New Zealand Journal of Marine and Freshwater Research, 2004, Vol. 38: 347–366
0028–8330/04/3802–0347

Water quality in low-elevation streams and rivers of New Zealand: recent state and trends in contrasting land-cover classes

SCOTT T. LARNED[1]
MIKE R. SCARSBROOK[2]
TON H. SNELDER[1]
NED J. NORTON[1]
BARRY J. F. BIGGS[1]

[1]National Institute of Water and Atmospheric Research Limited
P.O. Box 8602
Christchurch, New Zealand
email: s.larned@niwa.co.nz

[2]National Institute of Water and Atmospheric Research Limited
P.O. Box 11 115
Hamilton, New Zealand

Abstract River water quality in New Zealand is at great risk of impairment in low elevation catchments because of pervasive land-use changes, yet there has been no nationwide assessment of the state of these rivers. Data from the surface-water monitoring programmes of 15 regional councils and unitary authorities, and the National River Water Quality Network were used to assess the recent state (1998–2002) and trends (1996–2002) in water quality in low-elevation rivers across New Zealand. Assessments were made at the national level, and within four land-cover classes (native forest, plantation forest, pastoral, and urban). Finer-scaled assessments were made by subdividing the large number of pastoral sites into six climate classes, and seven stream orders. At the national level, median concentrations of the faecal indicator bacterium *Escherichia coli*, and dissolved inorganic nitrogen and dissolved reactive phosphorus exceeded guidelines recommended for the protection of aquatic ecosystems and human health. Water quality state varied widely within land-cover classes: *E. coli* and dissolved nitrogen and phosphorus concentrations in the pastoral and urban classes were 2–7 times higher than in the native and plantation forest classes, and median water clarity in the pastoral and urban classes was 40–70% lower than in the native and plantation forest classes. Water quality state in the pastoral class was not statistically different from that of the urban class, and water quality state in the plantation forest class was not statistically different from that of the native forest class. Significant trends in low-elevation rivers were limited to four parameters: flow (trending down in all instances), and temperature, clarity, and conductivity (trending up in all instances). The trends in flow, temperature, and clarity were apparent at the national scale, and within the pastoral class. The magnitudes of these trends were very low, corresponding to changes of ≤0.5%/year in parameter medians.

Keywords clarity; *Escherichia coli*; guidelines; land cover; land use; low-elevation; New Zealand; nutrients; state; streams and rivers; trends; water quality

INTRODUCTION

Agriculture, urban land use, and plantation forestry have been linked to reduced water quality and ecological degradation in New Zealand rivers (McColl & Hughes 1981; Cooper et al. 1987; Harding & Winterbourn 1995; Duggan et al. 2002; Quinn & Stroud 2002). The risk to water quality posed by these activities is particularly great in low-elevation catchments, where urban boundaries have expanded in recent years, acreage in exotic conifer plantations has increased, and agricultural practices have shifted from low-intensity grazing to intensive cropping and dairying (Taylor & Smith 1997; Nagashima et al. 2002; MAF 2003). As of 1997, 70% of the low-elevation land in New Zealand was developed for agriculture, plantation forestry, and

M03047; Online publication date 8 June 2004
Received 13 August 2003; accepted 2 February 2004

urban uses (Terralink International Ltd 2001: New Zealand Land Cover Database. www.terralink.co.nz/tech/data/lcdb/lcdb.htm). The remaining undeveloped area is primarily native forest and tussock.

Land-use pressures on low-elevation rivers exist across New Zealand, yet there has been no national-scale assessment of the links between low-elevation land cover and water quality. Such an assessment would entail examining spatial patterns in water quality, and determining how these patterns relate to land-cover patterns, whether water quality is improving or getting worse over time, and whether such trends are occurring in catchments dominated by particular land uses.

Previous assessments of land-cover effects on stream and river water quality in low-elevation catchments were made at relatively fine spatial-scales (4–18 streams, 0.2–400 km^2 catchment areas). These studies represent the current state of knowledge of the links between water quality and land cover in New Zealand. McColl et al. (1977) reported higher nitrate, dissolved reactive phosphorus (DRP), and total phosphorus (TP) concentrations in a stream in a pastoral catchment compared with streams in plantation forest and native forest catchments in the Taita basin near Wellington. Quinn et al. (1997) and Quinn & Stroud (2002) reported higher concentrations of dissolved inorganic nitrogen (DIN) and total nitrogen (TN), and lower clarity in streams in pastoral and plantation forest catchments than in native forest streams in western Waikato. Duggan et al. (2002) reported higher DIN, DRP, and TP concentrations in pastoral streams than in scrub and native forest streams near the South Island west coast. Townsend et al. (1997) reported higher TP in native forest streams than in plantation forest streams in the Taieri Stream catchment of Otago. Data in the study by Hamill & McBride (2003; table 3) indicate that DIN and DRP concentrations in 15 low-elevation pastoral streams and one urban stream were higher than in two undeveloped streams in Southland. Other studies of urban stream water quality in New Zealand have reported high concentrations of DIN, TN, TP, dissolved metals, and coliform bacteria in Auckland, Dunedin, Hamilton, and Rotorua (Hoare 1984; Williamson 1986; Hall et al. 2001; Mosley & Peake 2001).

Two coarse-scaled assessments of stream water quality have also been conducted in New Zealand, the "100 Rivers" study (Close & Davies-Colley 1990) and the National River Water Quality Network (NRWQN) study (Smith & Maasdam 1994). The intent of both studies was to characterise rivers across New Zealand, not to examine land-use effects. The 34 low-elevation sites in the 100 Rivers study, and the 23 low-elevation sites in the NRWQN study were predominately in pastoral catchments (82% in the former, 87% in the latter). Because of low spatial resolution and the predominance of a single land-cover class, these studies provide little basis for assessing land-use effects, or for extrapolating to other low-elevation rivers. The fine-scaled studies in the preceding summary cannot be used to make generalisations about larger areas. The trade-off between resolution and generality associated with coarse and fine-scaled studies could be reduced if assessments included a large number of monitoring sites, and these sites represented a wide range of land-cover classes.

In this study we examined the recent (1996–2002) state and temporal trends in water quality in low-elevation rivers across New Zealand. We considered low-elevation rivers to be those draining catchment areas where ≥50% of the mean-annual precipitation volume occurs at elevations ≤400 m a.s.l. (Snelder & Biggs 2002). The 400-m elevation criterion was developed for use in the River Environment Classification (REC) to group New Zealand rivers that receive run-off primarily from rain rather than snowmelt, and have flow regimes that closely track precipitation and evapotranspiration regimes (Snelder & Biggs 2002). The REC is described in detail below. We used water quality data provided by 15 regional councils and local authorities, plus NRWQN data collected by the National Institute of Water and Atmospheric Research Limited (NIWA), to develop a data set of sites in four land-cover classes: pastoral, urban, plantation forest, and natural (native forest and scrub). The study had three specific objectives: (1) establish and rank the water quality state of each land-cover class; (2) compare water quality state with guidelines recommended for protecting ecological integrity and human health in low-elevation rivers; and (3) identify temporal trends in water quality within each class.

METHODS

Sources of data and data-set reduction

We requested surface water quality data from regional councils, local authorities, and NIWA for all sites that had been monitored for at least 2 years at a frequency of 4 times/year or higher. We had no control over the quality of data provided to us, but

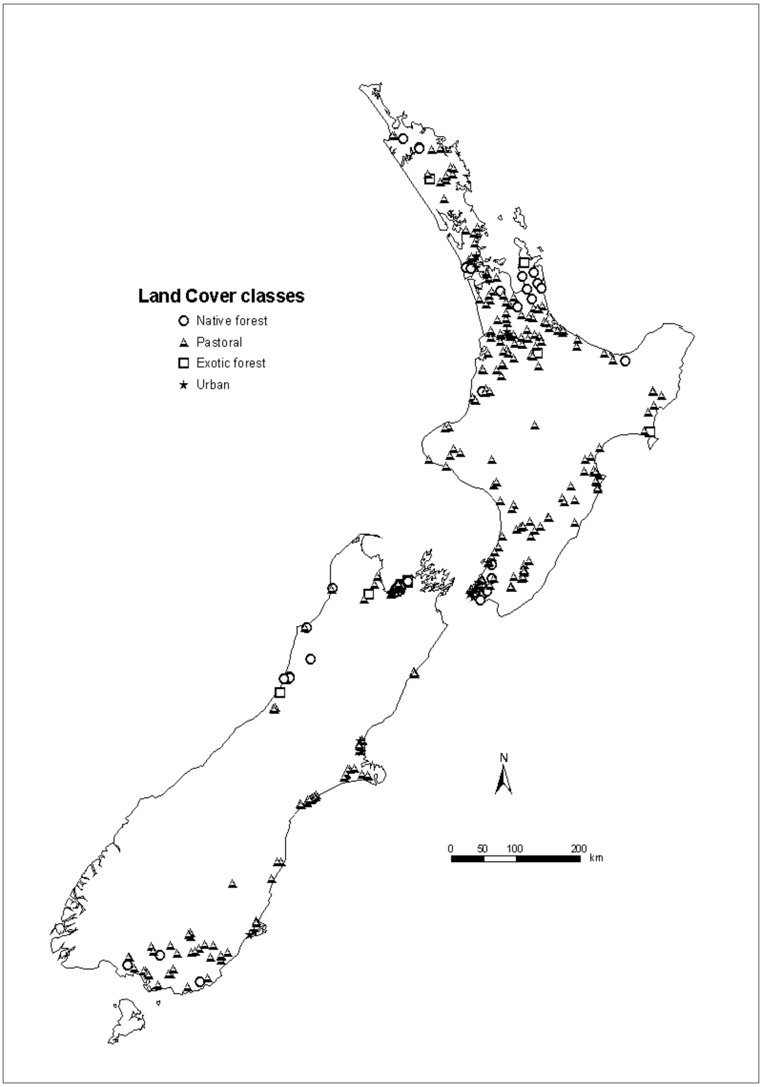


Fig. 1 Locations of 338 low-elevation river sites used in the water quality state assessments. Trend analyses were based on data from 229 of the sites. Land-cover classes from the River Environment Classification.

most council-run or contracted analytical laboratories have quality assurance programmes in place, and are accredited by International Accreditation New Zealand (the national authority for testing laboratories). The NRWQN quality assurance procedures are listed in Smith & McBride (1990) and Smith et al. (1996).

