

Farm staff, Field Service Centre staff and postgraduate students. Thanks to Barbara Dow and Alison Lister for their assistance with statistical analyses. Helpful review comments on the manuscript from Warwick Harris and Keith Widdup are gratefully acknowledged.

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## Reducing nitrogen fertiliser alters dairy shed effluent quality

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### Abstract

Dairy farmers using low rates of nitrogen fertiliser observed reduced odour in milking sheds, and a smaller growth response of pasture to effluent application. Effluent samples from four conventional (>100 kg N/ha/year, "high-fert") and three low-nitrogen (N) fertiliser (<50 kg N/ha/year, "low-fert") properties were collected in January 2017 and analysed for total N concentration, N form, mineral nutrient concentration and pH. Total effluent N concentration was comparable between both classes of farm. However, low-fert properties had a higher proportion of N in organic forms as opposed to ammoniacal-N than high-fert properties (mean 75% and 59% organic on low- and high-fert properties, respectively,  $P < 0.01$ ). Low-fert effluent also had a lower pH, higher P concentration, and nearer optimal N:P ratio than effluent from high-fert properties. It was hypothesised that reducing N fertiliser may result in more nutritionally-balanced effluent (N:P ratio), causing microbes to multiply more rapidly (lowering pH), storing N in microbial biomass, reducing ammonia emissions and odour, and reducing the risk of N leaching from effluent applied to pasture.

**Keywords:** dairy shed effluent, nitrogen fertiliser, ammonia, organic N, environmental loss

### Introduction

Dairy shed effluent is a valuable source of nutrients for pasture plants on dairy farms (DairyNZ 2015). However, it is also a potential source of nutrient loss predominately via the following pathways: ammonia volatilisation (Li *et al.* 2014), nitrate leaching, and phosphorus runoff. Effluent is highly variable, and the characteristics of effluent will alter the quantity of nutrients available for loss via each pathway, and therefore the potential for environmental losses (Houlbrooke *et al.* 2011).

Nitrogen (N) exists in effluent in both inorganic (ammonium ( $\text{NH}_4^+$ ) and nitrate ( $\text{NO}_3^-$ )) and organic forms. Inorganic N is susceptible to gaseous loss as ammonia ( $\text{NH}_3$ ) and leaching as nitrate. Although dissolved organic nitrogen can leach, as a general rule, organic nitrogen is unlikely to be lost unless it is first converted to inorganic forms (Houlbrooke *et al.* 2011).

Inorganic nitrogen is available to plants and can be rapidly used for growth. Therefore, effluent rich

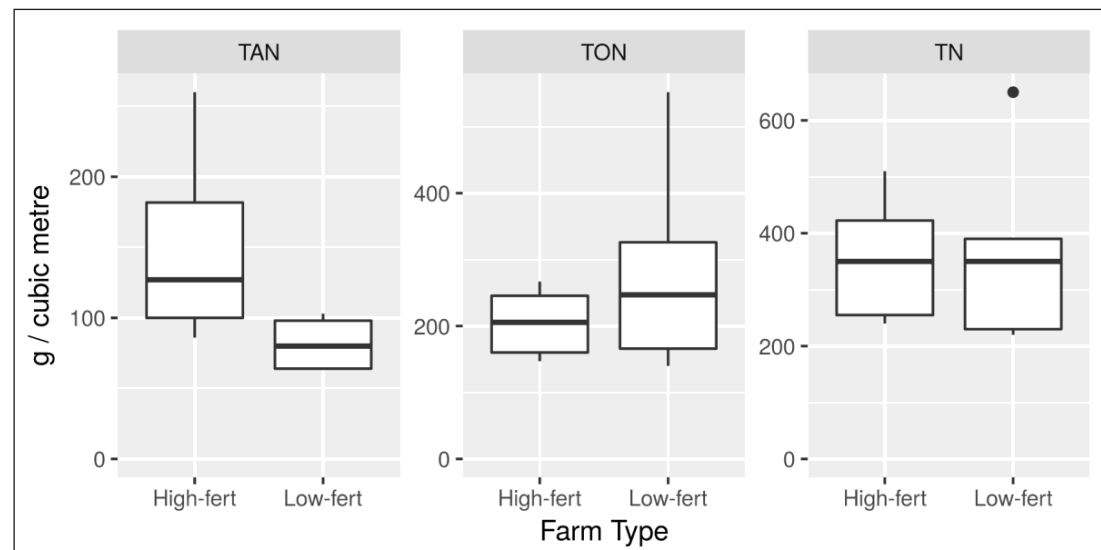
in ammonia and/or nitrate is likely to result in an immediate plant growth response. However, organic nitrogen (other than urea) is slowly released for plant use, and slow-release nutrients may result in a higher long-term pasture growth response per unit of N applied than soluble nutrient applications (Zaman *et al.* 2009).

