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Before and after riparian management: sediment and nutrient exports from a small agricultural catchment, Western Australia

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Abstract

Riparian vegetation can trap sediment and nutrients sourced from hillslopes and reduce stream bank erosion. This study presents results from a 10-year stream monitoring program (1991–2000), in a 6 km^2 agricultural catchment near Albany, Western Australia. After 6 years, a 1.7 km stream reach was fenced, planted with eucalyptus species and managed independently from the adjacent paddocks. Streamflow, nutrient and sediment concentration data were collected at the downstream end of the fenced riparian area, so there are data for before and after improved riparian management. Suspended sediment (SS) concentrations fell dramatically following improved riparian management; the median event mean concentration (EMC) dropped from 147 to 9.9 mg l^{-1}. Maximum SS concentrations dropped by an order of magnitude. As a result, sediment exports from the catchment decreased following improved riparian management, from over 100 to less than 10 kg ha^{-1} yr^{-1}. Observations suggest that this was the result of reduced bank erosion and increased channel stability. Riparian management had limited impact on total phosphorus (TP) concentrations or loads, but contributed to a change in phosphorus (P) form. Before improved riparian management, around half of the P was transported attached to sediment, but after, the median filterable reactive P (FRP) to TP ratio increased to 0.75. In addition, the median FRP EMC increased by 60% and the raw median FRP concentration increased from 0.18 to 0.35 mg l^{-1}. These results suggest that there was a change in the dominant P form, from TP to FRP. Changes in total nitrogen (TN) following improved riparian management were less clear. There were reductions in TN concentrations at high flows, but little change in the loads or EMC. This study demonstrates the benefits of riparian management in reducing stream bank erosion, but suggests that in catchments with sandy, low P sorption soils, there may be limitations on the effectiveness of riparian buffers for reducing P and N exports.
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1. Introduction

Degradation of stream water quality has been linked to livestock grazing in many parts of the world.

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However, these negative impacts can be reduced or eliminated with correct management. One tool available to farmers and catchment managers is improved riparian management. Riparian areas are lands directly adjacent to rivers and streams, and can potentially buffer streams from the impacts of agriculture. In this paper, the term *riparian buffer* refers to the management of riparian areas for water quality improvement.

Riparian buffers can improve stream water quality by a combination of physical, chemical and biological processes, and perform three key roles in minimising the impacts of agriculture on stream water quality. Firstly, they help stabilise stream channel morphology (Thorne, 1990). Secondly, they help protect streams from upland sources of pollution by physically filtering and trapping sediment, nutrient and chemicals (Osborne and Kovacic, 1993). Thirdly, they can displace sediment and nutrient producing activities away from the stream (Wenger, 1999).

Uncontrolled grazing of riparian areas by livestock, both cattle and sheep, can cause degradation of streams and their water quality. Comprehensive reviews on the impact of cattle on streams can be found in Meehan and Platts (1978), Kauffman and Krueger (1984), Trimble and Mendel (1995), Mosley et al. (1997) and Belsky et al. (1999), so only a brief summary is presented here. Cattle are attracted to riparian areas and may spend time in and around streams (Trimble and Mendel, 1995). Reasons for this attraction include shade, gentle topography, drinking water and palatable vegetation (Kauffman and Krueger, 1984). Their presence in or near streams can negatively affect water quality, channel morphology, hydrology, riparian soil structure, in-stream and stream bank vegetation. Causes of these negative impacts include: livestock urine and manure deposition into streams, in-stream trampling, increased bank erosion due to reduced vegetation, stream bank breakdown by livestock and soil compaction (Trimble and Mendel, 1995; Mosley et al., 1997; Belsky et al., 1999).

Information on the impact of sheep grazing riparian areas is limited, as sheep are not attracted to streams and seem to prefer drier soils. Thus the impact of sheep grazing on riparian areas may be minimal and not very different from non-use (Platts, 1989). However, if sheep are forced to graze heavily (Platts, 1981) or allowed to establish bed camps (where they concentrate urine and faeces) in riparian areas (Glimp and Swanson, 1994), their impact may be similar to cattle.

The exclusion of livestock from streams has resulted in the recovery of channel form (Howard-Williams and Pickmere, 1994; Williamson et al., 1996; Mosley et al., 1997), but the water quality benefits are much less certain. A few studies have documented large reductions in sediment yields as a result of riparian fencing, but few have assessed nutrient changes. Riparian fencing has been shown to reduce stream suspended sediment (SS) loads by between 40 and 80% (Owens et al., 1996; Williamson et al., 1996; Line et al., 2000). These reductions appear to be due to decreased stream bank erosion. Generally, once cattle are excluded from riparian areas there is a rapid transition from a wide, shallow stream with an unstable bed and heavily grazed and trampled banks, to a stream with more stable, vegetated banks (Howard- Williams and Pickmere, 1994). However, channel stability can be important in determining whether fencing will have any impact on channel form. Williamson et al. (1992) found that on larger channels, which were more actively meandering, riparian retirement had comparatively little benefit because any retirement or grazing effects were rapidly overtaken by channel migration.

Nutrient reductions following riparian buffer creation have been more variable. Both Line et al. (2000) and Williamson et al. (1996) reported significant reductions in P loads. On deep, free draining soils in New Zealand, Williamson et al. (1996) recorded small reductions in particulate N but increases in dissolved N. They suggested that the increase in dissolved N, mainly nitrate, could be due to reduced in-stream plant uptake or a consequence of historical land use. Line et al. (2000) recorded reductions of total Kjeldahl N (organic N plus ammonia) in a North Carolina catchment with a shallow loamy A horizon over clay, and suggest that they are most likely due to the SS reductions and therefore sediment associated N.

The study reported in this paper uses the 'before and after' approach to investigate the potential of riparian management to assist in meeting the nutrient targets set for the Oyster Harbour catchment, near Albany on the south coast of Western Australia.

Common experimental designs for water quality best management practice evaluation include paired catchments, upstream–downstream sites monitored before and after, or multiple catchment monitoring (Spooner and Line, 1993). The site used in this study was established in 1991 as part of a previous project investigating tributary water quality. In 1996, when the landowner decided to fence the riparian buffers, an opportunity arose for a before and after study, with a long before period, but with the constraint of no control or upstream monitoring site. Consequently, we pay particular attention to similarities and differences between the before and after periods.

Nutrient export targets were set by the Western Australia Environmental Protection Authority (WAEPA, 1990) to halt the decline of seagrass communities caused by excessive algal growth and eutrophication. Annual targets of <13.9 t of P (areal weighted equivalent: <0.05 kg ha^{-1}) and <107.9 t of N (areal weighted equivalent: <0.36 kg ha^{-1}) were set, and agriculture was allocated 99% of the target load requiring load reductions of $>50\%$ (WAEPA, 1990). Three key questions form the basis of this paper:

1. Has riparian management led to a reduction in SS exports?
2. Has riparian management led to a reduction in nutrient exports?
3. Are nutrient export targets being met?

2. Study catchment characteristics

The study was undertaken in a small (5.9 km^2) agricultural catchment near Albany, Western Australia (Fig. 1). The topography of the catchment is low gentle hills with one major granitic outcrop, Willyung Hill. Elevations range between 20 and 180 m above sea level (a.s.l.), but most of the catchment is below 70 m a.s.l.

The area has a Mediterranean climate with cool, wet winters and dry, warm to hot summers. Average annual rainfall at Albany Airport is 799 mm and average annual pan evaporation is 1383 mm (37.6 and 33.1 years of record, respectively; BOM, 2001). Most of the rain falls between April and October and during this time rainfall exceeds evaporation.

Soils are duplex sands. Duplex soils are texture contrast soils, in which the subsoil (B-horizon) is at least one and a half texture groups finer than the surface soil (A-horizon). Shallow sands overlie laterite, gravels and clay on the valley slopes and deep sands in the valley floors (Churchward et al., 1988). Soil samples collected throughout the Oyster Harbour catchment were previously reported on by Weaver and Reed (1998) and Table 1 summarises the results for samples collected in this study subcatchment. Generally, the soils had high P and potassium (K) and low sulphur (S). The majority of samples had ammonium oxalate extractable iron (Fe-AmOx) concentrations below 400 mg kg^{-1} and low P Retention Index (PRI, Allen and Jeffrey, 1990) indicating that they have low P sorption (Allen et al., 1991). A sample of stream bank material adjacent to the monitoring site was also analysed and had moderate PRI (Table 1).

