Temporal trends in macroinvertebrate metrics for some Waikato streams

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Abstract

A stratified approach based on sample size and level of confidence was applied to Spearman rank correlations to define both ecological and statistical significance of temporal trends in four invertebrate metrics at 49 Waikato, New Zealand, stream sites that had been monitored annually on 8-10 occasions. This approach suggested that ecological condition was stable over the monitoring period at around three-quarters of the sites, with the remainder of sites showing statistically and/or ecologically significant evidence of change. Relationships between landscape variables and the magnitude of change at sites showing "probable" or "clear" trends suggested that declines may have been greater in smaller, lower altitude streams with more developed catchments. Comparisons with other methods used to assess trends in macroinvertebrate community metrics in New Zealand indicated that the stratified approach provided a more liberal interpretation of possible trends. This approach may be useful for providing early warning of potential changes that may warrant management intervention where the reasons for decline are apparent.

Keywords: macroinvertebrate - MCI - EPT - Spearman correlation - landuse - New Zealand

Introduction

Ecological studies intended to determine the occurrence of temporal trends from annual monitoring programmes can be faced with a long wait before sufficient data are available to establish whether changes are significant based on conventional statistical approaches. Thomas (2005) estimated that up to 16 years' data were required to detect effects on fish in four North American streams receiving effluent inputs, depending on the magnitude of effect being assessed and the longevity of the target organisms. Moreover, ecological "noise" caused by natural spatial and temporal heterogeneity of instream environments may lead to inter-annual variability independent of anthropogenic stress that can obscure the existence of underlying trends where records of sufficient duration are not available to account for variations.

Various statistical methods have been applied to determine the significance of temporal trends, including several forms of regression analysis (e.g., Adams *et al.* 2002; Stark & Fowles 2006), coefficients of correlation or concordance (e.g., Scarsbrook *et al.* 2000a; Voelz *et al.* 2000), and time-series analyses (see Norris & Georges 1993). Non-parametric methods have been favoured for trend analyses of environmental data because of problems associated with non-normality, non-linearity, non-independence and missing values (Norris & Georges 1993). When doing correlation analyses, it is widely accepted that adjustments are needed to account for the increased possibility of making Type I errors where multiple comparisons are made, although the severity of required adjustments is a matter of debate (Death *et al.* 2000; Scarsbrook *et al.* 2000b). However, embedded in the array of statistically non-significant correlations from temporal studies that are still of short duration may be some relationships that are of ecological significance, and at the very least warrant further scrutiny from a management intervention perspective. Routine application of significance tests does not extract the maximum amount of information from environmental data and can result in misleading conclusions because a "practically" (i.e., ecologically) significant result may not necessarily be statistically significant (McBride 1993).

Invertebrates are often favoured for ecological monitoring because they are relatively easy to collect and identify, most are sessile with short life-histories and can respond rapidly to change, and they have a diversity of functional and effects traits that provide a variety of responses to changing environmental conditions (Hellawell 1978; Rosenberg & Resh 1993; Boothroyd & Stark 2000). Information on invertebrate community composition is condensed into "metrics" that can be related more directly to ecological condition than community data, and these metrics are widely used to report on changes in condition over time or identify differences in condition among sites. Because invertebrates respond to a range of interacting factors they are usually considered to provide a more holistic and cumulative understanding of ecosystem health than non-integrative measures such as water quality.

The regional authority Environment Waikato is responsible for ensuring that development of resources is conducted in a careful and sustainable fashion over a large (25,000 km^2) and ecologically diverse area of New Zealand's central North Island. As part of this responsibility, annual assessments of invertebrate community composition are conducted to document the condition of the region's streams and rivers (see Collier (2005) for a review). Environment Waikato's invertebrate monitoring has proceeded for at least 8 years with consistent sampling protocols at 49 sites. In statistical terms, 8 annual monitoring occasions represent a relatively small dataset for interpreting trends, but the desire for statistical rigour needs to be balanced with the requirement for information so that policies can be developed and actions taken to arrest any declining conditions. The aims of this paper are to (i) assess the presence of potential trends in the ecological condition of long-term monitoring sites on streams in the Waikato Region, and (ii) explore environmental factors linked to trends considered ecologically significant.

