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Invited paper

Nutrient management in New Zealand pastures—recent developments and future issues

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Abstract In this publication we review recent research and understandings of nutrient flows and losses, and management practices on grazed pastoral farms in New Zealand. Developments in nutrient management principles in recent years have seen a much greater focus on practices and technologies that minimise the leakage of nutrients, especially nitrogen (N) and phosphorus (P), from farms to the wider environment. This has seen farm nutrient management planning shift from a relatively small set of procedures designed to optimise fertiliser application rates for pasture and animal production to a comprehensive whole-farm nutrient management approach that considers a range of issues to ensure both farm productivity and environmental outcomes are achieved. These include consideration of factors such as multiple sources of nutrient imports to farms, the optimal re-use and re-distribution of nutrient sources generated within the farm (such as farm dairy effluent), identification of the risks associated with applying various nutrient forms to contrasting land management units, and an econometric evaluation of farm fertilisation practices. The development of nutrient budgeting and econometric decision support tools has greatly aided putting these more complex whole-farm nutrient management systems into practice. Research has also identified a suite of mitigation systems and technological measures that appear to be able to deliver substantial reductions in nutrient losses from pastoral farms. However, issues of cost, complexity, compatibility with the current farm system, and a perceived uncertainty of actual environmental benefits are identified as key barriers to adoption of some of these technologies. Farmers accordingly identified that their main requirement for improved nutrient management planning systems was flexibility in how they would meet their environmental targets. The provision of readily discernible information and tools defining the economic and environmental implications of a range of proven management or mitigation practices is a key requirement to achieve this.

Keywords adoption; farm dairy effluent; grazed pastures; mitigation; nitrogen; nutrient budgets; phosphorus; water quality

INTRODUCTION

Nutrient management is important for farm production and profitability, and the environment. On New Zealand pastoral farms, fertiliser represents either the second (sheep and beef) or third (dairy) major item of expenditure after debt servicing and labour costs (MAF 2007). Considerable research effort over the past 4 decades has been placed into understanding how nutrients cycle in grazed pastures and how

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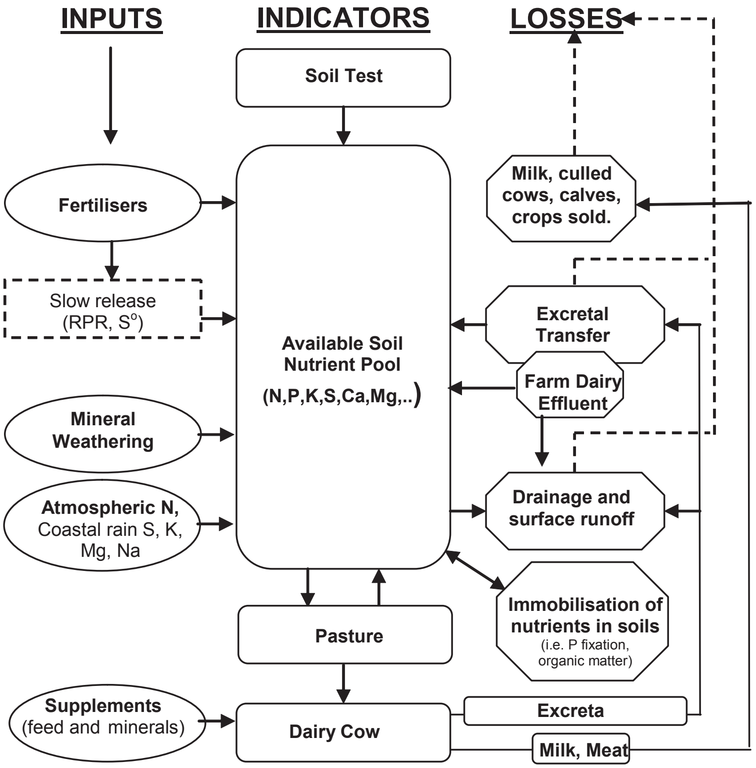


Fig. 1 Inputs, transfers and losses of nutrients from the soil-pasture-animal system on a standard dairy farm. RPR, reactive phosphate rock; S°, elemental sulphur.

Table 1 The fate of minerals ingested by lactating dairy cows (average pasture intake of 10 kg DM day^{-1}) (During 1984).

Element	Consumption (kg week^{-1})	Percentage in			
		Faeces	Urine	Milk	Retained
N	2.54	26	53	17	4
P	0.23	66	–	26	8
K	1.72	11	81	5	3
Mg	0.23	80	12	3	5
Ca	0.72	77	3	11	9
Na	0.27	30	56	8	6

fertiliser nutrients can be most efficiently used to optimise agronomic productivity. Earlier research on the recycling of nutrients in grazed pasture systems was reviewed by Haynes & Williams (1993), and in particular they reinforced the concept that nutrients in pasture ingested by the grazing animal are inefficiently utilised in growth, or milk and wool production. The majority of nutrients are excreted in dung or urine (e.g., Table 1). Much of the earlier research also focused on defining the relationships between soil test measures of plant available nutrients and pasture production under New Zealand conditions. This led to the development of a suite of soil sampling, testing and interpretation procedures which underpin most current farm fertiliser recommendations (Morton & Roberts 1999; see also reviews by Edmeades et al. 2005, 2006; Morton & Gillingham 2005). Much of the agronomic information derived from this earlier research was captured in the CFAS fertiliser recommendation model (Cornforth et al. 1982), which was later extended to an econometric modelling framework as the OVERSEER® fertiliser recommendation model (Metherell et al. 1997; Metherell 1999) and the OVERSEER® nutrient budget model (Ledgard et al. 1999a; Wheeler et al. 2003, 2006). The development of these decision support tools has shifted nutrient management decision making from simple fertiliser recommendations to more comprehensive farm nutrient management planning and nutrient efficiency evaluations on both a block (i.e., farm areas of similar soils and managements) and whole-farm basis, taking account of nutrient transfers onto and within the farm (Fig. 1). These systems have been augmented with plant (Sinclair et al. 1997a,b; Morton et al. 2001) and animal testing, particularly for trace nutrients, and are now being increasingly adopted into reporting systems by fertiliser company representatives and soil testing laboratories.

Most recently, greater attention has been focused on some of the off-site impacts of farming activities, particularly the consequences of nutrient enrichment of ground and surface waters and the contribution of greenhouse gases such as methane and nitrous oxide to global warming. In a sparsely populated country heavily dependent on its primary industries, New Zealand agriculture inevitably makes a significant contribution to these issues at a national level. Intensive agriculture is known to emit significant amounts of nutrients, particularly nitrogen (N) and phosphorus (P) (e.g., Ledgard et al. 1999b; Monaghan et al. 2005, 2007; Wilcock et al. 2006). Whilst these emissions are typically not large by agronomic standards (at least for P), the transfer of these pollutants from land to water can result in significant water quality impairment. Community concerns about water quality deterioration are reflected in various regional council water plans, many of which have the stated goal of reducing non-point source pollution of water bodies, particularly those that are highly valued. Consequently, greater research effort has been invested over the past decade in strategies to minimise nutrient losses from farming systems. This impetus has also fuelled the development of the nutrient management planning tools described above, particularly nutrient budgeting. Nutrient budgeting is a valuable tool to account for all nutrient input sources, to determine the efficiency of nutrient management on farms, and to examine the potential environmental impacts. To be effective, nutrient budgets must encompass the key drivers of nutrient flows on farms and incorporate the main management practices which determine nutrient losses. The main drivers of nutrient use efficiency and loss for a farming system are the:

(1) magnitude of nutrient input,
(2) inefficiencies in nutrient cycling, and
(3) timing of inputs and key management practices.

