creek. *U.S. Environmental Protection Agency, EPA-600/S2-82-090*

Quinn J.M. and Cooper A.B. (1997). Land-water interactions at Whatawhata, New Zealand: introduction and synthesis. *New Zealand Journal of Marine and Freshwater Research,* 31, 569-577.

Reed J., Depree C., Trip E., Chappell P. and Talbot N. (2008). Sampling receiving environments close to state highways. NIWA Client Report AKL2008-035 for New Zealand Transport Agency.

Rosenberg D.M. and Resh V.H. (1993). *Freshwater biomonitoring and benthic macroinvertebrates.* Chapman and Hall, New York.

Scott D., White J.W., Rhodes D.S. and Koomen A. (1994). Invertebrate fauna of three streams in relation to land use in Southland, New Zealand. *New Zealand Journal of Marine and Freshwater Research,* 28, 277-290.

Shutes R.B.E. (1984). The influence of surface runoff on the macroinvertebrate fauna of an urban stream. *The Science of the Total Environment,* 33, 271-282.

Smith M.E. and Kaster J.L. (1983). Effect of rural highway runoff on stream benthic macroinvertebrates. *Environmental pollution,* 32, 157-170.

Stark J.D. (1985). A Macroinvertebrate Community Index of Water Quality for Stony Streams. *Water & Soil Miscellaneous Publications,* 87, 1-53.

Stark J.D. (1993). Performance of the Macroinvertebrate Community Index: effects of sampling method, sample replication, water depth, current velocity, and substratum on index values. *New Zealand Journal of Marine and Freshwater Research,* 27, 463-478.

Stark J.D., Boothroyd I.K.G., Harding J.S., Maxted J.R. and Scarsbrook M.R. (2001). Protocols for sampling macroinvertebrates in wadeable streams. *Ministry for the Environment. Sustainable Management Fund Project No. 5103.* 57p p.

Stark J.D. and Maxted J.R. (2007). A biotic index for New Zealand's soft bottom streams. *New Zealand Journal of Marine and Freshwater Research,* 41, 43-61.

Statzner B. and Higler B. (1986). Stream hydraulics as a major determinant of benthic invertebrate zonation patterns. *Freshwater Biology,* 16, 127-139.

Suren A.M. (1991a). Assessment of artificial bryophytes for invertebrate sampling in two New Zealand alpine streams. *New Zealand Journal of Marine and Freshwater Research,* 25, 101-112.

Suren A.M. (1991b). Bryophytes as invertebrate habitat in two New Zealand alpine streams. *Freshwater Biology,* 26, 399-418.

Suren A.M. (2005). Effects of deposited sediment on patch selection by two grazing stream invertebrates. *Hydrobiologia,* 549, 205-218.

Suren A.M. (2000). Effects of urbanisation. In: *New Zealand stream invertebrates: ecology and implications for management.* ((Eds) K.J. Collier&M.J. Winterbourn.), pp. 260-288. New Zealand Limnological Society, Christchurch.

Suren A.M. and Elliot S. (2004). Effects of urbanisation on streams. In: *Freshwaters of New Zealand.* ((Eds) J.S. Harding; M.P. Mosley; C.P. Pearson and B.K. Sorrell.), pp. 764. New Zealand hydrological Society Inc and New Zealand Limnological Society Inc., Christchurch.

Suren A.M. and Winterbourn M.J. (1992). The influence of periphyton, detritus and shelter on invertebrate colonization of aquatic bryophytes. *Freshwater Biology,* 27, 327-339.

Townsend C.R. (1989). The patch dynamics concept of stream community ecology. *Journal of the North American Benthological Society,* 8, 36-50.

Vaughan C.C. (1986). The role of periphyton abundance and quality in the microdistribution of a stream grazer, *Helicopsyche borealis* (Trichoptera: Helicopsyche). *Freshwater Biology,* 16, 485-493.

Walsh C.J. (2000). Urban impacts on the ecology of receiving waters: a framework for assessment, conservation and restoration. *Hydrobiologia,* 431, 107-114.