The original data set comprised over 1000 river sites, and over 10 000 sampling events. The large amount of data allowed us to eliminate sites for which data were not current, or sampling was not conducted year-round. The trend data set was limited to sites that were sampled at monthly to bimonthly intervals from January 1996 to December 2002. The state data set was limited to sites that were sampled at monthly to quarterly intervals for at least 2 years, and for which there were at least 10 sampling dates, between December 1998 and December 2002.

The resulting data sets were inspected and some data were deleted or modified as follows. Measurements that were clearly erroneous or estimated were deleted. Data collected at high frequencies with automated loggers were converted to daily averages. Nutrient concentrations below detection limits were replaced with values equal to half the detection limit. The final data sets for the state and trend analyses consisted of 338 and 229 sites, respectively (Fig. 1). Not all monitoring programmes included the entire suite of water quality parameters considered in this report. Further, one or more parameter values were missing on some sampling occasions at some sites. Consequently, the total number of data points varies among parameters.

Site classification

Monitoring sites were assigned to land-cover classes and climate classes using the REC. The REC is a hierarchical system for classifying river segments using characteristics of catchments that strongly affect physical and biological conditions in rivers (Snelder & Biggs 2002). The hierarchical levels, from largest to smallest spatial scale, are climate, source-of-flow, geology, land cover, network position, and valley landform. Each level is composed of 4–8 categories that collectively delineate all of New Zealand's streams and rivers. The REC land-cover categories delineate river segments that are nested within the geology level, and geology categories are nested within the source-of-flow level, and so on. However, this approach subdivided our data set into 45 combinations, each represented by a small number of sites. As the focus of the study was land-cover patterns, and we wished to maximise replication, most of our analyses used sites grouped by land-cover category and pooled across higher levels (Table 1). Only the pastoral class had enough monitoring sites (259 in the state analysis, 166 in the trend analysis) to make robust assessments at finer spatial scales. Separate assessments were made within the pastoral class in each of the six REC climate classes (Table 1, Fig. 2).

Land-cover classification in the REC is based on proportions of various land-cover categories in the catchment of each segment of a river network. The source data for REC land-cover classification is the New Zealand Land Cover Database (Terralink International Limited). This database is based on false-colour satellite imagery and has a minimum mapping unit of 1 ha. Stream catchments are often composed of two or more land-cover categories, so a set of rules was established to assign river segments to single classes in a consistent way. Rules for assigning river segments (and the corresponding monitoring sites) to climate and land-cover classes are given in Table 1 and are discussed further by Snelder & Biggs (2002).

We also addressed the possibility that water quality is affected by stream size, and that data collected from streams of a limited size range resulted in biased assessments. The Strahler stream order was determined for each site in the pastoral land-cover class, and water quality state and trends were assessed separately for each stream order. Only the pastoral class included sufficient monitoring sites for this assessment; there were 8–79 sites in each of seven stream orders.

Before characterising water quality state across New Zealand, we checked the representativeness of the data set. If sites from a land-cover class comprise a larger proportion of the data set than that class comprises in terms of stream length in New Zealand, the data may give a biased view of nationwide water quality. We used the REC to compute the total length of perennial low-elevation streams in each land-cover class, then compared the proportions of stream length in each class with the proportions of monitoring sites in the same class. Table 2 summarises these comparisons. The proportion of pastoral sites in the data set was very similar to the proportion of low-elevation river length in the

Table 1 Summary of classes, notation, and rules for assigning monitoring sites to land-cover and climate categories of the River Environment Classification.

Classification level	Classes	Notation	Assignment rules
Land cover	Native forest	F	Spatially dominant category, unless A exceeds 25% of
	Exotic forest	E	catchment area, in which instance, class is A, or U
	Pastoral	P	exceeds 15% of catchment area, in which instance,
	Urban	U	class is U
Climate	Warm extremely wet	WX	Warm: mean annual temperature ≥12°C
	Warm wet	WW	Cool: mean annual temperature ≤12°C
	Warm dry	WD	Extremely wet: mean annual effective precipitation
	Cool extremely wet	CX	≥150 cm
	Cool Wet	CW	Wet: 50 >mean annual effective precipitation <150 cm
	Cool Dry	CD	Dry: mean annual effective precipitation <50 cm

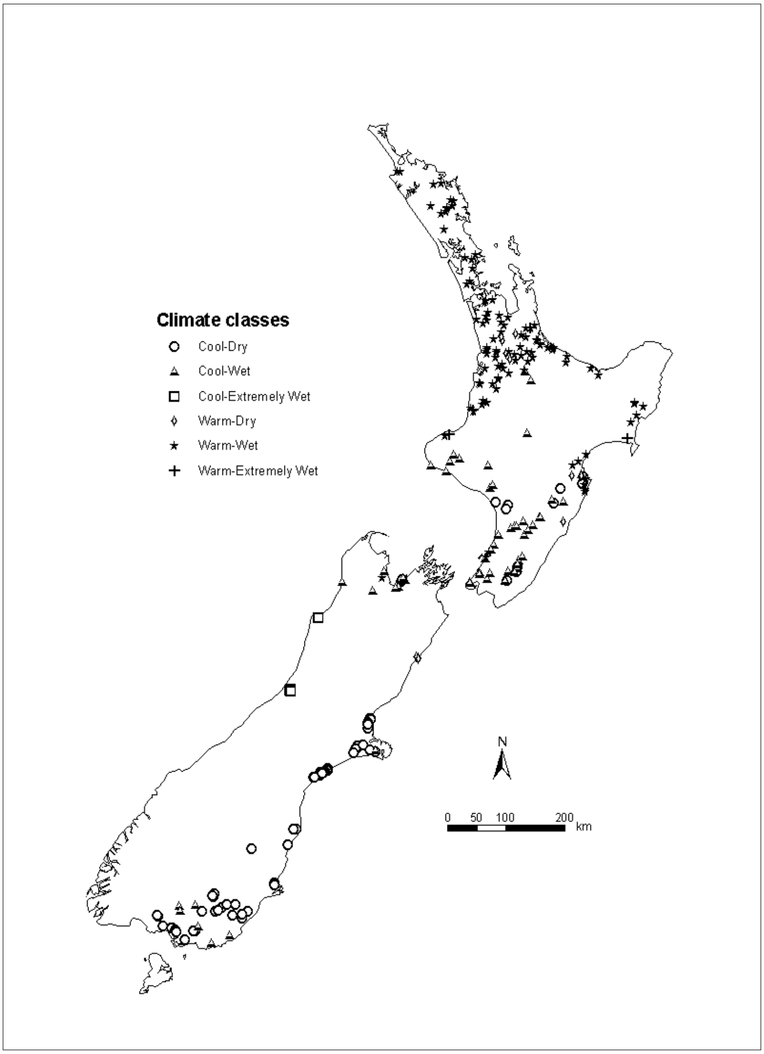


Fig. 2 Distribution of low-elevation pastoral river sites within River Environment Classification climate classes.

pastoral class. Exotic and natural forest sites were slightly under-represented in the data set, relative to their river lengths, and urban sites were over-represented. These comparisons suggest that national-scale estimates of water-quality state will be biased towards conditions in urban areas. The effect of this bias should be small, because the great majority of monitoring sites and river kilometres are in the pastoral class.

Data analyses

Eight water quality parameters were selected for the state analysis, based on their utility as indicators of environmental degradation, and on the number of monitoring programmes that include them: oxidised nitrogen (NO_X), ammonium (NH_4), DRP, TN, TP, *Escherichia coli*, electrical conductivity at 25°C, and water clarity measured by the black disk method (Davies-Colley 1988). Only the pastoral land-cover class had sufficient TN and TP data for analysis. Trend analyses were carried out using NO_X, NH_4, DRP, *E. coli*, conductivity, and clarity, plus temperature and flow. Trends in the latter two parameters were examined because they often reflect large-scale climatic variation. Without consideration of climate variation, some trends could be incorrectly attributed to changes in land cover or land-use management. Dissolved oxygen and pH data were available, but were not analysed because of systematic variation with temperature and time of day.

For state analyses, monitoring sites were used as replicates within land-cover classes, and data from individual sampling dates were treated as subsamples, i.e., parameter values for each site were averaged across dates, and the averages were used as single points in the analyses. For trend analyses, site-specific trends in parameter values over time were used as replicates.

