MASSEY UNIVERSITY
Farmed Landscapes Research Centre

Advanced
Sustainable Nutrient Management

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Table of Contents

Course Handbook

Welcome

We are sure you have made the right choice of study. If anything controls the wealth of this nation of ours it is sustaining the fertility of our soils and the productivity of our farming systems whilst endeavouring to reduce agriculture's deleterious impact on freshwater quality. We feel strongly that we need more people like you to develop advanced skills in the science of how soils and nutrient cycles function, so that our soil resources can be managed correctly to sustain our livelihood. You will have received a grounding in the skills required to sustainably manage nutrients in New Zealand Agriculture in the Intermediate SNM course.

This advanced course develops your problem solving skills.

Course Calendar

Month	Activity (see Course Timeline for details)
One	Study guide mailed out.
One	Work on Assignment A: Dairy Case Study.
Two/Three	Work on Assignment B: Arable Case Study.
Three	Work on Assignment C: Sheep and Beef Case Study.
Four	**Collect** data towards Assignment D: Personal Case Study Nutrient Management Plan. Prepare a presentation of your draft Assignment D report to bring to the contact course.
Four/Five	**Attend** 3-day contact course and **sit** final exam. **Send** Assignment D for marking (within two weeks after the contact course).

Course Aim

To provide students with an advanced knowledge of nutrient cycling pathways in New Zealand's farming systems allowing them to develop solutions for systems that have unacceptable nutrient loss to the wider environment.

Course Prescription

An advanced standard of sustainable nutrient management is developed for common New Zealand pastoral and arable farming systems. A study guide and Overseer software will assist the extramural student to develop nutrient management plans for actual pastoral and arable farming enterprises. The student's aim is to produce sustainable nutrient management plans that meet production goals whilst minimising the negative effects of nutrient losses on the wider environment.

During the short course, workshop discussions and presentations will expose the student to the latest in nutrient management research and explore solutions to nutrient losses from farming systems. The use of Overseer software and case study farm information will be used by the student to develop nutrient management plans that apply the objectives of *The Code of Practice for Nutrient Management*.

The course allows students to develop elective interests in relevant topical issues (e.g. nitrogen leaching mitigations, environmental soil testing, trace elements, nutrient loss limits).

Learning Outcomes

On completion of the course the student will be able to develop nutrient management plans for pastoral and arable farming systems that maximise the efficiency of nutrient use in the production system and minimise the loss of nutrients to the wider environment.

Assessment

To receive a Massey University Certificate of Completion in Advanced Sustainable Nutrient Management a student must present four (4) satisfactory reports (attaining at least a 50% average), attend the 3-day short course and pass (at least 50%) the 2-hour examination at the close of the course.

You should read through this introductory section on administrative details, course information and details about the assignments before you begin to study this course.

Please Note:

The delivery of this course in 2020 has been affected by the Covid-19 pandemic.

Whilst the Course Aim and Learning Objectives remain the same, and procedures for completing assignments remain similar, there is no contact course involved and the material that would have been presented at the contact course is now made available on-line.

Details about the delivery processes will be sent to you in a series of emails.

Assignments

Assignment No.	Form of Assessment	% of Total Mark
A	Dairy Case Study Nutrient Management Plan	10
B	Arable Case Study Nutrient Management Plan	10
C	Sheep and Beef Case Study Nutrient Management Plan	10
D	Personal Case Study Nutrient Management Plan	20
	Final examination	50
	Total	**100**

To pass Advanced Sustainable Nutrient Management students are required to:

1. Submit assignments by the due dates and attain at least a 50% average.
2. Sit the final examination and attain at least 50%.

Marking Assignments

The course co-ordinators will endeavour to mark your first assignment promptly and return it to you in time for you to make any necessary changes to your next assignment.
Grades will be awarded according to the following schedule of marks:

GRADE	MARK	COMMENT
A+	90+	Outstanding
A	85-89.99	
A-	80-84.99	Very good
B+	75-79.99	
B	70-74.99	Competent
B-	65-69.99	
C+	60-64.99	Acceptable
C	55-59.99	
C-	50-54.99	Poor effort
D	40-49.99	Unacceptable
E	below 40	

If you are having any problems with the course you should get in touch with the co-ordinators at an early stage to indicate this and allow remedial steps to be taken.

Study Guide and Course Timeline

Section	Title	Task/Activity	Month
1	**Introduction**	▸ Read Section 1.1. Refer to websites provided and make notes as directed.	One
2	**Research literature awareness**	Read the 6 selected papers (Readings 2.1-2.6) provided in the course study notes. Write brief notes, for your own study, on each of the following topics that correspond to each of the 6 readings: 2.2.1 The influence of rates and frequency of N fertiliser application on nitrate leaching. 2.2.2 The influence of fertiliser form and application on phosphate runoff. 2.2.3 The influence of soil test phosphate values on phosphate run-off. 2.2.4 Effect of duration controlled grazing on nutrient transfer 2.2.5 Effect of duration controlled grazing on nitrogen loss 2.2.6 The influence of nitrification inhibitors on nitrate leaching. 2.2.7 Plantain effects on milk production and urinary N excretion	One
3.	**Nutrient Management Plans**	▸ Read Section 3.1 of the study guide. ▸ Use Overseer software and the information provided in the course study notes and the case study information to prepare **Assignment A: Dairy Case Study Nutrient Management Plan report.** Assignment A and suggested Format for Case Study Nutrient Management Plan, will be emailed to you.	One
3.	**Sustainable Nitrogen Management in Arable Systems**	▸ Read Section 3.2 of the study guide ▸ Use Overseer software and the information provided in the course study notes and the case study information to prepare **Assignment B: Arable Case Study Nutrient Management Plan report.** Assignment B and suggested Format for Case Study Nutrient Management Plan, will be emailed to you.	Two/ Three

3.	**Personal case study farm**	‣ **Prepare Assignment D (draft): Personal Case Study Nutrient Management Plan**. Further instructions will be sent to you by email.	Four / Five
N.B.		**Section 4 and 5 are mailed to you on completion of Assignment B.**	
4.	**Nutrient loss limits**	Read the selected papers provided in the course study notes. Prepare brief notes on the following: 1. The concept of nutrient trading. 2. The information required to implement a nutrient trading programme. 3. The advantages of water pollution control by a nutrient trading programme compared to the previous method of control where each of the polluters was required to comply with mandatory limits on pollutants. ‣ **Prepare Assignment C: Sheep and Beef Case Study Nutrient Management Plan**.	Three
5.	**Sustainable use of trace elements**	5.1 Refer to selected readings from the text Mineral Nutrition of grazing ruminants by N.D. Grace. Make brief notes on the grazing animals' requirements for: a. Copper b. Cobalt c. Selenium d. Iodine **Bring notes to Short course.**	Four / Five
Contact Course		‣ Attend 3-day contact course. Further details will be sent to you by email. ‣ Final examination (2hrs)	Five

Contact the Farmed Landscapes Research Centre Office

If you have any questions about the course, including subject matter, please do not hesitate to contact the course co-ordinator, James Hanly (j.a.hanly@massey.ac.nz).

Teaching Materials

Study Notes and Case Study information

We expect you to refer to your copy of the study notes for both the previous introductory course on Sustainable Nutrient Management in New Zealand Agriculture and this, the advanced course. Useful publications to also refer to include:

Managing Mineral Deficiencies in Grazing Livestock. Occasional Publication No 15(2009).
N D Grace, S Knowles and A Sykes.

Soil Science, (2nd Edition) *Sustainable Production and Environmental Protection*, 1996, R. G. McLaren and K. C. Cameron, Oxford Press.

Requirements for Written Assignments

<u>Assignments that you submit must be your own work</u>. We appreciate that some people will benefit through mentoring by colleagues in their workplace, or through participation in study groups and we encourage this collaboration, however, it is intended that the assignments submitted must be an individual piece of work and that producing these should be a learning experience for each person enrolled on the course. Therefore, for all assignments <u>you must</u> generate your own Overseer reports and the written content should be your own original writing, except where references are provided.

Use of other people's ideas or material must be properly acknowledged with referencing. If no acknowledgement is made it will be assumed that the material is original. Please see Massey University's academic integrity website for information on correct referencing techniques. http://owll.massey.ac.nz/referencing/plagiarism.php

For Assignment D (Personal Case Study) this must be reporting on a different property for each person enrolled on the course.

Before submitting your assignment, **make sure that you have made a backup copy** so that you can produce another copy in case the original is lost in the post. Keep a detailed log of each assignment that you post in case of problems. All assignments are to be submitted as hard copies.

When completed, all assignments are to be **sent by courier** to:
Mr Lance Currie
FLRC, Room 1.32
AgHort Sciences Building
Riddet Road, Massey University
Palmerston North 4410

Policy on Late Submission

All assignments must be submitted by the due date. Late submissions will only be accepted if a prior agreement has been established between the student and the course co-ordinator.

Section

1

1. Introduction

1.1 Developments in the requirement for certified farm nutrient budgets

This course builds on your previous experience and knowledge of on-farm nutrient management and use nutrient budgeting. There are a range of national and international drivers that influence the development of changes in nutrient management in agriculture, examples of which are provided below.

National environmental quality examples

Nationally, implementation of the Resource Management Act (1991) by New Zealand Regional Councils to protect soil and water quality has seen the development the National Policy Statement for Freshwater Management (NPS-FWM) (2011). The revised NPS-FWM 2014 introduced the National Objectives Framework. This means that water bodies are managed as a Freshwater Management Unit (FMU), regional councils need to identify key attributes through consultation and set 'attribute states' with community and that these attribute states must meet or exceed the National Bottom Line. The national bottom line is the minimum acceptable standard (where specified).

The NPS-FWM has had several iterations and in Sep 2017, the Government set a target to make 90% of New Zealand's rivers and lakes swimmable by 2040. To achieve this, they require regional councils to improve water quality and regional councils to report on contributions to achieving regional targets every five years. The Government is currently reviewing the NPS-FWM and are proposing National Environmental Standards for Freshwater (Freshwater NES). A draft work programme called Essential Freshwater: Healthy Water, Action for Healthy Waterways is currently under discussion.

The Sustainable Dairying: Water Accord 2013 was formed in association with the National Policy Statement for Freshwater Management and the Land and Water Forum, which was established to foster collaboration between multiple stakeholders, including regional councils.

The accord has been successful at encouraging 98% of waterways on dairy farms to be fenced from stock, 100% of farms have been assessed for their effluent management and nitrogen benchmark information has been collected for 94% of dairy farms.

1.2 Activities

Read and become familiar with the Sustainable Dairying: Water Accord and the National Policy Statement for Freshwater Management.

http://www.dairynz.co.nz/media/3286407/sustainable-dairying-water-accord-2015.pdf

http://www.mfe.govt.nz/sites/default/files/media/Fresh%20water/nps-freshwater-ameneded-2017_0.pdf

After reading these two web resources, list 3 measures of water quality. Then list one land management change that will have an impact, either positively or, negatively, on all three measures. Also, become familiar with the policies around nutrient management on farms and within catchments from different regional councils. Read the websites of three regional councils and take notes on the differences between their land management rules, directed at maintaining or improving water quality. The regional councils throughout NZ include:

Northland Regional Council
Waikato Regional Council
Hawke's Bay Regional Council
Taranaki Regional Council
Greater Wellington Regional Council
West Coast Regional Council
Tasman District Council
Environment Southland

Auckland Council
Bay of Plenty Regional Council
Gisborne District Council
Horizons Regional Council
Environment Canterbury
Marlborough District Council
Otago Regional Council

Section
2

2. Research literature awareness

2.1 Introduction

The objective of this section is to provide examples of research literature (and their sources) that are relevant to advancing your knowledge in the area of nutrient management. The journals/proceedings in which these papers were published will provide other examples of current research. During the short course selected researchers will also talk to you about progress in their specific areas.

2.2 Activities

Read Readings 2.2.1 - 2.2.7.
(a selection of papers).

Then prepare a set of study notes on:

2.2.1 The influence of rates and frequency of N fertiliser application on nitrate leaching.
2.2.2 The influence of fertiliser form and application on phosphate runoff.
2.2.3 The influence of soil test phosphate values on phosphate run-off.
2.2.4 Effect of duration controlled grazing on nutrient transfer
2.2.5 Effect of duration controlled grazing on nitrogen loss.
2.2.6 The influence of nitrification inhibitors on nitrate leaching.
2.2.7 Plantain effects on milk production and urinary N excretion

Reading 2.2.1

Ledgard, S.F., Crush, J.R. and Penno, J.W. 1998. Environmental impacts of different nitrogen inputs on dairy farms and implications for the Resource Management Act of New Zealand. *Environmental Pollution* 102, Sl: 515-519.