Farm staff on properties in the study group using low rates of N fertiliser noticed a reduced odour in the milking shed, and also a reduction in the immediate visible pasture response to effluent application, compared with their expectations from past experience under conventional management (A. Lapping pers. comm.). This paper describes the results from a farmer-funded effluent analysis study designed to determine whether these observations were due to actual differences in the characteristics of effluent between farms using low or high rates of synthetic N fertiliser.

### Methodology and analysis

Seven dairy farms were chosen for this study. All were located on free-draining alluvial soil and annual rainfall was supplemented with irrigation. All farms were located between Greendale and Dunsandel, in Canterbury, New Zealand. On most farms herd size was 500-800 cows (one "low-fert" farm had 300 and one "high-fert" farm had 1500). All used twice-a-day milking at the time of sampling, most farms used a daily acid wash and two alkali washes per week (one high-fert farm used only one alkali wash per week). Insufficient data were available on wash-down water use to determine whether dilution differed between farms. Three farms applied low total rates of N fertiliser (30-43 kgN/ha/year, as both organic and synthetic forms), and four applied moderate to high rates (100-350 kgN/ha/year, primarily as urea). These two categories were referred to as low-fert and high-fert farms, where "fert" refers solely to nitrogen. Supplementary feed practices were comparable, with cows receiving a grain or PKE supplementary feed on most properties, although two low-fert and one high-fert property also fed seaweed and/or fish hydrolysate as an animal nutritional supplement.

Eleven effluent samples were collected on two dates in January 2017. Four farms (two low-fert, two high-fert) were sampled on 16 January. All seven farms were sampled on 30 January. Samples were



**Figure 1** Total ammoniacal nitrogen (TAN), total organic nitrogen (TON) and total nitrogen (TN) in effluent (High-fert, n=6; Low-fert, n=5). Boxes represent upper and lower quartiles and the central line is the median; whiskers represent data range and an outlier is marked with a dot.

collected in late morning (10 am-1 pm) on both dates, chilled immediately, and couriered to the laboratory that afternoon. Data from both sampling events were combined in this paper.

The effluent systems varied between farms. On each farm effluent would first pass through a stone trap, then a stirred saucer or pond. From this point treatment varied, effluent was either applied directly to paddocks or subjected to further treatment such as long-term storage or solids separation. For the greatest level of comparability between farms, all samples were collected from the saucer, which in most cases represented the effluent spread onto pasture.

Farmers were asked to stir the effluent before sampling. However, this did not always occur and unstirred saucers invariably had a crust. Samples were collected from either the total mixed slurry if stirred or from the liquid below the crust if unstirred.

Each effluent sample was analysed by Hill Laboratories for total nitrogen, phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg), sodium (Na), and sulphur (S). Filtered samples were analysed for nitrate/nitrite-N and ammoniacal-N. The organic N content was determined by subtracting nitrate/nitrite-N and ammoniacal-N from total N. Effluent pH was measured using an electronic pH meter. Results are presented on a wet volume basis, and were statistically analysed using t-tests in the software package 'R' (R Core Team 2016).

On each sampling date, people who had not collected the samples, were asked to order them from most to least pungent in a 'blind' odour test. Odour may be caused by

hydrogen sulphide as well as ammonia, so is not directly relevant to nitrogen alone, and practical issues were introduced by having two separate sampling dates. As a result, odour observations were informal and intended simply to determine whether the anecdotal accounts of farm staff regarding odour differences were noticeable in a side-by-side comparison.

