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The influence of land use, soil properties and seasonal factors on contaminant accumulation and loss from farming systems to water

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The influence of land use and soil properties on contaminant accumulation and loss from farming systems

Report prepared for Environment Southland

October 2014

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1. Summary

This review documents our current understanding of how contaminants from agricultural land enter waterways in Southland, with a specific focus on consideration of temporal patterns in the accumulation of contaminant sources on different farming types and how soil type, landscape features and land management influence the transport pathways involved in the delivery of these contaminants to water. The document was commissioned by Environment Southland for the purposes of informing Council policy, science and extension staff of the potential effects of farming systems on water quality and to identify the options available for avoiding or reducing some of these effects. Published and un-published literature, modelling tools and expert opinion are used to link the build-up and transport of nitrogen (N), phosphorus (P), sediment and faecal microorganisms to management practices on different farming systems. An assessment of the cost and effectiveness of some mitigation options that are currently available to reduce the impacts of farming on water quality is also provided. The report concludes with a list of suggested future research needs that will help fill some key knowledge gaps.

Local and national research shows that subsurface drainage is an important flow pathway involved in the transfer of water and contaminants from agricultural land to water (section 3). Artificial (mole-pipe) agricultural drainage systems are widespread in Southland and therefore particularly important contributors of nitrogen (N), phosphorus (P), sediment and faecal microorganisms loads to water bodies. Imperfectly or poorly-drained soils and soils with low infiltration rates (often as a consequence of animal treading damage) can generate significant volumes of surface runoff. Whilst this flow pathway does typically not remove a high proportion of the surplus rainfall received at a site, it is relatively enriched in P, sediment and faecal microorganisms. As documented in an earlier review (Monaghan et al. 2009a), literature indicates that N losses to water from dairy pastures are greater than from sheep or deer, although the actual amounts of N leached vary considerably depending on soil, climate and management factors. Recent and on-going research also shows that grazed winter forage crops can be important sources of N losses to water, particularly on free-draining soil types, and of P and sediment losses via surface runoff from gullies and swales.

Seasonal patterns of contaminant accumulation and loss are reviewed in section 4. There is a clear seasonal pattern of contaminant displacement from farms in Southland that is driven by climate variables that influence transport factors, namely surplus rainfall

and temperature. These seasonal losses have been documented in experimental studies conducted in Southland and Otago and clearly show that drainage and surface runoff events in late autumn, winter and early spring ***transport*** most of the contaminants discharged from farms to water. There is also a distinct seasonal pattern of N ***accumulation*** in soils over summer and autumn that governs the magnitude of potential N losses to water from Southland pastures (Figures 4.1 – 4.3). These seasonal patterns of accumulation and loss are important aspects that need to be considered when modelling and evaluating strategies that can potentially reduce losses. Section 5 categorises some of these source-management responses into options that focus on (i) optimising the timing of fertiliser and effluent applications to soils, (ii) reducing the amounts of excreta deposited to soil and pasture at critical times of the year, (iii) managing soils to protect soil structure, thus minimising surface runoff, (iv) manipulating animal diets to reduce N excretion, (v) using pasture species that have greater winter-activity and N uptake, and (vi) applying additives to soil to retain N or P.

Appended to this report are tables of practices that have been shown to improve the quality of water leaving dairy (Appendix I) or dry stock (Appendix II) farms. These are based on the MENU of practices (http://www.waikatoregion.govt.nz/menus) developed by Waikato Regional Council but have been re-freshed to reflect cost and effectiveness metrics for Southland farms. These assessments consider the flow pathway targeted by each mitigation and have been scaled to a whole-farm system equivalent.

2. Scope of Report

AgResearch has been asked to prepare a review that documents our current understanding of how contaminants enter waterways in Southland, with a specific focus on (i) consideration of temporal patterns in the accumulation of contaminant sources on different farming types, and (ii) consideration of how soil type and landscape features influence the transport pathways involved in the delivery of these contaminants to water. Accordingly, here we:

- Provide an overview of some of the basic concepts of source and transport factors that influence or determine how contaminants are delivered from farmland to water,
- Review the scientific literature and use modelling tools and expert opinion, where necessary, to link the build-up of N, P, and *E. coli* in soil to corresponding losses from different farming systems,

- Provide an assessment of the cost and effectiveness of some mitigation options that are currently available to reduce the impacts of farming on water quality,
- Identify remaining knowledge gaps where future research is needed.

Although the context of this report is the Southland province, research information from other relevant parts of the country is drawn upon where necessary to fill some key knowledge gaps.

This report is focused solely on agricultural non-point source discharges and does not address other potential sources of contaminants in the rural environment e.g. industrial discharges (e.g. whey), septic tanks, large colonies of wildfowl, etc. Nor does it consider the role of bank erosion as a source of P and sediment from agricultural landscapes. Particular attention is paid to research results obtained from grazing system studies that, as near as possible, replicate management conditions similar to those found on commercial farms. Emphasis is also placed on the influence of seasonal factors (and their management, where possible) that determine the potential for contaminant loss from farms to water.

List of abbreviations and terms used:

Runoff – the term used to describe that part of precipitation which ends up in streams or lakes (i.e. the combined flow of surface water, subsurface drainage and groundwater pathways, but not deep drainage).

Overland flow or *surface runoff* - that part of precipitation which flows overland to streams or directly to lakes.

Saturation excess runoff – surface runoff caused by the flow falling onto an already-saturated soil.

Infiltration excess runoff – surface runoff that occurs when rainfall intensity (mm/hr) exceeds soil infiltration rate. Pugging and compaction due to animal treading can lower soil infiltration rates to levels where this type of runoff process is likely.

Subsurface flow (or drainage) - that part of precipitation which infiltrates the soil and moves to streams or lakes as ephemeral, shallow, perched, mole-pipe or ground water flow.

Mole-pipe drainage - that part of precipitation which infiltrates the soil then moves through an artificial subsurface network of mole and pipe drains before draining to ditches or streams.

Critical Source Areas - small areas within a catchment where many of the less mobile stream pollutants, such as P, sediment, ammonium-N and faecal bacteria, are sourced and transported.

CSA – critical source area

DCD – dicyandiamide (a nitrification inhibitor)

E. coli – *Escherichia coli*, often used as an indicator of faecal pollution.

FMOs – faecal microorganisms

LUC – Land Use Capability

N – nitrogen; *TN* – total N; *DON* – dissolved organic N

P – phosphorus; *TP* – total phosphorus; *DRP* – dissolved reactive phosphorus

SS – suspended sediment; *TSS* – total SS; *VSS* – Volatile SS

FDE (Farm Dairy Effluent) - wash-down water collected from the dairy farm milking parlour and holding yard.

3. Source and transport factors governing contaminant losses to water from farming systems

3.1 Pathways of contaminant transport

For clarity, described below are the key pathways and terms used to describe the transfer of pollutants from land to water (and are shown pictorially in Figure 3.1):

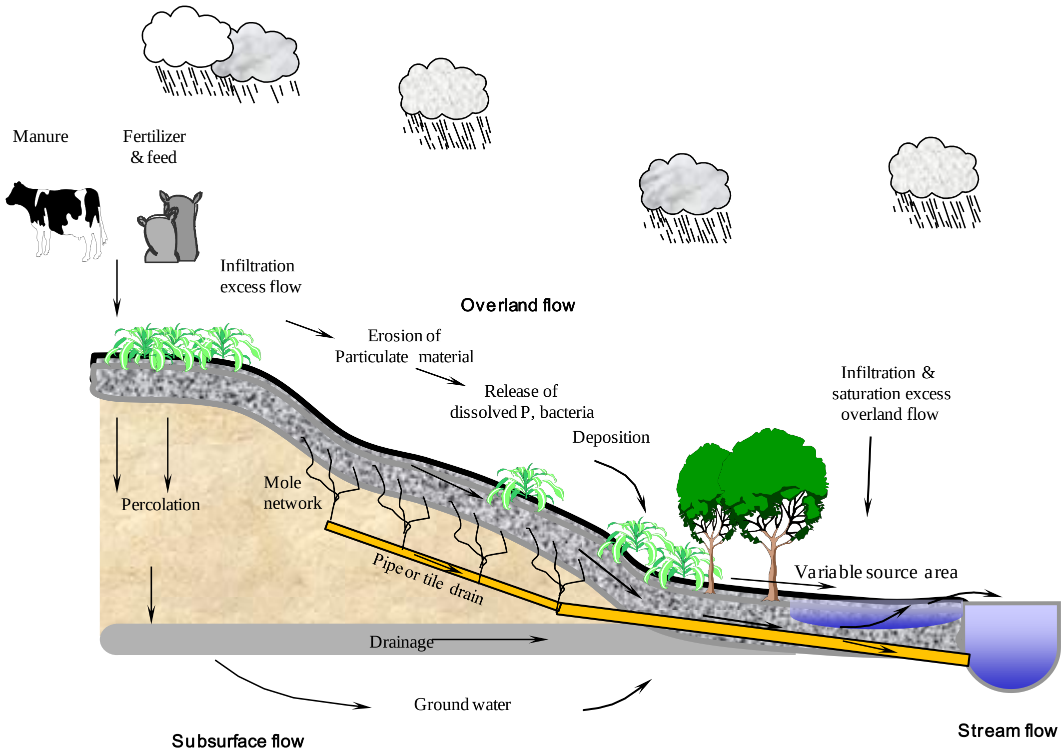


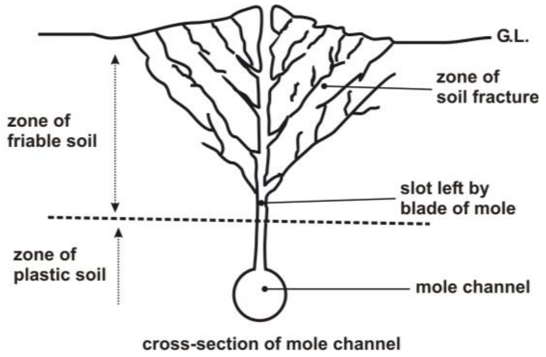
Figure 3.1. Conceptual diagram of processes that transport pollutants from the landscape to surface water (adapted from McDowell et al. 2004).

Percolation (or deep drainage), artificial drainage and surface runoff are the 3 main pathways involved in the transfer of pollutants from farms to water. The fate of excess water falling on a soil depends on a number of factors including the quantity, duration and intensity of precipitation, soil infiltration rate as determined by the volume of continuous macropores, the permeability of soil layers and the drainage coefficient of any artificial drainage system that may be present. Surface runoff from hard standing areas such as yards and lanes represent an additional pathway that can impact on water quality.

Mole-pipe drainage systems

With approximately 2 million ha of poorly- or imperfectly-drained soils in New Zealand, artificial subsurface drainage systems have been an important tool in the development of New Zealand's productive agricultural land (Bowler 1980). Fragic Pallic soils, like the Pukemutu and Waikoikoi silt loams, have a fragipan that is the determinative factor in the drainage process. A fragipan is a subsoil horizon which has a high bulk density and which is relatively hard when dry but softens when wet. Fragipans usually impede the downward movement of water and subsurface drainage systems of clay tiles or slottedplastic pipe are therefore usually a prerequisite to productive management of soils with these types of impermeable or slowly-permeable subsoil layers. The increased or quicker drainage that artificial drainage systems offer speeds the lowering of the water table and returns the soil to field capacity water content. The immediate result is that topsoil strength returns sooner after heavy rain, which creates more ‘safe’ autumn/winter/spring grazing days (Horne 1985). In addition, the topsoil is more rapidly returned to an aerobic state, which improves plant growth and general soil health.

In fine textured soils, where the natural drainage is very slow and a temporary perched water table exists in winter, the installation of a subsurface drainage system typically includes a subsurface network of mole drains. Moling is an operation in which a channel is extruded in the subsoil using a mole plough. The passage of the mole plough through the soil causes the soil to heave slightly, which results in a network of cracks that provide pathways for water to move rapidly through the soil and into the mole channel (Figure 3.2A). Mole channels are pulled above and generally at right angles to the collector pipe drains (Figure 3.2B). The pipes provide an outlet for water from the moles. The preferential movement of drainage water from surface soil through cracks to the mole drains transforms both the perched water and some surface runoff flow (generated in storm events) into accelerated subsurface drainage. There is a risk that this preferential flow of drainage bypasses the soil matrix allowing less time for the absorption and retention of contaminants in the soil. Past and current research indicates that subsurface drainage systems can be conduits for the rapid movement of nutrients (N & P), sediment and faecal organisms in drainage from grazed pastures to surface water. On artificially drained soils, mismanagement of land treatment systems for dairy farm effluent creates the potential for large concentrations of contaminants to move by this route. This has been documented for a range of sites in New Zealand (Houlbrooke et al. 2004a,b; Monaghan & Smith 2004; Monaghan et al. 2010). This research has in turn led to the development of guidelines and policies to minimise these transfers (e.g. Environment Southland 2010b). Considerable international research has also shown the importance of artificial drainage systems (such as tile drainage) as conduits for transfer of surplus rainfall and solutes from soil to water (e.g. Skaggs & van Schilfgaarde 1999; Deelstra et al. 2014).



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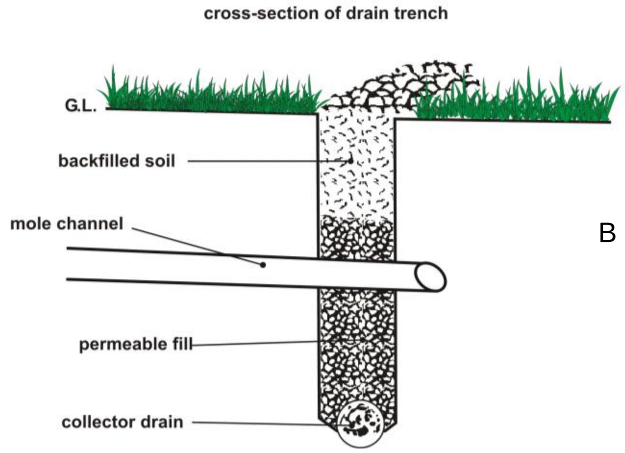

B

Figure 3.2. Diagram showing (A) the fissure network created by the mole plough and the cracks through which water flows preferentially, and (B) the position of a mole drain in soil above a pipe drain.

Surface runoff (also referred to as overland flow)

Surface runoff, or overland flow, may occur if soils become saturated (termed “saturation excess” surface runoff) or if rainfall intensities exceed soil infiltration rates (“infiltration excess” surface runoff). Saturation excess surface runoff typically occurs in low-lying parts of the landscape or areas that have a perched water table. Animal treading on wet soils whilst grazing pastures tends to damage soil structure and thus reduce soil infiltration rates. Consequently, it is not uncommon for rainfall intensity to exceed the infiltration rate, resulting in infiltration excess surface runoff. Field studies in Southland monitoring surface runoff from pastures grazed by cattle have shown that surface runoff is typically enriched in P (dissolved and particulate forms), sediment, faecal bacteria and ammonium-N, but has relatively little nitrate-N. This enriched surface flow is caused by the interaction of surface runoff with animal excreta and the fertile fine sediments located on the soil surface, particularly if soils have been damaged by hoof treading.

Although surface runoff typically has high concentrations of water contaminants, yields will depend on the quantities of surface runoff generated:

- At the Edendale grazing study (1996 – 1999), surface runoff from the artificially-drained soil types present accounted for less than 2% of the surplus rainfall received. This small volume of surface flow delivered less than 10 and 2% of the total P and inorganic N lost, respectively, in combined surface runoff and subsurface drainage flows (Smith & Monaghan 2003; Monaghan et al. 2005).
- At the Tussock Creek grazing study (unpublished results), surface runoff from the poorly- and artificially-drained Pukemutu soil accounted for 24% of the surplus rainfall received in 2002 and 2003. This surface flow delivered 56, 17 and 66% of the total P, inorganic N and sediment lost, respectively, in combined surface runoff and subsurface drainage flows.

