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Land use and land management risks to water quality in Southland

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Land use and land management risks to water quality in Southland

Report prepared for Environment Southland

April 2010

R M Monaghan, A Semadeni-Davies, R W Muirhead, S Elliott and U Shankar

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1. Summary

This review documents some of the effects of agricultural land use practices on water quality in Southland. This forms one of the information summaries that Environment Southland have requested as they prepare their Discharge Plan that will address discharges to land and the cumulative effects of intensive land use. In this report we (i) review the scientific literature and findings from on-going experimental trials that examine losses of the four contaminants specifically identified in Environment Southland's Water Plan (N, P, sediment and faecal micro-organisms), (ii) provide a comparative assessment of the relative risks of contaminant loss from different farming and management systems, (iii) provide an assessment of the cost, effectiveness and cost-effectiveness of some of the most promising Good Environmental Management Practices (GEPs) to mitigate the effects of farming on water quality, and (iv) make an assessment of the potential of some of the most cost-effective GEPs for improving water quality at a catchment scale, using the Oreti catchment as a case study example.

Research trials clearly show that subsurface drainage is the main pathway of N transfers from agricultural land to water, with nitrate accounting for between 80-90% of the dissolved N discharged in drainage. A review of the scientific literature indicates that nitrate leaching losses 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 (refer Tables 3.1-3.3 and Figure 4.1). For equivalent N inputs and cattle stocking rates, drainage N losses from free-draining soils tend to be greater than from poorly drained soils, due to greater soil denitrification rates in poorly drained soils. Grazed winter forage crops have been identified as having relatively large N leaching losses on a per hectare basis and are the subject of on-going research. Cropping systems also show a large potential to lose nitrate-N in drainage, although adherence to some key GEPs can greatly decrease these losses. In the case of P, sediment and faecal micro-organisms, additional pathways and sources contribute losses to water such as artificial subsurface drainage, overland flow and direct deposition. The yields of P, sediment and faecal micro-organisms discharged from land to water vary considerably depending on soil, climate and management factors. However, the variability introduced by these factors tends to obscure any obvious land use effects (refer Figure 3.2). Models are therefore important tools to account for variation in resources (soil, slope, rainfall) and management (e.g. nutrient inputs, FDE management, riparian protection) and thereby estimate nutrient and microbial losses from different farming systems.

Because individual contaminants have different sources and transport pathways, no single mitigation action can effectively decrease the losses of all contaminants at the same time, but some can have multiple benefits. Mitigations to decrease N losses need to target animal urine patches (nitrification inhibitors, off-paddock wintering) which are identified as the key source of N leached from grazed pastures. For cropping systems, 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 and using minimum tillage techniques. Management practices to decrease P and *E. coli* losses from pastoral land uses need to target stream fencing and the improved management of farm dairy effluent. Fortunately, many of the mitigation practices identified in this report will have multiple benefits. For example, in addition to significantly decreasing direct inputs of faecal P to streams, stream fencing will also decrease direct inputs of *E. coli* and N. Other benefits, such as protecting stream and riparian habitats from the harmful effects of bed and bank erosion due to animal treading, will also be incurred. However, it is also acknowledged that "best practice" in terms of on-farm management can only minimise losses to a point that is dependent on land use and soil and landscape features. Due to difficult-to-control pathways such as artificial drainage networks and/or large volumes of overland flow, contaminant losses to water from farms located on heavy soil types are likely to remain relatively high.

The effects of GEP uptake on water quality (TN, TP and *E. coli*) were modelled for the Oreti River catchment (3512 km^2) using the Catchment Land Use for Environmental Sustainability model version 3 (CLUES 3.0). This catchment modelling tool was used to compare the relative decreases in nutrient loads, yields and concentrations and *E. coli* loads that could be expected in the Oreti River catchment assuming the following mitigations were implemented: (i) stock exclusion (on LUC units 1-3), (ii) nitrification inhibitors (pastoral farms on LUC units 1-4), (iii) herd shelters (for dairy farms), (iv) improved farm dairy effluent (FDE) management and (v) the use of constructed wetlands. The main findings of this modelling exercise were:

- there was no one mitigation strategy that could substantially decrease all of the pollutant loads in the modelled scenarios;
- nitrification inhibitors (22% decrease) and herd shelters (16% decrease) were most effective for decreasing N losses in the modelled scenarios;

- stock exclusion (9% decrease), herd shelters (8% decrease) and improved FDE management (8% decrease) were most effective for decreasing catchment P losses in the modelled scenarios;
- wetlands (17% decrease) and improved FDE management (14% decrease) were most effective for decreasing catchment *E. coli* losses; and
- for greater decreases of all 3 pollutants, full stock exclusion will likely be required on higher LUC units. The implementation of combinations of GEPs will clearly also deliver greater decreases in pollutant loads.

In conclusion, this report's key messages are:

(i) the type of farming operation (e.g. sheep v. dairy etc) practised is an important determinant of contaminant losses from land to water,

(ii) some landscapes have greater risks of loss (e.g. sloping or poorly drained land) than others, and

(iii) these losses can be significantly modified according to land management, and

(iv) there remain a considerable number of knowledge gaps in our understanding of land-water transfers of stream contaminants (section 7).

2. Scope of Report

Environment Southland is currently preparing a Discharge Plan that will address discharges to land and the cumulative effects of intensive land use on water water quality. To assist with this project, AgResearch have been asked to prepare a report that documents the relative risks of different land uses and activities for water quality in the province. Accordingly, here we:

- Review the scientific literature to document losses of N, P, sediment and faecal bacteria to water measured in relevant experimental trials,
- Use the Overseer® Nutrient Budgeting Programme (hereafter referred to as *Overseer*) to provide a comparative assessment of the relative risks of N and P loss from different farming systems,
- Provide an assessment of the cost, effectiveness and cost-effectiveness of some of the most promising Good Environmental Management Practices (GEPs) that are currently available to mitigate the effects of farming on water quality, and

- Make an assessment of the potential of some of the most cost-effective GEPs for improving water quality at a catchment and regional scale.

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.

List of abbreviations used:

cfu – colony forming units; MPN – most probable number

CLUES – Catchment Land Use and Environmental Sustainability

CSA – critical source area

DCD – dicyandiamide (a nitrification inhibitor)

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

FDE – Farm dairy effluent

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

REC – river environment classification

SU/ha – stock units per hectare

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

GEPs - Good Environmental Practices. This termed is used to refer to management practices that deliver environmental benefit. This term is favoured above “Best Management Practice” (BMP) because “Best” implies better than others; in practice, often there are a number of management practices that can achieve good environmental outcomes. And “best” today may potentially be unsuitable in the future. A case in point is the 2-pond effluent treatment system promoted in the 1980s: although it represented a step forward in the management of FDE at the time, we now recognise that further improvements in FDE management can be achieved through the use of appropriate land application systems. Some of the GEPs of most relevance to Southland are described in Appendix I.

3. Pollutant losses to water from farming systems

3.1 Overview of contaminant pathways

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):

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. Overland flow is typically enriched in P (dissolved and particulate forms), sediment, faecal bacteria and ammonium-N, but little nitrate-N.

Subsurface flow (or drainage). That part of precipitation which infiltrates the soil and moves to streams or lakes as ephemeral, shallow, perched or ground water flow. In contrast to overland flow, subsurface drainage is usually the dominant pathway involved in the transfer of mobile pollutants such as nitrate from soil to water. In agricultural landscapes, the downward movement (or leaching) of subsurface flow can be intercepted by artificial drainage systems such as mole-pipe drains. Much local and international research has documented how this artificial drainage pathway can also deliver significant quantities of less mobile pollutants such as P, sediment and faecal bacteria to surface waters. Preferential flow through macropores to mole-pipe systems are attributes that allow these soil-water transfers to occur.

Variable (or Critical) Source Areas - it is recognised that many of the less mobile stream pollutants such as P, sediment, ammonium-N and faecal bacteria are not sourced from the entire catchment but instead from smaller areas within a catchment. The dominance of these small areas, often also 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 near stream channels or in low infiltration areas that are connected to the stream channel (McDowell and Srinivasan, 2009).

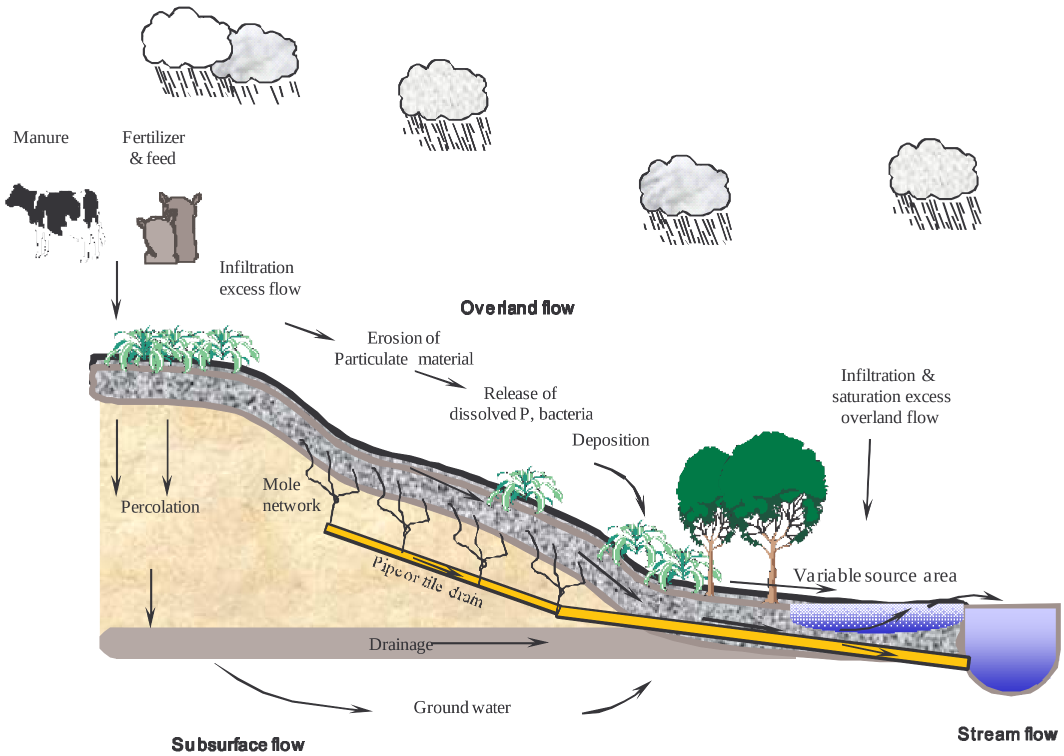


Figure 3.1. Conceptual diagram of processes that transport pollutants from the landscape to surface water (adapted from McDowell et al., 2004).

3.2 Sources of nitrogen

Within a grazed pastoral systems context, considerable research over the past four to five decades has helped us understand how nutrients cycle within agricultural ecosystems, and how we can optimize the efficiency of fertiliser nutrients for agronomic productivity. Earlier research on the recycling of nutrients in grazed pasture systems, reviewed by Haynes and Williams (1993), reinforced the concept that nutrients in pasture ingested by the grazing animal are inefficiently utilised in growth, or milk, meat and wool production. The majority of nutrients are excreted in dung or urine. Considerable research into N flows within grazed dairy pastures over the past three decades clearly shows that the amount of urinary N excreted by animals is 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 (e.g. Fraser et al. 1994). 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 on-set 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 (Ledgard et al., 1999a; 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 shallow stony soils compared to heavier-textured and/or poorly drained soils where a proportionally greater amount of soil nitrate is removed via denitrification processes (Scholefield et al., 1993).

Other non-pastoral land uses such as cropping and horticulture are also recognised as potentially significant sources of N losses to water. Their potential importance is also briefly described in sections 3.2.4 and 3.2.5 below.

3.2.1 N losses to water from cattle-grazed systems

There are 3 key dairy experimental sites examining nutrient losses to water that are of particular relevance to Southland: Edendale, Tussock Creek and Kelso. Although the latter is located in West Otago, it has a mole-pipe drained Pallic soil that is similar to many of those now used for dairy farming in Southland. Measured N losses from each of these multi-year trials under dairy grazing management are documented in Table 3.1. Some of the key points to note are:

- Nitrate-N represented between 80-90% of the dissolved N measured in drainage at the Edendale and Tussock Creek sites. Ammonium-N (3-4%) and dissolved organic N (8-10%) fractions were much smaller proportions of the total drainage N loads.
- Overland flow was not a major source of dissolved N exiting the Edendale (<1% of runoff N flux) or Tussock Creek (<10% of runoff N flux) sites.
- For equivalent N inputs and cattle stocking rates, drainage N losses from the more freely drained Edendale site (Woodlands-Waikoikoi soils) were greater than from the poorly drained Tussock Creek site (Pukemutu soil), most likely reflecting the greater soil denitrification rates in the Pukemutu soil.

