SURFACE FLOW CONSTRUCTED WETLANDS AS A DRAINAGE MANAGEMENT TOOL – LONG TERM PERFORMANCE

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Abstract

Subsurface tile drains are used extensively throughout many agricultural areas of New Zealand to reduce excessive wetness from grazed pastures. Along with transporting water, subsurface drains also transport nutrients which have leached from the soil, potentially bypassing natural attenuation processes in shallow groundwaters and riparian zones. These nutrients have the capacity to cause excessive "weedy-ness", algal growth, oxygen depletion and alterations to natural ecological processes in streams, lake and estuaries. Surface flow constructed wetlands intercepting and treating these drainage waters before discharge into surface waters are one of the tools employed available to farmers to manage nutrient losses. This paper quantifies long-term drain yields and nutrient removals associated with 2 constructed wetlands (situated in Waikato and Southland) planted with native raupo (*Typha orientalis*) and occupying 0.66–1.6% of their respective catchments. Previous results from these sites (plus an additional irrigated site), quantifying flows, and nitrogen and phosphorus yields from tile drains over continuous 3–5 year periods, and evaluating the capacity of the wetlands to reduce nutrient loads, have been published elsewhere. This paper presents results from an additional year of sampling undertaken 3 years after the earlier periods of study, and compares them to the earlier data. Overall annual drainage water yields ranged from 193–564 mm (16–51% of rainfall), with annual exports from the tile drains of 14–66 kg ha^{-1} of Total Nitrogen (TN) of which 58–97% was nitrate-N. The two constructed wetlands intercepting these flows removed 30–369 g TN m^{-2} (7–63%) of influent loadings annually. Annually, the tile drains exported 0.12–1.79 kg ha^{-1} of Total Phosphorus (TP) of which 33–93% was Dissolved Reactive Phosphorus (DRP). In contrast to the TN results, the phosphorus removal was generally poor in both wetlands, with twice as much TP exported overall than received in some years, although modest removals of up to 32% occurred in other years. These results demonstrate the need for research into reducing nutrient leaching from soils and improved nutrient attenuation within wetlands, particularly for phosphorus.

Nutrient removal performance from these and other wetlands, both nationally and internationally, have been used to develop guidelines for constructed wetlands treating agricultural drainage water.

Introduction

As one of the world's leading agricultural exporters, New Zealand works hard to maintain its "Clean & Green" image. Over the last 40 years there has been a major reduction in point-source agricultural pollution from farm dairies. Dairy farmers have either used waste stabilisation ponds to treat effluents before discharge (Sukias & Tanner 2005) or re-applied them to land to reduce direct pollution of surface waters, and utilise the fertilizer value of these wastes. With the overall reduction in direct, or point-sources, of pollution, there is a greater ability to measure the influence of diffuse nutrient losses, and grazing intensification in recent years has tended to exacerbate the problem. In New Zealand, as elsewhere in the rest of the world there are reports of ecosystem degradation associated with runoff from agricultural land (Carpenter et al. 1998, Novotny 2007, Rabalais et al. 2002). Drainage systems in agricultural landscapes can be a “mixed blessing”, with both positive and negative pollution outcomes. By providing routes to remove excess water from land, the water-table is lowered and infiltration rates increase, thus lowering the potential for surface run-off including sediments and associated contaminant losses. However the extensive drainage system in many parts of New Zealand have converted large areas of natural wetlands, with some shallow lakes having completely disappeared as water tables were lowered to develop agricultural land. Even localised drainage systems allow water to by-pass attenuation processes in riparian zones and natural wetlands (Armstrong & Garwood 1991, Ritter & Shirmohammadi 2001, Skaggs et al. 1994), thus off-setting some potential benefits drainage systems have in reducing diffuse pollution (Gilliam et al. 1999, Monaghan et al. 2005).

Typical nutrient leaching losses from intensive dairy pastures now commonly range from 20–60 kg N ha^{-1} and 0.1–8.5 kg P ha^{-1} (Ledgard 1999, McDowell, Monaghan & Wheeler 2005, Monaghan et al. 2008). Such elevated nutrient losses have resulted in degradation of water quality in many intensively-farmed catchments (MfE 2007, 2009).

Constructed and restored wetlands can however complement improved source control measures to reduce nutrient losses from agricultural landscapes and buffer impacts on receiving waters (Crumpton et al. 2006, Kovacic et al. 2000, Mitsch et al. 2001). Although the ability of wetlands to remove nutrients is well established, diffuse runoff flows and associated contaminant loads are inherently variable and so it is difficult to predict their performance under different climatic and landscape conditions. Quantitative performance information is required so that farmers, agricultural advisors, industry and regulatory agencies can assess the applicability and cost-effectiveness of wetlands in comparison to other mitigation options available to help achieve water quality targets.

We have undertaken research on the use of constructed wetlands to remediate subsurface drainage waters from agricultural land at three locations, two rain-fed (Toenepi in Waikato and Bog Burn in Southland) and one receiving irrigation of water and dairy shed effluent (Titoki in Northland) (Tanner et al. 2003, 2005a), with data from 2001 to the 2005/6 drainage season most recently summarised in Tanner and Sukias (2011). These results indicate consistent (but variable) TN attenuation performance, but negligible TP removal with the wetlands frequently contributing low levels of phosphorus. However phosphorus is a conservative element, without a sustainable gaseous removal mechanism. Thus the ability of wetlands to continuously remove phosphorus are limited. In contrast, nitrogen removal, via denitrification and other microbial pathways can theoretically operate sustainably and indefinitely within wetlands. This paper combines previous nitrogen and phosphorus removal results with those from a further annual period of wetland sampling (in the 2009/10, and 2010/11 drainage seasons respectively) from each of the two rain-fed sites. The objectives of this study were to quantify the annual nitrogen and phosphorus removal achievable using constructed wetlands and compare performance with that previously published, associated with the greater longevity of the wetland.