For each water quality parameter, median values of the sites in each land cover and climate class were compared with guideline values recommended for New Zealand rivers in the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC & ARMCANZ 2000) and the joint Ministry for the Environment/Department of Health microbiological guidelines (MfE & MoH 2002). An additional guideline for conductivity was derived from an empirical periphyton-conductivity relationship for New Zealand rivers (Biggs 1988). The guidelines are summarised in Table 3. It should be noted that two types of guidelines are included, "reference condition" guidelines derived from data collected at sites considered to be unmodified or slightly modified, and "effects-based" guidelines derived from relationships between water quality parameters and ecological or human health effects. The nutrient and clarity guidelines in Table 3 are based on reference conditions at NRWQN baseline sites below 150 m a.s.l., and the *E. coli* and conductivity guidelines are effects-based. *E. coli* is the preferred microbiological indicator of faecal contamination in New Zealand. Current guidelines are based on maximum *E. coli* concentrations in single samples (MfE & MoH 2003). Our assessments are based on long-term averages, so we have adopted a previous *E. coli* guideline that refers to medians (MfE & MoH 2002). Both of the *E. coli* guidelines are used to indicate unacceptable public health risk of microbiological contamination. The conductivity guideline is the level corresponding to a periphyton biomass of 35 g ash-free dry weight m^{-2}, based on data from 103 sites across New Zealand (Biggs 1988). The 35 g m^{-2} limit was recommended by the Ministry for the Environment for protecting trout habitat (Biggs 2000).

Statistical comparisons were made among land-cover classes, and for the pastoral class, among stream orders and climate classes. Because of non-normal distributions of some water quality parameters and varying numbers of sites per class, comparisons were made using non-parametric Kruskal-Wallis tests. We considered all differences with P values ≤0.05 to be statistically significant. In instances of significant Kruskal-Wallis tests, we made pair-wise comparisons of classes using non-parametric Tukey tests, adjusted for unequal sample sizes (Zar 1984).

We compared the average water quality state in New Zealand rivers from all elevations (based on NRWQN data) to that of New Zealand's low-elevation rivers (based on the data set developed in this study). We computed medians for parameters measured at all 77 NRWQN sites during the same period used for the present study, January 1999–November 2002 (NRWQN data from R. Wilcock, NIWA).

Assessments of temporal trends in water quality parameters from 1996–2002 were made using the Seasonal Kendall slope estimator (SKSE), which is the median of all within-month slopes for the study period. The SKSE accounts for seasonal variations in parameter values and gives robust trend estimates from non-normal data sets with some missing values (Helsel & Hirsch 1992; Smith et al. 1996). To make

Table 2 Proportions of water-quality monitoring sites in each land-cover class, compared with the proportions of low-elevation (<400 m a.s.l.) land area and low-elevation river length in the same classes. (SI, South Island; NI, North Island; NZ, New Zealand.)

	% sites in data set			% lowland area			% lowland river length		
Class	SI	NI	NZ	SI	NI	NZ	SI	NI	NZ
Exotic forest	6.5	2.3	3.8	5.1	8.4	6.9	4.5	6.9	6.0
Native forest	14.6	10.7	12.1	31.5	24.5	27.6	22.9	15.9	18.7
Pastoral	71.5	79.5	76.6	54.0	62.4	58.7	71.2	75.4	73.7
Urban	7.3	7.4	7.4	0.8	1.6	1.3	1.3	1.8	1.6

trends comparable across sites, each SKSE value was divided by the median value for the site, yielding a relative SKSE (RSKSE).

Nutrient, *E. coli*, clarity, conductivity, and temperature data used for trend analysis were flow-adjusted to reduce variability associated with fluctuating discharge. Such variability may be caused by dilution or erosion during high flows, and can obscure the monotonic, interannual trends that were of interest in this study. Flow adjustments were made using daily average flows for each sampling site and date. The flow data were acquired as follows. The locations of monitoring sites were compared to locations of flow recorders in New Zealand's National Hydrometric Network (Walter 2000). Where monitoring sites were at the same locations as flow recorders, daily average flows from the recorders were acquired. For monitoring sites that were not located at recorders, the most appropriate recorder in the area was selected, based on proximity and similarity in area, elevation, slope, and aspect between the monitoring site and neighbouring catchments with recorders. Daily average flows at the monitoring sites were then estimated by scaling the flows at the gauged sites. Scaling factors were specific water yield ($m^3\ s^{-1}\ km^{-2}$) and net precipitation (precipitation – potential evapotranspiration). Mean annual precipitation and evapotranspiration data are from Leathwick et al. (2003). About 75% of flow recorder sites were within 10 km of the corresponding monitoring site, and 50% were within 1 km.

Flow adjustments were made using the procedure described by Smith et al. (1996). Briefly, values of each water quality parameter were plotted against discharge, and the LOWESS smoothing procedure applied using visual analysis software (DataDesk 6.1; Velleman 1998). Residuals were calculated as the differences between observed and smoothed values, then the residuals were adjusted by adding the median for all data in the plot. Trend analyses were carried out using adjusted residuals.

Median RSKSE values were computed for each land-cover class, and for each climate class and stream order within the pastoral class. Confidence intervals were computed for each median representing ≥12 sites. Median RSKSE values whose confidence intervals did not include zero were considered statistically significant trends.

RESULTS

Water quality state

Differences among land-cover classes were significant for DRP, NO_X, NH_4, *E. coli*, and clarity (Kruskal-Wallis tests, $P < 0.05$), but not for conductivity. Pair-wise comparisons indicated that NO_X, NH_4, and *E. coli* concentrations were significantly lower, and clarity significantly higher, in the native forest and plantation forest classes compared with the urban and pastoral classes (Table 4). Differences between the urban and pastoral classes, and between the native and plantation forest classes were not significant for any parameter.

Median DRP, NO_X, NH_4, and *E. coli* concentrations in streams from the pastoral and urban land-cover classes exceeded recommended guidelines, and did not meet the guideline value for clarity (Fig. 3). In addition, the median *E. coli* concentration in native forest streams, and conductivity in plantation forest streams exceeded the guidelines. Urban and pastoral stream water quality was particularly low: the *E. coli* guideline was exceeded at all urban sites,

Table 3 Guideline water quality values for protection of New Zealand low-elevation river ecosystems, and human health. Nutrient and clarity guidelines are based on reference conditions; conductivity and *Escherichia coli* guidelines are based on relationships with periphyton biomass and public health risk, respectively. (NH_4, ammonium; NO_X, oxidised nitrogen; DRP, dissolved reactive phosphorus; TN, total nitrogen; TP, total phosphorus.)

Parameter and unit	Guideline value	Reference
NH_4 (g N m^{-3})	0.02	ANZECC & ARMCANZ (2000)
NO_X (g N m^{-3})	0.44	ANZECC & ARMCANZ (2000)
DRP (g m^{-3})	0.01	ANZECC & ARMCANZ (2000)
TN (g N m^{-3})	0.61	ANZECC & ARMCANZ (2000)
TP (g P m^{-3})	0.03	ANZECC & ARMCANZ (2000)
Clarity (m)	1.3	ANZECC & ARMCANZ (2000)
Conductivity (µS cm^{-1})	175	Biggs (1988, 2000)
Escherichia coli (/100 ml)	<126 (median)	MfE & MoH (2002)

and DRP, NO_X, and NH_4 guidelines at 92%, 86%, and 83% of urban sites, respectively. The *E. coli* guideline was exceeded at 96% of pastoral sites, and DRP, NH_4, and NO_X guidelines at 88%, 78%, and 64% of pastoral sites, respectively. When all cover classes were combined, the nationwide median DRP, NO_X, NH_4, and *E. coli* concentrations in low-elevation streams exceeded recommended guidelines, but met the guidelines for clarity and conductivity (Fig. 3).

Median DRP, NH_4, TN, TP, and *E. coli* concentrations in pastoral streams from all climate classes exceeded the recommended guidelines (Fig. 4). Median NO_X concentrations in the cool dry (CD) and warm dry (WD) classes (1.06 and 0.73 g m^{-3}, respectively) exceeded the 0.44 g m^{-3} guideline, but median NO_X concentrations in the cool wet (CW) class (0.49 g m^{-3}) and the warm wet (WW) class (0.41 g m^{-3}) were very close to the guideline. The high median NO_X concentration in the pastoral class as a whole (0.61 g m^{-3}) is largely a result of streams in the CD and WD classes. Median conductivity was below the guideline value in all climate classes.