ELSEVIER

Environmental Pollution 102, S1 (1998) 515–519

ENVIRONMENTAL POLLUTION

Environmental impacts of different nitrogen inputs on dairy farms and implications for the Resource Management Act of New Zealand

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Abstract

Nitrogen inputs and losses from white clover/ryegrass pastures grazed by dairy cows were measured over three years in a farmlet experiment near Hamilton, New Zealand. Three farmlets received nominal rates of nitrogen fertilizer of 0, 200 or 400 kg N ha^{-1} $year^{-1}$. Nitrate leaching from the free-draining soil averaged 40, 81 and 152 kg N ha^{-1} $year^{-1}$, respectively. The other main nitrogen losses/removals were in milk and transfer of excreta to lanes and the milking shed. Gaseous nitrogen losses were small but increased up to 5-fold with nitrogen fertilizer application. Groundwater nitrate–N concentrations increased with nitrogen application to an average of 19 mg l^{-1} in the 400 N farmlet. Data are discussed in relation to that for average dairy farms in New Zealand and European countries, and to the environmental implications for the Resource Management Act of New Zealand.

Keywords: Dairy farming; groundwater; nitrate leaching; nitrogen balance; white clover

Introduction

Dairy farming is one of the most intensive systems of pastoral land management. Milk production on dairy farms in the E.U. and U.S.A. has increased steadily since the 1950s due to intensification predominantly through increased use of nitrogen (N) fertilizer and imported supplementary feed (van der Meer and Wedin, 1989). This has led to concern about the environmental impacts from nitrate leaching to groundwater and gaseous N emissions (Strebel et al., 1989; Aulakh et al., 1992; Sutton et al., 1993).

In New Zealand, dairy farming is a major land use and is characterised by cows grazing throughout the year (i.e. no housing of cows at any time of the year) on pastures which depend on N_2 fixation by white clover (*Trifolium repens* L.) as the main N input. However, during the 1990's there has been a trend for intensification which has included increasing use of N fertilizer.

The potential amounts of N which may be lost from dairy farms into the environment can be calculated from the N surplus (i.e. N inputs – N outputs in milk, meat and feed). The N surplus on an average New Zealand dairy farm is comparable to that on Swiss farms where N fertilizer use is constrained, but is much lower than that on some English and Dutch dairy farms (Table 1). However, the N surplus gives no indication of the importance of the various processes of N loss or the direct impacts on groundwater or atmospheric quality.

This paper presents detailed research data on N inputs and losses from New Zealand dairy farmlets varying in rate of N fertilizer application, and the impacts on groundwater quality over a three-year period. It also describes the implications of farming practices to the Resource Management Act of New Zealand.

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Table 1

Farm characteristics and estimates of N inputs and outputs (kg ha^{-1} year^{-1}) for average dairy farms in New Zealand (Ledgard et al., 1998), Luzern, Switzerland (Thomet and Pitt, 1997), south-west England (Jarvis, 1993), and in the Netherlands on sandy soils (Aarts et al., 1992).

	New Zealand (1995/96)	Switzerland (1996)	S W England (1991)	The Netherlands (1983–1988)
Farm size (ha)	82		76	25
Cows per ha[a]	2.4	1.6	1.3	2.3
Milk (litres cow^{-1})	3182	5000	5554	5737
Total milk (litres ha^{-1})	7360	8000	7450	13195
Nitrogen inputs				
Fertilizers	40	27	250	331
N_2 fixation	140	90	10	0
Purchased feed	4	19	52	181
Atmospheric deposition	2	28	25	48
Miscellaneous	0	20	0	8
Total N inputs	186	184	337	568
Nitrogen output in products				
Milk	47	32	39	67
Meat	8	11	28	14
Total N output	55	43	67	81
Nitrogen surplus	131	141	270	487
N outputs/N inputs	30%	23%	20%	14%

[a]Some EU farms also have other cattle (e.g. English farms have 0.9 other livestock units ha^{-1}).

Methods

Farmlets

In June 1993, a long-term farmlet trial commenced at the Dairying Research Corporation's Number 2 dairy, near Hamilton, New Zealand (Penno et al., 1996). The trial included 3 self-contained farmlets (each with 6.47 ha in 16 separate paddocks) which were stocked with Friesian dairy cows at 3.3 cows ha^{-1} and received nominal rates of N fertilizer (urea) at 0, 200 or 400 kg N ha^{-1} year^{-1}. The fertilizer was applied in split applications of 22.5 or 45 kg N ha^{-1} at 1–4 days after each grazing, during all seasons except summer.

The dairy cows were rotationally-grazed on pasture throughout the year, with the rotation length varying between 21 days in mid-spring and 128 days in winter. All cows calved in late-winter and were milked twice-daily for 250–290 days, with the final milking date determined by pasture availability and cow body condition.

The site was flat and contained a permanent pasture of predominantly perennial ryegrass (*Lolium perenne* L.) and white clover. It was regularly fertilized with P, S and K to ensure non-limiting conditions. Prior to commencing the trial, lime was applied to raise the soil pH (1:5 soil:water) to the optimum level of 5.8–6.0. Annual rainfall over the 3 years from mid-1993 to mid-1996 averaged 1270 mm.

Measurements

Intake of pasture by cows was estimated by visual assessment of pasture before and after grazings, with calibration by cutting quadrats. Output of N in milk was determined from twice-weekly measurements of the volume and total N concentration of milk. Transfer of N in excreta from the paddocks to the lanes and the milking shed complex was assessed using visual recordings of the location of excreta deposition.

Measurements of N inputs and losses were confined to 4 replicate paddocks of each farmlet on a free-draining silt loam soil (Umbric Dystrochrept). Details of methodology were described by Ledgard et al. (1998). In brief, N_2 fixation by white clover was determined by ^{15}N isotope dilution using 8 microplots per farmlet and with estimates for whole plant material which included stolons and roots.

Denitrification was measured at approximately two-weekly intervals using an acetylene-inhibition technique (Ryden et al., 1987) involving field-incubation of 8 replicate samples with 12 soil cores (25 mm diameter, 0–75 mm depth) per sample. Volatilization of ammonia was measured after each grazing and N fertilizer application using a micrometeorological mass balance method, with samplers as described by Leuning et al. (1985). Masts containing 5 samplers were located upwind in ungrazed paddocks and 20 m within paddocks which had been grazed or fertilized.

Leaching losses in the grazed paddocks were determined using ceramic cup samplers (30 per farmlet) located at 1 m soil depth. Samples of solution were collected at approximately 2-weekly intervals and analysed for nitrate concentration by flow injection analysis. Water drainage was determined from the volume of water passing through lysimeters containing intact soil cores (0.4 m diameter × 1 m depth, 4 replicates) which received 0 or 400 kg N ha^{-1} $year^{-1}$. Groundwater nitrate concentrations under each farmlet were determined using piezometers (3 $farmlet^{-1}$; 6 m depth) with monthly sampling for nitrate analysis. All N transformations and losses were measured for 3 years after trial commencement.

Results

N inputs and outputs in products

In the 0 N fertilizer farmlet, N_2 fixation by white clover was the dominant N input and averaged 174 kg N ha^{-1} $year^{-1}$ (Table 2). Application of N fertilizer reduced the amount of biologically fixed N by 33% and 77% in the 200 and 400 N farmlets, respectively. ^{15}N measurements revealed that this was due to a decrease in clover growth of up to 70% and to substitution of uptake of added fertilizer N for N_2 fixation (data not presented). Inputs of N from other sources were small.

The main N output in products was in milk protein. However, the increase in milk-N in the N-fertilized farmlets corresponded to only 4–7% of the fertilizer N applied.

N losses, transfers and balances

Nitrogen loss by denitrification averaged 5 kg N ha^{-1} $year^{-1}$ in the 0 N farmlet and increased in the N-fertilized farmlets by the equivalent of 5–6% of the fertilizer N applied. Ammonia volatilization averaged 15 kg N ha^{-1} $year^{-1}$ in the 0 N farmlet and increased by 2 to 5-fold in the N-fertilized farmlets.

Nitrate leaching losses were large and increased by up to 4-fold with N fertilizer application. Leaching was confined to the period of net drainage which was predominantly in June–September. In all 3 years, the amount of drainage (measured by lysimetry) was high at 601–714 and 535–668 mm $year^{-1}$ in the 0 N and 400 N treatments, respectively.

The other major process of N loss from the measurement paddocks was by transfer of N via cow excreta to lanes and the milking shed, which averaged 13% of the total N excreted (87% was excreted in the paddocks). Estimates of the total amount of N returned in cow excreta averaged 460, 590 and 630 kg N ha^{-1} $year^{-1}$ in the 0, 200 and 400 N treatments, respectively. Excreta N in the milking shed effluent was pumped away to treatment ponds and was not recycled to the farmlets.

In all farmlets, the sum of the N outputs, transfers and losses was similar to the sum of the N inputs, bearing in mind the experimental error in estimating the various components.

Nitrate in groundwater

Nitrate–N concentrations in groundwater (1.6–4 m below the soil surface, depending on time of year) showed a significant ($P < 0.05$) increase with N fertilizer application by 16 months after trial commencement (Fig. 1). This effect was most apparent in the 400 N treatment. Nitrate–N concentrations showed a cyclical pattern over time, particularly in the N-fertilized farmlets, with peak values occurring in spring/early-summer.

Table 2

Milk production and estimates of N inputs and outputs (kg N ha^{-1} $year^{-1}$) for dairy farmlets receiving nominal rates of N fertilizer application of 0, 200 or 400 kg N ha^{-1} $year^{-1}$ (designated as 0 N, 200 N and 400 N, respectively). Data are the mean of 3 years.

	0 N	200 N	400 N	SED
Cows ha^{-1}	3.3	3.3	3.3	
Milk (litres cow^{-1})	3953	4735	4858	
Total milk (litres ha^{-1})	12955	15516	15921	
Nitrogen inputs				
Fertilizer	0	215	413	
N_2 fixation	174	117	40	15
Purchased feed	3	4	3	
Atmospheric deposition	2	2	2	
Total N inputs	179	338	458	
Nitrogen output in products				
Milk	75	89	92	
Meat	6	6	6	
Surplus silage[a]	0	15	28	
Total N output	81	110	126	
Nitrogen surplus	97	228	334	
N outputs/N inputs	45%	33%	28%	
Nitrogen losses/removals				
Denitrification	5	17	25	4
Volatilisation of ammonia	15	41	65	6
Leaching	40	81	152	24
Transfer to lanes/sheds	57	78	84	
Total N outputs+losses	198	327	452	
Nitrogen balance	–19	+11	+6	

[a]Surplus to requirements and "sold" from the farmlet.

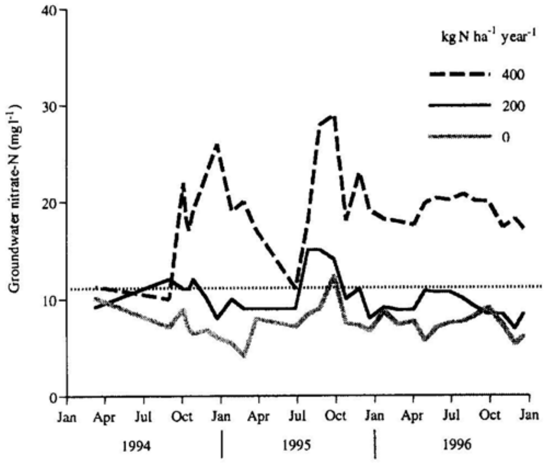

Fig. 1. Nitrate-N concentration in groundwater at 1.6–4 m depth below dairy farmlets receiving nominal rates of N fertilizer application of 0, 200 or 400 kg N ha^{-1} year^{-1}. Data are means for 3 replicate piezometers and have an average SED for each sampling of 4 mg l^{-1}. The dotted horizontal line is the E.U. and N.Z. recommended maximum concentration for potable water.

There was a trend for nitrate values to decline over time. The relatively high initial values were probably due to low drainage during winter 1993 (ca. 200 mm) when the trial started, whereas the winters of 1995 and 1996 had high drainage (600–710 mm) which diluted the leached nitrate. Over the experimental period, the mean nitrate–N concentrations were 7, 10 and 19 mg l^{-1} for the 0, 200 and 400 N farmlets, respectively.

Discussion

N balances and losses

The total N input into the 0 N farmlet was only 40% of that into the 400 N farmlet, whereas milk production from the 0 N farmlet was 83% of that from the 400 N farmlet (Penno et al., 1996). Economic analysis of the different farmlets, based on the relatively low unsubsidised milk payments to farmers in New Zealand, indicated that the 400 N farmlet was unprofitable whereas the 200 N farmlet showed a small increase in profitability over the 0 N farmlet (Penno et al., 1996).

This study highlighted the relatively high N efficiency of intensive dairy farms dependent on N_2 fixation by white clover as the main source of N. The 0 N farmlet had a 45% conversion of total N inputs into products (mainly milk). This was much higher than that for other farm systems with high N inputs in fertilizer or supplementary feed where the conversion was as low as 14% for Dutch dairy farms (Tables 1 and 2).

