Sukias, J.P.S. and Tanner, C.C. 2023. A CONSTRUCTED WETLAND RECEIVNG DAIRY FARM RUN-OFF: EFFECTIVENESS DURING HIGH FLOW EVENTS COMPARED WITH BASEFLOW. In: *Diverse solutions for efficient land, water and nutrient use*. (Eds. C.L. Christiansen, D.J. Horne and R. Singh). http://flrc.massey.ac.nz/publications.html. Occasional Report No. 35. Farmed Landscapes Research Centre, Massey University, Palmerston North, New Zealand. 12 pages.

A CONSTRUCTED WETLAND RECEIVING DAIRY FARM RUN-OFF: EFFECTIVENESS DURING HIGH FLOW EVENTS COMPARED WITH BASEFLOW

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Abstract

The efficacy of constructed wetlands (CWs) intercepting farm run-off during storm events has been questioned due to the reduced residence time available for treatment. While it is recognized that CWs can efficiently remove nitrate via denitrification during baseflow, their performance is less well studied during high flow events when high loads of particulate-associated pollutants and faecal bacteria are typically mobilized from the catchment. A multi-celled CW receiving runoff from farmland and a major laneway on a dairy farm was monitored for performance during two drainage years (2017 and 2019). The 2017 season had the highest precipitation since 2000 (25 events, 989 mm), while the 2019 season had the fourth lowest precipitation (7 events, 467 mm). Wetland outflow was even more extreme, at 96,500 m^3 and 10,500 m^3 respectively. During baseflow, runoff primarily comprised pastoral drainage emerging from upslope natural wetlands and direct groundwater inputs through the base of the wetland. Removal of nitrate appeared low during the 2017, as the upslope wetlands had already reduced nitrate in the surface inflows, while groundwater monitoring wells had not been installed. In 2019, when groundwater inputs were monitored in four piezometers, the reduction in incoming nitrate loads was able to be accurately estimated at ~70%. During rain events in both the years monitored, high loads of particulate associated pollutants washed off the laneway and into the CW, far exceeding inputs during baseflow. The initial cells of the CW acted as sedimentation basins, effectively capturing much of the sediment and associated pollutants. Removal efficacy of pollutants such as TSS for the two years was 80% and 65% respectively, with similar removals of TP and *E. coli*. In most instances, removal of other monitored pollutants exceeded 50%.

The study confirmed:

- The importance of laneways as point sources of particulate-associated pollutants.
- The sediment retention capabilities of CWs receiving surface run-off, and the value of incorporating an initial sediment trap.
- The value of monitoring groundwater seepage to properly characterize nitrate inputs to constructed wetlands.
- The influence of seasonal and inter-annual variability in CW input loads.

Introduction

The wetland complex was constructed in a valley as a series of five unlined cells separated by earthen bunds. The major baseflow into the wetland in terms of volume was the drainage from surrounding pastures, originating in a series of natural seepage wetlands in the headwaters.

The first three CW cells had smaller surface areas and were slightly deeper than the two downstream cells, with less permanent vegetation cover. These deeper cells functioned primarily as settling ponds. Cells 4 and 5 were shallower, had much larger surface areas and were almost completely vegetated. Four groundwater sampling wells were installed around the edge of the constructed wetland prior to the 2019 drainage year.

The primary catchment area for the constructed wetland is 45.9 ha of farmland. During rain events, the lower side laneway and side drain (Figure 1) intercept surface runoff from an additional 6.5 ha of farmland, this runoff and runoff generated on the laneway is directed into the wetland via the laneway input (total of 52.4 ha). Previously we noted that this laneway input was heavily contaminated with faecal material and was a major source of contaminant load to the wetland.