In this paper we review some of the more recent research and understandings which have focused on these drivers in New Zealand pastoral agriculture, particularly within the context of nutrient losses to water bodies and with particular emphasis on N and P, the two elements of most concern for nutrient enrichment of waters. Based on these drivers, some current and potential mitigation measures to reduce nutrient losses from farming systems are outlined. We also consider how future farm nutrient management systems may look if, as we anticipate, desired reductions in nutrient losses from farming systems are sought on a wider scale and more urgently than has occurred until now.

THE ROLE OF N AND P FERTILISERS IN NUTRIENT LOSSES TO THE ENVIRONMENT

The magnitude of nutrient inputs to a farm system is generally the main factor determining the nutrient surplus and therefore the potential for nutrient loss. This is evident for N in Table 2 where a three-fold increase in total N inputs resulted in a four-fold increase in N surplus, a four- to five-fold increase in gaseous and leaching losses, and a halving of the N use efficiency (Ledgard et al. 1999b). Two major inefficiencies in the cycling of N in grazed pastures and the conversion into product-N are (i) the high protein content of pastures compared to the dietary requirement of grazing animals, and (ii) the high concentration of excreted N in urine patches equivalent to up to c. 1000 kg N ha^{-1} (Haynes & Williams 1993). A large amount of research in New Zealand and overseas over the past 3 decades has clearly shown that the amount of N excreted by animals, and in particular urine N, is the most important determinant of N losses (including leaching, runoff and gaseous losses) from pastoral farms (e.g., reviewed by Ledgard 2001; Di & Cameron 2002a). Consequently, the amount of N excreted by animals is the primary driving factor of N losses rather than inefficiencies related to N fertiliser usage. The amount of N excreted is closely tied to the amount of N consumed by the animals which in turn is broadly related to the animal stocking rate. The main effect of fertiliser N use on N cycling efficiency in grazed pastures is thus an indirect one whereby N fertiliser inputs allow for an increase in pasture production, animal stocking rate and thus urine N

Table 2 N inputs and outputs from intensive dairy farm systems in New Zealand receiving N fertiliser at nil or 410 kg N ha^{-1} yr^{-1} (Ledgard et al. 1999b, unpubl. data). Bracketed values are range in N flows measured over 5 years.

	0 N	400 N
N inputs (kg N ha^{-1} yr^{-1})		
Clover N_2 fixation	160 (80–210)	40 (15–115)
Non-symbiotic fixation + atmospheric deposition	10	10
Fertiliser N	0	410
Purchased feed	0	41
N Outputs (kg N ha^{-1} yr^{-1})		
Milk + meat	78 (68–83)	114 (90–135)
Transfer of excreta to lanes/sheds	53 (41–63)	77 (72–91)
Denitrification	5 (3–7)	25 (13–34)
Ammonia volatilisation	15 (15–17)	68 (47–78)
Leaching	30 (12–74)	130 (109–147)
Immobilisation of fertiliser N		70 (60–84)
N balance (kg N ha^{-1} yr^{-1})	–11 (–74 to +47)	7 (–11 to +24)
Farm N surplus (kg N ha^{-1} yr^{-1})	92	387
N use efficiency (product-N/input-N)	46%	23%

excretion. The relationship between N losses and stocking rate is closer for sheep and beef farms because of the relatively small variation between farms in external N inputs, whereas on dairy farms there is a wide variation in external N inputs and in per-cow production and intake of feed-N (e.g., c. three-fold). Thus, stocking rate (animals ha^{-1}) is only a crude proxy for the magnitude of N loss to the environment for a dairy farm.

Fertiliser N use on farms in New Zealand has been steadily rising, although average annual inputs are currently only of the order of 15 and 110 kg N ha^{-1} on sheep-beef and dairy farms, respectively (from annual reports/summaries by Meat & Wool Economic Service and Dexcel ProfitWatch). Almost all New Zealand farmers use split applications of <50 kg N ha^{-1} application^{-1} and few use in excess of 250 kg N ha^{-1} on an annual basis. Research indicates that direct leaching of fertiliser N under these management conditions is generally low (Ledgard et al. 1999b; Di & Cameron 2002c; Monaghan et al. 2005), except for applications in late autumn/winter, when losses may be up to one-third (Ledgard et al. 1988). Increased N leaching from fertilisers or from greater excreta-N cycling results in an increase in leaching of associated cations, especially calcium (e.g., Rajendram et al. 1998), and these need to be replaced in the longer term to maintain production. Ammonia volatilisation from recently applied urea and ammonium-based fertilisers can be an important N loss pathway, particularly in situations where soil pH is high and warm conditions favour the removal of ammonia gas from the soil surface. Urea has the greatest potential for ammonia loss due to the pH increase associated with urea hydrolysis. In many instances this pH increase can range from 2 to 3 pH units, particularly on poorly buffered soils. Losses reported from pasture receiving urea fertiliser vary according to the amount and timing of the application, and a distinct seasonal variation is apparent in some of the studies reported in the literature. For applications of less than or equal to 50 kg N ha^{-1}, ammonia volatilisation losses typically range between 5 and 15% of the N applied (Black et al. 1984, 1985), although greater losses have been reported under warmer summer/autumn conditions (Theobald & Ball 1984). Ammonia losses from pastures fertilised with ammonium sulphate or calcium ammonium nitrate (CAN) are typically less than 2–3% of the N applied (Black et al. 1984, 1985; Theobald & Ball 1984). Losses from di-ammonium phosphate (DAP)-treated soils tend to be intermediate between those reported following urea and ammonium sulphate fertilisation (Black et al. 1985).

Fertiliser P use on New Zealand dairy farms has also been relatively high in recent years, leading to an increase in the proportion of farms with high soil P tests. The possibility of reducing fertiliser P use on some farms is backed up by results from a survey of over 240 000 soil test results for Olsen P over a 14-year period, where about 50% of dairy farms were above the biological and economic optimum values

(Wheeler et al. 2004). Nevertheless, New Zealand soils are far from being saturated in P, and P fertiliser inputs are essential to maintain productivity. Constraining P inputs to achieve economic optimum soil test levels can reduce the risk of P loss from soils. If best practice is followed, direct losses of P from fertiliser are generally less than 10% of the total P lost from pastures (McDowell & Catto 2005). Best practice implies that fertiliser is not spread too close to waterways and is applied more than 2 weeks before irrigation or at a time of year when P losses are unlikely—usually summer months when rainfall and overland flow are less. However, if this practice is not followed then P losses from fertilisers can account for the majority of P losses from the farm (Hart et al. 2004). For instance, applying superphosphate in winter resulted in 2.3–6.7% of the P applied at 50 kg ha^{-1} to be lost in either overland flow or drainage (Sharpley & Syers 1979), equating to 66–93% of annual P export from plots and catchments. In Australia, Bush & Austin (2001) and Nash et al. (2004) also found that soluble P fertiliser could account for a large proportion of the P lost from border-dyke irrigation bays when irrigation was applied within a few days of fertiliser application. Measurements of annual P losses in runoff from New Zealand sheep and cattle grazed pastures range from 0.1 to 3 kg P ha^{-1} yr^{-1} (Table 3). Factors such as slope, land-use intensity, rainfall energy, hydrological flow pathways, soil structural vulnerability, fertilisation and irrigation practices and riparian management strongly influence the magnitude of these losses and the proportion of loss which occurs in a dissolved form. There is a general trend evident in Table 3 that dissolved P forms represent a greater proportion of P loss as land-use intensity increases and erosion losses of sediment-associated P decrease.