The North Willyung catchment was cleared in the 1950s for pastoral use. The dominant land uses are cattle and sheep grazing of annual subterranean clover (*Trifolium subterraneum* L.) and ryegrass (Table 2). During the monitoring period several land use changes occurred. On Farm 2, in the upper reaches of the catchment, blocks of Tasmanian blue gum (*Eucalyptus globulus* Labill. subsp. *globulus*) were planted during 1996 and 1997, and sheep replaced cattle in 1998. Oats were grown on Farm 4 on some of the paddocks on the north-west slopes of Willyung Hill in April 1999.

Riparian vegetation condition was assessed using the criteria outlined by the Water and Rivers Commission (WRC, 1999), which reflects the general process of river valley degradation (Table 2). The system categorises riparian condition into grades on the basis of weed infestation, soil exposure and erosion: (A) pristine to slightly disturbed native bush, (B) degraded bush with a weedy understorey, (C) erosion prone foreshore with erosion and subsidence in a few places, (D) eroding ditch or weed infested drain. Each grade has three sub-categories denoted by 1, 2 or 3, for example (D1) eroding ditch, (D2) freely eroding ditch and (D3) weed infested drain.

On Farm 1 and the segment of Albany Airport in the catchment, stock do not have access to the stream channel, as the channel head is at the boundary between Farms 1 and 2. During the before period

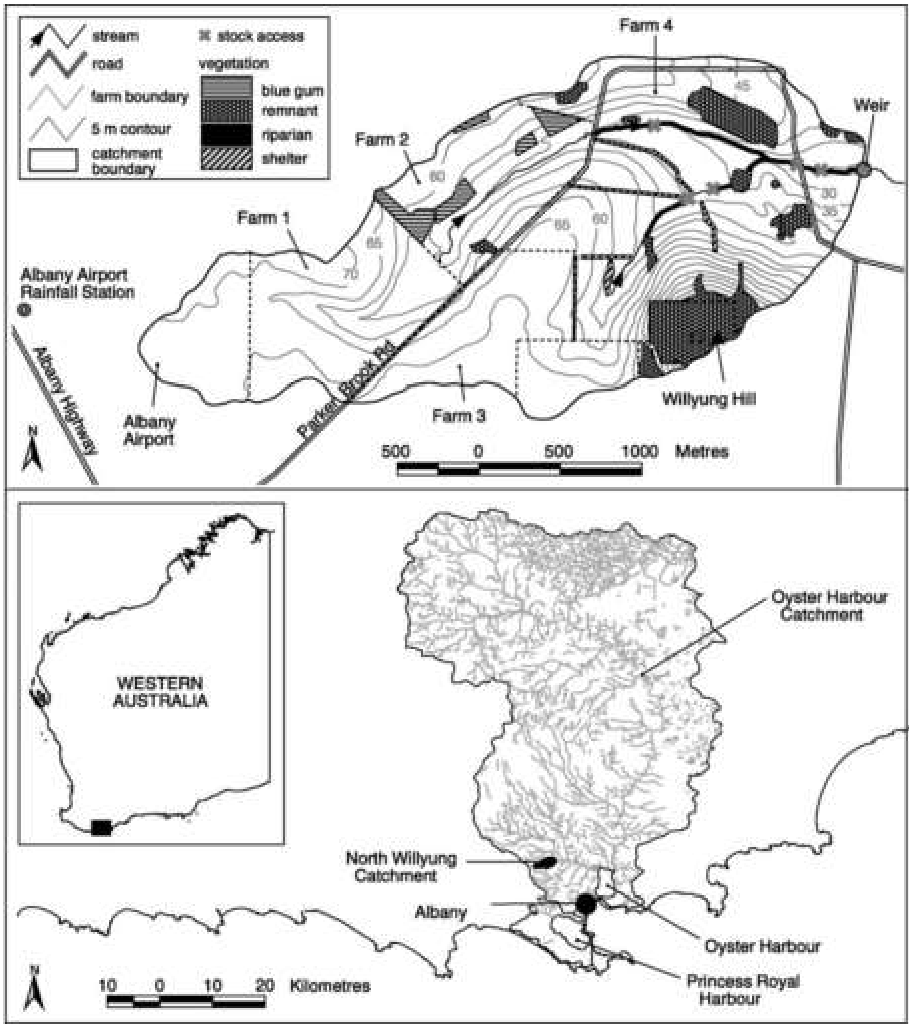

Fig. 1. North Willyung catchment showing the monitoring site, catchment vegetation, farm boundaries and stock access points. Accompanying maps place the North Willyung within the Oyster Harbour catchment and Western Australia.

the riparian vegetation on Farm 2 was graded as a freely eroding ditch (D2) as there was no fringing vegetation and erosion was out of control. In 1998, after sheep replaced cattle, the stream channel condition improved to grade D1. With time the stream channel became choked with reeds and rushes, reducing bank and bed erosion.

During the before period the riparian condition on Farm 4 was graded as a freely eroding ditch (D2). Remnant vegetation was limited to isolated trees on some reaches. The stream banks were heavily grazed and trampled, and the bed was unconsolidated sand. In 1996, the stream channel on Farm 4 was fenced along most of its length and planted with eucalypts (*E. globulus*, *E. saligna*, *E. rudis*, *E. camaldulensis*, *E. robusta*), wattles (*Acacia mearnsii*, *A. melanoxylon*), paperbarks (*Melaleuca quinquenervia*, *M. cuticularis*), pines (*Pinus radiata*), she-oak (*Allocasuarina cunninghamii*), and bottlebrush (*Callistemon glaucus*). Naturally regenerating native plants in the riparian buffer include swamp peppermint (*Agonis linearifolia*), pale rush (*Juncus pallidus*), paperbark

Table 1
Median and inter-quartile range (IQR) and P, K and S status for soil samples collected during late 1980s and early 1990s (Weaver and Reed, 1998)

Parameter	Units	Surface soil			Bank material[a]	
		No. samples	Median	IQR	Weighted total fraction	<75 μm fraction
pH($CaCl_2$)	–	69	4.1	0.45	–	–
Org C	%	56	3.52	0.37	1.3	2.6
P(HCO_3)	mg kg^{-1}	69	24	21	6	13
K(HCO_3)	mg kg^{-1}	69	141	62	–	–
Fe-AmOx	mg kg^{-1}	69	339	349	32	79
EC	mS m^{-1}	69	12.9	7.9	–	–
P-total	mg kg^{-1}	52	210	110	26	77
PRI[b]	ml g^{-1}	51	2.5	7.35	69	300
P-status	–	69	High	Medium–high		
K-status	–	69	High	High–high		
S-status	–	69	Low	Low–low		

[a] Bank material collected in this study and analysed using methods described by Weaver and Reed (1998).
[b] PRI, a single point P sorption measure is defined the ratio of P_{ads} (P adsorbed from the solution, μg g^{-1} soil) to P_{eq} (concentration of P remaining in solution at equilibrium, μg ml^{-1}) (Allen and Jeffrey, 1990).

(*Melaleuca rhaphiophylla*), myrtles (*Astartea fascicularis*, *Homalospermum firmum*) and ferns. Stock access was limited to five water points and numerous stock crossings were installed to remove stock and farm vehicles from the stream. The separation of the riparian area from the paddocks has led to an improvement in riparian condition. During the after period the condition was assessed as degraded and heavily weed infested (B2), as weeds are a significant component of the understorey, particularly taylorina (*Psoralea pinnata*), and kikuyu (*Pennisetum clandestinum*). The stream channel became choked in some sections and was excavated on several occasions to maintain the present channel course. Choking

Table 2
Landuse and riparian condition (WRC, 1999) and fertiliser use

		Farm 1	Farm 2	Farm 3	Farm 4
Stocking rate	Before	1 cow and calf/ha	1 cow and calf/ha	1 cow and calf/ha	1 cow and calf/ha
	After	No change	12 sheep/ha and tree farming	No change	No change
Riparian condition	Before	–	D2-ditch, freely eroding	–	D2-ditch, freely eroding
	After	–	D1-ditch, eroding	–	B2-degraded, heavily weed infested
Fertiliser	Type	–	5:1 super potash[a]	3:1 super potash[b]	5:1 super potash[a]
	Rate (kg ha^{-1})	–	160	55	180
	Applied P (kg)	–	990	240	3800
	History	None applied since 1988	Since 1980	Unknown	Since 1980
	Soil testing	No	Not since 1995	No	No

[a] Typical composition 7.6% P, 8.1% K, 9.6% S.
[b] Typical composition 6.8% P, 12.3% K, 8.6% S.

of the channel also increased in-channel water levels, and during winter the channel sometimes overflowed and was forced outside the fenced riparian buffer.