Methods

Study sites and constructed wetlands

Constructed wetlands treating subsurface drainage from intensively grazed (~2.8–3 cows ha^{-1}), ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*) dairy pastures were established in two prime dairy farming areas of New Zealand, namely Toenepi in the

Waikato, and Bog Burn site in the south of South Island (Figure 1). The treatment potentials of these wetlands were studied continuously over 4–5 annual periods (1 April–31 March), with an additional drainage season sampled after a 4 year interval (2003–2006/7, and 2010/11 at Bog Burn; 2001–2005/6, and 2009/10 at Toenepi). Both sites are rain-fed farms situated in Best Practice Dairying Study Catchments established by the New Zealand dairy industry to monitor long-term stream water quality associated with existing farm practices, and to promote adoption of more sustainable farming practices (Wilcock et al. 2007).

Figure 1. New Zealand map showing location of the two constructed wetland sites.

The basic specifications of the wetland treatment systems, percentage of the catchment areas they occupy, and climate characteristics have been published previously (Tanner & Sukias 2011), but are summarised in Table 1. Background information on the surrounding catchments and farm management practices are summarised in Wilcock et al (2006, 1999) and Monaghan et al. (2008, 2007). Both wetlands were planted with a local species of cattail (*Typha orientalis* C.Presl.), which came to dominate the vegetative cover (>90% of the wetland area).

Sampling and analysis

Wetland inflows and outflows were measured automatically at 15 minute intervals weirs linked to Unidata/NIWA model 2001 stage recording Hydrologgers (1 mm depth resolution; Unidata, Willetton, WA, Australia). Full drain-flows entered the Toenepi wetlands at a single point at the upstream end. Where drain flows at Bog Burn were less than 6 L s^{-1}, the full flow entered the wetland. Volumes higher than 6 L s^{-1} are diverted directly to an adjacent surface-flow drain, thus bypassing the wetland. The difference between the drain-flow and the wetland inflow at Bog Burn was used to estimate the proportion of drain-flow bypassed.

Samples of 220 ml were taken every 10–40 m^3 of cumulative flow (depending on site flow characteristics) and deposited (4 samples per bottle) in 1 L acid-washed bottles with mercuric chloride pre-added as a preservative. Approximately monthly during the main flow periods or when the telemetered data indicated that the samplers were approaching 96 samples (24 bottles) the samples were collected and returned to the laboratory where they were stored at 4 ºC until ready for processing.

Samples were analysed for total P (TP) and total N (TN), combined nitrate and nitrite N (henceforth denoted as nitrate-N), ammoniacal N (ammonium-N) and dissolved reactive P (DRP) using automated flow injection analysis (QuikChem 8000 FIA+, Lachat Instruments, Loveland, CO, USA). Organic N (Org-N) was calculated as TN minus dissolved inorganic forms of N (nitrate-N and ammonium-N). Areal mass loadings and removals of N and P for the wetlands were calculated by multiplying the cumulative flow for each measurement period by the flow-proportionally sampled concentrations of each nutrient species during that period. Specific details on analytical techniques are presented in Tanner and Sukias (2011)

Results and Discussion

Flows

Catchment drainage flow yields of 0.19–0.38 m y^{-1} at Toenepi from 2001-2006 comprised 16–33% of annual rainfall (Table 2). The 2009/10 drainage season gave a yield of 0.34 m y^{-1} or 36%, a slightly higher degree of drainage than previously recorded.

The Bog Burn site showed drainage flow yields of 0.18–0.56 m y^{-1} from 2003-2007 comprising 21–51% of annual rainfall (Table 2). The 2010/11 drainage season yielded 0.397 m y^{-1} (43% of annual rainfall). Hydraulic loadings to this wetland, which only occupied 0.7% of its drainage catchment, were moderated to ~400 mm d^{-1} by the high-flow diversion structure fitted at this site. This resulted in 16–37% of the annual drainage flow being diverted from the wetland, comprising 19–49% of the annual TN load and 15–46% of the annual TP load from the drainage system (see later sections). Detailed analyses of earlier rainfall and hydrological data can be found in Tanner and Sukias (2011).

Nitrate-N

As is common in tile drainage from fertile agricultural fields, nitrate was the predominant form of N at all sites, comprising >90% of the annual TN load at Toenepi (except 1st year of monitoring, 58%), and 70–78% of the annual TN load at Bog Burn (Fig. 3 and Table 3). The annual catchment nitrate-N yields (Table 2) at Toenepi ranged from 19–43 kg ha^{-1} and at Bog Burn from 10–27 kg ha^{-1} (the 2010/11 proving to be slightly higher than the previous highest season loading of 26 kg ha^{-1}). Mean annual flow-proportional nitrate-N concentrations in the tile drainage ranged from 10–12 and 3–6 g m^{-3} for Toenepi and Bog Burn respectively (Table 3). As noted in previous publications, this covers the range typically found for tile drainage from moderate to high-intensity dairy pastures in New Zealand (Houlbrooke et al. 2003, Monaghan et al. 2005), but is low-moderate compared to concentrations that can occur in drainage waters from intensive cropping systems subject to cultivation, higher fertiliser applications, and varying vegetative cover (Randall et al. 2003).

After passage through the wetlands annual nitrate-N loads were reduced by 11–52% at Toenepi, and 24–59% at Bog Burn (Table 3), reducing flow-proportional outflow concentrations (Table 3) to 5.9–10 g m^{-3} and 1.2–3.7 g m^{-3} respectively. The data for the additional years monitoring were within the ranges measured in the earlier periods.