Materials and methods

Study area

The Waikato Region covers a diverse area of New Zealand's central North Island between latitudes 36° and 39°S (Figure 1). Mean annual air temperature in most of this region is in the range 12.5 to 15.0 °C, but declines to <8.0 °C on southern mountain tops (2,797 m asl.) (Kilpatrick 1999). Average annual rainfall is variable, being lower in the north (1,000 – 1,500 mm pa) than on the Coromandel or West Coast (up to 2,500 mm pa), and exceeds 5,000 mm pa on southern mountaintops. Landforms range from active volcanoes and upland plateaux (c. 600 m asl) in the south of the region, to steep hill-country (300 - 600 m asl) along the west coast, through the central parts of the region and along Coromandel Peninsula, and extensive lowland wetlands and plains towards the north. The region is very diverse geologically, with extensive areas of volcanic rock (rhyolites, andesites, basalts and dacites) around the southern and western volcanoes and along Coromandel Peninsula (McCraw 1971), and limestone common in western parts of the region. Pre-European vegetation cover was predominantly podocarp-hardwood forest in hill-country and western areas, with extensive areas of beech in the south. Fernland / scrubland also occurred over wide areas prior to European colonisation, and sub-alpine grassland and scrubland still occur at higher altitudes on the southern mountains. Most of the Waikato Region has now been developed for pastoral agriculture or pine forestry, with small areas of urban development including Hamilton, New Zealand's 7th most populous city. Extensive remnants of original forest persist in upland parts of the southern and central region, and along Coromandel Peninsula.

The 49 stream sites considered for assessment of environmental trends were scattered throughout the region (Figure 1) and represented a range of stream types. Most sites were characterised by wet-warm climates, lowland sources of flow and volcanic acidic or hard-sedimentary geology, as defined by Snelder & Biggs (2002). Fourteen of the sites occurred in the West Coast management zone, followed by Hauraki (11), Waipa (10), and the combined upper, middle and lower zones of the Waikato River catchment (10) (Figure 1). Sites were evenly distributed in terms of size with 11-13 sites on streams of order 1-2, 3, 4 and 5, and one larger site on an order 7 river. Segment altitudes averaged 99 m asl and mean channel gradient was 16 mkm^{-1}. However, there was considerable variability among sites (9-480 m asl and <1-112 mkm^{-1}). A wide range of landuse types was represented in upstream

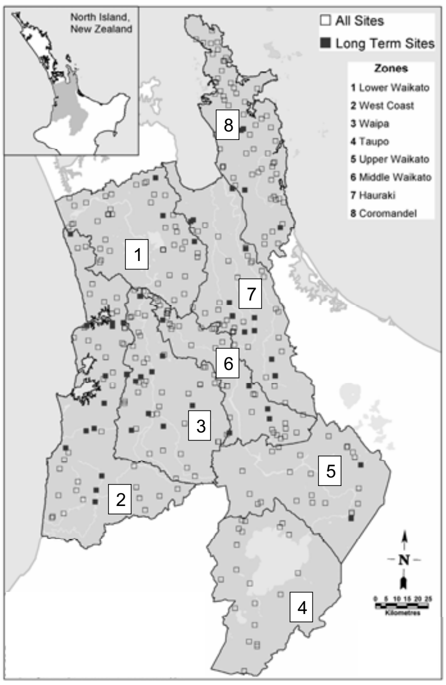

Figure 1. Location of Environment Waikato invertebrate monitoring sites in the Waikato Region of New Zealand.

catchments, although only one site had extensive urban development and 5 sites had greater than 10% exotic forestry cover upstream. Over all sites, pasture averaged 52% of upstream landuse and indigenous forest 35%. Three long-term reference sites were on small streams in native forested catchments (see Appendix I).

Sample collection and processing

Prior to 2002, field sampling for invertebrates involved collecting samples from habitat types in proportion to their occurrence using a 250 µm mesh triangular hand net. Following publication of Ministry for the Environment (MfE) protocols for sampling invertebrates in wadeable streams (Stark *et al.* 2001), a net mesh size of 500 µm was used and sampling in hard-bottomed streams was confined to riffles (see Collier & Kelly 2005 for further details). An assessment of the effect of changing protocols at six sites suggested average increases of 0.7 taxa for EPT* richness, 8 units for MCI values, and 10% for %EPT* using the new protocols (see Collier 2005). For the purpose of this analysis, only sites where comparable habitat types were likely to have been sampled in all years were retained for assessment of trends. Examination of time series plots indicated no marked shifts in metric scores associated with the change in protocols at these sites, and accordingly no adjustments were made to the data. The areas of habitat sampled equate to approximately 2 m^2 prior to 2002 compared to around 1-3 m^2 after this depending on whether a stream site was considered to be hard- or soft-bottomed (Collier & Kelly 2005).