**Results**

The forms of N present in the effluent are presented in Figure 1. Nitrate and nitrite concentrations were below the detection limit of the analysis for all samples. Both low-fert (LF) and high-fert (HF) farms had comparable mean total N (TN) concentrations (HF 368.0, LF 353.3 g/m<sup>3</sup>, P=0.87). However, high-fert farms had a higher mean total ammoniacal-N (TAN) content (LF 81.8, HF 148.5 g/m<sup>3</sup>, P=0.06), and by subtraction a lower mean total organic N (TON) content (LF 286.2, HF 204.8 g/m<sup>3</sup>, P=0.23), although only the TAN difference approached statistical significance.

The percentages of N present as ammoniacal-N (NH<sub>4</sub>) and organic N are presented in Figure 2. On high-fert farms 41% of the N in the effluent was present as ammoniacal-N, while on low-fert farms only 25% of the N was ammoniacal (the balance being organic), with this difference being highly significant (P<0.01).

The pH of the sampled effluent is presented in Figure 3. All effluent samples from low-fert farms had a pH<7, all samples from high-fert farms had a pH>7.2 (P<0.001). The pH differences did not appear to be due to differences in the wash cycle, as the wash cycles used on all farms were comparable.

The N:P ratio is used to understand the factors limiting microbial growth in aquatic ecosystems and wastewater treatment systems (Davies 2005). The N:P ratio in effluent from the low-fert farms averaged 7:1, while the high-fert farm effluent averaged 10:1 (ranging up to 15:1) (Figure 3), although this difference was not statistically significant (P=0.14).

The generally higher N:P ratio on the low-fert properties was driven by a higher total P content (LF 51.6, HF 36.0 g/m<sup>3</sup>, P=0.056) in the effluent from these properties (Figure 4). The low-fert farm effluent was also generally higher in Ca (LF 270, HF 105 g/m<sup>3</sup>, P<0.01), and of comparable K, Na and Mg (P>0.2) to the effluent from high-fert properties.

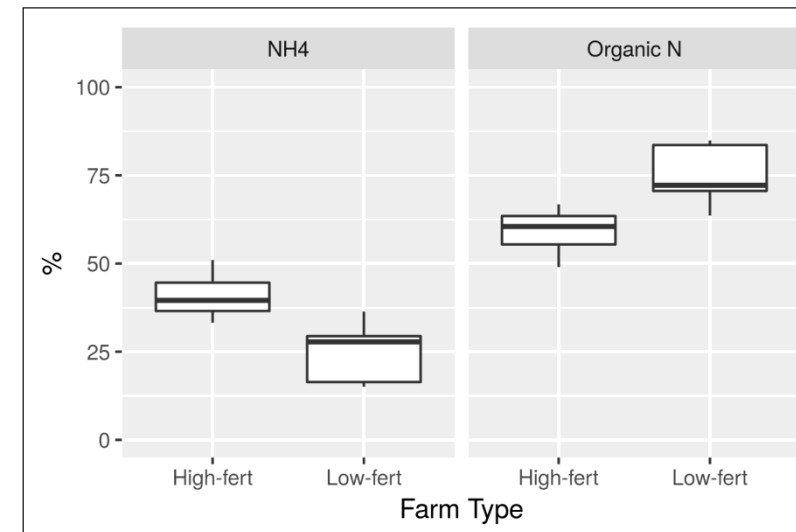
The relationship between the N:P ratio and the percentage of N present as ammoniacal-N is presented in Figure 5. As the N:P ratio increased, the ammoniacal-N content also increased, but the relationship was not strong (P=0.08, R<sup>2</sup>=0.21).

On both sampling dates, observers considered the effluent from high-fert farms to have a greater odour than the effluent from low-fert farms. Although some ponds had not been stirred before sampling, there was no effect of stirring on total N (P=0.8) or on any other factor.