Organic-N and Ammonium-N

With nitrate forming the majority of N exiting the drainage areas, other N forms were much lower, however in the first drainage season at Toenepi organic-N comprised 41% of the inflow load (27 kg N ha^{-1} y^{-1}), probably associated with soil disturbance while creating the drainage system (Tanner et al. 2005b). In subsequent years it has been negligible as has ammonium-N.

At Bog Burn organic-N comprised 18–27% of TN exiting the drainage system (3.5–6.7 kg N ha^{-1} y^{-1}). Annual loads of ammonium and organic-N often increased during passage though the wetlands (Fig. 3). Similar small increases in ammonium and organic-N have also been reported for other wetlands treating nitrate-rich flows, and are likely due to net release from sediments and decomposing vegetation (Davis & van der Valk 1978).

Total Nitrogen

Annual drainage TN yields of 20–66 kg ha^{-1} at Toenepi, and 14–34 kg ha^{-1} at Bog Burn fit within the normal range recorded for New Zealand dairy pastures in these regions under similar fertiliser and grazing regimes (Ledgard 1999, Monaghan et al. 2008, Monaghan et al. 2005).

The Toenepi and Bog Burn wetlands removed 7–63% and 26–42%, respectively, of the TN load that entered them over the annual periods sampled (Fig. 3).

Transient high loadings and retention of organic N in the Toenepi wetland during the first year, was followed by a year with low overall nitrate and TN removal.

Phosphorus

Mean annual flow-proportional TP concentrations in the tile drainage ranged from ~0.03–0.12 g m^{-3} at Toenepi, and 0.22–0.46 g m^{-3} at Bog Burn (Table 4). As shown in Figure 4, DRP comprised the major proportion of drainage mean annual flow-proportional TP loads at Toenepi (annual means 61–93%; 0.02–0.10 g DRP m^{-3}), compared to about half the loads at Bog Burn (annual means 36–62%; 0.09–0.28 g DRP m^{-3}).

Annual yields of TP in subsurface drainage ranging from 0.12–0.26 kg ha^{-1} at Toenepi, and 0.54–1.72 kg ha^{-1} at Bog Burn (Table 2), were low–moderate compared to typical losses of 0.1–8.5 kg P ha^{-1} recorded for New Zealand dairy farms (McDowell, Monaghan & Wheeler 2005, Monaghan et al. 2008). The annual yields of TP and proportion of DRP recorded in tile drainage flows over four years at Bog Burn were considerably higher than those reported (0.07–0.69 kg P ha^{-1}, and 20–35%) at a similar pastoral site ~40km to the southeast (McDowell, Monaghan, Smith, et al. 2005).

In most years monitored, mean annual flow-proportional TP concentrations (Table 4) and loads (Fig. 4&5) increased after passage through the wetlands. The Toenepi wetland showed greater than 70% net P generation in 3 out of the 6 years monitored, with the remaining 3 years showing no change or only very minor load reductions (<10%). The Bog Burn wetland reduced TP levels during the first year of monitoring then became net sources in subsequent years, however in the final year of monitoring (8^{th} drainage season), a TP load reduction of 29% was recorded (Fig. 5).

All the wetlands had the original topsoil from the site returned into the base of the excavated wetland as a plant growth media and source of organic carbon to support denitrification. Although these soils had been subject to annual fertilisation with superphosphate over many years, measured P retention capacities measured at the start of the trials using standard soil testing procedures on air-dried soil samples were in excess of 50% for the Toenepi soil, and 20-30% at Bog Burn (Tanner et al. 2005b, and unpublished data, M. L. Nguyen). Despite this, both soils proved to be net P sources rather than sinks under saturated wetland conditions in the present study.

There are a range of options available to reduce the potential for P generation during flooding of constructed wetland soils, including addition of redox-stable P-sorbing materials such as alum or limestone (Ann et al. 2000a, 2000b, Ballantine & Tanner 2010), and the use of subsoils with lower P status as growth media e.g. (Liikanen et al. 2004). Because of the lower organic matter content of subsoils, this latter option would likely result in reduced soil denitrification rates until sufficient organic matter could be built-up by plants growing in the wetland.

Wetland Maturation

Wetlands are complex biological systems which incorporate plant uptake and release of nutrients combined with microbial processing and physico-chemical processes to attenuate nutrient flows entering them. Maturation factors are likely to have influenced treatment responses during the course of the monitoring period, and were most noticeable in the first year of operation. Potentially important factors include: rapid initial nutrient uptake into establishing plant biomass (Tanner 2001), gradual accumulation of plant detritus as a carbon source and habitat for denitrifiers (Hume et al. 2002), and changes in soil P form and sorption/desorption potential related to flooding of former terrestrial soils (as discussed previously) and gradual saturation of available sorption sites (Richardson & Craft 1993). Differences in climate regime at each site may also have affected wetland treatment performance, via temperature or litter organic carbon quality effects on microbial decomposition and denitrification rates, or by influencing plant growth and associated nutrient uptake rates.

Nitrogen removal is theoretically more sustainable than with phosphate, with nitrate being converted into di-nitrogen gas via denitrification. This process requires an energy source such as that provided by decaying plant leaf litter. Thus as the organic mulch at the base of a wetland increases over years or decades, the available energy pool for denitrification is expected to increase. Natural wetlands however may continue to develop over many centuries, with some in the Waikato region approximately 15,000 years old (Clarkson et al. 2004). Thus the relatively short period of establishment and monitoring over which this study has been undertaken may leave further room for wetland development beyond the normal time-frame that funding agencies are prepared to fund scientific investigations. Nevertheless, we have been able to show nitrogen removal at both wetlands in each year of this study.

There were however no discernable patterns of increasing TN removal over the 8 and 9 years since wetland establishment.