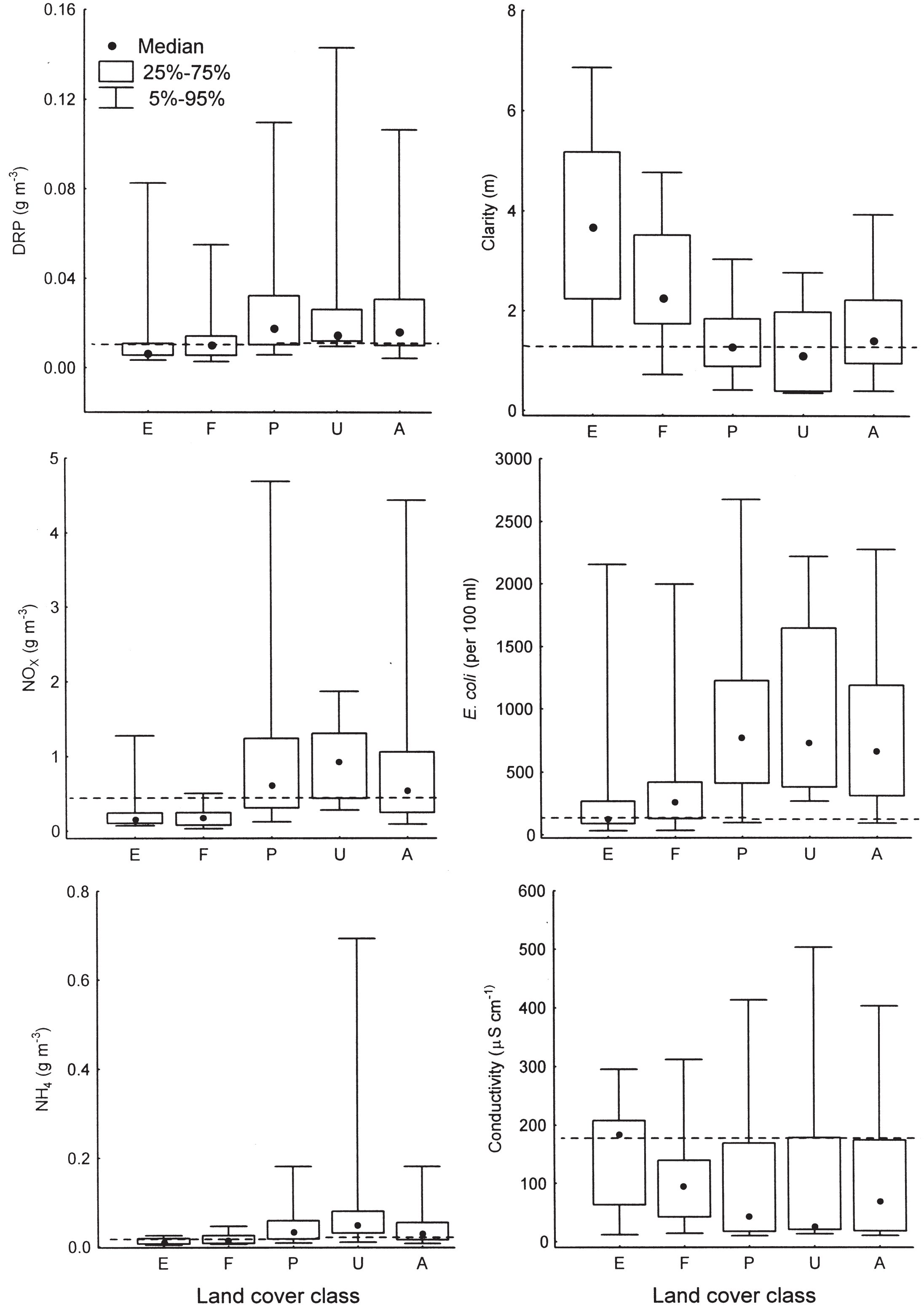
Differences among climate classes were significant for NO_X, NH_4, TN, and TP (Kruskal-Wallis tests, $P < 0.05$), but not for DRP, *E. coli*, clarity, or conductivity. Pair-wise comparisons indicated that the CD climate class had significantly higher NO_X, NH_4, and TN concentrations than the CW and WW classes, and the WD class had higher NH_4 concentrations than the CW and WW classes (Table 5). In contrast, the WD and WW classes had higher TP concentrations than the CD class. There were too few sites in the CX and WX classes to include them in the rank comparisons.

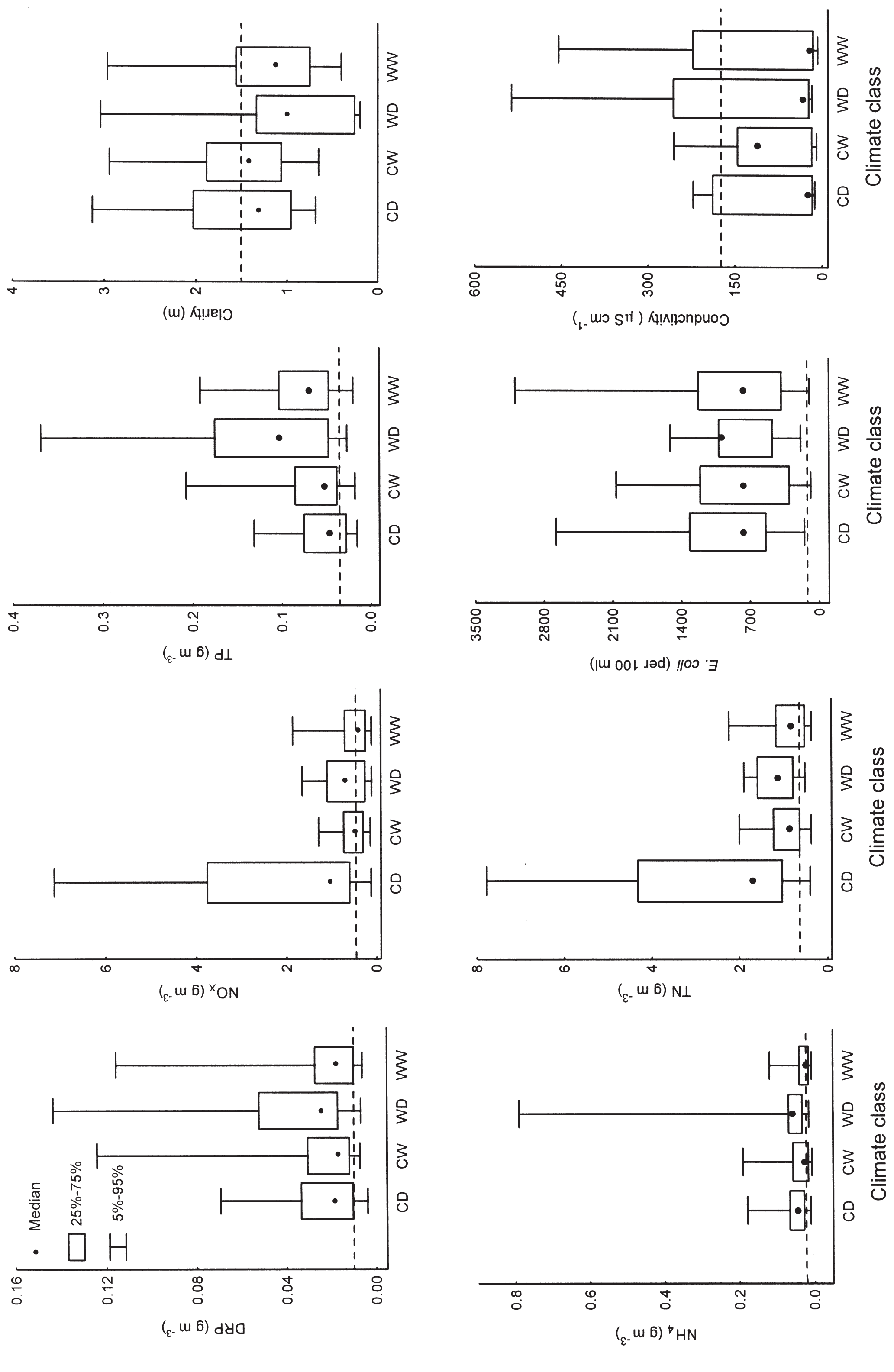
Median DRP, NO_X, TN, TP, and *E. coli* concentrations in pastoral streams from all stream orders sampled exceeded the recommended guidelines (Fig. 5). The median NH_4 concentration in 7th order rivers (0.02 g m^{-3}) was equal to the guideline, but median concentrations for all other orders exceeded the NH_4 guideline. Small streams (1st–3rd order) had median clarity values higher than the 1.3 m guideline, but median clarity values in 4th–7th order streams were below the guideline. A pattern of decreasing median clarity with increasing stream order is apparent in Fig. 5. A linear regression of clarity on stream order was significant ($P = 0.003$), and predicted a 13% change in clarity between each order. However, the regression had low explanatory power ($R^2 = 0.04$).

Fig. 3 Distributions of nutrient, clarity, *Escherichia coli*, and conductivity levels in low-elevation rivers in contrasting land-cover classes (E, plantation forest; F, native forest; P, pastoral; U, urban; A, all classes). Key in dissolved reactive phosphorus (DRP) graph applies to all graphs: points are medians, boxes show 50% of site means, whiskers show 90% of site means, dashed lines indicate guideline values from Table 3. ➤

Table 4 Water quality parameters ordered by land-cover class, from highest mean rank on left, to lowest on right. Significant pair-wise differences among classes are indicated by different superscript letters. (DRP, dissolved reactive phosphorus; NO_X, oxidised nitrogen; NH_4, ammonium; U, urban; P, pastoral; F, forest; E, exotic forest.)

	Rank			
Parameter	1	2	3	4
DRP	U^A (25)	P^A (249)	F^B (34)	E^B (12)
NO_X	U^A (22)	P^A (182)	E^B (10)	F^B (23)
NH_4	U^A (24)	P^A (259)	F^B (37)	E^B (11)
Escherichia coli	U^A (12)	P^A (177)	F^B (28)	E^B (12)
Conductivity	E^A (11)	F^A (23)	U^A (144)	P^A (10)
Clarity	E^A (13)	F^A (39)	P^B (206)	U^B (23)


Median
25%-75%
5%-95%
DRP (g m-3)
0.16
0.12
0.08
0.04
0.00
Clarity (m)
8
6
4
2
0
NOX (g m-3)
5
4
3
2
1
0
E. coli (per 100 ml)
3000
2500
2000
1500
1000
500
0
NH4 (g m-3)
0.8
0.6
0.4
0.2
0.0
Conductivity (µS cm-1)
600
500
400
300
200
100
0
E F P U A
Land cover class
Land cover class


DRP (g m^{-3})
Median
25%-75%
5%-95%
NO_x (g m^{-3})
TP (g m^{-3})
Clarity (m)
NH_4 (g m^{-3})
TN (g m^{-3})
E. coli (per 100 ml)
Conductivity (µS cm^{-1})
CD CW WD WW
Climate class

◄ **Fig. 4** Distributions of nutrient, clarity, *Escherichia coli*, and conductivity levels in pastoral streams from four climate classes (CD, cool dry; CW, cool wet; WD, warm dry; WW, warm wet). Dashed lines indicate guideline values from Table 3. Key in dissolved reactive phosphorus (DRP) graph applies to all graphs.