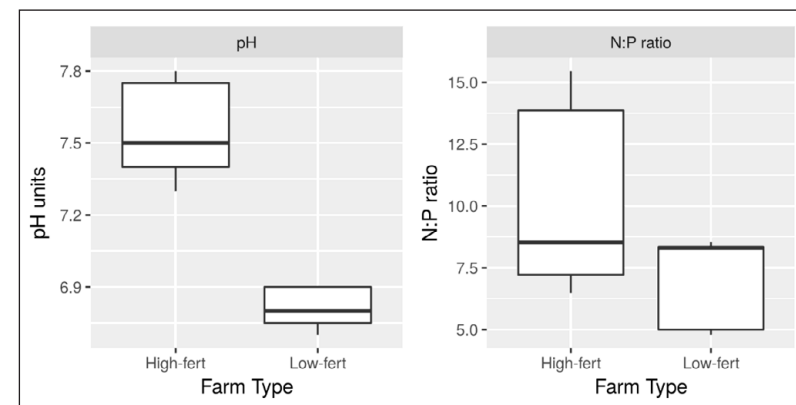
**Discussion**

There was no difference in total N contents. Rather, effluent from high-fert farms had a higher ammoniacal-N content and a higher pH than effluent from low-fert farms.

The higher the pH, the greater the proportion of ammoniacal-N that exists as dissolved ammonia (NH<sub>3</sub>) rather than ammonium (NH<sub>4</sub><sup>+</sup>). High ammoniacal-N and high pH in effluent from high-fert farms could result in greater ammonia emissions (Dexter *et al.* 2014).



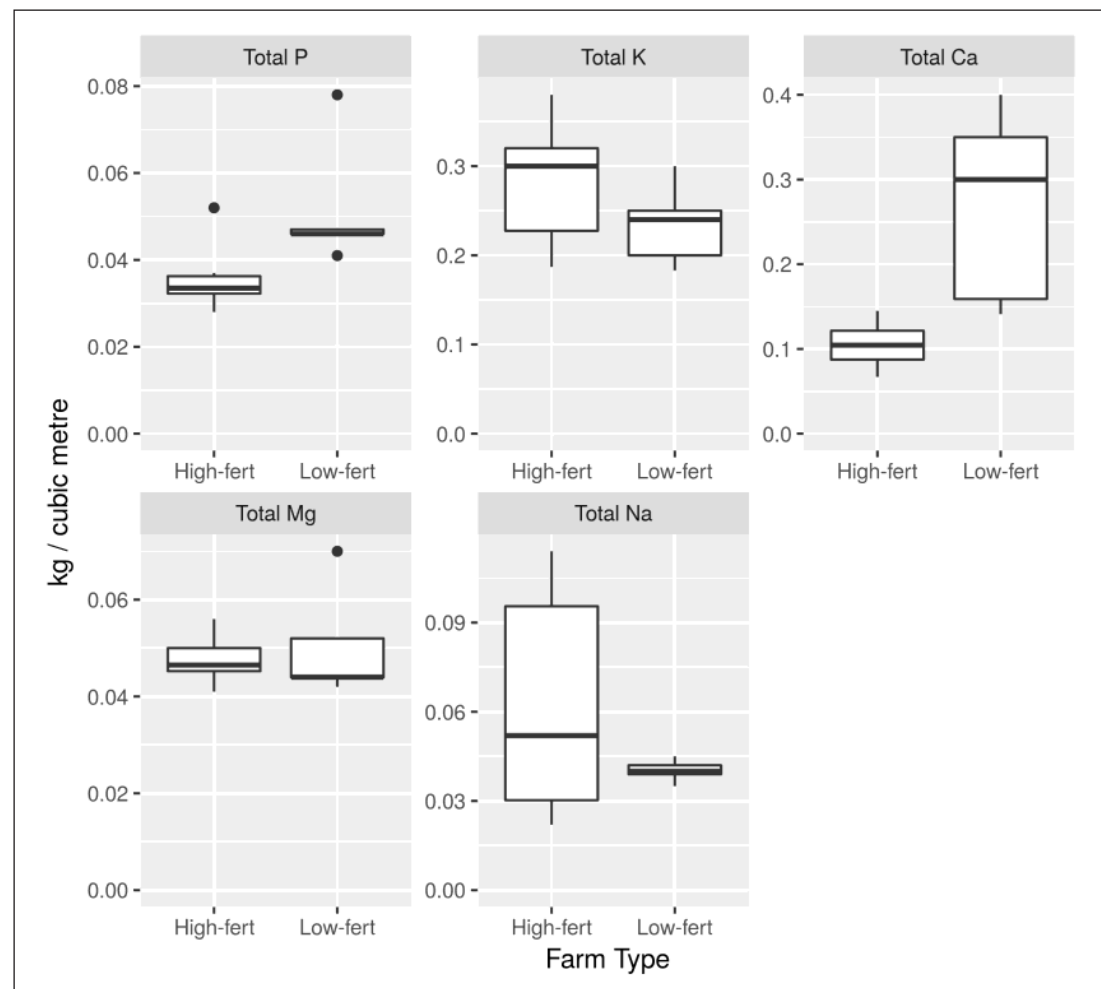
**Figure 2** Percentage of nitrogen (N) present in effluent as ammoniacal-N (NH<sub>4</sub>) and organic N (High-fert, n=6; Low-fert, n=5). Boxes represent upper and lower quartiles and the central line is the median; whiskers represent data range.



