BENEFITS OF WETLANDS IN FARMING LANDSCAPES – THE TOENEPI CATCHMENT

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Abstract
The Toenepi catchment study (1995-97) was the first to look exclusively at the effects of pastoral dairy farming on water and soil. The success of that study can be gauged by the number of studies that followed and the almost continuous monitoring of the catchment outlet since studies began there. Currently, Toenepi is one of the five best practice dairying catchments and results show that the stream has undergone substantial improvements in water and habitat quality as a result of fencing (>85%) and a major change in disposal of treated farm dairy effluents (FDE) from predominantly pond discharge to land irrigation. A riparian wetland near the top of the catchment has been monitored recently for its capacity to remove nutrients and faecal bacteria (*E. coli*). Three weirs and five sampling sites were used to monitor flow and water quality changes, respectively, over three years. Prevailing oxidising conditions meant that little denitrification occurred within the wetland and hence, little net N removal. However, the wetland removed about 95% total P and particulate C, and 70% each of particulate N and *E. coli*. A combination of oxidising conditions favouring phosphate sorption, and the long, narrow shape of the wetland, providing good conditions for trapping particulate material, achieved high removal rates. Dissolved P was mostly transformed to plant and microbial particulate forms whereas a sizeable fraction of N was converted to dissolved organic N and nitrate, neither of which was retained by the wetland. Wetlands like this one are common in the headwaters of dairy catchments in the Waikato region and can provide a useful means of mitigating catchment loads, particularly if coupled with down-slope wetlands where denitrification may occur to a significant extent.

Introduction

The Toenepi Stream and catchment has been monitored almost continuously for water quality and quantity since 1995. It was included in the "Best Practice Dairying Catchments" project in 2001, along with four other dairy catchments in which surveys of farming practices and soil quality are combined with the water monitoring data to define linkages between land use and water quality, e.g., fertiliser use and methods for treating and disposing of farm dairy effluent (FDE) (Wilcock et al. 1999; 2006). A summary of nine years of monitoring (Wilcock et al. 2006) established that water clarity measured by black disc had improved from 0.6 m to 1.5 m, and median ammonia-N and nitrate-N concentrations had declined by 70% and 57%, respectively. *Escherichia coli* concentrations had not abated and were on average 2-3 times the recommended guideline values for contact recreation. Likely causes of the improvement in water quality were an increased proportion of farms irrigating FDE to land with fewer discharges of oxidation pond effluent to the stream, and improved fencing of the main stream (Wilcock et al. 2006). A study of longitudinal inputs to the Toenepi Stream (Wilcock & Singleton 2006) established that loads of N, P and faecal bacteria in winter were mostly from headwater and upper catchment inflows, whereas spring loads were more uniformly distributed along the stream length, reflecting differences in farming activities for each season. Winter inputs of N comprise the bulk of annual N loadings and it was considered that targeting the headwater sub-catchment of Toenepi Stream for nitrate-N (NO_X-N) removal was likely to be a cost-effective approach to rectifying high NO_X-N inputs to this stream and others like it in the same region (Wilcock & Singleton 2006).

Natural wetlands are known for their capacity to remove N by denitrification in reducing soil conditions (e.g., Groffman et al. 2002; Zaman et al. 2008) and are a common feature of the upper Toenepi catchment. However, they are often a nuisance to farmers who have to extricate cattle from the wetlands. Consequently, many natural wetlands on dairy farms may either be drained or fenced off. In this study, we analyse the performance of a wetland seep-and-swale system from which stock has been permanently excluded and the riparian areas planted with native wetland species, to attenuate nutrients and *E. coli* in surface and subsurface flows from surrounding dairy pasture. We examine the wetland's capacity to reduce contaminant loads to downstream waters under different hydraulic loadings and ask "what would the consequences be of having all such wetlands in the headwater catchment treated this way?" The study looks at the efficacy of a targeted BMP approach compared with broad-brush, catchment-wide strategies.