Generally, the potential for P to be lost in either overland flow or drainage decreases exponentially with time after fertiliser topdressing, so that 30–60 days after topdressing the concentration of P lost in runoff from superphosphate treated plots had decreased to that of non-treated plots (McDowell et al. 2003b). The risk of P loss following fertiliser application depends on the solubility of the fertiliser applied, with superphosphate > serpentine

Table 3 Measured annual P losses from pastured fields and catchments in New Zealand (adapted and updated from Gillingham & Thorrold (2000) and McDowell et al. (2005c)). DRP, dissolved reactive phosphorus; TP, total phosphorus; SU, stock units; nd, no data.

Land use	Total P loss (kg ha^{-1} yr^{-1})	DRP/TP (%)	Reference
Field scale losses			
Border dyke irrigated dairy	3.4	>80*	Carey et al. (2004)
Mole-pipe drained dairy, Brown soil	0.29–0.37	33	Monaghan et al. (2005)
Mole-pipe drained dairy, Pallic soil			Houlbrooke et al. (2003)
Non-effluent area	0.34	59	
Effluent-treated area	0.86	38	
Dairy pasture, Pallic soil			R. M. Monaghan (unpubl. data)
Mole-pipe drained land	0.86	42	
Undrained land	0.42	45	
Catchment scale losses			
Sheep grazing (15 SU ha^{-1})	0.29	<38*	McColl et al. (1977)
Sheep and cattle grazing (11 SU ha^{-1})	1.60	<24*	Bargh (1978)
Sheep grazing (6–13 SU ha^{-1})	0.70	16	Lambert et al. (1985)
Sheep and cattle grazing (13 SU ha^{-1})	2.37	12	Quinn & Stroud (2002)
Cattle grazing (6–13 SU ha^{-1})	1.50	9	Lambert et al. (1985)
Sheep grazing (19 SU ha^{-1})	0.75	<20*	Smith (1987)
Deer grazing	0.56–0.90	4–5	McDowell et al. (2006)
Sheep and dairy grazing, forestry, Bog Burn	0.43	54	Monaghan et al. (2007)
Mixed land use (c. 12 SU ha^{-1}), Oteramika	0.5	nd	Thorrold et al. (1999)
Irrigated dairy and sheep grazing, Waikakahi	0.81	68	Wilcock et al. (2007)
Dairy grazing, Waiokura	0.86	23	Wilcock et al. (2007)
Dairy grazing, Toenepi	0.92	49	Wilcock et al. (2006)
Dairy grazing (high rainfall), Pigeon	2.02	45	Wilcock et al. (2007)

*Based on reported total dissolved P losses.

super > reactive phosphate rock (McDowell & Catto 2005). If superphosphate cannot be applied at a time when P loss is unlikely, or if soils are hydrophobic, consideration should be given to applying a lower solubility P fertiliser to minimise losses. The timing of inputs with respect to rainfall and soil moisture conditions is therefore an important consideration that can also markedly influence the efficiency of P use and the extent of P losses. The variability in and potential contribution of fertiliser P to total P losses from a typical grazed pasture are illustrated in Table 4, which shows the estimated contribution to farm P losses from various forms of P fertiliser applied at 30 kg P ha^{-1} at times when overland flow was likely (June) or unlikely (December) in Southland. Losses from the farm were on average 1 kg P ha^{-1} yr^{-1} for the 2 years of monitoring. This indicates that, if good practice is followed, superphosphate would contribute a maximum of 9% of farm P losses. However, this contribution would increase to 24% if best practice was not adopted and superphosphate was applied in June.

ADVANCES IN INTEGRATED NUTRIENT MANAGEMENT OF FARM DAIRY EFFLUENT

Typical New Zealand dairy farm milking practices lead to a considerable transfer of nutrients from pastures to the farm dairy shed. Based on the time cows spend in the dairy shed and collecting yards, it is estimated that on average 6% of dairy cow excreta is deposited on this area (Ledgard & Brier 2004). This represents a significant nutrient source that can supply valuable amounts of N, P, K and other elements to pasture, representing a potential saving of 10–15% in a farm's annual fertiliser requirements. Prior to the 1980s, raw farm dairy effluent (FDE) was disposed of in long ditches or by heavy rates of spray application to paddocks, with both systems creating the risk of raw effluent with biological oxygen demands (BOD) close to 2000 mg litre^{-1} leading to pollution of some surface waters (Macgregor et al. 1979). In the 1980s and 90s the two-pond system, which combines an anaerobic pond with a facultative pond (Sukias et al. 2001), was introduced as the preferred form of treatment to reduce the BOD of raw FDE before discharge to streams. The combination of anaerobic and aerobic ponds efficiently removes sediment, BOD and pathogenic bacteria, but despite some reduction high concentrations of N and P still remain in the effluent from aerobic ponds (Table 5; Hickey et al. 1989; Ledgard et al. 1996; Longhurst et al. 2000; Sukias et al. 2001). Further improvements in pond treatment of FDE have been achieved with the development of the Advanced Pond System (Craggs et al. 2004). This system appears to be most effective at reducing concentrations of faecal bacteria in pond discharges, but concentrations of N and P may remain relatively high.

From 1995 onwards, the two-pond treatment system, with discharge to a stream, began to be phased out because it was recognised that the nutrient-rich aerobic pond discharge had adverse environmental impacts on surface water quality (Hickey et al. 1989; Parminter 1995). Irrigation to land of raw FDE taken directly from a sump holding the daily yard wash-down, or partially-treated FDE taken from an existing two-pond system, is now the preferred treatment option for many regional councils. A lack of awareness of the nutrient content of FDE often leads to widespread nutrient enrichment of soil irrigated with FDE, especially on small land areas. This has been identified in nutrient budgets constructed for dairy farmers and confirmed by subsequent soil testing. The OVERSEER® nutrient budget model can be used to adjust fertiliser nutrient inputs to a dairy farm by accounting for additional nutrient inputs in FDE, particularly N and K, to the

Table 4 Estimates of direct losses of fertiliser P in overland flow from a grazed pasture in Southland under contrasting forms and timings of P fertilisation (from McDowell & Catto 2005).

Application date	Superphosphate (kg P ha^{-1} yr^{-1})	Serpentine superphosphate (kg P ha^{-1} yr^{-1})	Reactive phosphate rock (kg P ha^{-1} yr^{-1})
2002			
Jun	0.24	0.14	0.01
Dec	0.04	0.01	0.01
2003			
Jun	0.23	0.11	0.01
Dec	0.09	0.01	0.01

FDE blocks within the farm (Wheeler et al. 2003). The model can also be used to suggest remedial action for nutrient enriched areas, such as increasing the land area receiving FDE (see case study by Hedley et al. 2004). Most regional councils (e.g., Environment Waikato 1994; Selvarajah 1996) have established upper N loading limits of between 150 and 200 kg of FDE-N ha^{-1} yr^{-1} and many use this as a guideline for allocating land area for FDE irrigation. Adhering to N loading rates does not necessarily overcome surface water nutrient enrichment problems, especially if FDE irrigation events exceed the soil's infiltration or water holding capacities (e.g., during very wet soil conditions). This has been most evident in recent studies of mole and pipe-drained soils by Houlbrooke et al. (2003) and Monaghan & Smith (2004) where drainage discharges of raw or partially-treated FDE occurred during or immediately following effluent irrigation. These discharges can represent a large proportion of the quantities of P and faecal bacteria lost to waters when expressed at the farm scale.

