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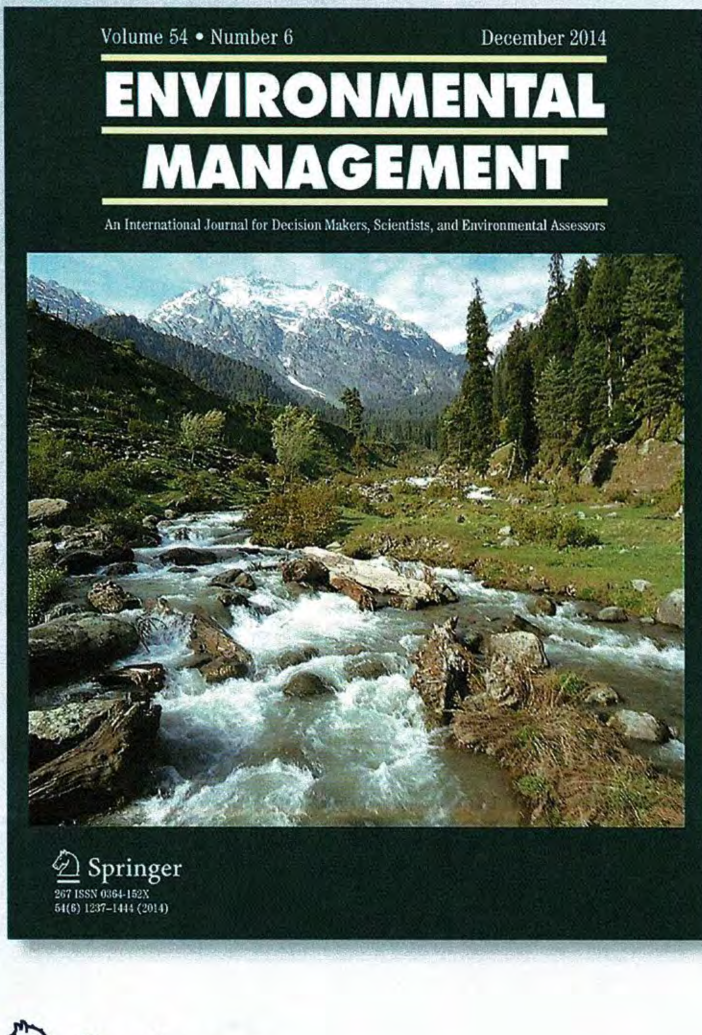

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Before and After Integrated Catchment Management in a Headwater Catchment: Changes in Water Quality

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Abstract Few studies have comprehensively measured the effect on water quality of catchment rehabilitation measures in comparison with baseline conditions. Here we have analyzed water clarity and nutrient concentrations and loads for a 13-year period in a headwater catchment within the western Waikato region, New Zealand. For the first 6 years, the entire catchment was used for hill-country cattle and sheep grazing. An integrated catchment management plan was implemented whereby cattle were excluded from riparian areas, the most degraded land was planted in *Pinus radiata*, channel banks were planted with poplar trees and the beef cattle enterprise was modified. The removal of cattle from riparian areas without additional riparian planting had a positive and rapid effect on stream water clarity. In contrast, the water clarity decreased in those sub-catchments where livestock was excluded but riparian areas were planted with trees and shrubs. We attribute the decrease in water clarity to a reduction in groundcover vegetation that armors stream banks against preparatory erosion processes. Increases in concentrations of forms of P and N were recorded. These increases were attributed to: (i) the reduction of instream nutrient uptake by macrophytes and periphyton due to increased riparian shading; (ii) uncontrolled growth of a nitrogen fixing weed (gorse) in some parts of the catchment, and (iii) the reduction in the nutrient attenuation capacity of seepage wetlands due to the decrease in their areal coverage in response to afforestation. Our findings highlight the complex nature of the water quality response to catchment rehabilitation measures.

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Keywords Land-use change · BACI · Catchment rehabilitation · Optical water quality · Phosphorus · Nitrogen

Introduction

Over the last 20–30 years improvements in catchment management, mainly in the form of rehabilitation measures such as riparian planting/fencing, erosion planting, and improved stock management, have become commonly applied methods for attempting to improve stream water quality and reducing the export of pollutants to downstream freshwater and marine receiving environments (Alexander and Allan 2007; Shah et al. 2007). Despite the popularity of these efforts, monitoring and reporting on their effectiveness is rare (Alexander and Allan 2007; Shields 2009). Among those studies that have reported their findings, planting of riparian buffers and/or the exclusion of stock from riparian areas are the most common catchment management/rehabilitation methods tested (McKergow et al. 2003; Sutton et al. 2010). Riparian buffers have been shown to be effective at reducing non-point source pollution in many agricultural catchments (e.g., Lee et al. 2003; Blanco-Canqui et al. 2004). Reductions in loads of sediment and fecal matter have consistently been shown to be one of the most effective outcomes of the removal of stock from within, and adjacent to streams (e.g., Williamson et al. 1996; McKergow et al. 2003; Muenz et al. 2006). The effectiveness of riparian buffers at reducing downstream concentrations and loads of different forms of nitrogen and phosphorus has, however, been shown to be variable (e.g., Burt et al. 1999; Wigington et al. 2003; McKergow et al. 2004). While riparian buffers can be effective in many environments and are a relatively easily implemented management tool they must still be considered a "last line of defense" with regards to the prevention of non-riparian zone sourced pollutants entering waterways.

A potentially more effective means of achieving improvements in stream water quality and reducing downstream export of sediment and nutrients is to take a "whole catchment" approach as advocated by proponents of integrated catchment management (ICM) (e.g., Walmsley 2002; Gregersen et al. 2007). While there is no precise definition of ICM, it is generally accepted to include the concept that water resources and other catchment ecosystem components (e.g., land, fauna, flora, and people) should be seen as connected and therefore managed holistically (Gregersen et al. 2007). Walmsley (2002) succinctly describes ICM as "managing for sustainable development at the catchment level, where water resources are viewed as the limited factor". As with many "single-approach" attempts at improving stream water quality (e.g., stock exclusion from riparian areas or riparian planting), there are also very few published accounts of the outcomes of the implementation of ICM plans on instream water quality using baseline data established prior to the interventions.

This study reports on 13 years (1995–2007) of monthly water quality monitoring results from an agricultural headwater catchment within the Waikato River catchment, North Island, New Zealand. In 2001 an ICM plan was implemented that significantly changed the land use. Accordingly, there are 6 years of data from before the land use changes, a transitional year (when the ICM plan was implemented), and 6 years of data from after the implementation of the land use changes. To our knowledge this is one of few studies that reports on the effect on instream water quality of an ICM plan in a headwater stream. The importance of headwater streams on downstream water quality is well appreciated (Alexander et al. 2007; Dodds and Oakes 2008). This study also appears to be one of the first to establish baseline water quality conditions over such a long period (6 years) before the implementation of the catchment management changes. The effect of the ICM plan on other aspects of the catchment, including the economic performance of the catchment as productive farm unit (Dodd et al. 2008c) and the effects on habitat and instream fauna and flora (Quinn et al. 2009), have been reported elsewhere.

Materials and Methods

Study Site and Integrated Catchment Management Plan

This study examines the long-term monthly water quality monitoring record of the 268 ha Mangaotama Stream catchment, a headwater stream within the Waikato River catchment, North Island, New Zealand (Fig. 1). From 1995 to 2000 the Mangaotama catchment was 100 % ryegrass-clover pasture and was a rotationally grazed mixed sheep and beef cattle grazing system. In 2001 an ICM plan was implemented within the catchment. Over the same period monthly water quality measurements were obtained from two control catchments, Whakakai Stream and Kiripaka Stream (Fig. 1). The Whakakai catchment (311 ha) is entirely vegetated in indigenous forest and has been relatively undisturbed by human activities for the last 80 years. The Kiripaka catchment (266 ha) is a mixed land use catchment that is part of the same farm unit as the Mangaotama catchment. All three catchments are dominated by steep to hilly topography, comprised of Mesozoic sedimentary sandstones and siltstones (graywacke and argillite) which have developed yellow brown earth soils (Quinn and Stroud 2002). The climate is humid temperate with a mean annual rainfall of 1,663 mm (1993–2010; NIWA, unpublished data for Whatawhata Research Station climate station) and a mean average temperature of 13.7 °C. For a more detailed physiographic and hydrological description

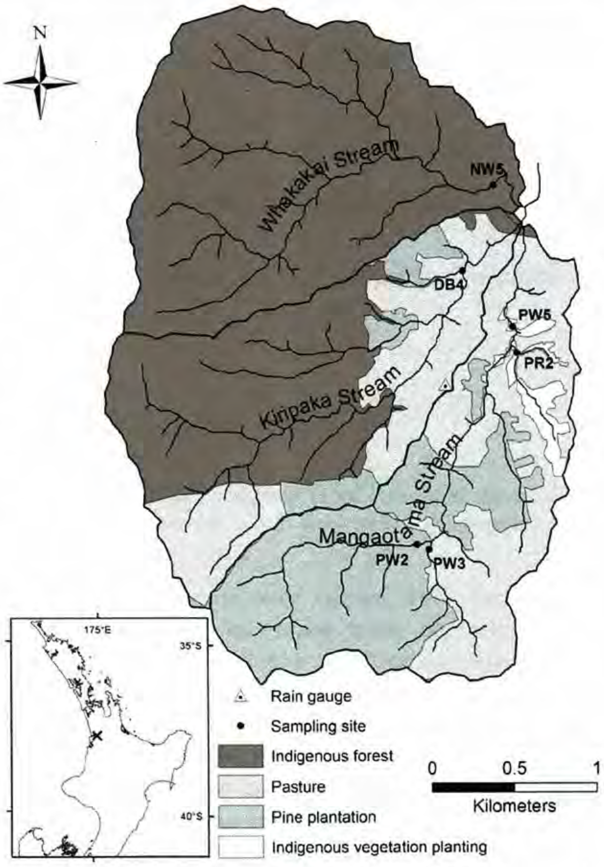

Fig. 1 Location map of study area showing the position of sampling sites and current (post-integrated catchment management plan) land cover. Sites DB4 and NW5 are control (C) sites

Table 1 Characteristics of study catchments

Sampling site	Catchment	Area (ha)	Median flow (L s^{-1})	95th percentile flow (L s^{-1})	Pre-ICM land use: %pasture:%pine:%indigenous	Post-ICM land use: %pasture:%pine:% indigenous
PW2	Mangaotama	95	n/a	n/a	100:0:0	0:100:0
PW3	Mangaotama	49	n/a	n/a	100:0:0	63:36:1
PW5	Mangaotama	268	40	219.1	99:0:1	38:58:4
PR2	Mangaotama	27	n/a	n/a	93:2*:5	70:0:30
DB4 (C)	Kiripaka	266	42	213.6	42:0:57	37:6:57
NW5 (C)	Whakakai	311	47	266.9	0:0:100	0:0:100

* Eucalypt not pine

of the study catchments refer to Dodd et al. (2008a) and Quinn and Stroud (2002).