Walsh C.J., Roy A.H., Feminella J.W., Cottingham P.D., Groffman P.M. and Morgan R.P.I. (2005). The urban stream's syndrome: current knowledge and the search for a cure. *Journal of the North American Benthological Society,* 24, 706-723.

Wehr J.D., Say P.J. and Whitton B.A. (1981). Heavy metals in an industrially polluted river, the Team. In: *Heavy Metals in Northern England: Environmental and Biological Aspects.* ((Eds) P.J. Say&B.A. Whitton.), pp. 99-107. University of Durham, Durham.

Wheeler A.P., Angermeier P.L. and Rosenberger A.E. (2005). Impacts of new highways and subsequent landscape urbanisation on stream habitat and biota. *Reviews in fisheries science,* 13, 141-164.

Whitton B.A., Say P.J. and Jupp B.P. (1982). Accumulation of zinc, cadmium and lead by the aquatic liverwort *Scapania. Environmental Pollution Series B,* 3, 299-316.

Winterbourn M.J. (1986). Effects of land development on benthic stream communities. *New Zealand Agricultural Science,* 20, 115-118.

Wolman M. (1954). A method of sampling coarse river-bed material. *Transactions of the American Geophysical Union,* 35, 951-956.

APPENDIX 1

Study Site photos

Mangaone Stream

Figure A1 Mangaone Stream, Site 2 above SH1

Figure A2 Mangaone Stream, Site 3 below SH1

Figure A3 Mangaone Stream looking northward, showing grassy verge

Figure A4 Mangaone Stream looking southward, showing grassy verge (mown)

Smith Creek

Figure A5 Smith Creek Site 3 just below the SH1 bridge

Figure A6 Smith Creek looking northward, showing grassy verge along the road side

Figure A7 Smith Creek looking southward

Figure A8 Smith Creek looking southward showing the sheep pass under SH1 (arrowed) that may have contributed localised sediment or nutrient inputs to change invertebrate communities slightly