Sample processing has been conducted on a fixed count basis since the start of the sampling programme. Initially, a minimum fixed-count of 100 plus a scan for rare taxa was conducted, but from 2002 the count was increased to at least 200 with a scan for rare taxa following MfE protocol P2 (Stark *et al.* 2001). For the purposes of trend analysis, the Species Diversity module in the computer program ECOSIM (Gotelli & Entsminger 2005) was used to generate 100-count data from 1000 iterations of the 200+ count data. All taxa counted were identified to the level of taxonomy listed in Collier & Kelly (2005), except prior to 2002 all chironomids were combined into a single taxon (see below). This list generally corresponds with the level of resolution used for calculation of the Macroinvertebrate Community Index (MCI) described in Stark *et al.* (2001), and it also identifies other taxa occurring in the region that have not been allocated MCI tolerance scores.

Data preparation and analysis

The 100-count data generated from ECOSIM were used to calculate the diversity metric Ephemeroptera, Plecoptera and Trichoptera (EPT*) richness, the compositional metric %EPT*, the tolerance metric MCI (Stark 1985), and a score based on axis 1 values of non-metric multidimensional scaling (NMDS) ordination of 17 standardised metrics (author's unpubl. data). Values for the NMDS condition score can range from 0 to 100 and are expressed as a percentage of the maximum values recorded at reference sites in native forested catchments, such that higher values are indicative of better ecological condition. EPT calculations excluded the caddisfly family Hydroptilidae (denoted by "*"), as recommended by Maxted *et al.* (2003). MCI, EPT* richness and

%EPT* are appropriate biological indicators for monitoring long-term trends because they are less susceptible to fluctuations in numbers of tolerant taxa, more robust to changes in sampling intensity, and less sensitive to changes in microscale habitat variables than many other metrics (Collier *et al.* 1998; Scarsbrook *et al.* 2000a). For MCI calculations, tolerance scores were the same as those listed in Stark *et al.* (2001) except the combined chironomid taxon was allocated a score of 5 representing the average for the sub-families Orthocladiinae, Tanypodinae and Podonominae, and for taxa listed in the subfamilies Chironominae and Diamesinae.

Temporal trends over 8-10 years were assessed using Spearman rank correlations between sampling year and metric values for each site. Spearman coefficient values were used to define four trend classes depending on the sample size available for analysis using different levels of certainty based on statistical significance and ecological significance assessed by professional judgement (Table 1). For all data, ecologically significant trends were considered at least "possible" where coefficient values exceeded 0.50, and "clear" trends were recognised where correlation coefficient probabilities were statistically significant when adjusted for multiple comparisons using the False Discovery Rate (FDR; Benjamini & Hochberg 1995; Garcia 2003; McBride 2005). Adjustments were made for comparisons among metrics within sites rather than for among sites because the aim was to determine whether trends were occurring at specific sites. Ecological significance was deemed at least "probable" depending on whether the coefficient was more or less than 0.7 where 8 or 9 years' data had been collected. Correlation thresholds around this value have been used by other workers to define high or environmentally significant relationships independent of statistical tests (e.g., Barbour *et al.* 1992; Maxted *et al.* 2000; Yuan & Norton 2003; Leathwick *et al.* 2005). Where 10 years' data had been collected, the significance of the Spearman coefficient was set at $\alpha = 0.05$ to define the cut-off between possible and probable trends because $r_s = 0.7$ becomes statistically significant for this α at a sample size of 9 (see Table 1).

Relationships between GIS-based environmental variables and the magnitude of change in invertebrate indicators between the initial and most recent monitoring dates expressed as a percentage of the initial value were also investigated using Spearman rank correlations for sites where declines were deemed probable/clear (see Table 1). Environmental factors used were stream

Table 1. Trend classes used to define the significance of relationships within sites for different sample sizes. FDR = False Discovery Rate which was used to adjust for multiple comparisons of metrics within sites.