A challenging aspect of modelling water and contaminant losses from land is defining the areas that contribute surface runoff flows to streams. The dominance of these areas, which are typically relatively small and therefore often termed critical source areas (CSAs), is dependent upon many factors, including soil type, topography, management (e.g., inputs of fertiliser and manure) and transport processes that are in turn dependent upon environmental and hydrological conditions. The interaction between these factors is complex and varies spatially and temporally. However, in general, CSAs are defined by a high concentration of pollutant available to flow and a high potential for flow, equating to a high potential for loss. Critical source areas are commonly located near stream channels or in low infiltration areas and gullies that are connected to the stream channel (McDowell and Srinivasan 2009). The development of spatially-explicit farm scale modelling tools such as the Mitigator model (under development through funding by Ballance AgriNutrients) will aid the process of identifying where in the landscape surface runoff is likely to occur and the management practices that may help to reduce contaminant losses in this flow.

Seepage and deep percolation

Sometimes, on sloping or undulating landforms, there can be significant seepage into surface waters from the surrounding soil. Some of this water may have originated as surface runoff or drainage in other parts of the landscape but it ultimately enters water courses more gradually as seepage from wet areas or underlying gravels. Seepage was a significant contribution to water flow in the small catchments studied by Sharpley (1977). For imperfectly or poorly drained soils with subsoils that are somewhat permeable, deep percolation can be a significant contributor to water loss from the profile. In these cases, the installation of mole-pipe drainage systems does not have the same marked effect on either the water table height or the quantity of surface runoff as it would for the soils discussed above with impermeable subsoils. For more permeable soils, it is the deep percolation - a relatively protracted process – which is the dominant pathway for the movement of surplus water from the soil profile to ground or surface waters.

In contrast to overland flow, deep percolation (or leaching) is usually the dominant pathway involved in the transfer of mobile pollutants from soil to water. Mobile solutes such as nitrate readily diffuse through soil and are therefore easily transported in surplus water percolating down the soil profile to below plant rooting depth, which for pastures is typically about 500 to 600 mm depth. The potential for displacement of soil nitrate-N by passing drainage water can be simply illustrated by considering drainage depth and the proportion of the soil volume participating in water transport. Assuming a surplus winter rainfall depth of 300 mm (typical for many parts of Southland), a soil total porosity of 50% and a water-filled pore space at field capacity of 80%, then the depth of solute displacement is calculated as 750 mm (= 300/(0.5*80%)). Although diffusion-dispersion processes and preferential flow pathways will modify this potential displacement, it serves to illustrate how far mobile solutes such as nitrate-N can move down a soil profile.

3.2 Key agricultural sources of water contaminants

The main contaminants of concern regarding diffusion pollution of water in NZ are nitrogen (N, in the ammonium- and nitrate-N forms), phosphorus, sediment and faecal bacteria. Although the potential impacts of these water pollutants are well known and described (e.g. Environment Southland 2010a), their sources and their management within an agricultural context are less well understood. Below is a brief summary of some of the basic scientific principles that have been shown to influence the sources of N, P sediment and faecal bacteria that may accumulate on farms and be potentially available for loss to water (as summarised in Figure 3.3).

Nitrogen

Losses of N to surrounding water mainly occur during autumn and winter when evaporation is low, precipitation is high and plant uptake of N is small. Considerable research into N flows within grazed pasture systems over the past three decades clearly shows that the amount of N excreted by animals, and in particular urine N, is usually the most important determinant of N losses (including leaching to deep drainage or runoff to stream and gaseous losses) (e.g., Ledgard 2001; Di and 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 main effect of fertiliser N use on N cycling efficiency in grazed pastures is therefore indirect, with N fertiliser inputs, which allow for an increase in pasture production and animal stocking rate, also increasing urine N excretion. A variety of studies have quantified the potential for N in urine to leach through pasture soils. In an experiment using ^{15}N-labelled urine, Fraser et al. (1994) estimated that 8% of urine N leached below 1.2 m depth after one year. The timing of urine deposition strongly influences the potential for urine N to leach through pasture soils, with large losses typically observed for urine deposited shortly prior to the onset of drainage. For clover-based dairy pastures, nitrogen fertiliser is generally not a major direct source of N loss, as it is applied in relatively low rates and used strategically to supplement N supply from biological N fixation. Direct leaching of fertiliser N is usually low if the rates and timing of applications are correct, (Ledgard et al. 1999; Monaghan et al. 2005; Di and Cameron 2002b). Soil type exerts a considerable influence on the amount of nitrate N leached from the soil profile, with greater losses observed for light free-draining soils compared to heavier-textured and/or poorly drained soils where a proportionally greater amount of soil nitrate is removed via denitrification processes. A summary of measured N losses in drainage and surface runoff from grazed experimental sites in Southland and Otago is shown in Table 3.1. With the exception of the Edendale site, where a mixed herd of non-lactating dairy and beef cattle was used to graze the experimental plots, these measured losses give an indication of the typical amounts of N lost to water from pastures and winter forage crops grazed by dairy cows.

The amount of N excreted by grazing cows is tied to the amount of N consumed, which in turn is broadly related to animal stocking rate. 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. However, there is a wide variation in external N inputs and per-cow production and intake of feed-N (e.g. c. 3-fold) on dairy farms. Because of this and other soil, climate and management factors, stocking rates (animals/ha) are therefore a crude proxy for the magnitude of N loss for a dairy farm. Large losses of nitrate-N in drainage may occur from areas used for forage crop grazing during winter when pasture growth rates are usually low (Monaghan et al. 2013; Shepherd et al. 2012; Smith et al. 2012). The greater losses from the wintering part of the system arise due to (i) relatively large amounts of mineral N remaining in the soil in late autumn following pasture cultivation and forage crop establishment the preceding spring, and (ii) the deposition of much excretal N onto the grazed forage crop during winter when plant uptake is correspondingly low or absent.

In recent times dairy grazing systems have evolved that utilise housing and high inputs of supplemental feed to manage dairy herds through winter months when pasture growth rates are low, soil pugging is often a concern and nutrient losses in drainage are greatest. These systems contrast with those typically found in warm-temperate regions of New Zealand where year round grazing is commonly practised because of a better match between herd feed demand and pasture growth rates. Whilst the use of animal housing in southern New Zealand has probably been driven by many additional considerations beyond the perceived environmental benefits mentioned above, these types of systems do confer advantages that, if well managed, can potentially increase the efficiency of N cycling within a farm system. One of these advantages is the ability to capture urinary N deposited during times of the year when the risk of loss in drainage is greatest. This seasonal pattern of risk and the management strategy to reduce this risk is described in more detail in sections 4.1 and 5.2, respectively.

Recent research in Southland and Otago indicates that paddocks used for summer-grazed forage crops may also have a relatively high potential for N leaching over the following winter (Chrystal et al. 2012 and unpublished results). The higher N leaching potential observed for the grazed summer forage crops reflects the asynchrony between plant demand for N and the potentially large amounts of mineral N that may accumulate in the soil during autumn. In the case of summer-grazed forage crops, there is a relatively narrow window of opportunity for the uptake of mineralised and urinary N by the re-sown pasture before winter rains arrive. Re-sowing these pastures as early as possible will help to minimise the amounts of unused mineral N remaining in the soil in late autumn, and thus potential leaching risk.

Table 3.1. Measured mean annual N losses in drainage and overland flow from grazed dairy study sites in Southland and Otago.

Site	N fertiliser input or grazing treatment	Stocking rate cows ha^{-1}	Drainage				Overland flow	Total dissolved N loss to water	Reference
			NO_3^--N leached kg N ha^{-1} $year^{-1}$	NO_3^--N conc. mg N L^{-1}	NH_4^+-N leached kg N ha^{-1} $year^{-1}$	DON[a] leached kg N ha^{-1} $year^{-1}$	NH_4^+ + NO_3^- kg N ha^{-1} $year^{-1}$	kg N ha^{-1} $year^{-1}$	
A. Pastures									
Edendale (1996-1999)	Nil fertiliser N	2.4	30	8.3	1.2	3	0.05	34	Monaghan et al. (2005) Smith & Monaghan (2003)
	100 kg N/ha/yr	2.8	34	9.2	1.5	3			
	200 kg N/ha/yr	3.0	46	12.5	1.7	5			
	400 kg N/ha/yr	3.3	56	15.4	1.7	5	0.08	63	
Tussock Creek (2001-2003)	Control	2.7	17	4.8	0.7	2.5	1.7	22	de Klein et al. (2006) & un-published data
	Restricted autumn grazing	2.7	11	3.1	0.4	2.5	-	-	
(2004-2013)	Control	2.6	14	3.8	0.9	-	-	-	Monaghan et al. (2009b) & unpublished data
	+DCD	2.6	9	2.4	0.9	-	-	-	
Kelso (2000-2007)	85 kg N/ha/yr	3.0	25	13.5	0.2	-	-	-	Monaghan & Smith (2004) & unpublished data
B. Winter forage (kale) crop									
Five Rivers (2009-2011)	~ 100 kg N $ha^{-1}yr^{-1}$	1330[b]	57	19.7	0.3	21	-	78	Smith et al. (2012)
Tussock Creek (2006-2008)	~100 kg N $ha^{-1}yr^{-1}$	1200[b]	52	10.8	0.3	-	-	-	Monaghan et al. (2013)

[a]dissolved organic N; [b]calculated grazing density (24 h) for a 13 T DM/ha kale crop and assuming 90% utilisation.

Arable agriculture has been shown to potentially have a high risk of N loss due to the difficulty of synchronising plant N uptake with N release from soil and crop residues. Much of this research has been undertaken in the US and Western Europe where large parts of the agricultural landscape may undergo frequent and intensive cultivation as part of crop establishment. Intensive cultivation accelerates the mineralisation of soil organic N and also requires that the land remains without growing plant cover for at least some of the time. If these periods coincide with times of drainage, there is large potential for the leaching of N due to the substantial amounts of mineral N that may accumulate in the soil as a result of mineralization and/or unused fertiliser N. Research undertaken in New Zealand (Francis et al. 1992; Francis et al. 1995) has also documented N leaching losses from arable cropping systems and highlights the importance of ensuring that the length of fallow period between cultivation and the onset of leaching is minimized as much as possible. Delaying cultivation of pasture until spring has been shown to reduce N leaching risk without causing any significant decrease in wheat yield or N uptake by the following crop. These trials and modelling analyses indicate (i) very large losses, sometimes in excess of 100 kg N $ha^{-1}yr^{-1}$, may occur from paddocks used for cropping, and (ii) the magnitude of N leaching loss is very dependent on climate, management and soil risk factors (e.g. Lilburne et al. 2003). The type of crop grown and presence of winter cover crops can also have a major influence on the amount of nitrate-N that is leached, due mainly to variations between crops in their synchrony of N demand with supply from the soil or fertiliser (Francis 1995; FAR 2008). Some of the key management practices that have been shown to minimise N leaching are the use of cover crops, ensuring fertilization rates are matched to crop demand (and applications avoid periods of high leaching), avoiding late summer/early autumn cultivation of pastures, using minimum tillage or no-till techniques and the application of the nitrification inhibitor DCD (Francis, 1995; Francis et al. 1998; FAR 2008; Myrbeck and Stenberg 2014; Syswerda et al. 2012). These practices are particularly recommended for cropping on shallow soils.

Phosphorus and sediment

Sources of P losses from farms tend to vary more than for N. Phosphorus losses depend heavily on spatial factors and the type of management practices employed on farm, such as how effluent or manures are handled, and the degree of protection of streams banks and beds from erosion and animal treading. Phosphorus losses from intensively grazed pastures arise from dissolution and loss of particulate material from the soil, washing-off of P from recently grazed pasture plants, dung deposits and fertiliser additions. With the exception of the latter, all are influenced by the action of grazing, whether it is the ripping of pasture plants or the influence of treading on soil erosion and surface runoff potential. Clover-based pasture dairy systems typically have relatively large P fertiliser usage to maintain adequate soil P fertility for optimum clover growth. Of the P recycled via the grazing cow, most is excreted in dung and in a soluble form (Kleinman et al. 2005). Dung therefore represents a concentrated form of readily available P that can have a large impact on surface water quality if voided directly into water. Stock access to streams, effluent pond treatment systems, and effluent/manure applications to land are therefore key land management practices that can potentially contribute substantially to farm P losses (Byers et al. 2005; Hickey et al. 1989).

Allowing stock access to streams has historically been one means of providing pastured animals with drinking water and comfort during hot weather, but is now recognized as a poor management practice from the standpoint of impacts on water quality and human and animal health. Practices such as excluding livestock from riparian areas, providing alternative sources of water and shade, and selection of feeding sites can have a profound effect on the environmental fate of nutrients from the excreta of pastured animals. McDowell and Wilcock (2007) monitored P losses from a 2,100 ha catchment in New Zealand containing dairy farms with seasonal milking. They observed elevated concentrations of total P in stream flow that were strongly correlated with stream sediment concentrations, attributing the sediment to trampling and destabilization of the stream bank by stock, as well as to other riparian management factors such as removal of riparian trees that stabilize banks. Elsewhere, James et al. (2007) estimated that 2,800 kg of P was defecated directly into pasture streams by dairy cattle every year in a 1,200 km^2 catchment in the northeastern USA with predominantly farms of low intensity grazing. An additional 5,600 kg P was deposited within a 10 m riparian area. Across the catchment, direct deposition of dung P into streams was equivalent to roughly 10% of the annual P loadings attributed to all agricultural sources.

Overland flow processes can also make a large contribution to the total P lost from dairy farms, unlike N. Although overland flow volumes are usually small relative to the volumes of water discharged in subsurface drainage, the entrainment of soil and dung P in this flow makes it a concentrated source of P and other potential stream contaminants such as ammonium-N and faecal micro-organisms. Despite much research on P loss from agricultural soils, the contributions from overland flow sources are still difficult to define because of problems associated with spatial and temporal variability, making sampling and measurement of flows under field conditions very difficult. Current understanding suggests that near stream areas are important sources of overland flow, as are areas of land underlain by artificial drainage systems, which act as direct conduits between soil and stream. These artificial subsurface drainage systems have been shown to act as important sources of P and sediment, presumably due to the entrainment of particulate and dissolved P as water moves through the macropores and fissures to tile or pipe drains (Chapman et al. 2005; Haygarth et al. 1998; Hooda et al. 1999; Monaghan et al. 2005; Sharpley & Syers 1979). Measurements of subsurface drainage and surface runoff flows at the Tussock Creek experimental site indicate that the mole-pipe network delivered 44 and 34% of the total P and sediment lost to water during 2002 and 2003 (Monaghan et al. 2014).

The issue of elevated levels of soil P is widely recognised as a significant source and unnecessary risk factor in P loss from farmland (e.g. Heckrath et al. 1995; McDowell et al. 2003a). The restoration of high P soils to levels that more closely match agronomic requirements, particularly within high risk areas, is an important measure that can reduce potential transfers of P from soil to water (Haygarth & Jarvis 1999). Direct losses of applied fertiliser P are another potentially important source of farm P losses. The greatest risks are when soluble forms of P are applied shortly before overland flow events, or if P fertiliser is inadvertently directly spread onto streams or wetlands. However, improved spreading technology and practices on most dairy farms now mean that accidental P applications to streams are usually small. McDowell et al. (2003b) showed how the potential for P losses in either overland flow or drainage from soils fertilised with superphosphate decreased exponentially with time post-application, so that after 30-60 days the concentration of P lost in runoff from superphosphate-treated plots equalled that of non-treated plots.

Losses of P from farmed landscapes are often closely linked to sediment losses because of the P attached to soil colloids and particulate material entrained in surface (and, to a lesser degree, subsurface) flow. Managements that target sediment and erosion control therefore also help to mitigate P loss. Field studies in Southland and Otago have documented yields of sediment in flow pathways from both pastures and grazed winter forage crops. Some important findings from these (and other) studies are:

- Subsurface drainage can be an important source of direct sediment discharges to waterways. Mean annual yields (2001 to 2003) of TSS and VSS in subsurface drainage from the Tussock Creek experimental site were 52 and 21 kg $ha^{-1}yr^{-1}$, respectively (Monaghan et al. 2014). Modelling of the Oteramkia catchment in eastern Southland suggested that subsurface drainage fluxes of this magnitude could potentially account for a significant proportion of the total catchment sediment yield, although the authors noted that there was much uncertainty attached to the sources and fluxes considered (Thorrold et al. 1998).