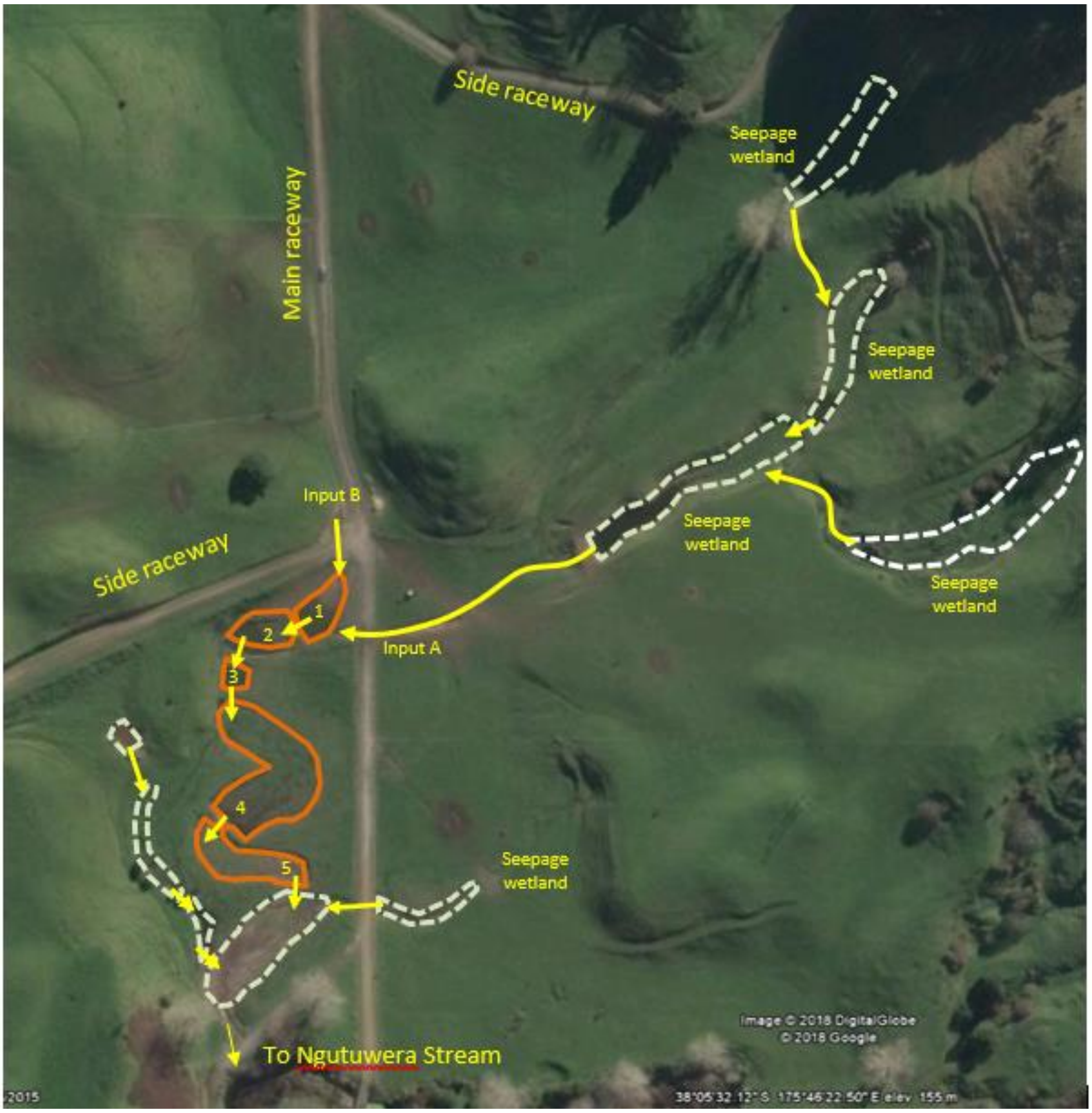

Figure 1. Layout of constructed wetland along with major inlets and outlets. The five monitored constructed wetland cells are defined with orange unbroken lines. Unmonitored wetland cells (constructed and natural seepage) are outlined in broken white lines. Laneways that contribute to the laneway input are identified. W1-W4 are the groundwater sampling wells.