Pauatahanui Stream

Figure A9 Pauatahanui Stream, Site 2 above SH58

Figure A10 Pauatahanui Stream, Site 3 below SH58

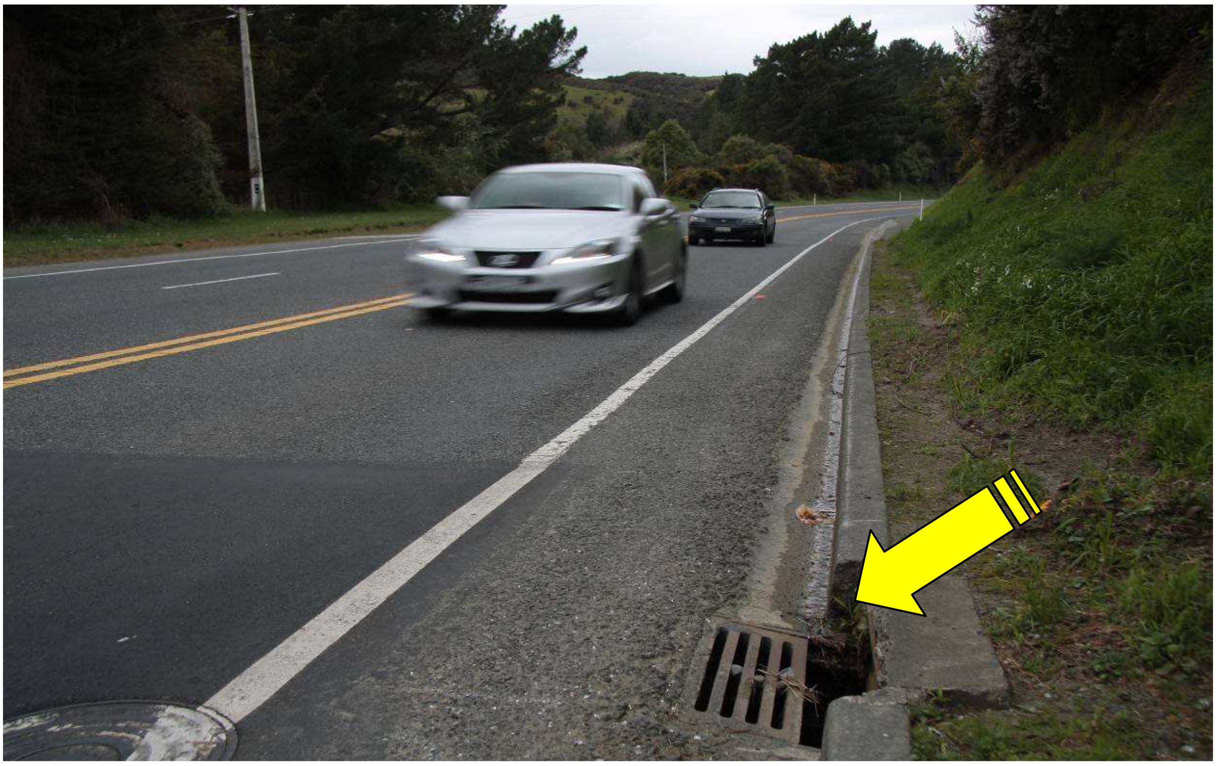


Figure A11 Roadside conditions at Pauatahanui Stream looking south at the base of a hill, showing the curb and channel and entry to the stormwater drain pipe that discharges o the stream (arrowed)

Figure A12 Roadside conditions at Pauatahanui Stream looking north back towards the stream at the base of the hill, again showing the channel carrying water to the stormwater drain pipe.