Table 5 Water quality parameters in pastoral streams, ordered by climate class, from highest mean rank on left, to lowest on right. Significant pair-wise differences following significant Kruskal-Wallis tests are indicated by different superscript letters. (NO_X, oxidised nitrogen; NH_4, ammonium; TN, total nitrogen; TP, total phosphorus; CD, cool dry; WD, warm dry; CW, cool wet; WW, warm wet.)

	Rank			
Parameter	1	2	3	4
NO_X	CD^A (76)	WD^{AB} (8)	CW^B (49)	WW^B (47)
NH_4	WD^A (13)	CD^A (76)	CW^B (56)	WW^B (97)
TN	CD^A (72)	WD^{AB} (7)	CW^B (32)	WW^B (27)
TP	WD^A (13)	WW^A (88)	CW^{AB} (38)	CD^B (72)

Differences among the stream orders in the pastoral class were significant for NO_X and TN (Kruskal-Wallis tests, $P < 0.05$). Pair-wise comparisons among orders indicated that the median NO_X concentration in 2nd order streams was higher than in 7th order rivers, and the median TN concentration in 2nd order streams was higher than in both 6th and 7th order rivers. No other pair-wise comparisons were statistically significant.

Comparisons of water quality in low-elevation rivers with rivers from all elevations (based on data from NRWQN) indicated that median NO_X and NH_4 concentrations in low-elevation rivers were 4 times higher than the NRWQN median, DRP and TN concentrations were 3 times higher than the NRWQN median, and the low-elevation TP concentration was twice the NRWQN median (Table 6). These comparisons indicate that low-elevation rivers are nutrient-enriched compared with rivers in New Zealand as a whole. Median clarity in the low-elevation streams (1.4 m) was similar to that of the NRWQN sites (1.7 m). Clarity in low-elevation rivers is primarily controlled by particulate and dissolved organic matter (Davies-Colley & Close 1990). However, a large proportion of NRWQN sites are located at higher elevations, where organic matter concentrations tend to be lower. Low clarity in high elevation rivers may instead be a result of high concentrations of inorganic sediments, including glacier-derived silts.

Water quality trends

During the 1996–2002 period, there were statistically significant positive trends in temperature and clarity and a negative trend in river flow at the national scale (Table 7). Within land-cover classes, there were negative trends in flow in the pastoral and urban classes, positive trends in clarity and temperature in the pastoral class, and positive trends in conductivity in the urban and native forest classes (Table 7). The positive clarity and temperature trends were not caused by flow reduction or flow-controlled turbidity or thermal storage, because the clarity and temperature data were flow-adjusted. The significance of trends in the plantation forest class could not be tested, as there were too few sites to compute confidence limits. In general, temporal trends were quite small, reflecting changes of 0.5% or less per year in the median value of each parameter. The median trend in *E. coli* concentrations in urban streams was –0.5%/year, but this trend was not significant because of the mixture of positive and negative trending sites within the class. No significant within-class or among-class trends in nutrient or *E. coli* concentrations were detected.

When sites in the pastoral land-cover class were grouped by climate class, a negative trend in flow was apparent only in the WW climate class (Table 8). This class included more pastoral sites (90 out of 181 pastoral sites) than the other climate classes, but was limited in geographic scope, as all of the WW sites were located on the North Island. There were positive trends in temperature in the CD and WW classes, and positive trends in clarity in the WD and WW classes. The CD and WD classes included sites on both the North and South Islands. There were no statistically significant trends in nutrient or *E. coli* concentrations within climate classes.

Trends within stream-orders in the pastoral land-cover class were generally consistent with trends in the pastoral class as a whole. There were negative trends in flow, and positive trends in temperature in 4th, 5th, and 6th order streams, and a positive trend in clarity in 4th order streams (Table 8). Trends in 1st and 2nd order streams were not analysed because there were too few sites in the data set.

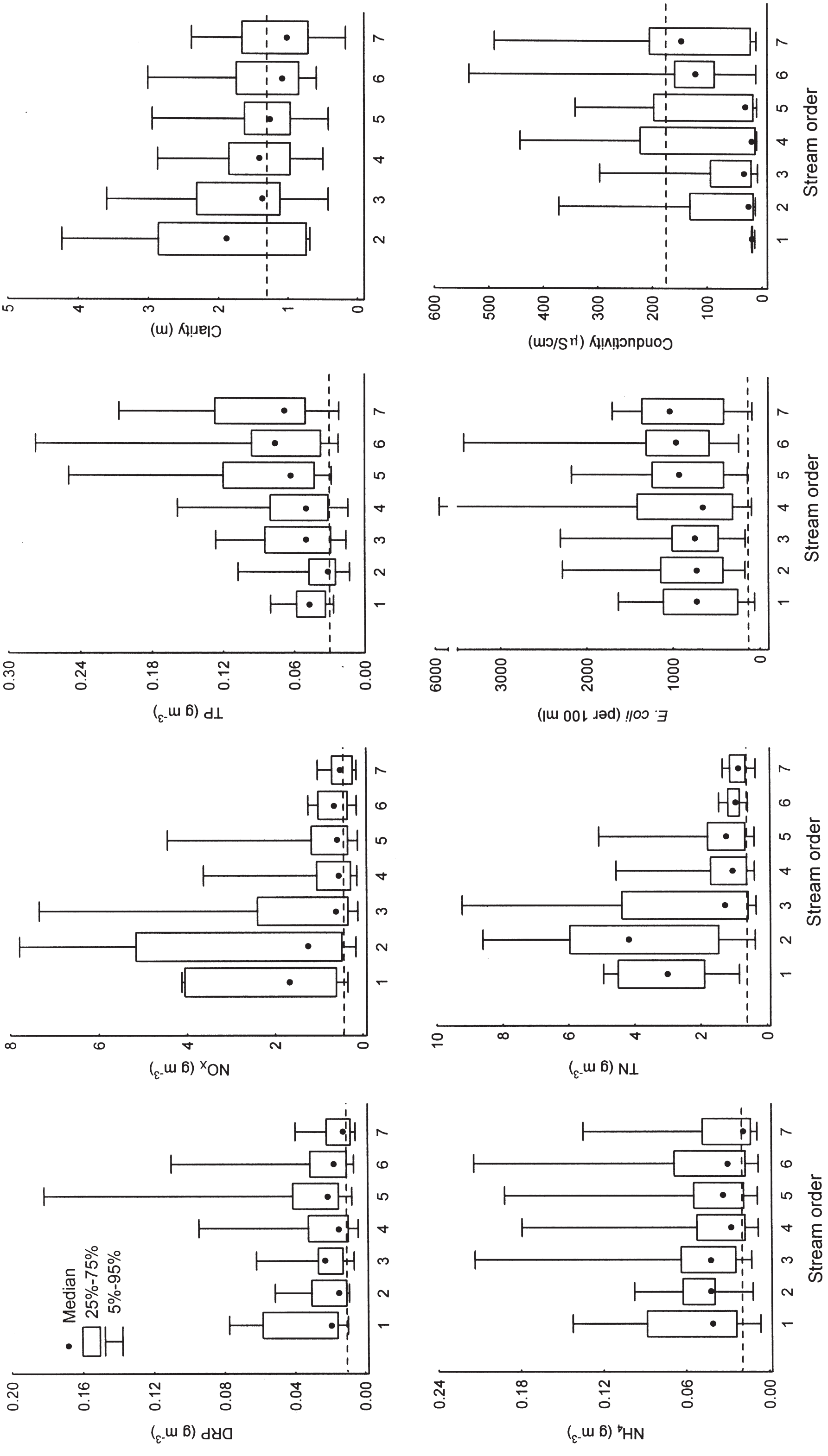

DRP (g m-3)
NOx (g m-3)
TP (g m-3)
Clarity (m)
NH4 (g m-3)
TN (g m-3)
E. coli (per 100 ml)
Conductivity (μS/cm)
Median
25%-75%
5%-95%
Stream order

Table 6 Water quality in low-elevation rivers of New Zealand (this study), and in rivers representing all elevation zones (NRWQN, National River Water Quality Network). Low-elevation river data is from December 1998 to December 2002. NRWQN data is from January 1999 to November 2002. Nutrient concentrations in g m^{-3}, clarity in m, *Escherichia coli* in number per 100 ml, conductivity in µS/cm. (NA, no data available; DRP, dissolved reactive phosphorus; NO_X, oxidised nitrogen; NH_4, ammonium; TN, total nitrogen; TP, total phosphorus; clar., water clarity; cond., conductivity.)

	DRP	NO_X	NH_4	TN	TP	Clar.	*E. coli*	Cond.
Low-elevation								
Median	0.016	0.55	0.029	1.03	0.06	1.4	664	68.6
Mean	0.033	1.08	0.058	1.71	0.07	1.7	906	113.6
SD	0.065	1.48	0.144	1.82	0.08	1.1	1033	124.6
N	320	237	322	158	254	281	229	188
NRWQN								
Median	0.006	0.15	0.007	0.33	0.03	1.7		118.6
Mean	0.011	0.26	0.011	0.41	0.04	2.3	NA	142.4
SD	0.014	0.33	0.013	0.39	0.04	1.8		93.7
N	77	77	77	75	76	77		77

Table 7 Trends (Relative Seasonal Kendall slope estimators) in flow and flow-adjusted water quality parameters in within and across land-cover classes. Values are % change in median parameter value per year. Arrows indicate directions of statistically significant trends. There were too few plantation forest sites to compute confidence intervals. (***, 99% confidence interval (CI) did not include zero; **, 95% CI did not include zero; *, 90% CI did not include zero; DRP, dissolved reactive phosphorus; NO_X, oxidised nitrogen; NH_4, ammonium; *E. coli*, *Escherichia coli*.)

