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

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SUMMARY: Riparian vegetation can trap sediment and nutrients coming from hillslopes and reduce stream bank erosion. This study presents results from a 10 year stream monitoring program in a small agricultural catchment near Albany, Western Australia. In 1996, a 1.6 km stream reach was fenced, planted with eucalyptus species and managed separately from the adjacent paddocks. Stream flow, nutrient and sediment concentration data were collected at the downstream end of the fenced riparian area between 1991 and 2000, so there are data for the "before" and "after" riparian management periods. Suspended sediment concentrations fell dramatically following riparian management; the average event mean concentration (EMC) dropped from 254 mg/λ to 15.8 mg/λ. Maximum suspended sediment concentrations dropped by an order of magnitude. As a result, sediment exports from the catchment reduced noticeably following riparian management, mainly due to reduced stream bank erosion. In contrast, riparian management had limited impact on nutrient exports. There was no detectable change in total phosphorus EMC, a 67% increase in filterable reactive phosphorus average EMC and a 37% decrease in average total nitrogen EMC between the before and after periods. This study demonstrates the benefits of riparian management in reducing stream bank erosion, and suggests that in this environment, with sandy soils with low phosphorus retention, there are limitations on the effectiveness of riparian management for reducing nutrient exports.

THE MAIN POINTS OF THIS PAPER ARE:

- Riparian fencing and livestock exclusion can reduce suspended sediment concentrations and exports by an order of magnitude on sandy, unconsolidated soils.
- Total nitrogen concentrations and loads decreased following improved riparian management.
- No detectable change in phosphorus concentrations or transport occurred, and filterable reactive phosphorus increased, possibly due to a reduction in the availability of sediment sorption sites.

1. INTRODUCTION

Degradation of stream water quality has been linked to livestock grazing in many parts of the world. However, these negative impacts can be reduced or eliminated with correct management. One management option available to farmers and catchment managers is fencing of riparian areas. Riparian areas are lands directly adjacent to rivers, streams, ponds or lakes and perform three key roles in minimising the impacts of agriculture on stream water quality. Firstly, they help maintain the hydrologic and geomorphic integrity of the stream channel and associated soil and vegetation. Secondly, they help protect streams from upland sources of pollution by physically filtering and trapping sediment, nutrient and chemicals. Thirdly, they can displace sediment and nutrient producing activities back from streams.

Uncontrolled grazing of riparian areas by livestock, such as cattle can cause degradation of streams and their water quality. Cattle are attracted to riparian areas and may spend time in and around streams (Trimble and Mendel, 1995). 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).

The exclusion of cattle from streams has resulted in the recovery of channel form (e.g. Howard-Williams and Pickmere, 1994; Williamson et al., 1996; Mosley et al., 1997), but the water quality benefits are less certain. A few studies have documented large reductions in sediment yields as a result of riparian fencing, but few have assessed changes in nutrient export. Riparian fencing has been shown to reduce stream suspended sediment loads by between 40 and 85 % on relatively stable channels (Williamson et al., 1996; Owens et al., 1996, Line et al., 2000). Such reductions appear to be mainly 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 (e.g. Howard-Williams and Pickmere, 1994). However, channel stability can be important factor in determining the effect of riparian fencing. Williamson et al. (1992) found that on larger, actively meandering channels, riparian retirement had comparatively little benefit because any retirement or grazing effects were rapidly overtaken by channel migration.

Nutrient reductions following fencing of riparian zones have been more variable. Both Line et al. (2000) and Williamson et al. (1996) reported significant reductions in phosphorus loads. However, reductions of nitrogen were not great, and there were also changes in chemical form. Williamson et al. (1996) noted that prior to fencing nitrate and organic nitrogen loads were roughly equal, but after fencing nitrate exports dominated nitrogen exports.