- Mean nitrate-N concentrations in drainage waters exceeded 10 mg L^{-1} at the Kelso site (10.8 mg L^{-1}), and at the Edendale site for the treatments where annual N fertiliser inputs were 200 kg N $ha^{-1}year^{-1}$ or greater, and for the "grazed" winter forage (kale) site.

The large nitrate leaching losses of 52 kg N ha^{-1} $year^{-1}$ recorded from the winter forage crop trial 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 low. These losses are high relative to those measured under pasture and make a disproportionately large contribution to total dairy system losses, considering winter forage crops commonly make up 10-20% of farm area. The application of DCD had no significant effect on decreasing N leaching losses from grazed winter forage crops at the Southland site (data not shown). In Southland, winter grazing forms part of the pasture renewal cycle where a fraction of the farm goes into a 'break crop' for two or three years each time pasture is renewed. This cycle results in the following sequence of activities:

(i) Spray off/plough in pasture

(ii) Establish crop (essentially a vegetable crop);

(iii) Graze crop in winter (intensive break feeding over winter);

(iv) Fallow until spring (remaining nutrients lost from topsoil)

(v) Re-establish crop and repeat from (ii).

Thus 'winter grazing' is a combination of activities that all have some degree of associated risk. Unfortunately there is only a small amount of data available to better define these risks.

Leaching losses have also been reported for a number of other dairy grazing field studies undertaken elsewhere in NZ. Many of these have been reviewed by others (e.g. Figure 3.2 and Table 1 in McDowell & Wilcock, 2008) and only some of the details are reproduced here in Table 3.2. It is important to distinguish between published studies that report annual nitrate losses under a range of field treatments, and those that examine in detail the processes responsible for the leaching of nitrate-N from soil . The latter are important for helping us to understand some of the key drivers affecting soil processes and leaching rates, but care should be taken when extrapolating lab or small plot scales studies to losses at field and farm scales. Results reported in Tables 3.1 and 3.2 are from grazing systems studies and thus come closest to representing what would occur on equivalent commercial farms.

Table 3.1. Measured mean annual N losses in drainage and overland flow from the Edendale, Tussock Creek and Kelso dairy study sites.

Site	N fertiliser input or grazing treatment	Stocking rate cows ha^{-1}	Drainage NO_3^--N leached kg N ha^{-1} $year^{-1}$	Drainage NO_3^--N conc. mg N L^{-1}	Drainage NH_4^+-N leached kg N ha^{-1} $year^{-1}$	Drainage DON[a] leached kg N ha^{-1} $year^{-1}$	Overland flow NH_4^+ + NO_3^- kg N ha^{-1} $year^{-1}$	Total dissolved N loss to water kg N ha^{-1} $year^{-1}$	Reference
A. Pastures									
Edendale (1996-1999)	Nil fertiliser N	2.4	30	8.3	1.2	3	0.05	34	Monaghan et al. (2005)
	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			Smith & Monaghan (2003)
	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	-	-	
(2004-2007)	Control	2.6	13	3.1	0.9	-	-	-	Monaghan et al. (2009a)
	+DCD	2.6	7	1.7	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									
Tussock Creek (2006-2008)	~100 kg N/ha/yr	1200[b]	52	10.8	0.3	-	-	-	Monaghan, unpublished data

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

Table 3.2. Annual nitrate leaching losses reported for cattle-grazed pastures in other parts of NZ.

Location	Measurement method	Treatment	Annual rainfall, mm	N input kg N $ha^{-1}yr^{-1}$ Fertiliser	Fixation	Drainage mm	N leached[a] kg N $ha^{-1}yr^{-1}$	Reference
Waikato 1994-1996 (dairy)	ceramic cup samplers	Control, 3.3 cows/ha	1384	-	174	646	40 (6)	Ledgard et al. (1999)
		200N, 3.3 "		215	117	601	79 (13)	
		400N, 3.3 "		413	40	585	150 (26)	
		400N, 4.4 "		411	37	n/a	133	
Manawatu 1975 (dairy)	mole-pipe	120N	958	120	n/a	168	8 (5)	Sharpley & Syers (1979)
		60N		60	n/a	159	9 (6)	
2002-2003		2.5 cows/ha	949	~130	n/a	228	26 (11)	Houlbrooke et al. (2003, 2008)
Taupo 2003-2005 (cattle)	ceramic cup samplers	All year grazing	1447	60	n/a	702	13 (2)	Betteridge et al. (2007)
		No winter grazing					5 (1)	
		No grazing					3 (<1)	
2004-2006		Young (5-18 month old) cattle; approx. 13 SU/ha	1510	0	n/a	939	8 (1)	Hoogendoorn et al. (2009)
Canterbury (dairy; 5-year average)	lysimeters	Free-draining soil	n/a	n/a (but aim to use <200N)	n/a	n/a	22	Cameron et al. 2008
		Poor-draining " (~4 cows/ha)					20	

[a]Nitrate concentrations in leachate are shown in parentheses and calculated from amount leached and drainage depth. n/a= data not available

Table 3.3. Annual nitrate leaching losses reported for sheep-grazed pastures in NZ.

Location	Measurement method	Treatment	Annual rainfall mm	N input as Fertiliser kg N $ha^{-1} yr^{-1}$	Fixation kg N $ha^{-1} yr^{-1}$	Drainage mm	Nitrate leached[a] kg N $ha^{-1} yr^{-1}$	Reference
Taupo	ceramic cups	5-18 month old stock; approx. 15 SU	1510	0	n/a	939	8 (<1)	Hoogendoorn et al. (2009)
Manawatu 1990	cores	Grass/clover	1040[b]	-	144	270	6 (2)	Ruz-Jerez et al. (1995)
		Herbal ley		-	152		7 (3)	
		Grass +400N		400	0		41 (15)	
Manawatu 1988	mole-pipe	SSP-S	1088	-	n/a	304	13 (4)	Heng et al. (1991)
		Elemental S		-	n/a	257	9 (4)	
1989		SSP-S	837	50	n/a	120	19 (16)	
		Elemental S		50	n/a	100	15 (15)	
1990		Control	1014	-	n/a	266	35 (13)	Magesan et al. (1994)
		120N		120	n/a	236	43 (18)	
1991		Paddock A	1007	-	n/a	339	23 (7)	Magesan et al. (1996)
		Paddock B		-	n/a	300	17 (6)	
Manawatu	mole-pipe	Grazed +irrig.	958[b]	-	n/a	403	28 (7)	Turner et al. (1979)
		Grazed, -irrig.		-	n/a	318	7 (2)	
		Ungrazed, +irrig.		-	n/a	340	3 (<1)	
		Ungrazed, -irrig.		-	n/a	216	3 (1)	
Manawatu	lysimeters	Nil PS, 0N	1290	0	27	589	6[c]	Parfitt et al. (2009)
		Nil PS, 300N		300	8		18[c]	
		34P, 41S, 0N		0	87		24[c]	
		34P, 41S, 300N		300	10		114[c]	
Canterbury	cores	lightly-grazed pasture	660	-	n/a	n/a	5	Adams & Pattinson (1984)

[a]Nitrate concentrations in leachate are shown in parentheses and calculated from amount leached and drainage depth; [b]from NIWA records; [c]includes dissolved organic N

Table 3.4. Annual P and sediment losses in drainage, overland flow or stream flow from some grazed pastures/cropland and catchments in NZ; results from additional studies are also documented in McDowell & Wilcock (2008).

Site	Region	Main treatment	TP loss kg P ha^{-1} $year^{-1}$	DRP loss kg P ha^{-1} $year^{-1}$	Sediment loss kg ha^{-1} $year^{-1}$	Reference
Paddock losses						
Edendale	Southland	[a]Drained, nil N	0.37	0.12	nd	Monaghan et al.
(1996-1999)	(cattle)	[a]Drained, 400N	0.29	0.10	nd	(2005); Smith &
		[b]Un-drained, nil N	0.09	0.03	nd	Monaghan (2003)
		[b]Un-drained, 400N	0.23	0.06	nd	
Tussock Creek	Southland	[a]Drained	0.85	0.39	86	Monaghan, un-published
(2000-2003)	(dairy)	[b]Un-drained	0.53	0.22	47	data
Kelso	Otago	[c]Control	0.20	0.02	7	Monaghan & Smith (2004) &
(2000-2003)	(dairy)	[c]Effluent-treated	0.41	0.05	7	McDowell et al. (2005)
Windsor	Otago	[b,d]Cattle winter crop	0.71	0.23	330	McDowell & Houlbrooke
		[b,d]Sheep winter crop	0.47	0.14	180	(2008)
		[b,d]Sheep pasture	0.31	0.17	32	
Lumsden	Southland	[b]Deer winter crop	2	0.13	1,012	McDowell & Stevens (2008)
Catchment yields						
Bog Burn	Southland	Mixed landuse catchment	0.43	0.23	58	Monaghan *et al*. (2007)
Oteramika	Southland	Mixed landuse catchment	0.85	Nd	237	Thorrold *et al*. (1998)
Waikakahi	Canterbury	Predominantly dairy	0.83	0.56	46	Monaghan *et al*. (2009b)
Inchbonnie	Westland	Dairy	5.02	2.27	883	Wilcock *et al*. (2007)
Waiokura	Taranaki	Dairy	0.86	0.20	190	Wilcock *et al*. (2007)
Toenepi	Waikato	Predominantly dairy	0.93	0.35	53	Wilcock *et al*. (2007)
Glenomaru	Southland	Deer	0.21	0.03	88	McDowell (2009a)
Waimea	Southland	Deer	0.22	0.04	11	McDowell (2009a)
Telford	Otago	Deer	3.0	0.04	3,940	McDowell (2009b)
		Deer (wallows removed)	0.22	0.02	1,560	McDowell (2009b)
Invermay	Otago	Deer	0.8	0.03	4,480	McDowell (2007)

[a]combined losses in mole-pipe drainage and overland flow; [b]overland flow losses only; [c]mole-pipe drainage losses only; [d]expressed as approximate annual losses

Some key points to note in Table 3.2 are:

- The study by Ledgard et al. (1999) is the most comprehensive N cycling study undertaken in NZ. The relatively high N leaching losses reported can be attributed to relatively high rainfall, the free-draining nature of the ash soils and the relatively high level of pasture utilisation and milk production obtained. Leaching losses represented *c*. 22, 23, 33 and 26% of total farm N inputs in the Control, 200N, 400N and 400N-4.4 cows/ha treatments, respectively. The lower leaching losses observed in the 400N-4.4 cows/ha treatment were due to the use of maize silage imported onto the farm and shows how diet can modify N leaching rates.
- The other data generally show lower N leaching losses, most probably due to a combination of some of the following factors: less freely draining soils (esp. the sites where mole-pipe drainage collection systems were used), lower rainfall and/or fertiliser N inputs, and perhaps less well developed pastoral soils that have higher rates of N immobilisation.

Excluding treatments where either (i) N fertiliser inputs exceeded 200 kg N $ha^{-1}year^{-1}$, or (ii) N losses were mitigated through the application of DCD or implementation of a restricted grazing strategy, the calculated mean annual nitrate leaching loss from the trials reported in Tables 3.1 and 3.2 was 26 kg N ha^{-1}.

3.2.2 N leaching losses from pastures grazed by sheep

Annual nitrate leaching losses reported for sheep-grazed pastures in NZ are summarised in Table 3.3, along with some relevant soil and meteorological parameters. Six of the 8 studies reported were situated in Manawatu (all very near Palmerston North), one was located near Taupo and one in Canterbury. National coverage is thus poor. The studies reported by Heng et al. (1991) and Magesan et al. (1994 and 1996) relate to two mole- and pipe-drained paddocks where nitrate leaching, in response to a number of treatment applications, was measured over the years 1988 to 1991. Losses from un-grazed pastures were very low at 3 kg N ha^{-1} $year^{-1}$ for the study reported by Turner et al. (1979). This observation is supported by findings from lysimeter studies which also observe very little N leaching from lysimeters that do not have urine applied. Excluding treatments where N fertiliser inputs exceeded 200 kg N ha^{-1} $year^{-1}$, the calculated mean annual nitrate leaching loss from the trials reported in Table 3.3 was 16 kg N ha^{-1}.