Methods

Experimental site

The wetland is located near the crown of the Toenepi catchment in the Waikato region near Morrinsville (Wilcock et al. 1999). Annual rainfall is 1377 mm and the topography is predominantly (>90%) gently sloping-to-undulating (slopes 5–10%), to rolling-to-steep (>15%) (Müller et al. 2010). The wetland is an approximately 600 m long upland seepage area within an established ryegrass (*Lolium perenne*) and clover (*Trifolium repens*) pasture catchment. The upper reach is an unfenced 10-20 m wide x 250 m long wetland that flows for 10 months of the year and is accessible to grazing cattle. The experimental site, separated from the upper reach by a boundary fence (Fig. 1), comprised a ponded section (c. 20 m x 20 m) extending down-slope as a 1-3 m wide x 340 m long swale. Wetland vegetation was mainly glaucous sweet grass (*Glyceria declinata*), jointed rush (*Juncus effusus*), sedge (*Carex* sp.), and lotus (*Lotus pedunculatis*). The dominant soils are clay textured ash soils (Morrinsville soils; NZ Soil Classification: Typic Orthic Granular Soils) (Müller et al. 2010). In 2007, the experimental wetland was fenced on both sides and 1700 native plants added (*Dodonea viscosa*, *Cordyline australis*, *Coprosma robusta*, *Pittosporum tenuifolium*, *Hoheria sextylosa*, *Plagianthus regius*, *Hoheria Populnea*, *Phormium tenax*, *Phormium cookyanum* and *Cortaderia fulvida*), providing a 5-10 m buffer zone between the wetland and the surrounding dairy pasture. Flow monitoring weirs were established in three positions (upstream, Q_2; middle, Q_3; and downstream, Q_5) and samples collected initially fortnightly and then at monthly intervals from the weirs, as well as from two other sites (S_1-S_5), giving longitudinal changes in flow and water quality in the wetland, during June 2007–June 2010 (Fig. 1). Flow for the middle weir (Q_3) was provided by an automatic flow recorder giving discharge at 10 min intervals (Müller et al. 2010). Distances between sites were 85 m (S_1-S_2), 156 m (S_2-S_3), 50 m (S_3-S_4) and 46 m (S_4-S_5). Samples were stabilised immediately after collection, either by filtration and refrigeration (dissolved constituents) or by freezing (total and particulate constituents), prior to analysis for total N (TN), nitrate plus nitrite N (NO_X-N), total ammoniacal N (NH_4-N), total phosphorus (TP), filterable reactive P (FRP), particulate C (PC) and N (PN) by standard methods (APHA, 1998). *Escherichia coli* concentrations were determined by the Colilert most probable number (MPN) method (IDEXX Laboratories, United States). Dissolved organic N (DON) was calculated from the difference between TN and the sum of NO_X-N, NH_4-N and PN. Dissolved oxygen was monitored (Hach portable DO meter model HQ40d) monthly at site 5 (S_5), and at the three weir sites (S_2, S_3 and S_5) during winter 2010. A survey determined the total wetland drainage area (catchment) at the downstream weir site (S_5) to be 11.4 ha and the surface water slope approximately 2.4%.

