

ASSESSING FARM-SCALE NUTRIENT FLOW PATHWAYS AND NITRATE ATTENUATION IN RANGITIKEI SAND COUNTRY

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Abstract

Recently, there has been significant agricultural development in lower North Island coastal dune areas, which have become viable for production under irrigation. However, little is known about transport and fate of nutrients lost from intensive farms in this environment. A case study farm was selected and monitored from April to September 2016, to assess flow pathways of nitrogen (N) and phosphorus (P) and N transformation ‘attenuation’ in an intensively-farmed, irrigated dairy farm within the Rangitikei dune country. A total of nine piezometers and twelve surface water sample sites were established to monitor shallow groundwater and surface waters in drains and stream flowing through the farm. The shallow groundwater piezometers were divided between four sites covering main soils types on the farm, and extended to depths of 3 and 6 m below ground level (bgl). The groundwater samples were collected monthly and analysed for nitrate-N, ammoniacal-N, total-N, dissolved reactive P and total-P to monitor nutrient flows. Dissolved organic carbon, manganese, iron and dissolved oxygen parameters were also measured and analysed to assess the reducing potential of the shallow groundwater. The surface water samples were collected fortnightly and analysed for different forms of N and P to assess any potential loss of nutrients through open surface drains and the freshwater stream that runs through the property.

The six-month monitoring of the farm found that the open surface drains on the property accumulate nitrate-N and could be a significant contributor of nitrate-N to the local stream. Also, dissolved reactive P was more common in shallow groundwater environment. The shallow groundwater appears to have a strong reducing environment, conducive for potential denitrification of nitrogen leached from the soils surface. This could have implications for land and N management strategies in the future. These research findings will help to develop appropriate in-field or edge-of-field management practices, and inform nutrient management plans for intensified land use to maintain or enhance water quality in the region.

Introduction

Background

Agricultural production is an important part of the primary sector’s exports in New Zealand. With the advent and increasing popularity of intensive irrigation systems, land that was previously inhospitable now has the potential to support dairy farm operations. The coastal sand belt located in the Rangitikei area of the Horizons region in the North Island is an example of this type of land. However, as this is a reasonably new development, its impact on local waterbodies (both surface and subsurface) is not fully understood, nor the nutrient flow pathways from source to sink.

The 'behaviour' of nutrients from farm processes on this sand country environment is of concern, as there is a year-round high water table on largely undeveloped soil beneath the farm. What are the key transport pathways and sensitive areas? The mainly sandy soils with low organic matter (OM) present a risk of nutrient loss (particularly nitrate-N) from the root zones, especially while surface soil OM is still developing (MPI, 2015). Therefore, the potential extent of this should be explored. Modelling is a desirable end-product of nutrient research, to map out nutrient flow pathways and apply data catchment-wide. This is important to: accurately estimate nutrient losses and outputs; gain an idea of the actual importance of denitrification; and understand how it is affected by subsurface hydrology. However, experimental data and analysis needs to be collected to form a base on which a model can be developed.

Also, a previous study (Collins, Singh, Rivas, Palmer, Horne, Roygard & Matthews, 2016) has indicated that the Rangitikei has a groundwater system conducive to denitrification, a step in the N cycle. This process would naturally convert nitrate in the system to less harmful forms, thereby attenuating nitrate-N concentrations in the subsystem. With N, finding a way to enhance attenuation in waterbodies could help minimise the damaging effects of N-loss. The capacity of this process is variable between and within catchments. Research into the denitrifying capacity of an area would help to understand the capacities of a catchment or perhaps match land use capabilities more effectively.

Methodology

Case study site

The case study farm is an intensive farm located in the Rangitikei coastal sand belt in the Santoft area, and has a total approximate area of 437 ha (406 ha effective). It is composed of sand dune complexes, and regionally-significant streams, lakes and wetlands supported by groundwater flows. The dairy unit has a series of drains that run parallel to and eventually join with the stream that flows through the property. Therefore, any contaminants from the farm and its drains have the potential to be carried to these larger surface water bodies.