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This study 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. Nutrient export targets were set by the Western Australia EPA (1990) to halt the decline of seagrass communities caused by excessive algal growth. Annual targets of <13.9 tonnes of phosphorus (<0.05 kg/ha) and <107.9 tonnes of nitrogen (<0.36 N kg/ha) were set for Oyster Harbour and agriculture was allocated 99% of the target load requiring load reductions of >50% (EPA, 1990). Three key questions form the basis of this paper:

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

2. STUDY CATCHMENT

The study was undertaken in a small (5.9 km^2) agricultural catchment near Albany, Western Australia (**Figure 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 (ASL), but most of the catchment is below 70 m ASL. Soils are duplex with shallow sands overlying laterite, gravels and clay on valley slopes and deep sands on the valley floors (Churchward et al., 1988). Soils in the catchment typically have low reactive iron (Fe) concentrations, indicative of low phosphorus retention (Weaver and Reed, 1998).

The area has a Mediterranean climate with cool, wet winters and dry, warm to hot summers. Average annual rainfall at Albany Airport is 803 mm and average annual pan evaporation is 1387 mm (36.8 and 32.3 years of record respectively; BOM, 2000). Most of the rain falls between April and October and there are an average of 175 rain days per year (BOM, 2000).

The North Willyung catchment was cleared in the 1950's for pastoral use. Present stocking rates are 1 cow and calf per hectare (Farms 1, 3, and 4) and 12 ewes per hectare (Farm 2). During the monitoring period several land use changes occurred. Blocks of blue gum (*Eucalyptus globulus*) were planted during 1996, and 1997, and sheep replaced cattle in 1998 on Farm 2. Riparian management was implemented on Farm 4 in 1996. The stream channel was fenced along most of its length, and the riparian area planted with eucalyptus species. Stock access to the stream was limited to five watering points and numerous stock crossings were installed to remove stock traffic from the stream (**Figure 1**). The remainder of the stream, on other farms, was left unfenced and cattle and sheep had unlimited access to the stream.

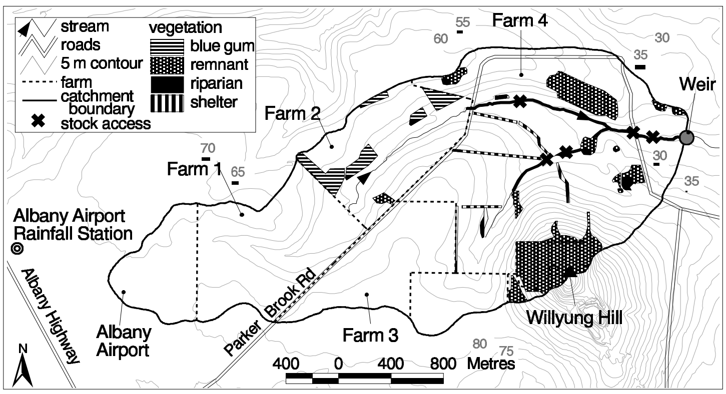

Figure 1: *North Willyung catchment.*

Fertiliser application timing and rates vary between farms and limited soil nutrient status testing is carried out. Farm 1 has received no fertiliser for the past 12 years. Farms 2 and 4 have received annual applications of 5:1 super potash in April at 160 kg/ha and 180 kg/ha, respectively. Farm 3 is also fertilised in April with 3:1 super potash at 55 kg/ha. On Farm 4 fertiliser is also applied on 30 ha of hay paddocks at 160 kg/ha in September. Lime has been applied on Farm 4 for the past 20 years; 0.2 t/ha/yr over the last 5 years and prior to that 0.1 t/ha/yr. Farm 2 also received lime in March 2000 at a rate of 2.5 t/ha.

3. METHODS

A monitoring station was established in 1991, and six years of flow and water quality data were collected before riparian management was initiated in 1996. Four more years of data were collected at the same site after riparian management was implemented.

3.1 Rainfall and flow measurement

Rainfall data from Albany Airport, (Bureau of Meteorology (BOM), station 9741), which is just outside the catchment boundary, was used in this study. A compound 30º/120º sharp-crested v-notch weir was installed (**Figure 1**) at the monitoring site at the start of 1991 and instantaneous water levels were measured and recorded by a Wesdata capacitance probe and logger until 1997. A 150º broad crested weir replaced the original structure in November 1996. From 1997 onwards, water levels were measured every 15 min by a pressure transducer and stored on a data logger. Continuous measurements of water level were converted to instantaneous flow using theoretical stage-discharge rating curves (Bos, 1989).