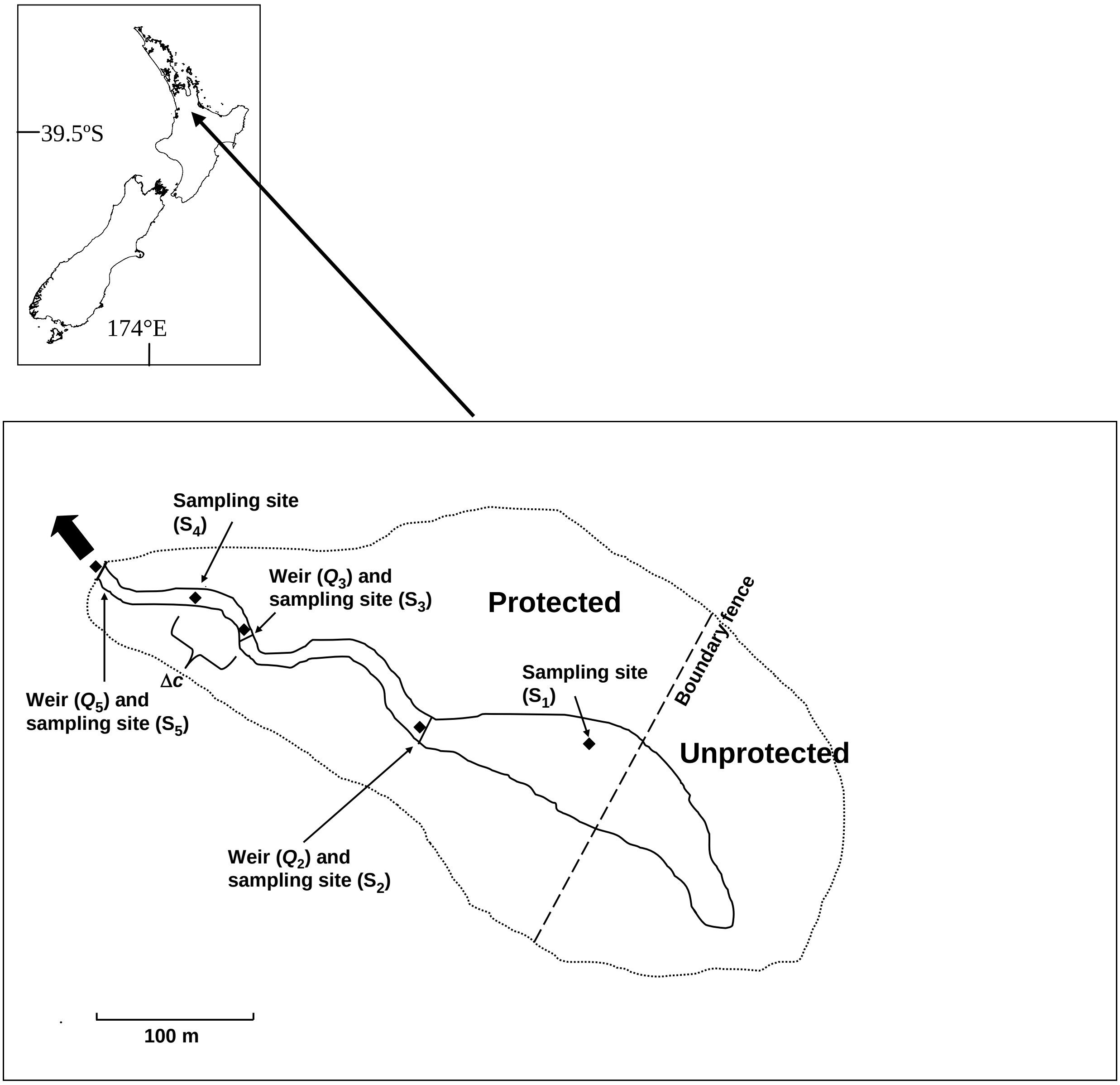

Fig. 1 Schematic of the Toenepi headwater wetland showing the approximate location of the three weirs (–) and five water quality sampling sites ⧫), as well as the reach between S $_3$ and S_4 where increases in concentrations (Δ*c*) of N, P and *E. coli* occurred. The catchment boundary (fine dotted line) and boundary fence (broken line) between the protected and the unprotected upstream part of the wetland are also shown, as well as the general location of the wetland in the North Island, New Zealand (inset).

Estimation of nutrient retention by the wetland

Nutrient retention occurs when the nutrient loading of waters draining from a wetland is less than the nutrient loading of waters entering the wetland. The term ‘retention’ includes physical, chemical and biological processes that retain contaminants in a form not readily released under normal conditions (Reddy et al. 1999). These include physical and chemical adsorption, plant uptake (of N and P) and microbially mediated processes like coupled nitrification–denitrification. Retention of N and P was calculated from differences in mass-flow (i.e. the product of instantaneous concentration and flow). Increases in flow (Δq) and concentration (Δc) between S_3 and S_4, were accounted for by a simple mass balance (Eq. 1).

$$\Delta c \Delta q = c_4 Q_4 - c_3 Q_3 \qquad (1)$$

c_3 and c_4 are concentrations (mg m^{-3} except for PC (g m^{-3}) and *E. coli* (MPN 100 mL^{-1})) at S_3 and S_4, and Q_4 is the mean of Q_3 and Q_5 (m^3). Differences between annual input (at sites S_1 to S_4) and output loads (at S_5) were calculated from products of flow-weighted mean concentration and true mean flow (Fergusson 1987) summed over 12 month periods (ΔT) (Eq. 2).

$$\Delta L_{1\text{-}5} = \left(\frac{\sum c_1 Q_2}{\sum Q_2}\right)\overline{Q}_2 \Delta T + \left(\frac{\sum c_4 Q_4 - c_3 Q_3}{\sum Q_4}\right)\overline{Q}_4 \Delta T - \left(\frac{\sum c_5 Q_5}{\sum Q_5}\right)\overline{Q}_5 \Delta T \qquad (2)$$