- The concentration of sediment in surface runoff is typically high relative to that found in subsurface drainage. This pathway can therefore be important for sediment transfers if surface runoff volumes are large. As noted above, surface runoff at the Tussock Creek experimental site accounted for 24% of the surplus rainfall received but delivered 66% of the sediment lost in combined surface runoff and subsurface drainage flows.
- The potential for sediment losses in surface runoff from grazed winter forage crops is very high. McDowell & Houlbrooke (2009) observed that sediment losses from cattle-grazed winter forage crops were greater than from sheep-grazed winter forage cropland, which in turn was greater than from sheep-grazed pastures. Orchiston et al. (2013) also reported very high losses of sediment in surface runoff from a paired catchment study following cattle grazing of a winter forage crop.
- Sediment losses from land grazed by deer can be particularly high (McDowell 2009a,b). This reflects the large amounts of erosion that deer can potentially cause, particularly when allowed access to streams and wet areas. Losses from winter forage crops grazed by deer can be especially high (McDowell & Stevens 2008) and probably exacerbated by the lack of ground cover and poor surface soil condition due to hoof treading.
- In a review of P and sediment losses from NZ hill country landscapes that were predominantly grazed by sheep-beef stock classes, McDowell & Wilcock (2008) documented annual P losses ranging from 0.1 to 2.1 kg $ha^{-1}yr^{-1}$ and annual sediment losses ranging from 22 to 2,740 kg $ha^{-1}yr^{-1}$; mean annual losses were 1 and 1,000 kg $ha^{-1}yr^{-1}$, respectively. Given the reduced intensity of land use on such landscapes, these losses are relatively high and reflect the greater potential for surface runoff due to increased slope.
- Few studies have been reported which document P and sediment losses from arable cropping systems in New Zealand. International literature suggests that this land use has the potential to deliver relatively large amounts of P and sediment to surface waters (e.g. Ulen et al. 2012). Soil tillage operations are important for the erosion process and plot scale measurements show that total P losses are often closely related to soil losses. Mitigation measures therefore aim to minimise or avoid tillage (particularly during autumn) and to include grass leys in crop rotations on landscapes that are prone to erosion (Schoumans et al. 2014). Data presented in Dymond (2010) indicates that measured erosion rates

from NZ cropping systems (presumably arable and horticultural) range from about 0.5 to 50 T $ha^{-1}yr^{-1}$, with an average of about 20 T $ha^{-1}yr^{-1}$. These losses are particularly large even by international standards and most probably reflect the potential for large losses from vegetable cropping systems that have not been well managed for erosion risk.

Faecal micro-organisms (FMOs)

The transfer of FMOs from land to water is an area of growing importance in the context of diffuse agricultural pollution. Contamination of water with enteric pathogens such as *Escherichia coli* (*E. coli*) 0157, *Salmonella spp*., *Campylobacter spp* and *Cryptosporidium parvum* has come under the spotlight in recent times as linkages between agricultural practices and pathogen dissemination in the wider community have been considered (Oliver et al. 2005a). Knowledge of the sources and pathways of FMO transfer from farmland to water is poor relative to the understandings for nutrients, particularly N. Dung can be a concentrated source of these organisms, and many of the land management practices which decrease P losses may therefore also decrease transfers of FMOs to waterways. Preliminary studies have identified that, like P, surface and subsurface flow pathways are important sources of FMOs (Oliver et al. 2005b). Unfortunately, we know little about the survival rates of FMOs on pasture and in soil, and of their mobility in overland and subsurface flows. Consequently, it is difficult to identify additional opportunities for management interventions that can decrease land-water transfers. Muirhead et al. (2006) demonstrated that *E. coli* bacteria are transported in overland flow as single cells rather than as flocs or attached to sediments, behaving similarly to solutes and negating opportunities for removal via settling or filtration by vegetation. Whilst it is difficult to envisage technological solutions that can reduce the survival or mobility of FMOs deposited in the field, opportunities do exist for treating and disinfecting manures and effluents collected in animal housing units or in the dairy yard (e.g. Craggs et al. 2004).

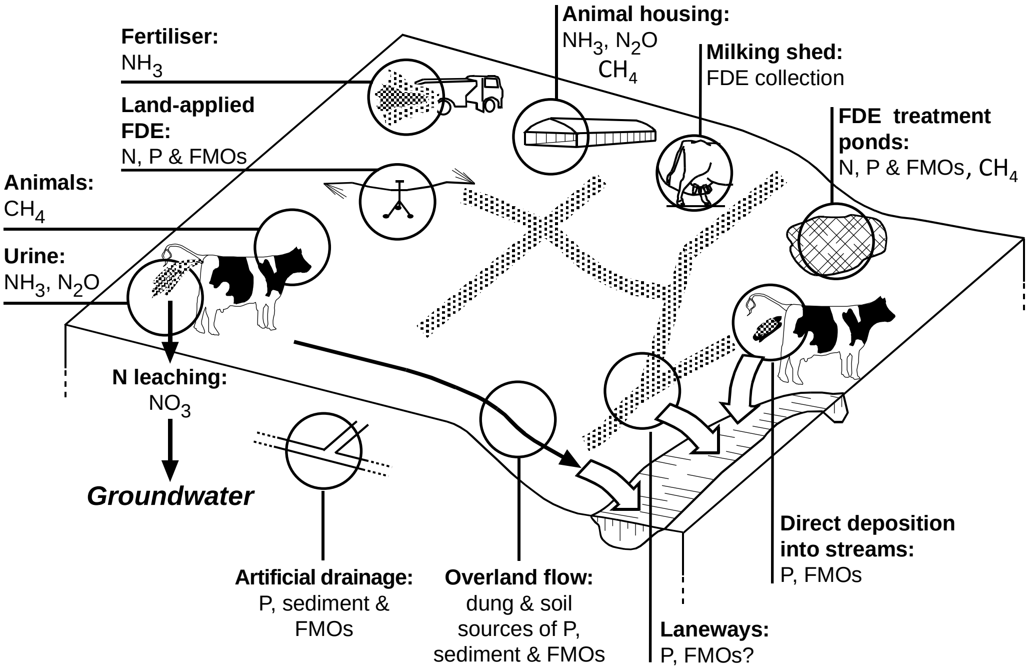

Figure 3.3. Schematic representation of the main sources of contaminant loss from grazed dairy farms.

4. Seasonal patterns of contaminant accumulation and loss

There is a clear seasonal pattern of contaminant displacement from farms in Southland & Otago that is driven by climate variables that influence transport factors, namely surplus rainfall and temperature. These seasonal losses have been documented in studies by Monaghan et al. (2005, 2009a, 2014), McDowell & Wilcock (2004) & Curran Cournane et al. (2011) and clearly show that drainage and surface runoff events in late autumn, winter and early spring transport most of the contaminants discharged from farms to water. Less evident are the seasonal patterns of how these accumulate in soils. This seasonal effect is particularly important for N and needs to be considered in modelling approaches that estimate N losses to water. These seasonal patterns and their implications for mitigation measures are described below.

4.1 Nitrogen

As discussed in section 3.2, animal urine patches contain large amounts of N, most of which becomes potentially available for plant uptake within 1 or 2 days of urine deposition. Unfortunately, the rates of N deposited in a urine patch usually far exceed immediate plant requirements. This imbalance between N supply and plant demand can be illustrated by comparing the equivalent N loading in a typical cow urine patch of 500 to 1000 kg N ha^{-1} with a typical pasture N demand of between 1 to 4 kg N ha^{-1} d^{-1}. The consequence of this imbalance is that much of the surplus N in the urine patch remains available for other N transformation and loss processes that occur in the soil, including leaching. Urinary N deposited close to winter is thus at greater risk of leaching loss than urinary N deposited many months before the arrival of surplus autumn and winter rainfalls. This principle has been illustrated in research by, amongst others, Cuttle & Bourne (1993) and Shepherd et al. (2011) and has guided the monthly modelling approach now incorporated into the Overseer® Nutrient Budgeting model (Cichota et al. 2014). Overseer® model estimates of monthly urinary N returns and N leaching are shown for some case study Southland farms in Figures 4.1 to 4.4 to further illustrate these principles. Measurements at the Tussock Creek experimental site have been used to guide and calibrate the model to ensure it performs adequately for Southland conditions. This local monitoring has shown a distinct pattern of N leaching from these mole-pipe drained silt loams soils, with most of the N displaced in early winter (May and June) drainage and only small amounts lost in spring drainage (Figure 4.5). This pattern of N elution from artificially-drained soil is similar to that observed by Christensen et al. (2011; shown in Figure 4.6) and reported by others (e.g. Heng et al. 1991; Scholefield et al. 1993). Preliminary monitoring at the Southland Dairy Demonstration Farm has also shown how most of the observed N leaching from the poorly drained soils that cover much of the farm occurred between May and the end of July 2012 (Cameron et al. 2014).

The first important principle evident in all of these Figures is the pattern of drainage displacement, with the bulk of annual drainage occurring from May until September. This agrees with observations from local studies that have monitored drain flows from mole-pipe drained soils (Monaghan and others) and is mainly driven by low evapotranspiration (ET) during this period. A second important principle evident in Figures 4.1 and 4.2 is the changing seasonal pattern of N returns to sheep pastures: the largest amounts of urinary N tend to be deposited in spring and lesser amounts in late autumn and winter. Lower urinary N returns to pasture during winter reflect the corresponding large returns to winter forage crop paddocks that contain relatively large amounts of standing feed (typically 10 to 15 T DM/ha) and thus dietary N, much of which is excreted by the non-lactating animal. A third and perhaps most important feature evident in Figures 4.1 to 4.3 is the greater proportional loss of urinary N deposited to pastures in autumn compared to that deposited in summer, or the preceding spring.

This pattern is most evident for the dairy farm case study example where more than one-third of urinary-N deposited in May is modelled to be lost in subsequent winter and

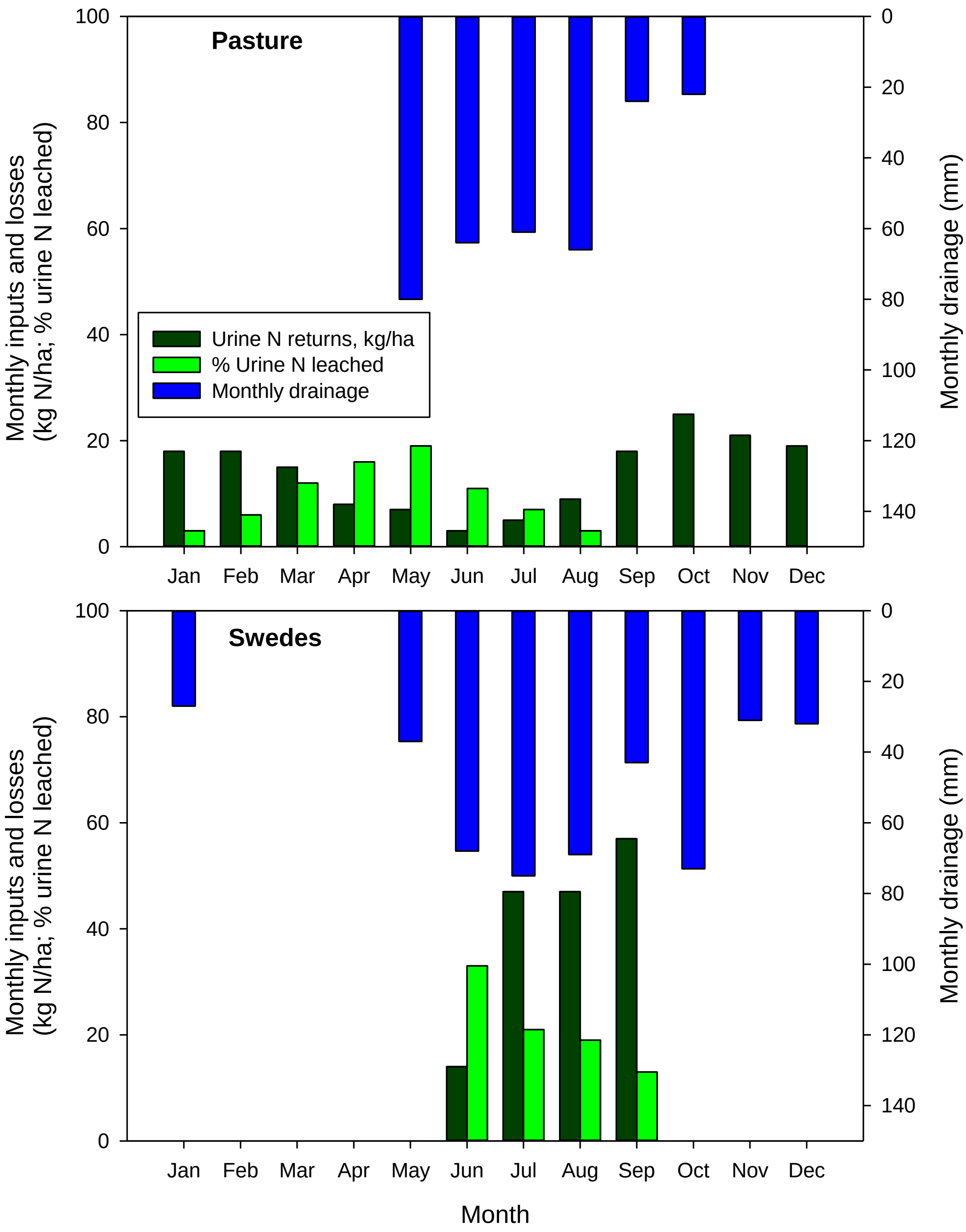

Figure 4.1: *Overseer* model estimates of monthly inputs and proportional losses via leaching of urine N deposited to pasture or swede paddocks on a typical intensive Southland sheep farm (Brown soil on LUC 2 land). Blue bars indicate typical monthly drainage values.

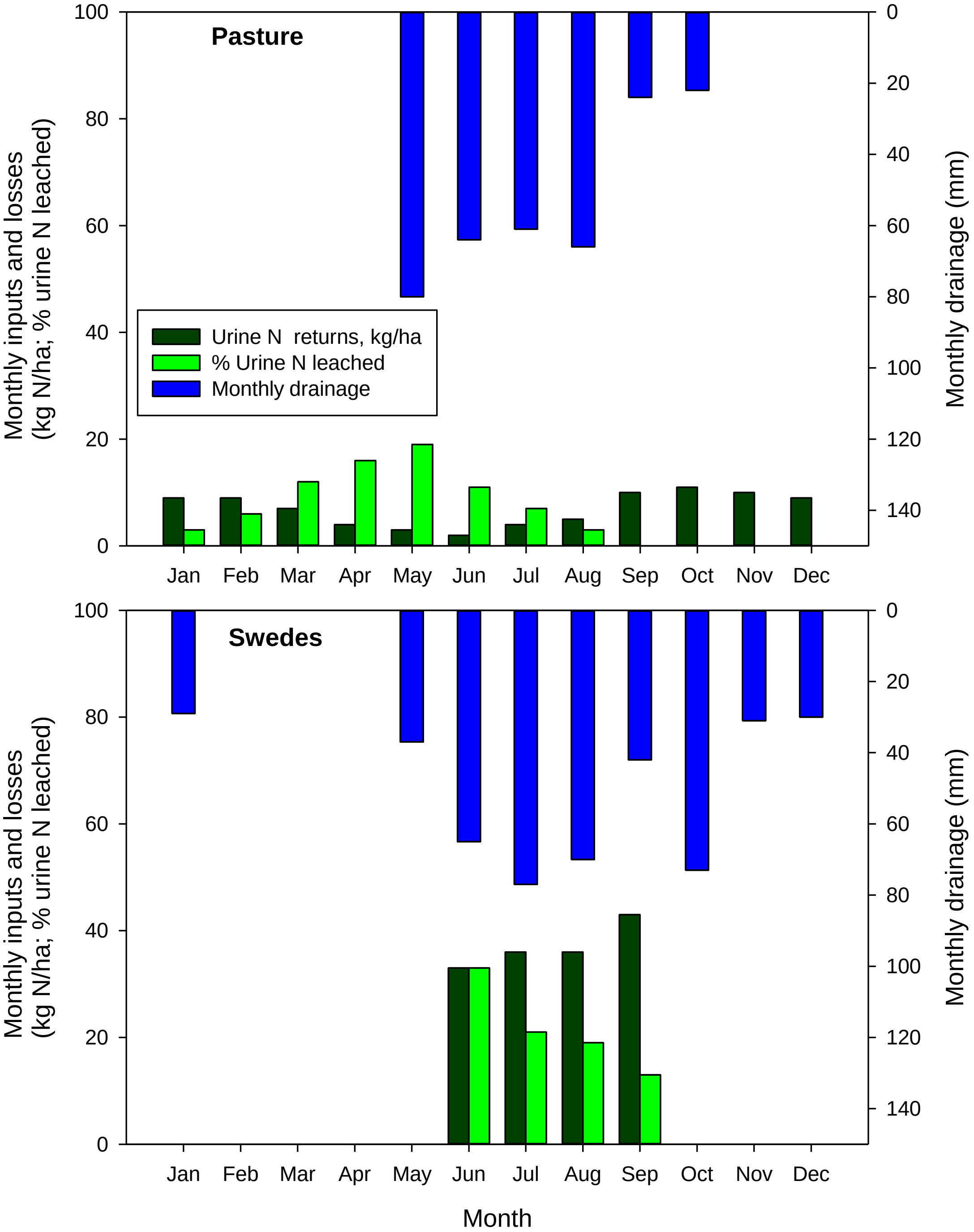

Figure 4.2: *Overseer* model estimates of monthly inputs and proportional losses via leaching of urine N deposited to pasture or swede paddocks on a typical hill country Southland sheep farm (Brown soil on LUC 5 land). Blue bars indicate typical monthly drainage values.