3.2 Water quality sampling

Samples for water quality analysis were collected by three methods: manual grab samples, rising stage samplers (Guy and Norman, 1970) and automatically. The time interval between consecutive samples varied from as long as 4 weeks, to as short as 45 minutes during storm events.

Manual grab samples were taken on each site visit. Rising stage samplers provided event samples 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 samplers were installed on the downstream side of the sharp-crested weir plate and samples were retrieved during each site visit. When the broad-crested weir was constructed, five rising stage samplers were installed at staggered depths upstream of the weir. An automatic sampler was installed in June 1997. During events, the sampler was activated after a predetermined stage rise in the previous hour. Initially, samples were taken at set time intervals depending on the rate of water level rise or fall (no less than 45 min apart). This scheme proved unsatisfactory, and the sampling strategy was modified to collect samples after a predetermined rise or fall in stage.

3.3 Water quality analysis

Samples were refrigerated on return to the laboratory and sub-samples filtered for filterable reactive phosphorus (FRP) determination. All samples were analysed for total phosphorus (TP), FRP and suspended sediment (SS). Total nitrogen (TN) was also analysed from November 1991 onwards. TP and TN concentrations were determined on unfiltered samples following persulphate digestion. FRP was measured on filtered samples following reaction with molybdate reagents (Murphy and Riley, 1962). SS concentrations were measured gravimetrically using a 1.2 μm GF/C filter paper and a drying time of 24 hours at 105ºC (APHA, 1978).

3.4 EMC and load calculations

Event mean concentrations (EMC) were calculated for events with more than two samples and a good quality flow record. The EMC was calculated by dividing the total load for each event by the total event volume.

Nutrient and sediment loads were calculated by direct computation (volume times concentration) for all sites. This analysis was carried out using HYDSYS (Hydsys, 1999). The discrete time series chemistry was converted to a continuous variable by linear interpolation and instantaneous loads were calculated directly at each instantaneous water level measurement. Caution should be exercised when comparing annual loads between the two periods as significantly more events were sampled after the automatic samplers were installed.

4. RESULTS

4.1 General physiographic changes

Prior to riparian management in 1996, the stream was wide and shallow. The stream banks were steep and unstable. Remnant vegetation was limited and the banks were heavily grazed and trampled.

After fencing the riparian zone was rapidly colonised by bracken, reeds and grasses. The stream channel became choked in some sections and was cleared on several occasions to maintain the present channel course. Choking of the channel also increased in-channel water levels, and during significant events the channel overflowed and was forced outside the fenced riparian area.

4.2 Rainfall and streamflow

Rainfall totals were generally below the long-term average, with the exceptions of 1992 and 1996, which were wetter than average (**Table 1**). Runoff coefficients were typically around 0.1 throughout the monitoring period. These figures are consistent with others calculated for southern Western Australia (e.g. Sharma et al., 1980; Salama et al., 1993).

Table 1: *Annual rainfall and streamflow summary.*

Year	Rainfall (mm)	Runoff (mm)	Runoff coefficient
1991	638.6†	108.9†	0.16†
1992	931.4	155.9	0.17
1993	821.0	138.5	0.17
1994	675.0	94.4	0.14
1995	720.0	73.3	0.10
1996	831.2	99.5	0.12
1997	724.2	93.7	0.13
1998	772.8	70.0	0.09
1999	730.6	70.6	0.10
2000	607.6	45.8†	0.08†

† for period when runoff and rainfall were both recorded, i.e. 1991 starts 24 April and 2000 ends Sept 1.