The coastal sand belt experiences low summer rainfall and the groundwater in this area is in high demand. There are two large centre pivots that cover most of the paddocks in the dairy unit to irrigate 297.5 effective ha of land, and 108.4 ha of the paddocks are also irrigated with effluent. The study site utilises land that has very sandy soils and dunes. Parts of the land have been re-contoured, meaning that dunes have been levelled and covered with peat. The site therefore has a combination of sandy loams, loamy sands and peaty loams, with weakly developed structure and poor topsoil development.

The milking platform works on a two-year lactation cycle, with a milking herd size of 3.1 peak cows per hectare, and approximately 20% of the herd milked through winter and through the following season. The young stock is wintered off-farm and cows are typically wintered on the milking platform (J. Hughes, personal communication, September 30, 2016).



Figure 1. Dairy unit water monitoring sites. Surface water sites are on the local stream (blue) and surface drains (red). The groundwater sites are shown in yellow.

Groundwater monitoring sites

A total of four groundwater sites were established to monitor shallow groundwater under selected paddocks, representing the main land use classes and soil types identified at the study farm (Figure 9). Each site was established by installing two to three PVC piezometers, at around 3 m and 6 m below ground level (bgl) (Table 1).

Table 1. Groundwater monitoring site descriptions for the Hyde Park dairy unit

Site	Piezometer codes	Piezometer depths (m bgl)	Soil type	LUC class
Site A	SA-1	1.9-2.4	Foxton brown sand + Pukepuke black sand	IIs9
	SA-2	4.55-5.05		
Site B	SB-1	2.9-3.4	Pukepuke black sand + Foxton brown sand	IIIs8/IIIw5
	SB-2	4.5-5.0		
	SB-5*	0-2.2		
Site C	SC-2	2.8-3.3	Foxton brown sand + Himitangi sand	IIIs8/IIIw5
	SC-3	5.7-6.2		
Site D	SD-1	2.5-3.0	Omanuka peat, permanently high water table	VIw2
	SD-2	5.0-5.5		

**SB-5 was installed half-way through the sample period, so data collected only covers from July-September, 2016*

Due to the electron acceptor hierarchy, parts of groundwater could be separated into pockets or ‘zones’ that use specific electron acceptors for reducing reactions (McMahon & Chapelle, 2007). Groundwater can, therefore, be spatially and temporally complex in terms of its potential denitrification in a given area. By having a transect of piezometer sites that cover a range of soil and land use capability classes, any potential spatial variation in shallow groundwater can be observed and assessed at the study farm.

Surface water monitoring sites

Surface water quality monitoring was used to determine the nutrient concentrations temporally and spatially across the farm's drains and the local stream. A total of twelve surface water monitoring sites were chosen based on the stream and two main drains on the farm. They were selected to assess any changes in nutrient concentrations along the stream's and drains' lengths, and any impacts of drain water on stream water.

Sampling regime

Groundwater sampling

The piezometers were used to retrieve monthly groundwater samples. The samples were filtered through 0.45 µm filters using a syringe into labelled 50 mL containers. Two replicates were collected per parameter. These filtered samples were analysed in the laboratory for forms of nitrogen and phosphorus (Rosen, Cameron, Taylor, & Reeves, 1999). All water samples were analysed for total N, nitrate-N, total P, and DRP concentrations.

When considering the impact of nitrate contamination in a catchment, factors that affect the nutrient's persistence in the environment must also be taken into consideration. Nitrate is reactive and redox sensitive, so can be denitrified or reduced to less harmful forms if the redox conditions are suitable. Therefore, the attenuating capacity of the subsurface hydrological environment was investigated using groundwater samples collected at the same time as nutrient samples. This was done by analysing shallow groundwater samples for certain cations and comparing the concentrations to set threshold values, in order to establish the type of redox process occurring in that area (McMahon & Chapelle, 2007) (Table 2). Manganese (Mn^{2+}) and iron (Fe^{2+}) ions, DO, nitrate-N and DOC concentrations were used as the redox-sensitive indicators of a reducing environment. Their concentrations help to determine whether groundwater conditions in the saturated environment are favourable for oxidation or reduction attenuation and what process is present (Stenger et al, 2013). The measured concentrations of redox species from the analysis were compared to threshold values identified in Table 5 to ascertain the kind of redox environment is present (McMahon & Chapelle, 2007).