Neither of the wetlands studied proved particularly effective at P removal, with overall generation during 3 of 5 years at Bog Burn, and 4 of 6 years at Toenepi. This differs from the positive P removal results reported for a range of other wetlands receiving agricultural drainage, run-off and river flows (Kadlec 2010, Kovacic et al. 2000, Mitsch et al. 2000), but is consistent with results from a number of other wetland studies (Coveney et al. 2002, Novak et al. 2004, Spangler et al. 1977). Many other studies reporting positive P retention involve treatment of run-off from cropped agricultural fields in which a larger proportion of the incoming P load is associated with suspended particulates, which are comparatively easily removed through settling and deposition. The systems we studied generally had low suspended solids loads, with P predominantly in dissolved forms that require adsorption and/or assimilation into microbial or plant biomass to be removed from the water column. Additional measures are clearly required to reduce P losses from these wetlands, or provide supplementary P removal. As an interim measure we advise mixing equal quantities of subsoil (with higher P retention capacity) with the local topsoil to provide the growth media for plants, however we are also investigating the practical efficacy of a wider range of P retentive soil amendments and granular filter materials (Ballantine & Tanner 2010).

Data from these and associated studies, both in New Zealand and internationally, have been used to develop guidelines for constructed wetland treatment of subsurface drainage from agricultural catchments (Tanner et al. 2010).

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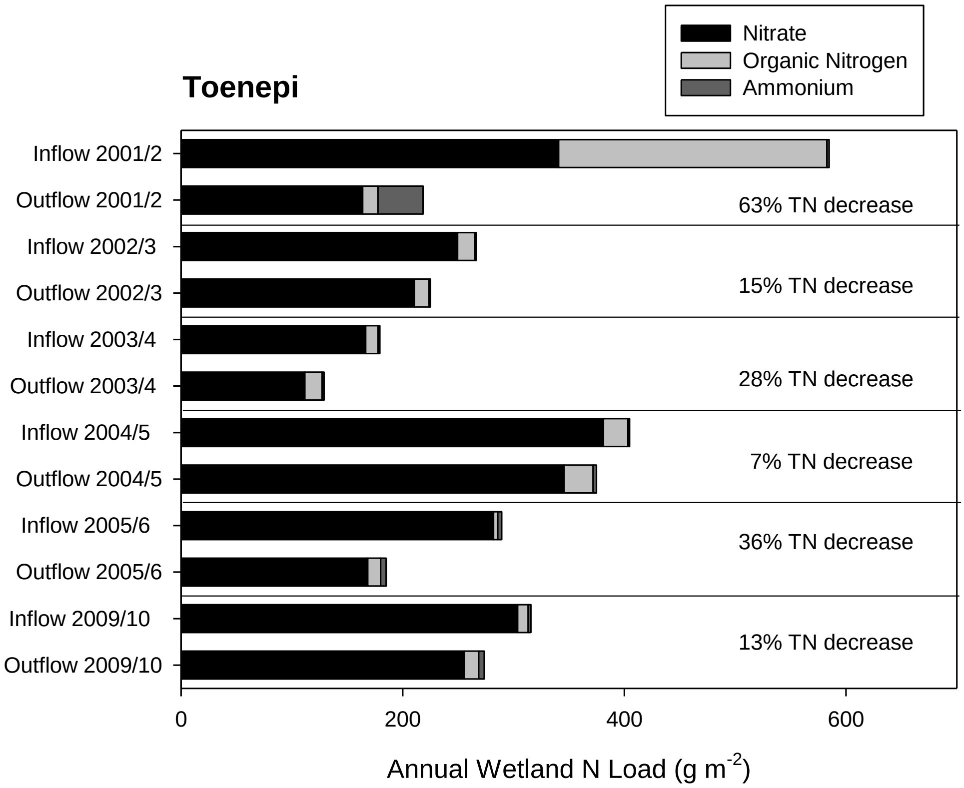

Figure 2. Annual aerial mass loading and outflow of nitrogen species at the Toenepi constructed wetland.

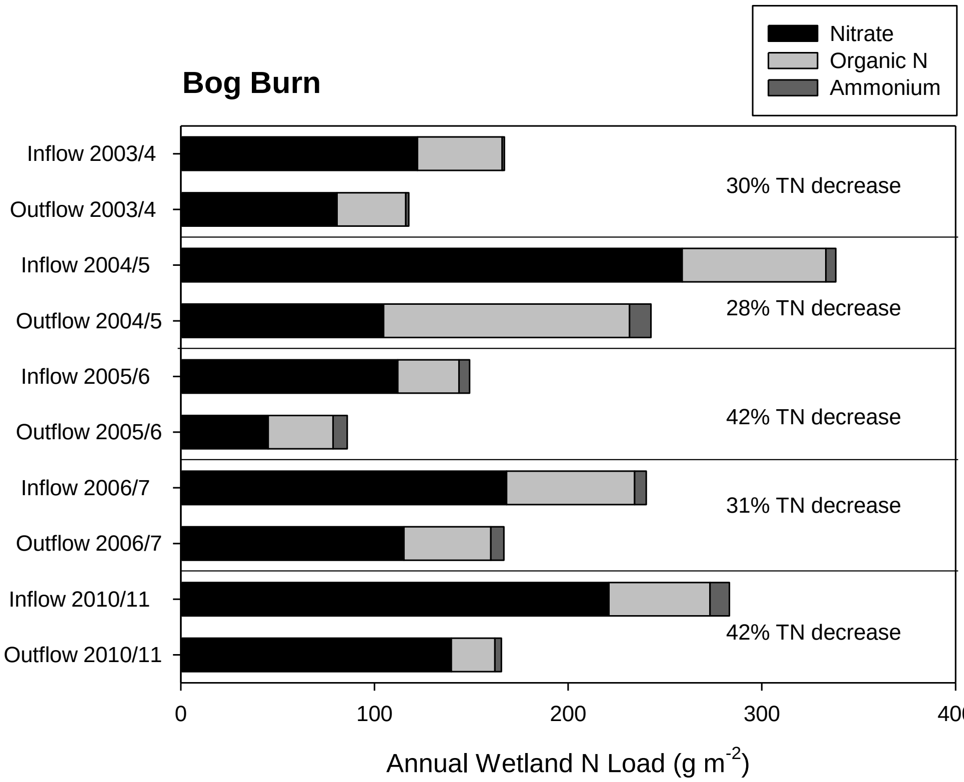

Figure 3. Annual aerial mass loading and outflow of nitrogen species at the Bog Burn constructed wetland.