$\overline{Q}_2$, $\overline{Q}_4$ and $\overline{Q}_5$ are mean annual flows based on the 10 minute flow record at Q_3. Loads were also calculated from the products of instantaneous concentrations and the mean discharge for each sampling period. This method is recommended by Johnes (2007) for P load estimation in lowland clay and permeable catchments with monthly sampling data. The method gave slightly smaller loads than Eq. 2 because it did not take into account the whole flow range (incorporated by $\overline{Q}_2$, $\overline{Q}_4$ and $\overline{Q}_5$ in Eq. 2) but gave identical values for % retentions of N and P. Annual retentions of N and P forms (kg yr^{-1}), and *E. coli* (billions yr^{-1}) were calculated as percentages of total inputs at S_1 for each year. Re-suspension of organic matter caused by sampling within the wetland at S_1 sometimes resulted in samples being very turbid by comparison with those taken at downstream sites and because of that, had extremely high concentrations of particulate N and P relative to those at sites S_2-S_5. It seems probable that the re-suspended material would quickly settle out and not be readily transported down the wetland. Therefore, to avoid overestimating wetland retentions that might be biased by high initial TN and TP concentrations at S_1 we also calculated annual input and output loads and cumulative retentions for the S_2-S_5 reach (Eq. 3), expressed as percentages of total input loads.

$$\Delta L_{2\text{-}5} = \left(\frac{\sum c_2 Q_2}{\sum Q_2}\right)\overline{Q}_2 \Delta T + \left(\frac{\sum c_4 Q_4 - c_3 Q_3}{\sum Q_4}\right)\overline{Q}_4 \Delta T - \left(\frac{\sum c_5 Q_5}{\sum Q_5}\right)\overline{Q}_5 \Delta T \qquad (3)$$

Mean flows for each year ($\overline{Q}_2$, $\overline{Q}_3$, $\overline{Q}_4$ and $\overline{Q}_5$) were obtained from daily flows, which were integrated from the 10-min flow recordings at Q_3, calibrated with instantaneous flows measured at the three weirs (Fig. 1), and from regressions of weir flows with measurements at a continuous flow recording site 6 km downstream (R^2>0.94 in all cases). Retentions were calculated for TN, NO_X-N, NH_4-N, TP, FRP, PC, PN and *E. coli* for the 2007-08, 2008-09 and 2009-10 years, as well as for the entire three-year period. Specific yields were calculated from annual output loads and the catchment area at S_5 (11.4 ha).

Results

Water quality concentrations varied widely at all sites, particularly for particulate forms (Table 1). Dissolved N was the largest component of TN (median 80%) with about 60% of this being DON, and the remainder comprising NO_X-N (16%) and NH_4-N (4%). Particulate N comprised c. 20% of TN. The composition of TN varied seasonally with DON always the dominant form and the second largest components being PN in summer-autumn, and NO_X-N in winter (Fig. 2). The proportion of TP that was FRP ranged from 5% in summer (December-February) to 25% in winter (June-August) and averaged about 10% year-round. The remaining TP comprised particulate and other dissolved forms, viz. dissolved organic P (DOP).

Median concentrations of NH_4-N and FRP exhibited little change down the wetland, but NO_X-N was more variable. Median concentrations of DON were consistently about 300 mg m^{-3} at all sites. By contrast, particulate forms (PN, TP and PC) generally decreased through S_1-S_3, increased at S_4 then declined at S_5, indicating that much of this material was retained and potentially recycled within the wetland. *E. coli* concentration ranges varied greatly at most sites, whilst median concentrations were generally low (20-50 MPN 100 ml^{-1}) and similar between sites (Table 1). Water flows (0-46 L s^{-1}) had a pronounced seasonal variation. Mean annual flow (±SD) at the recorder site Q_3 was 1.66±4.92 L s^{-1}, and mean summer (Dec-Feb) and winter (Jun-Aug) flows were 0.22±0.64 and 4.60±7.96 L s^{-1}, respectively. Flows at sites 2 and 3 were similar (Q_2:Q_3 = 1.03±0.05), but flows were consistently different between sites 3 and 5 (Q_3:Q_5 =1.41±0.06).

Dissolved oxygen in wetland water co-varied with flow (Spearman rank correlation coefficient 0.72, N=32), with winter values of 60-70% saturation and summer values of 2-4%. Thus, wetland waters were aerobic for much of the year when contaminants were mobile. Mass-flow distributions of TN and TP at each site declined initially between S_1 and S_3, then increased at S_4, and decreased again between S_4 and S_5, consistent with the concentration increase for most variables at S_4 and indicated by Δc in Figure 1.