Table 2. Framework for assessing subsurface redox conditions based on threshold concentrations, established by McMahan & Chapelle (2007)

Redox Process	Water Quality Criteria (mg/L)					Notes
	O ₂	NO ₃ ⁻ -N	Mn ²⁺	Fe ²⁺	SO ₄ ²⁻	
Oxic						
O ₂ reduction	≥0.5	-	<0.05	<0.1	-	-
Suboxic						
-	<0.5	<0.5	<0.05	<0.1	-	Further definition of redox process not possible
Anoxic						
NO ₃ ⁻ reduction	<0.5	≥0.5	<0.5	<0.1	-	-
Mn(IV) reduction	<0.5	<0.5	≥0.05	<0.1	-	-
Fe(III)/SO ₄ ²⁻ reduction	<0.5	<0.5	-	≥0.1	≥0.5	-
Methanogenesis	<0.5	<0.5	-	≥0.1	<0.5	-
Mixed						
-	-	-	-	-	-	Criteria for more than one redox process are met

Surface water sampling

Surface water from selected farm drains and the stream sites were sampled fortnightly for six months to capture changes from April to September. ‘Grab’ sampling was used, which means taking a quantity of water from the surface with a container. Plastic bottles (1L) were filled with flowing water without disturbing any settled sediment in the channel and filtered later in the laboratory using 0.45 µm filter paper and a vacuum flask. They were analysed for the same nutrient parameters as the groundwater samples.

Results and discussion

The concentrations of each nutrient parameter were analysed in each monthly groundwater sample and each fortnightly surface water sample. The detection limit of analysis is indicated on the graphs below by a black line. Below this point, the values are not fully reliable.

Nitrogen in shallow groundwater

Nitrate-N concentrations in many of the groundwater samples were measured below the detection limit (<0.125 ppm). In August 2016, the nitrate-N concentrations were measured from 0.26-0.83 ppm, and were highest at site SD-1 and SD-2. This peak was possibly due to the winter rainfall effect, with deep percolation and drainage causing an influx of nitrate-N into groundwater at this time. In terms of the standards and guidelines, the results are significantly below the WHO guideline of 11.3 ppm. Also, by surface water standards for ecological health, a majority of the sampling events were less than the standard of 0.44 ppm nitrate-N (Davies-Colley, 2000).