4.3 Stream chemistry

Riparian management has had a large impact on SS concentrations. The average EMC dropped by 94% from 254 mg/λ to 15.8 mg/λ (**Table 2**), and the median raw concentration dropped from 54.9 mg/λ to 7.3 mg/λ (**Figure 2**). The range of values also decreased by an order of magnitude. During events SS concentrations were often greater than 200 mg/λ before riparian management, but after riparian management only extreme outliers were this concentrated, and the majority of samples measured had SS concentrations less than 20 mg/λ.

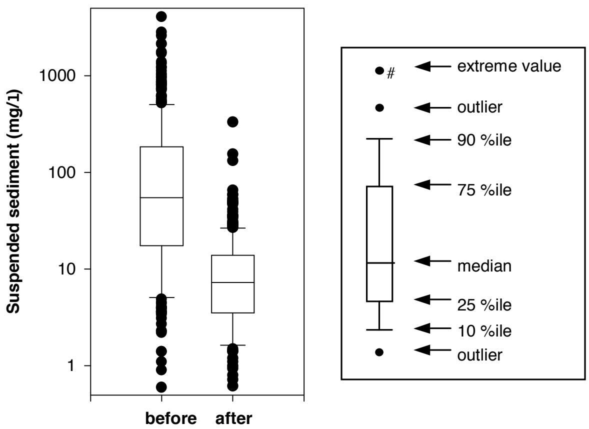

Figure 2: *Box plot of raw SS data for before and after periods and box plot explanation.*

Nutrient data shows that there was no detectable change in TP concentrations, an increase in FRP concentrations and a decrease in TN concentrations between the before and after periods. The EMC for TP were the same before and after improved riparian management. The range of TP concentrations measured did not alter either, a maximum of 2 mg/λ was the highest concentration measured for both periods (**Figure 3**).

Table 2: *Before and after EMC summary statistics and % reduction in average EMC.*

		SS EMC (mg/λ)	TN EMC (mg/λ)	TP EMC (mg/λ)	FRP EMC (mg/λ)
Before	*average*	254.42[a]	4.06 [b]	0.58	0.27[a]
	stdev	277.39	2.38	0.26	0.15
	no.	46	43	46	46
After	*average*	15.83[a]	2.55[b]	0.57	0.46[a]
	stdev	15.92	0.62	0.32	0.32
	no.	48	48	48	48
% reduction		94%	37%	2%	-67%

[a] average EMC are significantly different ($p<0.001$, t-test on log10 transformed data)
[b] median EMC are significantly different ($p<0.001$, Rank Sum Test on raw data, log10 transformed data failed normality test)

In contrast, FRP concentrations and concentration range increased following riparian management. The EMC increased 67% from 0.27 mg/λ to 0.46 mg/λ and the median raw concentration rose from 0.18 mg/λ to 0.35 mg/λ. The range of FRP concentrations also increased from 0-1 mg/λ to 0-1.7 mg/λ. An increase in the FRP:TP ratio also occurred (**Figure 3**). The increase in the ratio between the two monitoring periods suggests that more phosphorus is now travelling associated with particles <45 μm or in dissolved form.