Fertiliser application timing and rates varied between farms and as a management tool, the use of soil nutrient testing was limited. Farm 1 has not been fertilised since the late 1980s. Farms 2 though 4 are fertilised annually in April (Table 2) with super potash. Fertiliser is also applied annually on 30 ha of hay paddocks on Farm 4, in September. Lime has been applied annually on Farm 4 for the past 20 years; 0.2 t ha^{-1} annually over the last 5 years, and prior to this 0.1 t ha^{-1}. Farm 2 also received lime in March 2000 at a rate of 2.5 t ha^{-1}.

3. Methods

A monitoring station was established in 1991 (Fig. 1), and 6 years of flow and water quality data were collected before riparian buffers were created on Farm 4 in 1996. Four more years of data were collected afterwards, enabling us to compare water quality between the two periods.

3.1. Rainfall

Rainfall data from Albany Airport (Bureau of Meteorology station 009741) were used in this study. The rainfall station is just outside the catchment boundary as shown in Fig. 1. Rainfall is recorded by a tipping bucket rain gauge (0.2 mm per tip) and a manual rain gauge is read twice daily at 9 am and 3 pm.

Rainfall data were analysed using return period analysis with the HYDSYS module HYLP3, a Log Pearson Type 3 distribution analysis (HYDSYS, 1999). Annual maximum daily (to 9 am) rainfall totals were extracted directly from the time-series archive and analysed. Good quality and complete datasets were available for 32 years of the record.

3.2. Flow measurement

A compound 30°/120° sharp-crested v-notch weir was installed at the monitoring site at the start of 1991 and instantaneous water levels were recorded with a Wesdata capacitance probe and logger (Dataflow Pty Ltd) until 1997. A 150° broad crested weir replaced the original structure in November 1996 because the original structure was drowned during large events and sand filled the weir pool. Water levels were measured with a pressure transducer (Unidata 6508 C, 0–5 m) and stored on a data logger (Campbell Scientific CR10x) every 15 min from May 1997 to September 2000. Continuous measurements of water level were converted to instantaneous flow using theoretical stage-discharge rating curves (Bos, 1989). Errors in flow estimates for the installed discharge measurement structures were estimated to be in the order of ±5% (Bos, 1989).

Gaps in the flow record due to battery and logger failure (Table 3) were filled using either modelled data or data from a nearby catchment. For the period

Table 3
Annual rainfall and streamflow

Year	Rainfall (mm)	Runoff (mm)	No. days with estimated data	Rainfall/runoff coefficient
1991	638.6[a]	108.9[a]	111	0.16[a]
1992	931.4	155.9	138	0.17
1993	821.0	138.5	166	0.17
1994	675.0	94.4	0	0.14
1995	720.0	73.3	94	0.10
1996	831.2	99.5	271	0.12
1997	724.2	93.7	0	0.13
1998	772.8	70.0	7	0.09
1999	730.6	70.6	0	0.10
2000	607.6[a]	45.8[a]	5	0.08[a]

[a] For period when runoff and rainfall were both recorded, i.e. 1991 starts 24 April and 2000 ends 1 September.

1991–1993 gaps were filled with total daily runoff generated using SIMHYD, a daily rainfall–runoff model (Peel et al., 2001). Around 40% of the record was estimated using this method during the period 1991–1993 (Table 3). The model estimates streamflow from daily rainfall and areal potential evapotranspiration data, and is a simplified version of the conceptual daily rainfall–runoff model HYDROLOG (Porter and McMahon, 1975; Chiew and McMahon, 1994). The parameters in the model were first optimised using the good quality runoff record for the period January 1997–September 2000. The optimised parameter values were then used to estimate daily streamflow for periods of missing record. The model performance was assessed using the coefficient of efficiency and coefficient of variation. The coefficient of efficiency (E) describes the proportion of recorded streamflow variance that is described by the model (Nash and Sutcliffe, 1970). If the model exactly reproduced all the recorded monthly streamflow then E would equal 1.0. The coefficient of efficiency was 0.89.

For periods after 1994, a better way of filling data gaps existed as a monitoring site had been established at Seven Mile Creek, 6.9 km SW of the weir. Gaps during 1995 and 1996 were filled using mean hourly flow from the small catchment (4.33 km^2) at Seven Mile Creek (station 603018) managed by a Western Australian state government agency (Water and River Commission, WRC). Manual water level readings were taken at the weir on each site visit and so the known flow values were used to calibrate the record from Seven Mile Creek. The estimated data were then checked against modelled daily totals generated by SIMHYD. One continuous period of missing record was from 5 September 1995 to 23 August 1996 (17% infilling in 1995, 75% infilling in 1996). For the entire gap the SIMHYD estimated runoff volume was 56.91 mm and using Seven Mile Creek data the estimated runoff was 54.71 mm.

3.3. Water quality sampling

Samples for water quality analysis were collected by one of three methods: manual grab samples, rising stage samplers and automatically. The time interval between consecutive samples varied from as long as 4 weeks, to as short as 45 min during storm events.

Manual grab samples were taken on each site visit. Rising stage, or air displacement samplers (Guy and Norman, 1970) provided event samples in the period prior to automated sample collection. Samples were collected on the rising stage when the water level rose above the top point of the intake pipe on each bottle. Eight rising stage samplers were installed on the downstream wall of the sharp-crested weir and samples were retrieved during each site visit. When the broad-crested weir was installed, five rising stage samplers were installed at staggered depths upstream of the structure.

An automatic sampler (ISCO model 6300) was installed in June 1997. The sampler was used for storm event sampling and was activated by predetermined changes in water level.

3.4. Water quality analysis

Samples were refrigerated on return to the laboratory and sub-samples filtered for FRP determination. All samples were analysed for SS (APHA, 1989; method reference 2540-D with 1.2 μm GF/C filter paper), total persulfate N (TN, APHA, 1989; method reference 4500-N) and total persulfate P (APHA, 1989; method reference 4500-P) of unfiltered samples for TP, and of filtered samples for filterable reactive P (FRP, <0.45 μm). Detection limits (DL) were 0.01, 0.02 and 1 mg l^{-1}, for P, N and SS analyses, respectively. The term *filterable reactive P* is used here, rather than dissolved or soluble, as the filtrate could be a mixture of dissolved P and P attached to colloidal material that passes through the <0.45 μm filter.

3.5. Statistical analysis

Non-parametric tests were used, as no manipulation of data below the DL is required (Helsel and Hirsch, 1992). Mann Whitney Rank Sum Tests were used to compare two groups. A simple substitution of 0.5 DL was made for values below the DL for plotting purposes.

3.6. Load calculations

Interpolation and regression were used to estimate sediment and nutrient loads for each calendar year (the majority of events occur between April and October). Both methods have shortcomings (Cohn, 1995; Robertson and Roerish, 1999), but by utilising two different methods we can estimate the loads with some confidence.

Nutrient and sediment loads were calculated by interpolation for all sites using HYDSYS (Hydsys, 1999). The discrete time series chemistry was converted to a continuous variable by interpolation and instantaneous loads were calculated directly at each instantaneous water level measurement.