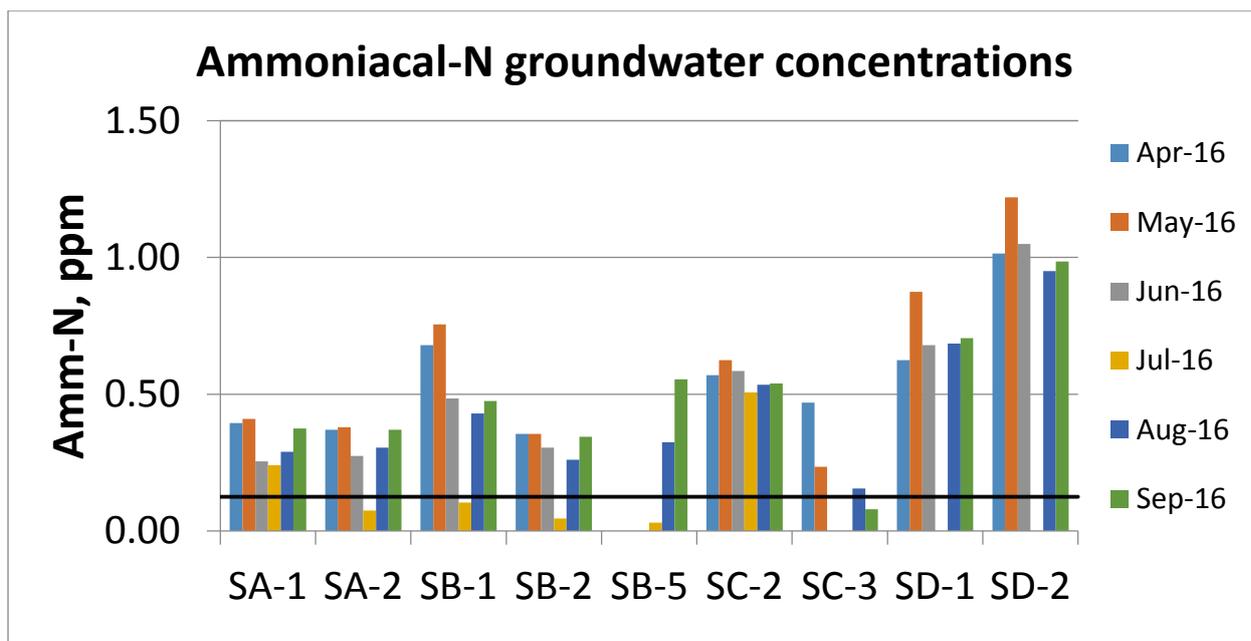


Figure 2. Ammoniacal-N concentrations in groundwater samples under Hyde Park dairy unit from April to September, 2016

The primary form of inorganic N in the shallow groundwater samples was ammoniacal-N, with sample concentrations ranging from <0.125-1.22 ppm over the sample period (Figure 1). The frequency of ammoniacal-N to the total-N concentrations from the same well and same sampling event were compared. This indicated that generally at least 50% of the total-N in samples could be explained by ammoniacal-N. In some cases, more than 80% of the total-N can be explained by ammoniacal-N. Nitrate-N was less notable, with only one sample event (August 16) showing consistent significant values. The ammoniacal-N concentrations were relatively higher at site SD-1 and SD-2 as compared to the other sites. The results from this site are greater than their Lower Rangitikei catchment average. These standards slightly exceeded the standard value of 0.021 ppm ammoniacal-N for freshwater ecological health (Davies-Colley, 2000).

Redox in shallow groundwater

The results obtained from analysing groundwater for DO, nitrate-N, Mn^{2+} and Fe^{3+} were applied to the framework created by McMahon & Chapelle (2008) (Table 3). This assessment provides an indication of the potential reducing capacity of shallow groundwater and therefore the potential for nitrate reduction and attenuation in the subsurface environment of the study area. The average values for each parameter are displayed, as well as the maximum and minimum values from the entire sample period. Most of the groundwater sites were assessed as anoxic and iron/sulphate reducing apart from site SC-3, which was assessed as mixed 'oxic-anoxic'.

Table 3. Averaged redox parameters for groundwater sites using the thresholds assigned by McMahon & Chapelle (2008)

Depth (m)	Redox variables	DO	NO ₃ ⁻ -N	Mn ²⁺	Fe ³⁺	Redox category	Redox process
	unit	ppm	ppm	ppm	ppm		
	Threshold value	0.5	0.5	0.05	0.1		
1.9-2.4	SA-1	0.41 (0.77-0.30)	0.14 (0.0-0.29)	0.46 (0.41-0.58)	7.24 (5.94-10.26)	Anoxic	Fe(III)/SO ₄
4.55-5.05	SA-2	0.19 (0.15-0.25)	0.12 (0.02-0.31)	0.51 (0.46-0.66)	7.84 (7.10-10.26)	Anoxic	Fe(III)/SO ₄
2.9-3.4	SB-2	0.38 (0.15-0.86)	0.13 (0.0-0.15)	0.20 (0.17-0.24)	4.25 (1.67-6.74)	Anoxic	Fe(III)/SO ₄
4.5-5.0	SB-3	0.21 (0.12-0.25)	0.11 (0.0-0.26)	0.31 (0.23-0.35)	6.03 (5.16-7.16)	Anoxic	Fe(III)/SO ₄
0.0-2.2	SB-5	0.46 (0.38-0.52)	0.14 (0.02-0.33)	0.49 (0.29-0.82)	4.23 (3.74-4.56)	Anoxic	Fe(III)/SO ₄
2.8-3.3	SC-2	0.32 (0.20-0.41)	0.18 (0.05-0.51)	0.67 (0.05-0.83)	4.91 (0.96-7.02)	Anoxic	Fe(III)/SO ₄
5.7-6.2	SC-3	1.33 (2.82-0.44)	0.12 (0.10-0.20)	0.23 (0.06-0.61)	3.57 (0.0-7.18)	Mixed (Oxic-Anoxic)	O ₂ ⁻ Fe(III)/SO ₄
2.5-3.0	SD-1	0.40 (0.25-0.60)	0.18 (0.06-0.60)	0.97 (1.0-0.94)	7.37 (7.92-5.90)	Anoxic	Fe(III)/SO ₄
5.0-5.5	SD-2	0.29 (0.16-0.51)	0.20 (0.01-0.13)	1.48 (1.13-1.59)	6.84 (7.06-6.10)	Anoxic	Fe(III)/SO ₄