Nitrogen surpluses within the farmlets increased by about 3.5-fold between the 0 and 400 N farmlets. This represented an increase in N surplus of 237 kg N ha^{-1} year^{-1}. All N outputs and losses increased with increasing N inputs, but the main increases were associated with gaseous N losses and nitrate leaching.

Gaseous N losses were small relative to other N losses, but both denitrification and volatilisation increased by approximately 5-fold between the 0 N and 400 N farmlets. High soil moisture status is a prerequisite for large N losses by denitrification (Aulakh et al., 1992) and this only occurred for short periods during winter in this free-draining soil. Volatilisation of ammonia from grazed pastures in the 0 N farmlet averaged 15 kg N ha^{-1} year^{-1} and was equivalent to 3.8% of the excreta-N deposited on the pasture, which is similar to estimates for Dutch dairy pastures by Bussink (1992). Ammonia N losses increased with N fertilizer application and averaged 12% of the urea-N applied.

The main potential for environmental impact from the farmlets was from nitrate leaching to groundwater, which accounted for approximately 40% of the total N surplus in all farmlets. There was a marked increase in nitrate leaching with increasing rate of N fertilizer which was equivalent to 19 and 27% of the applied N in the 200 and 400 N farmlets, respectively. Scholefield et al. (1993) also measured proportionally larger leaching losses with increasing rates of N application to grassland. This highlights the environmental risk associated with high N fertilizer use on dairy farms. In the 400 N farmlet, the nitrate-N concentration in groundwater increased to an average value of almost twice the commonly-accepted recommended maximum for potable water. Associated ^{15}N studies revealed that there was negligible direct leaching of fertiliser N applied at 200 kg N ha^{-1} year^{-1}, but that approximately 10% of the N applied at 400 kg N ha^{-1} year^{-1} was directly leached from the applications prior to or during winter (Ledgard et al., 1996). Thus, most of the leached nitrate was derived from cow urine patches where N is deposited at rates equivalent to approximately 1000 kg N ha^{-1} year^{-1} (Haynes and Williams 1993). This has important implications for the pasture management practices which can be adopted to reduce nitrate leaching to groundwater.

The challenge to researchers is development of dairy farming systems that cycle N internally very efficiently. Current research is evaluating methods to reduce N losses from dairy farms such as grazing management practices to reduce levels of potentially-leached N in soil from excreta in late-autumn/winter, and feed manipulation to reduce protein intake relative to energy intake and increase N absorption by cows (e.g. Aarts et al., 1992). Feed manipulation can also be used to increase excretion of N in dung relative to urine, which may improve N use efficiency. Other strategies are being assessed to reduce N loss from urine by increasing the spread and reducing the rate of N returned in urine e.g. using forages with diuretic properties.

Implications to resource management

In New Zealand, Parliament passed a Resource Management Act (RMA) in 1991 which promotes sustainable management of natural resources. The tenor of the Act is effect-based rather than being prescriptive. The RMA is enacted through Regional Plans of the Regional Councils throughout New Zealand. Most Regional Councils have set regulations for maximum N loadings in milking shed effluent application to pasture land of 150 or 200 kg N ha^{-1} $year^{-1}$, and two of the 16 Regional Councils have similar maxima for application of fertilizer N. The results from the farmlet trial showed that the effect of application of fertilizer N at 400 kg ha^{-1} $year^{-1}$ was to increase groundwater nitrate to above the recommended maximum for potable water and such high rates of application are actively discouraged. In practice, few dairy farmers in New Zealand use rates in excess of 150 kg N ha^{-1} $year^{-1}$ and over 80% apply less than 100 kg N ha^{-1} $year^{-1}$.

Regulation of N fertilizer inputs is contrary to the effects–based goals of the RMA and ignores the impact of the major N input from N_2 fixation in New Zealand's clover-grass pastures which can vary markedly between sites and years, e.g. 99 and 231 kg N ha^{-1} $year^{-1}$ in successive years in the 0 N farmlet (Ledgard et al., 1998). Nitrogen flows and losses will also be strongly influenced by local edaphic and site history factors. A better approach to minimising nitrate leaching may be to use N balances to indicate the potential for different management options to cause environmental damage. This places the emphasis on the efficiency of N use, and considers all N inputs rather than N fertilizer alone.

The fertilizer manufacturing industry in New Zealand has responded to concerns about ground and surface water quality in agricultural regions by funding the independent development of a Code of Practice for Fertilizer Use. This promotes responsible and effective management of nutrients to minimise their impact on the environment. This Code includes User Guides and Fact Sheets for use by farmers with information on efficient use of N fertilizer. The main mechanism for promoting dairy farming practices with low environmental impact which is advocated by Regional Councils, fertilizer industry and research organisations is farmer education through information and development of farm community groups whereby all parties participate in developing sustainable farming practices (e.g. O'Connor et al., 1997).

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Reading 2.2.2

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Potential phosphorus losses in overland flow from pastoral soils receiving long-term applications of either superphosphate or reactive phosphate rock

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Abstract The forms and concentrations of P in overland flow were measured from intact pastoral soils obtained from the Winchmore long-term P fertiliser trial. Treatments under evaluation were soils that received either 0, 188, 250 or 376 kg superphosphate ha^{-1} yr^{-1}, or 175 kg reactive phosphate rock (RPR) ha^{-1} yr^{-1}. The objective was to determine the magnitude of potential P transfers from soil to water following P fertilisation, and to determine if losses were different following RPR fertilisation compared with superphosphate. Overland flow was induced by the application of artificial rainfall at 15 mm h^{-1}, maintained for 1 h after flow commenced. Concentrations of dissolved reactive P (DRP) and total P (TP) mirrored the long-term application rates, although prior to a fresh application of P, soils with P applied in RPR form lost more P during an event than soils with the same rate of P applied as superphosphate. After a fresh application of RPR and superphosphate treatments, up to 5.4 mg TP litre^{-1} was lost in flow from the 376 kg superphosphate ha^{-1} yr^{-1} treatment, while P in flow from soils fertilised with RPR were commonly c. 0.11 mg litre^{-1}, but still greater than from the unfertilised control soils (0.02 mg litre^{-1}). Regression analysis indicated that DRP concentrations in flow from the fertilised soils were elevated above that lost before fertiliser application for a period of approximately 60 days. These results support earlier studies that demonstrate the greater risks of incidental P losses from soluble P fertilisers such as superphosphate (up to 60 days), and conversely the potential environmental benefits from RPR fertilisation of soils "at-risk" of P loss (e.g., where much overland flow occurs such as in very wet soils and near stream channels). However, if good management practice is followed then the difference in P loss between superphosphate and RPR treated soils should be minimal over a period of a year.

Keywords phosphorus; superphosphate; RPR; pasture; overland flow

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INTRODUCTION

The loss of phosphorus (P) from land to surface waters has the potential to impair water quality via eutrophication. Recently, intensification and expansion of farming systems in New Zealand, especially in the dairy sector, has required the use of P-based fertilisers to rapidly increase soil fertility. At times, P applications are made to soils already at an optimum soil P concentration for plant growth with the aim of building up soil P reserves. While this can be viewed as an insurance policy to keep sufficient P in reserve for later years, it also significantly increases the potential for P loss to overland flow (McDowell et al. 2001).

To abate the potential for P loss, it has been proposed that reactive phosphate rock (RPR) should be used as a substitute for superphosphate (e.g., Nguyen & Quinn 2001). This relies on the premise that RPR is less soluble, and thus less available for loss to flowing water, than superphosphate. After 4–6 years of Sechura RPR applications, pasture performance can be just as good as where superphosphate has been applied on soils with pH < 6.0 and rainfall > 800 mm (Sinclair 1994a,b).

Studies comparing organic farms, where RPR is used, with conventional farms, where superphosphate is used, have commonly noted a much greater build-up of P in soil in the conventional farms (Halberg 1999; Goulding et al. 2000). This has often been taken as an indicator of the more eco-friendly status of organic farms via increased nutrient efficiency and decreased P loss potential (Hansen et al. 2001). However, it has also been noted that the loss of P in subsurface flow from soils with RPR applied can be equal to, and at times greater than, soils with the same amount of P applied as superphosphate (Allinson et al. 2000).

Some research in New Zealand has advocated the use of RPR to minimise the potential P loss to surface waters, compared with the use of superphosphate. For example, in soils that had not previously received P as RPR (virgin soils), Nguyen et al. (1999) found that 100–200 times more dissolved reactive P (DRP) was lost in the short-term from plots supplied with P as superphosphate than similar plots that had received P as RPR. In an effort to determine the effect of long-term P applications (RPR and superphosphate) on P losses, Nyugen & Quinn (2001) and Nguyen et al. (2002) took soils from the Winchmore Irrigation Research Station that had a 20-year history of receiving either superphosphate or RPR. Following a fresh fertiliser application, a similar picture to that in the virgin soils emerged, where those soils with superphosphate applied lost much more P than those receiving RPR. However, measured losses of P, especially in particulate form, were much greater (>10-fold) than normally expected from grassland under natural conditions, even when compared with the large losses of P expected from an application of P shortly followed by a rainfall event (Preedy et al. 2001). This was probably a reflection of the methodology, whereby sieved soils without grass cover were rained upon at a very high rainfall rate (50 mm h^{-1}) to generate overland flow. As a consequence of the high soil loss, the concentrations and loads of P lost and differences between treatments with RPR or superphosphate applied are unclear. Thus, our objective was to determine the loss of P fractions (e.g., dissolved, particulate, and total) in overland flow generated from relatively low intensity rainfall (15 mm h^{-1}) with time from intact grassland soils with a long-term history of P applications either as RPR or superphosphate.

MATERIALS AND METHODS

Soils

The trial site was located at the Winchmore Irrigation Research Station, near Ashburton, New Zealand. The soil is a freely draining Lismore stony silt loam (Orthic Brown Soil [New Zealand soil classification]; Udic Ustochrept [United States Department of Agriculture soil classification]), typical of the Canterbury plains. Mean annual precipitation is approximately 745 mm. Irrigation is required typically 5–10 times during summer months to maintain pasture growth. This is supplied in flood-form using a border-strip system whenever soil moisture content (w/w) drops below 15% (0–100 mm depth).

Intact soil turfs (1050 mm long by 200 mm wide) were taken using a turf cutter with a sampling depth of 100 mm. Four replicate turf layers were extracted from each of four replicates of five treatments of varying P fertilisation histories: (1) Control: no P applied since 1952; (2) 188SP: P applied annually as superphosphate at 188 kg ha^{-1} (18–19 kg P ha^{-1}) since 1952; (3) 250SP: P applied annually as superphosphate at 250 kg ha^{-1} (23 kg P ha^{-1}) since 1981. Prior to this, 376 or 564 kg superphosphate ha^{-1} was applied annually from 1952–57, followed by no P from 1958–79 and an application of 850 kg superphosphate ha^{-1} in 1980; (4) 175RPR: P applied annually since 1981 at 175 kg Sechura RPR equivalent to 23 kg P ha^{-1}. Prior to this, 376 or 564 kg superphosphate ha^{-1} was applied annually from 1952–57, followed by no P from 1958–79 and an application of 750 kg Sechura RPR ha^{-1} in 1980; and (5) 376SP: P applied annually as superphosphate at 376 kg ha^{-1} (36–38 kg P ha^{-1}) since 1952.

In 1972, 4.4 Mg ha^{-1} of lime was applied to all treatments to redress the observed pH decline (Rickard & McBride 1987). Each plot consisted of a fenced-off border strip of approximately 0.1 ha and was mob-grazed by sheep except in winter months.

Overland flow

Each soil was placed in a plywood overland flow box (1050 mm long, 200 mm wide and 125 mm deep, with six 2-mm holes drilled in the bottom for drainage). Soils were stored outside at all times, open to the elements. After 1 week, soils were rained upon and overland flow sampled. Three days later a fresh application of 0, 188, 250, and 376 kg superphosphate ha^{-1} and 175 kg Sechura RPR ha^{-1} were made to the appropriate boxes. Soils were then rained upon after 7, 34, 112, 149, and 184 days and overland flow collected on each date.

Overland flow was generated by applying artificial rainfall (tap water containing P less than detection limit of 0.005 mg P $litre^{-1}$) at 15 mm h^{-1} to each boxed soil, inclined at 5% slope. The rainfall simulator used one TeeJet 1/4HH-SS30WSQ nozzle (Spraying Systems Co., Wheaton, IL) approximately 2.5 m above the soil surface to allow raindrops to gain terminal velocity (Sharpley et al. 1999). The nozzle, plumbing, in-line filter and pressure gauge were fitted onto a $3 \times 3 \times 3$ m aluminium frame with tarpaulins on each side to provide a wind screen. The simulated rainfall had drop size, velocity, and impact energies approximating natural rainfall (Shelton et al. 1985). The 15 mm h^{-1} rainfall intensity has a return frequency of 1–3 years. Samples of overland flow were taken for 1 h after flow had started.