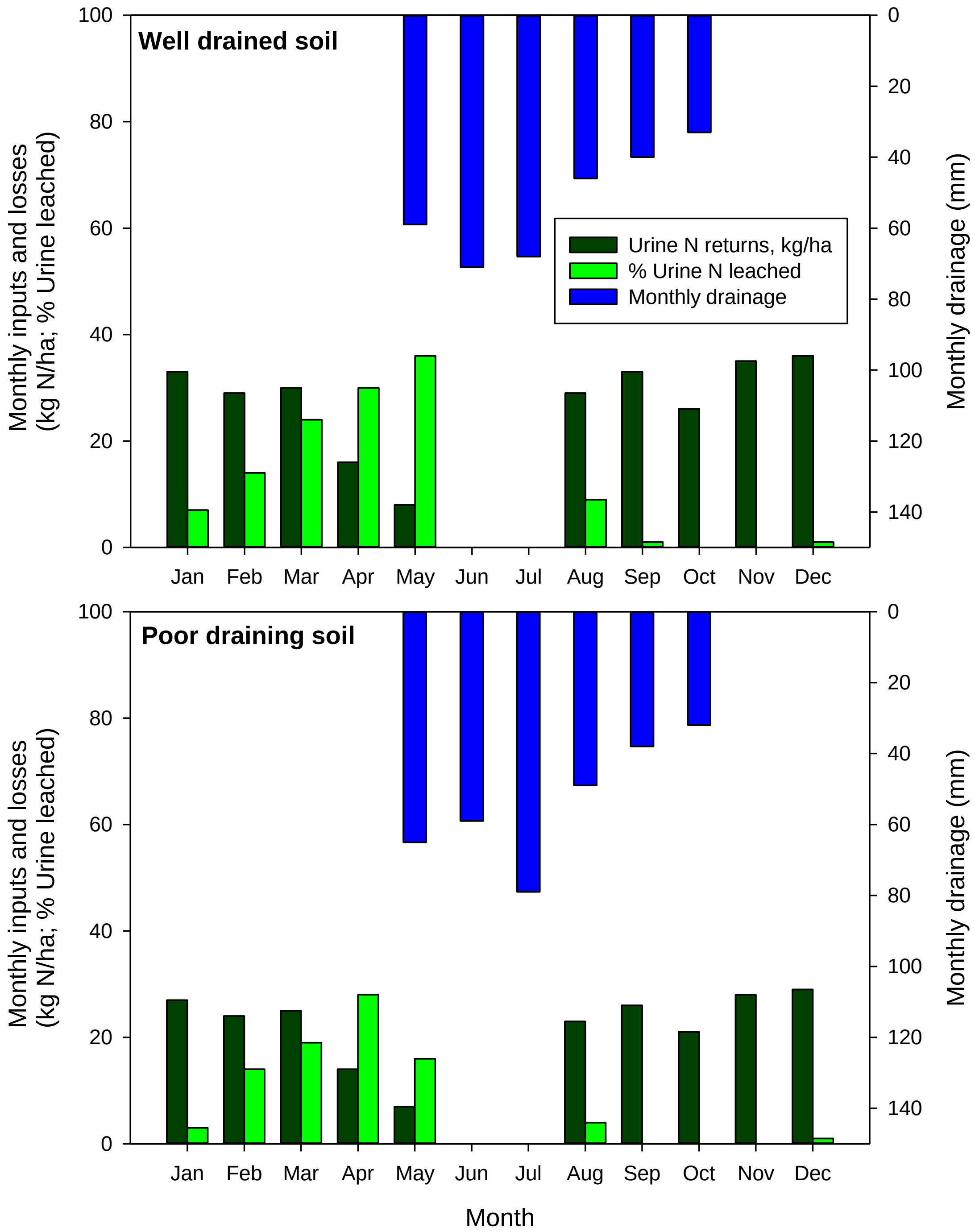

Figure 4.3: *Overseer* model estimates of monthly inputs and proportional losses via leaching of urine N deposited to pasture on a typical Southland dairy farm located on a well-drained Brown soil or poorly-drained Pallic soil. Blue bars indicate typical monthly drainage values.

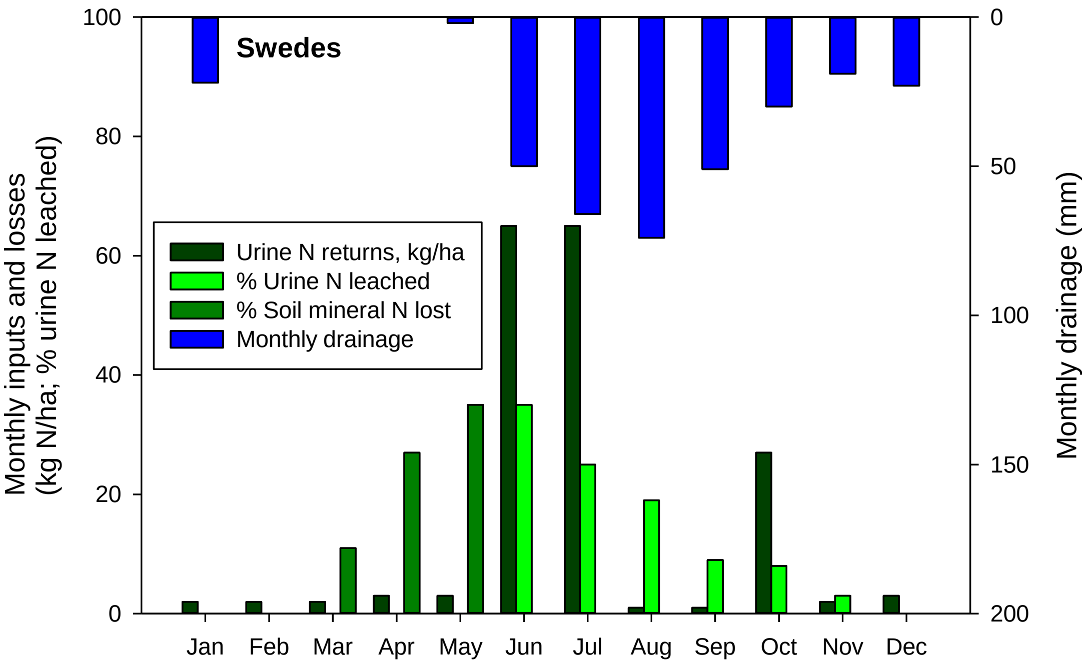

Figure 4.4: *Overseer* model estimates of monthly inputs and proportional losses via leaching of urine N deposited to a winter swede crop grazed by dairy cattle in Southland. Blue bars indicate typical monthly drainage values. Assumed soil type is an imperfectly drained Pallic silt loam.

spring drainage. In contrast, almost none of the urinary-N deposited in October or November is modelled to be lost in drainage. A management implication of this imbalance between autumn urinary N returns to soil and plant demand for N is that there is a greater potential to reduce annual N leaching losses by manipulating urinary deposition during the highest risk months of autumn and winter. For dairy systems, this could be achieved by minimising (autumn) or avoiding (eg winter) urinary returns to pasture by standing-off or housing cows (refer to section 5.2). Without this infrastructure, it is however difficult to manipulate this seasonal pattern of urinary N returns – animals need to eat (and excrete) through autumn and winter, albeit at lower intakes than typically observed in spring and summer months.

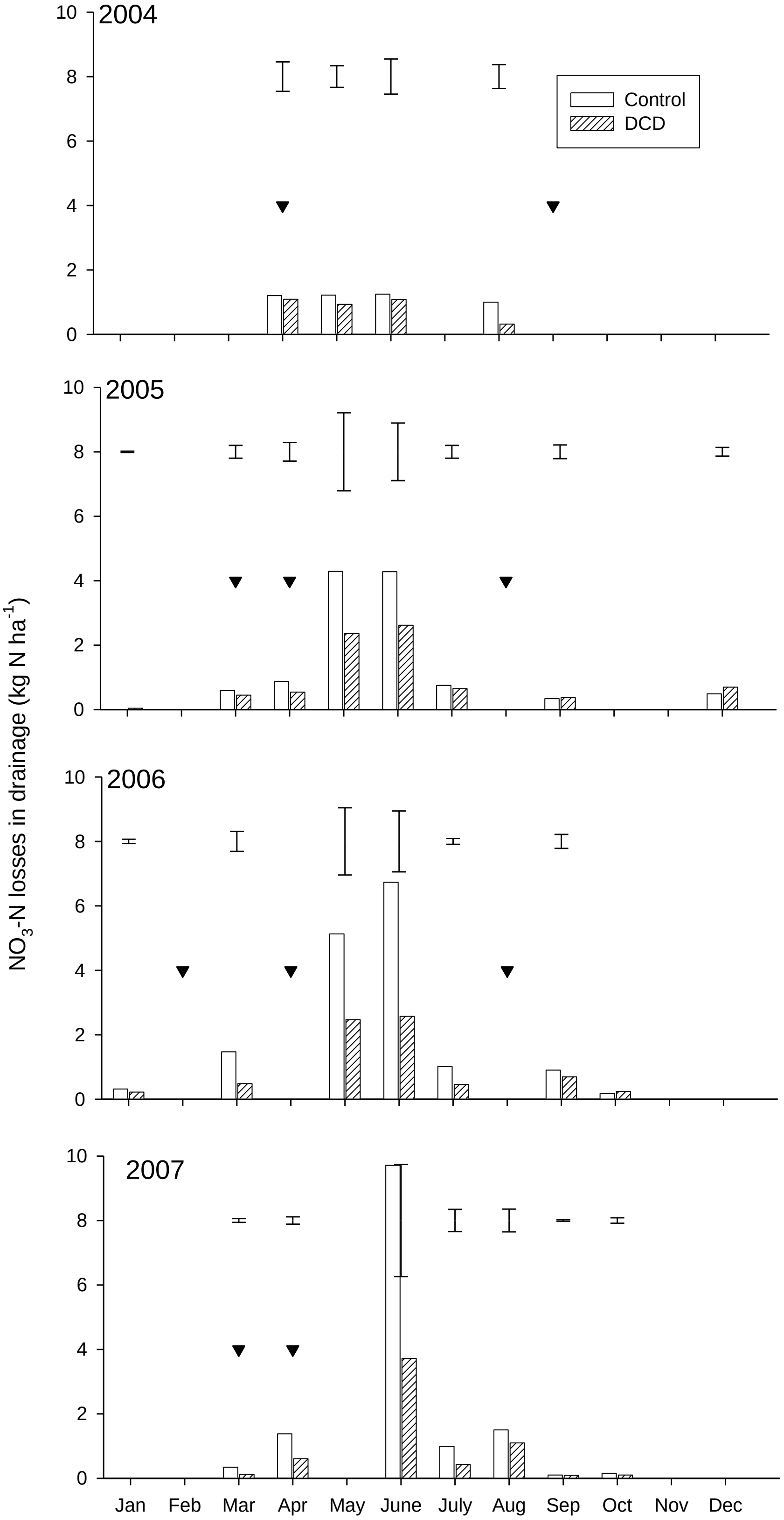

Figure 4.5: Measured monthly losses of N in drainage water from the Tussock Creek experimental site, 2004 to 2007 (from Monaghan et al. 2009b). Bars indicate SED values while ▼ show dates of DCD application.

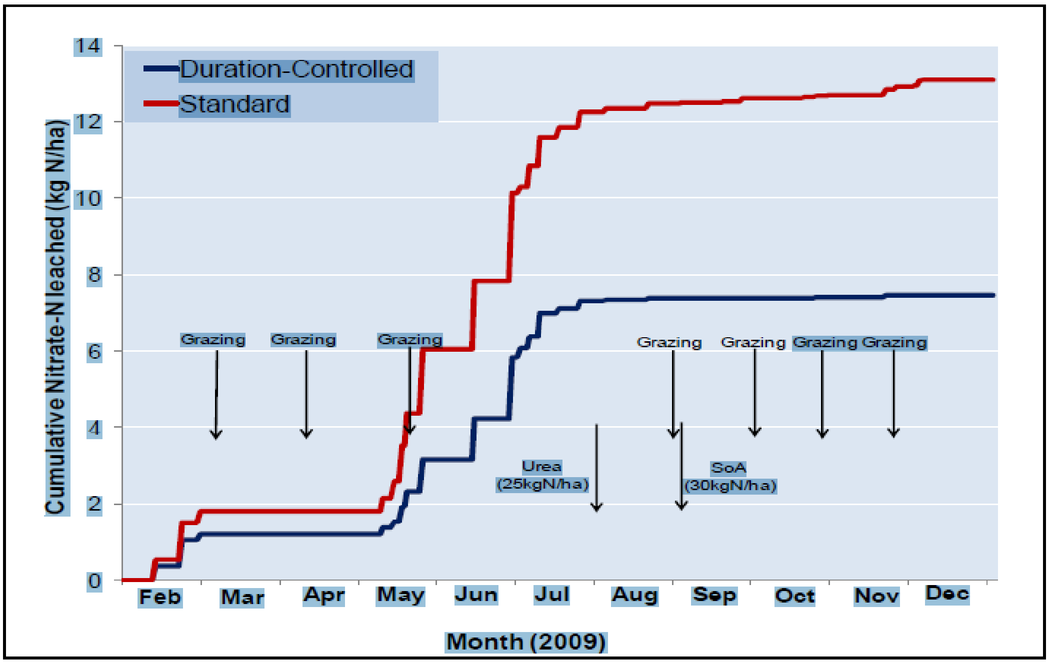

Figure 4.6: Cumulative nitrate-N leached in drainage water during 2009 for duration-controlled and standard dairy cow grazing treatments at Massey University's No. 4 Dairy Farm. (taken from Christensen et al. 2011). SoA = sulphate of ammonia fertiliser.

As discussed in section 3.2, N leaching losses from grazed winter forage crops in Southland are high relative to losses measured under grazed pasture (Monaghan et al. 2013; Smith et al. 2012; Smith & Monaghan 2013). This is also apparent when Figures 4.3 (N losses from pastures) and 4.4 (N losses from a winter forage crop) are compared. There is a very pronounced seasonal pattern of loss evident in Figure 4.4 due to potentially large amounts of mineral N remaining in the soil in late autumn following forage crop establishment and the deposition of much excretal N onto the grazed forage crop when it is grazed during winter when plant uptake is effectively absent.

4.2 Phosphorus & sediment

The seasonal patterns of P loss observed in studies by McDowell and others tend to be influenced more by transport factors than variations in the size of the different P sources available for loss in drainage or surface runoff. The transport of P from the landscape requires conditions conducive to either surface runoff or subsurface flow. The relative importance of these pathways for P transfers will depend on soil type and drainage status (see section 3.1), with the former usually being the more important pathway of transfer for well drained soils. For less well-drained soils requiring mole-pipe drainage, both surface runoff and subsurface drainage are important pathways of P transfer, with the relative importance of each changing for different times of the year. At the Tussock Creek dairy grazing study site, measured losses of P over autumn and winter months were approximately evenly spread between drainage and surface runoff (Figure 4.7). During spring, however, P was lost predominantly via surface runoff. On an annual basis, 56% of both DRP and TP yields were lost in surface runoff, despite subsurface drainage volumes being much greater. Sediment losses followed a similar pattern to that observed for TP (Figure 4.8), with relatively large yields in drainage and surface runoff recorded in May and June, and in surface runoff in September and October. Surface runoff accounted for 66% and 58 % of Total and VSS annual loads at this site. The greater significance of surface runoff as a pathway for P and sediment transfers in spring suggests that infiltration excess runoff becomes more important at this time of the year, probably due to the soil treading damage induced by cattle grazing during wet conditions (Houlbrooke et al. 2009). Protecting wet soils from such treading damage is therefore likely to help reduce these spring losses of P and sediment.