Throughout the study period the Mangaotama catchment was a mixed sheep and beef cattle enterprise stocked at about nine stock units (SU) per hectare with a sheep to cattle SU ratio of 60:40 (Dodd et al. 2008b). In 2000–2001 an ICM plan was implemented in the catchment. Poplar trees (*Populus deltoides*) were planted for erosion control during 2000 in erosion prone parts (mainly stream banks) of the area that remained in pasture. During August 2001 153 ha were planted in *Pinus radiata*, with the exception of a 10 m unplanted buffer on each side of the stream channels. In the lower reaches of the catchment, indigenous tree and shrub species were planted across 7 ha of the existing pastoral land, surrounding and linking an existing 5 ha of indigenous riparian forest patches (Fig. 1). Exclusion of all livestock from waterways was achieved for areas converted to pine forestry and areas of indigenous tree planting. Cattle, but not sheep, were excluded from riparian areas within the remaining pasture land. Although all livestock have been excluded from pine afforested areas, feral pigs use forested areas as cover and stream bank/wetland disturbance caused by wallowing and rooting has been observed in the pine forest areas. In addition to the aforementioned catchment rehabilitation measures, the beef herd was changed from Angus beef breeding to a Friesian-cross bull beef enterprise with animals brought in during autumn at 6 months old and sold at 18 months. The significance of this change is that it resulted in the smaller cattle being present in the catchment over the wetter winter period when rainfall is high and grass growth is limited.

Within the Mangaotama catchment there were four water quality sampling locations (PW2, PW3, PW5, and PR2; Fig. 1; Table 1). Prior to the ICM changes, the land use in the catchments of all four sampling sites was almost entirely pasture grazing. The ICM plan resulted in the catchment of PW2 being planted entirely in *P. radiata* (with the exception of the 10 m riparian buffer). At PW3, the catchment was converted to 36 % *P. radiata*, 1 % indigenous forest, and 63 % pasture grazing, while PW5 was converted to 58 % pine, 38 % pasture, and 4 % indigenous forest (mainly in riparian areas). Large reaches of the stream banks within the catchment of PW3 and PW5 had poplar trees planted on them In the PR2 catchment, a small area (~1 ha) of eucalypt trees was cleared and riparian planting (referred to above) was carried out using indigenous tree and shrub species. Small gaps in the livestock exclusion occurred along *c.* 100 m of stream located 500 m upstream of PW5 where the farm access track crossed the stream. Livestock were also able to access the unfenced lower true right tributary of PR2 that flowed through a steep gully with remnant native forest, comprised ~20 % of the catchment.

The Kiripaka catchment is a mixed-land-use catchment with a combination of grazing, pine plantation, and indigenous forest occurring (Table 1). Two land use changes occurred within the Kiripaka catchment in 2001: (i) 8 ha (<3 % of the catchment) of mature *P. radiata* forest was clear felled (this was replanted in 2002), and (ii) the cattle enterprise was modified as outlined for the Mangaotama catchment above. The Kiripaka catchment is considered to be a near-control catchment as the land use changes were either minor (i.e., change of cattle enterprise) or only affecting a small area of land (i.e., tree harvesting and replanting). The Kiripaka catchment was sampled near its confluence with the Mangaotama Stream (site DB4; Fig. 1).

The Whakakai catchment is entirely vegetated in indigenous re-growth indigenous Podocarp/broadleaf forest which was left to regenerate in the early twentieth century. Livestock do not have access to the forest, although feral pigs are present and some rooting and wallowing damage near channels has been observed. The Whakakai stream is therefore a control catchment. The Whakakai catchment was sampled near its outlet (site NW5; Fig. 1).

Data Collection and Laboratory Analysis

Monitoring of a number of measures of water quality has occurred continuously from 1995 to 2007 (inclusive). Water quality data are also available for all sites from 2008 to 2012; these data were not included in this paper because from 2008 to 2011 the pine plantation parts of the study catchment (~60 % of catchment area) underwent a number of cycles of thinning and pruning. This thinning and pruning is likely to have significantly affected some elements of water quality. Another manuscript that presents the results for this period is planned.

With the exception of site PR2 (where sampling commenced in September 2000) water quality samples were collected from each site on a monthly basis beginning in April 1995. Sampling took place regardless of flow/weather conditions and occurred at approximately the same time during each visit. This type of sampling tends to be biased toward baseflow conditions, particularly in small catchments where the duration of most flow events is short and the probability of a monthly visit intercepting high flows is low (Letcher et al. 1999).

Once collected, samples were immediately placed in an insulated storage bin containing an ice slurry. Samples were delivered to the NIWA—Hamilton Water Quality Laboratory on the day of collection. Samples were analyzed for oxidized N (hereafter referred to as *nitrate-N*), ammonium-N, total nitrogen (TN), dissolved reactive phosphorus (DRP), and total phosphorus (TP). Total organic nitrogen (TON) was approximated by subtracting nitrate-N and ammonium-N from TN. All ammonium-N, nitrate-N, and DRP samples were filtered with a Millipore® syringe and filter holder containing a GF/C glass fiber prefilter (47 mm diam., 1.2 μm pore size), and a Sartorius® cellulose acetate membrane filter (47 mm diam., 0.45 μm pore size). Details of all laboratory nutrient analyses and detection limits are presented in Table 2. During monthly site visits measurements of water clarity using the black disk visibility method (Davies-Colley 1988) were also made.

Table 2 Laboratory methods and detection limits for water sample nutrient analysis

Analyte	Method	Detection limit (μg L^{-1})
Nitrate-N	Lachat flow injection analyzer	1
Ammonium-N	Lachat flow injection analyzer	1
Total Kjeldahl N	Indophenol blue colorimetry (acid digested)	10
Total nitrogen (1995–2006)	Summing nitrate-N and total Kjeldahl N	10
Total nitrogen (2007)	Lachat flow injection analyser (digested with persulfate and reduced with cadmium)	10
Dissolved reactive phosphorus	Lachat flow injection analyzer	1
Total phosphorus	Molybdenum blue flow injection analyzer (persulfate digested)	1

Annual nutrient loads were calculated using the monthly grab sample data using the averaging estimator of Preston et al. (1989):

$$\text{Annual load} = k\sum_{m=1}^{12}\sum_{j=1}^{Nm} q_{jm}\left[\sum_{i=1}^{n_m}\frac{C_{ijm}}{n_m}\right], \tag{1}$$

where k = unit conversion factor, n = number of days sampled, N = total number of days, q_{jm} = total flow (L) during the sampling interval (as determined from the continuous flow record) jm, C_{ijm} = monthly concentration (μg L^{-1}). There are a large number of load estimation methods available for estimating loads from limited datasets, this method has been shown to be at least as reliable as other commonly used ones (Preston et al. 1989).

In addition to the above monthly grab samples, an automatic water sampler (at PW5; Manning model 4901) collected samples at 4.5-h intervals into bottles containing mercuric chloride preservative, with eight samples combined in each bottle. The bottles were collected at monthly intervals and combined in proportion to the flow recorded at the site during the filling of each bottle to make a single sample for the month. These samples were analyzed for a number of water quality parameters, however, due to problems with the sampler intake location (resulting coarse bed material being sampled) only dissolved nutrient parameters (nitrate-N and DRP) are reported for the automatic samplers.

$$\text{Annual load} = k\sum_{m=1}^{12} q_m C_m, \tag{2}$$

where k = unit conversion factor, m = month, q_m = monthly flow volume (L), C_m = monthly concentration (μg L^{-1}) as determined by 4.5 hourly composite sampling.

At PW5, DB4 and NW5 stage height has been recorded continuously (at 15 min intervals) since at least the start of the water quality monitoring (1995). Stage height was measured by NIWA Hydrologger water level recorders (1 mm resolution). PW5 and DB4 have composite rectangular weirs while NW5 has a bedrock control immediately upstream of a small waterfall. All sites have stage/discharge ratings that have been determined by manual gagings over a range of water levels. No continuous flow data are available for the PW2, PR2, and PW3 sampling sites.