Land cover	Flow	DRP	NO_X	NH_4	Temp.	Conductivity	Clarity	*E. coli*
Native forest	–0.17	0	–0.05	0.02	–0.01	0.11 ↑ *	0.01	–0.03
Pastoral	–0.35 ↓ ***	0	–0.01	–0.03	0.04 ↑ ***	0.01	0.48 ↑ ***	0
Urban	–0.46 ↓ *	0	0	0	0.05	0.02 ↑ *	0.27	–0.51
Plantation forest	–1.19 ↓ *	–0.07	–0.04	0	0.06	0.26	0.54	0.55
All classes	–0.33 ↓ ***	0.05	–0.04	–0.16	0.02 ↑ ***	0.07	0.16 ↑ ***	–0.29

Fig. 5 Distributions on water quality data in pastoral streams by stream order. Key in dissolved reactive phosphorus (DRP) graph applies to all graphs. Note axis break in *Escherichia coli* graph. Dashed lines indicate guideline values from Table 3. There were too few clarity data from 1st-order streams to plot.

DISCUSSION

Water quality in contrasting land-cover classes

There is a long-standing prediction among water quality researchers that stream water quality in developed catchments in New Zealand will be lower than in undeveloped catchments (McColl & Hughes 1981; Davis 1986; Hughes et al. 1986). This prediction has been tested at fine spatial-scales in paired and multiple-catchment studies. Nearly all fine-scaled comparisons concluded that water quality in low-elevation pastoral and urban catchments was worse than in undeveloped catchments. Our results indicate that many of the fine-scaled differences reported in these studies exist today at the national scale: DRP, NO_X, NH_4, and *E. coli* concentrations in the pastoral and urban classes were 2–7 times higher than in the native and plantation forest classes, and median clarity in the pastoral and urban classes was 30–60% of that in the native and plantation forest classes.

The similarities we observed in water quality in the urban and pastoral classes were unexpected. Urban catchments are characterised by high surface run-off and periodically high stormwater discharge, which led us to expect urban water quality to be worse than that in pastoral catchments (Paul & Meyer 2001). We know of no other water quality studies in New Zealand that included pastoral-urban comparisons, but several such comparisons have been made in North America and Europe. Results of these studies are not consistent: in some studies, urban streams had higher concentrations of suspended sediments, faecal bacteria, and some nutrients (e.g., Osborne & Wiley 1988; Arienzo et al. 2001; Sonoda et al. 2001); in other studies, higher concentrations occurred in pastoral streams (e.g., Lenat & Crawford 1994).

There are several possible explanations for the similarities we observed in the urban and pastoral classes. First, many urban catchments in New Zealand (defined in the REC as those with >15% urban land cover) include substantial proportions of pastoral land-cover. Water quality in these catchments represents the net effects of urban and pastoral land uses, and including such catchments in the urban class may reduce differences between the urban and pastoral classes. Second, some of the major pathways of water quality degradation are present in both pastoral and urban catchments. Both pastoral and urban streams have high wastewater input from non-point sources: tile drains and ditches in pastoral areas, and storm drains and leaking sewerage in urban areas. Both pastoral and urban catchments have lower interception and lower infiltration rates than forest catchments, resulting in greater overland flow and bank erosion (Paul & Meyer 2001). Third, the pastoral and urban classes are both broad categories that encompass a range of specific land uses (e.g., dairy farming, row cropping, manufacturing, urban housing) and hydrogeologic terrains. Differences in water quality may be more apparent when broad classes are subdivided by specific land use and terrain. Comparisons among these land uses would be a logical advance in water quality assessment, but suitable data for such detailed assessments are rare.

Within the pastoral land-cover class, nutrient concentrations differed widely among some climate classes. Median dissolved and total nitrogen concentrations in the CD climate class were over

Table 8 Trends (Relative Seasonal Kendall slope estimators) in flow and flow-adjusted water quality parameters in the pastoral land-cover class in 4 climate classes and 5 stream orders. Values are % change in median parameter value per year. Arrows indicate directions of statistically significant trends. There were too few sites in the CX and WX classes and in stream orders 1 and 2 to estimate trends. (***, 99% confidence interval (CI) did not include zero; **, 95% CI did not include zero; *, 90% CI did not include zero; DRP, dissolved reactive phosphorus; NO_X, oxidised nitrogen; NH_4, ammonium; *E. coli*, *Escherichia coli*.)

	Flow	DRP	NO_X	NH_4	Temp.	Conductivity	Clarity	*E. coli*
Climate								
Cool dry	–0.18	0.01	–0.01	–0.03	0.07 ↑ **	0.01	0.17	0.46
Cool wet	–0.25	0.02	0	–0.01	0.04	0.02	0.02	0.31
Warm dry	–0.39	–0.03	–0.27	–0.01	0.04	0.01	0.08 ↑ ***	–0.37
Warm wet	–0.35 ↓ ***	0	–0.01	–0.04	0.04 ↑ ***	0	0.27 ↑ ***	–0.01
Stream order								
3	–0.27	–0.01	–0.01	–0.03	0.02	0.04	0	0.18
4	–0.55 ↓ ***	0	0	–0.03	0.07 ↑ ***	0.01	0.12	–0.23
5	–0.41 ↓ ***	0.05	0	–0.08	0.08 ↑ **	0.05	0.18 ↑ ***	–0.04
6	–0.36 ↓ **	0.01	100–0.06	–0.03	0.06 ↑ **	0.03	0.30	–0.09
7	0.03	0.06	0.05	0	0.03	–0.01	–0.07	0.52

twice those in the CW and WW classes, and the median TP concentration in the WD class was over twice that in the CD class. The causes of these differences are not clear, but three general explanations should be considered. First, climatic processes such as solar heating may directly affect nutrient levels via temperature-regulated nutrient cycling or flow intermittency (e.g., Monteith et al. 2000). Second, agricultural and water management practices may vary at about the same scale as climate classes, and variation in these practices may have caused the differences we observed. There is currently not enough spatially-explicit information (e.g., fertiliser application and irrigation rates, stocking densities, riparian retirement) to test this proposition. Third, variation within and between climate classes may reflect variable groundwater influence on surface water quality. In particular, the high variability in TN and NO_X concentrations in the CD class may reflect differences among sites in groundwater input. Many low-elevation CD sites are on the alluvial plains of Hawke's Bay and the eastern South Island, and these sites may be effluent (net input of groundwater to stream channel) or influent (net loss of surface water to aquifer) (White et al. 2001). Effluent sites may be enriched in dissolved nitrogen as a result of both natural microbial degradation of particulate nitrogen in groundwater, and of human-caused nitrogen leaching, while influent sites are primarily affected by conditions in the catchment upstream (Burden 1984; Duff & Triska 2000).

There were few significant differences in water quality parameters between stream orders in the pastoral land-cover class, and no systematic changes across stream orders. In studies seeking to partition variability along river systems, stream order has been a good predictor for biological parameters, but a generally poor predictor for water quality (e.g., Naiman et al. 1987; Beecher et al. 1988). Larger-scale variables such as climate, catchment geology, and land use appear to have greater effects than stream order or size on water quality. The lack of systematic change in water quality with stream order in this study suggests that assessments at the scale of land-cover account for most fine-scaled systematic variation in water quality.

In our comparisons of water quality in plantation forest and native forest catchments, there were no statistically-significant differences among parameters. Previous comparisons in New Zealand have not yielded consistent results with which to compare our findings. Friberg et al. (1997) and Quinn et al. (1997) reported higher NO_X concentrations in pine plantation streams, and clarity was lower in pine plantation streams in the latter study. In studies by McColl et al. (1977) and Harding & Winterbourn (1995) differences in NO_X concentrations in pine plantation and native forest streams were not statistically significant. In the mid-elevation Purukohukohu experimental basin (530–650 m a.s.l.), DRP and TP concentrations in pine plantation streams were higher than in adjacent native forest streams, but NO_X and suspended sediment concentrations were higher in the native forest streams (Cooper et al. 1987; Dons 1987). Inconsistent results from paired plantation-native forest comparisons may be partly because of variability in stand age and management practices. Elevated nutrient and sediment input to streams is typical of pine plantations for several years after road construction, site preparation, planting, and clear-felling (Fahey & Coker 1992; Oyarzun & Peña 1995). As stands mature, interception and nutrient uptake rates increase, peak flows decline, and sediment and nutrient losses decline (Fahey & Rowe 1992). With regard to water quality, decades-old pine plantations appear to function like native forests, with low rates of sediment and nutrient loss compared with pastoral land (Friberg et al. 1997). In addition to stand age-related effects, fertilisation affects DIN and DRP losses from plantation forests to streams (Neary & Leonard 1978; Binkley et al. 1999). Stand history and management information is rarely included in water quality studies of plantation forests. If these details were made available, the precision of water quality assessments could be increased.