**Figure 3** Effluent pH and N:P ratio of effluent (High-fert n=6, Low-fert n=5). Boxes represent upper and lower quartiles and the central line is the median; whiskers represent data range.

This may explain the odour differences observed in the milking sheds by farm staff, and the odour difference between the samples.

Ammoniacal-N is soluble and immediately available for plant uptake. As a result, it is reasonable to assume that the higher the ammoniacal-N content of the effluent, the greater the expected initial pasture response (the author is not aware of any published direct measurements of the effect of effluent ammoniacal-N content on pasture response, this is simply a logical deduction). The different breakdown of ammoniacal-N and organic-N in effluent between the properties is therefore the most likely explanation for the reported differences in visible pasture growth response to effluent application.



**Figure 4** Total phosphorus (P), potassium (K), calcium (Ca), magnesium (Mg) and sodium (Na) in effluent. (High-fert, n=6; Low-fert, n=5). Boxes represent upper and lower quartiles and the central line is the median; whiskers represent data range and outliers are marked with dots.

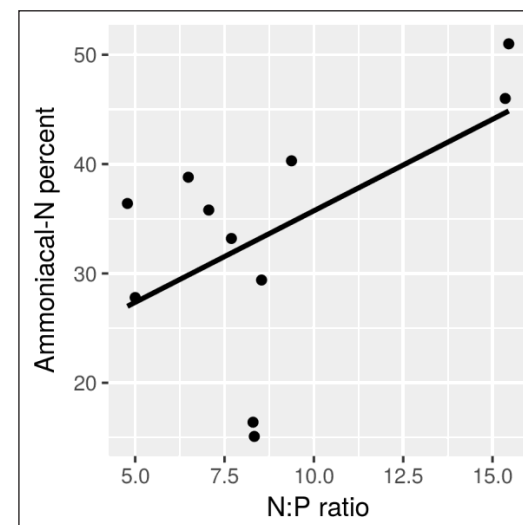
**Reasons for differences**

The organic N fraction contains urea, undigested herbage and other organic compounds from dung and microbial biomass. The observed difference in the ammoniacal-N/organic-N balance indicates that more N must exist in at least one of these fractions on low-fert properties.

A simplified description of N transformations in an effluent pond follows: urea is initially hydrolysed to ammonium (NH<sub>4</sub><sup>+</sup>), through the action of the urease enzyme. This is a rapid process, and raises the pH. Further aerobic microbial activity is a net acid producing process (HACH 2016). This is due to the nitrification of ammonium (NH<sub>4</sub><sup>+</sup>) to nitrate (NO<sub>3</sub><sup>-</sup>), the conversion of ammonium to amino acids and the production of CO<sub>2</sub> through respiration. Microbial activity, therefore, reduces the pH over time. Differences in pH between

the two classes of farm could be due to differences in urea hydrolysis (the process initially raising the pH) or microbial activity (the process lowering the pH again over time).