Multiple regression models were developed for each nutrient and SS for all sites (Table 4). The basic regression equation used was

$$\log C(t) = \beta_0 + \beta_1 \log Q(t) + \varepsilon$$

where $C(t)$ and $Q(t)$ are the concentration and discharge at time t, respectively, and β_0 and β_1 are coefficients estimated by linear regression and ε is the model residual. Additional explanatory variables were added to the $\log_{10} C - \log_{10} Q$ model to improve residual plots and model predictive capabilities (Table 4). All samples with good quality discharge data were used to develop the regression models. Only grab and automatically collected samples were used in the regression model development as rising stage samples are collected earlier in events and may be biased toward higher concentrations (Robertson and Roerish, 1999). In addition, rising stage samples are clustered at fixed flows and may exert leverage.

Regression was used only up to a certain discharge, above which the concentrations reached a plateau (based on a visual inspection). Above the threshold discharge value the mean concentration of all samples, including stage height samples, was used in the concentration time series. A summary of the selected model for each of the datasets is listed in Table 4. The model with the highest coefficient of efficiency (E, Nash and Sutcliffe, 1970) was selected if it had statistically significant F-stat and t-stats and residuals that did not display non-uniformity, heteroscedasticity, seasonality and non-normality. All selected models were subject to a reality check. If a model failed a reality check, for example, generating TP concentrations that were lower than estimated FRP concentrations, then the next best model was used. The non-parametric smearing estimate (Duan, 1983) was used to overcome retransformation bias, because of its simplicity and robustness (Helsel and Hirsch, 1992).

Agreement between the two methods was good, with the ratio of annual regression load to annual interpolated load ranging between 0.7 and 1.5 (Table 5). Estimation of the SS load using the median concentration did not perform as well, with the ratio of annual median estimated load to annual interpolated load varying between 0.2 and 2.8 (Table 5), so the interpolated SS loads are reported.

Significant proportions of the discharge data in the before period were estimated; 1994 was the only year

Table 4
Selected multiple regression models with flow threshold and mean concentration above threshold

Period		Model	r^2	E	Threshold ($\mathrm{l\ s^{-1}}$)	Mean conc. > threshold or median
Before	SS	No model (>5% samples < DL)	–	–	–	median 54.85 mg $\mathrm{l^{-1}}$
	TP	LC = 1.163LQ + 0.149SN − 0.178CS − 2.164	0.70	0.55	40	0.595 mg $\mathrm{l^{-1}}$
	FRP	LC = 1.189LQ + 0.003Q30 − 2.571	0.52	0.45	40	0.284 mg $\mathrm{l^{-1}}$
	TN	LC = 0.738LQ + 0.157Q30R − 0.896	0.50	0.54	40	4.114 mg $\mathrm{l^{-1}}$
After	SS	No model (>5% samples <DL)	–	–	–	median 7.3 mg $\mathrm{l^{-1}}$
	TP	LC = 0.476LQ − 0.342SN − 0.169CS − 0.19BI − 1.105	0.58	0.40	20	0.560 mg $\mathrm{l^{-1}}$
	FRP	LC = 0.062Q_i − 1.62	0.49	0.23	20	0.445 mg $\mathrm{l^{-1}}$
	TN	LC = 0.667LQ − 0.137BI − 0.576	0.75	0.67	30	2.575 mg $\mathrm{l^{-1}}$

LQ = $\log_{10}(Q_i)$ where Q_i = instantaneous discharge ($\mathrm{l\ s^{-1}}$); LC = $\log_{10}(C)$ where C = instantaneous concentration (mg $\mathrm{l^{-1}}$); BF = mean daily baseflow ($\mathrm{l\ s^{-1}}$), generated using HYBASE (Hydsys, 1999), a digital filtering algorithm of Lyne and Hollick (1979) with a filtering factor of 0.925 and 3 passes; BI: baseflow index, BI = BF/Q_m where Q_m is the mean daily discharge; Q_{30}: antecedent discharge, average of previous 30 days mean daily discharge; Q_{30}R: discharge ratio, $Q_{30}\mathrm{R} = Q_m/Q_{30}$; SN: sine seasonality function, $\sin(2\pi T)$ where T is date (years); CS: cosine seasonality function, $\cos(2\pi T)$.

Table 5
Summary of calculated loads (R = regression, I = interpolation) and loads comparison (ratio of R/I). Note 1991 and 2000 only have data for 8 months and all SS 'regression' loads were calculated using the median concentration

Year	SS (kg)			TP (kg)			FRP (kg)			TN (kg)			Runoff[a] (mm)
	R	*I*	Ratio	*R*	*I*	Ratio	*R*	*I*	Ratio	*R*	*I*	Ratio	
1991	35,000	206,000	0.2	280	360	0.8	150	200	0.7	2100	–	–	108.9
1992	50,000	92,000	0.5	400	370	1.1	210	230	0.9	2900	2100	1.3	155.9
1993	45,000	16,000	2.8	330	220	1.5	160	140	1.2	2200	1400	1.5	138.5
1994	30,000	120,000	0.3	180	240	0.8	90	100	0.9	1300	1500	0.9	94.4
1995	24,000	76,000	0.3	130	150	0.8	60	60	0.9	970	1300	0.8	73.3
1996	32,000	32,000	1.0	220	210	1.0	110	100	1.0	1600	1500	1.1	99.5
1997	4000	6600	0.6	230	190	1.2	170	150	1.2	990	1200	0.8	93.7
1998	3000	7200	0.4	170	120	1.4	120	87	1.3	680	720	1.0	70.0
1999	3000	3700	0.8	180	210	0.9	130	180	0.7	730	800	0.9	70.6
2000	1700	4100	0.4	110	110	1.0	80	77	1.0	470	520	0.9	45.8

[a] See Table 3 for number of days with missing or estimated data.

for which a continuous record was available (Table 3). A sensitivity analysis was carried out for the 1994 data to compare the loads generated using the continuous flow record with the mean daily flow record and the mean daily flow generated by the rainfall–runoff model SIMHYD. There were differences between the annual load estimates for each of the parameters, but all were between 0.85 and 0.98 of the Q_i load estimates, suggesting that loads estimated using coarser resolution modelled discharge data are acceptable.

3.7. Event mean and flow weighted concentrations

Event mean concentrations (EMC) were calculated using two different concentration datasets. For events with more than two samples and a good quality flow record interpolation was used to estimate EMC. The EMC was calculated by dividing the total load for each event by the total event volume. Events were defined as the period following rainfall, between a discharge rise greater than $1.5\ l\ s^{-1}$ in 15 min, and the time when the falling hydrograph limb was equal to the start discharge, or if another event occurred during the recession, when the falling limb discharge was at least 80% of the total rise. Flow weighted mean concentrations (FWMC) were calculated by dividing the total load by the total flow volume for the specified period, and were calculated for the interpolated loads only.

4. Results

4.1. Rainfall and streamflow

Rainfall totals for the monitoring period were generally below average with the exceptions of 1992 and 1996 which were wetter than average (Table 3). There were an average of 194 rain days per year, and 94 days per year when more than 2 mm was measured.

During the monitoring period there were generally four or five events each year that exceeded the 0.99 AEP calculated for daily rainfall totals (Table 6). Nine events exceeding the 0.99 AEP were recorded during 1992, and fewer large rainfall events were recorded during 1995, 1996 and 1997 compared with other years.

Four extreme daily rainfall totals (exceeding 0.2 AEP) were recorded during the monitoring period, and three of them during the before period (Table 6). The two extreme daily rainfall totals in 1991 were part of the same event; between 15:00 19 July and 24:00 21 July, 130 mm of rain was recorded. This event was the largest during the monitoring period and widespread flooding occurred throughout the catchment. The other two large rainfall events (1993 and 2000) did not result in widespread flooding as they occurred during late summer. In August 1992, a total of 85 mm over 2 days resulted in another extreme runoff event, although flooding was not as widespread as the July 1991 event.

Table 6
Annual exceedance probabilities (AEP) and return periods for daily rainfall totals at Albany Airport (1964–1999) and number of days with total rainfall exceeding the rainfall threshold for each return period

AEP (1/Y)	Return period (Y)	Rainfall (mm) (5%, 95% CI)	Number of days exceeding given rainfall									
			Before						After			
			1991	1992	1993	1994	1995	1996	1997	1998	1999	2000[a]
0.99	1.01	21.3 (+6.8, − 5.2)		5	2	1	1	1	3	2	3	2
0.9	1.11	27.2 (+2.9, − 2.6)	2	3		3	1	2		2	1	
0.5	2	38.9 (+4.0, − 3.6)		1	1					1		
0.2	5	51.1 (+6.2, − 5.5)	1									1
0.1	10	59.6 (+9.3, − 8.0)	1		1							
0.02	50	79.9 (+23, − 18.8)										

[a] Rainfall record ends 31 August 2000.