Methods

Wetland hydrology and volume measurement

Continuous flow monitoring stations (float and counter-weight attached to an encoder housed in a stilling well) were established prior to the 2017 drainage year at the outflow from cell 1, and the outflow of cell 5 (sampling locations 1 and 5 respectively in Figure 1). Flow from the cells exited though standard v-notch weirs. Flow data from cell 1 and cell 5 were used to create a water balance for the constructed wetland.

Referring to Figure 1 the hydrological and contaminant pathways were as follows:

▪ The flow volumes measured at cell 1 were a combination of farmland drainage inflow from the seepage wetlands (D) and runoff from the laneways (L). Groundwater seepage presumably also enters cell 1 directly at times, but this could not be separated from the dominant surface water input.

▪ Rainfall enters the wetland directly during rain events (measured at a nearby electronic weather station (Lichfield EWS), 4 km to the east).

▪ Evapotranspiration is the combination of evaporation[1] and transpiration[2]; estimates were obtained from the nearest station that measures evapotranspiration (Waikeria EWS, 33 km to the west). Evapotranspiration is negligible during rain events (0-2 mm/d).

▪ Groundwater can seep into the wetland through its base across a wide (but unknown) spatial extent[3].

▪ The net surface discharge from the wetland complex (C5 discharge) is:

$$Cell\ 5\ discharge = Cell\ 1\ discharge + Rainfall + \text{net}\ Groundwater\ inflow - ET$$

So that:

$$Groundwater\ inflow = Cell\ 5\ discharge + ET - (Cell\ 1\ discharge + Rainfall)$$

Although the non-laneway drainage inflow was not measured directly, it may be estimated as follows:

$$\text{D} = Cell\ 1\ discharge - L$$

The laneway inflow was transient, and only occurred during or subsequent to rainfall events. During the 2019 drainage year, bypass flow occurred once (see below). Under baseflow conditions (the bulk of the 2019 drainage year), the laneway inflow was zero, and the farmland drainage inflow was the same as the cell 1 discharge.

As Figure 2 indicates, under very heavy rainfall conditions a minor proportion of the laneway inflow bypassed the monitoring point (L) as well as cell 1, and entered the wetland complex downstream of the cell 1 discharge monitoring point.

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[1] Direct water losses from a water surface
[2] Water losses through plant leaves.
[3] Water loss via infiltration through the wetland base only occurs during periods of low groundwater levels during summer.

Surface inflow volume measurement

Discharge (Q) entering wetland cell 1 from the farm laneways (via a culvert) was measured using an ES&S PumpPro 6150 (which estimates water depth using the hydrostatic pressure of water above a submersed bubbler) at 2-minute intervals. Water depth was converted into discharge using the Manning formula[4] for flow in an open pipe as described by Bengtson (2000) (Eq. 1):

$$Q = (1/n)A(Rh^{2/3})S^{1/2}$$

Where:

▪ Q is the volumetric flow rate passing through the channel reach in m^3 s^{-1}.

▪ A is the cross-sectional area of flow normal to the flow direction in m^2.

▪ S is the bottom slope of the channel in m m-1 (dimensionless).

▪ N is the Manning Roughness coefficient, an empirical constant (dimensionless).

A Manning Roughness coefficient of 0.013 (appropriate for centrifugally spun concrete pipes) was used.

Estimate of laneway inflow when bypass flow was occurring

The farmland drainage inflow is the dominant surface water inflow in terms of volume. During rainfall events, the laneway input becomes significant, as runoff mobilises and transports materials from the laneway into the wetland.

During one high intensity rainfall event, a proportion of the laneway runoff was observed to have bypassed cell 1 and entered the wetland downstream of the cell 1 wetland inflow monitoring point. Estimation of the total volume which would have been present at cell 1 was estimated based on total flows measured at cell 5 during this event, and comparison with a similarly sized event which occurred 11 days later where bypass flow did not occur.