Statistical Analysis

For the purpose of determining any differences and/or trends in the various water quality variables, the data were divided into a "pre-ICM" period and a "post-ICM" period. The pre-ICM period was the period between April 1995

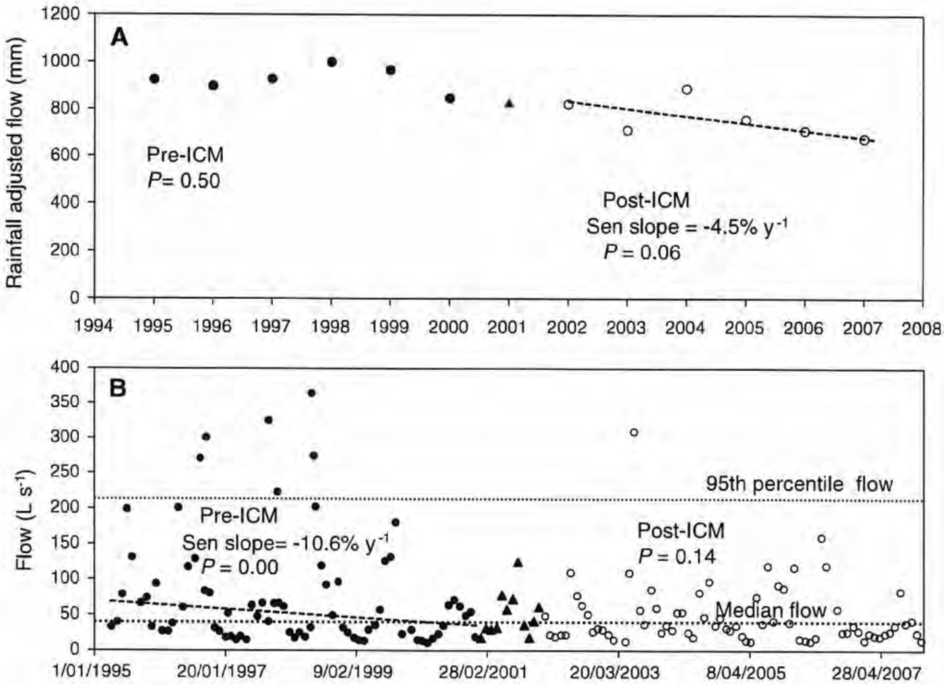

Fig. 2 **a** Rainfall-adjusted mean annual flow data for the pre-ICM (1995–2000; *solid circles*) and post-ICM (2002–2007; *hollow circles*) periods at PW5. **b** Instantaneous flow data (at the time of water quality sampling) at PW5 (April 1995–December 2007) (*dotted lines* indicate median (40 L s^{-1}) and 95th percentile (213 L s^{-1}) flows for PW5. *Dashed lines* are the relative seasonal Kendall Sen slope estimates (RSKSSE). *P* values were determined by the Mann–Kendall trend test

and December 2000 (5 years, 9 months duration). The post-ICM period was the period from January 2002 and December 2007 (6 years duration). The data from 2001 were excluded from the statistical analysis as this was considered to be a transitional year, during which many of the catchment land use changes took place.

The non-parametric Mann–Whitney test was used to test for differences in the median water quality values of the pre- and post-ICM periods. We selected the Mann–Whitney test in preference to a parametric approach (e.g., ANOVA) because water quality data are seldom normally distributed and is often affected by data censoring (e.g., reporting of minimum detection limits). Parametric tests lack statistical power when applied to non-normal data and are not appropriate to apply to censored datasets (Helsel and Hirsch 1992). The Mann–Whitney test presents the null hypothesis that there is no difference between the pre- and post-ICM median water quality parameter values. The alternative hypothesis is that the samples come from different distributions, or there is a difference in the median values.

The non-parametric Seasonal Kendall test was used to test separately for trends in pre- and post-ICM periods. The Seasonal Kendall test was preferred over linear regression as the latter does not account for the strong seasonal variability that is often exhibited in monthly water quality data, thus its ability to detect trends can be reduced (Smith et al. 1996). The Seasonal Kendall test poses the null hypothesis that there is no trend. The alternative hypothesis is that there is in fact a positive or negative trend.

The relative seasonal Kendall Sen slope estimator (RSKSSE) was used to assess the magnitude of trends as it allows direct comparison between sites. The seasonal Kendall Sen slope estimator (SKSSE) can be considered to be a non-parametric analogy of ordinary least squares regression (Helsel and Hirsch 1992) and trends are reported as determinand units per year. The RSKSSE is the ratio of the slope estimate for a site to the median determinand value for that site (reported as % y^{-1}) (Smith et al. 1996). Detailed descriptions of both the Seasonal Kendall Test and the RSKSSE are presented in Helsel and Hirsch (1992) and Smith et al. (1996).

Many water quality variables are dependent to some degree on river flow and it is generally accepted that the effect of flow should be removed before any trend analysis is carried out (Hirsch et al. 1982; Smith et al. 1996; Helsel and Hirsch 2002). We have therefore adjusted the water quality variable values using the adjustment procedure (power law used) of the non-parametric Mann–Kendall test (see Hirsch et al. 1982 and Helsel and Hirsch 2002). Flow adjustment removes the variation in measured water quality variables caused by different flow conditions at time of sampling. Removal of this variation reduces the background variability, therefore any trend present can be detected with more power (Helsel and Hirsch 2002). Helsel and Hirsch (2002) advise that flow adjustment should not be used where human activity has altered the probability distribution of discharge. For the PW5 site, it could be argued that there has been human intervention (e.g., afforestation) that has affected the flow and this is apparent

Table 3 Median pre-ICM (1995–2000) and post-ICM (2002–2007) water quality variable values for the six sampling sites

Variable	Water quality sampling site											
	PW2		PW3		PW5		PR2		NW5		DB4	
Raw data	Pre-ICM	Post-ICM	Pre-ICM	Post-ICM	Pre-ICM	Post-ICM	Pre-ICM	Post-ICM	Pre-ICM	Post-ICM	Pre-ICM	Post-ICM
n	66	65	66	65	64	66	4	66	59	68	64	66
Flow (L s^{-1})	N/A	N/A	N/A	N/A	*40.6 (−10.6)*	34.8	N/A	N/A	39.6	42.1	38.8	42.7
Water clarity (m)	0.53	**0.65**	0.59	**0.90**	0.75	**1.01**	1.01	0.90 (−6.7)	0.99	**1.35**	0.64	0.72
DRP (μg L^{-1})	20	**29 (8.0)**	*28 (−5.3)*	**32**	13	14 (3.9)	11	14 (8.4)	41	41	*27*	28
TP (μg L^{-1})	46	**59 (2.5)**	61	55	38	40	31	40	50	52	47	49
NO_3-N (μg L^{-1})	415	**801 (7.3)**	714	848	335	**564**	86	173 (5.6)	101	105	309	375
NH_4-N (μg L^{-1})	13	11	17	**11**	11	10	8	8 (−4.1)	3	3	8	7
TON (μg L^{-1})	189	**158**	211	**136**	192	**122**	123	111	72	**42**	*143 (−13.1)*	**99**
TN (μg L^{-1})	640	**911**	1050	1029	510	**691**	210	305	182	**155**	453 (−10.6)	511
Flow-adjusted												
Water clarity (m)	0.50	**0.65 (−6.2)**	0.59	**0.91**	0.75	**1.03**	N/A	0.90 (−7.8)	0.97	**1.33**	0.63	0.73
DRP (μg L^{-1})	20	**29 (8.1)**	*29 (−6.2)*	**32**	13	14 (4.2)	N/A	13 (7.8)	42	41	*28*	28
TP (μg L^{-1})	44	**58 (2.2)**	61	**53**	36	39	N/A	34	49	52	45	47
NO_3-N (μg L^{-1})	*410 (5.4)*	**842 (7.2)**	705	**865**	324	**571 (5.5)**	N/A	166 (12.3)	104	106	301	369
NH_4-N (μg L^{-1})	12	11	13	11	11	10	N/A	8 (−4.2)	3	3	8	7
TON (μg L^{-1})	178	149	191	**122**	181	**115**	N/A	94	72	**42**	*140 (−10.6)*	**91**
TN (μg L^{-1})	638	**929 (4.1)**	1023	1046	499	**691 (4.0)**	N/A	305 (7.7)	182	**148**	453 (−5.4)	492

Values in bold type indicate that that the median post-ICM value was significantly different from the pre-ICM value (Mann–Whitney test; $\alpha = 0.05$). Numbers in parentheses are the relative seasonal Kendall Sen slope estimates (RSKSSE; % y^{-1}) where there was a significant ($\alpha = 0.05$) trend in the data. Italicized values indicate there was a significant and meaningful trend in the pre-ICM data. A meaningful trend is defined as a RSKSSE greater than 1 % y^{-1}. Cells shaded dark gray indicate a "gradual change". Cells shaded light-gray indicate a 'step change'. Because of the limited number of pre-ICM samples no Mann–Whitney or pre-ICM Seasonal-Kendall tests were carried out for PR2 in the rainfall-adjusted mean annual flow at PW5 between 2001 and 2007 (Fig. 2a). Rainfall-adjusted mean annual flow is mean annual flow that has had the effect of any potential trend in annual rainfall removed by the linear adjustment procedure of the Mann–Kendall test outlined above. Despite this reduced flow at PW5, there is no significant trend in the flow measurements at the time the samples were collected (Fig. 2b). Therefore, while flow is likely to have reduced in response to the catchment land use changes, there is no significant trend in the data used to flow-adjust the water quality variables. We therefore have confidence that the flow-adjusted data are not reflecting any trend in the flow data. Nevertheless, to avoid doubt, the results for both the raw and the flow-adjusted water quality data are presented in Table 3.

As there were no flow data available for PW2, PW3, or PR2, and the fact that all three sites are located upstream of the PW5 hydrometric station, the flow data for PW5 were used to flow-adjust the water quality data for these sites.

Results

Water Clarity and Nutrient/Sediment Concentration Data

The pre- and post-ICM median water quality variable values and the results from the statistical tests are summarized in Table 3 and Fig. 3. For both the results of the Mann–Whitney test and the Seasonal Kendall test, statistical significance is inferred when the P value of a test is less than 0.05.

Site PW2

Site PW2, which was planted in 100 % (excluding 10 m riparian buffers) pine forest, experienced a number of significant changes in water quality. There were no statistically significant differences between the pre- and post-ICM median concentrations of both TON and ammonium-N. There was some change in the range of ammonium-N concentrations in the post-ICM period with fewer high outliers and a contraction in the interquartile range (Fig. 3a). The median water clarity at PW2 increased from 0.5 m in the pre-ICM period to 0.65 m in the post-ICM period (Fig. 3a). More detailed assessment of the water clarity time series (Fig. 4a) shows that water clarity trend during the pre-ICM period was non-monotonic with clarity initially declining (1995–1998) then increasing (1998–1999). Of particular note is that, despite the apparent improvements in water clarity (as illustrated in Fig. 3a), there was a negative monotonic trend of -6.2 % y^{-1} during the post-ICM period (Fig. 4a).