Correctly operated land application systems will minimise the risk of nutrients and harmful microorganisms leaching or draining into fresh water (Cameron et al. 1997). On artificially drained soils, recent research indicates that a deferred irrigation system can minimise the risk of direct drainage of effluent through the mole-pipe network. This involves the linkage of soil nutrient and water balances in the design and management of a sustainable land treatment system. The system involves storing FDE in a holding pond and then applying it strategically when the soil water deficit is sufficiently large to prevent the direct drainage of FDE. Initial studies with this deferred application strategy (Houlbrooke et al. 2003) showed an increase in the amount of total P lost in winter drainage by 0.46 kg P ha^{-1}. To minimise this small accelerated P loss (and ammonium-N and faecal bacteria losses), Hedley et al. (2005) and Monaghan & Smith (2004) proposed that the maximum FDE application depth is limited to the current soil water deficit in the effective pasture root zone. In this way nutrients are likely to be recycled more efficiently by pasture root systems. Houlbrooke et al. (2006) have demonstrated that individual sprinkler systems capable of intermittent or very low application rates (0.9–4 mm h^{-1}) are very effective at preventing both overland flow and the preferential flow of partially treated FDE to pipe drains on Pallic soils in Otago. More frequent scheduling of small irrigation depths requires FDE management systems that include customised pond storage, effluent block nutrient budgets, and a soil water balance monitoring system to guide irrigation scheduling. One of the consequences of meeting the proposed new "low rate deferred" irrigation criteria is that FDE pond storage capacity, or irrigation area, must increase on farms. The technique of Houlbrooke et al. (2006) of intermittent low application, however, increases the scheduling opportunities on soils with very low soil water deficits (winter and early spring) and may in some regions considerably decrease the pond storage capacity required.

Table 5 The improvement in water quality as farm dairy effluent is progressively pond-treated (anaerobic followed by aerobic pond) and land-treated (spray irrigation). Winter drainage was from grazed pastures having received aerobic pond effluent during summer soil water deficits at c. 35 mm yr^{-1} for 3 years. Bracketed values are estimates of nutrients (kg) discharged per 100 cows. Effluent was sprayed onto 4 ha per 100 cows and winter drainage was 250 mm. nd, no data.

	Farm dairy effluent*	Anaerobic pond discharge*	Aerobic pond discharge*	Winter drainage from grazed plots sprayed with aerobic pond effluent†
Biochemical oxygen demand (mg litre^{-1})	2000	190	128	nd
Total P (mg litre^{-1})	80 (110)	30 (40)	25 (35)	1 (10)
Total N (mg litre^{-1})	500 (700)	200 (280)	100 (140)	14 (140)
Ammonium-N (mg litre^{-1})	100 (140)	150 (210)	80 (110)	3 (30)
Nitrate-N (mg litre^{-1})	0	0	0	11 (110)
Faecal coliforms (100 ml^{-1})	2×10^7	1×10^8	3×10^3	2×10^2‡

*From Ledgard et al. (1996).
†Houlbrooke et al. (2003).
‡M. Hedley (unpubl. data).

MANAGEMENT OPTIONS FOR IMPROVED ENVIRONMENTAL EFFICIENCY

Products and practices for decreasing P losses

The loss of phosphorus from grazed pastures can be categorised into that lost from soil and dung or losses originating from added inputs of fertiliser or effluent. Of the many methods available to minimise P losses to waterways from New Zealand pastures, the best approach is to ensure that soil Olsen P is maintained within the range of concentrations considered optimal for pasture production and not excessive for any given soil type. Since the magnitude of P losses from soil via overland or subsurface flow is proportional to soil P concentration (McDowell et al. 2003a; Gillingham & Gray 2006), having an Olsen P concentration above optimum represents an unnecessary source of P loss and an unnecessary waste of the P inputs (e.g., fertiliser, effluent or dung). To ensure P is not accumulating in soils, a nutrient budget should be used to account for all P sources and to estimate P fertiliser requirements to maintain optimum soil P status (e.g., Wheeler et al. 2006). However, maintaining optimal soil Olsen P does not totally prevent P losses from occurring; moreover, some soils can lose a lot of P at optimal Olsen P concentrations for pasture production (e.g., soils with little Al and Fe oxides such as Podzols; McDowell & Condron 2004). Furthermore, if a soil is already P-enriched then it can take many years for Olsen P to decline.

To minimise the component of P loss contributed by factors other than soil P, a number of strategies have been trialled or proposed. These include: cultivation to redistribute high-P topsoil within the

Table 6 Relative P loss and effectiveness of different strategies for decreasing P loss from New Zealand pastures. Note that the relative contribution of fertiliser, dung or soil varies from 5–50%. RPR, reactive phosphate rock. –, not determined.

Strategy/treatment	Dissolved P loss (kg P ha^{-1} or % decrease)	Total P loss (kg P ha^{-1} or % decrease)	Reference
Constructed wetland	–92 to 85%	–426 to 77%	Sukias et al. (2006)
Effluent irrigation			
Travelling irrigator—aerated plot	–	1.5	Houlbrooke et al. (2006), unpubl. data
Travelling irrigator—non-aerated	–	4.3	
K-line—aerated plot	–	0	
K-line—non-aerated	–	1.3	
Sediment ponds			
Above sediment pond	0.04	0.21	McDowell et al. (2006)
Below sediment pond	0.05	0.14	
Fertiliser form			
25 kg P ha^{-1} superphosphate	0.13	0.43	Blennerhassett et al. (2007)
25 kg P ha^{-1} RPR	0.05	0.31	
Winter grazing management			
Grazing crop in winter	0.01	0.16	McDowell et al. (2005a)
Restricted grazing*	0.01	0.04	
P-sorbing amendments			
Fly-ash treated soils	–50 to 0%	0–44%	McDowell (2005)
Greywacke backfill drains	0.4	1.0	McDowell et al. (2005b)
Slag backfill drains	0.1	0.7	
Stream without P sock	0.125	0.5	McDowell & Hawke (2006)
Stream with P sock containing modified slag	0.081	0.4	

*Grazing limited to 3 h then moved to stand-off pad. Note data are for overland flow, which only contributed 5–10% of total P lost from a catchment with a winter forage crop (McDowell 2006).