Direct groundwater seepage inflows to the wetland

The net contribution of direct groundwater seepage into the base of the wetland was calculated by water balance as noted above. Four groundwater sampling wells (W1-W4; Figure 1) were used to sample incoming groundwater seepage. The relative contribution of the different groundwater flow areas (represented by each well) to the wetland was estimated on two occasions under baseflow conditions using a standard salt dilution technique (Lamontagne et al. 2002, Shafer et al. 2010).

Sampling

Water quality sampling was undertaken manually at the outflow of each cell, at each groundwater sampling well, and from the seepage input during “base-flow” periods (if flow

[4] Note: this equation is used when a pipe is less than half full, as it was throughout the monitoring period. An alternative equation for pipes more than half full is available in Bengtson (2000).

was occurring). Groundwater samples were collected during baseflow sampling as well as prior to rain event sampling.

Event samples were collected using four ISCO programmable autosamplers (Teledyne ISCO, Lincoln Nebraska, USA) according to a fixed sampling interval (estimated according to the anticipated length of each event derived from weather predictions). The four autosamplers collected water samples from the outlet of cell 1 and cell 5, the seepage input and the laneway input (Figure 1).

Surface samples collected using the autosamplers and groundwater samples were analysed in the laboratory for the same set of variables as those collected during manual baseflow sampling.

Input and output load estimation

We used a combination of two independent methods to estimate the performance of the wetland complex – the RiverLoad package, which provided annual estimates of surface inflow and outflow loads, and a stratified load estimation method (which was only used for groundwater estimation) because RiverLoad only measures surface water loads.

RiverLoad incorporates a range of different averaging methods (methods 1–6), ratio estimators (Beale ratio) and regression methods. Preliminary analysis steps in RiverLoad (which check for a relationship between discharge and concentration) showed that the simple regression methods were not suitable for the Lichfield wetland datasets because the inflow and outflow discharge-concentration correlations were low ($r<0.3$) and varied between events.

Using the stratified method, periods of inflow were arbitrarily separated into baseflow and event flows, depending on whether they were associated with a distinct rain event or not. Groundwater loads were estimated by using the total volume of groundwater entering the wetland. Groundwater flow volume was apportioned between the four inflow “zones” represented by the four monitoring wells based on relative flows measured by the salt dilution experiment. The total volume for each well or zone was multiplied by the concentration for each well, providing a total mass for each groundwater zone. These four groundwater contaminant mass estimates were summed to provide an estimate of the total mass of each contaminant for each event, and for the period of baseflow.

Table 2. Summary of different load estimation methods in the RiverLoad R package.

Method	Name	Description	Algorithm	Comments
1	Time weighted Q and C.	Mean C x mean Q at time of sampling.	$L = K\left(\sum_{i=1}^{n}\frac{C_i}{n}\right)\left(\sum_{i=1}^{n}\frac{Q_i}{n}\right)$	Reported to be precise, but can be biased and underestimate load.
2	Discharge weighted C.	Mean of instantaneous loads (Ci x Qi), all concentrations and flows equally weighted.	$L = K\left(\sum_{i=1}^{n}\frac{C_i Q_i}{n}\right)$	Large bias for discrete samples.
3	Mean discharge weighted C.	Each Ci x mean Q for interval between sample and previous sample.	$L = K' \sum_{i=1}^{n} C_{i\overline{\overline{Q_{i,i-1}}}}$	
4	Time weighted C.	Mean C x mean Q over the period.	$L = K\bar{\bar{Q}}\left(\sum_{i=1}^{n}\frac{C_i}{n}\right)$	Reported to be precise, but can be biased.
5	Time and discharge weighted.	Weights mean daily load by the mean of all measured flows.	$L = K\frac{\sum_{i=1}^{n} C_i Q_i}{\sum_{i=1}^{n} Q_i}\bar{\bar{Q}}$	Can result in large variability in load estimates.
6	Linear interpolation of C.	Simple linear interpolation between samples.	$L = K'' \sum_{i=1}^{n} C_j^{int} Q_j$	This method will underestimate unsampled events.
Beale ratio	Beale ratio (with bias correction).	Mean daily load (C x Q on days when samples taken) multiplied by flow ratio (average Q/average Q on sample days). A bias correction factor is included.	$L = Q\frac{\bar{l}}{\bar{\bar{q}}}\left[\frac{1+\frac{1}{n}\left[\frac{Cov(l,q)}{\bar{l}\bar{q}}\right]}{1+\frac{1}{n}\left[\frac{Var(\bar{q})}{\bar{q}^2}\right]}\right]$	Produces robust and statistically unbiased results.