Significant increases in the concentrations of DRP, TP, nitrate-N, and TN were detected at PW2 (Table 3; Figs. 3a, 4). The median DRP concentration increased from 20 μg L^{-1} in the pre-ICM period to a post-ICM median concentration of 29.0 μg L^{-1}. This increase in DRP also appears to account for most of the increase in TP (44–58 μg L^{-1}). The increase in DRP concentration occurred gradually, with a significant trend of 8.1 % y^{-1} detected over the post-ICM period (Table 3; Fig. 4b). The median concentration of nitrate-N increased by ~100 % from 410 to 842 μg L^{-1} (Table 3; Fig. 3). As with the increase in concentration of DRP and TP, the increase in nitrate-N accounts for most of the increase in TN also (638–929 μg L^{-1}). The increase in nitrate-N concentration occurred gradually over the post-ICM period with a trend of 7.2 % y^{-1} detected over the post-ICM period (Table 3; Fig. 4c). Interestingly, there was also a positive trend of 5.4 % y^{-1} in nitrate-N concentration during the pre-ICM period.

Site PW3

At site PW3, which had cattle excluded from riparian areas, poplar tree planted on stream banks and 40 % of its catchment planted in pine the median concentration of TON (191–122 μg L^{-1}; Fig. 3b) decreased significantly. The median water clarity improved by over 50 % from 0.59 to 0.91 m (Fig. 3b), although there was no trend in the post-ICM water clarity data (Fig. 5a). A similar non-monotonic trend as that measured at PW2 also occurred in the pre-ICM water clarity data, with clarity decreasing from 1995 to 1998 then increasing from 1998 to 1999 (Fig. 5a).

As with PW2, there was a significant difference between the pre- and post-ICM median concentrations of DRP at PW3 (Table 3; Fig. 3b). However, unlike PW2, DRP concentrations increased only marginally (29–32 μg L^{-1}) with no significant monotonic trend in the post-ICM data (Fig. 5b). There was, however, a negative trend (-6.2 % y^{-1}) in the pre-ICM DRP data. Despite the small increase in DRP, TP actually decreased by ~15 % (61–53 μg L^{-1}). Also similar to PW2, the median nitrate-N concentration at PW3 increased (but to a more limited extent), with the median increasing from 705 to 865 μg L^{-1} (Fig. 3b). There was no trend in post-ICM nitrate-N concentration data (Fig. 5c).

There were no statistically significant differences between the pre- and post-ICM median concentrations of TN.

Site PW5

At site PW5, which is near the outlet of the experimental Mangaotama catchment (downstream of PW2 and PW3), the median water clarity improved by ~40 % (0.75–1.03 m)

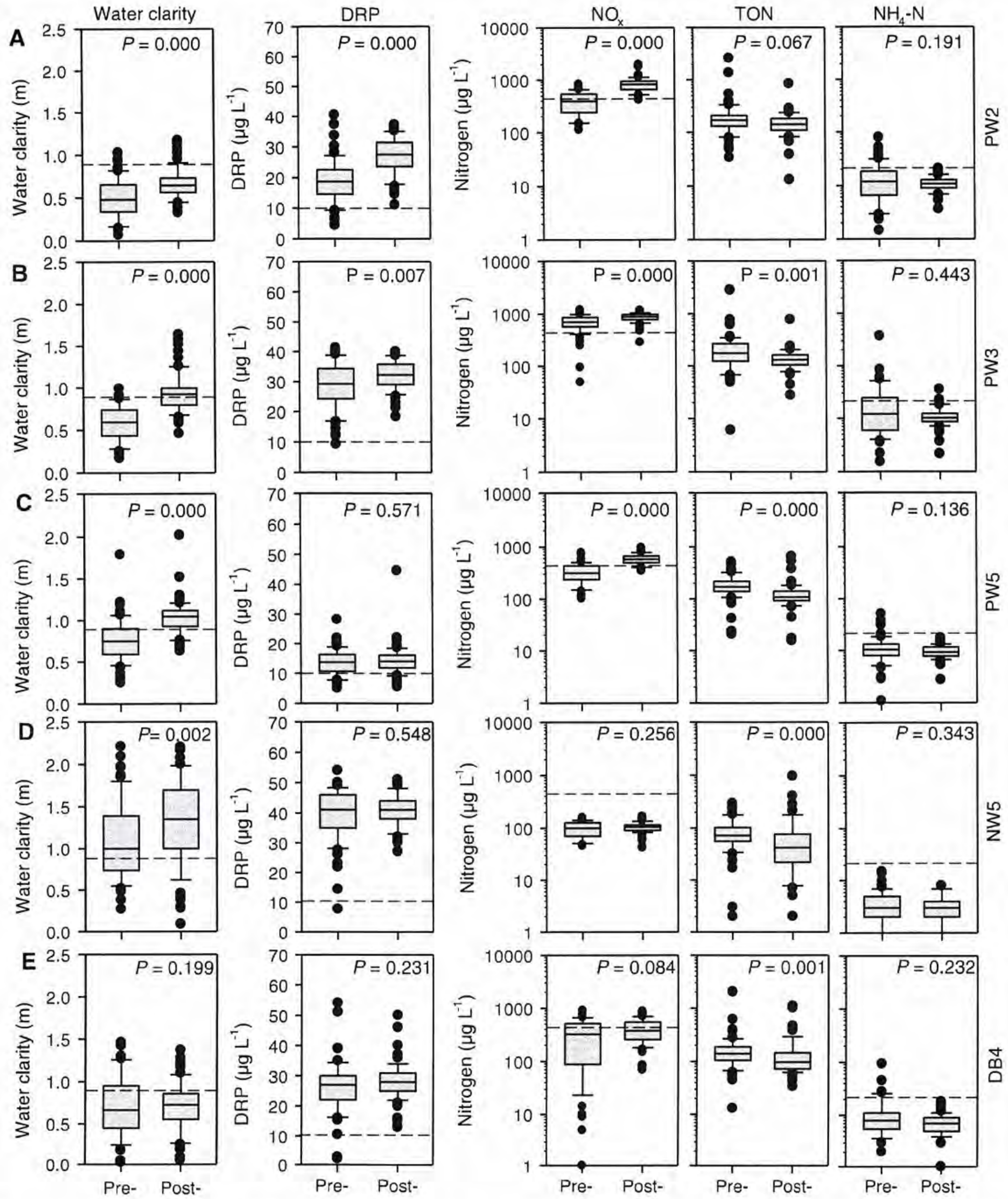

Fig. 3 *Box plots* of pre- and post-ICM (flow-adjusted) water clarity and concentrations of dissolved reactive phosphorus, nitrate-N, total organic nitrogen, and ammonium at **a** PW2, **b** PW3, **c** PW5, **d** NW5, **e** DB4. *P* values were determined by the Mann–Whitney *U* test. Site PR2 is not included as there were not enough pre-ICM data for any meaningful statistical analysis. The *bottom* and *top* of each box are the 25th and 75th percentiles. The *band inside* each box is the median. The *upper whisker* is the 90th percentile and the *lower whisker* is the 10th percentile. The *solid dots* are outliers. The *horizontal dashed line* indicates the 'trigger values' as identified by the Australian and New Zealand Guidelines for Fresh and Marine Water Quality ANZECC (2000). Trigger values are the 80th percentiles of 10 years of monthly data from three baseline sites (Davies-Colley 2000) that, if exceeded, would indicate preliminary benchmarks for relatively unimpacted New Zealand rivers have been crossed

(Table 3; Fig. 3c) although there was no trend detected in the post-ICM data (Fig. 6a). The median concentration of TON decreased by ~60 % (181–115 µg L^{-1}; Fig. 3c). The median concentrations of both nitrate-N and TN experienced positive trends in the post-ICM period. Nitrate-N concentration increased from 324 to 571 µg L^{-1} (Fig. 3c) at an average