3.2.3 N losses to water from pastures grazed by deer

Nitrogen losses to water from deer pastures have been reported in publications by McDowell & Wilcock (2008) and McDowell (2009). These document area-specific loads for streams draining catchments (4-280 ha) located in Southland and Otago. Unpublished data have also been collected at a deer grazing trial near Taupo (Coby Hoogendoorn *pers comm.*). Reported leaching losses range between 3 and 19 kg N ha^{-1} $year^{-1}$. These relatively low yields of N reflect the low intensity of extensive deer farming (but not finishing properties), which often occupies Class IV to VII land, and the fact that N returns in deer urine are closer to those of sheep than cattle.

3.2.4 N leaching losses from arable farms

To the authors' knowledge, most of the water quality-related research reported for arable cropping systems focuses on N leaching losses, and to a lesser extent, sediment. Much of this comes from research from 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 un-used fertiliser N.

Research undertaken in Canterbury has documented N leaching losses from arable cropping systems. Some of the pertinent points from this research are summarised briefly below:

- Francis et al. (1992) measured N leaching losses of 78, 40 and 5 kg N ha^{-1} following the cultivation of a temporary leguminous pasture in March, May or July, respectively. Similar findings reported by Francis et al. (1995a and 1995b) also highlighted the importance of ensuring that the length of fallow period between cultivation and the on-set of leaching is minimized as much as possible. Indeed, in the study reported by Francis et al. (1992), delaying cultivation of the pasture until spring did not cause any significant decrease in wheat yield or N uptake by the following crop. Francis et al. (1995b) reported leaching losses from the March fallow treatment ranged from 72-106 kg N ha^{-1}. In contrast, losses from the May fallow treatment ranged from 8-52 kg N/ha.

- Francis et al. (1994) compared N leaching losses from fallow fields and leguminous and non-leguminous grain crops. Leaching losses declined in the order fallow > legumes > non-legumes (110 > 72 > 37 kg N/ha, respectively).
- A long term tillage trial, known as the Millennium Tillage Trial, found that the presence of winter cover crops had a marked effect on N leaching losses. Over a seven-year period (2001-2007), winter N leaching averaged 28 kg N ha^{-1} $year^{-1}$ under a range of different tillage practices (intensive, minimum and no-tillage) from winter fallow soil. The presence of winter cover crops decreased this 7-year average loss to 15 kg N ha^{-1} $year^{-1}$ (FAR, 2008). When comparing tillage methods, intensive tillage followed by winter fallow resulted in the highest N leaching losses (30 kg N ha^{-1} $year^{-1}$). However, N leaching was greatest from no-tillage treatments when comparing soils supporting winter cover crops. This result was thought to be due to the poor winter crop establishment on the no-tillage plots due to slug damage.
- Thomas et al. (2005) provide some modelled estimates of N leaching losses from "typical" cropping rotations in Canterbury. Over the whole rotation, the average N leaching loss estimated by the *Overseer* model was *c*. 48 kg N ha^{-1} $year^{-1}$, equivalent to about 34% of the fertiliser N applied.
- Using the GLEAMS simulation model, Lilburne et al. (2003) demonstrated the importance of having plant uptake of N during autumn and winter months to minimise N leaching losses under wheat production. This work also recognised the greater N leaching risk posed by cropping on shallow soil types.

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 techniques and the application of the nitrification inhibitor DCD (Francis, 1995; Francis et al., 1998; FAR 2008). These practices are particularly recommended for cropping on shallow soils. On a positive note, research suggests that well-managed cropping systems should leach relatively little nitrate-N if close attention is paid to the above management considerations.

3.2.5 N leaching losses from other land uses

Vegetable and bulb production

Intensive field vegetable production systems have the potential to lose very large amounts of N via nitrate leaching. This is due to the ample quantities of fertiliser which are often used to grow the crop and the large amounts of N that can be left behind in crop residues. The net result is that large amounts of residual soil mineral N often remain in the soil after harvest of the crop. There are, however, surprisingly few published studies where direct measurement of N (or P) losses to water have been made from such production systems.

Neeteson et al. (1999) document the amounts of residual soil mineral N left behind following a range of vegetable crops and use a computer model to derive estimates of nitrate-N leaching during the following winter and spring. Crops such as spinach, leeks, celeriac and cauliflower were shown to leave large amounts of residual soil mineral N in the soil following harvest, generally ranging between 50 and 220 kg N ha^{-1}. Modelling analysis suggested that subsequent leaching losses from these crops were also very high and of a similar magnitude to the amount of soil mineral N left in the soil post-harvest.

More locally, Francis et al. (2003) document N leaching losses from winter potato and winter greens (spinach, cauliflower or cabbage) grown near Pukekohe. On average, potato fields received the greatest amount of N fertiliser (481 kg N ha^{-1}), had the greatest soil mineral N content in June (184 kg N ha^{-1}) and had the greatest leaching loss (114 kg N ha^{-1}). These leaching losses were *c*. 7.5-fold greater than nitrate-N leaching losses measured on a nearby dairy farm. Losses from the winter greens were intermediate between the potato and dairy land uses. The large leaching losses from the winter crops were attributable to the large applications of fertiliser N before winter and the rapid mineralisation of residues from the previous greens. The authors noted that the measured losses from the potato and spinach crops were similar to those measured in other studies (P.H. Williams, unpubl. data & Williams et al. 2000).

Bulb production is a growing industry in Southland. Unfortunately we have a very poor understanding of the likely impacts of this land use activity on water quality in the province. As for cropping systems, judicious tillage and fertiliser management practices may be able to minimise the risks associated with this farming activity.

Forestry

Catchment monitoring studies indicate that N leaching losses from forestry land uses vary according to land use history, the management of the forest and the time elapsed since planting. It is generally believed that changing land use from pasture to forestry decreases nutrient losses from the land. Decreased fertiliser inputs, decreased rates of nitrogen fixation and soil erosion, and removal of the grazing animal are key factors that decrease nutrient yields from forested catchments (Quinn & Ritter 20003). Although logging of the forest does increase nutrient yields, this effect is generally short-lived and does not negate the benefits of decreased yields compared to pastoral land if assessed over the life-time of the forestry rotation (i.e. planting, forest growth, logging and re-planting) (Quinn & Ritter 2003). Evidence from recent Environment Court proceedings concerning the effects of land use on Lake Taupo water quality provides a useful summary of the likely N yields to water from forested land (Environment Court 2008). During the science caucusing process, scientists generally agreed that pines planted into improved pasture may lose between 8 to 12 kg N $ha^{-1}year^{-1}$. This will eventually decrease to a long-term equilibrium rate of 2-3 kg N ha^{-1} $year^{-1}$. These long-term forest losses are small compared to losses assigned to other land uses in the Lake Taupo catchment (e.g. 29 and 9 kg N ha^{-1} $year^{-1}$ for dairy and non-dairy pastures, respectively). Given the higher rainfall and free-draining pumice soils in the Taupo region, we would also note that the N yields reported above for Taupo are likely to be higher than expected for Southland.

Shrub-land

The above Environment Court proceedings also drew attention to the role that woody nitrogen-fixing species such as gorse and broom may have as sources of N leaching into Lake Taupo. Unpublished technical evidence collected by Scion suggests that N leaching losses from land covered in gorse may be a significant source of N in catchments that has been over-looked until now. However, we note that the assigned N leaching rates for woody nitrogen-fixing species such as gorse and broom used in the Lake Taupo decision (Environment Court 2008) were the same as for unimproved land i.e. 2 kg N ha^{-1} $year^{-1}$.

3.3 Sources of phosphorus & sediment

Sources of P losses from dairy farms tend to vary more than for N. Phosphorus losses depend heavily on spatial and temporal factors and the type of management practices employed on farm, such as how FDE 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 fertilizer additions (McDowell et al., 2007). These authors estimated the relative proportions of fertilizer-, dung-, plant- and soil-derived P lost from a non-effluent paddock at an Olsen P of 30 mg kg^{-1} to be 10, 30, 20 and 40%, respectively. Wide variations in these proportions will occur depending on soil Olsen P concentration, stocking rate and the timing of fertilizer applications relative to runoff. All except from fertilizer additions 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, FDE pond treatment systems, and FDE/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).

Non-pastoral land uses can also be important sources of P and sediment, particularly via soil erosion. This is discussed further in section 3.3.2.

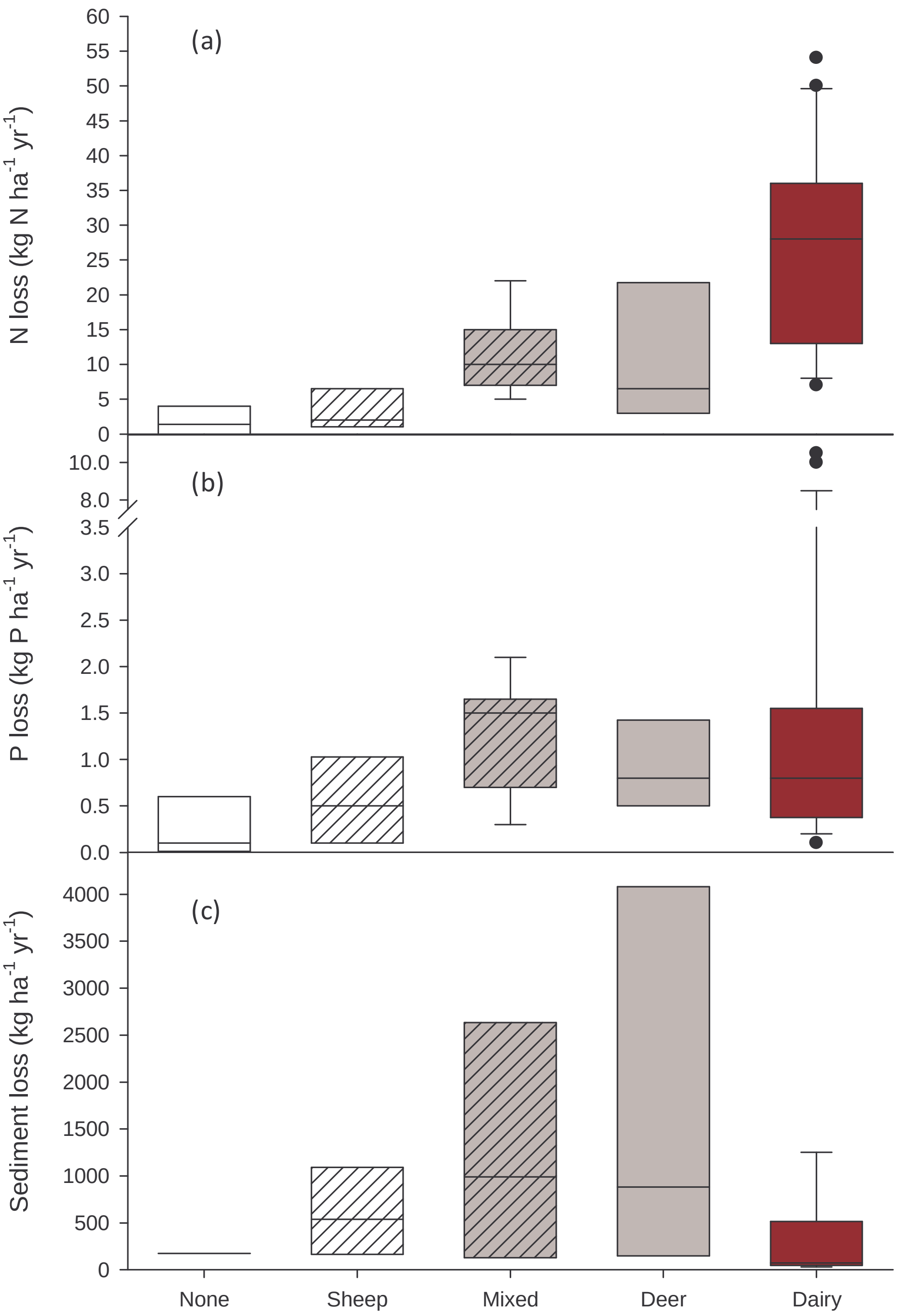


Figure 3.2. Box plots showing the median, bounded by the 25th and 75th percentiles, the 10th and 90th percentiles as whiskers, and any outliers as dots, for (a) N, (b) P, and (c) sediment annual loads for each class of land use. "None" refers to non-agricultural rural land uses, such as exotic plantation and native forest, while 'mixed' refers to a catchment with more than one type of land-use class (from McDowell & Wilcock 2008).