The rainfall simulation and soil boxes used in this study were designed to study interactions between soil and overland flow and mechanisms controlling the release of soil P to overland flow (Sharpley 1985). Although these boxes accurately represent most field processes, and as such mimic processes involved in saturation-excess overland flow, they are not designed to quantify P losses on the field scale *per se* (Sharpley et al. 1982).

Water and soil analyses

Soil analyses were determined on air-dried, crushed, and sieved samples taken from the 0–75 mm depth prior to fertilisation. These samples were analysed for bicarbonate-extractable P (Olsen P; Olsen et al. 1954) and total P following digestion of 250 mg ground (<1 mm) air-dry soil with 5 ml modified aqua regia (4:1 v/v $HCl:HNO_3$). An estimate of P in overland flow was also made by shaking 100 mg of soil in 30 ml of deionised water for 45 min, filtering (<0.45 μm) and determining DRP. This is defined as H_2O-P.

Overland flow samples were analysed for DRP within 24 h and total dissolved reactive P (TDP) after Kjeldahl digestion (Taylor 2000) within 48 h. An unfiltered sample was also digested and total P (TP) measured within 7 days. Fractions were defined as dissolved unreactive (largely organic) P (DOP) (TDP-DRP) and particulate P (PP) (TP-TDP). Suspended sediment was determined by weighing the residue left after filtration through a GF/A glass fibre filter paper.

Statistical analyses

An initial exploration of the data indicated that a power function represented a good fit of the data, while having a minimal number of parameters ($y = \alpha t^{-\beta}$: where y is concentration, t is time α, and β are functions related to the initial value of y and the decrease in y with t, respectively. Statistical comparisons were performed using Genstat v. 6.0 (Genstat Committee 2002) or SPSS v. 6.0 (SPSS Inc., 1999).

RESULTS AND DISCUSSION

Soil P status

Soil chemical parameters are given in Table 1. The soil total P concentration increased with the historical P application rate. More total P was present in the RPR-treated soils (175RPR) than in the soils receiving an equivalent annual application of P as superphosphate (250SP). However, the mean Olsen P concentration of the 250SP plots was significantly greater than the 175RPR plots. This reflects the insoluble nature of RPR, both in soil and in the bicarbonate extract (Perrott et al. 1992; Fleming et

Table 1 Soil chemical characteristics for each P application rate. SED is the standard error of the difference between treatments. Application rate refers to the annual application rate (kg ha^{-1}) of superphosphate or RPR (175). SP, superphosphate; RPR, reactive phosphate rock.

Application rate	pH	Organic C (g kg^{-1})	Total N (g kg^{-1})	H_2O-P (mg $litre^{-1}$)	Olsen P (mg kg^{-1})	Total P (mg kg^{-1})
0	5.73	41.1	3.45	0.016	5.8	785
188SP	5.70	41.7	3.83	0.059	18.5	1004
250SP	5.73	40.0	3.68	0.056	20.3	1047
175RPR	5.63	41.9	3.82	0.055	18.8	1121
376SP	5.68	39.6	3.65	0.157	53.0	1274
SED	0.03	1.3	0.01	0.012	2.0	31

al. 1997). The mean concentration of Olsen P was similar in the 188SP and 250SP plots. This is also reflected in values for water soluble P, and may reflect the greater proportion of P in organic and recalcitrant forms in soils with less readily soluble P applied (Metherell et al. 1997).

P in overland flow

Concentrations of P fractions in overland flow from all events were least in the control (no P applied) plots (Table 2). The minimum concentration of DRP lost from this treatment was less than 20% of the next greatest minimum DRP loss from the 175RPR treated plots. Overall, mean DRP concentrations made up 11, 49, 66, 35, and 73% of total P losses for the 0, 188SP, 250SP, 175RPR, and 376SP plots, respectively. This was dominated by the large initial concentration of P lost (maximum P concentrations in Table 2) in soluble form as more fertiliser was applied (except for the 175RPR treated plots). This also resulted in a decreasing proportion of P lost as PP with increasing P applied. This proportion decreased from 71% in the control treatment to 21 and 23% in the 175RPR and 376SP treatments, respectively. Although P is lost from pastures mainly as dissolved P (up to 95%; Sharpley et al. 1994), particulate P can be lost where slopes are steeper or rainfall more intense. For example, Smith (1989) noted that up to 70% of the TP lost from a steep sloping (therefore erosion prone) catchment (15°) in Waikato was in the form of PP. However, the magnitude of PP loss in our study was much less

Table 2 Summary statistics for P fractions (mg litre^{-1}) lost in overland flow from each P treatment for all events. SED is the standard error of the difference between treatments. Application rate refers to the annual application rate (kg ha^{-1}) of superphosphate or RPR. DRP, dissolved reactive P; OP, organic P; TDP, total dissolved P; PP, particulate P; TP, total P.

P fraction	Application rate	Minimum	Maximum	Flow weighted mean
DRP	0	0.011	0.046	0.023
	188SP	0.079	1.866	0.494
	250SP	0.106	2.777	0.743
	175RPR	0.051	0.156	0.106
	376SP	0.219	5.403	1.378
	SED	–	–	0.589
OP	0	0.016	0.059	0.039
	188SP	0.016	0.291	0.110
	250SP	0.017	0.384	0.117
	175RPR	0.024	0.130	0.061
	376SP	0.018	0.088	0.034
	SED	–	–	0.046
TDP	0	0.027	0.100	0.062
	188SP	0.125	2.156	0.632
	250SP	0.123	3.161	0.880
	175RPR	0.110	0.286	0.179
	376SP	0.298	5.491	1.460
	SED	–	–	0.599
PP	0	0.069	0.229	0.151
	188SP	0.113	1.160	0.369
	250SP	0.083	0.411	0.239
	175RPR	0.076	0.219	0.122
	376SP	0.058	1.489	0.427
	SED	–	–	0.185
TP	0	0.169	0.265	0.214
	188SP	0.239	3.316	1.002
	250SP	0.259	3.572	1.119
	175RPR	0.251	0.393	0.302
	376SP	0.410	6.980	1.886
	SED	–	–	0.759

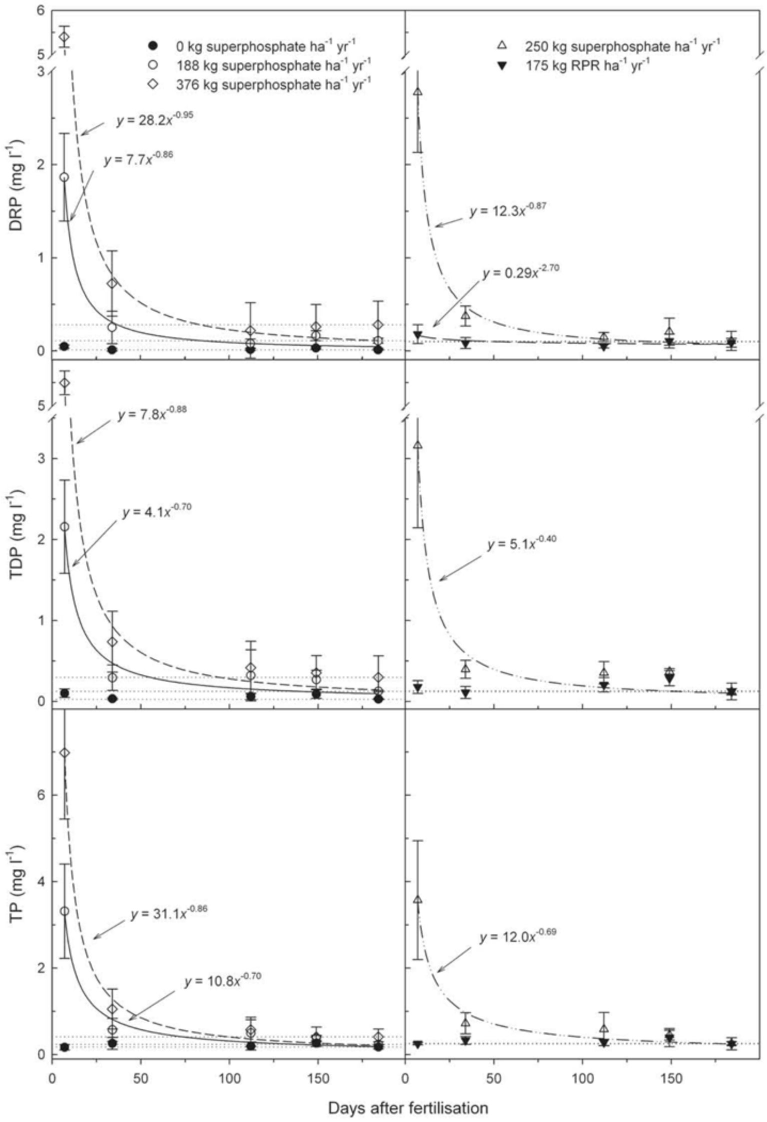

Fig. 1 Concentrations of P fractions ± 95% confidence intervals (DRP, dissolved reactive P and TP, total P; PP, particulate P [TP-TDP, total dissolved P] and OP, organic P [TDP-DRP]) with time since fertilisation. The dotted horizontal lines represent mean P concentrations before fertilisation (note 188SP always lies between the 0 and 376SP treatments). Only significant $P < 0.05$ line fits are shown. SP, superphosphate.

Table 3 Concentrations of P fractions in each treatment before a fresh application of fertiliser was made. SED is the standard error of the difference between treatments. See Tables 1 and 2 for keys to abbreviations.

Treatment	DRP	OP	TDP	PP	TP
0	0.011	0.016	0.027	0.148	0.175
188SP	0.109	0.016	0.125	0.114	0.239
250SP	0.106	0.017	0.123	0.136	0.259
175RPR	0.104	0.015	0.119	0.157	0.276
376SP	0.281	0.018	0.298	0.111	0.410
SED	0.055	0.011	0.058	0.041	0.047

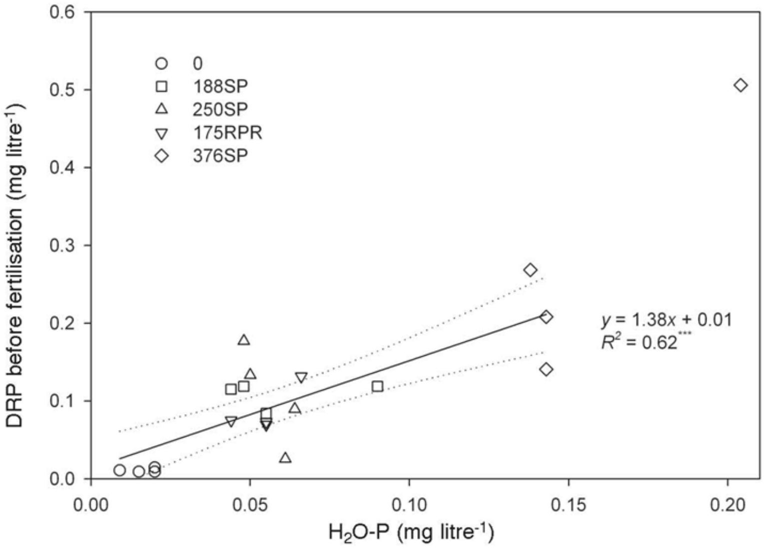

Fig. 2 Relationship between H_2O-P (DRP, dissolved reactive P in laboratory extraction of soil) and DRP in overland flow before fertiliser was applied. Note, one outlying data point with much leverage is not included in the regression. Dotted lines represent the 95% confidence intervals. ***, $P < 0.001$.

(suspended sediment [SS] was below the detection limit of 20 mg litre^{-1}) and agrees well with that reported by Gillingham et al. (1997) who noted a 12% loss of PP as a proportion of TP in overland flow 5 days after fertilisation with superphosphate. In contrast, Nguyen et al. (2002), noted much greater losses of PP (c. 40–50% from superphosphate and 95% from RPR-treated Winchmore soils) from plots inclined at a similar slope but applying higher rainfall than used here. This reflects the lack of pasture cover in the Nguyen et al. (2002) study and the resulting direct loss of particulate fertiliser and soil material in flow.

Following the application of superphosphate fertiliser to soil, P loss in overland flow decreased with time to a minimum near the last event (Fig. 1). In the control treatment the concentrations of PP and TP, but not dissolved P fractions, were lowest at the first event. The rate of decline in P concentrations in the superphosphate-amended treatments is described by the magnitude of the β constant in the fit of the power function to the data ($y = \alpha x^{\beta}$). The data in Fig. 1 shows that the rate of decline was greater (β more negative) as more P was applied. In the 175RPR treatment, P concentrations in overland flow were considerably less than those measured in the superphosphate-amended soils during the first two samplings after fertiliser addition. This is a reflection of the slow kinetics of RPR solubility (Di et al. 1994) and indicates that the potential for P loss from 175RPR applied plots soon after fertilisation is much decreased compared with soils recently amended with superphosphate, and by inference, other soluble P fertilisers.