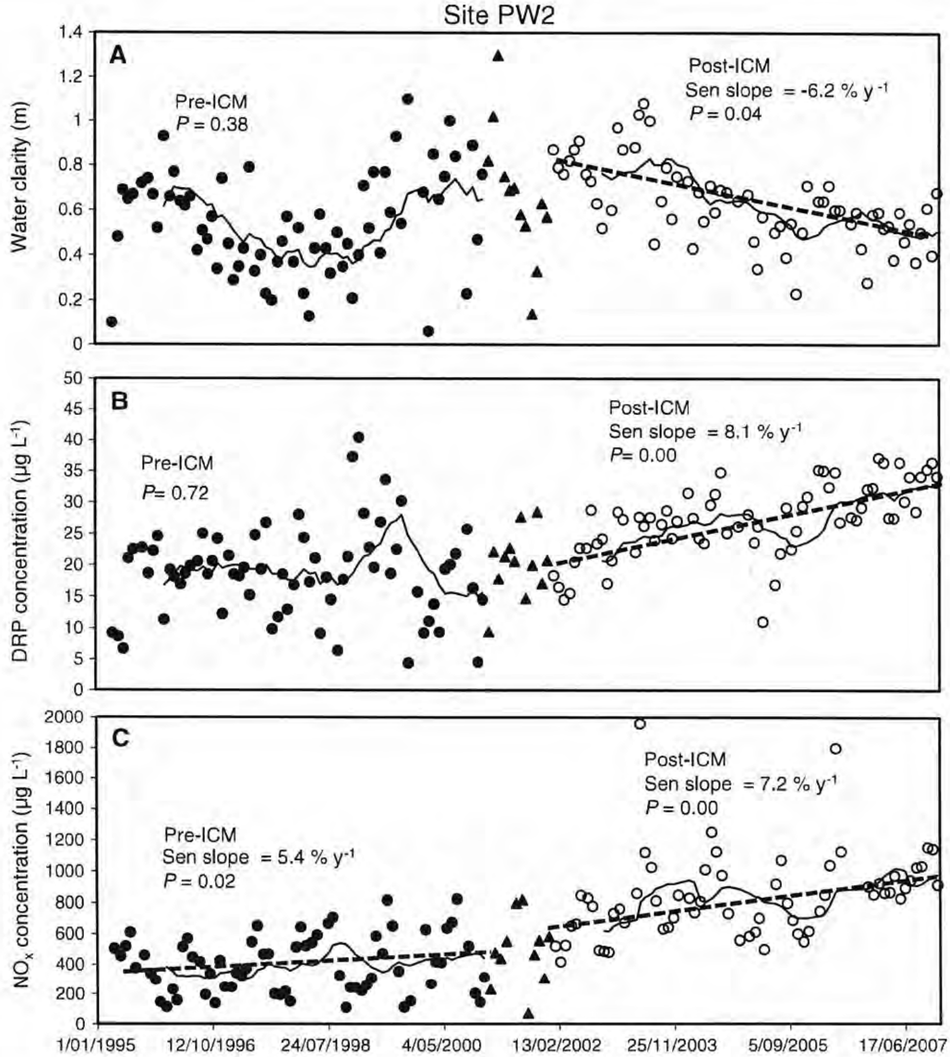

Fig. 4 Flow-adjusted (using PW5 flow data) monthly measurements of **a** water clarity, **b** dissolved reactive phosphorus (DRP), and **c** nitrate-N (NO_x) at PW2 (April 1995–December 2007). *Solid circles* = pre-ICM data; *solid triangles* = transitional year (2001); *hollow circles* = post-ICM data. *Solids lines* are 10 point running mean average and *dashed lines* are the relative seasonal Kendall Sen slope estimates (RSKSSE). *P* values were determined by the Seasonal Kendall test

rate of 5.5 % y^{-1} (Fig. 6c), while median TN concentration increased from 499 to 691 μg L^{-1} at an average rate of 4.0 % y^{-1}. There were no statistically significant differences between the pre- and post-ICM median concentrations of DRP (Fig. 3c), TP, and ammonium-N (Fig. 3c). There was, however, a significant positive trend (4.2 % y^{-1}) in the post-ICM DRP data (Fig. 6b). There were fewer high outliers and a contraction in the inter-decile range in ammonium-N concentrations in the post-ICM period (Fig. 3c).

Site PR2

At PR2, where the main ICM changes were stock exclusion and riparian planting with indigenous tree and shrub species, only four samples were acquired during the pre-ICM period. Accordingly, there were not enough data to carry out any meaningful statistical analysis of the pre-ICM data. Testing for differences in the median pre- and post-ICM concentrations was also not possible. Seasonal-Kendall tests were, however, possible for the post-ICM data (Table 3). Significant trends were detected in six out of the nine measured variables. Water clarity became gradually more degraded through the post-ICM period, with it reducing at a rate of 7.8 % y^{-1} (Fig. 7a). DRP increased at rate of 7.8 % y^{-1} (Fig. 7b), although no trend was detected in TP. Both nitrate-N and TN illustrated positive trends, with nitrate-N increased at a rate of 12.3 % y^{-1} (Fig. 7c) while TN increased at 7.7 % y^{-1}.

Also of note is that the post-ICM concentrations of most variables at PR2 are conspicuously lower than those from PW2, PW3, and PW5 (Table 3). Of particular note is nitrate-N which is 3–5 times lower (166 μg L^{-1}) than that from the other three experimental sites (PW2: 842 μg L^{-1}; PW3: 865 μg L^{-1}; PW5: 571 μg L^{-1}).

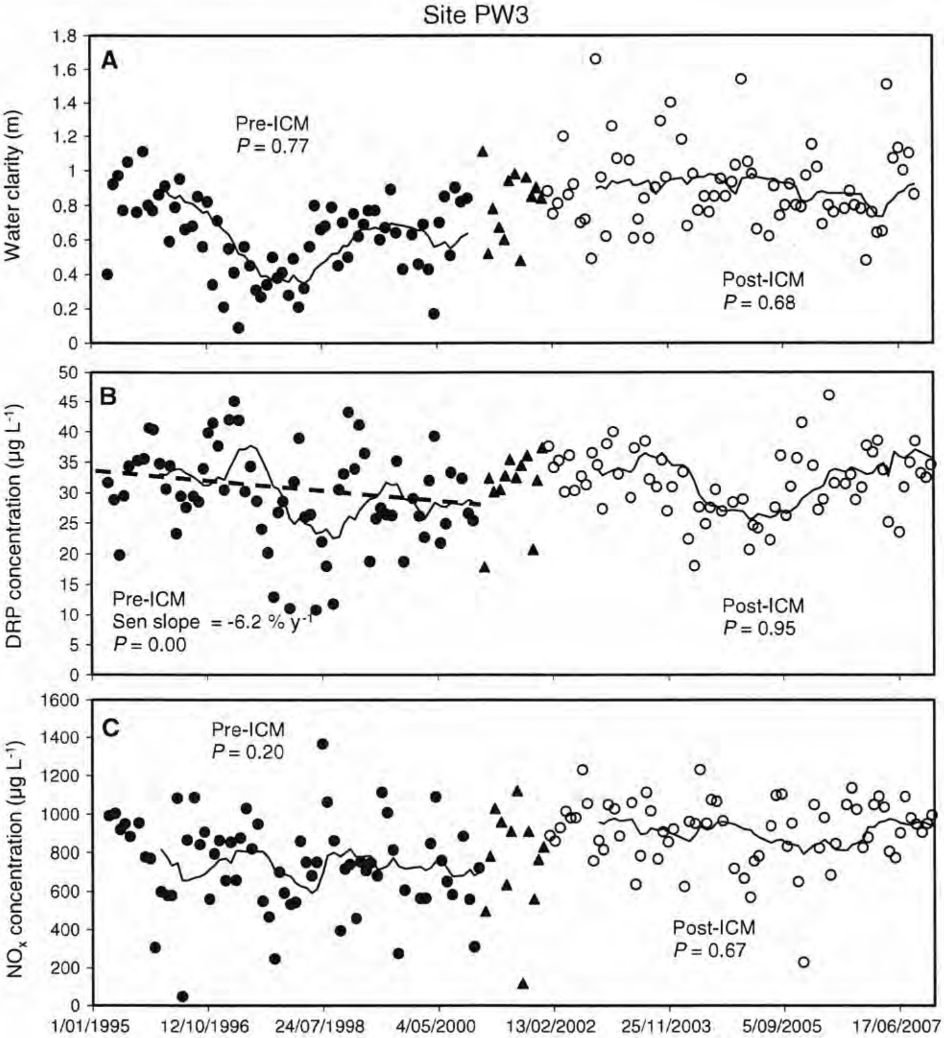

Fig. 5 Flow-adjusted (using PW5 flow data) monthly measurements of **a** water clarity, **b** dissolved reactive phosphorus (DRP), and **c** nitrate-N (NO_x) at PW3 (April 1995–December 2007). *Solid circles* = pre-ICM data; *solid triangles* = transitional year (2001); *hollow circles* = post-ICM data. *Solids lines* are ten point running mean average and *dashed line* is the relative seasonal Kendall Sen slope estimate (RSKSSE). *P* values were determined by the Seasonal Kendall test

Control Sites (NW5 and DB4)

At both the indigenous forest catchment control site (NW5) and the mixed land used catchment control site (DB4), there were few statistically significant differences in the water quality variables calculated for the pre- and post-ICM periods (Table 3). At NW5 water clarity improved by ~25 % (0.97–1.3 m), although the time series data indicated that water clarity appeared to fluctuate during both the pre- and post-ICM periods. The median concentration of TON also decreased by ~40 % from 72 to 42 µg L^{-1}, although there was no trend in either the pre- or post-ICM datasets. Because TON comprises a higher proportion of the TN at NW5 than the other (agricultural) sites a significant decrease in TN concentrations was also detected in median TN concentrations (182–148 µg L^{-1}).

The only water quality variable to have a significant difference between the pre- and post-ICM medians at DB4 was TON, which decreased by ~45 % from 140 to 91 µg L^{-1}. TON at DB4 was trending down at an average rate of −10.6 % y^{-1} during the pre-ICM period. There was no trend in the post-ICM TON dataset.

Dissolved Nutrient Loads

There is reasonably good agreement between the DRP and nitrate-N loads determined by the two sampling methods (Fig. 8a, b). Because of the general lack of sampling during the highest flows, and the moderately strong positive relationship between flow and nitrate-N at PW5 ($R^2 = 0.50$), both load calculation methods are likely to underestimate the true nitrate-N loads (see Stone et al. 2000). The close agreement between the two sampling methods gives us confidence in the relativity of the calculated loads. Although DRP does not appear to vary significantly with flow ($R^2 = 0.07$), the monthly composite