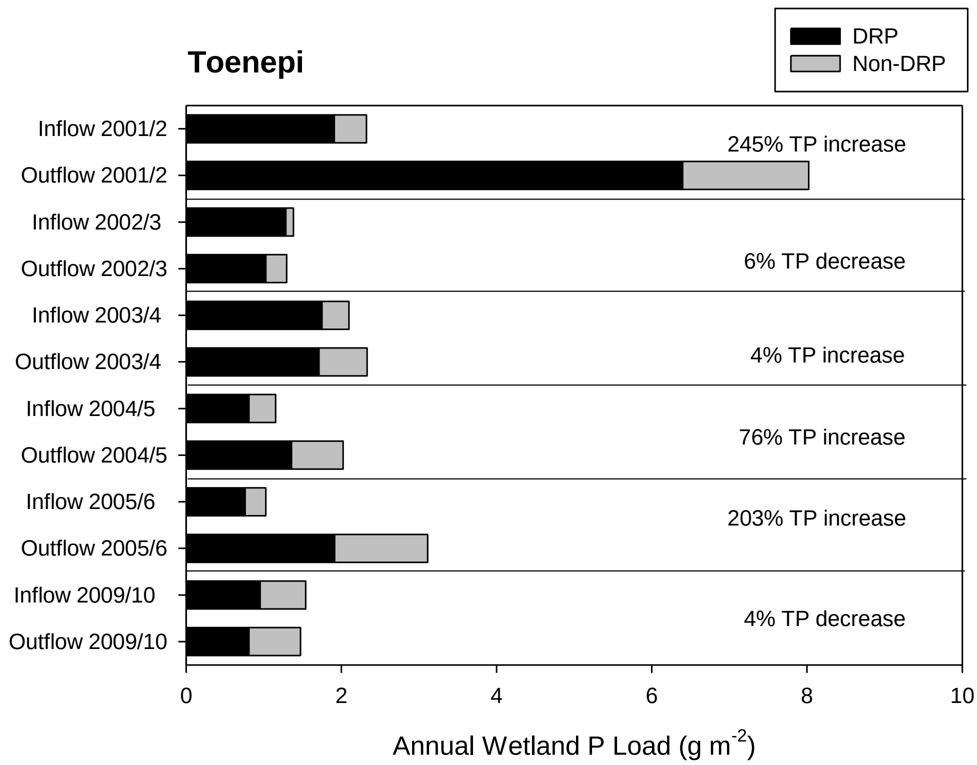

Figure 4. Annual aerial mass loading and outflow of phosphorus species at the Toenepi constructed wetland.

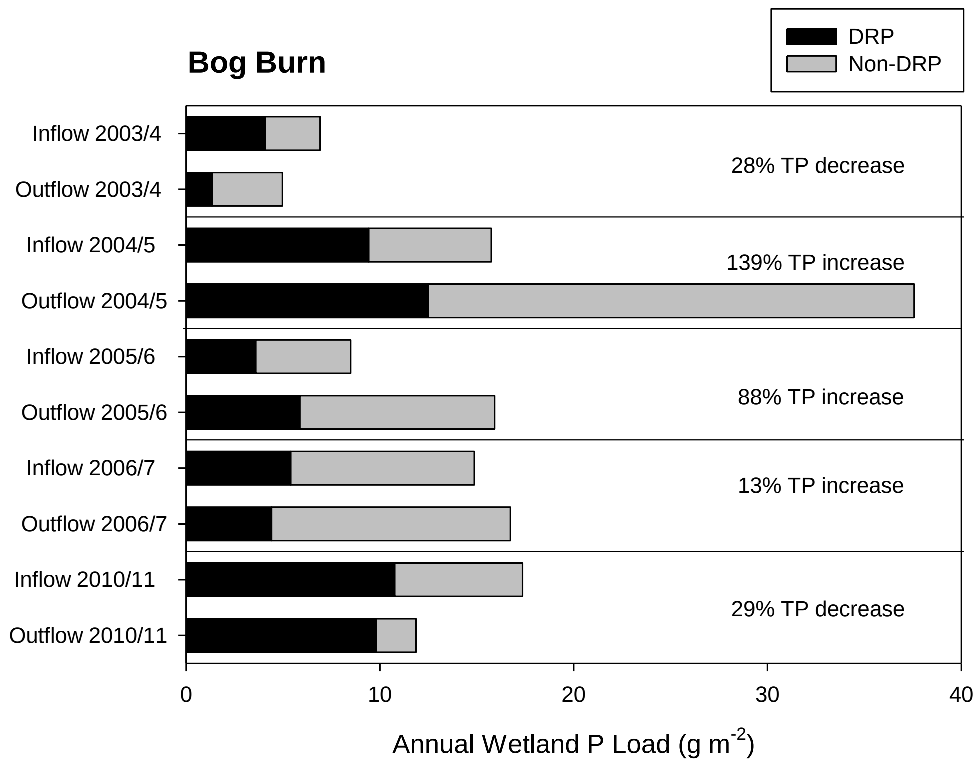

Figure 5. Annual aerial mass loading and outflow of phosphorus species at the Bog Burn constructed wetland.