The quantity of urea available to be hydrolysed will alter the degree to which the pH is raised via urea hydrolysis. The partitioning of N between urine and dung can differ due to pasture composition or supplementary feed, resulting in a higher proportion of consumed N remaining in the digestive tract of the grazing animal, and leaving as complex organic compounds in dung, rather than as urea in urine (Carulla *et al.* 2005). If the low-fert properties have a higher proportion of N in dung and a lower proportion of N in urine, this could explain both the differences in ammoniacal-N/organic-N fractions in effluent between the two classes of property, and the differences



**Figure 5** Percentage of nitrogen as Ammoniacal-N, as a function of effluent N:P ratio (High-fert, n=6; Low-fert, n=5).

in effluent pH. Mackle *et al.* (1996) found minimal difference in dung and urine N partitioning from cows fed on high- or low-N fertilised pasture, however, the pastures in that study received the majority of fertiliser in June and July, so the results do not necessarily reflect the effect of N fertiliser under present-day practice, where N fertiliser is not applied during winter months.

The rate of urea hydrolysis to ammoniacal-N can slow in acidic conditions, however, Dexter *et al.* (2014) found no measurable change until pH was reduced to 2.5, well below the pH of the low-fert effluent. Any alteration in urea hydrolysis is most likely due to the quantity of urea available, rather than factors affecting the rate of urea breakdown.

In wastewater treatment operations, a reduction in pH generally indicates microbial activity driving aerobic oxidation (C. Salkeld pers. comm., Rolleston Wastewater Treatment Plant). Aeration may also alter pH, however, pond stirring had no significant effect on the results. The wash cycles did not differ in a way that could explain the pH differences. As a result, the lower pH values on the low-fert properties could also be due to a greater quantity of microbial activity having taken place in the effluent pond before the samples were collected.

Effluent from low-fert farms had N:P ratios near the optimal ratio for wastewater treatment of 5:1 (Davies 2005), indicating there was sufficient P to allow the majority of N to be incorporated into microbial biomass. High-fert effluent N:P ratio was double the optimal level, indicating microbial activity would be P-limited, with microbes only able to convert a smaller fraction of the ammoniacal-N into organic forms. Increased

N:P ratios were associated with higher ammoniacal-N (Figure 5). However, poor correlation at low N:P ratios suggests this would not explain all the differences observed.

Effluent from low-fert farms had higher P and Ca (Figure 4). Microbes require many nutrients to function, and nutrients other than P could limit microbial activity on some properties. Microbes also require a supply of energy and carbon, for example, readily digestible carbohydrates. There could also be differences in the forms of organic material present in the effluent that may alter microbial activity.

The pH and N form differences observed are therefore consistent with both differences in N partitioning between urine and dung entering the effluent pond, and differences in subsequent microbial activity (as evidenced by the N:P ratio differences). The differences are most likely due to a combination of both mechanisms. As the partitioning of N between urine and dung was not measured, there were insufficient data to determine the relative contribution of each mechanism.

Effluent characteristics may be affected by other factors as well as pasture. N fertiliser practices are not changed in isolation; farmers reducing N fertiliser may implement a number of complementary changes including different fertilisers, animal supplements, grazing management, or other practices, while pastures and soils may also alter over time under different management. For instance, one difference worth noting is the inclusion of a seaweed and/or fish hydrolysate animal supplement on two low- and one high-fert farm, which was applied to PKE before feeding to cattle. This did not appear to have a strong effect on N or pH, results aligning with fertiliser practices rather than supplementation (data not presented), however, as the selection of farms was not designed to accurately compare supplementary feed practices this cannot be discounted. All between-farm comparisons are always confounded by both known and unknown factors; this study was a farmer-funded observational study not a controlled trial.

**On-farm implications**

This study demonstrated clear and significant differences in the forms of N present in effluent on farms using low or high levels of nitrogen fertiliser. These differences are not presently accounted for when considering the environmental implications of fertiliser usage.

Organic N from effluent results in substantially lower losses of N to water than ammoniacal-N applied at the same rate (Di *et al.* 1998). Effluent containing a higher proportion of N in organic forms may, therefore, cause lower environmental losses per unit of N applied than effluent containing more ammoniacal-N per unit of N

applied (or as a % of the applied N), depending on the land-application method used.

A slower rate of fertiliser N release generally increases N use efficiency and pasture yields (Zaman *et al.* 2009). Organic N is a slow-release form of N, so a higher organic N content can be expected to increase long-term pasture production.

Reductions in nitrogen fertiliser application rates may therefore reduce N loss from land-applied dairy shed effluent, and may increase pasture production on paddocks receiving effluent.