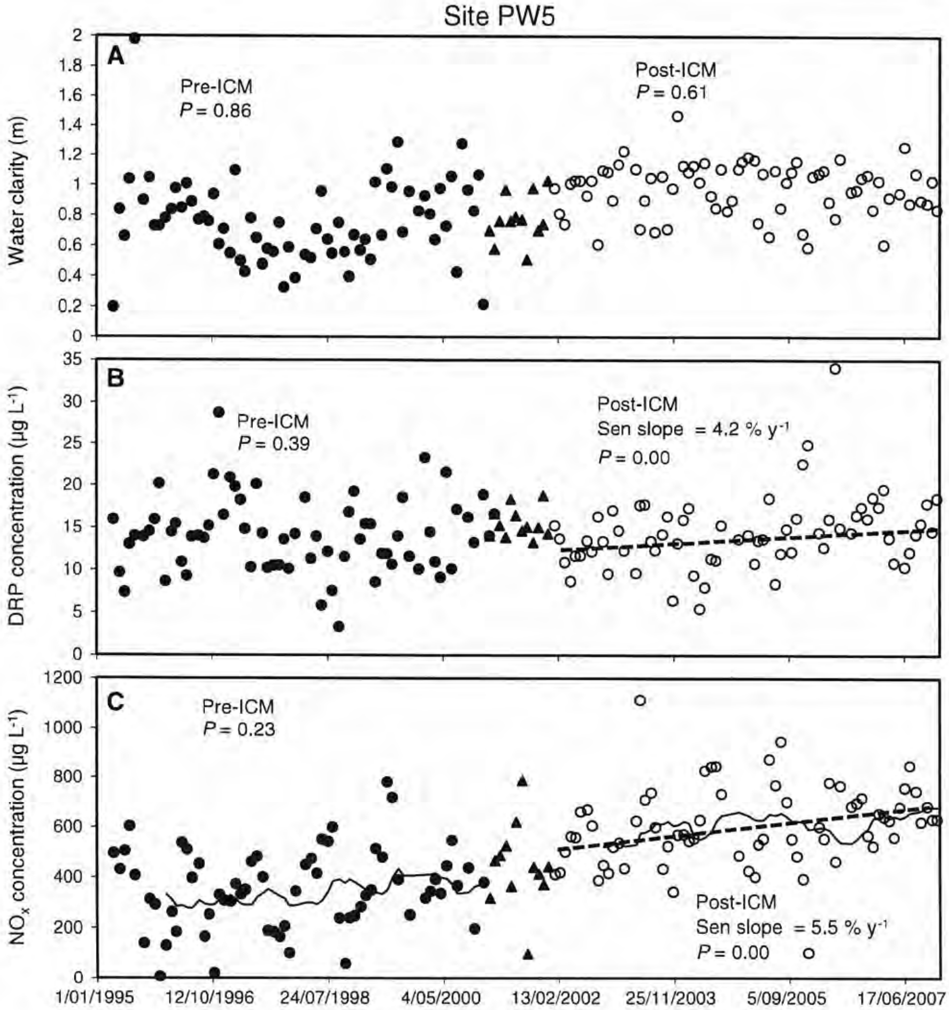

Fig. 6 Flow-adjusted monthly measurements of **a** water clarity, **b** dissolved reactive phosphorus (DRP), and **c** nitrate-N (NO_x) at PW5 (September 2000–December 2007). *Solid circles* = pre-ICM data; *solid triangles* = transitional year (2001); *hollow circles* = post-ICM data. *Solid lines* are ten point running mean average and *dashed lines* are the relative seasonal Kendall Sen slope estimates (RSKSSE). *P* values were determined by the Seasonal Kendall test

sampling derived loads are consistently higher than the monthly grab sampling derived loads. This suggests that above the flow range which has been currently sampled (by monthly sampling) there may be some change in the flow-DRP concentration relationship that results in relatively higher DRP concentrations for a given flow.

Using the Mann–Whitney test, no significant differences in the pre- and post-ICM median concentrations of either DRP ($P = 0.618$) or nitrate-N ($P = 0.631$) were detected. Furthermore, no significant trends (0.05 significance level) were detected in either the pre- or post-ICM periods for either load calculation method (Table 4). There was, however, a significant negative trend in DRP loads at the 0.1 significance level.

Discussion

Long-term studies that monitor the effects of changes in catchment management on water quality are rare. This "before and after" study monitored the effects on water quality of implementing an integrated management plan in a small headwater catchment in New Zealand over a 13-year period. Due to the relatively short agricultural history and low intensity land uses, the water quality of many New Zealand agricultural streams is good compared to many northern hemisphere countries. Despite the comparatively low levels of many pollutants, the elevated levels experienced in many agricultural catchments do have the potential to adversely affect the ecosystems of recently pristine streams.

The Australian and New Zealand Environment and Conservation Council and Agriculture and Resource Management Council of Australia and New Zealand formulated the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC 2000). These guidelines identify "trigger values" for measures of water quality within lowland (<150 m elevation) rivers in New Zealand (Table 5). These trigger values are the 80th percentiles of 10 years of monthly data from three baseline

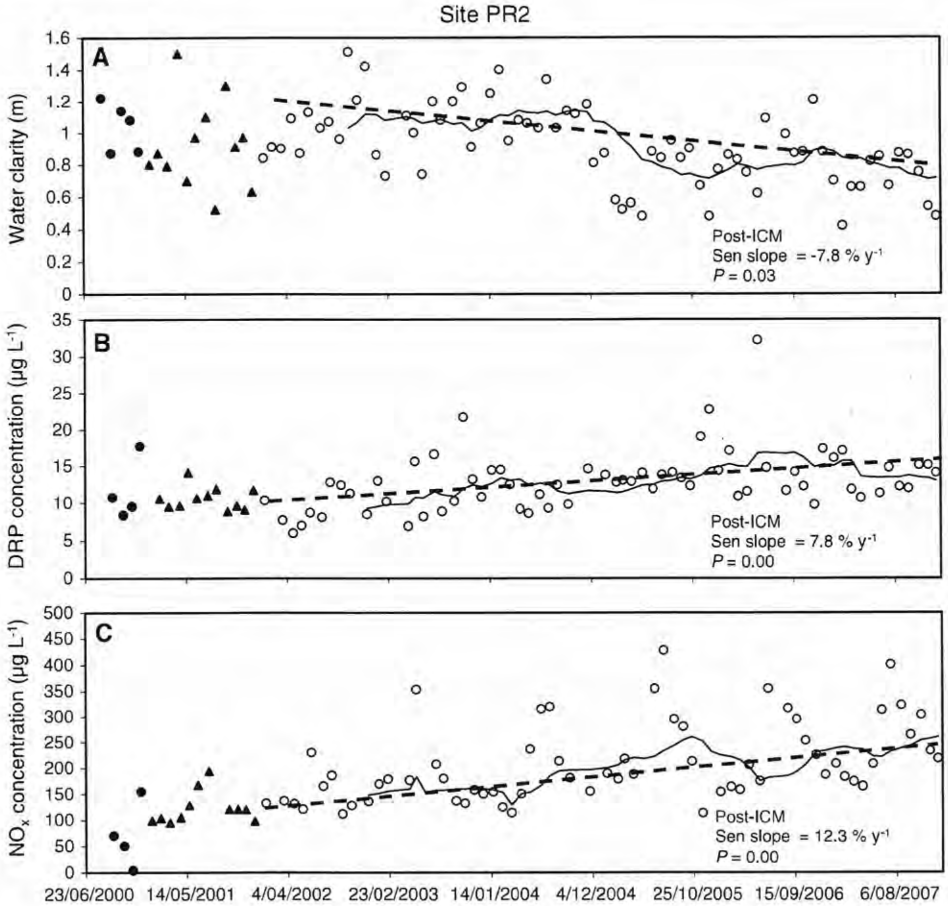

Fig. 7 Flow-adjusted (using PW5 flow data) monthly measurements of **a** water clarity, **b** dissolved reactive phosphorus (DRP), and **c** nitrate-N (NO_x) at PR2 (April 1995–December 2007). *Solid circles* = pre-ICM data; *solid triangles* = transitional year (2001); *hollow circles* = post-ICM data. *Solids lines* are ten point running mean average and *dashed line* is the relative seasonal Kendall Sen slope estimate (RSKSSE). *P* values were determined by the Seasonal Kendall test

sites (Davies-Colley 2000) that, if exceeded, would indicate preliminary benchmarks for relatively unimpacted New Zealand rivers have been crossed.

For sampling sites draining agricultural or former agricultural-based catchments (i.e., PW2, PW3, PW5, and DB4) the trigger values are exceeded for most measured parameters (ammonium-N being the notable exception) (Fig. 3). With the exception of DRP, all the measured parameters at the indigenous forest site (NW5) are within the guideline levels. The precise reason for the comparatively high levels of DRP at NW5 is unknown but it may be related to the some geological difference within its catchment.

One of the main catchment management changes implemented within the Mangaotama catchment was the removal of cattle access to, and planting (or natural regeneration) within, riparian areas. Reductions in sediment concentrations and loads have consistently been shown to be one of the most effective outcomes of the removal of stock from within, and adjacent to, streams. The improved water quality at PW3 and PW5 shows that this has also largely been the case in the Mangaotama catchment. However, at PW2, where the entire catchment was planted in pine and a 10 m riparian strip allowed to regenerate naturally, water clarity did not improve. Despite the detection of a statistically significant increase in water clarity at PW2 (between the pre- and post-ICM periods), water clarity trended downward during the post-ICM period. The statistical detection of an improvement in water clarity between the pre- and post-ICM periods at PW2 may be an artifact of the non-monotonic trend (of unknown cause) in pre-ICM water clarity. Furthermore, at PR2 where stock were excluded from 80 % of riparian areas and indigenous vegetation (trees and shrubs) was planted, there was a decline in water clarity. Although there is no meaningful pre-ICM data available, the post-ICM data at PR2 indicate a significant negative trend in water clarity.