Only a small portion of rainfall becomes streamflow at this site. The North Willyung stream flowed continuously during the study period and in winter the stream was responsive to rainfall (Fig. 2). Rainfall–runoff coefficients were calculated on an annual basis, but limited to periods when rainfall and runoff data were both collected. Runoff coefficients ranged between 0.08 and 0.17 during the monitoring period (Table 3). These figures are consistent with those calculated for other small catchments in south–west Western Australia (Sharma et al., 1980; Peel et al., 2001).

Although rainfall totals were similar during the before and after periods, more runoff was recorded during the before period and so runoff coefficients are generally higher. Decreased annual yields during the after period may be a result of increasing water use in the riparian zone, errors in discharge measurement

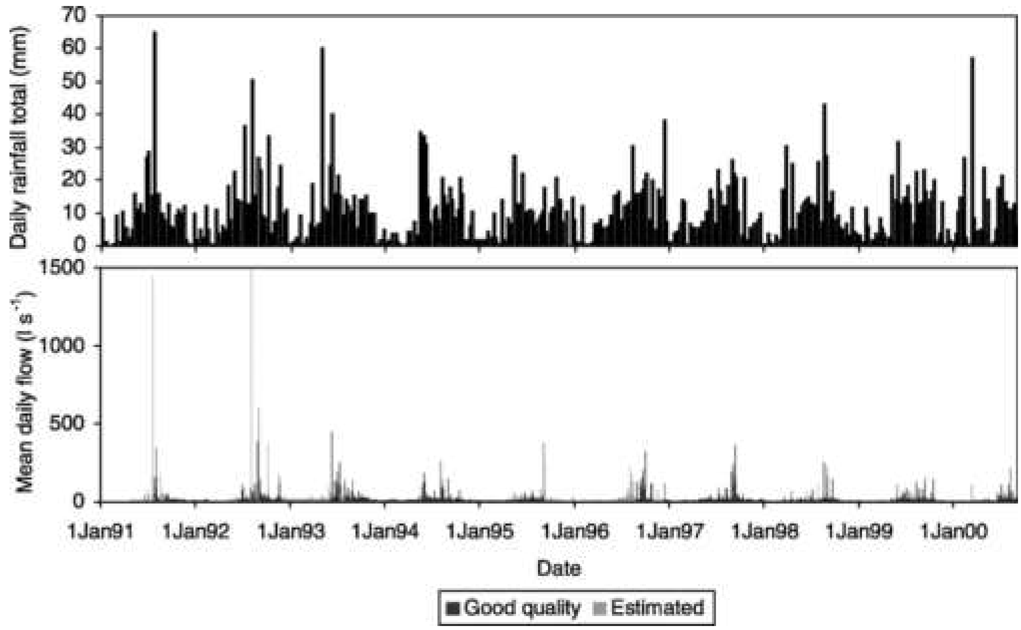

Fig. 2. Daily rainfall totals at Albany Airport, and mean daily flow at the weir, 1991–2000.

and extreme events. Higher runoff coefficients for 1991 and 1992 are largely due to the two extreme events recorded during these years.

Individual years do, however, have similar runoff behaviours, and these are compared in the following sections. Both 1994 and 1997 had runoff totals of 94 mm and a similar number of events were sampled (16 of 25 events in 1994 and 15 of 21 events in 1997). Peak discharges of the sampled events during the two years were not significantly different (Mann Whitney, $p = 0.649$). Comparisons are also made between 1995, 1998 and 1999 which all had annual runoff totals around 70 mm. Thirteen of 19 events with good quality flow records were sampled during 1995. During 1998, 16 of 19 events were sampled, and in 1999, 22 of 26 events were sampled. No statistical differences were evident between the peak discharges of sampled events during 1995 and 1998 (Mann Whitney, $p = 0.948$), and 1995–1999 (Mann Whitney, $p = 0.138$).

4.2. Suspended sediment

Improved riparian management had a large impact on SS concentrations (Fig. 3, Table 7). The median EMC decreased significantly by 93% (Table 7), the median raw concentration decreased significantly from 54.9 to 7.3 mg l^{-1} (Fig. 3, Mann Whitney, $p < 0.001$) and the average FWMC decreased from 148 to 13 mg l^{-1}. The range of values also decreased by an order of magnitude. Before the riparian buffers were present, concentrations during events were often greater than 200 mg l^{-1}, but after the riparian buffers were created only the extreme outliers were this concentrated, and the majority of samples measured had SS concentrations less than 20 mg l^{-1} (Fig. 3).

Hidden within the summary figures is the inter-annual variability and differing runoff volumes. Decreased runoff during the period following improved riparian management may have further contributed to the reductions observed in SS. However, by comparing concentrations for years with similar runoff totals and event sizes we can verify the trends seen in the summary figures. Concentration–discharge plots (Fig. 4) for all of the samples collected during 1994 and 1997 show that for a given discharge there was an order of magnitude decrease in SS in 1997. An order of magnitude decrease in SS concentration for a given discharge is also evident in Fig. 5, which compares the drier years (1995 compared with 1998 and 1999).

SS exports decreased dramatically during the monitoring period. The highest annual loss was measured in 1991, when a single extreme event occurred and there was widespread sand deposition over the floodplain. The average sediment load prior to riparian management was 114 kg ha^{-1} yr^{-1} (excluding 1991), although annual variability was large (Fig. 6). After improved riparian management the average sediment load dropped to less than 10 kg ha^{-1} yr^{-1} and annual variability was reduced.

4.3. Phosphorus

Nutrient data show that there was no change in TP concentrations and an increase in FRP concentrations between the before and after periods (Fig. 3, Table 7). The TP median EMC and FWMC were similar for the before and after periods, at around 0.51 (Table 7) and 0.39 mg l^{-1}, respectively. The range of TP concentrations altered slightly, but were significantly different due to an increase in the 25th percentile concentration (Fig. 3). Above 0.25 mg l^{-1} (after 25th percentile), there was no significant difference between the before and after TP concentrations (Mann Whitney, $p = 0.378$).

In contrast, FRP concentrations doubled and the concentration range increased after the riparian buffers were established. The median raw concentration increased from 0.18 to 0.35 mg l^{-1} (Fig. 3, Mann Whitney, $p < 0.001$) and the median EMC increased by 60% from 0.23 to 0.37 mg l^{-1} (Table 7, Mann Whitney, $p < 0.001$). Maximum FRP concentrations also increased from 1 to 1.7 mg l^{-1}. An increase in the FRP/TP ratio also occurred (Fig. 4) from a median of 0.5 to more than 0.75. Figs. 5 and 6 confirm that for a given discharge, TP concentrations have remained similar while the proportion of TP moving as FRP (FRP/TP ratio) has increased.