In a study in Otago, Curran-Cournane et al. (2011) found that most surface runoff occurred in winter when soil moisture contents were at or above field capacity. A seasonal effect was observed and showed that although the greatest P loads occurred in winter, the greatest P concentrations occurred in summer months, under infiltration-excess conditions. These summer losses could pose a risk to receiving waterways because increased light and warmth may induce an algal response compared to winter. The findings demonstrated that it was a case of not ‘what’ stock type grazed (sheep, cattle or deer), but ‘when’ stock grazed paddocks relative to soil moisture that influenced P and SS lost in surface runoff. Regression analysis identified significant decreases in the concentrations and loads of P (Figure 4.9) and SS (data not presented) with days since grazing. Taken together, their results also support the use of mitigation strategies such as restricted or nil grazing in winter when soil moisture has reached field capacity to minimise P and SS loss to surface water, regardless of stock type.

Because dung is potentially an important source of P that is transported in flow to streams, careful management of farm dairy effluent (FDE) is required. For Southland, research has guided Regional Council policies that help to ensure FDE is not a source of P (and FMO) contamination of streams (Houlbrooke & Monaghan 2009). The current policy framework has a clear seasonal aspect whereby FDE scheduling and hydraulic loading rates are determined by soil water deficit criteria. This approach effectively means that FDE storage and low rate or low depth applications are required during wet spring and autumn conditions to avoid direct (or “incidental”) FDE transfers from land to water, as have been documented in studies by Monaghan et al. (2010), Houlbrooke et al. (2004a,b) and elsewhere.

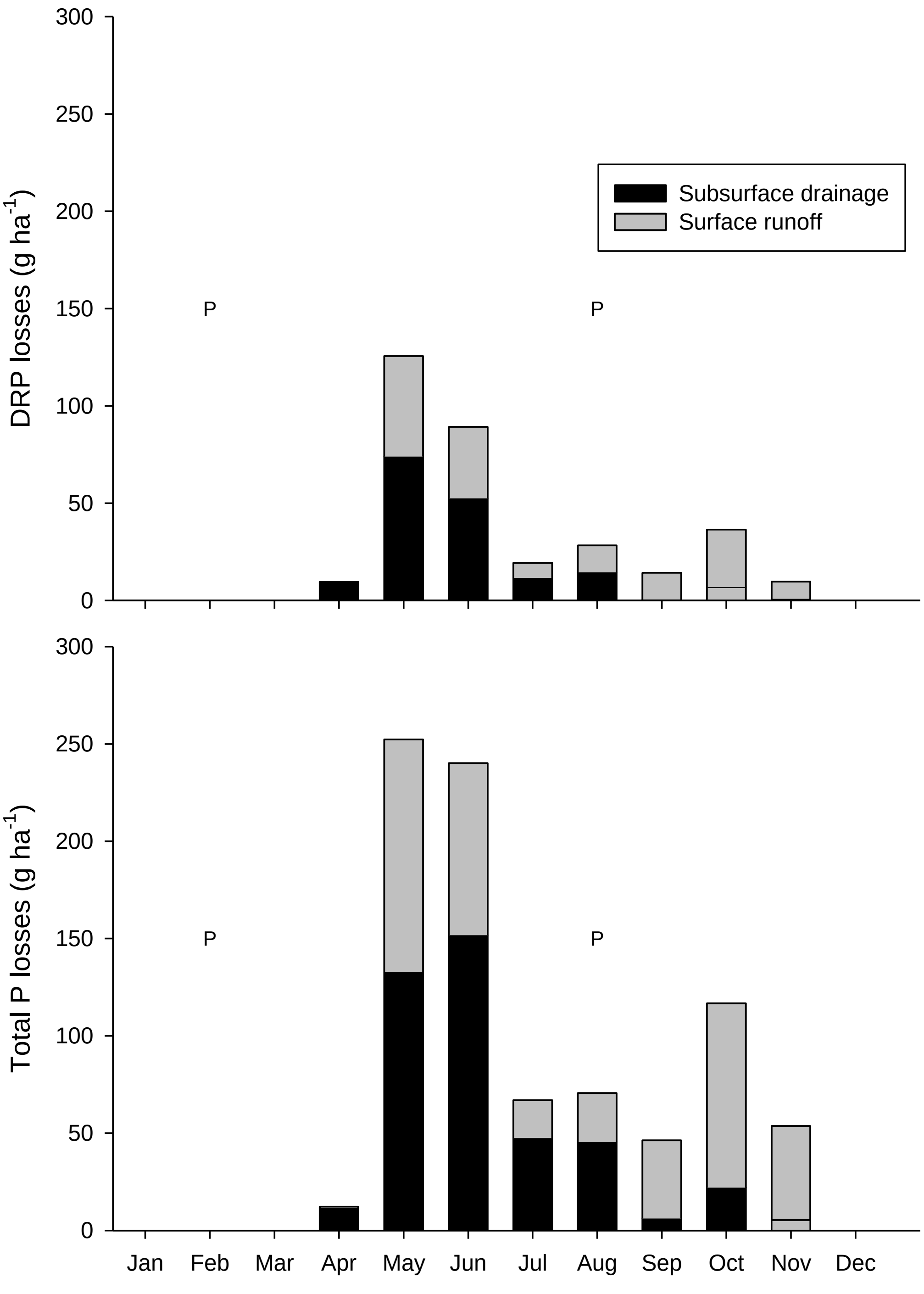

Figure 4.7. Mean monthly losses of P (g P ha^{-1}) measured in subsurface drainage and surface runoff at the Tussock Creek dairy grazing trial site (Monaghan et al. 2014). Note that “P” indicates the application of P fertiliser.

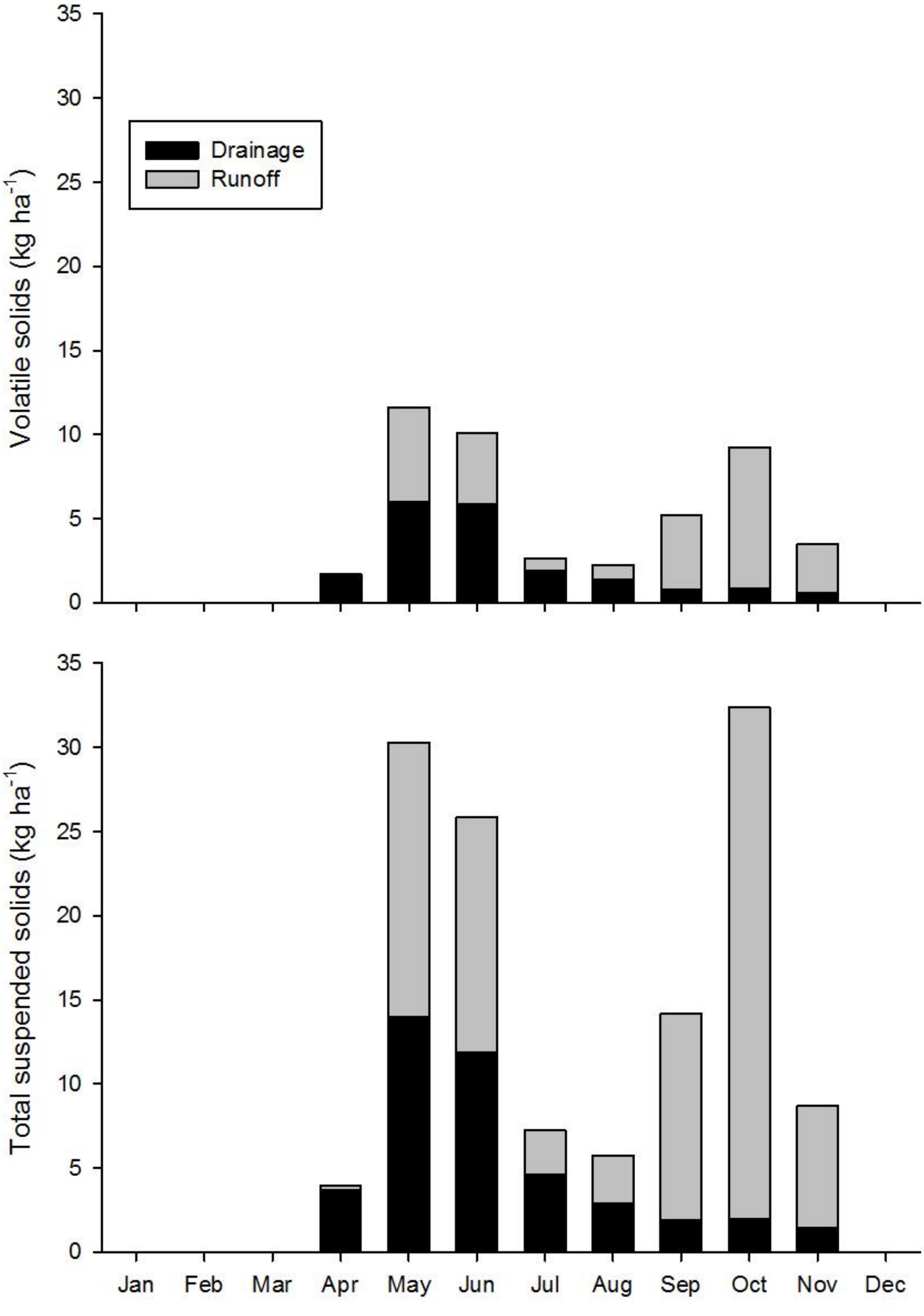

Figure 4.8. Mean monthly losses of suspended sediment (kg ha^{-1}) measured in subsurface drainage and surface runoff at the Tussock Creek dairy grazing trial site (Monaghan et al. 2014).

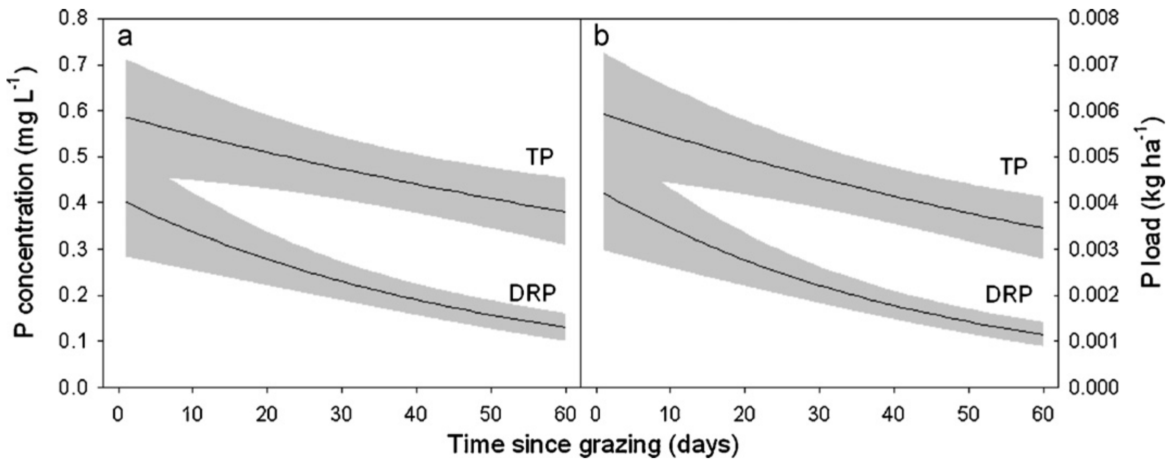

Figure 4.9: Regression of mean concentrations (a) and loads (b) of dissolved reactive phosphorus (DRP) and total phosphorus (TP) in surface runoff against time (days) since grazing by sheep, cattle or deer. Re-drawn from Curran Cournane et al. (2011).

Another example of incidental P loss that requires a seasonal consideration of source and management is the application of P fertiliser to land. Research has shown a high potential for P loss from recently-applied soluble P fertilisers (Figure 4.10). This potential for loss will be realised if application timings coincide with surface runoff events. McDowell et al. (2005) accordingly defined months with the greatest risk of surface runoff, and thus potential for P loss, for each region of New Zealand. For Southland, May through to September (inclusive) were deemed to be months of greatest risk. Consideration of less soluble forms of P fertiliser or changing application timings to a month with lower risk of surface runoff are 2 strategies that can help to reduce the risk of incidental losses of fertiliser P (McDowell & Catto 2005; McDowell & Smith 2012).

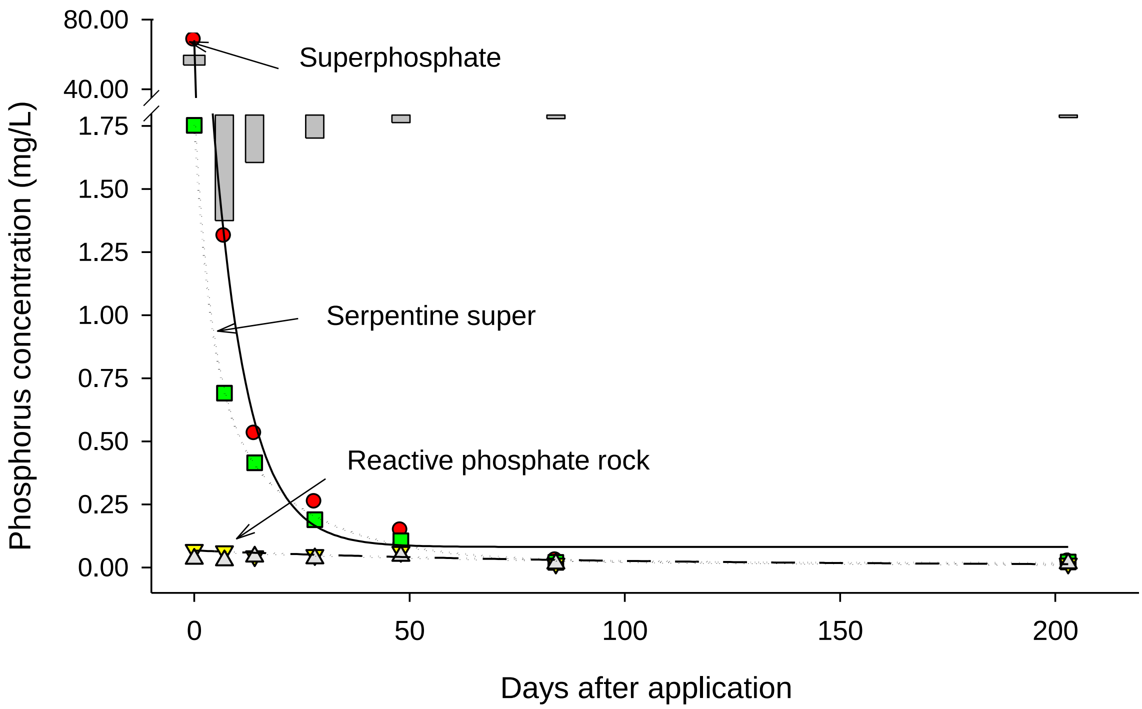

Figure 4.10. P concentrations in surface runoff following the application of superphosphate (red circles), serpentine super (green squares) and reactive phosphate rock (yellow triangles) (from McDowell & Catto 2005). Grey bars indicate least significant difference values at $P < 0.05$.

4.3 Faecal microorganisms

Although we have less knowledge about losses of FMOs from pastures to water than we do for N and P, our current understanding would suggest that both transport and source factors are important determinants of the numbers of FMOs transferred in drainage or surface runoff. Measurements of *E. coli* losses in subsurface drainage or surface runoff at the Tussock Creek dairy grazing study site show a similar pattern as observed for P, with losses during autumn and winter months being approximately evenly spread between drainage and surface runoff (Figure 4.11). During spring, however, *E. coli* losses were predominantly via the surface runoff pathway. On an annual basis, 75% of the annual *E. coli* yield was lost in surface runoff, despite subsurface drainage volumes being much greater. Greater concentrations of *E. coli* in surface runoff than in subsurface drainage have also been noted by Oliver et al. (2005)(cited in Muirhead & Monaghan 2012). As noted for P and sediment, the greater significance of surface runoff as a pathway for *E. coli* transfers to water in spring suggests that infiltration excess runoff becomes more important at this time of the year.