The relative P concentrations lost by each treatment were similar at the first (before fertilisation, Table 3) and last overland flow events. Indeed, by the third rainfall event (112 days) P losses were not significantly different from those measured before fertilisation (the range of P concentrations before fertilisation is given by the dotted lines in Fig. 1 and data in Table 3). Thus, the potential for P loss from soils treated with superphosphate is only greater during this initial period, and during most of the year the potential for P loss is no greater than in those soils with RPR applied.

Using SS concentration and P enrichment data from the literature (Sharpley & Smith 1989; McDowell & Sharpley 2001), a water extraction procedure was designed to estimate the concentration of P in overland flow. A reasonable relationship was obtained when this water extraction data (H_2O-P) was compared with the DRP concentrations measured in overland flow before the fertilisation event (Fig. 2). This relationship excluded one outlier from the analysis, due to its remote location and greater leverage compared with other data, which increased the slope from c. 1.3 to c. 1.9. This relationship would appear useful for estimating the potential P loss in overland flow.

The mean concentrations of DRP lost in overland flow, even from the most impoverished control soils, were in excess of the limit recommended for the eutrophication of lowland rivers in New Zealand

(0.01 mg litre^{-1}; ANZECC 2000). However, our study was not designed to reflect P losses at large scales, and many other processes such as uptake by sediments and dilution by subsurface flow can occur and alter P concentration in flow before the stream is reached.

Implications for management

The loss of P in overland flow following recent P additions in fertilisers and manure is well known and has been termed "incidental transfers" (e.g., Haygarth & Sharpley 2000). Such incidental transfers are known to occur in pasture and cultivated soils when flow occurs from land that has had a recent application of fertilisers or manure (Preedy et al. 2001). Our results support the conclusions of Nguyen et al. (1999) and Nguyen & Quinn (2001) who demonstrated that the potential for, and magnitude of, incidental P losses from RPR fertiliser was much less than from superphosphate. Data shows that the greater potential for P loss from the 250SP soils only lasted until the third rainfall event, at which time P loss was similar to the 175RPR soils. Using the fit of the data to the power function, it is predicted that DRP loss from the RPR- and superphosphate-treated soils would not be significantly different from one another after approximately 60 days. Indeed, other studies have shown that in the medium to long-term, P loss from RPR- and superphosphate-treated soils are similar if P is allowed sufficient time to react with the soil (Nguyen et al. 1999).

In systems under best practice, superphosphate is commonly applied during the drier times of the year, thus minimising the risk of an incidental P loss. This infers that the direct loss of P from fertiliser applications is low and should not be significantly different from soils treated with RPR. Numerous studies of P loss in New Zealand have shown that direct losses following fertiliser addition are commonly low and only increase if P is applied during wet months or just before a storm event. For example, McColl et al. (1975) showed that during two post-fertiliser floods P loss was 1.4 and 0.55% of P applied. Similarly, Wilcock et al. (1999) found maximum concentrations of DRP and TP in the Toenepi Stream, Waikato, New Zealand coincided with the application of fertiliser in the autumn and spring when soils were wet. In contrast, in a 3-year trial at Edendale, Southland, incidental P losses were minimal and most P loss occurred during winter and spring and not in summer where superphosphate had been applied as per best practice (Smith & Monaghan 2003).

Therefore, if a management strategy were to be adopted to minimise incidental P losses from superphosphate fertiliser, it should advocate application during drier times of the year when such losses are less likely. In most cases, farmers will apply P in the summer, avoiding times of the year at risk of P loss (e.g., autumn and winter). In areas where irrigation (particularly border-dyke irrigation) is used, P should be applied only after irrigation in the summer and timed as far from a future irrigation as possible. To further minimise the potential for P loss, RPR could be considered instead of superphosphate where suitable soils (e.g., pH < 6.0) were already at or near an agronomic optimum (e.g., 20–45 mg Olsen P kg^{-1} depending on soil group; Roberts & Morton 1999) or in areas of "at risk" soils, such as when P fertiliser applications are needed on wet, heavy soils or near-stream areas. Other management practices could also minimise P loss by overland flow to waterways such as the use of riparian strips or buffer strips where no P is applied. Conversely, if good management practice is followed then the difference in P loss from RPR- and superphosphate-treated soils should be little over the period of a year.

CONCLUSIONS

Concentrations of P fractions (DRP, TDP, and TP) in overland flow increased with P application rates from 0 to 36 kg P ha^{-1} yr^{-1}. Considerably greater concentrations of P were measured in overland flow collected from soils fertilised with superphosphate than from those fertilised with RPR. This enhanced loss from the superphosphate-amended soils lasted for up to 60 days after fertiliser application. Therefore, the application of superphosphate (particularly at high rates) should be avoided during periods of likely overland flow, and thus P loss. Alternatively, it would appear that the use of RPR fertiliser can greatly decrease the risk of incidental (short-term) transfers of P from soils to waterways, although actual losses of P to waterways will depend on the field topography, soil, and vegetation conditions specific to each site. As such, if good management practice is used (e.g., not applying superphosphate to soils at risk of P loss in the short-term, such as in wet periods) then the difference in P loss from RPR- and superphosphate-treated soils should be minimal.

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Reading 2.2.3

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Estimating phosphorus loss from New Zealand grassland soils

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Abstract The potential for phosphorus (P) loss from New Zealand grassland soils was assessed using a combination of measured soil chemical properties and concentrations of dissolved reactive P (DRP) determined in drainage and overland flow from simulated rainfall experiments. Soil analyses included Olsen P, calcium chloride (0.01*M* $CaCl_2$) and water extractable DRP, P sorption index (PSI), and % P retention. Results confirmed that DRP concentrations in drainage and overland flow were closely related to $CaCl_2$- and water-extractable DRP in soil, respectively. The preliminary data indicated that the potential concentration of DRP in subsurface and overland flow from pasture soils that have not been recently grazed and at a small scale (e.g., overland flow from 1-m lengths) could be estimated from Olsen P and PSI (or P retention) data according to the following equations:

DRP (subsurface flow) = 1.480 (Olsen P/PSI)
= 0.069 (Olsen P/P retention) + 0.007

DRP (overland flow) = 0.495 (Olsen P/PSI)
= 0.024 (Olsen P/P retention) + 0.024.

Keywords phosphorus; pasture; nutrient loss

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INTRODUCTION

Phosphorus (P) loss from soil to water is implicated in the accelerated eutrophication of surface waters. The potential for loss increases with soil P concentration, but can vary with the transport pathway (Heathwaite & Dils 2000). For example, P loss in overland flow commonly has a greater particulate P component due to erosion than does P loss in subsurface flow (Sharpley et al. 1993). However, the ionic strength of subsurface flow can often be greater as a result of more contact time with soil than occurs with overland flow (except perhaps in preferential or by-pass flow). Both the soil to solution ratio and the ionic strength are known to effect the concentration of P in solution (Evens & Sorensen 1984; Koopmans et al. 2002). Fortunately, laboratory tests have been derived to estimate potential P loss in subsurface flow from the plough layer and tentatively also for overland flow at small scales (McDowell & Sharpley 2001).

There is a need to estimate the potential for a soil to contribute P to overland and subsurface flow from easily measured parameters. Work in New Zealand and overseas has advocated the use of common agronomic (e.g., Olsen or Mehlich-3 extractable P) or environmental (e.g., acid ammonium oxalate extraction for P sorption of Al and Fe oxides) soil tests as methods to judge when an increase in P loss potential may occur (Breeuwsma & Silva 1992; Pote et al. 1996; McDowell & Condron 1999). However, while such methods have management potential for soils where this relationship is known, this potential cannot at present be extrapolated with certainty to all soils. Furthermore, such tests may not be suitable for all soils (e.g., acid ammonium oxalate extractions for P saturation; Hughes et al. 2000), or may not reliably estimate the magnitude of P loss. With this in mind the objectives of this study were to design simple laboratory tests to estimate P in subsurface and overland flow and then to show how readily available soil test P data such as Olsen P and P retention can be used to estimate P in subsurface and overland flow for a range of New Zealand grassland soils of different P status.

Table 1 Selected physiochemical parameters of the 44 grassland soils used in this study.

Soil	Classification	pH	Organic C ($g\ kg^{-1}$)	Olsen P ($mg\ kg^{-1}$)	Total P ($mg\ kg^{-1}$)
Cargill silt loam	Acidic Mafic Brown	5.5	57.8	29	1169
Egmont black silt loam	Typic Orthic Allophanic	5.7	53.0	7	629
Eyre silt loam	Weathered Orthic Recent	5.8	34.3	73	1255
Fork silt loam	Typic Orthic Brown	6.0	25.2	20	690
Haldon stony silt loam	Typic Immature Pallic	5.5	48.5	15	707
Hamilton clay loam	Typic Orthic Granular	5.7	93.6	11	1559
Himatangi sand	Typic Sandy Brown	7.0	29.0	46	862
Hurunui stony loam	Typic Orthic Brown	5.6	78.7	32	904
Kiripaka silt loam	Typic Orthic Allophanic	6.4	102.8	57	2461
Lismore stony silt loam	Pallic Firm Brown	5.6	39.0	18	707
Mahoenui silt loam	Typic Orthic Recent	5.3	54.4	16	560
Mangamahu silt loam	Mottled Orthic Brown	5.3	39.3	10	401
Mangatea clay loam	Pallic Orthic Brown	5.4	85.9	8	682
Mapua sandy loam	Mottled Albic Ultic	5.2	19.5	10	116
Mataura silt loam	Typic Orthic Recent	6.1	30.5	11	585
Maunu silt loam	Typic Orthic Oxidic	5.7	102.3	10	1413
Mihiwaka silt loam	Acidic Mafic Brown	5.8	38.5	30	707
Ngakuru loam	Typic Orthic Allophanic	6.5	84.3	117	1862
Northope silt loam	Typic Orthic Gley	6.7	17.0	10	633
Oamaru clay loam	Weathered Rendzic Melanic	6.8	43.2	92	1291
Okarito peaty silt loam	Silt-mantled Perch-gley Podzol	5.4	130.4	37	597
Oruanui silty sand	Podzolic Orthic Pumice	5.3	88.9	50	1127
Patoka fine sandy loam	Buried-allophanic Orthic Pumice	5.7	93.4	14	1585
Portobello silt loam	Typic Mafic Brown	5.3	46.4	18	658
Poukawa peaty loam	Peaty Orthic Gley	6.0	31.9	24	1218
Pukaki silt loam	Humose Orthic Brown	5.2	49.1	11	663
Pukemutu silt loam	Argillic-mottled Fragic Pallic	6.1	17.5	16	390
Richmond silt loam	Typic Orthic Gley	5.7	36.5	15	813
Riponui clay	Yellow Albic Ultic	5.8	21.6	15	317
Rotoiti sandy loam	Typic Orthic Pumice	5.5	50.9	51	560
Stratford sandy loam	Typic Orthic Allophanic	5.4	67.8	42	2746
Taihape silt loam	Mottled Orthic Brown	6.0	50.5	10	341
Taupo silty sand	Immature Orthic Pumice	5.1	55.4	80	1311
Te Kauwhata clay loam	Typic Orthic Granular	5.7	50.0	19	938
Te Kuiti sandy loam	Typic Orthic Allophanic	5.8	34.7	14	573
Temuka silt loam	Typic Orthic Gley	6.5	39.6	54	1056
Waikiwi silt loam	Typic Firm Brown	5.8	64.2	8	414
Waikoikoi silt loam	Mottled Fragic Pallic	6.2	22.3	26	609
Waiotira clay loam	Typic Acid Brown	5.3	46.5	18	682
Waipawa silt loam	Typic Argillic Pallic	5.5	38.4	18	524
Waitahuna silt loam	Mottled Fragic Pallic	5.8	39.9	88	1236
Warepa silt loam	Mottled Fragic Pallic	5.6	43.0	39	633
Wharekoke silt loam	Perched-gleyed Densipan Ultic	5.4	33.6	18	439
Wingatui silt loam	Weathered Fluvial Recent	5.6	35.5	25	877
Woodlands silt loam	Typic Firm Brown	5.1	58.8	8	487

MATERIALS AND METHODS

Soil extractions

Samples (0–7.5 cm) of 44 soils currently under grassland were collected from different areas of New Zealand (Table 1), air-dried, crushed, and sieved (<2 mm). Each soil was analysed for pH in water (1:2.5 soil to solution ratio), and organic C by LECO® combustion. A range of P analyses were conducted.