Unnamed Stream SH58

Figure A13 Unnamed Stream under SH58, Site 2 above the road bridge

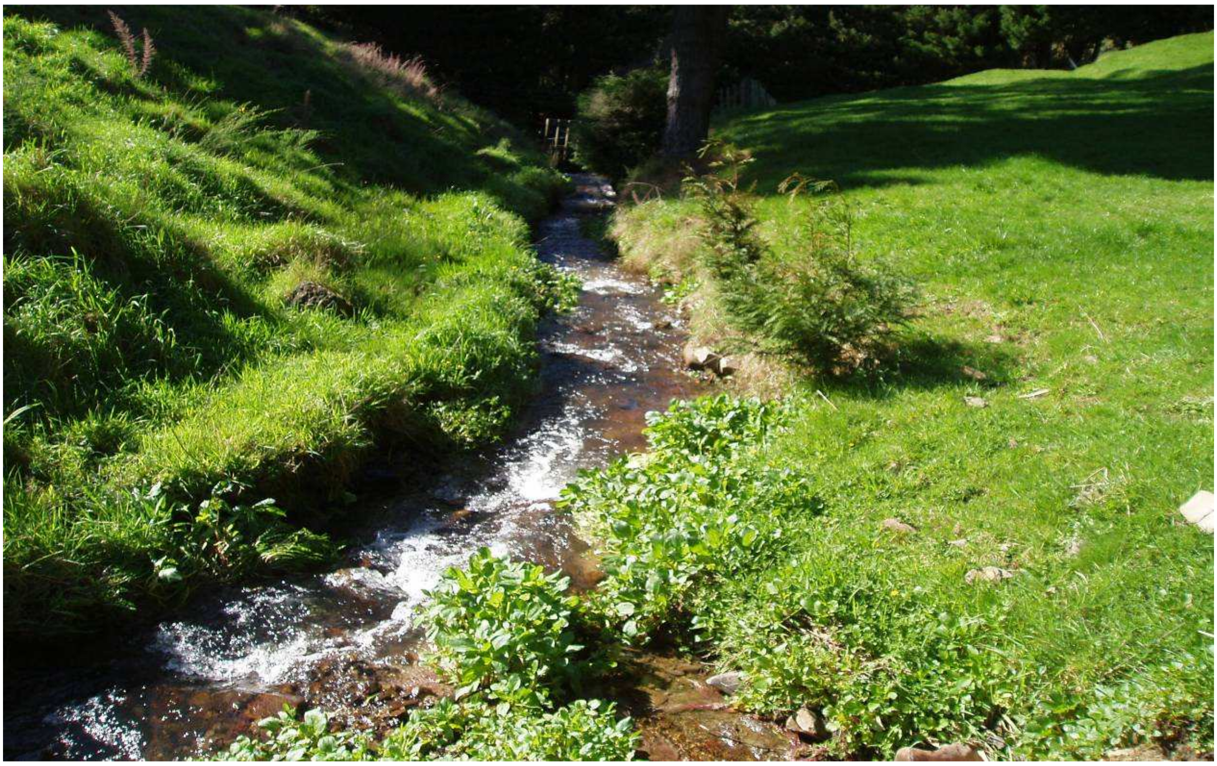


Figure A14 Unnamed Stream under SH58, Site 3 below the road bridge

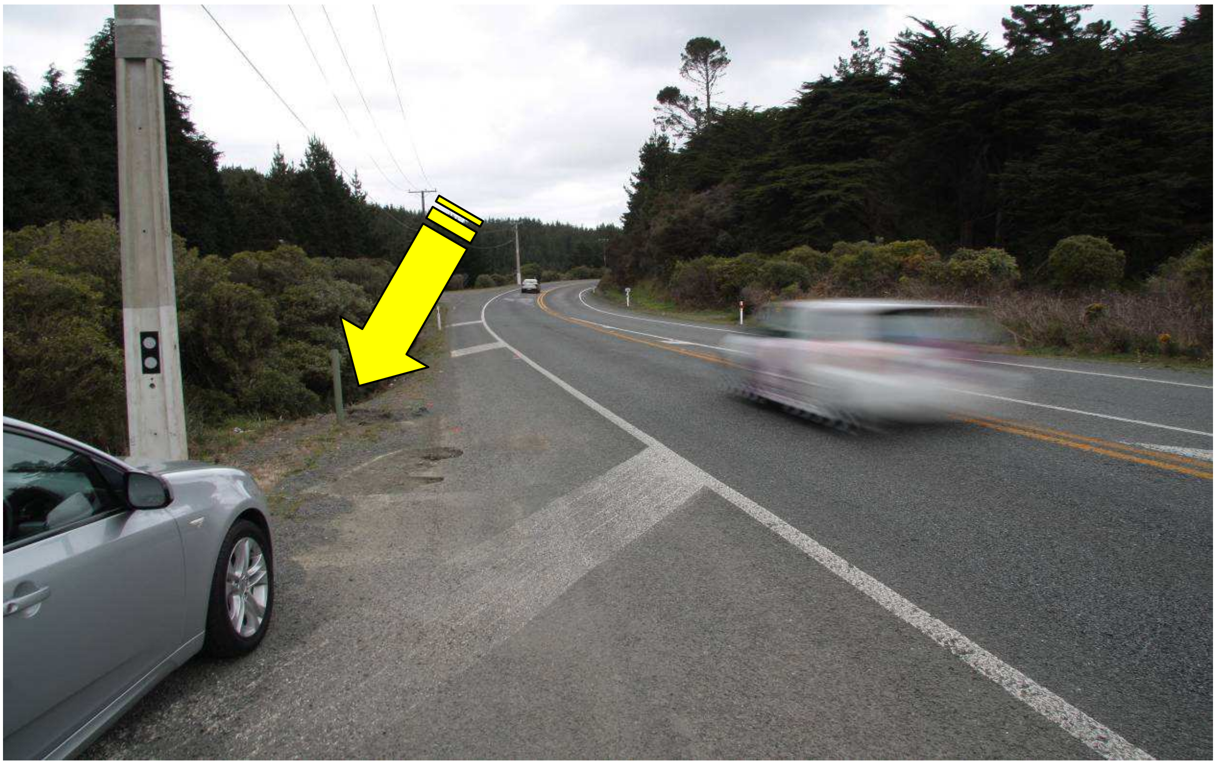


Figure A15 Roadside conditions at the small unnamed SH58 stream looking south towards a slight uphill bend in the road. The location of the stream below the road is shown (arrowed).