	Trend Class			
n	**Stable**	**Possible**	**Probable**	**Clear**
8 or 9	$r_s \leq 0.50$	$0.50 > r_s < 0.70$	$0.70 \geq r_s < r_{s(\mathrm{FDR};\ \alpha=0.05)}$	$r_s > r_{s(\mathrm{FDR};\ \alpha=0.05)}$
10	$r_s \leq 0.50$	$0.50 > r_s < r_{s(\alpha=0.05)}$	$r_{s(\alpha=0.05)} \geq r_s < r_{s(\mathrm{FDR};\ \alpha=0.05)}$	$r_s > r_{s(\mathrm{FDR};\ \alpha=0.05)}$

order, altitude, gradient and upstream landcover variables derived from the database underlying the River Environment Classification (Snelder & Biggs 2002).

Results

The four measures of ecological condition displayed a high degree of variability within years but there were few extreme outliers (indicated by the whiskers and asterisks, respectively, in Figure 2). The interquartile ranges of metrics over the sampling period varied from 2-12 taxa for EPT* richness, 10-68% for %EPT*, 70 to 133 units for MCI, and 57-85% for the NMDS condition score (Figure 2). Overall, median annual values for each metric in each year were not strongly related (r_s typically <0.5), except for the NMDS condition score and EPT* richness where a rank correlation of 0.77 was obtained (n = 11). Median EPT* richness (6), %EPT* (31) and MCI (111) based on synthesised 100-invertebrate counts were similar to median values reported by Scarsbrook *et al.* (2000a) for long-term monitoring sites throughout New Zealand (8, 39% and 103, respectively).

Generally, most invertebrate-based measures of condition showed a similar pattern with respect to the percentage of

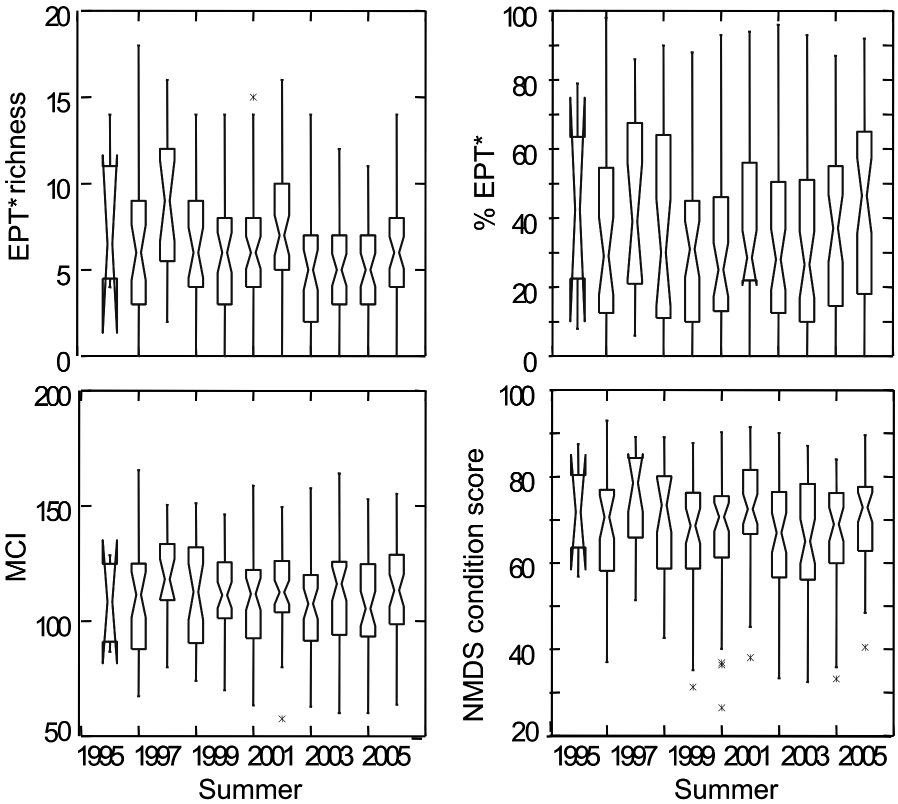

Figure 2. Box plots of four measures of ecological condition based on aquatic invertebrate communities sampled on 8-10 occasions over 11 years. Boxes indicate interquartile range with medians at the narrowest point; whiskers represent 1.5 x the interquartile range, and asterisks indicate extreme outliers. Note that EPT* excludes Hydroptilidae.