Total P exports from the catchment were similar throughout the monitoring period, generally between 0.2 and 0.6 kg ha^{-1} yr^{-1} (Fig. 6). Total

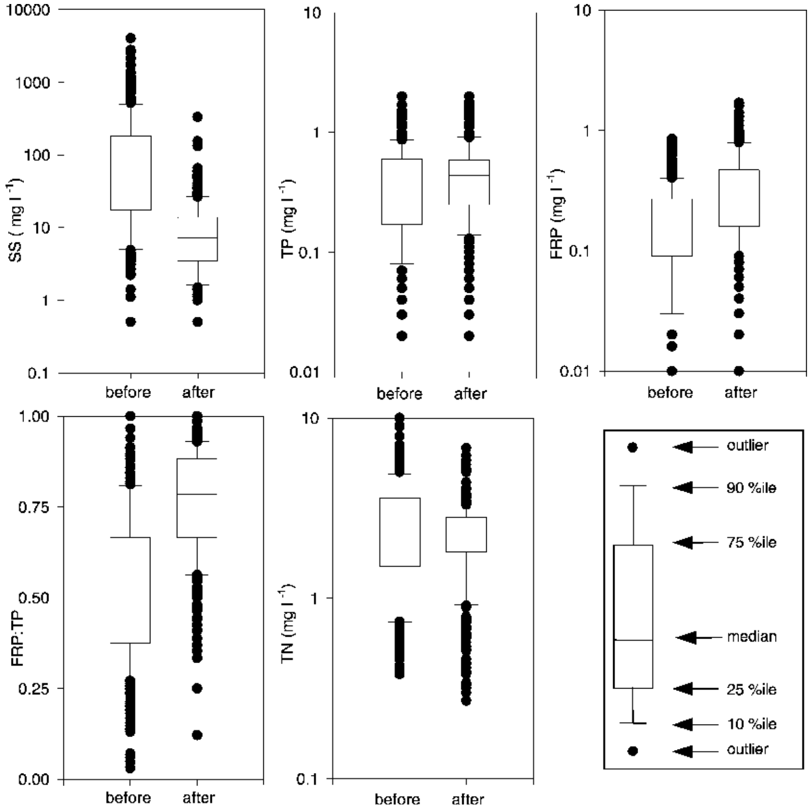

Fig. 3. Box plots of raw SS and nutrient concentrations (all samples) for before and after riparian buffers. All comparisons are significantly different (Mann Whitney, $p < 0.001$).

P exports exceeded the areal weighted equivalent export target of $<0.05\ \mathrm{kg\ ha^{-1}\ yr^{-1}}$ set by the WAEPA (1990) by up to an order of magnitude. FRP exports were variable (Fig. 6), between 0.1 and $0.35\ \mathrm{kg\ ha^{-1}\ yr^{-1}}$, also exceeding the target value throughout the monitored period.

4.4. Nitrogen

Changes in TN concentrations are less clear. While the median EMC and median raw concentration reductions are significantly different, Figs. 4 and 5 suggest that the key change to TN

Table 7
EMC summary statistics before and after riparian management

		SS EMC (mg l^{-1})	TP EMC (mg l^{-1})	FRP EMC (mg l^{-1})	TN EMC (mg l^{-1})
Before	Median	146.7[a]	0.51	0.23[a]	3.26[a]
	IQR	315.1	0.39	0.19	2.2
	Number	45	45	45	42
After	Median	9.9[a]	0.48	0.37[a]	2.51[a]
	IQR	13.9	0.30	0.30	0.56
	Number	48	48	48	48
% Reduction		93	6	−60	23

[a] Median EMC are significantly different (Mann Whitney, $p < 0.001$).

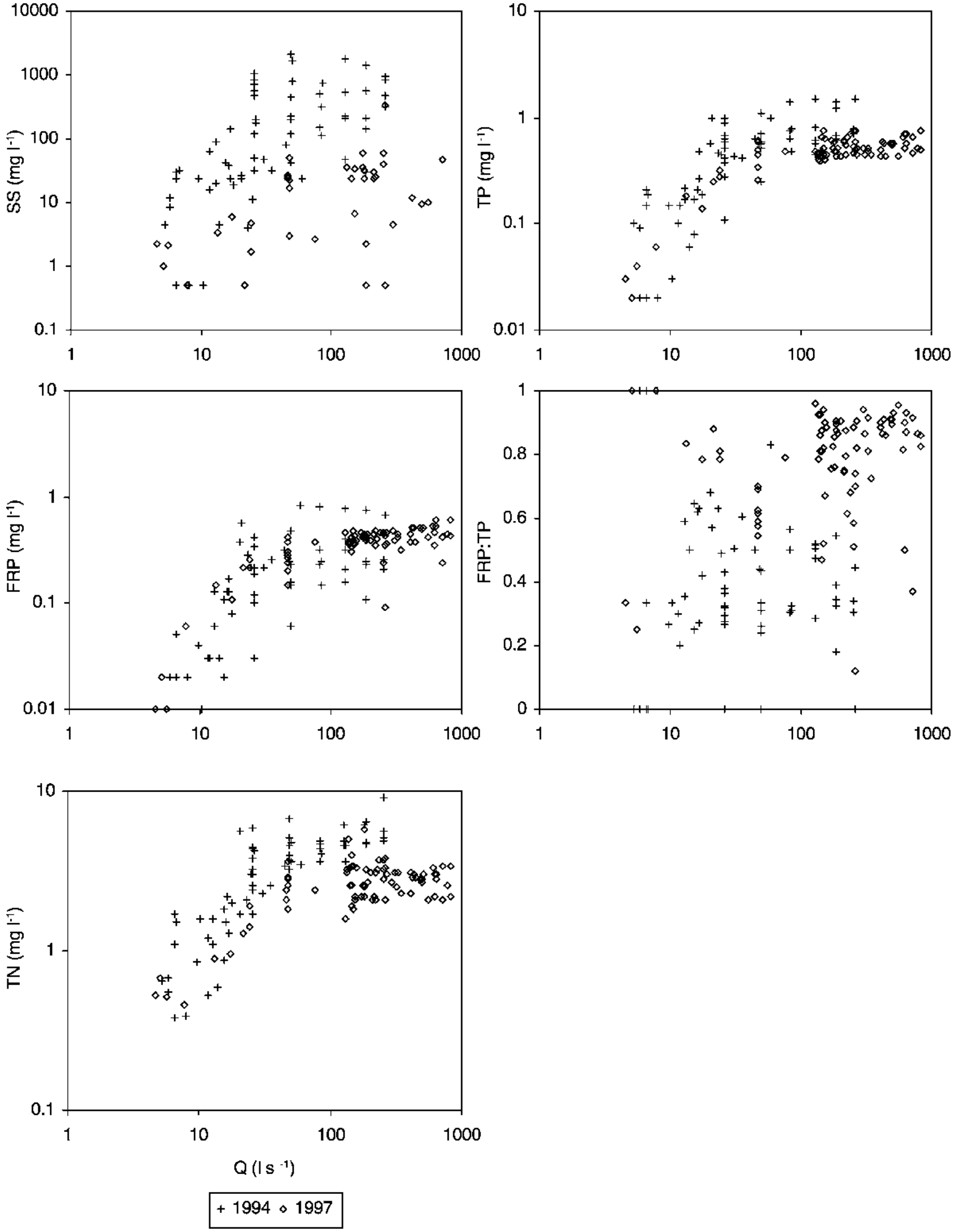

Fig. 4. Concentration–discharge relationships for all samples collected during 1994 and 1997.