This lack of improvement in the water clarity at PW2 and decline in water clarity at PR2 suggests that the stream

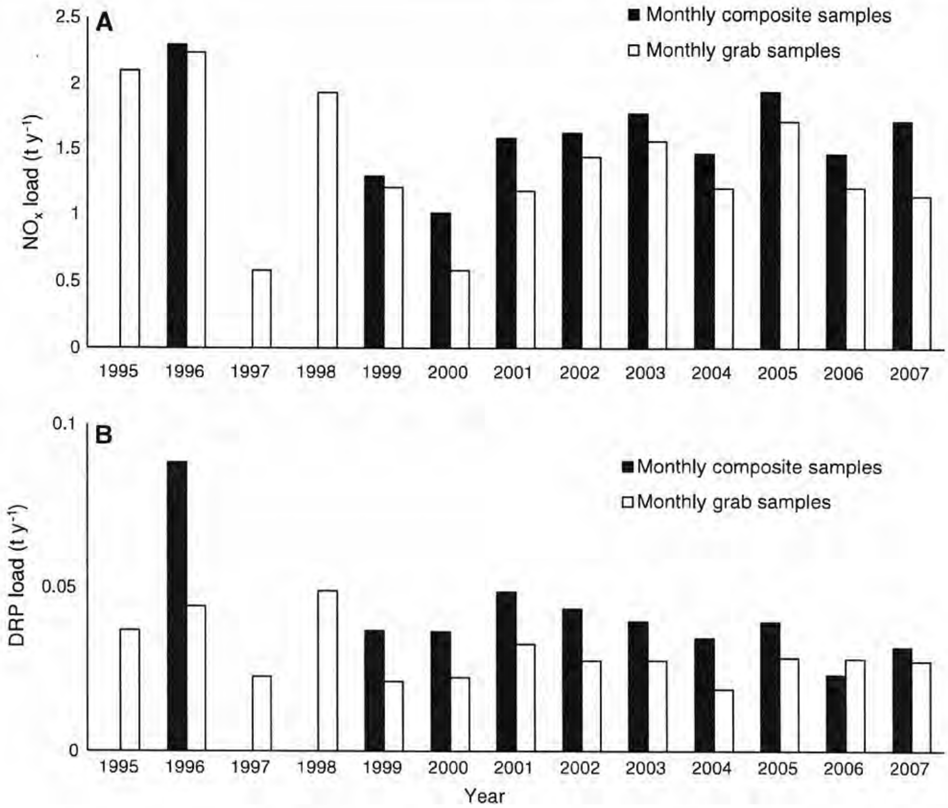

Fig. 8 Specific annual yields of **a** nitrate-N (NO_x) and **b** dissolved reactive phosphorus (DRP) as determined from monthly composite samples and monthly monitoring grab samples

Table 4 Mann–Kendall trend test *P* values for detection of trends in DRP and nitrate-N (NO_x) loads (determined from grab samples and composite samples) for both the pre-ICM and post-ICM periods

Period/load method	Mann–Kendall trend test *P* value	
	DRP	Nitrate-N
Pre-ICM		
Grab samples ($n = 6$)	0.45	0.26
Composite samples ($n = 3$)	0.54	0.30
Post-ICM		
Grab samples ($n = 6$)	0.50	0.45
Composite samples ($n = 6$)	0.09	0.50

Table 5 Water quality guideline levels (trigger values) for the protection of New Zealand lowland stream (<150 m elevation) ecosystems (ANZECC 2000)

Analyte and unit	Guideline value
Water clarity (m)	0.8
Dissolved reactive phosphorus ($\mu g\ L^{-1}$)	10
Total phosphorus ($\mu g\ L^{-1}$)	33
Ammonium-N ($\mu g\ L^{-1}$)	21
Nitrate-N ($\mu g\ L^{-1}$)	444
Total nitrogen ($\mu g\ L^{-1}$)	614

The levels are the 80th percentiles of 10 years of monthly data from three baseline sites (Davies-Colley 2000) that, if exceeded, would indicate preliminary benchmarks for relatively unimpacted New Zealand rivers have been crossed

banks may not have reduced supplying sediment to the same degree as they have in the pasture dominated subcatchment (PW3). The exclusion of cattle from riparian areas throughout the Mangaotama catchment in 2001 probably resulted in a relatively rapid reduction in sediment derived from degraded channels and near-channel sources. However, in those parts of the catchment where riparian areas were either planted or were allowed to regenerate naturally (i.e., catchments of PR2 and PW2, respectively), stream banks became shaded due to the growth of weeds, shrubs, and trees. In headwater streams such shading can result in increased sediment input from bank erosion due to the reduced viability of groundcover vegetation (such as grasses) that armor stream banks against erosion (Davies-Colley 1997; Trimble 1997). Evidence of this phenomenon within the Mangaotama catchment was presented by Hughes et al. (2012). They argued that the persistence of clock-wise hysteresis in the discharge-suspended sediment concentration relationship throughout the pre- and post-ICM periods indicated bank or near-bank sources dominated. The continued dominance of bank or near-bank sources, despite the removal of cattle from riparian areas, was attributed to the increased susceptibility to bank erosion due to the shading of groundcover vegetation. This argument is supported by previous work that found stream channels became progressively shaded as riparian areas that were retired from grazing naturally regenerated (Quinn et al. 2009). In headwater streams, stream power is usually insufficient to erode banks without some prior loosening of bank material (Lawler 1995; Abernethy and Rutherfurd 1998). The loss of dense grass groundcover on and near stream banks, as a result of ground level shading by riparian shrubs and weeds, may have resulted in increased loosening of bank material by preparatory erosion processes (e.g., desiccation, rainsplash, and micro-rill development of bare banks). Subsequent periods of elevated flow can easily transport this loosened bank material (Prosser et al. 2000). Although the effect of decreased resistance to erosion would be of most significance during flow events, it is possible that there is some on-going residual effect on water clarity and this has been detected by the monthly water quality sampling.

In an attempt to improve stream bank stability, poplar trees were planted on stream banks within the catchments of PW3 and PW5 (excluding the pine afforested PW2 catchment). Poplar trees have been used extensively throughout New Zealand to reduce erosion, including stream bank erosion (Wilkinson 1999). Improvements in water clarity at PW3 and PW5 occurred in a step change manner. Such a change (where there was a significant difference between the pre- and post-ICM median values but no significant trend was detected in the post-ICM data) suggests that the improvements occurred soon after the implementation of the ICM plan. While the beneficial impacts of poplar planting in the catchment cannot be completely discounted, this step change response suggests that the removal of cattle from riparian areas had the greatest impact. There is little evidence to suggest that there are any on-going improvements in water clarity due to poplar growth. This is consistent with the hypothesis that preparatory processes are likely to be the most important bank erosion process occurring in many headwater streams. The erosion control benefits of poplar trees in this catchment are therefore likely to be minimal as such large trees with extensive roots systems provide more effective protection against fluvial scour or mass failure mechanisms (Abernethy and Rutherfurd 1998, 2000).

Our findings highlight the need for catchment managers to consider a range of factors (e.g., stream size and dominant erosion processes) when considering catchment rehabilitation measures. The objectives of rehabilitation measures, such as riparian planting also need to be carefully considered before they are implemented. If the major goal is to rapidly improve the water clarity or reduce sediment loads, then the potential consequences on the existing riparian vegetation and river morphology need to be taken into account. Reduced sediment delivery from river banks is a potential long-term outcome of stock exclusion and replacement of grasses with shrubs and trees in riparian areas. Nevertheless, it should be appreciated that the replacement of dense riparian groundcover vegetation (such as grazed pasture) with shrubs and trees may (at least in the short-term) result in conditions that promote the widening of headwater channels toward their naturally greater width under forest lighting conditions (Davies-Colley 1997; Anderson et al. 2004). Clearly, riparian planting is often used as more than just a tool to reduce the amount of sediment being transported by a river (e.g., habitat improvement and nutrient uptake). In these cases, a decrease in water clarity (and transport of particulate material) by bank degradation (until some new river bank morphology equilibrium is reached) may be an acceptable consequence of achieving the primary rehabilitation objective.

As with water clarity, the effects of implementing catchment management measures on other aspects of water quality was also equivocal. There was a variable response in the concentrations of the different forms of P and N at all four experimental sites. At PW2, where the upstream catchment area was planted in *P. radiata* (with the exception of a 10 m riparian buffer that was allowed to regenerate naturally), concentrations of DRP, TP, nitrate-N, and TN all had positive post-ICM trends and all median post-ICM concentrations were higher than the pre-ICM median concentrations. At PR2, where extensive riparian planting with indigenous vegetation occurred, there were positive trends in the post-ICM concentrations of DRP, nitrate-N, and TN. These findings are contrary to previous research in New Zealand that has consistently found lower levels of N and P leaching from afforested areas than other agricultural land uses. Menneer et al. (2004) reviewed the impacts of different land uses on N and P losses and found that the mean N leaching loss from forestry areas was an order of magnitude less than land used for sheep grazing, dairying and cropping. The Menneer et al. (2004) review also indicated that the concentrations of both dissolved and total P are markedly lower in streams under pine forest than those under pasture. The legacy effect of previous land uses on nutrients accumulated in pasture soils and groundwater is important, however, with Davis et al. (2012) finding that pine forest established on former pasture land had higher nitrate-N leaching than pine forest established on indigenous forest or shrub land.

We propose that there may be a combination of factors contributing to the positive trends in forms of P and N at PR2 and PW2. First, in regards to both N and P, stream channels became progressively shaded as riparian areas that were retired from grazing naturally regenerated (Quinn et al. 2009). Light measurements indicate that by 2007, lighting declined from 40 to 50 % (pre-ICM) to 5 % at

PR2, ~30 % at PW2 and PW3 and 19 % at PW5 (Authors' unpublished data). Stream channel shading by weeds and regenerating indigenous vegetation and reductions in the periphyton biomass and/or macrophytes have previously been measured within the PW2 and PR2 catchments (Quinn et al. 2009). A number of studies have attributed increases in concentrations of N and P to shading-induced reductions in macrophytes and periphyton. Working at experimental sites within the Mangaotama catchment, Quinn et al. (1997) demonstrated that nitrate-N uptake rates by periphyton decreased gradually with increasing levels of shade and Cox and Rutherford (2012) found that experimental removal of watercress eliminated previously high nitrogen attenuation (6–11 % mass loss over 100 m). Matheson et al. (2012) showed rapid reductions in nitrate-N and dissolved P uptake as stream lighting decreased. In a long-term study in the Taupo district of New Zealand, Howard-Williams and Pickmere (1999) also attributed reductions in instream uptake of nitrate-N and DRP to riparian shading. Similar findings have also been found in other parts of the world (e.g., Sabater et al. 2000; Bukaveckas 2007).

Within the Mangaotama catchment (with exception of DRP at the PW5 site) DRP and nitrate-N concentrations increased significantly at all sites. Our results suggest a possible relationship between riparian shading and the percentage increase in nitrate-N concentration. If we use the proportion of channels (within a site's catchment) with riparian vegetation (planted or naturally regenerating) as a surrogate of channel shading, we find a possible correlation. At PW2 there was ~100 % riparian regeneration and nitrate-N increased by ~105 %; at PW3 there was ~35 % riparian regeneration and nitrate-N increased by ~23 %; at PW5 there was ~51 % riparian regeneration and nitrate-N increased by ~76 %.