sites that appeared to be stable or display changes in condition (Table 2; a complete list of correlation coefficients is provided in Appendix I). Analysis of trend classes indicated that in general terms, ecological condition at around three-quarters of sites was "stable" (inconclusive evidence of change over the monitoring period), with the remainder showing evidence of potential change (average of all metrics = 28% of sites). Of the apparently changing sites, around 70% showed signs of net declines and 30% showed signs of net increases in overall ecological condition. Had only sites that were statistically significant following adjustment for multiple comparisons been considered, it would have been concluded that around 90% of sites were stable and up to 10% were changing (Table 2). The stratified approach combining possible, probable and clear trends provided a more liberal interpretation in terms of the percentage of sites considered to display trends compared to the approach of Scarsbrook *et al.* (2000a) who used an unadjusted α of 0.10 (Table 3). If only probable and clear changes are considered, the percentage of sites considered to display trends was similar using the two approaches.

The magnitude of change between minimum and maximum values recorded over the monitoring period for sites displaying possible-clear trends averaged 6.8 taxa for EPT* richness, 43% for %EPT*, 31 units for the MCI and 24% for the NMDS condition score (Table 3). The absolute magnitude of change in metrics at sites suggesting temporal trends was similar to the magnitude of change recorded at three reference sites that had been monitored over the long-term. However, in percentage terms (i.e., (maximum – minimum) / maximum * 100), the magnitude of change for EPT* taxa richness was considerably less at reference sites, whereas the other metrics were

Table 2. Percentage of sites (n = 49) classified according to different trend classes for four invertebrate-based measures of ecological condition. EPT* = Ephemeroptera, Plecoptera and Trichoptera (excluding Hydroptilidae); MCI = Macroinvertebrate Community Index.

	EPT* richness	Percent EPT*	MCI	NMDS condition score
Stable[1]	65	78	76	71
Possible[1] decline	16	4	8	10
Probable[1] decline	4	2	4	6
Clear[1] decline	10	4	6	4
Overall[2] decline	30	10	18	20
Possible[1] improvement	4	2	6	6
Probable[1] improvement	0	6	2	0
Clear[1] improvement	0	4	0	2
Overall[2] increase	4	12	8	8
Total trends				
α = 0.05; FDR corrected	10	8	6	6
α = 0.10[3]	18	20	12	20
Stratified approach[2]	34	22	26	28

[1], see Table 1
[2], sum of possible, probable and clear
[3], Scarsbrook *et al.* (2000a)

similar or greater at reference sites compared to all sites showing trends (Table 3). Apparent trends for some metrics were detected at two of the three reference sites monitored over the long term. At one site %EPT* and MCI displayed increasing trends, whereas the other site also had increasing MCI but declining EPT* taxa richness and NMDS condition scores. These trends appeared to reflect in part the effects of flooding prior to sampling period in recent years. Relationships between landscape variables and the magnitude of change in invertebrate indicators displaying probable-clear declines indicated r_s values of $\geq$ 0.7 for MCI versus stream order (r_s = -0.79), and % upstream catchment area in urban (r_s = -0.71) or pastoral (r_s = -0.80) landuse, and between the NMDS condition score and stream order (r_s = -0.74) and elevation (r_s = -0.70) (n = 5 for all coefficients). Insufficient probable/clear declines were detected for %EPT* (n = 2) to enable analysis.

Discussion

Scarsbrook *et al.* (2000a) investigated national patterns in invertebrate community metrics at 66 sites sampled annually from 1989 to 1996, and reported variable responses depending on the metrics examined. They noted an increase in the percentage of baseline sites with MCI scores indicative of clean water that coincided with reported general improvements in water quality. Some concordance was also noted between trends in metric values identified using the stratified analytical approach described in the present study and water quality trends in the Waikato reported by Vant & Smith (2004) who associated increasing concentrations of total nitrogen and phosphorus with changes in landuse, particularly pastoral intensification and land drainage. The significance of land use was supported in our study by the association between percentages of upstream catchment area in pasture or urban development and the magnitude of change in MCI values at sites experiencing

Table 3. Magnitude of change over time between minimum and maximum values recorded for metrics at all sites where possible, probable or clear trends were detected (Trends), and for reference sites only.