Retention of contaminants

Mass-flows of TN and TP generally decreased from S_1 to S_3, increased at S_4 as a result of inflow between S_3 and S_5, and then decreased between S_4 and S_5. First-order retention coefficients (Reddy et al. 1999) were calculated for TN and TP mass-flows between reaches S_1-S_3 and S_2-S_3, where flows were mostly constant, and S_4-S_5. For TN the coefficients (±95% CL) were 0.0054±0.0020, 0.0027±0.0016 and 0.0026±0.0016 m^{-1} for S_1-S_3, S_2-S_3 and S_4-S_5, respectively, whereas the corresponding values for TP were 0.014±0.003, 0.009±0.004 and 0.014±0.008 m^{-1}. These coefficients indicate greater retention of TN between the first two sites than at sites downstream of S_2, possibly due to settling of disturbed suspensoids. TP retention coefficients were similar for the three reaches, indicating similar rates of loss per unit length throughout the wetland. Retention of TN for the entire wetland reach (S_1-S_5) was c. 50%, but much less when initial losses between the first two sites were discounted (5% over three years). Nitrate mass-flow exhibited an overall net gain in NO_X-N (c. 20 and 30% for reaches S_1-S_5 and S_2-S_5) that was not solely attributable to mineralisation of NH_4-N (Table 2). TP retentions were 97% for the reach S_1-S_5 and 93% (S_2-S_5) and similar to PN and PC retentions. *E. coli* retentions differed markedly for the three years and were undoubtedly influenced by the pattern of cattle grazing within the wetland catchment (Collins et al. 2007). Greater description of interannual variations in retentions and other details of the study is given in a manuscript submitted for publication (Wilcock et al. 2011).

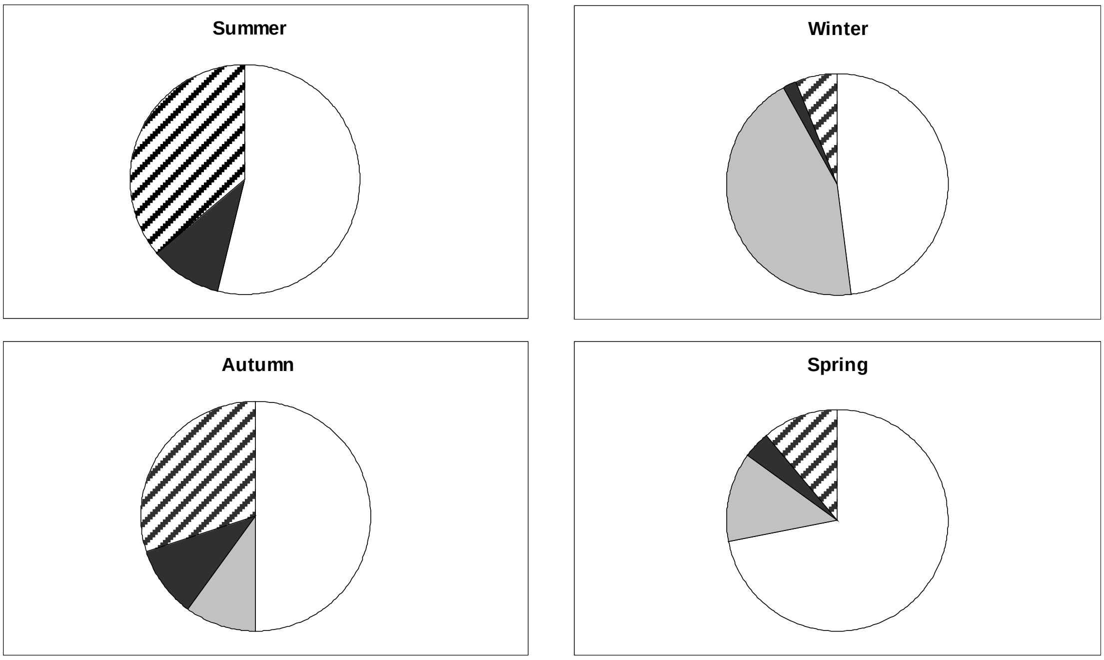

Fig. 2 Seasonal proportions of total nitrogen components at all sites in the experimental wetland during the study period June 2007-June 2010. Key: DON dissolved organic N (white); nitrate plus nitrite N NO_X-N (gray); total ammoniacal N NH_4-N (black) and particulate N PN (diagonal stripes).