In the algorithms, Ci is the instantaneous sample concentration, Qi is the instantaneous discharge at time of sampling, n is the number of samples collected, K is a conversion factor to account for measurement units, $\bar{q}$ is the mean flow for times when measured and $\bar{l}$ is the mean load for times when samples were collected.

Results

Weather and drainage flow volumes

Rainfall during the 2019 drainage year (1242 mm) was 87% of the average annual rainfall (1 422 mm) for the period 2000-2019. Rainfall in the 2019 drainage year was the fourth lowest recorded during the 20-year period. The calculated runoff for this period (467 mm) (Figure 4) was the lowest estimated over the 20 years of record.

As a result of generally dry conditions, outflow from the first wetland cell was intermittent between 5 July 2019 and 25 October 2019. The longest period of continuous outflow from cell

1 during this period was 40 days. Discharge occurred from cell 1 over the equivalent of 69.5 days during the 2019 drainage year.

Despite the short periods of surface outflow from cell 1, outflow from cell 5 occurred continuously from 10 June 2019 until 31 October 2019 (145 days), maintained by groundwater entering the wetland system downstream of cell 1. Table 3 provides a hydrological balance for the wetland complex.

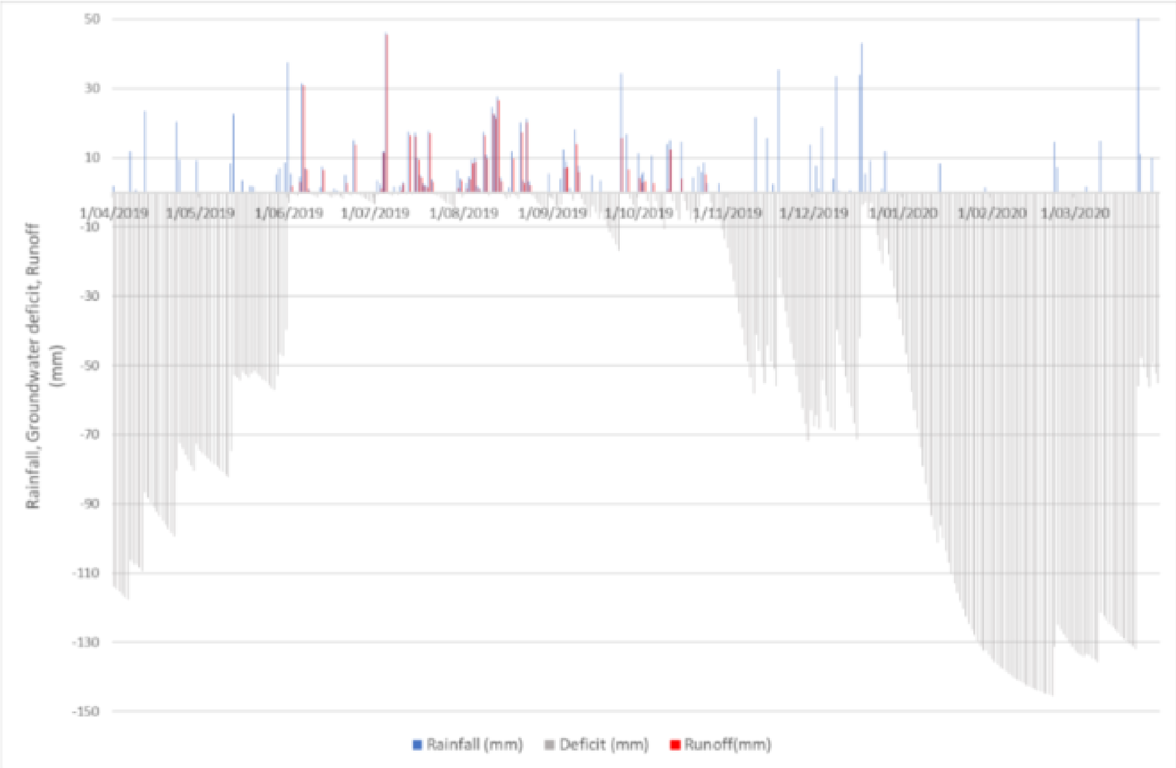

Figure 2. Rainfall, groundwater deficits and runoff at Lichfield EWS weather station. Data retrieved from NIWA national Cliflo database.

Table 3. Hydrological balance of the wetland (2019).

Source of water	Volume (m³)	Comment
Cell 1 discharge	3,900	Direct measurement
Plus groundwater inflow	5,140	Estimated by difference
Plus direct rainfall	4,250	Estimated from remote site
Less estimated evapotranspiration	-2,790	Estimated from remote site
Outflow, Cell 5	2,090	Direct measurement

CW contaminant loads and removal efficacy

The 4 most appropriate estimates from the Riverload modelling are summarised in Tables 4-7. The median of these four methods was used to calculate wetland efficacy.