Unlike N, overland flow accounts for a large proportion of the total P lost from dairy farms. Although overland flow volumes are usually small relative to the volumes of water discharged in sub-surface drainage, the enrichment of P in the topsoil relative to deeper soil layers and 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 understandings suggest that near stream areas are important sources of overland flow, as are impervious areas connected to the stream and areas of land underlain by artificial drainage systems (McDowell and Srinivasan 2009). Where artificial sub-surface drainage systems exist, loads of P lost can still be the same as losses in overland flow from un-drained land, presumably due to the entrainment of particulate and dissolved P as water moves through the macropores and fissures to tile or pipe drains (Haygarth et al., 1998; Hooda et al., 1999; Monaghan et al., 2005; Sharpley & Syers 1979).

3.3.1 Documented losses of P and sediment from pastoral farms

In contrast to N, fewer studies have been undertaken to document annual yields of P and sediment loss from pastoral farms. Those that have been reported in the literature are summarised in Table 3.4. Also shown in Table 3.4 are the measured losses of P and sediment from trial work undertaken in Southland by AgResearch but not yet published in the science literature. Some of the key points to note in Table 3.4 are:

- Overland flow was not a major source of P in runoff exiting the drained plots at Edendale, contributing only 10% of the Total P load recorded. However, where plots where left un-drained and received high inputs of N fertiliser (and consequently high stocking rates), total P losses in overland flow increased to *c*. 0.23 kg P $ha^{-1} year^{-1}$.
- Mole-pipe drainage at the Tussock Creek site discharged approximately 35 kg sediment ha^{-1} $year^{-1}$. This represented 40% of the total sediment discharge from the drained plots, with the balance (51 kg ha^{-1} $year^{-1}$) coming via overland flow. The mean annual loss of sediment in overland flow from un-drained plots at this site was 47 kg ha^{-1}, despite overland flow volumes in this treatment (67 mm $year^{-1}$) being greater than in drained plots (54 mm $year^{-1}$).
- In contrast to Tussock Creek, total P and sediment losses in mole-pipe drainage from the dairy pasture (no FDE applied) at the Kelso site were 0.20 and 7 kg ha^{-1} $year^{-1}$, respectively. For the FDE-treated plot at this site, mean annual

total P losses increased to 0.41 kg ha^{-1}, due mostly to the preferential flow of some of the applied FDE through the mole-pipe network.

- Dissolved reactive P (DRP) represented between 14-55% (mean of 33%) of the paddock-scale P losses from pastures (excluding deer) reported in Table 3.4. Of the catchment-scale P yields (excluding deer catchments) reported in Table 3.4, this range was 23-67% (mean of 45%).
- Some of the other important treatment and landuse contrasts evident in Table 3.4 are:
    - P and sediment losses from cattle-grazed winter forage crops were greater than from sheep-grazed winter forage cropland.
    - These losses from sheep-grazed winter forage cropland were greater than from sheep-grazed pastures.
    - P and sediment losses from land grazed by deer were particularly high at up to 3 and 4,480 kg ha^{-1} $year^{-1}$, respectively. These reflect the large amounts of erosion that deer can potentially cause, particularly when allowed access to streams and wet areas. Accordingly, the mean ratio of DRP:TP loads for the 5 deer catchment studies reported in Table 3.4 was 0.093. Losses from winter forage crops grazed by deer were also particularly high and probably exacerbated by the lack of ground cover and poor surface soil condition due to hoof treading.
    - Poorly-timed applications of FDE to the Kelso trial site increased P losses from 0.20 to 0.41 kg $ha^{-1}year^{-1}$.
    - The very high rainfall (4,500 mm $year^{-1}$) in the Inchbonnie catchment (West Coast) contributed to the very high P and sediment yields recorded.
    - Although leaving the soils at Edendale and Tussock Creek in an un-drained state did increase P losses in overland flow, total P discharges in overland flow and mole-pipe drainage from these un-drained treatments were lower than from the equivalent drained soils. This suggests that installing mole-pipe drainage systems in poorly-drained soils in Southland is likely to slightly increase P losses, rather than decrease losses.

Some of the key conclusions and practical implications from the studies reported above are:

(i) Mole-pipe drainage can make a relatively large contribution to P losses from drained soils.

(ii) These losses are exacerbated by poorly-timed applications of FDE.

(iii) In terms of P losses, a general order of ranking would be: deer pasture & winter forage cropland > cattle-grazed winter cropland > sheep-grazed winter cropland = FDE-treated pasture> cattle-grazed pasture > sheep-grazed pasture.

(iv) For sediment, this ranking would be: deer > cattle-grazed winter cropland > sheep-grazed winter crop-land > cattle-grazed pasture = sheep-grazed pasture.

The above rankings are broadly consistent with those evident in the box plots presented by McDowell & Wilcock (2008; Figure 3.2). These box plots also show that there are very large ranges in reported losses of N, P and sediment within any given land use. These large ranges indicate how resource (soil, slope, rainfall) and management (e.g. nutrient inputs, riparian protection) risks combine to greatly modify pollutant losses for any given land use. Due to these resource and management factors, models that account for this variability are essential for predicting losses from farming systems.

3.3.2 Losses of P and sediment from non-pastoral land uses

In a review of P transfers from agricultural land, Haygarth & Jarvis (1999) note that tillage practices on cropped land can have a direct effect on P losses. The loss of crop cover is a particularly important issue that can leave bare soil exposed to the erosive force of rain drop impacts. Although intensive cultivation can disturb the soil increasing the risk of erosion, the stratification and enrichment of P in the topsoil, and subsequent overland flow, means that minimum tillage may not yield as much benefit as some think. However, contour ploughing is a simple precaution that can help to decrease runoff and therefore the risk of P loss. The reported losses of P from tilled or cropped land that were documented by Haygarth & Jarvis (1999) ranged between 0.1 and 6.2 kg P ha^{-1} $year^{-1}$. The latter yield was recorded in surface runoff from a corn field in Ohio, USA. The application of manures to cropland can also increase the risk of P transfer to water, either due to large losses that may occur shortly after applying manure to land, or due to the build-up of soil P concentrations following repeated applications of manure-P to land. This latter risk is a particular concern for many feedlot farms in the USA, where insufficient land may be available to avoid applying the large amounts of manure-P generated by the confined animals (pigs and poultry in particular).

Forestry blocks can also be important sources of P and sediment loss to water. Some documented losses are reported below:

- Cooper & Thomsen (1988) report P and sediment losses from the Purukohukou native forest of 0.12 and 27 kg ha^{-1} $year^{-1}$, respectively. They observed that although total P yields from the pine catchment were 4-fold higher than from native forest in the 5 years after logging, these were still 4-fold lower than recorded for pasture.
- Quinn & Stroud (2002) report P and sediment losses from a native forest catchment at Whatawhata of 0.58 and 320 kg ha^{-1} $year^{-1}$, respectively.
- Quinn & Ritter (2003) observed how increases in catchment N and P yields after logging are short-lived and that high yields in year 1 are compensated for by lower yields in subsequent years. In a comparison of pine, native and pasture catchments, they noted that P losses from pines in the 9-12 year period after logging were *c*. 18-fold lower than recorded for the pasture catchment..Losses from the native forest catchment were broadly similar to those lost from the pine.
- Wilcock (1986) reports median specific yields of P from native forests, un-disturbed exotic forests, and disturbed (i.e., recently logged) exotic forests of 0.2, 0.1 and 0.5 kg ha^{-1} $year^{-1}$, respectively. The reported equivalent median yields for sediment losses were 300, 500 and 700 kg ha^{-1} $year^{-1}$, respectively.

3.4 Sources of faecal bacteria

There are even fewer studies, compared to N and P, that have documented losses of faecal micro-organisms from pastoral farms. For reasons of cost and sampling logistics, *Escherichia coli* (*E. coli*) is the microbial organism most commonly measured in runoff waters and is used as an indicator of faecal pollution rather than an indicator of pathogenic risk *per se*. There are a number of guidelines that are used as threshold guidelines for microbial water quality. Some of the most frequently used ones are indicated below:

- Water used for human consumption should have no faecal micro-organisms present.
- Concentrations of faecal coliform bacteria in stock drinking water should not exceed 1000 per 100 ml.
- The 95^{th} percentile of samples taken from water bodies used for contact recreation should not exceed 260 *E. coli* per 100 ml.

The concentrations and loads of *E. coli* in mole-pipe drainage have been measured at the Kelso and Tussock Creek experimental sites. These measurements were also made on overland flow samples collected at the latter site to identify whether mole-pipe drainage or overland flow was the more important pathway of *E. coli* losses at a paddock scale. The main treatment (un-replicated) effect evaluated at the Kelso site was the addition of FDE to one of the plots. Over the first measurement period (2001-2003), the load and volume-averaged concentration of *E. coli* in mole-pipe drainage from the plot that did not receive FDE was 9.2×10^{10} cfu ha^{-1} $year^{-1}$ and 520 cfu 100 ml^{-1}, respectively. For the effluent-treated plot, these values increased to 1.7×10^{11} cfu ha^{-1} $year^{-1}$ and 9,540 cfu 100 ml^{-1}, respectively. When a low rate (K-line) sprinkler system, instead of a travelling irrigator, was used to apply FDE to both plots throughout 2004-2007, mean annual loads and volume-averaged concentrations of *E. coli* decreased to 4.1×10^{10} cfu ha^{-1} $year^{-1}$ and 2,100 cfu 100 ml^{-1}, respectively. Results from this experimental work were used to construct the EFFDRAIN model which has been used to predict farm scale-equivalent loads of effluent pollutants lost in mole-pipe drainage under contrasting scenarios of pond storage and FDE application depths and rates (Monaghan & Smith 2004; Monaghan et al. 2010, *in press*).

The concentrations and loads of *E. coli* in mole-pipe drainage and surface runoff from the experimental plots at Tussock Creek have been continuously measured over the 2001-2003 period. This experimental site is managed as an established dairy pasture stocked at 2.7 cows ha^{-1}. In contrast to the Kelso site, no applications of FDE have been made to the trial plots. Figure 3.3 shows how *E. coli* concentrations in both drainage and surface runoff flows were (i) relatively high, and (ii) generally decreased as the interval between grazing and subsequent drainage or runoff events increased. Surprisingly, there did not appear to be any significant relationship between event size and *E. coli* concentration (data not shown). The mean volume-averaged concentrations of *E. coli* in collected mole-pipe drainage and surface runoff flows were 4.9×10^{3} and 2.6×10^{4} cfu 100 ml^{-1}, respectively. Of the calculated mean paddock yield of 2.1×10^{11} cfu ha^{-1} $year^{-1}$, approximately 66% of the yield was derived from surface runoff with the remainder coming via mole-pipe drainage. These results demonstrate the large contribution that surface runoff makes to paddock scale losses of faecal micro-organisms. Leaving the soil un-drained increased *E. coli* yields in surface runoff by 23%, although is likely to decrease *E. coli* yields in subsurface drainage. Although subsurface drainage losses of *E. coli* were not measured in the un-drained soil treatment, we might conclude that, while leaving the soil at Tussock Creek in an un-drained state is likely to increase *E. coli* losses in surface runoff, it is unlikely to significantly increase paddock yields overall.