Table 1: Summary of treatment wetlands characteristics at the two test sites.

Wetland site	Location coordinates	Annual range mean monthly air temperatures (°C)	Typical annual rainfall (m)	Monitoring period	Wetland basal area (m^2)	System length: width ratio	Contributing catchment (ha)	Percentage of catchment area	Study site features
Toenepi, Waikato	37° 44’ S 175° 35’ E	9.2–18.6	1.1-1.2	2001-2006	293 (143, 150)*	8.5 (4.3, 4.2)*	2.6	1.1 %	Catchment soil type: Rotokauri silt loam, Acidic Orthic Gley (NZ), Aeric Haplaquent (USDA). Rainfed pasture; toeslope interceptor drain; mild central North Island inland climate.
Bog Burn, Southland	46° 02’ S 168° 12’ E	5.3–14.5	0.8-1.1	2003-2007	112.5	4.5	1.7	0.66 %	Catcment soil type: Pukemutu silt loam, Argillic-mottled Fragic Pallic (NZ), Typic Fragiochrept (USDA). Rainfed pasture; grid of mole and tile drains, diversion of extreme high flows; cool southern South Island climate with low-intensity rainfall.

* Wetland comprised of two cells in series

Table 2: Summary of annual rainfalls, catchment drainage and nutrient yields, and measured wetland influent hydraulic loading during the monitoring periods.

Site/	Hydrology ($m\ y^{-1}$)			Nutrients ($kg\ ha^{-1} y^{-1}$)		
Annual period	Rainfall	Catchment drainage yield[1]	Wetland hydraulic loading	Catchment nitrate-N yield	Catchment TN yield	Catchment TP yield
Toenepi						
2001/2	1.212	0.317 (26%)	28.09	38.4	65.9	0.26
2002/3	1.145	0.234 (20%)	20.78	28.1	30.0	0.16
2003/4	1.174	0.193 (16%)	17.12	18.8	20.2	0.24
2004/5	1.147	0.380 (33%)	33.74	43.0	45.6	0.13
2005/6	1.165	0.301 (26%)	26.75	31.8	32.6	0.12
2009/10	0.950	0.343 (36%)	30.40	34.2	35.6	0.17
Bog Burn						
2003/4	0.874	0.183 (21%)	23.19 (84%)[2]	10.1 (80%)[2]	13.7 (81%)[2]	0.54 (85%)[2]
2004/5	1.101	0.564 (51%)	66.93 (79%)[2]	26.3 (65%)[2]	33.4 (67%)[2]	1.38 (76%)[2]
2005/6	1.036	0.380 (37%)	39.27 (68%)[2]	12.9 (57%)[2]	17.3 (51%)[2]	1.04 (54%)[2]
2006/7	0.931	0.350 (38%)	34.36 (64%)[2]	19.6[3] (57%)[2]	27.6[3] (58%)[2]	1.69[3] (58%)[2]
2010/11	0.922	0.397(43%)	38.17 (63%)[2]	27.6 (53%)[2]	34.2 (55%)[2]	1.72 (64%)[2]

[1] Apparent water yield from contributing catchment; values in parenthesis are drainage yield as a percentage of rainfall and irrigation, if any.
[2] High flow diversion operating at wetland inlet. Values in parenthesis are percentage that was not diverted and entered the wetland.
[3] Value corrected from Tanner and Sukias (2011).

Table 3: Summary of mean flow-proportional concentrations (g m^{-3}) and percentage change for total nitrogen (TN) and constituent fractions in the wetland inflows and outflows.

	Nitrate-N			Organic-N			Ammonium-N			TN		
Annual period	inflow	outflow	change	inflow	outflow	change	inflow	outflow	change	inflow	outflow	change
Toenepi												
2001/2	12.13	5.86	-52%	8.62	0.50	-94%	0.07	1.45	+1902%	20.82	7.72	-63%
2002/3	12.01	10	-17%	0.75	0.63	-16%	0.06	0.06	+3%	12.81	10.69	-17%
2003/4	9.73	6.32	-35%	0.66	0.84	+27%	0.09	0.09	+1%	10.48	7.25	-31%
2004/5	11.3	10.02	-11%	0.66	0.77	+16%	0.04	0.09	+110%	12.00	10.87	-9%
2005/6	10.54	6.11	-42%	0.14	0.42	+197%	0.13	0.18	+37%	10.82	6.71	-38%
2009/10	9.97	8.51	-15%	0.31	0.43	+38%	0.08	0.17	+110%	10.37	9.10	-12%
Bog Burn												
2003/4	5.27	3.48	-34%	1.89	1.53	-19%	0.05	0.07	41%	7.21	5.08	-29%
2004/5	3.87	1.62	-58%	1.11	1.97	+77%	0.08	0.17	127%	5.05	3.76	-26%
2005/6	2.85	1.17	-59%	0.81	0.86	+7%	0.14	0.19	35%	3.80	2.22	-42%
2006/7	4.90	3.70	-24%	1.92	1.44	-25%	0.17	0.22	25%	6.99	5.36	-23%
2010/11	5.79	3.66	-37%	1.37	0.59	-57%	0.26	0.09	-67%	7.42	4.34	-42%

Table 4: Summary of mean flow-proportional concentrations (g m^{-3}) and percentage change for dissolved reactive and total phosphorus in the wetland inflows and outflows.