1) Olsen P was determined using a soil:solution ratio of 1:20 and quoted on a weight (mg kg^{-1}), not volume (mg $litre^{-1}$) basis, negating the influence of bulk density (Olsen et al. 1954).
2) $CaCl_2$-P (an estimate of P in subsurface flow; McDowell & Sharpley 2001), was measured using a soil to 0.01*M* $CaCl_2$ solution ratio of 1:5 and a shaking time of 30 min before measuring dissolved reactive P (DRP) in the filtrate (<0.45 μm).
3) H_2O-P (an estimate of P in overland flow), was determined using a soil to deionised water ratio of 1:300 and a shaking time of 45 min before measuring DRP.
4) PSI (P sorption index), was determined from the amount of P sorbed, x (mg P kg^{-1}), from an addition of 1.5 g P kg $soil^{-1}$ (as KH_2PO_4 in 0.01*M* $CaCl_2$) after shaking for 24 h at a soil to solution ratio of 1:20. The PSI was then calculated as the quotient $x \log C^{-1}$, where C is the solution concentration (mg $litre^{-1}$) in the filtrate (Whatman #42). This quotient is highly correlated to the P sorption maximum in a wide range of soils (Bache & Williams 1971).
5) Percent P retention (%P remaining after equilibration with a soil P saturating solution, buffered at pH 4.6) was assessed according to the methods outlined by Saunders (1964).
6) Total P was determined by the digestion of 0.15 g of soil (ground < 500 μm) with 5 ml concentrated HNO_3:HCl in a 1:4 mix (Crosland et al. 1995).

All P determinations (except P retention; Saunders 1964) were made using the method of Watanabe & Olsen (1965). All soil extractions were made in duplicate except for the $CaCl_2$-P and H_2O-P extractions which were conducted in triplicate.

Additional analyses

For the calibration of laboratory tests to estimate overland and subsurface flow, data from the literature was supplemented by rainfall simulation studies generating overland flow from 11 intact grassland soils selected to represent a wide range of New Zealand soil types under pasture. The soils were Woodlands (Typic Firm Brown), Waikiwi (Typic Firm Brown), Mataura (Typic Orthic Recent), Northope (Typic Orthic Gley), Pukemutu (Argillic-mottled Fragic Pallic), Waikoikoi (Mottled Fragic Pallic), Waitahuna (Mottled Fragic Pallic), Lismore (Pallic Firm Brown), Rotoiti silt loam (Typic Orthic Pumice), Taupo sandy silt (Immature Orthic Pumice) and Ngakuru loam (Typic Orthic Allophanic) soils (Table 1). All soils were taken in triplicate from field sites around the country to a 5 cm depth using either a turf cutter or spade during winter of 2001. The exception to this general rule was the Waitahuna and Lismore soils which were taken to 7.5 cm depth and the Rotoiti, Taupo, and Ngakuru soils which were taken in summer 2002. Pasture was trimmed to a uniform 5 cm height before soils were placed into boxes 1 m long by 20 cm wide and 7.5 cm deep with six small (2 mm diameter) holes drilled for some drainage. Soil samples for analysis were taken to the full depth of each soil box from the upslope end.

Overland flow was generated by applying artificial rainfall (tap water, P less than detection limit of 0.005 mg P $litre^{-1}$) at 1.5 cm h^{-1} to boxes, inclined at 5% slope and within 1 week, and generally within 3 days, of collection. The rainfall simulator uses one TeeJet 1/4HH-SS30WSQ nozzle (Spraying Systems Co., Wheaton, IL) positioned approximately 250 cm above the soil surface in order for raindrops to gain terminal velocity (Sharpley et al. 1999). The nozzle, plumbing, in-line filter, and pressure gauge were fitted onto a 305 × 305 × 305 cm aluminium frame. The simulated rainfall had drop-size, velocity, and impact energies approximating natural rainfall (Shelton et al. 1985). Samples of overland flow were collected for 1 h after flow had started; a subsample was taken and filtered (<0.45 μm), and P determined in duplicate.

Additional data for DRP loss in subsurface flow from Woodlands silt loam soils were taken from McDowell & Monaghan (2002; Fig. 1A). The rainfall simulation and soil boxes used in this study accurately mimick mechanisms controlling the release of soil P to saturation-excess overland flow in the field (Sharpley 1985, 1995). Recent work has shown the use of small scale plots and rainfall simulation can be used to estimate P losses from dairy pastures (Cornish et al. 2002). However, they are not designed to quantify P losses on the field scale *per se* (Sharpley et al. 1982).

All summary statistics (mean, standard error, minimum, and maximum) and regression analyses were made using SPSS v. 10.0 (SPSS Inc. 1999).

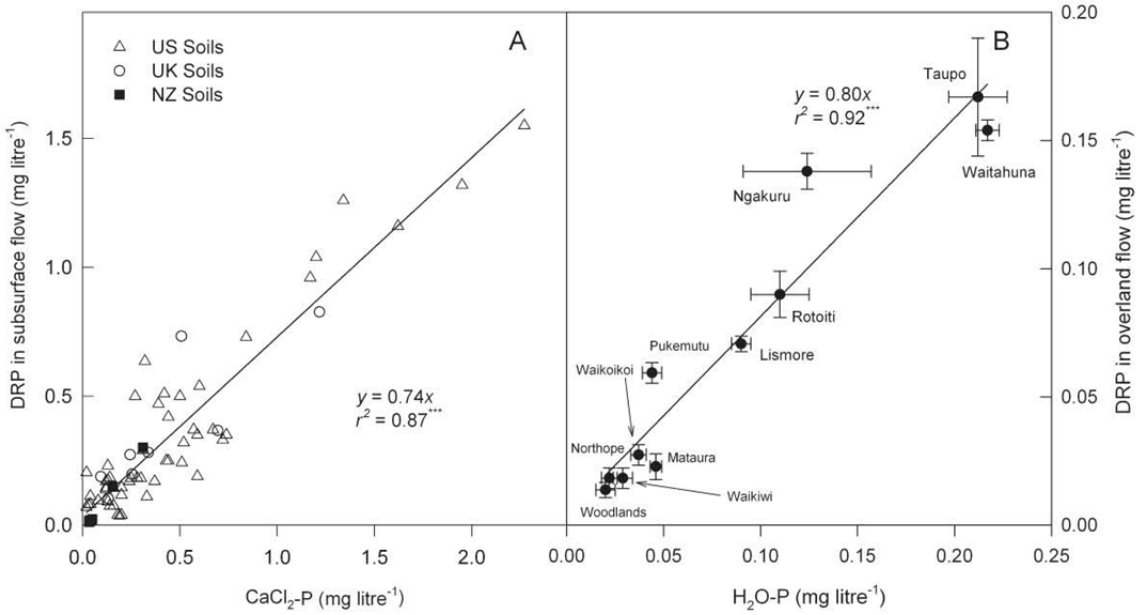

Fig. 1 Relationship between dissolved reactive P (DRP) in subsurface flow and $CaCl_2$-P for the United States (McDowell & Sharpley 2001), United Kingdom (McDowell & Sharpley 2001), and New Zealand (McDowell & Monaghan 2002; McDowell unpubl.) soils (A), and between DRP in overland flow and H_2O-P for selected New Zealand grassland soils (B). Error bars are the standard errors for three replicates.

RESULTS AND DISCUSSION

Laboratory extractions for P loss estimations

Soil chemical data are given in Tables 1 and 2. The soils cover the majority of the soil groups found in New Zealand. Since all of the soils were under grassland when sampled, it is not surprising to see the least variation is evident in pH, while other chemical parameters such as Olsen P, organic C, and % P retention reflect their pedological origin and probably recent fertiliser management (Table 2). The soils selected to evaluate a quick laboratory test to predict P in overland flow represent a wide ranging subset of soils from this group, while data from the literature is used to show the validity of an established quick laboratory test for P in subsurface flow (McDowell & Sharpley 2001; McDowell & Monaghan 2002).

To truly predict P desorption into solution, and thus into subsurface and overland flow, the medium must reflect the cation status as well as the ionic strength of the aqueous phase of the system (Ryden & Syers 1975; Beauchemin et al. 1996). Thus, for subsurface flow, a short-term (30 min) extraction of soil with 0.01*M* $CaCl_2$ at a soil to solution ratio of 1:5 was designed by Schofield (1955) to simulate the correct ionic strength for soil solution in near-neutral pH and calcareous soils. This test has been used recently for the prediction of P behaviour and concentration in subsurface flow in the United Kingdom, United States, and New Zealand (McDowell & Condron 1999; McDowell & Sharpley 2001; Blake et al. 2002; McDowell & Monaghan 2002; McDowell unpubl.). The data for these studies is plotted in Fig. 1A and shows a good relationship between P in subsurface flow and $CaCl_2$-P. However, this data only pertains to P lost from the top 20–30 cm of surface soil. This does not imply that, in general, this estimate corresponds to the quantities of P leaving the soil profile in subsurface flow, unless intercepted by preferential flow pathways and/or tile drains (Heckrath 1997).

For overland flow, preliminary tests designed to estimate P have traditionally used much wider soil to solution ratios and lower ionic strengths to simulate, in general, less soil contact time, compared with subsurface flow. For example, Ryden et al. (1971a,b) used a water extraction at a soil to solution ratio of

1:400 to estimate the contribution of P into stream flow from an eroding urban stream bank. A water extraction has also been used by other workers (e.g., Sissingh 1971; Sharpley et al. 1982; Koopmans et al. 2001). Sharpley et al. (1982) also recognised that in order to estimate P in overland flow, allowance must be made for the selective erosion of P-rich fines in flow, effectively concentrating P in flow, compared with the suspension of whole source soil in flow, i.e., an enrichment ratio.

With these mechanisms in mind, we took rainfall simulation data from Sharpley & Smith (1989), Pote et al. (1999), McDowell & Sharpley (2001), and McDowell et al. (2003) and calculated the mean likely enrichment ratio (degree of P enrichment of sediment in overland flow compared with source soil) for more than 200 soils under pasture. This was calculated as approximately three for a 45-min overland flow event. By combining the enrichment ratio with data for the mean suspended sediment concentration in overland flow from 90 grassland soils from Southland (0.1 g litre^{-1}) under a low rainfall intensity of 1.5 cm h^{-1} (compared with USA studies that commonly use > 5 cm h^{-1}) a soil to water ratio of 1:300 was derived (McDowell et al. 2003). Once filtered, the P measured in this extract was termed H_2O-P.

Using this laboratory extraction procedure, H_2O-P in surface soil was estimated and compared with that generated from a 45-min rainfall simulation of 11 intact grassland soils (each with three replicates) from across New Zealand with a range of P concentrations in topsoil. As with the relationship between $CaCl_2$-P and DRP in subsurface flow, a good relationship was gained between DRP in overland flow and H_2O-P (Table 1; Fig. 1B).

Predicting $CaCl_2$-P and H_2O-P

Several workers have attempted to relate soil P concentration and P saturation to P loss in overland and subsurface flow. For example, in the Netherlands, P in subsurface flow is related to the degree of P saturation of Al and Fe oxides estimated from an acid ammonium oxalate extraction (Breeuswma & Silva 1992). Work in the United States, United Kingdom, and Australasia has related P loss to soil test P concentrations with varying degrees of success (e.g., Mehlich-3 (Pote et al. 1996, 1999); Olsen P (Gillingham et al. 1997; McDowell & Condron 1999)).

Following an analysis of the data for correlations between $CaCl_2$-P or H_2O-P and the various soil chemical parameters tested, Olsen P, PSI, and P retention were found to be significantly correlated ($P \leq 0.05$; r = 0.635, 0.489, 0.512, respectively for $CaCl_2$-P; 0.760, 0.387, and 0.418, respectively for H_2O-P). Several studies have related the equilibrium P concentration at zero net sorption or desorption (EPC_0; deemed likely to represent the behaviour of P in flow) to measures of soil test P and concluded that a measure of P sorption was also necessary to fully predict EPC_0. For example, Sallade & Sims (1997) and Hughes et al. (2000) both incorporated the PSI along with a soil P test (Mehlich-I extractable and Olsen extractable P, respectively) to predict EPC_0. The PSI is known to be closely correlated with P sorption capacity and is a quick and reliable indicator for the potential of a soil to change its ability to retain P following P additions (Indiati & Sharpley 1997). By combining terms in various combinations within a stepwise multiple regression, a plot of the quotient of Olsen P and PSI against CaCl-P or H_2O-P was found to be highly significant

Table 2 Summary statistics for the 44 grassland soils used in the study.