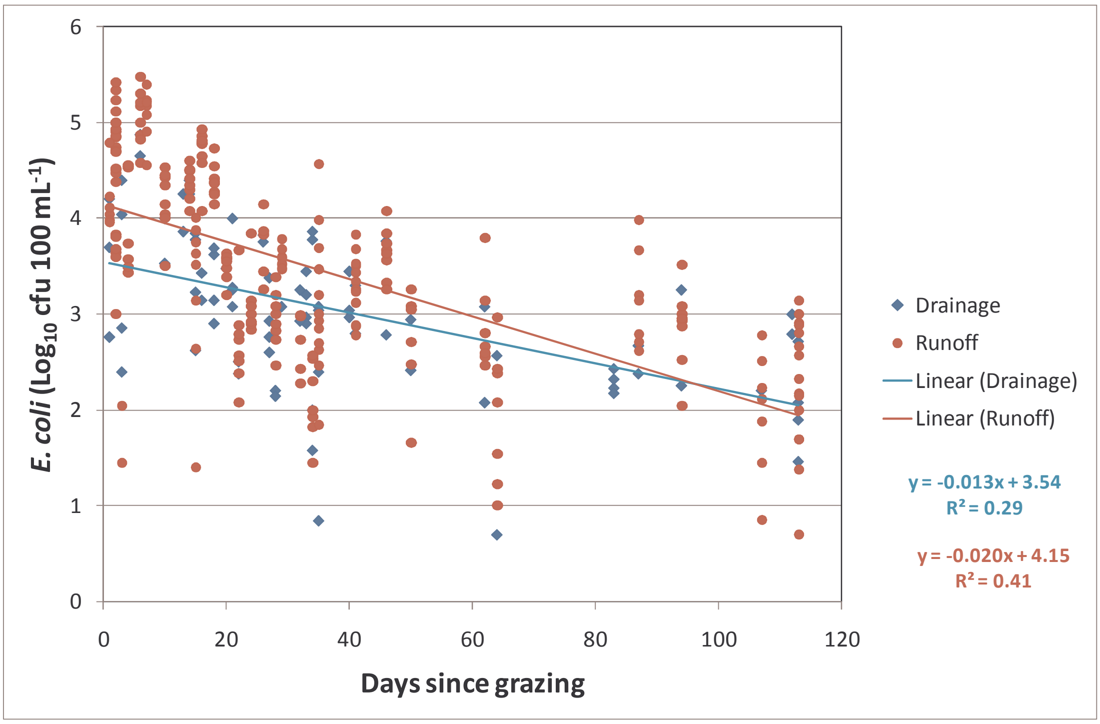


Figure 3.3. *E. coli* concentrations in mole-pipe drainage and surface runoff from a Southland dairy pasture (Tussock Creek experimental site, 2001-2003; Monaghan et al. *unpublished data*).

Whilst relatively high concentrations of *E. coli* have been measured in drainage and surface runoff at the Kelso and Tussock Creek sites, it should be noted that most of these flow events typically occur from late autumn until mid spring. Discharges during summer months, when river and stream recreational use is highest, are fortunately infrequent. The environmental risks posed from winter and spring discharges of *E. coli* in farm runoff are difficult to assess. However, flood events have very poor microbial water quality, probably due to *E. coli* reservoirs in stream sediments (Muirhead et al., 2004; Davies-Colley et al., 2008), but this moves rapidly through the stream network. As a general rule, storm-flow loads are important for water quality in receiving water bodies such as lakes and coastal areas where they may affect fishing, shellfish collection or aquaculture, whereas base-flow loads are important for water quality in the stream itself (Muirhead et al., 2008).

Another factor that complicates our understanding of *E. coli* (and P) loss from farms is the fact that not all areas of the landscape contribute to flow pathways of loss. Those that do are termed critical source areas and are characterised as being directly “connected” to water bodies (McDowell & Srinivasan 2009). Examples of critical source areas for *E. coli* are those areas of the landscape that have mole-pipe drained soils, areas where overland flow can escape directly to streams, farm tracks and deer wallows (e.g. Monaghan & Smith 2009; McDowell, 2008). To complicate things even further, risk modelling analysis by Muirhead et al. (2010) suggests that water fowl can also be a potentially significant source of faecal bacteria in rural streams (Figure 3.4). The context surrounding this analysis suggests inputs by water fowl become important when (i) currently recommended GEPs, such as improved effluent management, stock exclusion and the elimination of stock crossings, have been implemented, and (ii) large populations of fowl inhabit stream reaches.

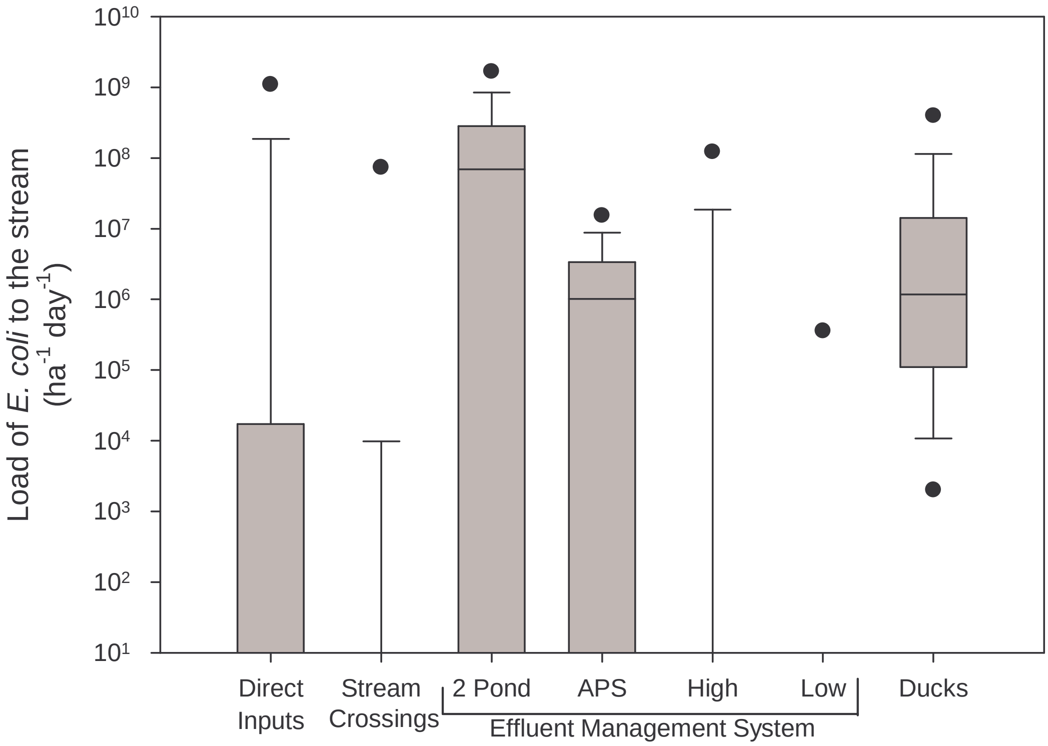


Figure 3.4. Distributions of the modelled daily loads of *E. coli* discharged to the stream from different farm managements or from ducks living in the stream. Horizontal lines are the median values, boxes are the 25th to 75th percentiles, whiskers the 10th and 90th percentiles and points the 5th and 95th percentile values. Modelling analysis from Muirhead et al. (2010) and based on an average of 4 ducks per stream kilometre.

Recent studies have also documented some of the risks associated with intensive land management and faecal pollution of ground and surface waters elsewhere in New Zealand. Close et al. (2008) report some of the effects of intensive dairy farming and border dyke irrigation on *E. coli* and *Campylobacter* concentrations in shallow groundwater within the Waikakahi catchment, South Canterbury. Of the 135 groundwater samples taken, *E. coli* and *Campylobacter* were detected in 75% and 12% of the cases, respectively. Border dyke irrigation was identified as an important factor contributing to the increased risk of infection from *Campylobacter* during the irrigation season. Studies by Malcolm Mcleod from Landcare Research have identified coarse soil structure (large structural cracks) or soils with a drainage impediment (containing wetting and drying cracks) as contributing to the rapid transport of faecal micro-organisms (contained in applied FDE) via preferential flow (McLeod et al. 2008, McLeod et al. 2004, Aislabie et al. 2001). Based on this work, some of the soil orders/subgroups and soil characteristics identified as having a high risk of preferential flow include:

- Organic soils,
- Ultic soils
- Granular soils
- Melanic soils
- Podzol soils
- Gley and perch-gley soils
- mottled subsoils
- peaty soils
- skeletal and pedal soils
- soils with a slowly permeable layer
- soils with coarse soil structure
- soils with a high K_{SAT}:K_{-40} ratio.

These risk attributes are discussed in more detail within a Southland context in the report by Houlbrooke & Monaghan (2009; pages 15 and 16).

Animal access to streams for drinking or crossing provides a direct input of faecal material to water (McDowell, 2008; Davies-Colley et al., 2004). Ruminant faeces are a concentrated source of faecal indicator organisms: concentrations ranging from 2-200 x 10^5 and 2-400 x 10^4 cfu g^{-1} wet faeces have been reported for cattle and sheep faeces, respectively (McDowell et al., 2008). Assuming each cattle and sheep faecal deposit weighs approximately 2 and 0.1 kg, respectively (Haynes & Williams, 1993), these values translate to loads of up to 8 x 10^9 and 2 x 10^9 cfu per faecal pat, respectively. These relatively large numbers demonstrate how direct deposition of faecal material into streams can increase the concentrations of faecal micro-organisms in water-ways draining pastoral land (Muirhead et al., 2008).

Wilcock et al. (2007) document concentrations of *E. coli* that were measured in streams draining 5 predominantly dairy farmed catchments in NZ. Median concentrations for the Toenepi (Waikato), Waiokura (Taranaki), Pigeon (West Coast), Waikakahi (Canterbury) and Bog Burn (Southland) catchments were 367, 1250, 640, 290 and 530 MPN 100 ml^{-1}, respectively. Suggested key sources of this stream pollutant included discharges from 2-pond effluent treatment systems (Toenepi & Waiokura), irrigation wipe-off water (Waikakahi), mole-pipe drainage (Bog Burn) and surface runoff from grazed pastures (all catchments).

4. Relative risks of different land uses and activities

This section makes an assessment of the relative risks posed by contrasting land use activities in Southland. For N and P losses, we use the *Overseer* model to make some projections of N and P losses from contrasting land use scenarios within the setting of the Bog Burn catchment. The lower part of this study catchment is covered by the poorly-drained Pukemutu soil, which is also present at the Tussock Creek experimental site. Modelled N losses are also presented for the Oteramika catchment in Eastern Southland, where the more freely-drained Edendale silt loam is the major soil group present.

In the case of *E. coli* losses, we use an inventory of identifiable potential sources to indicate relative losses from some of the model farms evaluated. As stated earlier, models are important tools that are needed to account for some of the resource (soil, slope, rainfall) and management (e.g. nutrient inputs, FDE management, riparian protection) risks that are known to contribute to the variability observed in nutrient and microbial losses from different farming systems.

Overseer estimates of N and P losses for model farm systems set within the Bog Burn catchment are shown in Figure 4.1. For this exercise we assumed that the hypothetical deer units would be located on the steeper country in the head-waters of the catchment. Farms are loosely based on data reported in MAF Monitor Farm reports (MAF 2007), Monaghan et al. (2007), literature values and local knowledge.

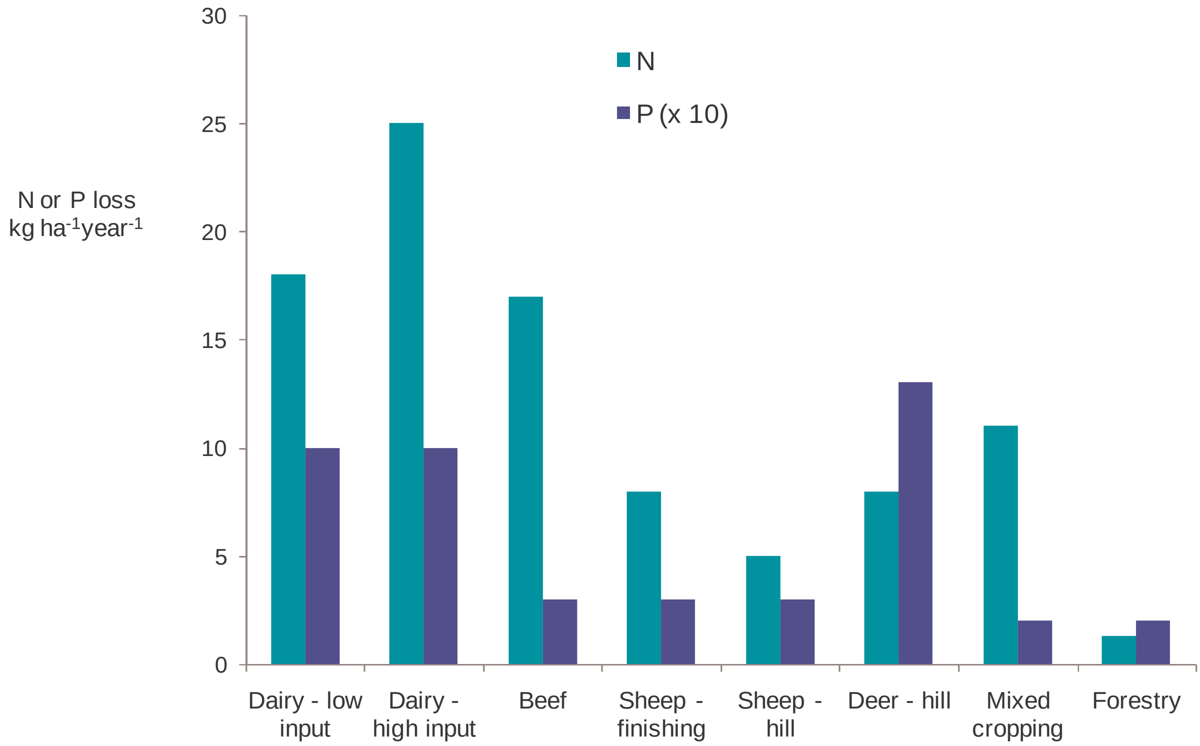


Figure 4.1. Modelled estimates of N and P (x 10) losses for contrasting model farm types set within the Bog Burn catchment, Southland.