Annual period	DRP inflow	DRP outflow	DRP change	TP inflow	TP outflow	TP change
Toenepi						
2001/2	0.068	0.229	+236%	0.083	0.287	+247%
2002/3	0.062	0.049	-21%	0.067	0.062	-8%
2003/4	0.102	0.092	-10%	0.123	0.124	+1%
2004/5	0.024	0.039	+64%	0.034	0.059	+72%
2005/6	0.028	0.069	+144%	0.038	0.113	+194%
2009/10	0.031	0.027	-14%	0.051	0.049	-3%
Bog Burn						
2003/4	0.176	0.058	-67%	0.298	0.214	-28%
2004/5	0.141	0.193	+37%	0.235	0.582	+147%
2005/6	0.092	0.152	+66%	0.216	0.411	+90%
2006/7	0.157	0.142	-10%	0.433	0.538	+24%
2010/11	0.282	0.257	-9%	0.455	0.311	-32%

References

Ann, Y.; Reddy, K.R.; Delfino, J.J. (2000a). Influence of chemical amendments on phosphorus immobilization in soils from a constructed wetland. *Ecological Engineering* 14: 157–167.

Ann, Y.; Reddy, K.R.; Delfino, J.J. (2000b). Influence of redox potential on phosphorus solubility in chemically amended wetland organic soils. *Ecological Engineering* 14: 169–180.

Armstrong, A.C.; Garwood, E.A. (1991). Hydrological consequences of artificial drainage of grassland. *Hydrological processes* 5: 157–174.

Ballantine, D.J.; Tanner, C.C. (2010). Substrate and filter materials to enhance phosphorus removal in constructed wetlands treating diffuse farm runoff: A review. *New Zealand Journal of Agricultural Research* 53(1): 71–95. <http://dx.doi.org/10.1080/00288231003685843>

Carpenter, S.R.; Caraco, N.F.; Correll, D.L.; Howarth, R.W.; Sharpley, A.N.; Smith, V.H. (1998). Nonpoint pollution of surface waters with phosphorus and nitrogen. *Ecological Applications* 8: 559–568.

Clarkson, B.; Schipper, L.A.; Lehmann, A. (2004). Vegetation and peat characteristics in the development of lowland restiad peat bogs, North Island, New Zealand. *Wetlands* 24(1): 133–151.

Coveney, M.F.; Stites, D.L.; Lowe, E.F.; Battoe, L.E.; Conrow, R. (2002). Nutrient removal from eutrophic lake water by wetland filtration. *Ecological Engineering* 19(2): 141–159.

Crumpton, W.G.; Stenback, G.A.; Miller, B.A.; Helmers, M.J. (2006). Potential benefits of wetland filters for tile drainage systems: Impact on nitrate loads to Mississippi River Subbasins. *Department of Ecology, Evolution and Organismal Biology, Iowa State University IOW06682* Ames, IA.

Davis, C.B.; van der Valk, A.G. (1978). The decomposition of standing and fallen litter of *Typha glauca* and *Scirpus fluviatilis*. *Canadian Journal of Botany* 56: 662–675.

Gilliam, J.W.; Baker, J.L.; Reddy, K.R. (1999). Water quality effects of drainage in humid regions. *In*: Skaggs, R.W.; Schilfgaarde, J.v. (eds). Agricultural Drainage. Agronomy Monograph 38, pp. 801-830. American Society of Agronomy, Madison, WI.

Houlbrooke, D.J.; Horne, D.J.; Hedley, M.J.; Hanly, J.A.; Snow, V.O. (2003). "The impact of intensive dairy farming on the leaching losses of nitrogen and phosphorus from a mole and pipe drained soil." Presented at the Proc. N. Z. Grass. Assoc., Ashburton, NZ.

Hume, N.P.; Fleming, M.S.; Horne, A.J. (2002). Denitrification potential and carbon quality of four aquatic plants in wetland microcosms. *Soil Science Society of America Journal* 66: 1706–1712.

Kadlec, R.H. (2010). Dynamics in event-driven wetlands. *Ecological Engineering* 36: 503-516.

Kovacic, D.A.; David, M.B.; Gentry, L.E.; Starks, K.M.; Cooke, R.A. (2000). Effectiveness of constructed wetands in reducing nitrogen and phosphorus export from agricultural tile drainage. *Journal of Environmental Quality* 29: 1262–1274.

Ledgard, S.F. (1999). Nitrogen inputs and losses from clover/grass pastures grazed by dairy cows, as affected by nitrogen fertilizer. *Journal of Agricultural Science* 132:215-225.

Liikanen, A.; Puustinen, M.; Koskiaho, J.; Vaisanen, T.; Martikainen, P.; Hartikainen, H. (2004). Phosphorus removal in a wetland constructed on former arable land. *Journal of Environmental Quality* 33: 1124–1132.

McDowell, R.W.; Monaghan, R.M.; Smith, L.C.; Koopmans, G.F.; Stewart, I. (2005). Enhanced losses of phosphorus in mole-tile drainage water following short-term applications of dairy effluent to pasture. *In*: Burk, A.R. (ed.). Water pollution: new research, pp. 55–76. Nova Science Publishers, New York, USA.

McDowell, R.W.; Monaghan, R.M.; Wheeler, D. (2005). Modelling phosphorus losses from pastoral farming systems in New Zealand. *New Zealand Journal of Agricultural Research* 48: 131-141.

MfE (2007). Environment New Zealand 2007. Ministry for the Environment. Wellington, NZ,

MfE (2009). Water quality in selected dairy farming catchments: a baseline to support future water-quality trends assessments. Environment, M.f.t. Wellington, NZ,

Mitsch, W.J.; Day, J.W.; Gillingham, A.G.; Groffman, P.M.; Hey, D.L.; Randell, G.W.; Wang, N. (2001). Reducing nitrogen loading to the Gulf of Mexico from the Mississippi River Basin: strategies to counter a persistent ecological problem. *Bioscience* 51: 373-388.

Mitsch, W.J.; Horne, A.J.; Nairn, R.W. (2000). Nitrogen and phosphorus retention in wetlands- ecological approaches to solving excess nutrient problems. *Ecological Engineering* 14: 1–7.