The oxidative-reductive potential (ORP) values were consistently negative, indicating reducing conditions. The DOC levels from the groundwater system at Hyde Park were measured higher from 8.01-13.22 ppm, and the analysis also indicates a large concentration of Fe³⁺ ions ranging from 3.57-7.84 ppm. Therefore, there is a suitable supply of electron donors for both heterotrophic microbial processes using DOC and autotrophic microbial processes using inorganic substrates like iron or sulphate.

Overall, based on the six-month sampling and analysis presented above, it is likely that nitrate would be disposed to reduction in shallow groundwater under Hyde Park dairy unit. This is promising, as nitrate attenuation characteristics could be valuable to intensive dairy farming where losses to the subsurface are inevitable. By expanding and outlining the redox in a catchment, the information could be put forward towards planning and management that is more tailored to a catchment's natural nitrate attenuation capacity. This information would be useful when making decisions on acceptable levels of intensification or suitable land use types in different parts of a catchment. However, the information here is only an assessment of the conditions in the catchment and is not a true measurement of denitrification capacity. To expand on the potential for denitrification, *in-situ* investigations like the push-pull test should be employed (Collins et al., 2016).

Investigating *in situ* denitrification capacity further would be beneficial, as the amount of ammoniacal-N in the samples compared to nitrate-N raises questions about the form of reduction occurring. One reason for higher ammoniacal-N concentrations could be dissimilatory nitrate reduction to ammonium (DNRA) rather than denitrification or a combination of the two when nitrate becomes a limiting factor (McMahon & Chapelle, 2008; Rivett et al., 2008). This is another means for nitrate-N to be transformed under low-oxygen conditions, terminating at the ammoniacal-N form. Denitrification removes nitrogen completely from the subsurface system if the nitrate-N is transformed to N₂ (Behrendt, 2014). DNRA can actually preserve inorganic, reactive and bioavailable nitrogen in a system, meaning harmful effects on water quality can still occur (McMahon & Chapelle, 2008; Rivett et al., 2008).