plough layer (Sharpley 2003), restricting the grazing of cropland by cattle in winter (McDowell et al. 2005a), the use of less water-soluble P fertilisers instead of superphosphate (McDowell et al. 2003b; Table 4), improved effluent storage and application technology, soil aeration to increase the contact area of effluent with soil (Houlbrooke et al. 2006), adding P sorbing compounds to effluent ponds before discharge, ensuring that the irrigation requirements of border-dyke soils are matched with delivery volumes or installing bunding so that no outwash occurs, and border-dyke irrigating no earlier than 7 days after application of fertilisers or grazing to ensure minimal pickup of dung and fertiliser granules (Carey et al. 2004). However, inevitably some P will be lost from pastures, hence other technologies have been trialled to improve P capture from drainage, overland flow or in stream flow. In drainage, McDowell et al. (2005b) trialled the use of steel slag as back-fill in tile-drains, which decreased dissolved P loss by up to 90% compared to greywacke gravel backfill. McDowell (2005) spread Fe-, Al- and Ca-enriched coal fly ash to decrease the concentration of dissolved P in overland flow, and McDowell & Hawke (2006) used processed steel slag encased in cloth and placed on the stream bed ("P socks") to mop up P from stream flow. Additional in-stream or near-stream technologies that decrease largely particulate P losses include sedimentation ponds (McDowell et al. 2006) and wetlands (Sukias et al. 2006), although their efficiency depends on flow rate—decreasing as flow increases. Furthermore, with time, the ability of sediment and associated P to be trapped decreases as wetlands and ponds fill up, which can cause them to become a net source of P loss. Buffer strips or riparian areas also suffer from variable trapping capacity, although this can be overcome if the buffer strip is treated as a cut and carry system and the P removed, or the P-rich topsoil is periodically redistributed in the plough layer during re-grassing. However, much of the work indicating buffer strips are effective for mitigating P loss overlooks the fact that, unless there were significant erosion losses involved, the majority of P lost from pastures is in a dissolved form and would pass through vegetation. Furthermore, micro-topography typically causes flow to converge before exiting paddocks to the point where either flow rate is fast enough to overwhelm any interaction with the strip (Verstraeten et al. 2006) or the strip is by-passed altogether. A summary of the effectiveness of these technologies and strategies is given in Table 6.

The effectiveness of each mitigation strategy can be enhanced if it is used in areas where P losses are greatest. To do this we need to know not only where P is enriched but also when and where the main transport pathways within the catchment are active. The integration and overlap of P sources with hydrological pathways has been termed "critical source areas" (CSAs) and in theory these areas account for the majority of P losses that come from a minority of the catchment area (McDowell et al. 2004). Ideally, once these areas are known, appropriate mitigation strategies can be tailored to stop P loss. For example, Gillingham & Gray (2006) determined that P travelled only a few metres from fertiliser spread in bands across a seasonally dry hill-country catchment in summer, suggesting that much of the P lost from fertiliser was only from areas near the stream and not from all of the hill-slope. Future research developments are likely to focus on improving our ability to identify the locations of these CSAs and further evaluation of the effectiveness of some of the above mitigation strategies that are targeted at these landscape locations (Srinivasan & McDowell 2007).

Products and practices for decreasing N losses

Soil N process inhibitors

Within the last 5 years, much research has been undertaken that has examined the role of N process inhibitors for improving N use efficiencies in pastoral agriculture. One mitigation technology that has recently been developed to reduce nitrate (NO_3^-) leaching and nitrous oxide (N_2O) emissions from grazed pastoral soils is the use of the nitrification inhibitor dicyandiamide (DCD) to slow the conversion of NH_4^+ to NO_3^-. Because most soils in temperate regions of the world have a net negative charge, the positively charged NH_4^+ ion is held onto the negatively charged soil exchange surfaces, giving a greater opportunity for it to be taken up by plants, immobilised into soil organic matter, or fixed into certain 2:1 type clay mineral interlayers, rather than being leached. Dicyandiamide inhibits the first stage of nitrification, i.e., the oxidation of NH_4^+ to NO_2^-, by rendering the ammonia mono-oxygenase enzyme in *Nitrosomonas* bacteria ineffective (Amberger 1989). DCD is biodegradable and decomposes in the soil into NH_4^+ and CO_2. Although DCD has been used in the past to help increase the efficiency of N supply from fertilisers or manures, the results have been variable (e.g., Amberger 1989; Williamson et al. 1998). Until recently its use to reduce NO_3^- leaching losses from animal urine in grazed pasture systems had not been developed or rigorously tested.

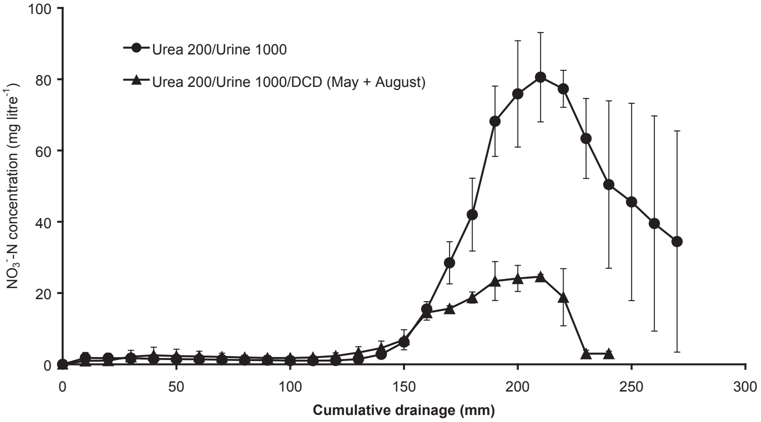


Fig. 2 Nitrate-N concentrations in drainage water from Templeton soil lysimeters which received N fertiliser (200 kg N ha^{-1}) and dairy cow urine (1000 kg N ha^{-1}) with and without dicyandiamide (DCD) (Di & Cameron 2004a).

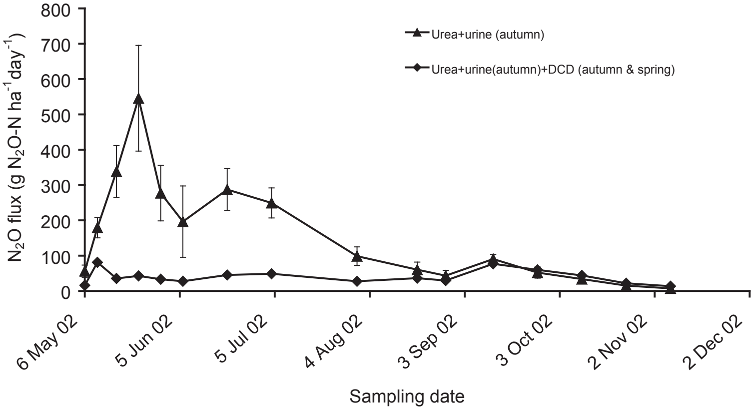
Di & Cameron (2002b) tested the potential of DCD for reducing NO_3^- leaching from dairy cow urine patches using Lismore stony silt loam monolith lysimeters in Canterbury and showed that NO_3^- leaching could be reduced by an average of about 69% for dairy cow urine applied in the autumn or spring. These early results were later confirmed by subsequent studies using deeper Templeton sandy soil lysimeters which showed that DCD reduced NO_3^- leaching from cow urine-N patches by 68–76% (Di & Cameron 2004a, 2005) (Fig. 2). Different formulations of DCD products have been evaluated for their effectiveness in nitrification inhibition (Smith et al. 2005). The longevity of the effect is believed to vary according to temperature and rainfall. In practice, most nitrate leaching occurs from urine patches deposited over a series of grazings during autumn and winter. Therefore, the benefit of DCD use will depend on factors including the timing of DCD applications relative to grazing. DCD is therefore currently used to strategically target this critical period from late autumn to early spring (Di & Cameron 2004a, 2005). Research has shown that N_2O emissions from urine-N patches are also reduced by about 70% with the use of DCD (Fig. 3) (Di & Cameron 2003; Di et al. 2007). An additional benefit with the use of DCD in grazed pastures is the reduction of cation leaching losses associated with the reduction of NO_3^- leaching (Di & Cameron 2004a,b, 2005). Because DCD can help with the retention of nutrients, particularly N, in the top layer of the soil, thus making them available for pasture uptake for a longer period, the use of DCD can also result in increased pasture production. Pasture yield responses of greater than 20% have been recorded with the use of DCD in lysimeter and drainage plot studies (Di & Cameron 2004a, 2005; Moir et al. 2007). However, further studies are required to evaluate pasture responses under field grazing conditions in a wider range of soil and climatic conditions. In addition, long-term studies at a farm systems level where DCD is repeatedly used over a number of years are required to fully evaluate its long-term effectiveness.