	Mean value	Range	Mean percentage change	Number of sites
EPT* taxa richness				
Trends	6.8	2-12	70.8	17
Reference sites	10.0	9-11	18.2	3
%EPT*				
Trends	43.2	2-79	82.9	11
Reference sites	53.3	15-83	81.9	3
MCI				
Trends	31.0	17-46	25.6	13
Reference sites	32.8	17-55	68.4	3
NMDS condition score				
Trends	23.9	9-32	29.9	14
Reference sites	19.6	9-25	65.5	3

probable / clear declines over the monitoring period. Declines in ecological condition also appeared to be greater for some metrics in smaller streams in lower elevation settings.

The concept of ecological significance when applied to trend analysis needs to consider not only the strength of a relationship but also the magnitude of change encountered over the monitoring period. For example, one site displaying a probable / clear decline in EPT metrics changed by only 2 taxa and 2% over the monitoring period, but these were the maximum values recorded at that site. Although this was an isolated example in the dataset analysed, it emphasises the importance of defining the magnitude of change in metric values that can be considered ecologically significant. Nevertheless, the average and range of magnitude of changes at all sites displaying temporal trends were similar to those recorded at unimpacted reference sites, suggesting that dynamic changes also occur in the absence of direct anthropogenic pressure, particularly following large floods. The magnitude of change in EPT* taxa richness was much less in percentage terms at reference sites compared to all sites showing apparent trends, suggesting that this metric may be particularly useful for discriminating temporal change from background variability.

Scarsbrook *et al.* (2000a) used unadjusted Spearman rank correlations and $\alpha = 0.10$ to discriminate the occurrence of trends in invertebrate metrics measured annually over seven years, a similar timescale to that of the present study. Their approach provoked discussion about the importance of adjusting for multiple comparisons when using correlation analyses to infer trends versus the need for a balanced approach to include practical ecological significance (see Death *et al.* 2000; Scarsbrook *et al.* 2000b). The stratified approach to interpreting ecological significance used in the present study tripled the percentage of sites considered to be potentially changing over time compared to a statistical approach adjusting for Type I error rates among metric comparisons, and provided a more liberal interpretation of trends to that provided by the approach of Scarsbrook *et al.* (2000a) (see Table 2).

When applying these results in a resource management context, it is perhaps more pertinent to consider the relative weights given to apparent trends that are considered possible, probable or clear in terms of scrutiny or intervention, rather than focussing purely on statistical significance. Evidence of possible or probable changes may not provide sufficient basis for implementing large-scale management or policy changes, but the potential early warnings they provide can highlight the need for closer scrutiny at specific sites. Such warnings may present an opportunity to ameliorate impacts before they worsen to the extent of signalling a statistically significant and potentially irreversible decline where the agents of change are apparent. Similarly, early indications of the trajectories of change following rehabilitation initiatives may empower community groups or suggest the need for assessing other factors constraining ecological recovery. As time progresses and the sample sizes of temporal studies increase, new challenges will be created using stratified approaches for trend analysis in order to deal with statistically significant relationships that may not be ecologically significant.

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References

Adams, S.M., Hill, W.R., Peterson M.J., Ryon, M.G., Smith, J.G. & Stewart, A.J. (2002). Assessing recovery in a stream ecosystem: applying multiple chemical and biological endpoints. *Ecological Applications* 12: 1510-1527.

Barbour, M.T., Plafkin, J.L., Bradley, B.P., Graves, C.G. & Wissmar, R.W. (1992). Evaluation of EPA's rapid bioassessment benthic metrics: metric redundancy and variability among reference stream sites. *Environmental Toxicology and Chemistry* 11: 437-449.

Benjamini, Y. & Hochberg, Y. (1995). Controlling the false discovery rate: a practical and powerful approach to multiple testing. *Journal of the Royal Statistical Society Series B* 57: 289-300.

Boothroyd, I. & Stark, J. (2000). Use of invertebrates in monitoring. In *New Zealand stream invertebrates: ecology and implications for management* (eds. K.J. Collier & M.J. Winterbourn), pp. 344-373, New Zealand Limnological Society, Christchurch.

Collier, K.J. (2005). Review of Environment Waikato's Regional Ecological Monitoring of Streams (REMS) programme: Past practices and future directions. Environment Waikato technical report TR05/48, Environment Waikato, Hamilton. (http://www.ew.govt.nz/publications/technicalreports/tr0548.htm).