Phosphorus in shallow groundwater

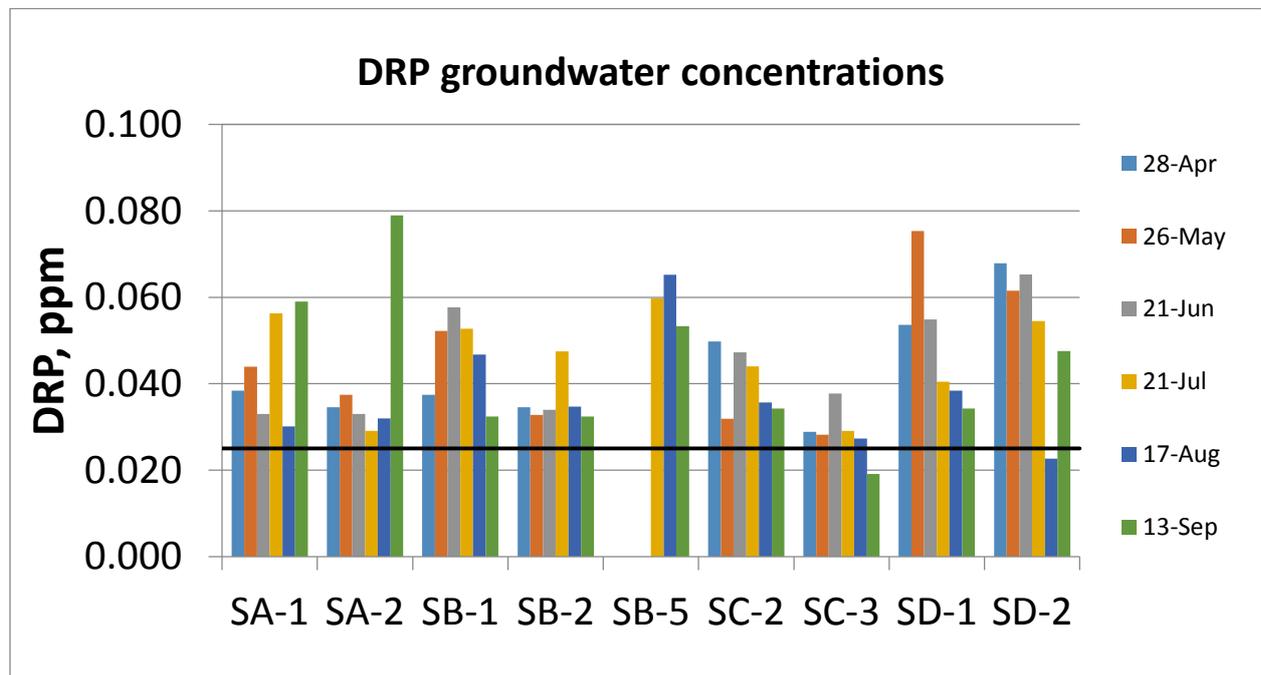


Figure 3. DRP concentrations in groundwater samples under Hyde Park dairy unit from April to September, 2016

Figure 2 shows that there were significant levels of DRP were present in shallow groundwater samples at the study farm. A majority of the P in groundwater samples was present as organic P, which is not readily available like DRP but can become available over time.

Nutrient concentrations in surface waters

Nitrogen in surface waters

In general, nitrogen was present more as nitrate-N in surface water samples, particularly in samples from the drain. The range of nitrate-N in drains was from <0.25-6.59 ppm, while ammoniacal-N concentrations ranged from <0.25-0.62 ppm.