Further research is needed to confirm these effects (either more broadly over a larger number of farms or in a controlled study), to better understand the mechanisms driving these observations, and to investigate the wider farm system implications of these findings.

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## Post-sampling procedure affects the measured nutritive value of perennial ryegrass (*Lolium perenne*)

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### Abstract

Estimates of pasture nutritive value are affected by how samples are stored and dried before laboratory analyses. To determine which post-sampling procedures best preserved nutritive value, a range of procedures were compared for one diploid and one tetraploid perennial ryegrass cultivar in June 2016 and April 2017. Treatments included different combinations of transportation from the field to the laboratory either in liquid nitrogen or in a chilly-bin with ice-packs, storage at room temperature (19°C), a chiller (3°C), standard freezer (-18°C) or a -80°C freezer, and drying either in a freeze-drier or an oven at 65°C for 48 hours. Samples were sent to a commercial laboratory for near-infrared reflectance spectroscopy analysis.

Values were higher for one or more of metabolisable energy, soluble sugars, non-structural carbohydrate, and organic matter digestibility, and lower for neutral detergent fibre and ash, when comparing transportation in liquid nitrogen with a chilly-bin, freezer storage (-80°C and standard freezer) with chiller and room temperature storage, and freeze-drying with oven-drying (P<0.01). Freeze-drying was the most critical factor in preserving the nutritive value.

**Keywords:** freeze-drying, oven-drying, liquid nitrogen, pasture sample storage, nutritive value preservation

### Introduction

Perennial ryegrass (*Lolium perenne*) is the single most important plant species to the New Zealand economy, generating more than \$14 billion/annum (<http://mpi.govt.nz/document-vault/14527>). The nutritive value of perennial ryegrass-based pastures is one of the key drivers of livestock performance (Feedsmart 2012); accurate information on pasture nutritive value is critical to predict livestock performance and farm profitability.

Assessment of nutritive value for perennial ryegrass-based pastures requires the harvested pasture sample to be transported from the paddock to the laboratory. Usually, the pasture sample is then dried and ground, before being sent to a commercial laboratory, for either near-infrared reflectance spectroscopy (NIRS) or wet chemistry. These results are presented as a nutritive value profile, typically comprising metabolisable energy (ME), soluble sugars (SolSug), crude protein

(CP), acid detergent fibre (ADF), neutral detergent fibre (NDF), non-structural carbohydrate (NSC), organic matter digestibility (OMD), crude fat (CFat), nitrogen (N) and ash.

NIRS is an accurate and cost-effective way to analyse pasture nutritive values, especially when larger sample sets are being analysed (Corson *et al.* 1999). During NIRS, a sample is assessed in terms of its absorption properties in the near-infrared electromagnetic region, which depends on the chemical composition of the sample (Corson *et al.* 1999; Deaville & Flinn 2000; Alomar *et al.* 2003).

Changes in the chemical composition of the sample between harvest and laboratory analysis may result in an inaccurate estimate of nutritive value. A post-sampling procedure (i.e. transportation, storage and drying before laboratory analyses) may be simple, where a harvested pasture sample is transported from the paddock to the laboratory in a chilly-bin with ice-packs, placed directly into a 65°C oven for 48 hours and then ground. Alternatively, a post-sampling procedure may be intensive where a cut pasture sample is snap-frozen in the paddock using liquid nitrogen and on return to the laboratory, stored at -80°C until the sample is freeze-dried and ground.

The latter method should give the most accurate estimates of nutritive value as the pasture sample remains frozen from the time it is cut in the field until after freeze-drying. The rapid freezing in liquid nitrogen halts plant respiration instantly, when compared with conventional freezing at -18°C, giving less cell disruption and protein denaturation (Alomar *et al.* 2003). In comparison to oven-drying, freeze-drying also results in less changes to the protein structure (Alomar *et al.* 2003). Presumably, these methods would better preserve key nutritive value attributes such as metabolisable energy and soluble sugars. However, an intensive post-sampling procedure may not always be achievable; there may be budget limitations and other constraints to accessing liquid nitrogen and freeze-driers, especially in remote rural areas.

This study compared post-sampling procedures with different combinations of transportation from the field to laboratory, storage methods in the laboratory and drying methods before commercial analyses. It was envisaged that results would act as a guide for choosing