Table 4. RiverLoad estimates of TSS load. "Method" refers to the RiverLoad method.

Measurement location	Method	TSS load estimates (kg)	Median TSS load estimate (kg)
Cell 1	Four	2 970	3 940
	Five	4 790	
	Six	3 090	
	Beale	4 980	
Groundwater	Sum of (vol x conc.) values	0	0
Subtotal inflow			3940
Cell 5	Four	1 490	1 360
	Five	1 370	
	Six	635	
	Beale	1 350	
Subtotal outflow			1 360
Inflow-outflow			**2 580**
Attenuation efficacy			65%

Table 5. RiverLoad estimates of TN load. "Method" refers to the RiverLoad method. The groundwater component is the sum of four components of the total groundwater load, each calculated as the product of the proportions of total groundwater and median groundwater TN concentrations using the stratified method.

Measurement location	Method	TN load estimates (kg)	Median TN load estimate (kg)
Cell 1	Four	52.7	62.6
	Five	72.5	
	Six	48.0	
	Beale	74.9	
Groundwater	Sum of (vol x conc.) values	50.8	50.8
Subtotal inflow			113.4
Cell 5	Four	52.6	43.4
	Five	43.6	
	Six	31.7	
	Beale	43.1	
Subtotal outflow			43.4
Inflow-outflow			**70.0**
Attenuation efficacy			62%

Table 6. RiverLoad estimates of TP load. "Method" refers to the RiverLoad method. The groundwater component is the sum of four components of the total groundwater load, each calculated as the product of the proportions of total groundwater and median groundwater TP concentrations using the stratified method.

Measurement location	Method	TP load estimates (kg)	Median TP load estimate (kg)
Cell 1	Four	14.7	18.7
	Five	22.7	
	Six	13.8	
	Beale	23.7	
Groundwater	Sum of (vol x conc.) values	1.6	1.6
Subtotal inflow			20.3
Cell 5	Four	9.39	7.01
	Five	7.09	
	Six	4.16	
	Beale	6.93	
Subtotal outflow			7.01
Inflow-outflow			**13.3**
Attenuation efficacy			65%

Table 7. RiverLoad estimates of *E. coli* load. "Method" refers to the RiverLoad method. The groundwater component is the sum of four components of the total groundwater load, each calculated as the product of the proportions of total groundwater and median groundwater *E. coli* concentrations using the stratified method.