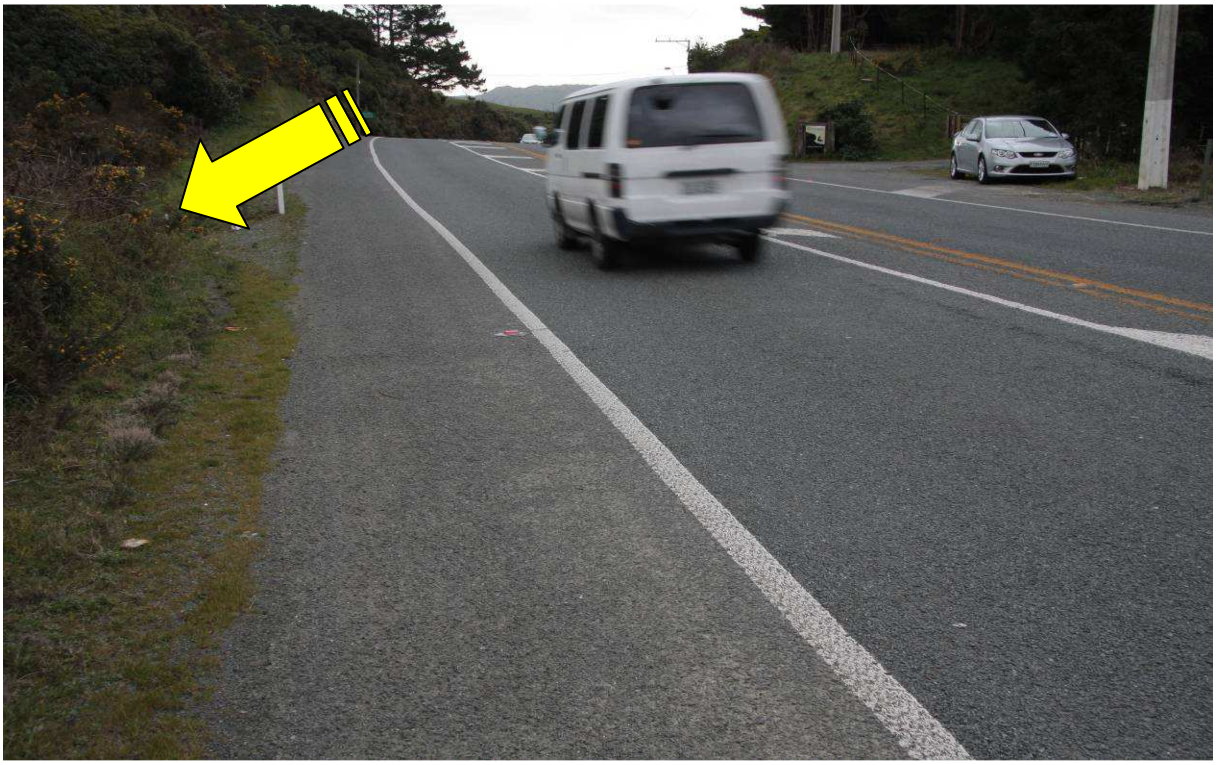


Figure A16 Roadside conditions at the small unnamed SH58 stream looking north away from the slight uphill bend in the road. The location of the stream above the road is shown (arrowed).

North Branch – Mahurangi River, SH1

Figure A17 North Branch, Mahurangi River, Site 2 above the road bridge

Figure A18 North Branch, Mahurangi River, Site 3 below the road bridge

Figure A19 Roadside conditions at the North Branch of the Mahurangi River looking north towards a slight uphill bend in the road.

Figure A20 Roadside conditions at the North Branch of the Mahurangi River looking south towards Warkworth.

South Branch – Mahurangi River, SH1

Figure A21 South Branch, Mahurangi River, Site 2 above the road bridge

Figure A22 North Branch, Mahurangi River, Site 3 below the road bridge