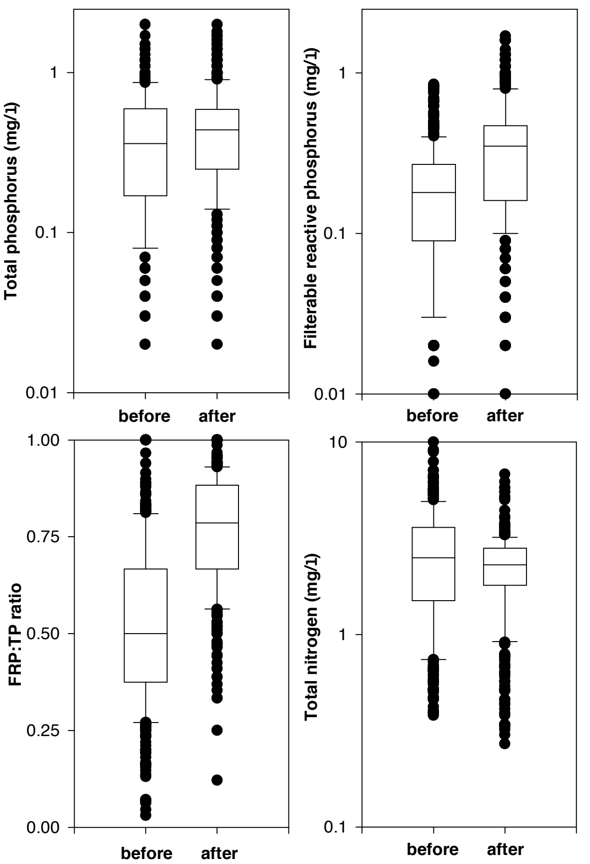

Figure 3: *Box plots of raw nutrient concentrations (all samples) for before and after.*

The TN concentration decreased between the before and after periods; the EMC decreased from 4.1 mg/λ to 2.5 mg/λ and the median raw concentration decreased from 2.5 mg/λ and 2.3 mg/λ. The range of TN concentrations was also affected by riparian management (**Figure 3**). The high TN concentrations (>8 mg/λ) observed in the period before riparian management were not measured afterwards.

4.4 Catchment exports

Annual loads have been estimated by direct computation and it is therefore likely that total loads for the before period are underestimated, as fewer events were sampled during this period.

Suspended sediment 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 153 kg/ha/yr, although annual variability was large (**Figure 4**). After riparian management the average sediment load dropped to 9 kg/ha/yr and annual variability was reduced.

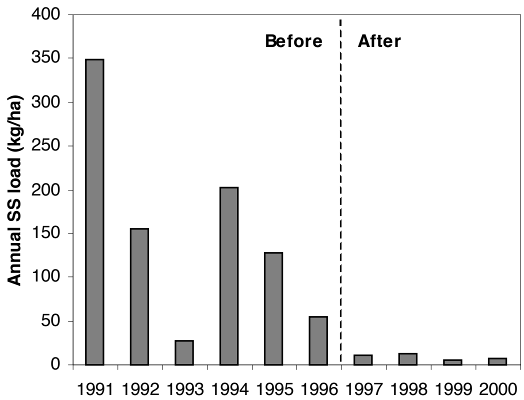

Figure 4: *Annual total SS exports.*

Since riparian management was implemented annual TN exports have decreased steadily from around 3 kg/ha to around 1 kg/ha, although annual variability was large (**Figure 5**). Throughout the monitored period the TN loads exceeded the target of <0.36 kg/ha/yr set by EPA (1990).

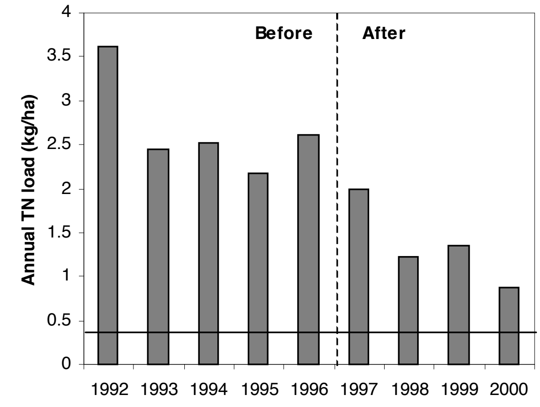

Figure 5: *Annual TN exports and target export (solid line).*

Total phosphorus exports from the catchment were similar throughout the monitoring period, generally between 0.2 kg/ha/yr and 0.5 kg/ha/yr. Total

phosphorus exports exceeded the export target of <0.05 kg/ha/yr set by EPA (1990) by up to an order of magnitude. Filterable reactive phosphorus exports remained around 0.2 kg/ha/yr throughout the monitoring period.

5. DISCUSSION

This study compared sediment and nutrient concentrations and loads before and after riparian management was implemented on a 1.6 km stream reach. There are uncertainties in attributing causes to the observed changes because we were unable to control other land management practices such as stocking rates, stock types and fertiliser practices. 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 do not appear to be attracted to streams 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 most notable rise in nutrient concentrations occurred between the start and end of the stream reach on Farm 4.