Measurement location	Method	*E. coli* load estimates (MPN)	Median *E. coli* load (MPN)
Cell 1	Four	1.73×10^{13}	3.78×10^{13}
	Five	4.92×10^{13}	
	Six	2.63×10^{13}	
	Beale	5.21×10^{13}	
Groundwater	Sum of (vol x conc.) values		4.68×10^{10}
Subtotal inflow			3.78×10^{13}
Cell 5	Four	1.09×10^{13}	1.33×10^{13}
	Five	1.57×10^{13}	
	Six	5.78×10^{13}	
	Beale	1.63×10^{13}	
Subtotal outflow			1.73×10^{13}
Inflow-outflow			**2.45×10^{13}**
Attenuation efficacy			65%

Comparison of 2017 and 2019 drainage years

Key hydrological characteristics of the two drainage years are summarised in Table 8. Estimated annual loads and removal efficacies for 2017 and 2019 are summarised in Table 9.

Table 8. Summary of composition and total nett cell 5 outflow volumes.

Number of events	2017 water volumes (m^3)				Number of events	2019 water volumes (m^3)			
	Surface		Groundwater (+rain-E.T.)	Total		Surface		Groundwater (+rain-E.T.)	Total
	Baseflow	Event				Baseflow	Event		
25	33 520	33 430	29 550	96 500	7	1 380	2 540	6 580	10 500

Table 9. Comparison of estimated annual loads and removal efficacies, 2017 and 2019 drainage years. Negative values indicate the wetland was a nett source of contaminant during a year.

Variable	2017				2019			
	Annual load			Removal efficacy (%)	Annual load			Removal efficacy (%)
	In	Out	Removal		In	Out	Removal	
Nitrate-N (kg)	86.1	158.6	-72.5	-84%	48.9	15.1	33.8	69%
Ammonia-N (kg)	36.8	8.4	28.4	77%	5.11	2.78	2.33	46%
Organic-N (kg)	446.1	117.9	328.2	74%	51.3	25.4	25.9	50%
TN (kg)	569.0	284.9	284.1	50%	113.4	43.4	70.0	62%
DRP (kg)	8.0	8.4	-0.4	-6%	2.58	1.59	0.99	38%
TP (kg)	122	18	104	85%	20.3	7.0	13.3	66%
TSS (kg)	30 980	6 280	24 700	80%	3 940	1 360	2 580	65%
E. coli (MPN)	1.60×10^{12}	2.44×10^{11}	1.36×10^{12}	85%	3.78×10^{13}	1.33×10^{13}	2.45×10^{13}	65%

Discussion

Comparison of 2017 and 2019 results

The total volume of drainage in 2017 was approximately 9 times larger than in 2019, and there were almost four times as many rainfall events (causing discharge to exceed 3 L s^{-1}). In addition, the groundwater input in 2017 was approximately equal to surface inflow, whereas in 2019, groundwater inflow was approximately half (55%) of surface inflows, and during several events, the inflow did not even reach the 3.0 L s^{-1} value used to define the threshold between baseflow and event flow in 2017. These differences have a bearing on wetland performance in the two periods.

TSS loads in 2017 and 2019

The hydrology in 2017 was event-dominated, which explains much of the difference in TSS load estimated for the two periods.

Although approximately eight times more sediment entered the wetland complex in 2017 than in 2019, the mass of TSS leaving the wetland was approximately 4.5 times greater in 2017 than in 2019, and the total mass retained within the wetland was approximately 10 times greater in 2017 than in 2019.