Urease inhibitors have also been used to reduce N losses to the environment, particularly, ammonia volatilisation to the atmosphere (Watson et al. 1998). Urease inhibitors work by inactivating the urease enzymes in the soil, thus slowing down the rate of hydrolysis of urea to ammonium. Singh et al. (2006) described how the dual application of the urease inhibitor Agrotain (N-(n-butyl) thiophosphoric triamide) and the nitrification inhibitor DCD reduced both N_2O and NH_3 losses from plots treated with urine at the rate of 600 kg N ha^{-1}. The potential benefits of using soil N process inhibitors has been clearly shown and research is on-going to evaluate their practical use and effectiveness in grazed pasture systems. The retention of N within the pastoral system from the use of soil N process inhibitors has implications for subsequent N cycling and losses, and long-term studies are required to define the long-term benefit of these mitigation technologies.

Plant and animal options to reduce N losses

The physical and chemical characteristics of pasture plants can influence the potential for N leaching losses in grazed pasture systems. Potentially desirable plant physical characteristics include high root density, increased root depth and active winter

Fig. 3 N_2O emissions from a Lismore stony soil which received dairy cow urine with and without dicyandiamide (DCD) (Di & Cameron 2003).



growth, while N-efficient chemical characteristics in plants include high sugar content, reduced N concentration and presence of tannins. Crush & Easton (2004) identified ryegrass selections with differences in rooting characteristics which were associated with differences in N recovery from soil. Grass species with active winter growth will have a greater ability to absorb N from soil during this high leaching risk period than winter-dormant species. Research at sites with cold winter conditions indicate that high-sugar grasses may have potential for increased efficiency of N cycling through greater N recovery by animals and relatively less N being excreted in urine than in dung (e.g., Miller et al. 2001). Research with tannin-containing plants has shown the same potential benefits for N cycling (Barry et al. 1986). However, none of these plant species or characteristics has been evaluated in grazed pasture systems in order to measure their effects on N losses to the environment.

The choice of animal type or animal amendment are strategies that both have the potential to alter the spatial distribution of animal excreta, or the fate of the N excreted, and reduce N leaching. Betteridge et al. (2005a) measured N leaching losses in pastures grazed by beef cows, sheep or deer with the same level of animal N intake and found that N leaching per hectare from beef cows was twice that from sheep or deer. This was attributed to excretion of urine by cattle in fewer patches at higher N concentrations resulting in lower grass N recovery and greater leaching than for sheep and deer. Potentially, this beneficial effect of increased area of urine spread at lower N concentrations could be achieved through supplementation of diuretic compounds such as salt. A recent study with cows supplemented with salt showed increased water intake, greater urination frequency and lower urine-N concentration (Ledgard et al. unpubl. data) indicating the potential benefits to N cycling. Other detailed animal metabolism studies have shown that N process inhibitors can be delivered to animals and excreted intact in the urine resulting in inhibition of nitrification of urine-N on deposition to soil (Ledgard et al. 2007). Field grazing studies to evaluate the effects of these animal amendment options, and an improved grass selection, on N leaching in the Lake Taupo catchment has recently commenced.

System changes to improve the efficiency of N use

As most N leaching in winter drainage occurs from animal urine patches, particularly those deposited at grazings closest to the onset of drainage, management systems which target the amount of urinary N deposited, or modify the timing of deposition can thus greatly influence these losses. On New Zealand dairy farms, the use of brought-in supplementary feed (mainly as pasture or maize silage) is increasing, but remains low relative to other countries, at 1.1 t DM ha^{-1} yr^{-1} (Dexcel ProfitWatch data) in 2004/05. This represents N and P inputs to the dairy farm of only 22 and 3 kg ha^{-1} yr^{-1}, respectively. However, there is a wide range in the amount of supplement used, with most farms being very low but some using up to 20 t DM ha^{-1} yr^{-1} and where it is the main source of nutrient input to the farm. Within a dairy farm, integration of a low-protein forage such as maize silage has the potential to increase the overall efficiency of nutrient conversion into milk, particularly when compared to other sources of extra feed such as N-boosted pasture (i.e., increasing N fertiliser use). However, efficiency indicators need to account

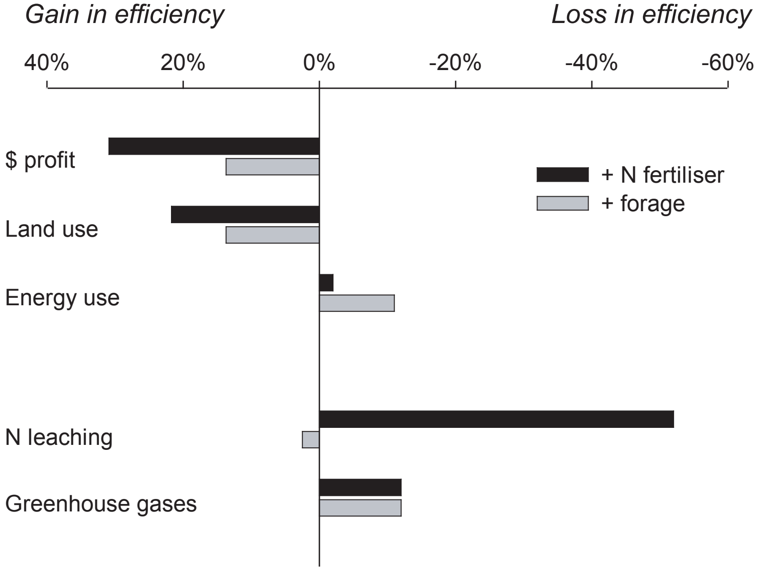


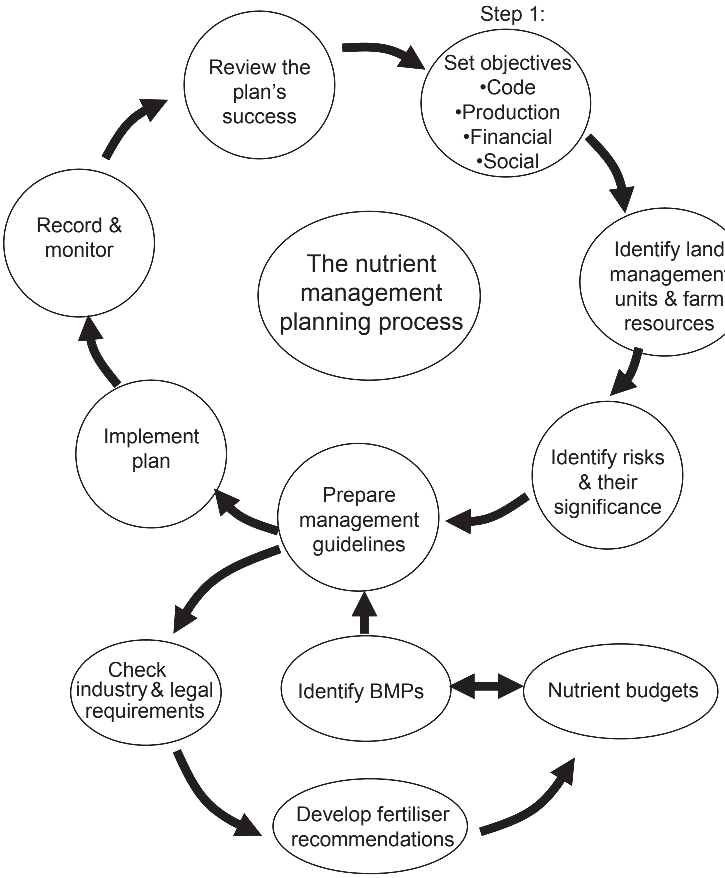
Fig. 4 Effect of intensification from 850 (base) to 1020 kg milksolids ha^{-1} yr^{-1}, using extra N fertiliser (+200 kg N ha^{-1} yr^{-1}) or forage (+2 t DM ha^{-1} yr^{-1} as maize + oats silage), on efficiency of resource use and environmental emissions. Data refer to the whole dairy system (dairy farm + grazing + forage land) (Ledgard et al. 2003).