Data from the Tussock Creek study site was used to calibrate a simple two-reservoir model of *E. coli* sources found in grazed pastures (Muirhead & Monaghan 2012). This model has been used as a tool to synthesise our current knowledge and explore potential options for reducing FMO losses. Some conclusions from this analysis are:

- As observed for P, concentrations of *E. coli* in overland flow and subsurface drainage decreased with time since last grazing (up to 120 days in the case of *E. coli*; refer to Fig. 3 in Muirhead & Monaghan 2012).
- Data clearly showed that there was an environmental reservoir of *E. coli*, other than those remaining in faecal pats deposited to pasture, that was displaced in drainage after 120 days post grazing.
- Mechanically disrupting cow pats did not hasten *E. coli* die-off rates and thus did not appear to reduce the risk of *E. coli* loss to water.
- A suggestion for future research was to focus on preventing the build-up of the faecal microbial reservoir in the months preceding runoff events. This strategy obviously overlaps with the restricted grazing strategies mentioned above for N and P. Given the greater sensitivity of receiving waters to faecal pollution during warmer seasons, such an approach should probably focus on management of soils and pastures during spring to both reduce the size of the faecal source and to protect wet soils from treading damage (thus minimising surface runoff).

An additional source of FMO losses to water from farms are the lanes trafficked by stock as they move between paddocks or to the milking parlour. Poorly-sited and/or poorly designed lanes that discharge directly to streams have the potentially to deliver significant numbers of FMOs to streams during low flow conditions. Monaghan and Smith (2012) documented the concentrations and loads of N, P, sediment and *E. coli* in overland flow from 7 laneways sites located on 4 dairy farms in the Bog Burn catchment. This flow was very enriched in nutrients and faecal bacteria, with mean concentrations similar to those found in farm dairy effluent. Concentrations were observed to be higher at sampling sites close to the milking parlour, presumably due to the greater frequency of excreta depositions in these more highly trafficked lengths of the farm laneway network. Although assessments of laneway contributions to catchment scale discharges of stream pollutants suggested this source made a relatively minor contribution when expressed on an annualised basis, laneways were predicted to be a potentially important source of stream pollutants during summer when stream flows are relatively low and temperatures are warmer. Unlike within-field surface runoff and subsurface drainage pathways, which are not active year-round, overland flow from lanes was observed to occur in all months of the year. When laneway loads measured over summer were expressed as a proportion of measured summer catchment loads, lanes could have contributed between 4 and 76% of P loads, or between 6 and 100% of SS loads, assuming that 5 or 100% of laneway areas discharge directly to Bog Burn stream, respectively. Calculations of laneway contributions to catchment *E. coli* loads suggested that lanes could account for all the *E. coli* discharged from the catchment over the summer period, although this assessment was confounded by the likely die-off of *E. coli* between on-farm sources and the catchment outlet. Given that this pathway was the only one likely to be active during warmer summer months when stream flows are typically low, the seasonal analysis described above indicates that laneway sources of stream pollutants have greater ecological importance than suggested by simple consideration of annualised fluxes. Strategies that can reduce contaminant transport via this pathway are therefore to be encouraged, and include such measures as bunding or lane re-contouring to re-direct runoff away from streams, creating wetlands, sediment traps or grass buffer areas between lanes and streams, and, ideally, ensuring lanes are appropriately sited and not in close proximity to stream margins.

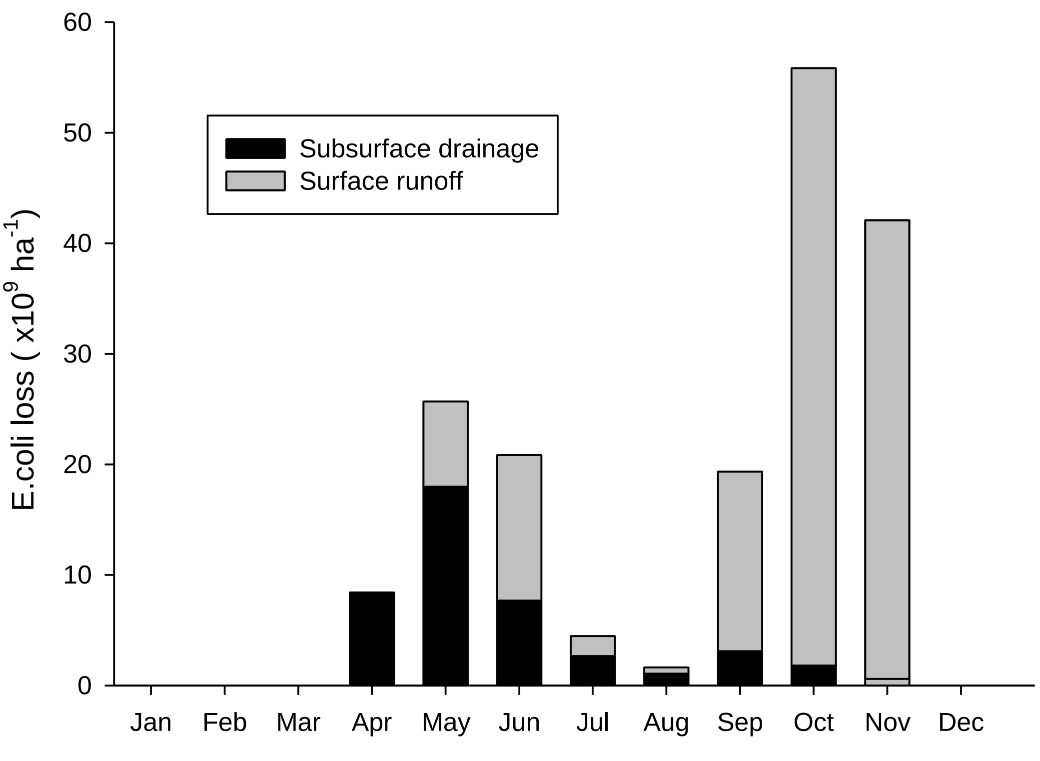

Figure 4.11. Mean monthly losses of *E. coli* (x 10^9 ha^{-1}) measured in subsurface drainage and surface runoff at the Tussock Creek dairy grazing trial site (Monaghan et al. 2014).

5. Management options that seasonally target interventions to reduce the risk of contaminant accumulation and loss

Many of the practices currently recommended for minimising the risk of pollutant transfer from farms to water target particular times of the year when source or transport risks are greatest. These are categorised and described below and also listed in Table 5.1. Edge-of-field or off-paddock solutions such as wetlands, buffer strips and sediment traps are not considered.

5.1 Scheduling fertiliser and effluent applications

The most obvious set of measures that seek to optimise the timing of management interventions are those concerning fertiliser inputs and effluent returns to land:

- For N fertilisers, recommended application rates and timings are tailored to ensure plant demand and N supply is optimally matched. Temperature is therefore a key decision trigger whereby applications are only recommended once soil temperatures are above 5°C in spring, or 8°C in autumn. It is recommended that applications are also avoided in months when the risk of transport in overland flow or subsurface drainage is high. For Southland environments, the practical implications of these recommendations are that N fertiliser applications are best avoided during late autumn and winter months.
- For P fertilisers, consideration of less soluble forms of P fertiliser or changing application timings to a month with lower risk of surface runoff are 2 strategies that can help to reduce the risk of incidental losses of fertiliser P.
- Because FDE is potentially an important source of *E. coli*, ammonium-N and P that can be transported in flow to streams, FDE storage and low rate or low depth applications are required during wet spring and autumn conditions to avoid direct (or "incidental") FDE transfers from land to water. In addition to the temperature thresholds described above for N fertiliser, adherence to soil type and soil wetness triggers will also help to minimise these incidental transfers of stream pollutants (refer to Houlbrooke & Monaghan 2009).

5.2 Reducing the amounts of excreta deposited to soil and pasture

As discussed in section 3.2, much of the N leaching risk in grazed pastures is attributable to the relatively high equivalent N loading rates in the urine patch, which typically range between 300 and 1000 kg N ha^{-1}. Because pastures and crops have only a limited (and growth-dependent) ability to utilise this urinary N, there is less opportunity for the plants to capture urinary N deposited shortly before winter compared to urinary N deposited the preceding spring and summer. In fully or partially grazed systems, a strategy for reducing N losses is to minimise urine deposition during periods of high N loss risk, by either removing the animals from pasture at certain times or by extending the existing housing period. Measurement and modelling of these "restricted grazing" strategies have been shown to reduce N leaching losses and/or surface runoff (Cardenas et al. 2011; Christensen et al. 2011, 2012; de Klein et al. 2006; Ledgard et al. 2006). The size of these reductions depends on the duration and timing of the restricted grazing period. Disproportionately greater benefits have been observed if grazing was restricted shortly preceding or during periods when losses were likely i.e. when drainage was occurring. Stand-off pads (preferably covered), herd shelters and wintering barns are some of the infrastructure options that are required for an off-paddock animal confinement system to work effectively. A restricted autumn grazing regime was evaluated at the grazing study site at Tussock Creek in Southland during 2001 to 2003. The 4 hour grazing strategy showed some promise for mitigating N losses, reducing nitrate-N and Total Soluble N (TSN) yields in drainage by 39 and 36%, respectively (de Klein et al. 2006; Monaghan et al. 2014). This grazing approach was selected as a compromise between minimising the time spent grazing (and urinating) in the 2 to 3 months prior to the onset of winter drainage, and ensuring that cows had opportunity to utilise as much of the pasture on offer as possible.

The increasing use of animal wintering barns and shelters is an important development in Southland dairy farming. Whilst these facilities are typically constructed for a range of reasons that include productivity and labour considerations, whole-system modelling assessments of their use indicates that they can potentially deliver significant reductions in the amounts of N, P, sediment and FMOs transferred from farms to water. This potential environmental benefit is delivered via a couple of mechanisms:

- As noted above, housing animals allows for the capture, storage and later return of dung and urine to pastures, thus avoiding N deposition to bare soils at times when the risk of drainage is greatest and plant uptake lowest.
- Housing animals in off-paddock facilities avoids causing the soil treading damage that contributes to increased volumes of surface runoff; potentially

large losses of sediment, P and *E. coli* from wintering areas can therefore be avoided (see section 3.1).

Realising the above environmental benefits does however depend on how well off-paddock facilities are designed and managed, particularly the methods used to handle, store and land-apply collected excreta. Economic imperatives may also drive a process whereby barns and shelters are used to intensify the farm system through greater use of imported feed; this effect may off-set some of the environmental benefits noted above. Decisions associated with off-paddock wintering approaches are therefore complex and costly; the particular system a farmer selects will be determined by their individual circumstances and preferences (Dalley et al. 2012).

5.3 Managing soils to minimise surface runoff

The patterns of contaminant loss evident in Figures 4.6, 4.7 and 4.9 suggest that management practices that can protect soils from treading damage in spring, and thus reduce surface runoff, are likely to have greatest benefit for reducing the impacts of P, sediment and *E. coli* losses on water quality. This benefit is likely due to the effects of both reducing the total annual amounts of contaminant loss in surface runoff and the timing of the intervention – spring surface runoff events are likely to have a greater impact on water quality in the following warmer summer months compared to late autumn and winter runoff events. The link between soil structure/quality and surface runoff risk has been reported by Curran Cournane et al. (2011) who observed significant negative relationships between soil macroporosity and the concentrations or loads of DRP and TP lost. McDowell et al. (2003b) also reported a relationship between soil macroporosity and concentrations and loads of DRP, TP, and SS for grassland and cultivated soils subjected to simulated rainfall. An inverse relationship between P and SS loads and soil saturated hydraulic conductivity (K_{sat}) was noted by Curran Cournane et al. (2011) (Figure 5.1) and McDowell et al. (2003b) also reported that an increase in soil treading damage decreased K_{sat}, producing greater volumes of surface runoff. These results are not unexpected since a decrease in soil infiltration rate would produce a greater volume of surface runoff, assuming saturation-excess conditions are not causing the runoff. McDowell and Houlbrooke (2009) also suggested the use of restricted grazing as a means of decreasing P and SS from grazed winter forage crops.

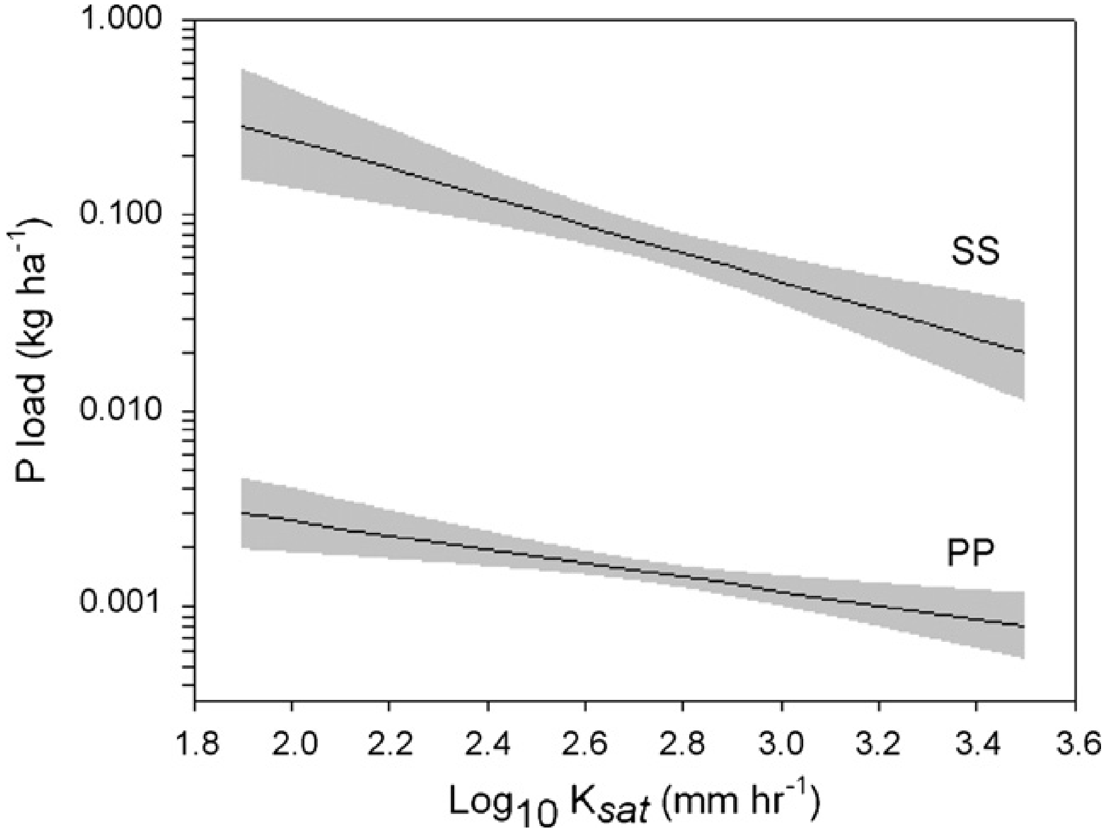

Figure 5.1. Regression relationships between mean soil saturated hydraulic conductivity (K_{sat}; 0–5cm depth) and the daily loads of particulate phosphorus (PP) and suspended sediment (SS) lost in surface runoff. Shading represents 95% confidence intervals. Re-drawn from Curran Cournane et al. (2011).

In addition to ***timing*** as an issue to consider for the strategy of protecting soils from treading damage (and thus surface runoff risk), consideration of ***spatial*** factors also needs to be taken into account to maximise the cost-effectiveness of measures for reducing the time animals spend on paddocks when soils are wet. Targeting the implementation of an off-paddock strategy to CSAs where the risk of runoff is greatest will help to ensure that maximum benefit is gained for least cost and logistical complexity. A relevant example of this is reported by Orchiston et al. (2013 and unpublished data) who observed greatly reduce yields of sediment (90%) and P (83%) in surface runoff from a winter forage crop that was grazed in a manner that minimised grazing time and soil treading damage in the CSA of the headwater study catchment. A similar strategy of protecting CSAs in gullies, seeps and swales is also likely to reduce contaminant losses in surface runoff from pastured landscapes.