Collier, K. & Kelly, J. (2005). Regional guidelines for ecological assessments of freshwater environments. Macroinvertebrate sampling in wadeable streams. Environment Waikato technical report TR2005/02, Environment Waikato, Hamilton. (http://www.ew.govt.nz/publications/technicalreports/tr0502.htm).

Collier, K.J., Wilcock, R.J. & Meredith, A.S. (1998). Influence of substrate type and physicochemical conditions on macroinvertebrate faunas and biotic metrics of some lowland Waikato streams. *New Zealand Journal of Marine and Freshwater Research* 32: 1-19.

Death, R.G., Henderson, I.M. & Joy, M.K. (2000). Comment: Problems with Type I errors in water quality analysis. *New Zealand Journal of Marine and Freshwater Research* 34: 745-746.

Garcia, L.V. (2003). Controlling the false discovery rate in ecological research. *Trends in Ecology and Evolution* 18: 553-554.

Gotelli, N.J. & Entsminger, G.L. (2005). EcoSim: Null models software for ecology. Version 7. Acquired Intelligence Inc. & Kesey-Bear. Jericho, VT 05465. (http://garyentsminger.com/ecosim/index.htm).

Hellawell, J.M. (1978). *Biological surveillance of rivers.* Water Research Centre, Medmenham & Stevenage, U.K.

Kilpatrick, R. (1999). *Bateman contemporary atlas of New Zealand.* David Bateman, Auckland.

Leathwick, J.R., Rowe, D., Richardson, J., Elith, J. & Hastie, T. (2005). Using multivariate adaptive regression splines to predict the distributions of New Zealand's freshwater diadromous fish. *Freshwater Biology* 50: 2034-2052.

Maxted, J.R., Barbour, M.T., Gerritsen, J., Poretti, V., Primrose, N., Silvia, A., Penrose, D. & Renfrow, R. (2000). Assessment framework for mid-Atlantic coastal plain streams using benthic macroinvertebrates. *Journal of the North American Benthological Society* 19: 128-144.

Maxted, J.R., Evans, B.F. & Scarsbrook, M.R. (2003). Development of standard protocols for macroinvertebrate assessment of soft-bottomed streams in New Zealand. *New Zealand Journal of Marine and Freshwater Research* 37: 793-807.

McBride, G.B. (1993). What do significance tests really tell us about the environment? *Environmental Management* 17: 423-432.

McBride, G.B. (2005). *Using statistical methods for water quality management. Issues, problems and solutions.* John Wiley & Sons, Hoboken, New Jersey.

McCraw, J.D. (1971). The geological history of the Waikato River basin. In *The waters of the Waikato* (ed. C. Duncan), pp. 11-23, The University of Waikato, Hamilton.

Norris, R.H. & Georges, A. (1993). Analysis and interpretation of benthic macroinvertebrate surveys. In *Freshwater biomonitoring and benthic macroinvertebrates* (eds. D.M. Rosenberg & V.H. Resh), pp. 234-286, Chapman & Hall, New York.

Rosenberg, D.M. & Resh, V.H. (1993). Introduction to freshwater biomonitoring and benthic macroinvertebrates. In *Freshwater biomonitoring and benthic invertebrates* (eds. D.M. Rosenberg & V.H. Resh), pp. 1-9, Chapman & Hall, New York.

Scarsbrook, M.R., Boothroyd, I.K.G. & Quinn, J.M. (2000a). New Zealand's National River Water Quality Network: long-term trends in macroinvertebrate communities. *New Zealand Journal of Marine and Freshwater Research* 34: 289-302.

Scarsbrook, M.R., Quinn, J.M. & Boothroyd, I.K.G. (2000b). Reply: A balanced approach to interpretation of data. *New Zealand Journal of Marine and Freshwater Research* 34: 746-749.

Snelder, T.H. & Biggs, B.J.F. (2002). Multiscale river environment classification for water resources management. *Journal of the American Water Resources Association* 38: 1225-1239.

Stark, J.D. (1985). *A macroinvertebrate community index of water quality for stony streams.* Water & soil miscellaneous publication 87, Ministry of Works and Development, Wellington.

Stark, J.D., Boothroyd, I.K.G., Harding, J.S., Maxted, J.R. & Scarsbrook, M.R. (2001). *Protocols for sampling macroinvertebrates in wadeable streams.* New Zealand Macroinvertebrate Working Group report no. 1. Ministry for the Environment, Wellington.