Parameter	Mean	Standard Error	Minimum	Maximum
pH	5.7	0.07	5.1	7.0
Organic C (g kg^{-1})	53.7	3.88	19.5	130.4
Olsen P (mg kg^{-1})	30	3.9	7	117
Total P (mg kg^{-1})	866	78.0	116	2746
$CaCl_2$-P (mg litre^{-1})	0.143	0.032	0.003	0.944
H_2O-P (mg litre^{-1})	0.068	0.010	0.011	0.338
P retention (%)	33	3.1	7	85
PSI	677	65.1	144	2123

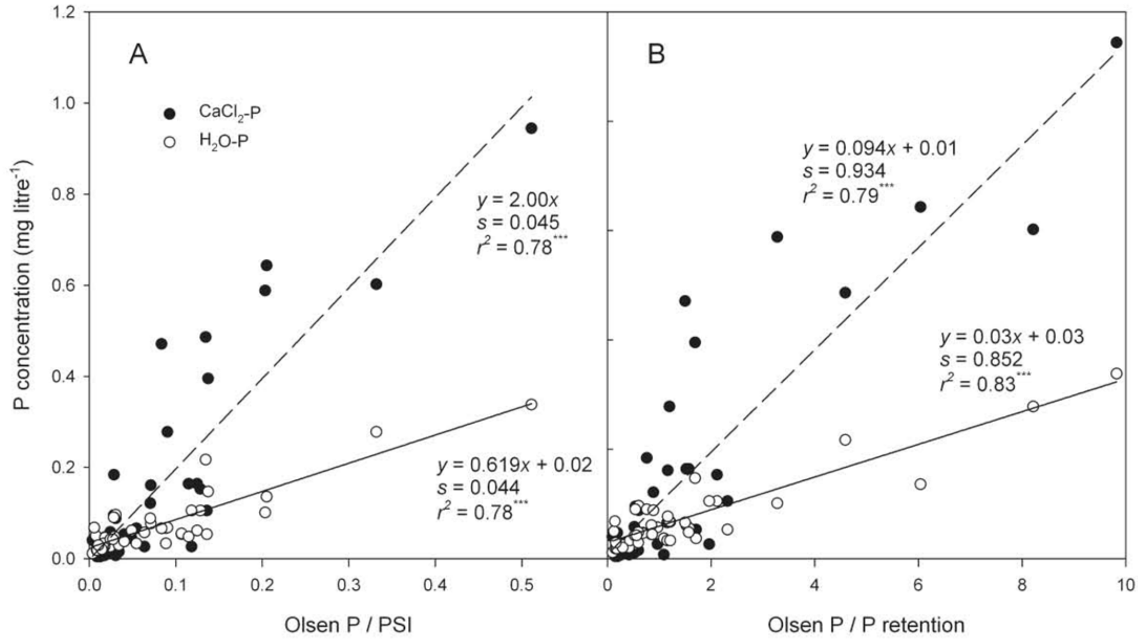

Fig. 2 Linear regressions for the relationships between dissolved reactive P (DRP) concentration in $CaCl_2$-P or H_2O-P and the quotients of Olsen P and PSI (A), and Olsen P and % P retention (B), for 44 New Zealand grassland soils.

($P < 0.001$) with the added benefit of simplicity and no superfluous terms. We therefore propose this as an easy method of predicting $CaCl_2$-P or H_2O-P (Fig. 2A). However, the linear regression shown in Fig. 2A is dependent upon a few data points with much leverage, especially for $CaCl_2$-P data. As such, the relationships are preliminary (note standard errors, *s*).

In New Zealand, the P retention test developed by Saunders (1964) has been widely used in soil classification and agronomic advice. As expected, the PSI and P retention values were closely correlated ($r = 0.905$; $P < 0.001$), and hence the quotient of Olsen P and P retention is also a suitable method for the prediction of $CaCl_2$-P or H_2O-P (Fig. 2B). Although some attempts to use P retention for the prediction of P loss have been made, they either involve complex equations, only relate to one flow pathway, or have thus far been without validation (Hart et al. 2002; Hedley et al. 2002).

The choice between using the Olsen P and PSI or P retention quotient to estimate $CaCl_2$-P or H_2O-P is almost arbitrary; both are simple measures that are either easily adopted or used. However, in New Zealand many soils already have their P retention values determined. The quotient of Olsen P and P retention could thus provide the most useful method of determining the potential concentration of H_2O-P and $CaCl_2$-P. However, caution should be employed when extrapolating this to P in overland and subsurface flow for New Zealand pastoral soils. Recent evidence by Koopmans et al. (2001, 2002) and McDowell & Sharpley (2001) indicates that differing rainfall intensity could affect the concentration of P in flow. We have only demonstrated the relationship and link between H_2O-P and P in overland flow for one rainfall intensity. Although work over the last 30 years has shown that the difference in concentration in soil solution ratios above 1:100 is small, there is still potential for some differences to occur (Ryden et al. 1971a,b). Furthermore, we have demonstrated that the results presented here pertain to P loss at one scale, overland flow boxes 1-m long. Recent work has shown that scale can affect the concentration of P in flow (McDowell & Sharpley 2002). Additional work is required to see by what degree the relationships change under the range of rainfall intensities possible in the New Zealand environment, and different scales in the field. Furthermore, the relationship is only applicable for pasture soils that have not been recently grazed (<10–15 days since grazing), since P from dung may significantly increase P loss. This work should, therefore, be considered as still evolving. In the meantime, by combining equations in Fig. 1 and 2, a preliminary relationship for the estimation of DRP in overland and subsurface flow from Olsen P and PSI/P retention data can be generated:

DRP concentration (overland flow)
= 0.495 (Olsen P/PSI) + 0.016;
= 0.024 (Olsen P/P retention) + 0.024.

DRP concentration (subsurface flow)
= 1.480 (Olsen P/PSI)
= 0.069 (Olsen P/P retention) + 0.007.

CONCLUSIONS

Dilute $CaCl_2$ solution and deionised water extractions can be used for the prediction of P concentrations in subsurface and overland flow, respectively. However, for subsurface flow this method pertains only to P lost from topsoil (30 cm) and has not been validated for lower depths. The laboratory test developed (1:300 soil to solution mixture, shaken for 45 min) appeared to estimate well the DRP in overland flow from 11 soils generated by simulated rainfall from 1-m long boxes. These data were used to derive relationships between DRP in overland and subsurface flow from soils that have not been recently grazed and selected soil chemical parameters (Olsen P, PSI/P retention).

However, caution should be employed when using these relationships as they are likely to change as additional soils become available for validation and they are tested under different rainfall regimes (which would likely alter the soil to solution ratio) and scales. Consequently, further work should be directed at determining the likely influence of these parameters, and these relationships should be viewed as preliminary.

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Reading 2.2.4

Christensen CL, Hedley MJ, Hanly JA, Horne DJ (2018) Duration-controlled grazing of dairy cows. 1: Impacts on pasture growth, cow intakes and nutrient transfer. *New Zealand Journal of Agricultural Research*, 1-25.

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RESEARCH ARTICLE

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Duration-controlled grazing of dairy cows. 1: Impacts on pasture growth, cow intakes and nutrient transfer

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ABSTRACT

Standing cows off pasture after 4 hours of grazing (duration-controlled (*DC*) grazing) was researched over 3 years (2008–2011) to compare the losses of nutrients in drainage and surface runoff water with losses under standard grazing (*SG*). Pasture growth rates, nutrient concentrations and apparent intake by cows were used to model nutrient removals from, and returns to, pasture. Pre- and post-grazing covers and apparent pasture intakes were similar for both treatments. Quantities of N, P and K returned to the *DC* plots in excreta and effluent were 34%, 0% and 45% less than those returned to *SG* plots. This reduction in nutrient returns was associated with a nil, 20% and 9% decrease in annual pasture growth on the *DC* plots in the three respective years. Reductions in annual pasture growth under *DC* grazing may be avoidable if nutrient removals are balanced with returns in effluent and fertiliser.

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RESEARCH ARTICLE

Check for updates

Duration-controlled grazing of dairy cows. 2: nitrogen losses in sub-surface drainage water and surface runoff

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ABSTRACT

The annual quantities of nitrate-N (NO_3^--N) and total N (TN) leached from mole and pipe drained dairy pastures on a Pallic soil, were reduced by an average of 52% and 42% ($p < .05$), respectively, by standing cows off pasture to ruminate and rest after grazing (Duration-controlled grazing, DC), compared to the standard grazing management (SG) of leaving cows at pasture between milkings. For the SG treatment, measured NO_3^--N leached was 13, 8 and 21 kg N ha^{-1} and total N leached was 18, 13 and 26 kg N ha^{-1} in 2009, 2010 and 2011, respectively. Annual surface runoff contained <1 kg NO_3^--N ha^{-1} y^{-1} and <3 kg TN ha^{-1} y^{-1}. The deposition of urine in the autumn period had the greatest influence on the quantity of NO_3^- leached in winter drainage. DC grazing is a practical method to reduce N leaching from grazed dairy pastures.

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Reducing nitrate leaching losses from a Taupo pumice soil using a nitrification inhibitor *eco-n*

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Abstract

The decline in water quality in Lake Taupo has been attributed to nitrogen (N) leaching from surrounding land areas. Pastoral agriculture has been identified as a significant contributor to this N transfer to the lake through animal urine deposition. There is therefore an immediate need for new management options to reduce N losses. The objective of this study was to measure the effectiveness of using a nitrification inhibitor (*eco-n*) to reduce nitrate leaching losses from a pasture soil of the Taupo region. A 3-year study was conducted using 20 lysimeters on Landcorp's 'Waihora' sheep and beef farm, within 10 km of Lake Taupo. The results show that animal urine patches were the main source of nitrate leaching (>95% of the total annual loss) and that *eco-n* significantly ($P<0.05$) reduced nitrate leaching losses from urine treated lysimeters. When the lysimeter results were combined with a detailed GPS survey and GIS analysis of urine patch coverage of the farm it is concluded that *eco-n* reduced annual nitrate leaching losses by between 23 and 32%, with an average reduction of 27%. Thus *eco-n* represents a practical technology that pastoral farmers could adopt today, to assist them to meet new water quality standards in sensitive catchments near Lake Taupo and the upper Waikato River.

Keywords: nitrate, urine patch, nitrification inhibitor, eco-n

Introduction

Environment Waikato has set a target 'to reduce the manageable sources of nitrogen flowing into Lake Taupo by 20%' (Environment Waikato 2007). Environment Waikato's variation No.5 to the Waikato Regional Plan contains new policy and rules to manage land use in the Taupo catchment area. The new rules cap the annual average amount of nitrogen leached from farmland. This represents a considerable challenge to the farmer who wants to increase production because most of the nitrogen that is leached from pastoral farms comes from urine that is deposited by grazing animals, rather than directly from fertiliser (Scholefield *et al.* 1993; Ledgard *et al.* 1999; Di & Cameron 2002a, 2002b).

One of the management options that has the potential to reduce nitrate (NO_3^-) leaching from agricultural land is the use of nitrification inhibitors which slow down the conversion of ammonium (NH_4^+) to NO_3^- in the soil (Amberger 1989). Recently, Di and Cameron (2002c, 2004) demonstrated that NO_3^- leaching losses from grazed dairy pastures on sedimentary soils can be substantially decreased by treating the soil with a nitrification inhibitor, DCD. The objective of this trial, therefore, was to measure the effectiveness of using a DCD-based nitrification inhibitor (*eco-n)* to reduce nitrate leaching from a typical pasture soil in the Taupo region.

Since it is practically impossible to collect all of the water that drains from a paddock of free-draining soil, such as a Taupo pumice soil, it is very difficult to quantify the amount of nitrate leaching losses that occur at the paddock scale. A further complication to quantifying leaching losses at the paddock scale is that in a grazed pasture there are two distinct contrasting areas; (i) animal urine patch areas and (ii) inter-urine patch areas. The difference between these two areas is commonly observed as relatively large differences in pasture growth. The differences in nitrate leaching losses between these different areas are also considerable, with over 90% of nitrate leaching being reported to originate from urine patch areas and less than 10% originating from inter-urine patch areas (Ledgard *et al.* 1999; Di & Cameron 2002a, 2002b).

One approach, in order to try to overcome these problems, is to use lysimeters to measure the leaching losses from stratified samples representing urine patch and inter-urine patch areas and to 'scale-up' the results to the paddock scale by using information about the percent area coverage of urine patches.

Materials and Methods

Lysimeter experiment

A typical grazed pasture site was identified on Landcorp's 'Waihora' farm, in the Lake Taupo district. The soil type is a free-draining yellow brown pumice soil, with soil fertility levels of pH 5.9, Olsen P 22 µg P/mL and Sulphate-S 20 µg S/g. The pasture was a mixture of perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*). The stocking rate was typical of this class of hill country in this region with 13 ssu (standard stock units) per hectare.

Twenty lysimeters (300 mm diameter x 500 mm deep) were collected from the site in April 2003. Obvious urine and dung patch areas were avoided and the lysimeters were collected by placing a PVC cylinder on the soil surface, digging around the casing, making sure to minimise disturbance to the soil structure inside, and gradually pushing the casing down by small increments (Cameron *et al.* 1992). Once the casing had reached the desired depth (500 mm) the soil monolith was cut at the base with a cutting plate, secured on the lysimeter casing and lifted out of the collection site. The gap between the soil core and the casing was sealed using petroleum jelly to stop edge-flow effects.

The lysimeters were then installed in field facilities where the lysimeters were placed on either side of a trench with the soil surface of the lysimeters at the same level as the surrounding paddock. The space outside the lysimeter casings was backfilled with soil to the same level as the surface of the lysimeters and the surrounding paddock. The lysimeters were thus exposed to the same climatic conditions as the soil and pasture in the surrounding paddock.