Table 1 Water quality data summary statistics (range and **median**) for experimental wetland sites 1-5 (S_1-S_5) from monthly grab samples collected between June 2007–June 2010. Key: NH_4-N total ammoniacal N; NO_X-N nitrate plus nitrite N; PN particulate N; TN total N; FRP filterable reactive P; TP total P; PC particulate C; *E. coli*. Concentration units are mg m^{-3} except for PC (g m^{-3}) and *E. coli* (MPN/100 ml).

	NH_4-N	NO_X-N	PN	TN	FRP	TP	PC	*E. coli*
S_1	9-1420	1-3510	23-48900	210-75200	1-20	12-58200	0-3380	2-20000
	34	**10**	**969**	**1860**	**5**	**460**	**7.0**	**52**
S_2	9-840	1-3070	13-1620	262-6990	1-10	10-12300	0-183	1-2900
	45	**54**	**107**	**726**	**4**	**44**	**1.5**	**31**
S_3	6-125	1-2900	9-758	293-3320	1-8	9-685	0-23.2	1-450
	18	**58**	**29**	**553**	**5**	**16**	**0.2**	**30**
S_4	8-1220	1-3040	15-20800	429-8770	2-13	10-26000	0-726	1-15500
	25	**151**	**330**	**815**	**6**	**89**	**3.5**	**20**
S_5	1-795	1-2950	10-4820	336-8440	1-39	7-1060	0-37	1-2000
	27	**45**	**117**	**776**	**6**	**50**	**0.9**	**26**

Table 2 Annual % retentions of total N (TN), nitrate+nitrite N (NO_X-N), total P (TP), filterable reactive P (FRP) and *E. coli* between sites S_1 and S_5 (upper numbers) and between sites S_2 and S_5 (lower numbers) during the three-year period followed by one standard deviation. Annual input (IN) and output (OUT) loads are: kg for total N (TN), nitrate+nitrite N (NO_X-N), ammonium N (NH_4-N), total P (TP), filterable reactive P (FRP) and particulate N (PN); MPN x10^9 for *Escherichia coli* (*E. coli*) and tonnes for particulate C (PC)

	2007-2010		Three-year
	IN	OUT	Retention (%)
TN	420±40	220±20	48±10
	230±20		5±1
NO_X-N	150±15	185±20	-20±3
	140±14		-30±5
NH_4-N	6±2	5±2	17±7
	8±2		37±10
TP	210±21	6±1	97±14
	80±8		93±13
FRP	0.7±0.2	0.9±0.3	-25±15
	0.6±0.2		-50±25
E. coli	480±50	100±10	80±10
	290±30		65±9
PN	190±30	10±2	95±20
	46±7		78±16
PC	7.9±1.2	0.13±0.03	98±20
	2.9±0.4		96±20

Discussion

On average, the wetland-swale removed 93% of TP and 65% of *E. coli* from the wetland between S_2 and S_5, and acted as an effective trap for suspended particulate N such that about 80% of waterborne PN was retained within the wetland. Only 5% of TN transported down the narrow swale below S_2 was retained or removed, which may be attributable in part to a smaller cross-section area and hence, higher velocities and shorter retention times (Wilcock et al. 2002; Jordan et al. 2003). The protected wetland had a surface area of about 800±100 m^2, an average depth 0.5±0.2 m, and a mean annual flow (±95% CL) at Q_3 of 1.66±0.30 L s^{-1}, giving a mean residence time (volume÷flow) of 3±1 d and a mean velocity of 0.001±0.0006 m s^{-1} (106±64 m d^{-1}). First-order rate constants for TN and TP based on the steady-state plug flow model (Kadlec 1994) were 50-100 and 170-270 m yr^{-1}, respectively. In-stream removal of N by plants in New Zealand streams declines linearly with flow (Howard-Williams and Pickmere 2010) such that a flow rate of about 2 L s^{-1} corresponds to a first-order rate constant of 50-100 m yr^{-1}, as was observed here. A key finding of our study is that very high retentions of TP, PN and *E. coli* occurred because large fractions of N, P and *E. coli* were in particulate form and able to be trapped during passage through the long wetland path length (360 m). From a consideration of nutrient spiralling, the average distance travelled by a molecule before being removed from the water column (or uptake length) is given by the reciprocal of the spatial first-order retention coefficients (m^{-1}) (Newbold et al. 1981). Average uptake lengths for TP and TN were 70-110 m and 185-385 m, respectively, equivalent to 2-5 cycles for TP and 1-2 cycles for TN for nutrients travelling from S_1 to S_5. Results from this study are compared with other wetland studies (Table 3), where pollutant retention varies greatly according to hydraulic loading, duration of inundation, pollutant loading and contact time with wetland plants and soils (Reddy et al. 1999; Knox et al. 2008). The comparison highlights the capacity of our wetland to trap and retain particulate forms to a high degree but perform rather poorly with dissolved forms of N and P. A review of 57 wetlands (Fisher and