The significant reduction in SS concentrations and exports following riparian management found in this study is 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 average of 128 $kg/km^2/yr$ to 20 $kg/km^2/yr$ (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 the average annual sediment losses from the catchment by 40 %, from 2.5 Mg/ha to 1.4 Mg/ha (Owens et al., 1996).

Observations suggest that the change in SS concentrations and load is 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 zone, 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 erosion is an unlikely sediment source in this catchment as limited surface runoff coupled with a flat floodplain with thick pasture and riparian grass cover significantly reduces the opportunity.

Although rainfall totals were similar during the before and after periods, more runoff was recorded during the before period. While decreased runoff during the period following improved riparian management may have further contributed to the reductions observed in SS and TN, there is evidence that the SS reductions are valid. Runoff totals for 1991 through 1993 were around 150 mm annually, in contrast to totals of less than 90 mm/yr during the after period. However, by comparing annual exports for 1994/5 and the after period when runoff totals were similar, there is still support for the hypothesis that large reductions in SS were observed following riparian fencing.

The potential of riparian buffers to reduce nutrient exports in an environment with these characteristics is less certain. Phosphorus exports remained the same throughout the study period, suggesting that phosphorus is not derived from channel material. A parallel decrease in TP concentration with SS would have been observed if the phosphorus was derived from channel material. Phosphorus is most likely leached from the pasture through the sandy, low Fe soils, which may have reached phosphorus saturation (Weaver and Reed, 1998). The FRP:TP ratio may have increased because the sediment concentrations have decreased, reducing the availability of sorption sites for leached soluble phosphorus.

Total nitrogen concentration and export reductions are most likely due to a combination of a reduction in the amount of cattle urine and faeces entering the stream, increased trapping of particulate nitrogen in surface runoff and in-stream nutrient uptake.

Total phosphorus and TN exports from this catchment are lower than those reported for other small grazed catchments in southern Australia (e.g. Nelson et al., 1996; Fleming and Cox, 1998). However, regardless of this fact, they may still be too high for the sensitive receiving environment of Oyster Harbour. Nutrient exports from this catchment exceeded the catchment wide target levels set for the Oyster Harbour catchment (EPA, 1990), identifying the North Willyung as a nutrient “hot spot”. The greatest return, in terms of water quality improvements, will come from focusing nutrient management tools on small, degraded catchments with high exports, such as the North Willyung. This is particularly important for sub-catchments in close proximity to receiving water bodies, as there is little potential for improving the water quality of larger rivers.

The hydrological flowpaths on these duplex soils may affect the potential of riparian buffers for reducing nutrient exports. For example, a high proportion of runoff travels as subsurface flow (George and Conacher, 1993) and subsurface flowpaths dominate over surface runoff. This may reduce the functionality of riparian buffers, particularly the direct filtering of surface runoff. However, an additional function of riparian buffers is the displacement of sediment or nutrient producing activities away from streams. Removal of fertiliser application from riparian areas increases the transit time between pasture and stream and reduces the likelihood of direct fertiliser application to the stream. Direct application may be particularly important where aerial top-dressing is practiced (e.g. Cooke, 1988).

6. CONCLUSIONS

This study on the water quality impacts of improved riparian management has shown that reductions in SS and TN can be achieved with riparian fencing, tree planting and livestock exclusion. The riparian buffers reduced the average SS EMC by 94 % from 254 mg/λ to 16 mg/λ and the average TN EMC from 4 mg/λ to 2.5 mg/λ. Reductions in TP concentrations were not evident following the same management change. The amount of phosphorus moving associated with particles <45 μm or in dissolved form increased following riparian management. Riparian fencing and livestock exclusion are likely to be valuable for reducing stream sediment and nitrogen exports. However, reductions in phosphorus are less certain in catchments dominated by leaching through soils with low phosphorus retention, and subsurface hydrological pathways. Additional water quality control tools may be required to reduce agricultural phosphorus exports in catchments with these characteristics.

7. ACKNOWLEDGEMENTS

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