In terms of N losses, it is evident that losses are greater for cattle than for sheep or deer. Losses are also lower for the less intensively farmed sheep hill country farm than an equivalent lowland finishing unit. These relative losses reflect the role that animal urine has as a source of N loss from pastoral farms; in addition to having a larger urine patch and greater N loading within a cow urine patch, dairy farms also tend to have higher rates of pasture production, and thus higher stocking rates and more urine patches per unit area. The net consequence of these greater loadings and numbers of urine patches is a greater N leaching loss. Nitrogen losses from deer systems tend to be similar to those from equivalent sheep farms due to similar rates of N intake and excretion.

Although not dealt with explicitly in this report, the main effect of irrigation on contaminant losses is to increase animal stocking rates and thus urinary returns. The net consequence of this is an expected increase in the amounts of N leached from pastures. First principles would suggest that this irrigation x stocking rate response is also likely to lead to increases in losses of faecal micro-organisms, although there is little data to support this. McDowell & Rowley (2008) showed that increasing frequency of flood irrigation or stocking rate increased P losses in wipe-off (range 0.7 to 12.6 kg P ha^{-1} $year^{-1}$).

Findings from the Oteramika Catchment study (Thorrold et al 1998) show a broadly similar pattern of N loss from contrasting farm types (Figure 4.2). This study also considered some additional sources of leached N, such as the irrigation of whey effluent from the Edendale dairy factory and from septic tanks. The latter source was estimated to make a small but significant contribution to whole catchment N losses.

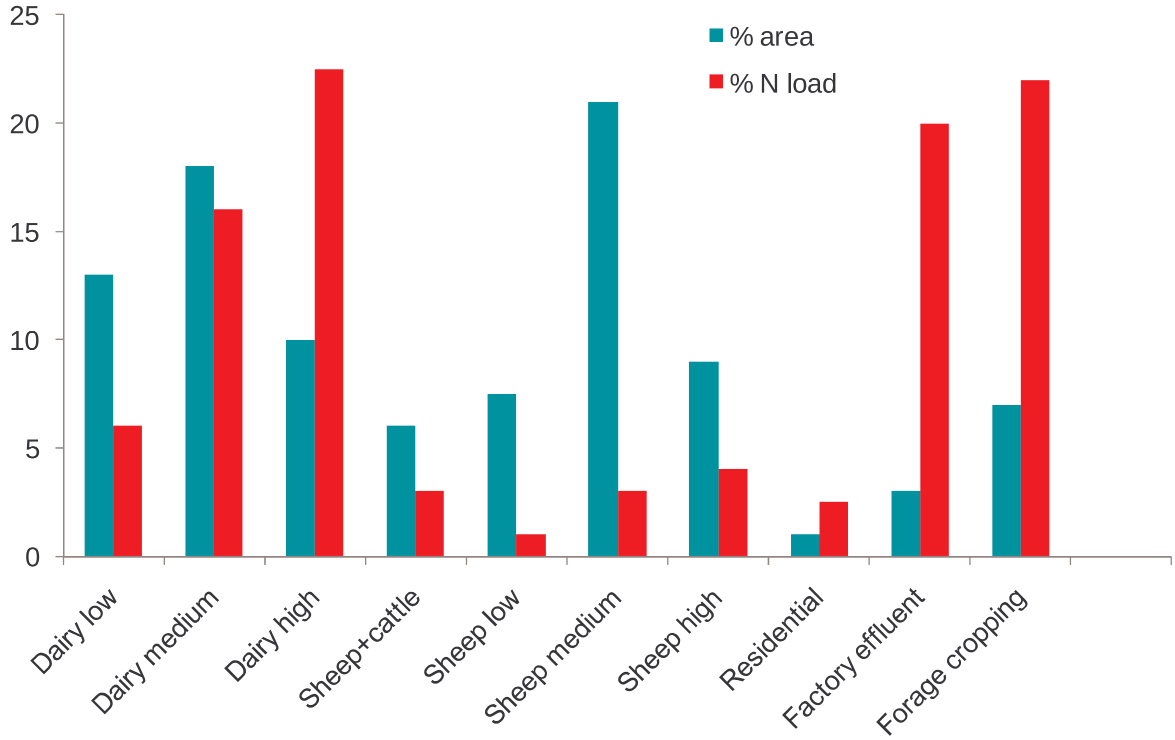


Figure 4.2. Modelled estimates of N losses from a range of sources within the Oteramika catchment study, Southland (Thorrold et al. 1998). Taller red bars compared to green indicate sources that make a proportionately greater contribution to catchment N loads than suggested by their areal extent i.e., they have a high N loss per unit land area.

The wide range in modelled P losses shown in Figure 4.1 reflects the increased complexity and multiple sources of P losses from farms. For example, a key source of P loss from deer farms is assumed to be due to erosion caused by deer wallowing and pacing. These losses are exacerbated by increased slope, direct deposition of faeces in wallows, decreased soil structural resilience and poor drainage. The relatively high losses modelled for dairy farms are due to contributions from rain-fed mole-pipe drainage, overland flow and losses of FDE applied to poorly- and/or artificially-drained soils when soil conditions are too wet. In contrast, the relatively low losses estimated for sheep land use reflects the lower soil Olsen P concentrations and stocking rates typically found on these farms.

The modelled N and P losses for Forestry land use in Figure 4.1 are estimates representing the average for a plantation life cycle (i.e. spread over growth and harvest phases). These losses were derived from the literature (Wilcock, 1986; Cooper and Thomsen, 1988; Quinn and Ritter, 2003) and are generally low relative to the modelled losses for other farm types.

4.1 Key sources of pollutants: dairy farms

The pie charts shown in Figure 4.3 illustrate the relative importance of the different sources of N, P and *E. coli* discharged from a model dairy farm. This method of presenting data can help to identify where mitigation efforts can be most effectively targeted. The information used to construct the pie charts in Figure 4.3 comes from an inventory of measured and modelled data. For overland flow and mole-pipe drainage sources, summary measured data were taken directly from the Tussock Creek dairy grazing trial. Due to similar soil types, this data agrees well with the model estimates derived using Overseer for the model Bog Burn dairy farm presented in Figure 4.1. In the case of pollutant losses due to the preferential flow of FDE through mole-pipe drainage systems, the EFFDRAIN model (Monaghan & Smith 2004) was used to calculate yields based on measured climate data and soil water deficits recorded at the Tussock Creek site and assuming that one month's effluent pond storage was used on the model dairy farm. Direct deposition of pollutants due to cows grazing un-fenced stream reaches was modelled using the algorithms contained in the BMPToolbox (Monaghan, 2009; Bagshaw 2002), which in this case assumed that cows deposited 3% of their excreta in the stream. Given that approximately 85% of stream reaches on dairy farms in the Bog Burn catchment are fenced, direct excretal deposition is much higher than typically expected, but is used here to illustrate the large potential effect that animal access to streams can have on pollutant losses from farms. Note also that our modelling analysis here does not consider any additional effects that animal access to streams can be expected to have e.g. stream bank and bed erosion due to hoof damage leading to increased losses of P and sediment.

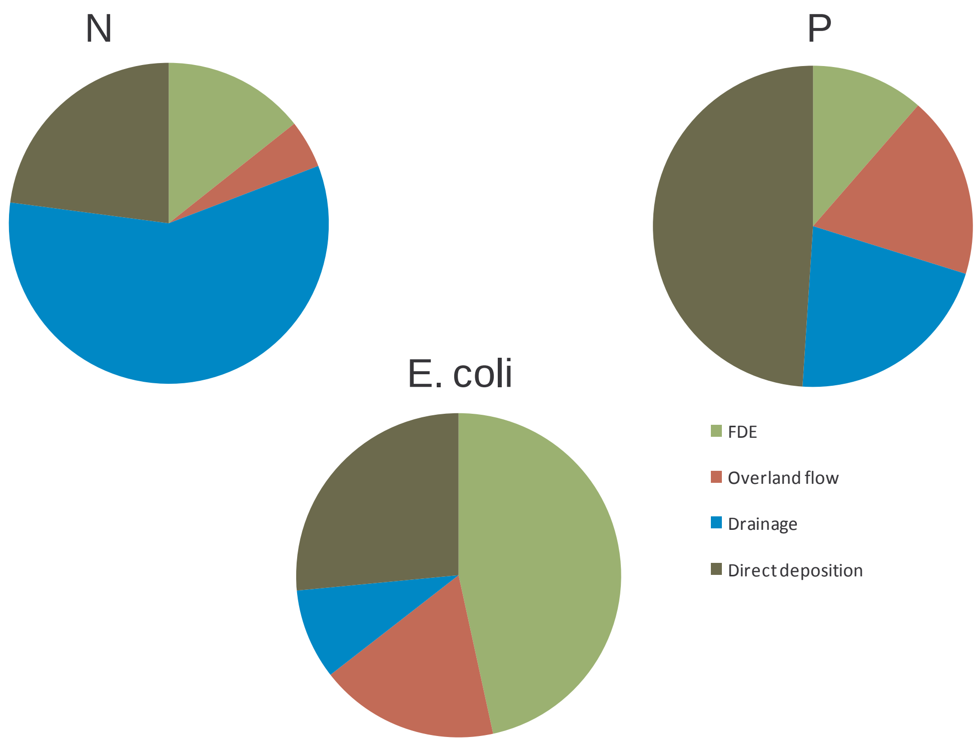


Figure 4.3 Estimated sources of N, P and *E. coli* discharges to water from a Bog Burn dairy farm: (i) direct deposition of faeces to un-fenced streams, (ii) drainage, (iii) overland flow, and (iv) incidental losses of contaminants due to the preferential flow of FDE through mole-pipe drains (one month pond storage assumed). Note that un-restricted access of cows to streams has been assumed to demonstrate the large potential effect this can have on whole-farm contaminant losses.

In the case of N, we can see that rainfall-induced subsurface drainage represents the major source (60%) of N losses to waterways from our model dairy farm. Effective strategies for mitigating this load therefore need to target the urine N deposited in the paddock that contributes to this drainage N load. This is why technologies such as nitrification inhibitors (Di & Cameron 2002; Monaghan et al. 2009a) and off-paddock grazing strategies have been developed and researched in recent times (de Klein et al. 2005). Direct drainage of FDE through the mole-pipe network (14%) and the direct deposition of excreta to streams (23%) make up most of the remainder of the farm N load to water, with overland flow representing only 5% of the total N loss.

The key source of P loss from our (un-fenced) model farm is the direct deposition of faecal P to streams due to cows having unrestricted access. This is estimated to contribute 50% of the whole-farm P loss to water. In contrast, mole-pipe drainage (21%), overland flow (18%) and direct drainage of FDE (11%) represent smaller sources of P loss from our model farm. For our inventory of *E. coli* losses, the situation is different again, with direct drainage of FDE accounting for 47% of the farm load. Direct deposition of faeces to streams (26%), overland flow (18%) and mole-pipe drainage (9%) represent much smaller sources of whole-farm *E. coli* losses to water.