Acreman 2004) found that 80% of them reduced N loadings with a mean retention (±SD) of 67±27%, and 84% reduced P loadings by 58±23%. Fisher and Acreman (2004) speculated that P binding sites in wetlands may become saturated and limit their efficacy for P removal. The wetland discussed here has been in a developed pasture catchment for c. 100 years, receiving regular inputs of superphosphate fertiliser at an average rate of 60-80 kg P ha^{-1} yr^{-1} for many decades (Wilcock et al. 2006). Our monitoring of the wetland does not indicate a decline in the capacity of the wetland to remove P that is largely converted to particulate form. We cannot discount that the three weirs may have acted as sediment traps, thereby contributing to the observed retentions. However, long meandering wetlands within pasture catchments such as this have many natural barriers as well as those created by culverts and farm tracks so that the weirs were in effect just three additional impediments to flow.

Table 3 Comparison between retentions (%) of TN total N, NO_X-N nitrate plus nitrite N, NH_4-N total ammoniacal N, FRP filterable reactive P, TP total P, PC particulate C and *E. coli* for this study (sites S_2-S_5) over the three-year period 2007-2010, with others natural and constructed wetland studies used for treating agricultural runoff.

TN	NO_X-N	NH_4-N	FRP	TP	PC	*E. coli*	Reference
35-42	60	–	35-42	35-42	–	68	Knox et al. 2008
14	52	25	18	27	34	–	Jordan et al. 2003
55	51	38	41	34	71	–	Kadlec and Knight 1996
68	33	93	-49	76	–		Tanner et al. 2005[2]
5	-30	37	-50	93	96	65	This study 2007-2010
–	–	–	–	37	–	–	Bhada et al. 2010

[1] Cited in Jordan et al. 2003; [2] Retentions calculated for two years

Other studies in this catchment (Stenger et al. 2008; Müller et al. 2010) reported that inputs of N and P to either this wetland or to headwaters of the Toenepi Stream generally, were predominantly NO_X-N and FRP in shallow groundwater and TP in surface runoff. During high water table conditions in winter stream recharge is mostly from shallow (<3 m), nitrate-bearing groundwater. Mean (±SD) annual groundwater NO_X-N concentrations in the catchment were 0.53±1.40 g m^{-3} (N=33) during 2002-2004 and 3.38±1.17 g m^{-3} (N=18) in 2007 (Stenger et al. 2008), compared with the wetland mean surface water NO_X-N concentration of 0.30±0.65 g m^{-3}. Groundwater N input loads estimated from the product of the average inflow (1.7 L s^{-1}) and the average shallow groundwater NO_X-N concentration were 30-200 kg N yr^{-1} and were comparable with input and output loads for the wetland (Table 2). Ammonium N concentrations in groundwater immediately adjacent to the wetland were generally less than 0.01 g m^{-3} (Stenger et al. 2008), whereas median concentrations in the wetland were 0.02-0.05 g m^{-3} and peak values 0.8-1.4 g m^{-3} (Table 1). Ammonium is a favoured form of N for uptake by plants and microbes (Silver et al 2001) so that inputs of NH_4-N would be transformed to DON and PN with the latter able to be trapped within the wetland. Notably, the overall retention (removal or uptake) of NH_4-N was not enough to account for the production (negative retention) of NO_X-N (Table 3), but the results do indicate nonetheless that nitrification occurred but without significant denitrification within the wetland.

Dissolved organic N, a major component of TN at all sites, passed through the wetland with little reduction in concentration and was a major cause of the poor N removal. Much of the DON in forest streams is considered to originate from soil as refractory humic and fulvic acids and may behave conservatively (Brookshire et al. 2005). We propose that a similar situation occurs in pasture wetlands, whereby terrestrial N deposited from livestock excreta enters as DIN and is transformed initially to plant biomass, a portion of which enters the wetland soil-detritus matrix. In-stream leaching of this soil-detrital matrix then provides the DON that moves unattenuated through the wetland.