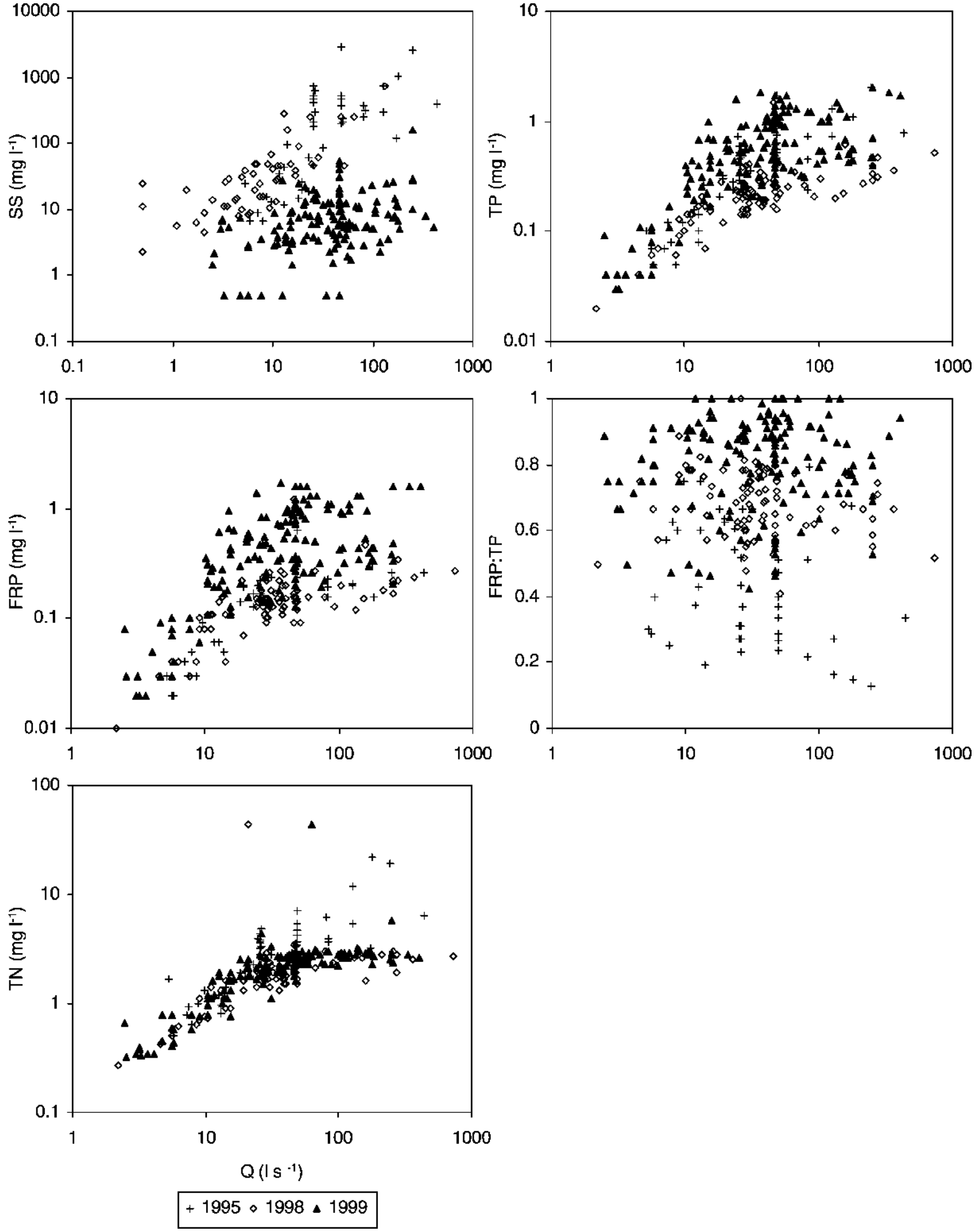

Fig. 5. Concentration–discharge relationships for all samples collected during 1995, 1998 and 1999.

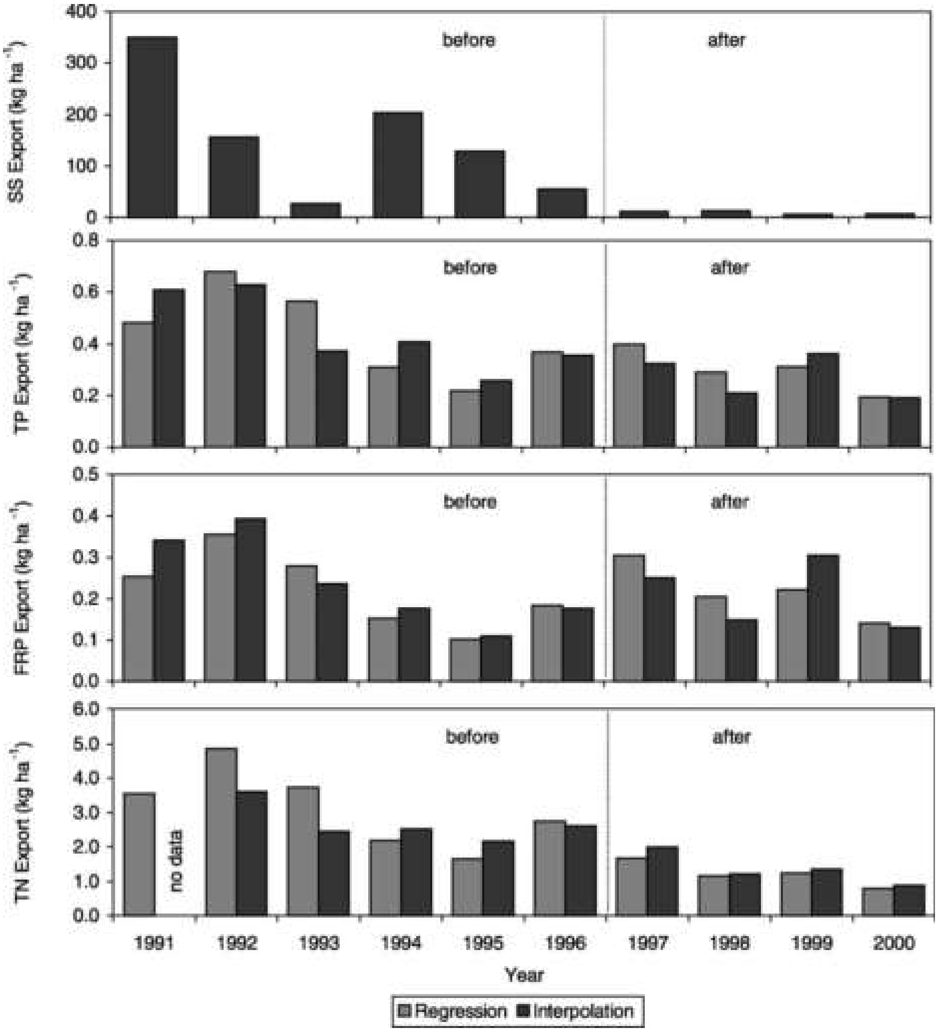

Fig. 6. Annual exports of SS and nutrients for the 'before' and 'after' periods.

concentrations has been a reduction at higher discharges. The high TN concentrations (>8 mg l^{-1}) observed in the period before riparian buffers were not measured afterwards (Fig. 3).

Annual TN exports were similar throughout the monitoring period, generally between 1 and 3 kg ha^{-1} yr^{-1}, although annual variability was large (Fig. 6). The high loads in 1991 and 1992 were probably due to the large events. Throughout the monitored period the TN loads exceeded the areal weighted equivalent target of <0.36 kg ha^{-1} yr^{-1} set by the WAEPA (1990).

5. Discussion

This study compared sediment and nutrient concentrations and loads before and after riparian management on a 1.7 km stream reach. SS concentrations and exports were reduced by at least an order of magnitude. Observations suggest that this was the result of reduced bank erosion and increased channel stability. Before riparian management the channel was degraded, with steep, bare eroding banks. Following fencing and planting of the riparian area, the channel form stabilised. Coarse unconsolidated soils, such as the sandy soils in this catchment, are susceptible to erosion by cattle (Mosley et al., 1997). Surface sediment delivery is an unlikely sediment source in this catchment as limited surface runoff coupled with a flat floodplain with dense perennial pastures and riparian grass cover significantly reduces the opportunity.

The significant reductions in SS concentrations and exports following riparian management, found in this study are similar to findings from previous studies. Riparian fencing, with buffers of varying widths, reduced sediment exports from a 79 km^2 catchment in New Zealand by 84%, from an estimated 1.28 to 0.20 kg ha^{-1} (Williamson et al., 1996). Similarly, fencing of riparian areas and exclusion of cattle from the stream in a 26 ha Ohio catchment of unimproved pasture reduced annual FWMC by 60% and the average annual sediment losses from the catchment by 40%, from 2500 to 1400 kg ha^{-1} (Owens et al., 1996).

The potential of riparian buffers to reduce P exports in catchments with sandy, low P retention soils, is less certain. Total P concentrations and loads did not alter significantly after the riparian buffers were fenced and planted. In contrast, FRP concentrations increased significantly, and exports during the period 1997–1999 were greater than those between 1994 and 1996.

Concentrations in soils, bank material and runoff provide some insights into likely sources of P in this catchment. During the before period around half of TP travelled with sediment >0.45 μm and the SS 25th and 75th percentile concentrations were 17 and 183 mg l^{-1}, respectively. If this SS was derived from surface soil (Table 1), then the >0.45 μm P concentration would be between 0.004 and 0.038 mg l^{-1}, and the TP concentration between 0.007 and 0.077 mg l^{-1}. If the SS was derived from bank material (Table 1), then >0.45 μm P concentration would be between 0.001 and 0.014 mg l^{-1} and the TP concentration between 0.003 and 0.028 mg l^{-1}. The TP 25th and 75th percentile concentrations were 0.17 and 0.60 mg l^{-1}, respectively, at least an order of magnitude greater than the estimated surface soil and bank material derived P, so we can conclude that there are additional P sources.