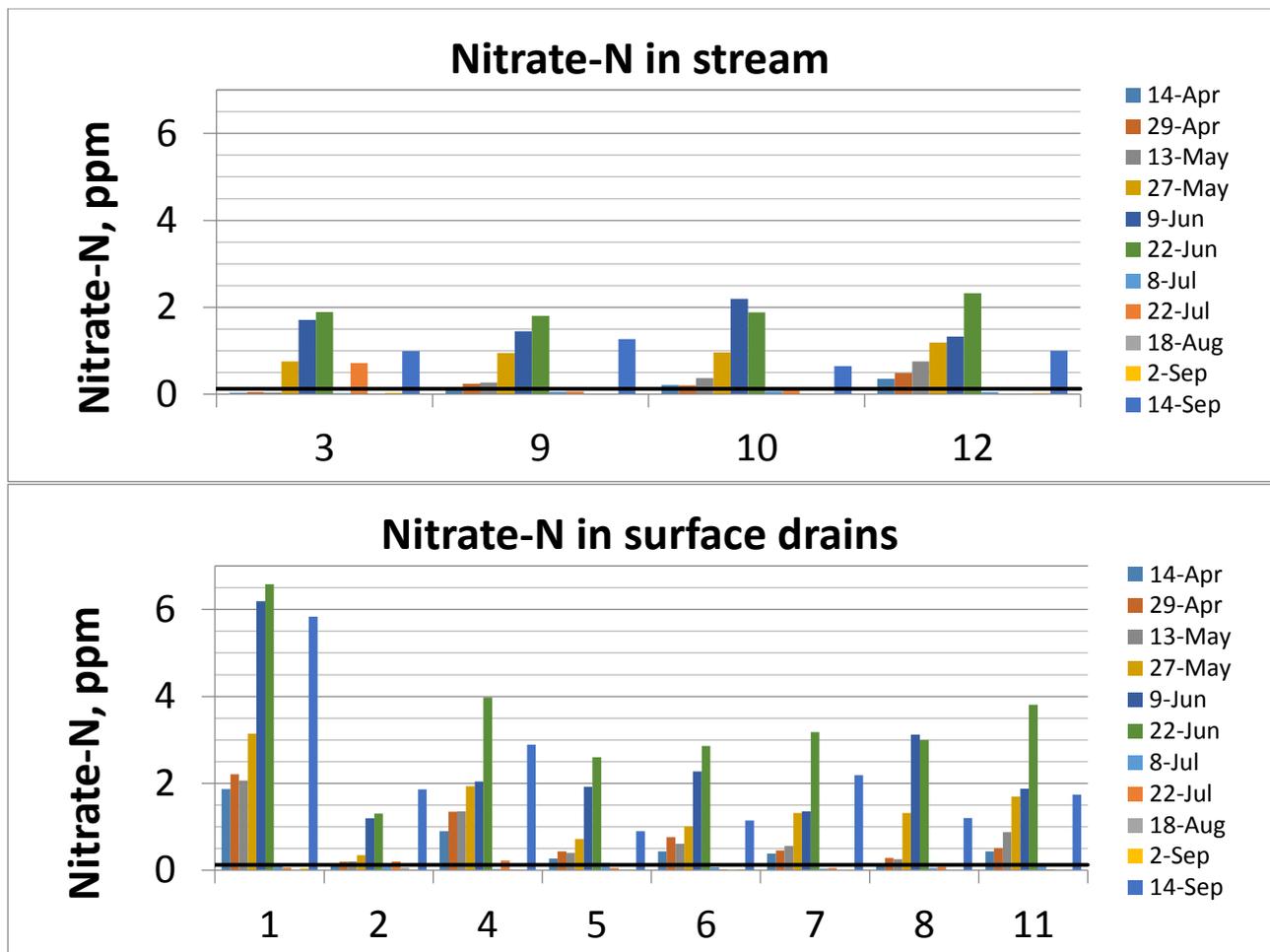


Figure 4. Nitrate-N in surface water in Hyde Park dairy unit

Figure 3 reproduces nitrate-N concentrations measured at surface water sites at the study farm. For the sampled sites along the stream, there was a steady increase in the concentrations as the sampling moved into the wetter months, then a sudden drop in the concentrations again by July 2016. There was a spike in nitrate-N concentrations again at the second September sampling. Site 12 on the stream is situated at the property boundary after all of the drains join the stream. Samples from this site began to exceed the nitrate-N standard from April 29 sample event. However, sample sites on the stream before the main drains joins it only begin to exceed the nitrate-N standard value from May 27. From the May 27 sampling to June 22, all monitoring sites along the stream did not meet the nitrate-N standard value of 0.44 ppm (Davies-Colley, 2000).

In some sampling events, nitrate-N concentrations increased gradually along the stream. For instance, the concentrations of the four monitoring sites from site 3 (upstream) to site 12 (downstream) were 0.76, 0.95, 0.96 and 1.19 ppm respectively on May 27. A study by Cameron et al. (2014) also showed this trend during periods of high stream and drain flow, which is the case for the results for Hyde Park. However, this trend is not always the case, as the sample events on June 9 and September 14 showed inconsistent nutrient concentrations along the stream. For instance, the sampling event that occurred on June 9. This trend may have been affected by dilution from drain inputs when drains were flowing more, or groundwater may be recharging surface water at certain points, diluting nitrate-N or adding to it.