Monaghan, R.M.; de Klein, C.A.M.; Muirhead, R.W. (2008). Prioritisation of farm scale remediation efforts for reducing losses of nutrients and faecal indicator organisms to waterways: A case study of New Zealand dairy farming *Journal of Environmental Management* 87(4): 609-622.

Monaghan, R.M.; Paton, R.J.; Smith, L.C.; Drewry, J.J.; Littlejohn, R.P. (2005). The impacts of nitrogen fertilisation and increased stocking rate on pasture yield, soil physical condition and nutrient losses in drainage from a cattle-grazed pasture. *New Zealand Journal of Agricultural Research* 48: 227-240.

Monaghan, R.M.; Wilcock, R.J.; Smith, L.C.; Tikkisetty, B.; Thorrold, B.S.; Costall, D. (2007). Linkages between land management activities and water quality in an intensively farmed catchment in southern New Zealand. *Agriculture, Ecosystems and Environment* 118: 211–222.

Novak, J.M.; Stone, K.C.; Szogi, A.A.; Watts, D.W.; Johnson, M.H. (2004). Dissolved phosphorus retention and release from a coastal plain in-stream wetland. *Journal of Environmental Quality* 33: 394–401.

Novotny, V. (2007). Diffuse pollution from agriculture: ecological sustainability or food production or both. *Water21* (Apr): 52–59.

Rabalais, N.N.; Turner, R.E.; Wiseman, W.J., Jr. (2002). Gulf of Mexico Hypoxia, A.K.A. "The Dead Zone". *Annual review of ecology and systematics* 33: 235–263.

Randall, G.W.; Vetsch, J.A.; Huffman, J.R. (2003). Nitrate Losses in subsurface drainage from a corn-soybean rotation as affected by time of nitrogen application and use of Nitrapyrin. *Journal of Environmental Quality* 32: 1764–1772.

Richardson, C.J.; Craft, C.B. (1993). Effective phosphorus retention in wetlands: fact or fiction? *In*: Moshiri, G.A. (ed.). Constructed wetlands for water quality improvement, pp. 271–282. Lewis Publishers, Boca Raton, FL.

Ritter, W.F.; Shirmohammadi, A. (2001). Agricultural drainage and water quality. *In*: Ritter, W.F.; Shirmohammadi, A. (eds). Agricultural nonpoint source pollution: watershed management and hydrology, pp. 207-231. Lewis Publishers, Boca Raton, FL.

Skaggs, R.W.; Breve, M.A.; Gilliam, J.W. (1994). Hydrologic and water quality impacts of agricultural drainage. *Critical Reviews in Environmental Science and Technology* 24: 1–32.

Spangler, F.I.; Fetter, C.W.; Sloey, W.E. (1977). Phosphorus accumulation-discharge cycles in marshes. *Water Resources Bulletin* 13: 1191–1201.

Sukias, J.P.S.; Tanner, C.C. (2005). Ponds for livestock wastes. *In*: Shilton, A. (ed.). Pond Treatment technology, pp. 408–432. *Integrated Environmental Technology Series*. IWA Publishing, London.

Tanner, C.C. (2001). Growth and nutrient dynamics of soft-stem bulrush is constructed wetlands treating nutrient-rich wastewaters. *Wetlands ecology and management* 9: 49–73.

Tanner, C.C.; Nguyen, M.L.; Sukias, J.P.S. (2003). "Harnessing constructed wetlands to reduce nutrient export from the agricultural landscape." Presented at the NZWWA 45th Annual Conference and Expo, Auckland.

Tanner, C.C.; Nguyen, M.L.; Sukias, J.P.S. (2005a). "Export of nitrogen in subsurface drainage from irrigated and rain-fed dairy pastures and its attenuation in constructed wetlands." Presented at the Fertiliser and Lime Research Centre Workshop: Developments in Fertiliser Application Technologies and Nutrient Management, Palmerston North, February 2005.

Tanner, C.C.; Nguyen, M.L.; Sukias, J.P.S. (2005b). Nutrient removal by a constructed wetland treating subsurface drainage from grazed dairy pasture. *Agriculture, Ecosystems and Environment* 105: 145-162.

Tanner, C.C.; Sukias, J.P.S. (2011). Multi-year nutrient removal performance of three constructed wetlands intercepting tile drain flows from grazed pastures. *Journal of Environmental Quality* in press.

Tanner, C.C.; Sukias, J.P.S.; Yates, C.R. (2010). New Zealand Guidelines for Constructed Wetland Treatment of Tile Drainage. *National Institute of Water and Atmospheric Research NIWA Information Series No. 75* Hamilton, New Zealand. 54 p., http://www.niwa.co.nz/our-science/freshwater/tools/water/pollution-management/tile-drain-wetland-guidelines.

Wilcock, R.J.; Monaghan, R.; Thorrold, B.S.; Meredith, A.S.; Betteridge, K.; Duncan, M.J. (2007). Land-water interactions in five contrasting dairying catchments: issues and solutions. *Land Use and Water Resources Research* (7:2.1-2.10.).

Wilcock, R.J.; Monaghan, R.M.; Quinn, J.M.; Campbell, A.M.; Thorrold, B.S.; Duncan, M.J.; McGowan, A.W.; Betteridge, K. (2006). Land-use impacts and water quality targets in the intensive dairying catchment of the Toenepi Stream, New Zealand. *New Zealand Journal of Marine and Freshwater Research* 40: 123–140.

Wilcock, R.J.; Nagels, J.W.; Rodda, H.J.E.; O'Connor, B.; Thorrold, B.S.; Barnett, J.W. (1999). Water quality of a lowland stream in a New Zealand dairy farming catchment. *New Zealand Journal of Marine and Freshwater Research* 33: 683–696.