Allowing cattle access to streams has historically been one means of providing pastured cattle with drinking water and comfort during hot weather, but is now recognized as a poor management practice from the standpoint of nutrient loss as well as cattle health. Practices such as fencing out 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 cattle. McDowell and Wilcock (2007) monitored a 2,100 ha catchment in New Zealand containing dairy farms with seasonal milking. They observed enriched 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 north-eastern 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.

In contrast to the findings of McDowell & Wilcock (2007), monitoring of the Bog Burn catchment in Southland indicated that trapped sediment was derived from topsoil entering the stream either via mole-pipe drainage or overland flow (McDowell & Wilcock, 2004). However, streams within this catchment are mostly fenced, thus eliminating some of the key sources identified by McDowell & Wilcock (2007).

The benefits to water quality from preventing livestock access stock to streams are much broader than just decreased nutrient and faecal bacteria inputs that result when excretal deposits are avoided. Quinn et al. (2009) document some of these wider benefits of riparian reforestation and stock exclusion within the context of New Zealand hill country. These benefits include decreases in water temperatures and streambed cover by fines and macrophytes, and improvements in stream macroinvertebrate populations.

Belsky et al. (1999) also document some of the wider influences (generally deleterious) that livestock have on riparian ecosystems in the western United States.

The above analysis demonstrates how the key sources of stream pollutants vary depending on the contaminant of interest. This is somewhat unfortunate because it implies that no single mitigation action on a dairy farm can effectively decrease the losses of all contaminants at the same time. Hence, mitigations to decrease N losses need to target animal urine patches (nitrification inhibitors, off-paddock wintering), whereas mitigations to decrease P and *E. coli* losses need to target stream fencing and the improved management of FDE. However, it is important to acknowledge that each mitigation will have multiple benefits e.g. stream fencing to decrease P losses will also decrease *E. coli* and N losses, by 26% and 13% in the case of our model farm, respectively.

4.2 Key sources of pollutants: sheep farms

A similar inventory of key sources can be also constructed for a model sheep farm, again using the Bog Burn catchment and soils as a case study setting. Although we have relatively little experimental data to back up our inventory assumptions (particularly for P and *E. coli*), we can use the *Overseer* model and the BMPToolbox to make an assessment of the effects of allowing sheep grazing access to streams. Some of the assumptions we have made here to perform this assessment are (i) 1 out of every 150 dung pats deposited by sheep lands directly in the stream, (ii) background N and P losses in overland flow and subsurface drainage combined are 8 and 0.4 kg ha/yr, respectively (Fig. 4.1) and (iii) background *E. coli* losses in overland flow and subsurface drainage combined are 1.4×10^{11} cfu ha^{-1} $year^{-1}$. The latter assumes that *E. coli* losses from a mole-pipe drained Pallic soil in Southland under dry stock farming are lower than measured for dairy pastures, and decreased in proportion to the lower sheep stocking rate assumed (14 SU ha^{-1}) relative to dairy land use (20 SU ha^{-1}).

The pie charts shown in Figure 4.4 illustrate the relative importance of the different sources of N, P and *E. coli* discharged from the model sheep farm. As noted for dairy, subsurface drainage represents the major source (80%) of nitrate-N lost from our model farm; direct inputs of excretal N to the stream (11%) and overland flow (8%) contribute the remainder. In the case of direct excretal deposition to the stream, we have assumed that 1 out of 150 dung pats is deposited into un-fenced streams. This is based on direct deposition rates for cattle (Bagshaw 2002), but attenuated to reflect the decreased tendency for sheep to spend time in riparian areas. In the case of P losses, mole-pipe drainage, direct deposition and overland flow make approximately equal contributions to the farm discharge. For our inventory of *E. coli* losses, direct deposition of faeces to streams contributes only 10% of the annual load, with overland flow representing the major pathway (60%) of *E. coli* losses to water from our model farm.

It is important to note that, while the pie charts in Figures 4.3 and 4.4 clearly illustrate the key sources/pathways of contaminant loss on an annual basis, they do not account for important seasonal effects. For example, although direct deposition represents only 10% of annual farm *E. coli* losses from our model sheep farm, it probably represents a much more important source of loss during critical summer months when (i) streams are of greatest recreational value and (ii) the other sources and pathways (overland flow and subsurface drainage) are unlikely to be active. FRST-funded research programmes are underway to attempt to better describe the importance of these seasonal effects.

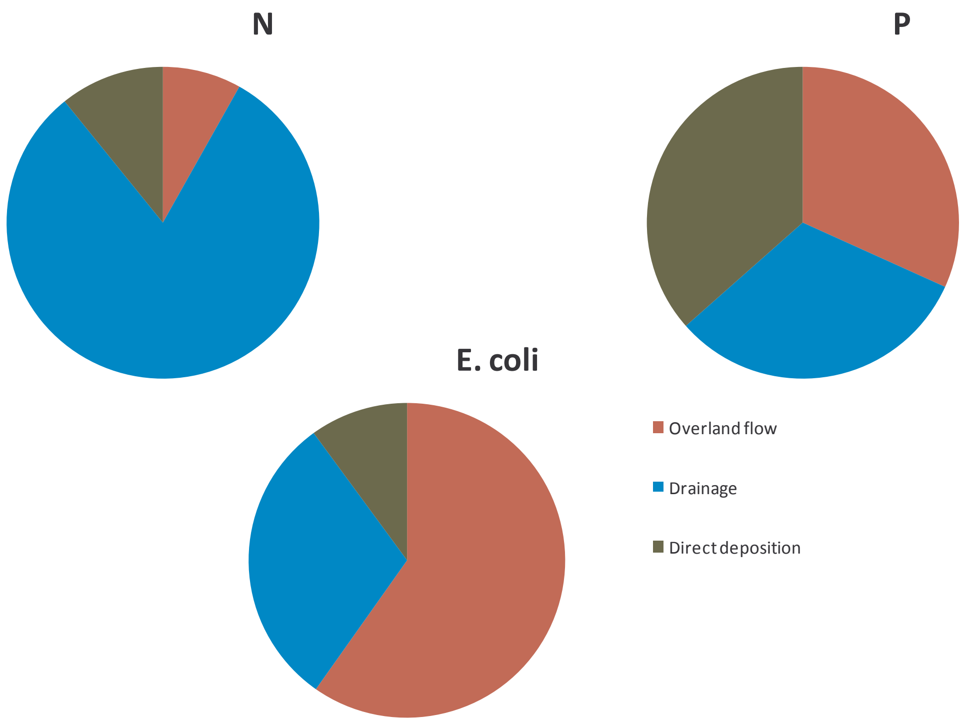


Figure 4.4 Estimated sources of N, P and *E. coli* discharges to water from a Southland sheep farm: (i) direct deposition of faeces to un-fenced streams (deposition

of 0.7% of excreta assumed), (ii) drainage, and (iii) overland flow. Note that un-restricted access of sheep to streams has been assumed to demonstrate the potential effect this can have on whole-farm contaminant losses.

4.3 A simplified assessment of potential relative effects of contrasting land uses and management practices

Based upon the literature reviewed in section 3 of this document, and the comparative assessments shown above, we have attempted to provide a simple index in Table 4.1 showing the effects that different land uses and land use practices have on contaminant losses from farms. It is important to recognise that landscape and management factors strongly influence absolute losses. However, this simplified assessment does illustrate the potential effects of land use and practice for equivalent landscape and management factors. It is evident from Table 4.1 and Figure 4.1 that changes in land use can have a profound effect on contaminant losses to water. Some obvious examples include:

- Switching from low intensity land uses, such as sheep or forestry, to dairy farming will result in greater losses of N to water.
- These shifts are also likely to result in greater losses of P, sediment and faecal micro-organisms if they occur on heavy soils and/or management practices on the converted farms are poor.
- Changing land use to deer farming is likely to increase losses of P and sediment unless riparian areas and seeps/bogs are well protected.
- Changing from sheep farming to mixed cropping is also likely to lead to increased losses of some contaminants, although this will be heavily dependent on how the cropping system is managed. Factors that exacerbate N leaching losses on the mixed cropping farm include having a large proportion of the farm in crop, pre-winter tillage, applying high rates of N fertiliser to crops and practising conventional cultivation instead of direct drilling. Losses of P and sediment from the mixed cropping farms will also depend heavily on management factors such as cultivation of slopes, sowing up and down slopes and leaving soil exposed during periods of surplus rainfall. Conversely, the use of GEPs such as contour ploughing, cover crops, and expert fertiliser decision support tools may help to decrease losses.

Table 4.1. A simplified index of the potential relative effects of different land uses and management practices on farm contaminant losses for equivalent landscape and management factors (flat topography assumed for all farm types). More crosses = greater losses.

Land use	Soil type	Management practice	N loss	P loss	*E. coli* loss	Sediment loss
Dairy	free-draining	< 2 weeks FDE storage; no stock exclusion; forage crop wintering	X X X X X X	X X	X X X	X X X
		- with stock exclusion	X X X X X	X	X X	X X
		- stock exclusion & FDE storage	X X X X	X	X	X
		- & off-paddock wintering	X X X	X	X	X
Dairy	poor-draining	< 2 weeks FDE storage; no stock exclusion; forage crop wintering	X X X X X	X X X X X	X X X X X	X X X X X
		- with stock exclusion	X X X X	X X X X	X X X X	X X X
		- stock exclusion & FDE storage	X X X	X X X	X X	X X
		- & off-paddock wintering	X X	X X	X X	X
Sheep	free-draining	15 SU ha^{-1}; streams unfenced	X X	X X	X X	X
	free-draining	15 SU ha^{-1}; streams fenced	X X	X	X	X
	poor-draining	15 SU ha^{-1}; streams unfenced	X X	X X X	X X X	X X

		15 SU ha^{-1}; streams fenced	X X	X X	X X	X X
Cropping	shallow soils	Sub-optimal[a]	X X X X X X	X	X	X X
		Optimal (fert., cultivation timing)	X X X	X	X	X
	deeper soils	Optimal (fert., cultivation timing)	X X	X	X	X
Deer	all	15 SU ha^{-1}; streams un-fenced	X X X	X X X X X	X X X X X	X X X X X X
	free-draining	15 SU ha^{-1}; streams fenced	X X	X	X	X X
	poor-draining	15 SU ha^{-1}; streams fenced	X X	X X X	X X X	X X X
Horticulture		Winter vegetables	**Potentially very high [b]**	**Potentially very high [b]**	**n/a**	**Potentially very high [b]**
Forestry			X	X	X	**?[b]**

[a]with respect to (i) fertilization amounts and timing, and (ii) tillage method and timing.

[b]very dependent on management practices (e.g. forestry harvesting). For horticultural crops, fertilisation timing and rates, residue management and cultivation practices will heavily modify losses.

5. Recommended Good Environmental Practices to decrease pollutant losses

Based on our review and inventory analyses above, we can recommend some key GEPs that decrease contaminant losses from farms (Table 5.1). Also provided are some rankings according to the relative cost-effectiveness of each GEP. This metric provides an assessment of where we are likely to get the "biggest bang for buck". In other words, a high cost-effectiveness implies that relatively large decreases in contaminant losses can be achieved per $ of mitigation expenditure. Estimates of the annualised net cost of implementing each mitigation measure was used to calculate a range of cost-effectiveness values based upon information within the BMPToolbox (Monaghan, 2009). For simplicity, this assumes a number of default costs, such as the opportunity cost of capital (8%), depreciation, maintenance, additional labour and feed, and revenue foregone as a result of land lost to production. Any financial benefits expected from implementing measures are deducted from the net overall annualised cost. These benefits can be particularly important where a measure increases productivity (e.g. extra pasture growth from the use of nitrification inhibitors) or decreases farm operational costs such as avoiding off-farm cow wintering fees if the animals are wintered under a Herd Shelter on the home farm. Strictly speaking, any assessment of mitigation costs and effectiveness should be conducted on a farm-specific basis due to the variable nature of farm management systems and landscape features. However, on a regional basis, such an exercise would be impossible to do for every permutation. Hence, the information in Table 5.1 provides an indicative assessment of the relative cost-effectiveness of a range of GEPs relevant to "typical" Southland farms.