The drain containing the monitoring sites 1, 4, and 11 showed a decrease in nitrate-N concentrations from upstream (site 1) to downstream (site 11). For instance, the nitrate-N concentrations on July 22 were 6.59, 3.98, and 3.81 ppm at sites 1, 4 and 11, respectively. These figures show that higher values of nitrate-N concentrations came from site 1. This site is at the beginning of the property, and is adjacent to one paddock used for pasture and another used for cropping. As this site is representative of the ‘beginning’ of that drain on Hyde Park, this suggests that either something up-gradient of the dairy unit is affecting composition, or there is something in the vicinity that is having such an effect. During the entire survey period, this drain was directly adjacent to a paddock that had been crop, with a large amount of bare soil. N mineralisation was likely to be occurring in this paddock due to decomposing matter and fertiliser use, which could have moved to the drain via the subsurface pathway identified above. This land use can be considered a probable source and poses the question of what is happening to this large quantity of N from upstream to downstream. In contrast, the drain with monitoring sites 2, 5, 6, and 7 displayed a general trend of increasing nitrate-N concentrations along the drain. This drain terminates at site 10 on the Koitiata stream, where this accumulated nitrate-N enters into the stream system. When comparing the nitrate-N concentrations of the stream to the drains, the concentrations in the drains were sometimes significantly greater. Stream values ranged from <0.25-2.32 ppm, while drain values ranged from <0.25-6.59 ppm.

The percentage of total-N that could be explained by nitrate-N was calculated using the measurements presented in the graphs. The average percentage of total-N that is nitrate-N was 44% for the stream measurements and 52% for the drain measurements. Compared to this, the average percentages of total-N that could be explained by ammoniacal-N was 1.8% for the stream measurements and 6.3% for the drain measurements. This clearly highlights nitrate-N as the main form of inorganic N loss to surface water.

Subsurface movement is a transport pathway that can contaminate surface water. Generally, nitrate-N leaches into the subsurface and is then transported to surface waters by subsurface drainage or base flow (Wang, Geo, Li, Zhang & Wang, 2015; Randall & Mulla, 2001). It is much less likely to be lost via surface runoff as its mobility makes it prone to leaching into the soil profile (Randall & Mulla, 2001). By having a drain network that extensively cover the far reaches of the property, there is opportunity for subsurface nitrate-N to move to surface water. From there, drains can act as direct conduits and therefore can be a point-source of contaminants into adjoining waterways. Activity on land that surrounds a drain upstream can affect the downstream system and therefore the waterways they flow into (Ministry for the Environment, 2001).

Phosphorus in surface waters

During the survey period, no samples from any surface water site on the property gave DRP concentrations greater than the detection limit apart from at site 9 on May 27th. At this time, the concentration was 0.058 ppm DRP. Meanwhile, there were some events where there were significant levels of total-P in stream and drain samples from the end of June. Before this point, concentrations from surface water appeared too insignificant to be read by the autoanalyser that was used.

Conclusion

The main findings of this case study are:

- The main common form of N in groundwater was ammoniacal-N, and nitrate-N in surface water;
- DRP was more consistently present in groundwater samples, and very rarely measurable in surface water samples;
- Levels of nitrate-N, ammoniacal-N, and total-N in groundwater were significantly lower compared to their levels measured in surface drains;
- The redox conditions of groundwater appear to be conducive to processes that would reduce nitrate-N, via denitrification and/or;
- DNRA is a reducing process in the subsurface environment.

The study clearly identifies surface drains as a main pathway for nutrient loss from the study farm. It raises the question of whether having deep open drains is beneficial in terms of nutrient management. In this situation, it is likely that the drain network is acting as an accumulator and direct conduit for diffuse N pollution from the surrounding farmland that would otherwise go to groundwater. There are some areas that could be explored further to expand the research.

- The level of connectivity between surface and ground water could be assessed, in order to fully understand the threat of nutrients in groundwater to surface water quality;
- Carry out an *in situ* measurements of shallow groundwater reducing processes and capacity, to fully qualify the likelihood of high nitrate attenuation in the subsurface environment;
- As the groundwater in this area appears to be reducing and conducive to natural nitrate-N attenuation, the validity and potential benefits of controlling surface drainage into drains could be explored;
- Carry out a feasibility analysis establishing wetlands to treat surface drainage water to reduce nutrient loadings for intensive farms in the region; and
- Investigate the potential for using this information to inform and alter nutrient and land management planning, and the validity of including information like this in a nutrient budgeting programme like OVERSEER®.

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