The strategy of managing wet soils to minimise surface runoff has some overlap with the strategy for reducing the amounts of excreta deposited to pasture (section 5.2) in that both require some sort of off-paddock facility for locating and feeding animals when they are not grazing pasture or forage. Given financial and logistical considerations, these strategies are therefore likely to be more relevant to dairy cows than other stock types, although they may potentially be relevant to some beef cattle and deer farming systems. It is possible that we may increasingly see more “hybrid” grazing systems on some farms as they seek to:

- Reduce urinary N returns to pasture in autumn and winter by reducing the time animals spend grazing each day as a way of reducing N leaching risk (section 5.2).
- Reduce the risk of P, sediment and *E. coli* losses in surface runoff in winter and spring by minimizing soil treading damage during wet conditions (this section).
- Provide a feeding facility to offer cows supplemental feeds in a way that minimises feed wastage and maximises the recovery of excretal nutrients.

As noted earlier however, the evolution of these types of hybrid production systems can result in unintended consequences that have the potential to off-set some of the environmental gains that are initially apparent.

5.4 Diet manipulation to reduce N excretion

Another seasonally-relevant option that can help to reduce N losses from pastures is to substitute autumn feeds that have a relatively high N content, such as N-fertilised pasture, with low-protein supplemental feeds such as grain or cereal silages. The principle of this strategy is that a lower dietary N content should result in reduced N excretion by the animal, which in turn should reduce the risk of N leaching loss. Maximum benefit is likely if this strategy is targeted to the autumn period when the risk of urinary N loss is greatest (Figure 4.3). 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 for the whole system, including the land used to grow the forage which can have relatively high N leaching losses (Ledgard et al. 2006). 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. 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.

The inclusion of crops such as fodder or sugar beet in the diet should also potentially result in reduced N excretion by the animal, which in turn should reduce the risk of N leaching loss. On-going research in the P21 programme (and elsewhere) is seeking to quantify this effect for crops used as winter feed; to my knowledge there is as yet no published material that documents the magnitude of this effect. These forage crops that have lower N contents could also potentially be used to reduce dietary N contents during summer and autumn, although this benefit could be more than off-set by the introduced risk of N leaching from the crop paddock itself (assuming it has been grazed to harvest the available feed). As noted in section 3.2, recent research in Southland and Otago has observed that summer-grazed forage crop paddocks have a relatively high potential for N leaching over the following winter (Chrystal et al. 2012 and unpublished results). A strategy that could overcome this potential risk is to mechanically harvest and feed out the low-N crop, thus avoiding the return of concentrated urine patches to bare soil shortly ahead of periods of autumn and winter drainage.

The application of the plant growth hormone gibberellic acid (GA) to pastures (marketed as "ProGibb" in New Zealand) has been suggested as a strategy that could potentially lower dietary protein concentrations and thus N excretion. Crude protein responses following GA application have however been variable (e.g. van Rossum et al. 2013) and to this author's knowledge no study has shown a link between GA application and reduced N leaching.

5.5 Using pasture species that have greater winter-activity and N uptake

Recent research by Malcolm et al. (2014) and others suggests that the strategic use of selected pasture species with high plant winter activity (plant growth/root metabolic activity) can help to mitigate N leaching losses. They observed that N leaching losses were 24–54% lower beneath urine-treated lysimeters containing an Italian ryegrass-white clover (WC) pasture mix than other pasture species, including a perennial ryegrass/WC treatment. Much of this benefit was attributed to the greater winter daily N uptake rate of the Italian ryegrass/WC mix rather than differences in specific root architecture (e.g. deep roots). Whilst this research is still at a preliminary stage and based on plot-scale findings, it does suggest that appropriate pasture species selection may provide another option for farmers wishing to minimise N leaching losses. On-going research is seeking to quantify this potential benefit in grazing studies undertaken at a farmlet systems scale. This next phase of research includes a consideration of the proportional area of a farm that can be feasibly maintained in pasture species that have greater winter activity.

5.6 Applying additives to soil to retain N or P

There has been much interest in the past decade in the role of soil additives that have the potential to retain N and P in less mobile forms. One mitigation technology that has recently been developed to reduce nitrate 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^+-N to the more mobile NO_3^--N form. Much of the initial DCD research was carried out at a plot-scale by Di & Cameron (e.g. 2004, 2005) and showed large reductions in N leaching from urine applications in late autumn and early spring. This led to a few grazing systems studies that strategically targeted autumn applications as a way to maximise the effectiveness of DCD. Monitoring at the Tussock Creek site from 2004 and 2007 showed that the application of a granular form of DCD reduced N losses in drainage by between 21 and 56%, depending on the year of study (Monaghan et al. 2009b). Because much of the annual drainage loss occurred during May and June, it was recommended that maximum effectiveness was achieved for DCD applications following pasture grazings in March and May. Due to recent concerns about DCD residues in milk, this product has however unfortunately been removed from the market.

Urease inhibitors have also been promoted and used as an option to reduce N losses to the environment, particularly, ammonia volatilisation to the atmosphere (Watson et al. 1998). However, their effectiveness at reducing N losses to water from grazed pasture systems remains uncertain.

Of the P-sorbing materials evaluated as options for capturing P in flow or retaining P in soil (McDowell & Nash 2012), none appear to be practical and cost-effective options at this point in time. McDowell & Houlbrooke (2009) reported that aluminium sulphate (alum) decreased P losses in surface runoff by about 30% when applied after animals grazed a winter forage crop in North Otago but cautioned that additional work was required to determine its long-term effectiveness and use in other soils and under different management regimes. In the high rainfall West Coast environment, McDowell (2010) noted that alum did not decrease P losses from a grazed dairy pasture, most likely due to aluminium (Al) being washed off in surface runoff before it could bind to the soil. At this point in time it would therefore appear that alum is not a feasible and cost-effective option to be recommending for reducing P losses from grazed pastures.

6. Mitigation methods and effectiveness

For completeness, lists of the mitigation options that are relevant to minimising the impacts of dairy and dry stock farms are contained in Appendices I and II, respectively. These are based on the MENU of practices collated by Waikato Regional Council and DairyNZ (http://www.waikatoregion.govt.nz/menus), but have been refreshed to reflect cost and effectiveness metrics for Southland farms. The menu includes a general rating of effectiveness for each practice based on recent research and best guess. Assessments consider the flow pathway targeted by each mitigation and have been scaled to a whole-farm system equivalent.

Table 5.1. Seasonally-targeted interventions that reduce the risk of contaminant accumulation and loss from grazed pastures. Cost and effectiveness estimates for each are given in the Menus of Practices for Southland farms attached to the end of this report (Appendices I and II). Recommended scheduling guidelines for applications of N fertiliser are assumed to be followed (as per section 5.1)

Strategy	Target pathway[1]	Barriers to adoption	Ancillary benefits
N mitigation:			
Off-paddock cow wintering	SSDR	Cost of off-paddock facility; additional farm complexity	Will reduce losses of FMOs, SS and P
Restricted grazing of winter forage crops	SSDR & OF	"	"
Restricted autumn grazing of pasture	SSDR	"	"
Autumn substitution of N-fertilised pasture with low N feeds	SSDR	Cost of low N feed; ability to feed it out with minimal wastage	Can improve animal energy intakes
P and sediment mitigation:			
Low solubility P fertiliser	OF		
P fertilization outside risk months	OF		
Restricted spring grazing on wet soils	OF	Cost of off-paddock facility; additional farm complexity	Will help protect soils and pastures from treading damage (production benefit).
Deferred and/or low rate effluent irrigation	SSDR & OF	Cost of pond storage; additional labour possibly needed	Will reduce losses of FMOs and N

[1]SSDR = subsurface drainage; OF = overland flow.

7. Future research needs

Many knowledge gaps remain in our understanding of contaminant losses from farms to water and how management interventions may reduce these losses. There is also a great deal of uncertainty attached to estimates of losses using currently available modelling tools. Whilst acknowledging that much remains to be learnt to improve our understanding of the potential impacts of farming systems, below are some suggested priority areas for further research. Some of these research questions are being addressed in on-going research programmes and are highlighted accordingly.

- The impacts of sheep wintering practices on water quality are poorly defined. A better understanding of these impacts is desirable given (i) large numbers of sheep are still wintered in Southland, and (ii) it is likely that this part of the sheep farming system is where the most cost-effective measures for mitigating water quality impacts can be targeted. A focus on both N losses in subsurface drainage and sediment, P and FMO losses in surface runoff from CSAs is required.
- Future research should also focus on preventing the build-up of the faecal microbial reservoir in the months preceding runoff events. This strategy obviously overlaps with the restricted grazing strategies relevant for N and P mitigation; given the greater sensitivity of receiving waters to faecal pollution during warmer seasons, this strategy should probably be focussed on management of soils and pastures during spring to both reduce the size of the faecal source and to protect wet soils from treading damage (thus minimising surface runoff).
- It is evident that some confinement systems used for wintering cows are also being employed to make wider changes to the farm system. These changes include milking cows for longer, feeding greater amounts of supplement and using the pad/housed system to protect wet soils and pastures during spring. Such adjustments have wider implications for the environment that are not easily understood by the scientific community nor are adequately captured by current modelling frameworks. Investments in confinement facilities are also costly and require careful consideration and financial planning. More research is need to better understand the full environmental, economic and social implications of using these facilities.
- There is a need to link our knowledge of the fluxes in seasonal losses and understanding of transport pathways, as documented in this report, with seasonal trends in ground- and surface water quality observed in larger scale

monitoring undertaken by Environment Southland. This information can then be used to more effectively guide community processes that seek to attain water quality values and goals for the Southland community.

Additional research questions that need addressing but are perhaps of lesser immediate priority or are relevant to issues beyond just water quality are:

- Preliminary research suggests that minimum or no-tillage management techniques may help to minimise nitrate leaching losses from winter forage crops. These benefits need to be quantified within the context of a fully grazed paddock where urinary N returns are an additional source of leached N.
- A better understanding of the characteristics of manures collected from animal confinement systems such as barns and standoff pads in cooler climates typical of a Southland winter is required. Quantification of emission factors for NH_3, CH_4 and N_2O emissions from stored manures is of particular priority. This data is essential if we are to be able to holistically assess environmental efficiencies/unintended consequences and to ensure that suggested improved wintering practices do not result in "pollution swapping" e.g. implementing measures that reduce N leaching but also result in increased emissions of greenhouse gases. On-going research within the P21 programme and other GHG research programmes are attempting to fill this knowledge gap. Improved understanding of the management systems required for handling, storing and re-applying these winter manures and effluents to is also required.
- The high capital cost of existing confinement systems such as barns is a barrier to rapid adoption of this management system. Development of low cost standoff options would provide alternatives that may be more palatable to the farming sectors, especially non-dairy farming enterprises.

8. References

Bowler, D.G. (1980). The drainage of wet soils. Hodder and Stoughton, Auckland. 259p.

Byers, H.L. Cabrera, M.L. Matthews, M.K. Franklin, D.H. Andrae, J.G. Radcliffe, D.E. McCann, M.A. Kuykendall, H.A. Hoveland, C.S. and Calvert, V.H. (2005). Phosphorus, sediment, and *Escherichia coli* loads in unfenced streams of the Georgia Piedmont, USA. Journal of Environmental Quality 34, 2293-2300.

Cameron, K.C., Di, H.J., Roberts, A., Beale, N., Weir, J., Borrie, N. (2014). Monitoring effects of Southland Demonstration Farm on stream water nitrate. In: Nutrient management for the farm, catchment and community (Eds L.D. Currie and C L.

Christensen). http://flrc.massey.ac.nz/publications.html. Occasional Report No. 27. Fertilizer and Lime Research Centre, Massey University, Palmerston North, New Zealand. 6 pages.

Cardenas L.M., Cuttle S.P., Crabtree B., Hopkins A., Shepherd A., Scholefield D., del Prado A. (2011). Cost effectiveness of nitrate leaching mitigation measures for grassland livestock systems at locations in England and Wales. Science of the Total Environment 409, 1104-1115.

Chapman, A.S., Foster, I.D.L., Lees, J.A., Hodgkinson, R.A., (2005). Sediment delivery from agricultural land to rivers via subsurface drainage. Hydrological Processes 19, 2875-2897.

Chrystal, J.M., Monaghan, R.M., Dalley, D., Styles, T. (2012). Assessments of N leaching losses from six case study dairy farms using contrasting approaches to cow wintering. Proceedings of the New Zealand Grassland Association 74:51-55.

Christensen C.L., Hedley M.J., Hanly J.A., Horne D.J. (2012). Nitrogen loss mitigation using duration-controlled grazing: field observations compared to model outputs. Proceedings of the New Zealand Grassland Association 74, 115-121.

Christensen, C.L., Hanly, J.A., Hedley, M.J., Horne, D.J. (2011). Nitrate leaching and pasture production from two years of duration-controlled grazing. In: Adding to the knowledge base for the Nutrient Manager (Eds L.D. Currie and C L. Christensen). http://flrc.massey.ac.nz/publications.html. Occasional Report No. 24. Fertilizer and Lime Research Centre, Massey University, Palmerston North, New Zealand. 8 pages.

Cichota, R., Snow, V.O., Vogeler, I., Wheeler, D.M., Shepherd, M.A. (2012). Describing N leaching from urine patches deposited at different times of the year with a transfer function. Soil Research 50, 694-707.

Craggs, R.J. Sukias, J.P. Tanner, C.T. Davies-Colley, R.J. (2004). Advanced pond system for dairy-farm effluent treatment. New Zealand Journal of Agricultural Research 47, 449-460.

Curran-Cournane, F., McDowell, R., Littlejohn, R., Condron, L. (2011). Effects of cattle, sheep and deer grazing on soil physical quality and losses of phosphorus and suspended sediment losses in surface runoff. Agriculture, Ecosystems and Environment, 140: 264--272.

Cuttle, S.P. and Bourne, P.C., (1993). Uptake and leaching of nitrogen from artificial urine applied to grassland on different dates during the growing season. Plant and Soil 150, 77-86.

Dalley, D., Geddes, T., Pinxterhuis, I., Chrystal, J., Styles, T. (2012). Wintering system performance – what do you need to know to achieve your farm systems targets? Proceedings of the South Island Dairy Event. http://side.org.nz/past-proceedings/

Deelstra, J., Iital, A., Povilaitis, A., Kyllmar, K., Greipsland, I., Blicher-Mathiesen, G., Jansons, V., Koskiaho, J., Lagzdins, A., (2014). Hydrological pathways and nitrogen runoff in agricultural dominated catchments in Nordic and Baltic countries. Agriculture, Ecosystems & Environment 195, 211-219.

de Klein, C.A.M. Smith, L.C. and Monaghan, R.M. (2006) Restricted autumn grazing to reduce nitrous oxide emissions from dairy pastures in Southland, New Zealand. Agriculture, Ecosystems and Environment 112, 192-199.

Di, H,J. and Cameron, K.C. (2002a) Nitrate leaching in temperate agroecosystems: sources, factors and mitigating strategies. Nutrient Cycling in Agroecosystems 64, 237-256.

Di, H.J. and Cameron, K.C. (2002b). Nitrate leaching and pasture production from different nitrogen sources on a shallow stony soil under flood irrigated dairy pasture. Australian Journal of Soil Research 40, 317-334.

Di, H.J. and Cameron, K.C. (2004). Treating grazed pasture soil with a nitrification inhibitor, 'eco-nTM, to decrease nitrate leaching in a deep sandy soil under spray irrigation – a lysimeter study. New Zealand Journal of Agricultural Research 47, 351-361.

Di, H.J. and Cameron, K.C. (2005). Reducing environmental impacts of agriculture by using a fine particle suspension nitrification inhibitor to decrease nitrate leaching from grazed pastures. Agriculture, Ecosystems and Environment 109, 202-212.

Dymond, J.R. (2010). Soil erosion in New Zealand is a net sink of CO_2. Earth Surface Processes and Landforms 35, 1763-1772.

Environment Southland and Te Ao Marama Inc. (2010a). Our Ecosystems: How healthy is the life in our water and our freshwater ecoystems? Part 2 of Southland Water 2010: Report on the State of Southland's Freshwater Environment. Environment Southland, Invercargill. ES publication number 2011/7 ISBN 0-909043-45-0. 122p.

Environment Southland (2010b). Regional Water Plan for Southland (amended in accordance with Council and Environment Court decisions). Publication No 2013-09, ISBN 0-909043-07-0. http://www.es.govt.nz/media/35406/regional_water_plan_-_june_2014.pdf.