Streams draining undeveloped catchments are often used as reference sites for assessing or predicting effects of land use on water quality, and for developing guidelines for environmental protection (Clark et al. 2000). However, water quality in undeveloped catchments may exhibit high spatial variability, corresponding to variation in climate, vegetation, and lithology. Spatially variable water quality in undeveloped catchments can reduce the utility of guidelines (Rohm et al. 2002); guidelines based on average conditions from a broad range of undeveloped catchments may be overly strict for some assessments, and overly lenient for others. Variability in water quality across undeveloped catchments in New Zealand is high, as indicated by coefficients of variation for water quality parameters in the native forest class in our study. These ranged from 0.5 for clarity to 2.1 for DRP. High among-site variability suggests that national averages for low-elevation, native forest streams will be of limited use as water quality guidelines. The guidelines in current use in New Zealand for nutrient concentrations and clarity are very broad-scaled, distinguishing only upland and lowland rivers, and the lowland river guidelines were derived from data from only four NRWQN sites (ANZECC & ARMCANZ 2000).

There is a clear need to identify low-elevation reference sites in different areas of New Zealand, and to establish regional reference conditions based on these sites. It is also imperative that consistent criteria be defined for selecting reference sites (Hughes et al. 1986). No low-elevation rivers are truly pristine (i.e., unaffected by human activity), so the types and severity of human impact that are acceptable at reference sites must be specified. Techniques for comparing water quality and biological parameters in reference and test sites have received considerable attention (e.g., Reynoldson et al. 2001), but detailed criteria for selecting reference sites are rarely provided. Rather, reference sites are generally described as "minimally disturbed" (e.g., Davies et al. 2000). If reference sites are selected for New Zealand regions, we recommend the development of a full set of reference-condition guidelines (i.e., guidelines for all common water quality parameters). Alternatively, a full set of effects-based guidelines may be developed. Effects-based guidelines relate ecological properties such as eutrophication and hypoxia to the water quality parameters that control those properties. In the present study, we combined effects-based and reference-condition guidelines because of the absence of a complete set of either type. Effects-based and reference-condition guidelines differ in derivation and aim; the former are derived from correlations and experiment results and are used to indicate impairment thresholds, whereas the latter are derived from reference site data and are used to indicate natural conditions. Because of these differences, future assessments should seek to use one type consistently.

Trends

Significant trends in low-elevation streams from 1996 to 2002 were limited to four parameters, flow (trending down in all instances), and temperature, clarity, and conductivity (trending up in all instances). The trends in flow, temperature, and clarity are apparent at the national scale, and within the pastoral class. The pastoral class dominated the trend data set with 72% of monitoring sites, and it is possible that changes in pastoral land uses are responsible for the trends that occurred at both the class scale and the national scale. We suggest below that this is not so; our argument is based on a recent trend analysis of NRWQN data that examined the influence of the El Niño/Southern Oscillation (ENSO) climate system on water quality in New Zealand (Scarsbrook et al. in press).

Between 1989 and 2001, temporal trends in flow, temperature, and clarity at NRWQN sites were closely related to the Southern Oscillation Index (SOI), a standard measure of ENSO intensity. River flow was positively related to SOI in northern New Zealand, and negatively related to SOI in southern New Zealand (Scarsbrook et al. in press). These relationships are consistent with the effects of ENSO on precipitation across New Zealand (Salinger & Mullan 1999). Flow-adjusted water temperatures and clarity levels increased with ENSO intensity throughout the country. When the NRWQN sites in the analysis were divided into "baseline" and "impact" sites (upstream and downstream from developed areas, respectively), the relationships between water quality trends and SOI were largely unchanged. These observations suggest that large-scale climate trends controlled flow, temperature, and clarity trends.

Climate conditions in New Zealand during the period of the current study, 1996–2002, were characterised by alternating El Niño and La Niña conditions, with brief neutral periods (Island Climate Update, http: www.niwa.co.nz/NCC/ICU). Partly as a result of this cycle, there was no monotonic trend in ENSO intensity during the study period (Australia Bureau of Meteorology, http: www.bom.gov.au/climate). The weak trends we observed in flow, temperature, and clarity (< 0.4 %/year change in each parameter) may reflect the net effects of the ENSO cycle.

Positive trends in conductivity were detected in the urban and native forest land-cover classes. Conductivity has been recommended as a surrogate for nutrient enrichment, because it is often correlated with periphyton, but is not strongly affected by in-stream nutrient uptake (Biggs & Price 1987). However, the fact that conductivity increased in both the native forest and urban classes suggests that land-use intensification was not the major or sole cause of the trend. Climatic factors such as air temperature and precipitation affect conductivity via rock weathering and atmospheric input; temporal changes in these factors may have been partly responsible for conductivity trends (Johnson et al. 1994).

A caveat is required concerning the preceding arguments. No direct comparisons have been made between trends in water quality across New Zealand, and trends in the land-use attributes that are likely to affect water quality, such as human populations, livestock densities, irrigated acreage, or clear-felling. At present, there is insufficient data about these attributes for national-scaled trend analyses. Regional studies have had more success, as some regional land-use trends have been quantified (e.g., Hamill & McBride 2003).

The small temporal trends we report stand in sharp contrast to the large differences in water quality state among land-cover classes. The few trends that were detected appear to be more closely related to climate variability than to land-use change. A reasonable interpretation of these results is that among-class differences in water quality state developed before the 1996–2002 period, and current differences are relatively stable. If this is accurate, then recent changes in land-use practices, climate conditions, and other sources of anthropogenic and natural variability have had only minor effects on low-elevation stream water quality, or have occurred at scales too small to be detected by this large-scale study.

ACKNOWLEDGMENTS

Many thanks to the regional, district and city council staff that provided water quality data, and details of analytical methods. Thanks also to our colleagues at NIWA for their contributions: Katie Image and Keri Niven classified sites, organised data and computed river kilometres; George Payne computed trend statistics; Helen Hurren and Mark Weatherhead produced GIS maps; Ross Woods, Roddy Henderson, and Kathy Walter prepared flow estimates; and Graham McBride reviewed the manuscript. We thank Adrian Meredith and an anonymous referee for helpful reviews. Financial support was provided by the New Zealand Ministry for the Environment, and the Environmental Hydrology and Habitat Hydraulics Programme (Contract C01X0215) of the Foundation for Research, Science and Technology.

REFERENCES

ANZECC & ARMCANZ 2000: Australian and New Zealand guidelines for fresh and marine water quality. Volume 1. The guidelines. Canberra, Australia, Australia and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand.

Arienzo, M.; Adamo, P.; Rosario Bianco, M.; Violante, P. 2001: Impact of land use and urban runoff on the contamination of the Sarno River Basin in southwestern Italy. *Water, Air and Soil Pollution 131*: 349–366.

Beecher, H. A.; Dott, E. R.; Fernau, R. F. 1988: Fish species richness and stream order in Washington State streams. *Environmental Biology of Fishes 22*: 193–209.

Biggs, B. J. F. 1988: Algal proliferations in New Zealand's shallow stony foothills-fed rivers: toward a predictive model. *Verhandlungen der Internationalen Vereinigung für Theoretische und Angewandte Limnologie 23*: 1405–1411.

Biggs, B. J. F. 1995: The contribution of flood disturbance, catchment geology and land use to the habitat template of periphyton in stream ecosystems. *Freshwater Biology 33*: 419–438.

Biggs, B. J. F. 2000: New Zealand periphyton guideline: detecting, monitoring and managing enrichment of streams. Wellington, Ministry for the Environment. 122 p.

Biggs, B. J. F.; Price, G. M. 1987: A survey of filamentous algal proliferations in New Zealand rivers. *New Zealand Journal of Marine and Freshwater Research 21*: 175–191.

Binkley, D.; Burnham, H.; Allen, H. L. 1999: Water quality impacts of forest fertilization with nitrogen and phosphorus. *Forest Ecology and Management 121*: 191–213.

Burden, R. J. 1984: Chemical zonation in groundwater of the Central Plains, Canterbury. *Journal of Hydrology (New Zealand) 23*: 100–119.

Clark, G. M.; Mueller, D. K.; Mast, M. A. 2000: Nutrient concentrations and yields in undeveloped stream basins of the United States. *Journal of the American Water Resources Association 364*: 849–860.

Close, M. E.; Davies-Colley, R. J. 1990: Baseflow water chemistry in New Zealand rivers. 1. Characterisation. *New Zealand Journal of Marine and Freshwater Research 24*: 319–341.

Cooper, A. B.; Hewitt, J. E.; Cooke, J. G. 1987: Land use impacts on streamwater nitrogen and phosphorus. *New Zealand Journal of Forestry Science 17*: 179–192.

Davies, N. M.; Norris, R. H.; Thoms, M. C. 2000: Prediction and assessment of local stream habitat features using large-scale catchment characteristics. *Freshwater Biology 45*: 343–369.

Davies-Colley, R. J. 1988: Measuring water clarity with a black disk. *Limnology and Oceanography 33*: 616–623.

Davies-Colley, R. J.; Close, M. E. 1990: Water colour and clarity of New Zealand rivers under baseflow conditions. *New Zealand Journal of Marine and Freshwater Research 24*: 357–365

Davis, S. F. 1985: Effects of land development on aquatic ecosystems. *New Zealand Agricultural Science 19*: 156–161.

DoH 1992: Provisional water quality guidelines for recreational and shellfish- gathering waters in New Zealand. Wellington, Public Health Services, Department of Health. 17 p.

Dons, A. 1987: Hydrology and sediment regime of a pasture, native forest, and pine forest catchment in the central North Island, New Zealand. *New Zealand Journal of Forestry Science 17*: 161–178.