Four treatments, each with five replicates, were allocated to the lysimeters in a randomised design: (i) urine alone (equivalent to 700 kg N/ha/yr in May); (ii) urine (May) plus *eco-n* (10 kg/ha in both May and August); (iii) no urine; and (iv) no urine plus *eco-n* (10 kg/ha in both May and August). The application rate of 700 kg N/ha was equivalent to the typical N loading rate under a beef cattle urine patch (Haynes & Williams 1993). The DCD was applied at the rate of 10 kg active ingredient per ha as a fine particle suspension on to the surface of the lysimeters following the urine application on the same day. The DCD treatment was repeated in early August in line with recommended use of the *eco-n* product. Pasture cover at the time of DCD application was short (equivalent to c. 1200 kg dry matter/ha) to simulate conditions after grazing and to ensure that the DCD treatment reached the soil surface. The herbage on each lysimeter was cut periodically to simulate typical grazing practice. Because animal urine deposition is unlikely to occur on the same spot each year, subsequent urine application treatments were made onto the 'no urine' treatment lysimeters of the previous year.

The total amount of water draining from the lysimeters over each entire year was measured on a weekly basis (or more frequently if heavy rain persisted during any one week) and a sub-sample of 100 mL was collected for analysis. The concentration of nitrate-N in the leachate was determined by standard FIA analysis (Tecator Inc., Sweden) and the amount of nitrate-N leached was calculated from the NO_3^- concentration and the drainage volume collected from each lysimeter on each occasion. Analysis of variance was performed using Genstat (Version 9, Lawes Agricultural Trust) and least significant differences (LSD) calculated.

GPS/GIS measurement of urine patch coverage

A new method, using global positioning system (GPS) and geographic information system (GIS) technology, was used to quantify the spatial distribution and area coverage of urine patches deposited by grazing animals (Moir *et al.* 2006). Six 10 x 10 m (100 m^2) plots were marked out on typical areas of the Waihora farm. Three plots were located on flat areas (0-3% slope), while the remaining three were located on easy rolling to medium hill slopes (3-7% slope). Urine and dung patches inside each plot were recorded using survey-grade differential GPS at regular time intervals through the year. Real-time kinematic (rtk) GPS was used to enable the exact location of the centre of each urine and dung patch to be recorded. Grazing information was recorded in order to relate area coverage data to stocking rate and total grazing days.

The radius of each patch was measured with a measurement arm attached to the base of the GPS pole and was also recorded (in cm) in the data logger to enable area to be calculated. Patches were identified either by pasture growth response to urine, or dung, or by any obvious fresh dung deposition. Measurements were generally made 2-3 weeks after the paddock had been grazed so that pasture growth response areas to urine and dung could be easily observed. The sampling time interval was set at a minimum of 3 months.

Results and Discussion

Lysimeter results

The results for year one show that animal urine patches were the main source of nitrate leaching (>95% of the total annual loss) in this Taupo pumice soil (Fig. 1). The results also show that the application of the nitrification inhibitor *eco-n* significantly ($P<0.05$) reduced nitrate leaching losses from urine patch areas. In 2003-4 the amount of NO_3^- leached from urine treated lysimeters was reduced from 306 to 213 kg N/ha/yr with *eco-n* (representing a 30% reduction), in 2004-5 the reduction was from 289 to 195 kg N/ha/yr (33% reduction), and in 2005-6 the reduction was from 133 to 81 kg N/ha/yr (39% reduction). The variation in amount of nitrate-N leached each year appears to be related to the amount of rainfall received, with the greatest nitrate-N loss occurring in year 1 when the greatest amount of drainage occurred (635 mm), compared to year 2 (563 mm) and year 3 (486 mm) when lower nitrate-N losses occurred.

These results relate to leaching losses of urine-N deposited in May. Losses from urine-N deposited at other times of the winter/ early spring drainage period may be different. Unfortunately, there are no other data available to compare these losses against and it is therefore difficult to speculate about what the losses would be at other

Figure 1 Amount of nitrate-N leached in drainage water from the Taupo lysimeters as affected by urine treatments and *eco-n* nitrification inhibitor over the 3 year period of the study.

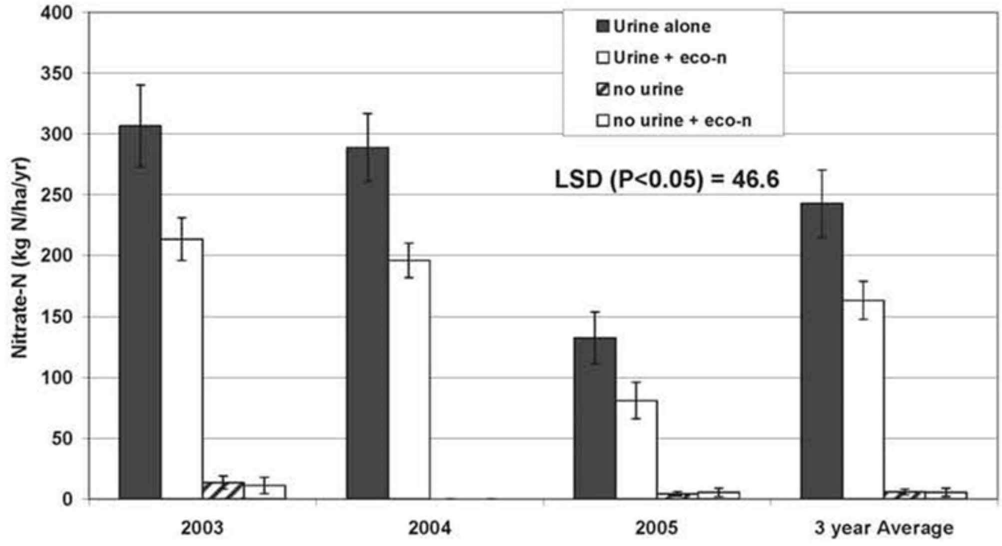

Table 1 Average annual cumulative urine patch coverage.

	Total plot area covered (%)	SE
Flat area	19.3	1.4
Hill slope	14.3	1.7

Table 2 Slope classes and percentage of the Waihora farm area.

Slope Class	Slope (%)	Percentage of total area	Area (ha)
1	0 - 3.0	56.8	1956
2	3.1 - 7.0	38.9	1338
3	7.1 - 15	4.4	150
Total area			3,444

times. However, it is worth noting that other work has shown that at winter soil temperatures typical of the Taupo district (<8°C) the effect of the *eco-n* inhibitor can last for 2 to 3 months (Di & Cameron 2004b). In addition, work involving a 2.5 week delay in animal urine deposition relative to inhibitor treatment has shown that nitrous oxide gas emissions were reduced by 73%; indicating that the inhibitor was still effective after this time period (Di *et al.* 2007). Therefore it is likely that an application of the inhibitor in May still has the potential to reduce nitrate-N leaching losses from urine that is deposited during winter.

Although the percentage reductions in nitrate leaching reported here are less than previously reported (e.g. 75% in Di & Cameron 2004a) they still represent a statistically significant ($P<0.05$) reduction in nitrate leaching loss (Fig. 1). The reasons for the lower percentage reductions are unclear at this stage but may be due to differences in soil type and/or climatic conditions. Current work indicates that the actual percentage reduction is not necessarily related to the urine application rate and that reductions of over 50% can still be achieved in sheep urine patches where the N loading rate is equivalent to around 300 kg N/ha/yr.

Urine patch coverage results

The average annual cumulative coverage of urine patches on the flat areas was 19.3% and on hill slopes the average annual urine patch coverage was 14.3% (Table 1). Dung patches were excluded from data analysis.

The annual cumulative urine patch coverage for the different slope classes calculated from the research plots, situated on the respective slopes, was then used to calculate the annual urine patch coverage for the full Waihora farm. The different land slope class distributions on the Waihora farm are shown in Figure 2 and areas of each slope class are given in Table 2.

In total, 18.0% (569 ha) of the Waihora farm would be covered by urine depositions each year (Table 3).

Paddock scale nitrate leaching losses

The GPS/GIS urine patch spatial data and the lysimeter trial data were combined together to calculate the nitrate leaching losses at the paddock scale using the following equation (Di & Cameron 2000):

$$N_L = N_{L1} \times A_1 + N_{L2} \times A_2$$

Where N_{L1} and N_{L2} are the N leaching losses from the

Figure 2 Slope class distribution on the Waihora farm.

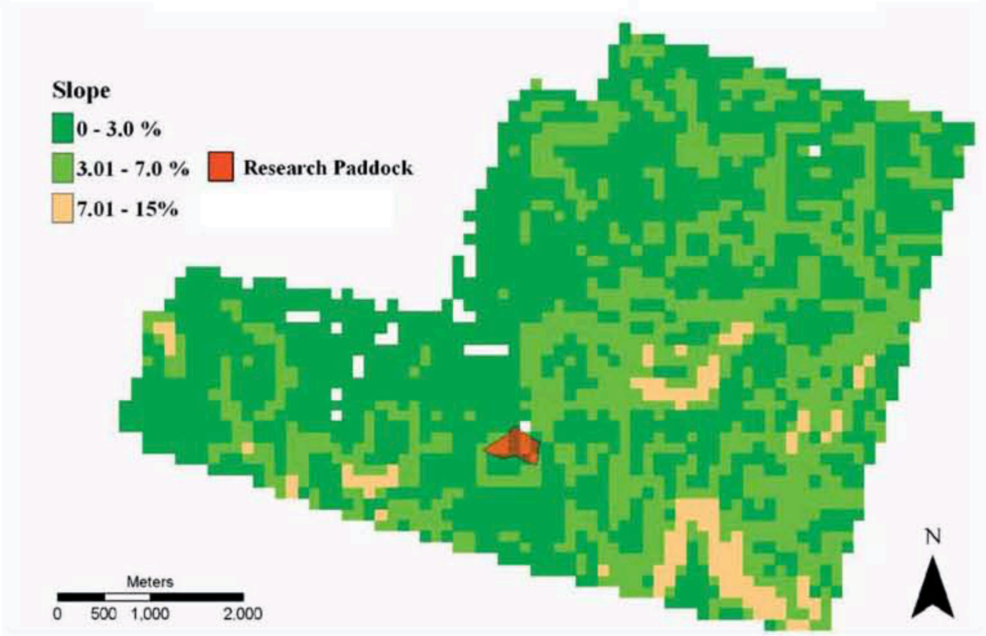

Table 3 Farm area annually covered by urine patches according to slope class.

Slope class	Slope (%)	% of total area covered by urine patches	Total area covered by urine patches (ha)
Flat Areas	0 - 3.0	11.5	377
Hill Slopes	3.1 - 7	6.5	213
Total area		18.0	590

Table 4 Estimate of the effect of *eco-n* on the annual amount of NO_3-N leached from the Waihora farm in the 2003-4 and 2005-06 seasons.

	2003-4 kgN/ha/yr	2004-5 kgN/ha/yr	2005-06 kgN/ha/yr
No *eco-n*	39.9	25.9	15.9
Plus *eco-n*	29.4	17.6	12.2
eco-n reduction (%)	26	32	23

urine patch area (A_1) and the non-urine patch area (A_2), respectively.

Based on the drainage results from the lysimeters and the 25 year average climate data from NIWA, it was found that drainage (and thus nitrate leaching) predominately occurs from urine deposited on this farm during May to October. Therefore the 6-month urine patch area (A_1) prone to leaching is approximately 9% (i.e. 50% of the total value given in Table 3 above) with the remainder of the area being inter-urine patch areas.

The results (Table 4) show that the use of *eco-n* nitrification inhibitor technology has the potential to reduce nitrate leaching losses from the Waihora farm by between 23 and 32%, with an average reduction of 27%. This reduction in nitrate leaching is greater than the 20% target set by Environment Waikato, making this a viable mitigation strategy for use in the Taupo catchment.

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RESEARCH ARTICLE

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Milk production and urinary nitrogen excretion of dairy cows grazing plantain in early and late lactation

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ABSTRACT

The effects of 50% or 100% of a herbage diet of plantain on milk production and urinary nitrogen (N) concentration were measured in two experiments for late (autumn 2015) and early (spring 2015) lactation dairy cows. Three groups of 12 mixed age Friesian × Jersey dairy cows were offered a perennial ryegrass-white clover pasture, or pure plantain or 50% perennial ryegrass-white clover and 50% pure plantain by ground area (50–50 pasture–plantain). Urine N concentration was lower in both experiments ($P < .001$) for plantain (2.4 and 2.2 g N/L) and 50–50 pasture–plantain (3.6 and 3.4 g N/L) than pasture (5.4 and 4.7 g N/L). Cows on plantain or 50–50 pasture-plantain produced at least as much milk as those on pasture in both experiments. Plantain may offer environmental benefits to dairy systems by reducing the N concentration of urine deposited on the soil from grazing cows.

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