for the whole system, including the land used to grow the forage which can have relatively high N leaching losses (Ledgard et al. 2006a). Ledgard et al. (2003) examined efficiency indicators for dairy farms and the effect of intensification of dairying to achieve a 20% increase in milk production using forage (maize+oats) or N fertiliser. In this study, a Life Cycle Assessment method was used to account for all contributors to nutrient use and emissions and showed differences in whole-system efficiency for different indicators (Fig. 4). Estimated N leaching losses were much higher for the +N system than the +forage or base farm system at 75 versus 44 and 45 g N leached per kg milksolids, respectively. However, N fertiliser was the more cost-effective intensification option, demonstrating that there can be conflicts between economic and environmental efficiency.

Management practices to reduce the amount of N leaching from animal urine patches over the winter drainage period offer large potential to reduce total N leaching losses. Dairy farm system grazing studies (de Klein et al. 2001, 2006; Chadwick et al. 2002) have shown that restricted grazing management strategies where animals are removed from pasture onto a feed-pad from autumn until calving (4 months), with collection of effluent and re-application during spring/summer, can reduce N leaching by up to 60%. Similar reductions in N leaching when animals are wintered off have been reported for pastures grazed by beef cattle (Betteridge et al. 2005b). Alternative practices such as restricted autumn and/or winter grazing of pastures all have a cost. Desk-top analyses by de Klein et al. (2001) and Monaghan et al. (2007) indicate that the costs of construction and use of winter feed-pads may be compensated for by a small increase in production due to more efficient use of the nutrients in effluent returned to pasture, or due to savings in cow wintering costs associated with off-farm grazing during winter months. In practice, the use of feed-pads or strategic stand-off pads has increased on New Zealand dairy farms, although this has occurred through the desire to reduce pugging damage of pasture, improve animal welfare or in conjunction with increased use of supplement feeding during the milking season. The effectiveness and cost of mitigation practices, such as the use of stand-off/feed pads, differ and the preferred option, or options, will vary between farms depending on economics and practicality. Farmers and researchers are looking for options to increase profitability without increasing environmental impacts.

Preliminary analyses combining estimates of N leaching using the OVERSEER® nutrient budget model and production using the STOCKPOL model (Marshall et al. 1991) with detailed economic analyses have looked at a number of management options for beef farms (Smeaton & Ledgard 2004). Similar analyses, but instead using the UDDER model (Hart et al. 1998; Larcombe 1999) to characterise milk production, have also been performed for dairy systems (Monaghan et al. 2007). These analyses

Fig. 5 A conceptual overview of the nutrient management planning process (re-drawn from NZFMRA 2006). BMPs, best management practices.



indicate that an increase in efficiency, in terms of $ profit kg^{-1} N leached, can be achieved through combinations of intensification and mitigation practices e.g., increased feed inputs as well as strategic use of stand-off pads over winter.

FUTURE ISSUES

Nutrient management in a tighter regulatory environment

Nutrient management on New Zealand farms was historically viewed as optimising fertiliser application rates for pasture production. This has now been transformed into a comprehensive whole-farm nutrient management approach designed to meet multiple goals of optimising pasture growth whilst ensuring good environmental outcomes are achieved. The New Zealand Fertiliser Manufacturers' Research Association (FertResearch) has developed an integrated nutrient management system involving soil testing, an econometric fertiliser recommendation model, the OVERSEER® nutrient budget model and the FertResearch Code of Practice for Fertiliser Use (Fig. 5). The latter includes guidelines on fertiliser management to minimise environmental impacts. The main dairy company (Fonterra) which collects milk from about 95% of all dairy farms has recently developed a Clean Streams Accord with local and central government agencies. The Accord signatories will require all dairy farmers to be using nutrient budgeting to manage nutrient use on farms by May 2008. Various regional councils are also promoting the use of nutrient budgeting, and Environment Waikato's regional plan specifies that farms using N fertiliser at more than 60 kg N ha^{-1} yr^{-1} require a nutrient budget. Clearly, the use of nutrient budgeting on New Zealand farms will increase in the future as industry quality assurance schemes and regional council compliance officers will require farmers to demonstrate that on-farm nutrient use does not compromise soil and water quality. These nutrient budgets and nutrient management systems will be more than just a sheet-of-paper to ward off regulators and will instead be used by farmers to shape farm management strategies that increase whole-farm nutrient use efficiency and meet catchment-specific water quality targets. Experience informs us that some catchments are more sensitive to nutrient pollution than others, and some are specifically sensitive to P (e.g., Lake Brunner) whilst others may be more sensitive to N (e.g., Lake Taupo).

In some catchments that are particularly highly valued and sensitive to further nutrient enrichment, regional council consent will be required to continue farming in the future, and will be based on whether the farm has an appropriate nutrient management plan. Environment Waikato has developed a variation to their regional plan specifically for the Lake Taupo catchment which aims to cap N leaching losses from farms, using N budgets as part of the process to ensure this is achieved. Similarly, Environment Bay of Plenty has developed a rule to cap both N and P outputs from farms around five Rotorua lakes. The draft regional plan of Environment Canterbury also includes specific areas with groundwater aquifers showing increasing nitrate concentrations and where farmers will be required to prepare an N budget and identify mitigation practices depending on the estimated nitrate-N concentration in drainage water. The nutrient management compliance process is most advanced with Environment Waikato, which will require farms in the Lake Taupo catchment to be benchmarked based on their average N leaching, calculated using the OVERSEER® nutrient budget model, between 2001 and 2005. After 1 July 2007, farms will be capped at this average N leaching value and will be required to have an approved Nitrogen Management Plan (NMP) outlining farm characteristics and management practices which would enable them to farm within their N cap. The potential for trading N leaching credits between farms within the catchment is also proposed. A preliminary study of issues relating to implementation of NMPs on five farms in the catchment revealed that farmers differed greatly in the implications of benchmarking on the need for change to their current systems and in the likelihood of uptake of new management or mitigation practices on farm (Ledgard et al. 2006b). Farmers identified that the main requirement for an NMP process was flexibility in how they would meet their N cap. This meant that the N budget model used in the NMP process would need to be regularly updated by incorporating new research and mitigation practices as they are developed and proven. An important aspect of this nutrient management consent process, which internalises nutrient losses into the cost of farming, is that it alters the benefits and economics of incorporating mitigation options into the farm system. When nutrient emissions from farms are capped, intensification may only be possible in conjunction with the use of mitigation practices. For example, on a sheep and beef farm in the Lake Taupo catchment, an increase in productivity and profitability may be achievable by increasing stocking rate, lambing percentage and lamb finishing weights in conjunction with improved winter grazing management practices such as the use of stand-off pads to reduce N leaching from cattle on farm (Ledgard et al. 2006b).