Shallow (1.5 m) groundwater FRP concentrations in the wetland catchment were relatively constant year-round (0.02±0.01 g m^{-3}), and stormflow accounted for 63 and 48% of the FRP and TP losses, respectively (Müller et al. 2010). Mean FRP concentration at all sites in the wetland was 0.006±0.0.006 g m^{-3} (Table 1), indicating that much of the input FRP was transformed to other P forms, such as organic P (e.g., nucleic acids, polyphosphates, phosphorus esters and phosphonates) (Ahlgren et al. 2005) or immobilised by being bound to soil cations (Reddy et al. 1999). An analysis of organic P in leachate from a grassland soil using ^{31}P nuclear magnetic resonance analysis found that 55-76% occurred in unreactive particulate monoester and diester forms (Toor et al. 2003). Under favourable conditions the esters were hydrolysable with different phosphatises to inorganic P forms (Toor et al. 2003). On that basis it would seem that P other than FRP in our wetland was similarly in unreactive particulate forms that were subsequently trapped or otherwise retained within the wetland. N and P entering the wetland predominantly in DIN and FRP underwent transformation to particulate forms, firstly through assimilation in plant and microbial biomass and then via conversion to detrital organic matter, either in suspension or associated with wetland sediment (Wilcock and Croker 2004; Silver et al. 2010).

The wetland's riparian protection was also important in managing pollutant exports to Toenepi Stream. Permanent fencing excluded stock and thereby minimised direct inputs of faecal matter as well as particulate N and P associated with excreta (deKlein and Ledgard 2005; Collins et al 2007). Vegetated riparian zones provide filtration and settling of particulates in surface runoff (viz. faecal bacteria and particulate P) as well as presenting a barrier and possible denitrification zone for shallow groundwater inputs of nitrate (McKergow et al. 2008). The complete wetland, including the unfenced upstream section, occupied about 1% of its catchment area. Much of the upper Toenepi Stream catchment is rolling land with wetland swales in valley floors (Wilcock et al. 1999; Müller et al. 2010) so that if all headwater wetlands were given the same degree of riparian protection as in this study, loadings from the headwater catchment might be collectively reduced by about 5% for N, and 90% for P and *E. coli*. Specific yields for the entire wetland were 4-9 kg N ha^{-1} yr^{-1}, 0.1-0.2 kg P ha^{-1} yr^{-1} and $\approx 10^9$ *E. coli* ha^{-1} yr^{-1} compared with typical dairy catchments yields, viz. 10-30 kg N ha^{-1} yr^{-1}, 0.5-1.0 kg P ha^{-1} yr^{-1} and 10^{14} *E. coli* ha^{-1} yr^{-1} (Wilcock et al. 2007). Further reductions in N might be achieved with additional wetlands located on flatter land downstream of the upland swales, where residence times and anoxic conditions may promote denitrification or other processes promoting nitrate removal (Tanner et al. 2005; Zaman et al. 2008).

Conclusions

Runoff inputs of N, P and faecal bacteria from pasture entering surface waters are a mixture of particulates transported in overland flow events, and solutes (viz. FRP, NO_X-N and NH_4-N) in shallow groundwater. Concentrations of FRP in the wetland were very much lower than in the nearby groundwater, indicating that a large fraction of input P was transformed to particulate forms that was subsequently trapped and retained within the wetland. Long narrow wetlands with sufficiently long hydraulic residence times, such as the one studied here, seem well suited to trapping and retaining P and faecal bacteria with overall retentions of >90%.

Nitrogen retentions were small (c. 5%) because the wetland transformed DIN to DON; the largest component of TN (60%) and this was not as readily retained as PN. The predominantly aerobic conditions prevented denitrification being a significant mechanism for N removal and may well have facilitated nitrification with the result that nitrate loads increased somewhat with passage down the wetland. It is highly likely however, that denitrification was a major mechanism for removing NO_X-N from groundwater as it entered the wetland in subsurface flow. Addition of an anoxic wetland downstream may effectively remove nitrate (and hence up to 44% of TN during winter) via denitrification and plant uptake. The risks of wetlands contributing to greenhouse gas emissions due to incomplete denitrification processes have to be evaluated.

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