Duff, J. H.; Triska, F. J. 2000: Nitrogen biogeochemistry and surface-subsurface exchange in streams. *In*: Jones, J. B.; Mulholland, P. J. *ed*. Streams and ground waters. San Diego, California, Academic Press. Pp. 197–220.

Duggan, I. C.; Collier, K. J.; Champion, P. D.; Croker, G. F.; Davies-Colley, R. J.; Lambert, P. W.; Nagels, J. W.; Wilcock, R. J. 2002: Ecoregional differences in macrophyte and macroinvertebrate communities between Westland and Waikato: are all New Zealand lowland streams the same? *New Zealand Journal of Marine and Freshwater Research 36*: 831–845.

Edwards, P. J.; Kochenderfer, J. N.; Seegrist, D. W. 1991: Effects of forest fertilization on stream water chemistry in the Appalachians. *Water Resources Bulletin 27*: 265–274.

Fahey, B. D.; Coker, R. J. 1992: Sediment production from forest roads in Queen Charlotte Forest and potential impact on marine water quality, Marlborough Sounds, New Zealand. *New Zealand Journal of Marine and Freshwater Research 26*: 187–195.

Fahey, B. D.; Rowe, L. K. 1992: Land use impacts. *In*: Mosely, M. P. *ed*. Waters of New Zealand. Wellington, New Zealand Hydrological Society. Pp. 265–284.

Friberg, N.; Winterbourn, M. J.; Shearer, K. A.; Larsen, S. E. 1997: Benthic communities of forest streams in the South Island, New Zealand: effects of forest type and location. *Archiv für Hydrobiologie 138*: 289–306.

Hall, M. J.; Closs, G. P.; 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.

Hamill, K. D.; McBride, G. B. 2003: River water quality trends and increased dairying in Southland, New Zealand. *New Zealand Journal of Marine and Freshwater Research 37*: 323–332.

Harding, J. S.; Winterbourn, M. J. 1995: Effects of contrasting land use on physico-chemical conditions and benthic assemblages of streams in a Canterbury (South Island, New Zealand) river system. *New Zealand Journal of Marine and Freshwater Research 29*: 479–492.

Helsel, D. R.; Hirsch, R. M. 1992: Statistical methods in water resources. Amsterdam, Elsevier.

Hoare, R. A. 1984: Nitrogen and phosphorus in Rotorua urban streams. *New Zealand Journal of Marine and Freshwater Research 18*: 451–454.

Hughes, H. R.; Boshier, J. A.; Lumley, M. 1986: The impact of land use changes on water quality. *New Zealand Agricultural Science 20*: 94–97.

Hughes, R. H.; Larsen, D. P.; Omernik, J. M. 1986: Regional reference sites: a method for assessing stream potentials. *Environmental Management 10*: 629–635.

Johnson, C. E.; Litaor, M. I.; Billett, M. F.; Bricker. O. P. 1994: Chemical weathering in small catchments: climatic and anthropogenic influences. *In*: Moldan, B.; Cerny, J. *ed*. Biogeochemistry of small catchments: a tool for environmental research. Chichester, John Wiley. Pp. 323–341.

Leathwick, J.; Wilson, G.; Rutledge, D.; Wardle, P.; Morgan, F.; Johnson, K.; McLeod, M.; Kirkpatrick, R. 2003: Land environments of New Zealand. Auckland, David Bateman.

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

McColl, R. H. S.; Hughes, H. R. 1981: The effects of land use on water quality—a review. *Water and Soil Miscellaneous Publication No. 23*. Wellington, New Zealand, Ministry of Works and Development. 59 p.

McColl, R. H. S.; White, E.; Gibson, A. R. 1977: Phosphorus and nitrate run-off in hill pasture and forest catchments, Taita, New Zealand. *New Zealand Journal of Marine and Freshwater Research 11*: 729–744

MAF 2003: National exotic forest description. 19th ed. Wellington, New Zealand, Ministry of Agriculture and Forestry. 64 p.

MfE & MoH 2002: Microbiological water quality guidelines for marine and freshwater recreational areas. Wellington, New Zealand, Ministry for the Environment and Ministry of Health. 121 p.

MfE & MoH 2003: Microbiological water quality guidelines for marine and freshwater recreational areas. Wellington, New Zealand, Ministry for the Environment and Ministry of Health. 159 p.

Monteith, D. T.; Evans, C. D.; Reynolds, B. 2000: Are temporal variations in nitrate content of UK upland freshwater linked to the North Atlantic oscillation? *Hydrological Processes 14*: 1745–1749.

Mosley, L. M.; Peake, B. M. 2001: Partitioning of metals (Fe, Pb, Cu, Zn) in urban run-off from the Kaikorai Valley, Dunedin, New Zealand. *New Zealand Journal of Marine and Freshwater Research 35*: 615–624.

Nagashima, K.; Sands, R.; Whyte, A. G. D.; Bilek, E. M.; Nakagoshi, N. 2002: Regional landscape change as a consequence of plantation forestry expansion: an example in the Nelson region, New Zealand. *Forest Ecology and Management 163*: 245–261.

Naiman, R. J.; Melillo, J. M.; Lock, M. A.; Ford, T. E.; Reice, S. R. 1987: Longitudinal patterns of ecosystem processes and community structure in a subarctic stream continuum. *Ecology 68*: 1139–1156.

Neary, D. J.; Leonard, J. H. 1978: Effects of forest fertilisation on nutrient losses in streamflow in New Zealand. *New Zealand Journal of Forest Science 8*: 189–205.

Osborne, L. L; Wiley, M. J. 1988: Empirical relationships between land use/cover and stream water quality in an agricultural watershed. *Journal of Environmental Management 26*: 9–27.

Oyarzun, C. E.; Peña, L. 1995: Soil erosion and overland flow in forested areas with pine plantations at coastal mountain range, central Chile. *Hydrological Processes 9*: 111–118.

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

Quinn, J. M.; Stroud, M. J. 2002: Water quality and sediment and nutrient export from New Zealand hill-catchments of contrasting land use. *New Zealand Journal of Marine and Freshwater Research 36*: 409–429.

Quinn, J. M.; Cooper, A. B.; Davies-Colley, R. J.; Rutherford, J. C.; Williamson, R. B. 1997: Land use effects on habitat, water quality, periphyton, and benthic invertebrates in Waikato, New Zealand, hill-country streams. *New Zealand Journal of Marine and Freshwater Research 31*: 579–597

Reynoldson, T. B.; Rosenberg, D. M.; Resh, V. H. 2001: Comparison of models predicting invertebrate assemblages for biomonitoring in the Fraser River catchment, British Columbia. *Canadian Journal of Fisheries and Aquatic Sciences 58*: 1395–1410.

Rohm, C. M.; Omernick, J. M.; Woods, A. J.; Stoddard, J. L. 2002: Regional characteristics of nutrient concentrations in streams and their application to nutrient criteria development. *Journal of the American Water Resources Association 38*: 213–237.

Salinger, M. J.; Mullan, A. B. 1999: New Zealand climate: temperature and precipitation variations and their links with atmospheric circulation 1930–1934. *International Journal of Climatology 19*: 1049–1071.

Scarsbrook, M. R.; McBride, C. G.; McBride, G. B.; Bryers, G. G. 2003: Effects of climate variability on rivers and consequences for long-term trend analysis. *Journal of the American Water Resources Association 39*: 1435–1447.

Smith, D. G.; McBride, G. B. 1990: New Zealand's national water quality monitoring network: design and first year's operation. *Water Resources Bulletin 26*: 767–775.

Smith, D. G.; Maasdam, R. 1994: New Zealand's National River Water Quality Network 1. Design and physico-chemical characterisation. *New Zealand Journal of Marine and Freshwater Research 28*: 19–35.

Smith, D. G.; McBride, G. B.; Breyers, G. G.; Wisse, J.; Mink, D. F. J. 1996: Trends in New Zealand's National River Water Quality Network. *New Zealand Journal of Marine and Freshwater Research 30*: 485–500.

Snelder, T. H.; Biggs, B. J. F. 2002: Multiscale river environment classification for water resource management. *Journal of the American Water Resources Association 38*: 1–15.

Sonoda, K.; Yeakley, J. A.; Walker, C. E. 2001: Near-stream landuse effects on streamwater nutrient distribution in an urbanizing watershed. *Journal of the American Water Resources Association 37*: 1517–1532.

Taylor, R.; Smith, I. 1997: The state of New Zealand's environment. Wellington, Ministry for the Environment.

Townsend, C. R.; Arbuckle, C. J.; Crowl, T. A.; Scarsbrook, M. R. 1997: The relationship between land use and physicochemistry, food resources and macroinvertebrate communities in tributaries of the Taieri River, New Zealand: a hierarchically scaled approach. *Freshwater Biology 37*: 177–191.

Velleman, P. F. 1988: The DataDesk handbook. Ithaca, New York, Odesta.

Walter, K. M. 2000: Index to hydrological recording sites in New Zealand. *NIWA Technical Report 73*. Christchurch, New Zealand. 216 p.

White, P. A.; Clausen, B.; Hunt, B.; Cameron, S.; Weir, J. 2001: Groundwater-surface water interaction. *In*: Rosen, M. R.; White, P. A. *ed*. Groundwaters of New Zealand. Wellington, New Zealand Hydrological Society. Pp. 133–160.

Williamson, R. B. 1986: Urban stormwater quality II. Comparison of three New Zealand catchments. *New Zealand Journal of Marine and Freshwater Research 20*: 315–328.

Zar, J. H. 1984: Biostatistical analysis. 2nd ed. Englewood Cliffs, New Jersey, Prentice-Hall.