Stark, J.D. & Fowles, C.R. (2006). *An approach to the evaluation of temporal trends in Taranaki State of the Environment macroinvertebrate data.* Report no. 1135, Cawthron Institute, Nelson.

Thomas, J.F. (2005). How long should a long-term river study be? *Journal of Freshwater Ecology* 20: 367-379.

Vant, B. & Smith, P. (2004). *Trends in river water quality in the Waikato Region, 1987-2002.* Environment Waikato technical report 2004/05. Environment Waikato, Hamilton.

Voelz, N.J., Sieh, S-H. & Ward, J.V. (2000). Long-term monitoring of benthic macroinvertebrate community structure: a perspective from a Colorado River. *Aquatic Ecology* 34: 261-278.

Yuan, L.L. & Norton, S.B. (2003). Comparing responses of macroinvertebrate metrics to increasing stress. *Journal of the North American Benthological Society* 22: 308-322.

Appendix I. Spearman correlation coefficients between monitoring years (*n*) and four metric scores derived from 100-count invertebrate data. *, possible trend; **, probable trend; ***, clear trend (see Table 1). Locations of sites can be found in Collier (2005).

Site	*n*	EPT* richness	%EPT*	MCI	NMDS condition score
1055-2	9	-0.20	-0.13	-0.55*	-0.13
1055-3	10	-0.47	-0.17	-0.02	-0.65**
1158-7	9	-0.09	0.32	0.20	-0.07
1172-6	10	-0.43	-0.66**	-0.02	-0.72**
1174-10	9	-0.71**	-0.25	-0.75**	-0.67*
1236-4	8			-0.25	-0.41
124-4	8	-0.48	-0.14	0.00	-0.64*
1247-3	9	-0.24	-0.22	-0.07	-0.40
1249-15	9	-0.25	-0.08	-0.53*	0.17
1249-32	8	-0.79***	-0.76***	-0.74***	-0.45
1252-3	10	0.04	0.12	-0.35	0.12
1253-8	9	-0.53*	-0.60*	-0.02	-0.55*
1253-9	8	0.07	-0.43	0.16	-0.26
125-4‡	8	-0.28	0.55*	0.57*	-0.23
1257-4	8	-0.17	0.86***	-0.52*	0.29
1284-1	9	0.51*	0.70**	0.60*	0.48
1293-8	9	-0.37	-0.37	-0.45	-0.26
1414-1‡	9	0.10	-0.21	-0.15	0.03
195-1	8	0.40	0.29	0.17	0.43
220-1	9	-0.52*	0.03	0.02	-0.44
23_2	9	0.35	0.17	-0.08	0.22
240-5	8	-0.79***	-0.19	-0.88***	-0.76***
256-2	9	-0.75***	-0.38	-0.44	-0.80***
36-1	8	-0.11	-0.26	-0.43	-0.38
365-1	9	0.02	0.20	-0.50	-0.03
4-2	8	-0.34	0.50*	-0.77**	-0.02
407-1	8	0.28	0.76**	-0.23	0.69*
413-2	10	-0.58*	-0.24	0.06	-0.64**
428-3	9	-0.54*	-0.63*	-0.57*	-0.63*
428-5	8	-0.91***	0.07	0.12	-0.01
429-3	9	-0.27	-0.27	-0.37	0.05
433-2	9	-0.62*	0.03	0.13	0.17
453-8	9	0.19	0.23	0.26	0.27
476-1	9	-0.54*	0.30	0.40	0.10
477-14‡	9	-0.58*	0.44	0.52*	-0.54*
477-5	9	0.48	0.37	0.22	0.38
481-11	8	-0.33	-0.44	-0.42	-0.44
493-1	9	-0.29	-0.49	-0.28	-0.48
495-1	8	-0.51*	-0.14	0.17	-0.24
514-1	9	-0.29	-0.10	-0.17	-0.35
531-4	9	0.01	0.34	0.47	0.13
539-1	9	-0.25	0.80***	0.34	0.75***
556-9	9	-0.74**	0.09	-0.37	-0.49
619-20	9	-0.04	0.14	0.07	-0.22
749-10	9	-0.73**	-0.73***	-0.73***	-0.33
753-7	9	-0.13	0.27	0.33	0.23
786-2	10	0.47	0.69**	0.48	0.56*
786-22	8	0.51*	0.21	0.71**	0.67*
976-2	8	0.21	0.01	-0.37	0.31

‡, reference site