FAR (2008). Millenium Tillage Trial – Nitrate leaching over winter (2001-2007). Foundation for Arable Research Factsheet No. 80, July 2008.

Fraser, M. Cameron, K.C., Sherlock, R.R. (1994). Lysimeter study of the fate of nitrogen in animal urine returns to irrigated pasture. European Journal of Soil Science 45, 439-447.

Francis, G.S. (1995). Management practices for minimising nitrate leaching after ploughing temporary leguminous pastures in Canterbury, New Zealand. Journal of Contaminant Hydrology 20, 313-327.

Francis, G.S., Haynes, R.J., Sparling, G.P., Ross, D.J., Williams, P.H. (1992). Nitrogen mineralization, nitrate leaching and crop growth following cultivation of a temporary leguminous pasture in autumn and winter. Fertiliser Research 33, 59-70.

Francis, G.S., Haynes, R.J., Williams, P.H. (1995). Effects of the timing of ploughing-in temporary leguminous pastures and two winter cover crops on nitrogen mineralization, nitrate leaching and spring wheat growth. Journal of Agricultural Science 124, 1-9.

Francis, G.S., Bartley, K.M., Tabley, F.J., (1998). The effect of winter cover crop management on nitrate leaching losses and crop growth. Journal of Agricultural Science 131, 299-308.

Haygarth, P.M. Hepworth, L., Jarvis, S.C. (1998). Forms of phosphorus transfer in hydrological pathways from soil under grazed grassland. European Journal of Soil Science 49, 65-72.

Haygarth, P. M., and Jarvis, S. C. (1999). Transfer of phosphorus from agricultural soils. Advances in Agronomy 66, 195-249.

Heckrath, G. Brookes, P.C. Poulton, P.R., Goulding, K.W.T. (1995) Phosphorus leaching from soils containing different phosphorus concentrations in the Broadbalk Experiment. Journal of Environmental Quality 24, 904-910.

Heng, L.K., White, R.E., Bolan, N.S., Scotter, D.R. (1991). Leaching losses of major nutrients from a mole-drained soil under pasture. New Zealand Journal of Agricultural Research 34, 325-334.

Hickey, C.W. Quinn, J.M., Davies-Colley, R.J. (1989). Effluent characteristics of dairy shed oxidation ponds and their potential impacts on rivers. New Zealand Journal of Marine and Freshwater Research 23, 569-584.

Hooda, P.S. Moynagh, M. Svoboda, I.F. Edwards, A.C. Anderson, H.A., Sym, G. (1999). Phosphorus loss in drainflow from intensively managed grassland soils. Journal of Environmental Quality 28, 1235-1242.

Houlbrooke, D.J. and Monaghan, R.M. (2009). The influence of soil drainage characteristics on contaminant leakage risk associated with the land application of farm dairy effluent. AgResearch Client Report to Environment Southland, October 2009. 32p.

Houlbrooke, D.J. Horne, D.J. Hedley, M.J. Hanly, J.A., Snow, V.O. (2004a). A review of literature on the land treatment of farm-dairy effluent in New Zealand and its impact on water quality. New Zealand Journal of Agricultural Research 47, 499-511.

Houlbrooke, D.J. Horne, D.J. Hedley, M.J. Hanly, J.A. Scotter, D.R., Snow, V.O. (2004b). Minimising surface water pollution resulting from farm-dairy effluent application to mole-pipe drained soils. I. An evaluation of the deferred irrigation system for sustainable land treatment in the Manawatu. New Zealand Journal of Agricultural Research 47, 405-415.

James, E. Kleinman, P. Veith, T. Stedman, R., Sharpley, A. (2007). Phosphorus contributions from pastured dairy cattle to streams of the Cannonsville Watershed, New York. Journal of Soil and Water Conservation 62, 40-47.

Kleinman, P.J.A. Wolf, A.M. Sharpley, A.N. Beegle, D.B., Saporito, L.S. (2005). Survey of water-extractable phosphorus in livestock manures. Soil Science Society of America Journal 69, 701-708.

Ledgard, S.F. (2001). Nitrogen cycling in low input legume-based agriculture, with emphasis on legume/grass pastures. Plant and Soil 228, 43-59.

Ledgard, S.F. Penno, J.W. and Sprosen, M.S. (1999). Nitrogen inputs and losses from clover/grass pastures grazed by dairy cows, as affected by nitrogen fertiliser application. Journal of Agricultural Science, Cambridge 132, 215-225.

Ledgard, S.F. Finlayson, J.D. Gavin, J. Blackwell, M.D. Carran, R.A. Wedderburn, M.E., Jollands, N.A. (2003). Resource use efficiency and environmental emissions from an average Waikato dairy farm, and impacts of intensification using nitrogen fertiliser or maize silage. Proceedings of the New Zealand Grassland Association 65, 185-189.

Ledgard, S., Sprosen, M., Judge, A., Lindsey, S., Jensen, R., Clark, D., Luo, J. (2006). Nitrogen leaching as affected by dairy intensification and mitigation practices in the Resource Efficient Dairying (RED) trial. In 'Implementing sustainable nutrient management strategies in agriculture'. (Eds LD Currie, JA Hanly) pp. 263-268. (Fertiliser and Lime Research Centre, Massey University: Palmerston North, N.Z.).

Malcolm, B.J., Cameron, K.C., Di, H.J., Edwards, G.R., Moir, J.L., (2014). The effect of four different pasture species compositions on nitrate leaching losses under high N loading. Soil Use and Management 30, 58-68.

McDowell, R. W. (2010). Evaluation of two management options to improve the water quality of Lake Brunner, New Zealand. New Zealand Journal of Agricultural Research 53(1): 59-69.

McDowell R.W. (2009a). Maintaining good water and soil quality in catchments containing deer farms. Int. J. River Basin Mgmt 7:187-195

McDowell R.W. (2009b). The use of safe wallows to improve water quality in deer farmed catchments. New Zealand Journal of Agricultural Research 52, 81-90.McDowell (2009b). NZJAR 52: 81-90

McDowell, R.W. and Catto, W. (2005). Alternative fertilisers and management to decrease incidental phosphorus loss. Environ. Chem. Lett. 2, 169-174.

McDowell, R.W. and D.J. Houlbrooke (2009). Management options to decrease phosphorus and sediment losses from irrigated cropland grazed by cattle and sheep. Soil Use and Management 25(3): 224-233.

McDowell, R.W. and Nash, D., (2012). A review of the cost-effectiveness and suitability of mitigation strategies to prevent phosphorus loss from dairy farms in New Zealand and Australia. Journal of Environmental Quality 41, 680-693.

McDowell, R.W. and Smith, L.C., (2012). Potential water quality impact and agronomic effectiveness of different phosphorus fertilisers under grazed dairying in Southland. Proceedings of the New Zealand Grassland Association 74, 225-230.

McDowell, R.W, and Srinivasan, M.S. (2009). Identifying critical source areas for water quality: 2. Validating the approach for phosphorus and sediment losses in grazed headwater catchments. Journal of Hydrology 379, 68-80.

McDowell R.W. and Stevens D.R. (2008). Potential waterway contamination associated with wintering deer on pastures and forage crops. New Zealand Journal of Agricultural Research 51, 287-290.

McDowell, R.W. and Wilcock, R.J. (2008). Water quality and the effects of different pastoral animals. New Zealand Veterinary Journal 56(6), 289-296.

McDowell, R.W. and Wilcock, R.J. (2007). Sources of sediment and phosphorus in stream flow of a highly productive dairy farmed catchment. Journal of Environmental Quality 36, 540-548.

McDowell, R.W. and Wilcock, R.J. (2004). Particulate phosphorus transport within stream flow of an agricultural catchment. Journal of Environmental Quality 33, 2111-2121.

McDowell, R.W., B.J.F. Biggs, A.N. Sharpley, L. Nguyen (2004). Connecting phosphorus loss from agricultural landscapes to surface water quality. Chemistry and Ecology 20: 1-40.

McDowell, R.W. Monaghan, R.M., Morton, J. (2003a). Soil phosphorus concentrations to minimise P loss to surface waters in Southland. New Zealand Journal of Agricultural Research 46, 239-253.

McDowell, R.W., Drewry, J.J., Paton, R.J., Carey, P.L., Monaghan, R.M., Condron, L.M., (2003b). Influence of soil treading on sediment and phosphorus losses in overland flow. Australian Journal of Soil Research 41, 949–961.

Monaghan, R.M., Smith, L.C., Muirhead, R.W. (2014). Pathways and loads of N, P, sediment and Escherichia coli transfers from a grazed dairy pasture to water. Paper in preparation.

Monaghan, R.M., Smith, L.C., de Klein, C.A.M. (2013). The effectiveness of the nitrification inhibitor dicyandiamide (DCD) in reducing nitrate leaching and nitrous oxide emissions from a grazed winter forage crop in southern New Zealand. Agriculture, Ecosystems & Environment 175: 29-38.

Monaghan, R.M., Smith, L.C. (2012). Contaminant losses in overland flow from dairy farm laneways in southern New Zealand. Agriculture, Ecosystems & Environment 159:170-175.

Monaghan, R.M., Houlbrooke, D.J., Smith, L.C. (2010). The use of low-rate sprinkler application systems for applying farm dairy effluent to land to reduce contaminant transfers. New Zealand Journal of Agricultural Research 53: 389-402.

Monaghan, R.M., Semadeni-Davies, A., Muirhead, R.W., Elliott, S., Shankar, U. (2009a). Land use and land management risks to water quality in Southland. AgResearch Client Report for Environment Southland. 88 pages.

Monaghan, R.M., Smith, L.C., Ledgard, S.F. (2009b). The effectiveness of a granular formulation of dicyandiamide (DCD) in limiting nitrate leaching from a grazed dairy pasture. New Zealand Journal of Agricultural Research 52: 145-159.

Monaghan, R.M. Paton, R.J. Smith, L.C. Drewry, J.J., Littlejohn, R.P. (2005). The impacts of nitrogen fertilisation and increased stocking rate on pasture yield, soil physical condition and nutrient losses in drainage from a cattle-grazed pasture. New Zealand Journal of Agricultural Research 48, 227-240.

Monaghan, R.M. and Smith, L.C. (2004). Minimising surface water pollution resulting from farm-dairy effluent application to mole-pipe drained soils. II. The contribution of preferential flow of effluent to whole-farm pollutant losses in subsurface drainage from a West Otago dairy farm. New Zealand Journal of Agricultural Research 47, 417-429.

Muirhead, R.W., Monaghan, R.M. (2012). A two reservoir model to predict *Escherichia coli* losses to water from pastures grazed by dairy cows. Environment International 40, 8-14.

Muirhead, R.W. Collins, R.P., Bremer, P.J. (2006). The association of *E. coli* and soil particles in overland flow. Water Science & Technology 54, 153-159.

Myrbeck, Å., and Stenberg, M. (2014). Changes in N leaching and crop production as a result of measures to reduce N losses to water in a 6-yr crop rotation. Soil Use and Management, 30(2), 219-230. doi: 10.1111/sum.12118.

Oliver, D.M. Clegg, C.D. Heathwaite, A.L., Haygarth, P.M. (2005a). Assessing the potential for pathogen transfer from grassland soils to surface waters. Advances in Agronomy 85, 125-180.

Oliver, D.M., Heathwaite, A.L., Haygarth, P.M., Clegg, C.D. (2005b). Transfer of *Escherichia coli* to water from drained and undrained grassland after grazing. Journal of Environmental Quality 34, 918-924.

Orchiston, T.S., Monaghan, R.M., Laurenson, S. (2013). Reducing overland flow and sediment losses from winter forage crop paddocks grazed by dairy cows. In Accurate and efficient use of nutrients on farms (Eds L.D. Currie and C L. Christensen). http://flrc.massey.ac.nz/publications.html. Occasional Report No. 26. Fertiliser and Lime Research Centre, Massey University, Palmerston North, New Zealand. 7 pages.

Scholefield, D., Tyson, K.C., Garwood, E.A., Armstrong, A.C., Hawkins, J., Stone, A.C. (1993). Nitrate leaching from grazed grassland lysimeters: effects of fertilizer input, field drainage, age of sward and patterns of weather. Journal of Soil Science 44, 601-613.

Schoumans, O.F., Chardon, W.J., Bechmann, M.E., Gascuel-Odoux, C., Hofman, G., Kronvang, B., Rubæk, G.H., Ulén, B., Dorioz, J.M. (2014). Mitigation options to reduce phosphorus losses from the agricultural sector and improve surface water quality: A review. Science of the Total Environment 468–469, 1255-1266.

Sharpley, A.N. (1977). Sources and transport of phosphorus and nitrogen in a stream draining a dominantly pasture catchment. PhD thesis. Massey University, NZ.

Sharpley, A.N. and Syers, J.K. (1979). Loss of nitrogen and phosphorus in tile drainage as influenced by urea and grazing animals. New Zealand Journal of Agricultural Research 22, 127-131

Shepherd, M., Phillips, P., Snow, V. (2011). The challenge of late summer urine patches in the Waikato region. In Adding to the knowledge base for the nutrient manager. (Eds. L.D. Currie, and C.L. Christensen), Occasional Report No. 24. Fertiliser and Lime Research Centre, Massey University, Palmerston North, New Zealand. http://flrc.massey.ac.nz/publications.html

Shepherd, M.A., Stafford, A., Smeaton, D. (2012). The use of a nitrification inhibitor (DCn™) to reduce nitrate leaching from a winter-grazed forage crop in the Central Plateau. Proceedings of the New Zealand Grassland Association 74, 103-108.

Skaggs, R.W. and van Schilfgaarde, J. (1999). Agricultural Drainage. Number 38 in the AGRONOMY series of Monograhs. American Society of Agronmy, Inc.; Crop Science Society of America, Inc.; Soil Science Society of America, Inc; Madison, Wisconsin, USA.1328p.

Smith, L.C. and Monaghan, R.M. (2003). Nitrogen and phosphorus losses in overland flow from a cattle-grazed pasture in Southland. New Zealand Journal of Agricultural Research 46, 225-238.

Smith, L.C. and Monaghan, R.M. (2013). Comparing OVERSEER® estimates of N leaching from grazed winter forage crops with results from Southland trial sites. AgResearch Client Report RE500/2013/123 to Environment Southland, September 2013. 9p.

Smith, L.C., Orchiston, T., Monaghan, R.M. (2012). The effectiveness of the nitrification inhibitor dicyandiamide (DCD) for mitigating nitrogen leaching losses from a winter grazed forage crop on a free draining soil in Northern Southland. Proceedings of the New Zealand Grassland Association 74:39-44.

Syswerda, S.P., Basso, B., Hamilton, S.K., Tausig, J.B., Robertson, G.P. (2012). Long-term nitrate loss along an agricultural intensity gradient in the Upper Midwest USA. Agriculture, Ecosystems & Environment 149, 10-19.

Thorrold, B.S., Rodda H., Monaghan R.M. (1998). Modelling land use effects on water quality in the Oteramika catchment. Client Report to Southland Regional Council on the Oteramika Trial Catchment Project. 47p.

Ulén, B., Bechmann, M., Øygarden, L., Kyllmar, K. (2012). Soil erosion in Nordic countries – future challenges and research needs. Acta Agriculturae Scandinavica, Section B — Soil & Plant Science 62, 176-184.

van Rossum, M.H., Bryant, R.H., Edwards, G.R. (2013). Response of simple grass-white clover and multi-species pastures to gibberellic acid or nitrogen fertiliser in autumn. Proceedings of the New Zealand Grassland Association 75, 145-150.

Watson, C.J. Poland, P., Allen, M.B.D. (1998). The efficacy of repeated applications of the Urease inhibitor N-(n-butyl) thiophosphoric triamide for improving the efficiency of urea fertiliser utilization on temperate grassland. Grass and Forage Science 53, 137-145.

Wheeler, D., Ledgard, S.F., R.M., Monaghan, R.M., McDowell, R., de Klein, C. (2006). *Overseer* development – what it is, what it does. *In* Implementing Sustainable Nutrient Management Strategies in Agriculture. (Eds. L.D. Currie and J.A. Hanly). Occasional Report No. 19. Fertiliser and Lime Research Centre, Massey University, Palmerston North, New Zealand. pp. 231-236.