From a farm productivity perspective, the increasing intensification on some farms using large amounts of supplementary feed and associated nutrient inputs can lead to large nutrient surpluses, and in the case of potassium can increase the risk of metabolic diseases (e.g., Ledgard et al. 2006b). In some countries, this intensification has led to large nutrient disequilibria with some regions producing feed-crops through high fertiliser use and transfer of this feed to other regions with intensive feedlot-type farms where effluent generates large nutrient surpluses and environmental problems (Sims 2005). From a nutrient efficiency perspective, the integration of effluent from the intensive animal production units back onto the forage crop areas to minimise fertiliser use is a preferred practice and by default will help to minimise many of the water quality problems related to the handling of effluent produced from confined animal feeding operations.

In the future, within pastoral farms in New Zealand there is potential for much greater spatial optimisation of nutrient management. Geographical Information System (GIS) technology means that targeted and precise application of fertilisers is feasible, which could ensure that they are not placed in critical source areas of nutrient loss, or that applications are placed according to plant needs. This could also apply to other inputs such as FDE. The automation of FDE scheduling and application using real-time daily weather records and computer simulation models or sensors that track storage pond volumes and soil moisture deficits to identify opportunities for irrigation is also likely. A GPS smart irrigator could identify its position in the landscape and customise applications to reflect soil chemical and physical characteristics. The automation of FDE management systems offers advantages in terms of reduced farm labour requirements and fewer opportunities for operator error. Technologies that reduce the volumes of FDE produced at the milking shed, such as covered yard holding areas that do not require daily wash-down, are also likely to greatly reduce the frequency of unwanted discharges of FDE from soil to water. While these systems may not currently appear economically attractive, with the likelihood of increased regulation impinging on farm nutrient management, such complex systems allow nutrient inputs to pastures to be placed with precision, according to need and in a manner that poses little risk of runoff to waterways.

Adoption of improved nutrient management practices and new technologies

A wide range of nutrient management practices have been identified to increase nutrient efficiency and reduce environmental emissions but their adoption by farmers has been variable. For example, while the benefits of effective land application of FDE have been well demonstrated, the extent of farmers breaching regional council requirements, as reported in compliance surveys for Canterbury and Waikato in 2006, has been surprisingly high. In a series of surveys of dairy farmers, Bewsell & Kaine (2005) found that decisions around FDE management were not driven by environmental considerations but by issues of on-farm practicality and fitting into the context of the farm (i.e., consideration of constraints such as labour, equipment availability, soils, topography, etc.). They found that the attitudes of farmers to sustainability and the environment had, at best, a limited role to play in their consideration of adoption of improved nutrient management practices.

Farmer education on environmental issues and nutrient management practices is important and needs to be provided in a way that identifies the on-farm context and implications to farm productivity and profitability. However, in some situations this will be insufficient to meet requirements for environmental protection (Bewsell & Kaine 2005). In that case, other policy instruments may be required such as regulations (as in the Taupo and Rotorua lake catchments) or incentives. For example, Environment Bay of Plenty are proposing the use of economic incentives including reduced rate payments for the adoption of best nutrient management practices.

In recent social research with sheep and beef farmers in the Taupo and Rotorua lake catchments, Botha & Parminter (2006) identified three main barriers to adoption of new technologies for reduced nutrient losses, being cost, compatibility with their current farm system (i.e., is the technology relevant to their farm, and can it be implemented?), and uncertainty of actual environmental benefits. In another farmer group study in the Taupo catchment, Dooley et al. (2005) applied Multiple Criteria Decision Making methodology to evaluate the potential for adoption of a wide range of alternative farm practices and found considerable variation between farmers in the preferred practices and criteria for decision making. Economic criteria only constituted roughly one-third of the total contribution to decision making, and other important factors included skill requirement, labour, lifestyle, risk and complexity. They concluded from the variability in criteria between farmers that a range of technologies or improved management practices will be required, which will need to be tailored to individual farmers. The future challenge for researchers, extension groups and policy organisations in the area of nutrient management will be to provide farmers with readily discernible information on the economic and environmental implications of a range of proven management or mitigation practices. This will require application of accepted nutrient management tools, consideration of specific environmental targets and alignment with farmers' goals and farming systems to encourage adoption of new technologies and improved nutrient management practices.

Additional environmental drivers

The future importance of effective nutrient management is likely to extend to encompass a wider consideration of whole-system resource (energy, water, nutrients) use efficiency and reduction in effects on a range of environmental indices, including greenhouse gas emissions. New Zealand is signatory to the Kyoto Protocol and is committed to reducing total greenhouse gas emissions to 1990 levels during the first commitment period from 2008–12. The latest national greenhouse gas inventory (Ministry for the Environment 2005) shows that nearly half (49.4%) of New Zealand's total greenhouse gas emissions in 2003 were from the agriculture sector and, of this, N_2O accounted for about one-third (34.9%), mostly emitted from animal excreta in grazed pastures. As climate change effects are starting to be recognised across the globe, it is likely that increasingly stringent greenhouse gas reduction targets will be put in place. With N_2O being such a substantial part of New Zealand's total greenhouse gas inventory, the pressure will be on for the development of additional nutrient management technologies which can reduce emissions from grazed pasture systems. A spin-off from the development of many mitigating technologies for N leaching is that they can also reduce both direct and indirect N_2O emissions. For example, dairy farm system studies showed that incorporation of winter stand-off pads or intensification using low-protein maize silage reduced whole-farm system emissions per kg milksolids for N leaching (Ledgard et al. 2006a) and N_2O emissions (Luo et al. 2006). The use of nitrification inhibitors is another example of a technology that delivers benefits of both reduced N leaching and reduced N_2O losses (Di & Cameron 2005; discussed in previous section).

SUMMARY

Developments within the last 10–15 years have seen the management of nutrients on New Zealand pastoral farms broaden from a focus on productivity responses to fertilisers to include consideration of the environmental impacts of a wider range of farming practices such as grazing, effluent and soil management. This metamorphosis has occurred in response to public concerns about the deterioration in water quality in some streams, rivers, lakes and aquifers, particularly within intensively-farmed catchments. The on-going intensification of farming practices has inevitably placed increasing pressure on water resources, with increased losses of N and P observed as land use has intensified. A range of mitigation practices and technologies has been developed to reduce these losses, although many incur some increase in net cost and complexity to the farming business. New Zealand land users are likely to be faced with a more restrictive future regulatory environment in which implementation of improved farm nutrient management practices will be mandatory. Implementation of particular mitigation practices will depend on a number of factors, in particular clearly defined environmental targets and penalties for non-compliant practices versus the relative cost-effectiveness of each mitigation technology, its compatibility with the current farm system, and whether regional councils develop whole catchment nutrient management and trading plans depending on their sensitivity to further nutrient enrichment. In addition to objectives for water quality, future developments in farm nutrient management approaches are likely to see greater emphasis placed on farming systems which meet targets for greenhouse gas emissions and energy efficiency and/or self-sufficiency.

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