In addition, P exports remained the same throughout the study period, despite a significant decrease in SS, suggesting also that little P was derived from channel material. A parallel decrease in TP with SS would have been observed if the P was derived from channel material. Phosphorus is most likely leached through the sandy, low Fe soils, which may have reached P saturation (Table 1) and receive annual applications of fertiliser (Weaver and Reed, 1998). The FRP/TP ratio has most likely increased because the amount of exposed channel sediment has decreased, reducing the availability of sorption sites for leached soluble P.

Finally, there has been no change in the fertiliser history for the past 20 years and little soil testing has been conducted, although there was evidence in 1991, that the soils had high P status. High status, using Fe-AmOx classes and soil bicarbonate P, represents soil where a response to P applications will not be seen (Weaver and Reed, 1998).

There are several reasons why riparian buffers in this catchment may not reduce P exports. First, P losses from the catchment are dominated by FRP and filterable P loads are less amenable to reduction by riparian vegetation than particulate P loads (Muscutt et al., 1993) and retention is often rather low (Dillaha et al., 1989; Vought et al., 1994; Uusi-Kämppä and Yläranta, 1996). Secondly, the hydrological flowpaths on these duplex soils may affect the potential of riparian buffers for reducing nutrient exports. Subsurface flowpaths can dominate over surface runoff in duplex soils (George and Conacher, 1993). Monitoring of surface and subsurface flow in a nearby catchment shows that subsurface flowpaths can dominate P transport in this environment (unpublished data). Subsurface pathways may also dominate P transport in this catchment and this would reduce the functionality of riparian buffers, particularly the direct filtering of surface runoff. Thirdly, the soils in this catchment cannot retain large amounts of P. Soil samples collected in the catchment show that the soils have low P sorption, generally being below 400 mg kg^{-1} Fe-AmOx and having a very low PRI (Table 1). Using the single point P sorption measure, PRI, two thirds of samples collected in the North Willyung catchment could be described as very weakly adsorbing or desorbing to weakly adsorbing.

The TN data was less clear, although there were reductions in TN concentrations at higher discharges after the riparian buffers were planted. Reductions at higher discharges may be due to a reduction in particulate N with reduced stream bed and bank erosion, and reduced animal manure supply after stock exclusion. Williamson et al. (1996) observed decreases in particulate N and Line et al. (2000) recorded decreases in TKN and associated these decreases with reduced erosion following riparian fencing and livestock exclusion. Reductions in TN concentrations and loads were anticipated following improved riparian management due to reductions in the amount of cattle urine and faeces entering the stream, increased trapping of particulate N in surface runoff, increased in-stream nutrient uptake, removal of pasture legumes (which may bleed N) from the stream banks and improved denitrification conditions. Given that only the high TN concentrations were reduced it is unlikely that the denitrification potential increased following improved riparian management.

There are uncertainties in attributing causes to the observed changes due to differences in rainfall and runoff between years, errors in measurement, and changes in management and history. Farmers altered land management practices during the study period, for example, stock type changes (Table 2). In addition, only one segment of the stream was retired from grazing, and management of the upstream area, in particular Farm 2, did not remain constant during the after period. Sheep replaced cattle in paddocks with stream access on Farm 2 in 1998. Sheep are not attracted to streams like cattle, and prefer to graze drier soils. Despite this change in land management, unpublished data collected at additional sites in the catchment between 1997 and 2000 show that the highest nutrient concentrations occurred between the start and end of the stream reach on Farm 4 (unpublished data).

The introduction of automated sampling may also have influenced the results of this study. Before 1997 only grab and stage height samples were collected and some events were therefore not sampled. Comparison of concentrations collected by stage height and auto samplers during the after period shows that there were only significant differences for SS samples. The 146 auto samples (median 8.2 mg l^{-1}) had significantly lower concentrations (Mann Whitney, $p < 0.001$) than the 49 stage height samples (median 23 mg l^{-1}). However, both after sample types had concentrations significantly lower (Mann Whitney, $p < 0.001$) than the 199 before stage height samples (median 137 mg l^{-1}) suggesting that either sampling method would have produced the conclusion that SS concentration reductions were at least an order of magnitude and therefore cannot be disputed. Concentration reductions are also independent of any errors in discharge measurement or load calculations.

Replacement of the discharge structure may introduce errors into the load estimates. The original discharge structure (a compound 30°/120° sharp-crested v-notch weir) was drowned during large events and sand filled the weir pool. In 1996, when the potential for a before and after investigation arose the structure was replaced with a 150° broad crested weir, a design better suited to the flow and high sediment load conditions. However, errors in discharge measurement are likely to be small in comparison with inter-annual variability.

Total P and TN exports from this catchment are lower than those reported for other small grazed catchments in southern Australia (Nelson et al., 1996; Fleming and Cox, 1998). However, they may still be too high for the sensitive receiving environment of Oyster Harbour. Areal weighted equivalent export targets of <0.36 kg N ha^{-1} yr^{-1} and <0.05 kg P ha^{-1} yr^{-1} were set for the Oyster Harbour catchment (WAEPA, 1990). Although such guideline values are unlikely to be applicable to the entire catchment, particularly areas remaining uncleared, they can help to identify whether a catchment is in excess or deficit of the targets. The North Willyung catchment is clearly exporting excess nutrients to the King River. This is of particular concern as the monitoring site is 1.6 km from the King River, which exits into Oyster Harbour a further 5.2 km downstream. It is important to reduce nutrient exports, particularly from catchments in close proximity to receiving water bodies, as there may be limited opportunities for assimilation, as travel times are shorter.

Riparian buffers at the catchment scale are rarely as effective as plot scale investigations suggest they might be. A buffer's ability to trap pollutants may be exceeded by concentrated flow or saturation of riparian soils (Fitzpatrick (1984), cited in Norris, 1993; Dillaha et al., 1989). In this catchment, water logging of riparian areas during the winter may further reduce the ability of riparian vegetation to filter sediment and nutrients from surface runoff. In addition, drains and ditches are used throughout the catchment to reduce the severity and occurrence of water logging. Bypassing of runoff via drains and ditches can reduce the effectiveness of riparian buffers (Hitchman and Haycock (1994), cited in Haycock and Muscutt, 1995; Blackwell et al., 1999).

Even within first order catchments water quality management must be catchment wide and not rely on one measure, for example riparian buffers. Riparian buffers have been shown to be successful for reducing sediment exports in the North Willyung, however, additional water and water quality management is required to address excessive N and P exports.

For P, source control may be one of a few options available. Reassessment of the timing, rate and method of fertiliser applications, along with less water soluble and nutrient specific fertilisers may provide some opportunity for reducing P exports (Weaver et al., 1988; Weaver and Reed, 1998). Increasing the use of perennial pastures may also assist by reducing discharge, and therefore nutrient loss from the catchment. Historical P accumulation in the soils and waterways of catchments such as the North Willyung, mean that even with radical changes in fertiliser management, exports may not reduce to target levels for some time (Ritchie and Weaver, 1993). Further work on source identification is necessary for targeting measures to reduce TN concentrations and exports.

Riparian buffers were valuable for reducing stream sediment, and possibly N exports in this catchment, largely by removing or displacing sediment producing activities back from the stream. Riparian buffers may also remove the risk of direct fertiliser application to the stream and slow the transport between pasture and stream. Direct application may be particularly important where fertilisers are broadcast by truck or aerial top-dressing (Cooke, 1988).

6. Conclusion

This study on the water quality impacts of improved riparian management has shown that large reductions in SS can be achieved with riparian fencing, tree planting and livestock exclusion. SS concentrations and loads were at least 90% lower after the riparian buffers were created. Reductions in TP exports were not evident and the amount of P moving associated with particles <0.45 μm or in dissolved form increased following improved riparian management. Nitrogen concentration reductions were apparent at high flows.

Riparian fencing and livestock exclusion are likely to be valuable for reducing stream sediment. However, in catchments with sandy, low P retention soils, which receive annual fertiliser applications for legume based pastures, which be dominated by subsurface hydrological pathways, reductions in P and N are less certain. Additional water quality control tools will be required to reduce agricultural P and N exports in catchments with these characteristics.

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