The reduction in uptake of dissolved N and P by periphyton and macrophytes under the increasingly shaded, post-ICM conditions is expected to have altered the timing, mode, amount and form of nutrient export from the catchments, resulting in higher baseflow concentrations. In the pre-ICM conditions of high light input to the streams, dissolved nutrients incorporated into the organic pools in abundant periphyton and macrophyte biomass (thus lowering baseflow instream concentrations) are expected to be: (i) scoured from the streambed and banks during spates (Biggs and Thomsen 1995), or sloughed during senescence, and either deposited to floodplain as detritus or exported downstream; and (ii) returned to the terrestrial environment through livestock browsing of macrophytes or aquatic insect emergence (Small et al. 2013). Scour of periphyton by spates occurs rapidly as resistance thresholds are exceeded (Biggs and Thomsen 1995), so that export of this material is unlikely to be captured by routine monthly sampling.

A further relevant factor at PR2 and PW2 that may be contributing to increasing concentrations of N is the growth of the nitrogen fixing legume gorse (*Ulex europaeus*). Gorse is an invasive weed in New Zealand and its low palatability to stock means that, without careful management, it spreads readily on grazing land (Magesan et al. 2012). Before the ICM changes, areas of poor quality pasture within the Mangaotama catchment (including within the PW2 catchment) were affected by gorse growth (Dodd et al. 2008b). While the PW2 catchment was planted in pine forest, there are significant patches (>10 % of the catchment area) where no pine was planted (likely due to dense gorse growth after control measures ceased) and it was left to regenerate naturally. These areas, together with the 10 m buffer adjacent to streams (that was also was left to regenerate naturally) means that there were significant areas where gorse either remained or had the opportunity to become established. Field reconnaissance and analysis of satellite imagery suggests that gorse is (or was) present in the areas where pine was not planted. Gorse is also present in the planted riparian areas within the PR2 catchment. Gorse is capable of fixing large amounts of nitrogen (200 kg ha^{-1} y^{-1}) (Magesan et al. 2012) and this can be readily leached out of soils. Dyck et al. (1983) found that average nitrate-N concentrations of 5.1 g m^{-3} in the soil water under areas of gorse while they were only 0.006 g m^{-3} from stands of *P. radiata* in the central North Island of New Zealand. Magesan and Wang (2008) also found nitrate-N leaching rates of between 36 and 64 kg ha^{-1} y^{-1} from two sites in the Rotorua lakes catchment, also in the central North Island of New Zealand.

Although ryegrass/clover mix pasture (commonly found in New Zealand) can also fix large amounts of nitrogen, a previous study from within the same study catchment found N fixing by clover of between 55 and 85 kg ha^{-1} y^{-1} (Ledgard et al. 1987). They also found that due to removal by grazing and transfer to stock camp areas, accumulation rates of soil N on steep hillslopes is likely to be very slow (~7 kg ha^{-1} y^{-1}).

A further possible explanation to the increasing concentrations of N at PW2 and PR2 may be the change in functioning of small seepage wetlands, in response to the growth of indigenous forest (at PR2) and *P. radiata* plantation (at PW2). Nitrate-N removal by such wetlands under baseflow conditions has been found to exceed 75 % (Downes et al. 1997; Rutherford and Nguyen 2004). The Mangaotama catchment is interspersed with small seepage wetlands (see Rutherford and Nguyen 2004). Previous research has suggested that the reduction in areal coverage of these wetlands in response to afforestation may be responsible for increased downstream nitrate-N concentrations (Smith 1992). This is because a reduction in the amount of subsurface water passing through these wetlands may result in a decreased input into streams of water with low nitrate-N concentrations (due to removal of nitrate-N by denitrification). These seepage wetlands were (and still are) particularly common throughout the PW2 catchment, however, we do not have any information on any possible change in their areal extent through the study period.

We have suggested a number of possible explanations for the positive post-ICM trends in nitrate-N concentrations within the catchments of PR2 and PW2. However, at both PR2 and PW3, there were also similar positive trends in DRP concentrations during the post-ICM period. DRP concentrations are unlikely to be affected by either gorse growth or seepage wetlands in the same way that nitrate-N is. Therefore, at least for DRP, it would appear that the positive trends may be largely explained by the shade-induced reduction in in-stream processing. If this hypothesis is correct we may expect DRP concentrations to increase further with increasing levels of riparian shading. Although this may be counteracted to some degree by the eventual reduction in the nutrients accumulated in pasture soils and groundwater during the previous agricultural phase.

Despite the findings that the low-flow concentrations of some forms of N and P have increased at various points, including nitrate-N and TN near the catchment outlet (PW5), no significant increase in annual nitrate-N or DRP loads were detected over the study period by two load estimation methods (Table 4). The absence of an increase in annual nitrate-N loads may be related to the reduction in (rainfall adjusted) mean annual flow recorded at PW5 during the post-ICM period. That is, while concentrations have increased this has been counteracted by the reduction in the amount of runoff generated due to the growth of ICM plan initiated tree planting. No significant trend was detected in annual DRP loads. Again, the combination of increasing DRP concentration and reduction in mean annual flow over the post-ICM period may account for this.

No significant difference was detected in the pre- and post-ICM concentrations of ammonium-N at any of the sampling sites. Despite this, a contraction in the inter-quartile ranges of ammonium-N at both PW2 and PW3 and in the inter-decile range at PW5 was observed. In agricultural catchments the major source of ammonium-N is animal excreta. Once cattle were removed from streams and riparian areas, a major source was removed resulting in lower maximum ammonium-N concentrations. Conversely, the increase in the minimum ammonium-N concentrations can be attributed to the reduction in nutrient uptake by macrophytes and periphyton caused by the increased riparian shading.

At the control sites most measured water quality variables remained static. While there was no measured degradation in any of the variables, there were significant improvements in TON, TN, and water clarity at NW5 and TON at DB4. Due to the absence of human disturbance (including stock grazing) in the NW5 catchment, the concentrations of all water quality measures are lower at NW5 than the other sites, therefore relatively small catchment disturbances (e.g., feral animal damage or storm induced erosion scars) that may have an on-going effect on water quality, may have a more pronounced effect on the statistical analysis. This may explain why significant differences have been detected in the pre- and post-ICM median concentrations of TON and TN and water clarity values. At DB4 the difference in pre- and post-ICM median TON concentrations is due to there being a significant negative trend in TON during the pre-ICM period. The reason for this negative trend in the pre-ICM TON concentration is unknown. However, because the TON concentrations for the late pre-ICM period (1999–2001) are similar to those for the entire post-ICM period it is reasonable to suggest that there has in fact been no meaningful change in TON concentrations in the post-ICM period.

Previous research illustrates that the relationship between catchment rehabilitation and changes in stream nutrient concentrations is a complex one, with nutrient concentrations rarely decreasingly immediately or linearly in response to rehabilitation measures (Liljaniemi et al. 2003; McKergow et al. 2003). Our findings also illustrate this complexity, while there were some reductions in concentrations of the different forms of N and P at some sites, many sites experienced either unchanged or increased concentrations. While increased nutrient concentrations may not necessarily be an expected (or desirable) outcome of the implementation of catchment rehabilitation measures, it does not necessarily mean that such interventions are a failure. As measured here, increased nutrient concentrations may not necessarily result in increased nutrient loads. This is because the ICM plan initiated tree planting reduced total runoff from the catchment as a result of increased evapotranspiration. Furthermore, while the concentrations of some forms of N and P (particularly nitrate-N at PW2) increased markedly, the concentrations remain well below what is considered to be toxic to freshwater organisms in New Zealand (Hickey and Martin 2009). The major concern with excess dissolved N and P in waterways is the overgrowth of algae, which alter the riverbed habitat and can decrease the dissolved oxygen content of the water and therefore adversely affect the health of in-stream biota. Measured reductions in the amount of periphyton and macrophytes in treated reaches and improvements in macroinvertebrate community metrics (Quinn et al. 2009) suggest that this is not an issue here.

Furthermore, there may be site specific issues that have resulted in the nutrient increases we have reported here. Increases in nitrate-N may be, in part, a result of the unchecked growth gorse in areas not planted in pine and the widespread occurrence of small seepage wetlands. Where relevant, careful consideration of such factors in other catchments may be worthwhile to ensure rehabilitation goals are not compromised. In New Zealand, gorse is endemic and leaving pastoral land to regenerate naturally is likely to result in conditions that are (at least in the short-term) favorable to its growth and spread. Gorse is, however, shade intolerant, therefore minimizing the amount of land that is left to regenerate naturally may be a method to ensure gorse does not become problematic. Moreover, in catchments were seepage wetlands are important, the effect of afforestation on their ability to attenuate dissolved nutrients (particularly N) needs to be appreciated.

Conclusions

Thirteen years of water quality monitoring data from both before and after the implementation of an ICM plan provided a rare opportunity to quantify the effects of such changes over the long-term. The removal of cattle from riparian areas had an immediate and positive effect on stream water clarity. In contrast, where riparian areas were planted (or naturally regenerated) with trees and shrubs and livestock were excluded, the stream water clarity decreased. We attribute this to a reduction in groundcover vegetation (caused by shading by weeds, trees and shrubs) that armor stream banks against preparatory erosion processes. The response of different forms of N and P to the catchment management changes has been complex with concentrations increasing at some sites. These increases were attributed to: (i) the reduction of instream nutrient uptake by macrophytes and periphyton due to increased riparian shading; (ii) uncontrolled growth of a nitrogen fixing weed (gorse) in some parts of the catchment, and (iii) the reduction in the nutrient attenuation capacity of seepage wetlands due to the decrease in areal coverage of seepage wetlands in response to afforestation. These findings highlight the complexity of the response of stream water quality to catchment management changes and illustrate the need for catchment managers to consider a range of factors when planning catchment rehabilitation measures.

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