The performance of the wetland was better in 2017 (80% removal) than in 2019 (65%). These results indicate that the wetland complex is able to retain TSS over a wide range of hydrological conditions, and that performance may be maintained by managing (reducing) the mass of material trapped within the first cells of the wetland. Performance in 2019 was degraded by one large event which mobilised materials retained from previous events (possibly even the 2017 drainage year). TSS removal could be maximised by regularly removing accumulated material from the first wetland cells to maintain a target minimum storage volume.

Total phosphorus loads in 2017 and 2019

The load of TP to the CW is primarily associated with particulate materials, so the factors that determined TSS performance are likely to influence TP removal efficacy as well. The wetland retained a greater proportion of TP in 2017 (85%) than in 2019 (66%), analogous to TSS removal.

Nitrogen loads in 2017 and 2019

The dominant form of nitrogen in the TN baseflow load was nitrate-N, which is soluble and unlikely to be influenced appreciably by sedimentation processes. Although the total volume of water passing through the wetland complex was approximately 9 times greater in 2017 than in 2019, the surface water component was 17 times greater in 2017 than in 2019 (66 950 m^3 and 3 900 m^3 respectively). The volume of groundwater entering the wetland in 2017 was approximately 4.5 times greater than in 2019 (29 550 m^3 and 6 580 m^3 respectively), which substantially increased the nitrate-N load during 2017 relative to 2019. The efficacy of denitrification is dependent on several factors: the mass of nitrate-N in the inflow, the availability of organic carbon, and the residence time for microbially-mediated process to occur. The large volume of water passing through the wetland in 2017 resulted in reduced retention times and contributed to the relatively poor performance observed (the wetland appeared to be a net source of nitrate-N in 2017). Another factor was the uncertain estimate of mass of nitrate-N transported into the wetland complex as groundwater.

In 2019, however, the mass of nitrate-N introduced into the wetland complex as groundwater was better-estimated, the smaller hydrological load allowed longer residence times in the wetland, and the mass of nitrate-N was more likely to be balanced by that of organic carbon; these factors probably contributed to the higher removal efficacy observed in 2019.

Wetland performance during high flow events

There have been questions from the agricultural sector, regulators and policy-makers regarding the performance (efficacy of contaminant removal) of wetlands that receive a large proportion of incoming loads during brief rain events. These stakeholders have raised the following questions:

- Will attenuation and removal mechanisms be overwhelmed by high water and contaminant inflows?

▪ What effect will preceding loads have on future performance?

▪ What can be done to maintain acceptable wetland performance?

Results obtained over two very different drainage years indicate that removal performance of a wetland receiving high TSS loads can remain high, provided the wetland has been designed to facilitate retention of sediments (e.g., by incorporating ponds or basins specifically for the capture of TSS). Wetland designs that assist with maximising TSS removal are likely to improve TN and TP removal efficacies as well. There was evidence of reduced efficacy of attenuation of particulate material during the 2019 drainage season, as well as resuspension of solids previously accumulated during one event. Removal of deposited sediment from sedimentation ponds or basins will help reduce mobilisation of previously deposited particulate material, as well as the discharge of soluble forms of P and N that may be generated from the sediments under favourable biogeochemical conditions.

The total mass of nutrients stored in above ground plant biomass is small compared with the total nutrient store in the wetland, and thus manual removal of plant biomass (i.e. harvest and removal) is unlikely to substantially improve constructed wetland nutrient removal. In addition, the carbon component of wetland plant material deposited in the wetland represent a vital energy source for denitrification. Further, a good cover of mature wetland plants reduces the opportunity for invasion of weed species. Thus we do not recommend harvest and removal of wetland plant materials as a nutrient removal strategy.

Acknowledgements

The constructed wetland project (comprising design, construction and monitoring phases) was undertaken as a partnership involving DairyNZ, Baldwin Family Trust, Opus International Consultants (Hamilton) and Hill Laboratories, with additional support from Waikato Regional Council and NIWA.

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