The recommended GEPs shown in Table 5.1 are limited to those that have been shown to be effective in the field and cheap to implement. Some are effective in decreasing multiple contaminants (e.g. the improved effluent management practices) while others target a specific contaminant (e.g. nitrification inhibitors). Nutrient budgeting and stock exclusion are GEPs common to all pastoral land uses that are known to be highly cost-effective. Improved effluent management practices are another set of measures that are highly cost-effective mitigation measures for dairy farms. The next tier of GEPs that can be described as being of "medium" cost effectiveness include "facilitated" wetlands and nitrification inhibitors (all pastoral land uses) and off-paddock dairy grazing systems such as wintering shelters and/or restricted autumn grazing practices. The bottom tier of mitigation measures that are relatively cost-ineffective within a Southland context include grass buffer strips, incorporation of low N feeds into the diet and the use of constructed wetlands. It should be noted that our assessment of the cost-effectiveness of constructed wetlands does not fully capture the ancillary benefits of these structures such as habitat, biodiversity and aesthetic values and sediment capture.

Table 5.1. Suggested GEPs that can decrease contaminant losses from farms. A brief description of each GEP is provided in Appendix I.

Land use	GEP	Cost-effectiveness
Dairy	Nutrient budgeting	High
	Stock exclusion from streams	High
	Effluent storage[a]	High
	Low rate effluent application[a]	High
	Nitrification inhibitors	Medium
	Off-paddock wintering[b]	Medium
	Restricted autumn grazing	Medium
	Facilitated wetlands[c]	Medium
	Elimination of stock stream crossings	Varies according to bridging costs
	Constructed wetlands	Medium-low
	Grass buffer strips	Low
	Incorporating low N feeds into diets	Low
Sheep/beef/deer	Nutrient budgeting	High
	Stock exclusion from streams	High
	Facilitated wetlands	Medium
	Nitrification inhibitors	Medium
	Elimination of stock stream crossings	Varies according to bridging costs
	Constructed wetlands	Medium-low
	Grass buffer strips	Low
Cropping	Use of fertiliser calculators	High
	Judicious timing of cultivation	High
	Decreased tillage intensity	High
	Cover crops	High

[a]refer to Houlbrooke & Monaghan (2009) for more detailed assessments of soil-topographical categories where these improved effluent management systems are required; [b]assuming cows are wintered in/on structures where full effluent containment is achieved; [c]wetlands targeted at naturally poorly drained and relatively un-productive parts of the landscape such as seeps and bogs (McKergow et al. 2008).

6. Effects of GEP uptake at a catchment/regional scale

The effects of GEP uptake on water quality (TN, TP and *E. coli*) for the Oreti River catchment (3512 km^2) was simulated by NIWA using the Catchment Land Use for Environmental Sustainability model version 3 (CLUES 3.0). CLUES was run with default settings (no mitigation) and six mitigation scenarios to simulate: stock exclusion (current and future levels); nitrification inhibitors; herd shelters; improved farm dairy effluent (FDE) management; and constructed wetlands.

6.1 CLUES 3.0 overview

CLUES is a modelling system for assessing the effects of land use change on water quality and socio-economic factors at a minimum scale of sub-catchments (~10 km^2 and above). CLUES was developed by NIWA in collaboration with Lincoln Ventures, Harris Consulting, AgResearch, HortResearch, Crop and Food Research, and Landcare Research for the Ministry of Agriculture and Forestry (MAF) and the Ministry for the Environment (MfE). CLUES couples a number of existing models within a GIS-platform and is provided to users as a front-end interface within ArcGIS (Figure 6.1). Water quality results provided by CLUES are:

- Nutrient loads (kg year^{-1}) - in-stream cumulative loads for total nitrogen (TN) and total phosphorus (TP) for each river reach.
- Sediment load (kilo-tonnes/year) - in-stream cumulative load of total suspended solids (TSS) for each river reach
- *E. coli* loads (10^{15} or one “peta” of organisms/year) – in-stream cumulative organism count for each river reach
- Nutrient concentration (mg/m^3) - in-stream N and P median concentrations for each river reach.
- Nutrient yields (kg/ha/year) - nutrient load divided by the contributing area. Provided in two forms:
  Cumulative yield - the in-stream cumulative yield which represents the total yield for each reach and its up-stream tributaries.
  Generated yield - the yield generated by each sub-catchment which is delivered to the stream network.
- Generated Sediment yield (tonnes/ha/year) - yield of TSS generated by each sub-catchment. This information can be used to identify sources of sediment.

- Total nitrogen loss risk (scale from very low to very high) - the leaching risk for nitrogen based on land use from EnSus.

The CLUES interface has tools which allow users to develop land use change scenarios. The use of these tools has been demonstrated by Semadeni-Davies et al. (2009a) for the Waikato River catchment. The current study uses a pre-release version of CLUES (CLUES 3.0) which also allows users to vary stocking rates and apply mitigation factors to simulate the impacts of various farming practices on water quality.

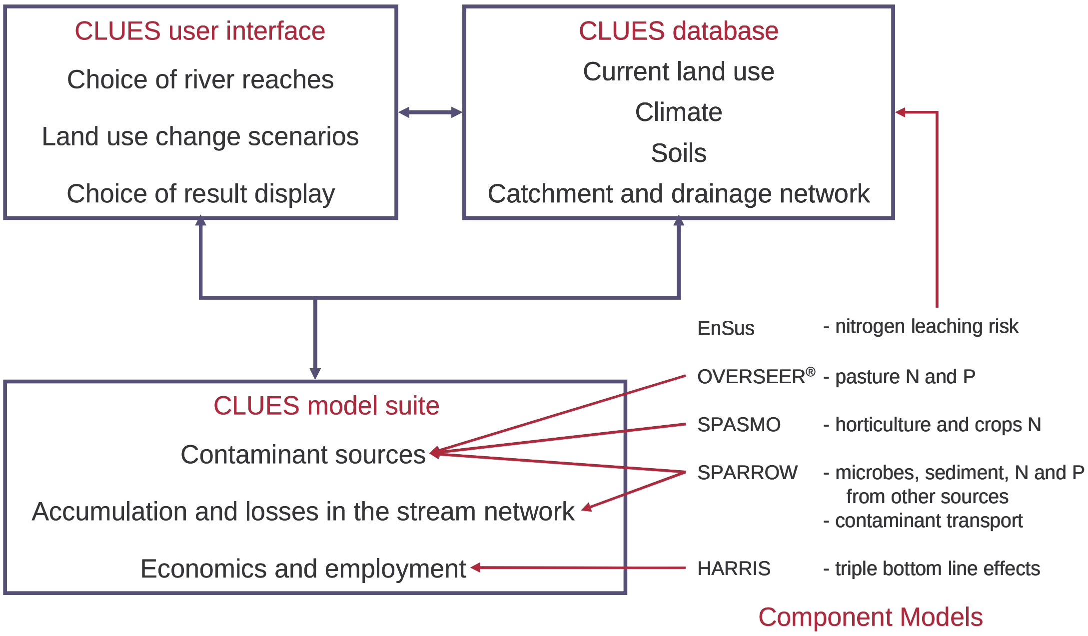


Figure 6.1 CLUES modelling framework (from Semadeni-Davies et al., 2009b).

CLUES integrates the following models into one tool within a GIS platform:

- **SPARROW** (Spatially Referenced Regression on Watershed attributes) - predicts annual average stream loads of TN, TP, sediment and *E. coli*. It includes extensive provisions for stream routing and loss processes. This modelling procedure was originally developed by the USGS (Smith et al. 1997) and has since been applied and modified in the New Zealand context, in extensive liaison with the developers. **SPARROW** has been applied to N and P in the Waikato (Alexander et al. 2002) and subsequently to the whole New Zealand landscape (Elliott et al. 2005). A national model for *E. coli* is now available (Alexander & McBride in prep., obtained by fitting models to a national microbial water quality dataset reported by Till et al. 2008). The **SPARROW** sediment transport routines were assessed by Elliott et al. (2008) and simulations compared favourably with measured sediment load data.

- **SPASMO** (Soil Plant Atmosphere System Model, HortResearch) - calculates the N budget for five horticultural enterprise scenarios. Detailed simulations for many cases (combinations of crops, climate, fertiliser use) have been run (using a daily time step) to build look-up tables that CLUES queries. It has been validated against data from grazed pasture (Rosen et al. 2004) and pasture treated with herbicide (Close et al. 2003; Sarmah et al. 2004).

- **OVERSEER**® (AgResearch, Wheeler et al. 2006) - computes nutrient leaching for dairy, sheep and beef and deer. It provides annual average estimates of N losses from these land uses, given information on rainfall, soil order, topography and fertiliser applications. Within CLUES, **OVERSEER** losses vary as a function of soil order, rainfall, stocking rate, land use class and region. For other variables, such as fertiliser application rates, typical values are used based on the region and land use.

- **TBL** (Triple Bottom Line, Harris Consulting) - estimates economic output from different land use types (pasture, horticulture, forestry and cropping), in terms of Cash Farm Surplus (CFS), Total GDP and Total Employment from that land use, given as a function of output. The calculations are based on the MAF farm monitoring models.

- **EnSus** (Environmental Sustainability, Landcare Research) - provides maps of nitrogen leaching risk, used as an adjunct to interpretation of CLUES results. It is based on studies of nitrogen losses at national and regional scales (Hewitt and Stephens, 2002; Parfitt et al. 2006).

CLUES does not contain a groundwater model. Rather, it is assumed that water percolating into the ground will emerge in the same surface water catchment.

The base areal unit of CLUES is the sub-catchment which comes from the NIWA River Environment Classification (REC) of the national stream and sub-catchment network[1]. Each sub-catchment is associated with a river reach and has a unique identity number — there are 6696 reaches in the Oreti catchment. Predictions of the water quality and financial indicators given above can be made for any reach.

[1] http://www.niwa.co.nz/ncwr/rec

Geo-spatial data needed to run CLUES are provided at national, regional, catchment and sub-catchment levels. Terrain data is at 30m resolution. In addition to REC, data sets provided are current land use, runoff (derived from rainfall less evapotranspiration), slope, soil data (from the Land Resources Inventory, LRI, Fundamental Soils Layer[2] – Wilde et al., 2004) and point sources and lakes. The current land use layer provided with CLUES was developed with extensive reference to the LCDB2 (Land Cover Database)[3], AgriBase (AsureQuality Ltd)[4], and LENZ (Land Environments of New Zealand)[5] land use geo-databases. Considerable effort was expended, with Landcare Research, to ensure that the spatial data coverage was as accurate as possible. Pastoral land use is based on 2001 conditions, and will not reflect recent dairy conversions. Further details on the modelling framework can be found in Woods et al. (2006).

New to CLUES 3.0 is the ability to create farm practice scenarios which enhance or mitigate contaminant yields at the sub-catchment scale. These can be applied to river reaches using interactive selection tools or by supplying CLUES with a scenario table for those catchments affected. Percentage changes in stocking rates, nutrient losses to water and *E. coli* release from dairy, sheep and beef and deer farms can be used to simulate farm practices. These tools are at the heart of the present report.

6.2 Comparisons between CLUES predictions and water quality observations in the Oreti Catchment

CLUES was run for the entire Oreti catchment and compared with available nutrient measurements in order to assess model performance. The comparison was made at three locations (Figure 6.2; Table 6.1) in order to assess model performance in the Oreti catchment. Two of the sites (Riverton Highway Bridge and Lumsden) are in the National Water Quality Network (Smith and Maasdam, 1994; Ballantine and Davies-Colley, 2009), and the measurements are based on 20 years of data. The third site (Bog Burn) was monitored as part of the Best Practice Catchments for Sustainable Dairying programme (Wilcock et al., 2007; Parshotam and Elliott, 2009). For the comparison, CLUES was run using default settings (i.e., no mitigation) and for the current level of stock exclusion (see Section 6.3.1). The stock exclusion analysis is included in the

[2] http://soils.landcareresearch.co.nz/contents/index.aspx

[3] http://www.mfe.govt.nz/issues/land/land-cover-dbase/classes.html

[4] http://www.asurequality.com/corporate/it_services/agribase.cfm

[5] http://www.landcareresearch.co.nz/services/informatics/LENZ/about.asp

comparison as approximately 75% of dairy cattle and 35% of dry stock (sheep and beef) are currently excluded in areas with a land use capability (LUC) rating of 1 to 3 (Environment Southland: Morgan, G., pers. comm., 2010).

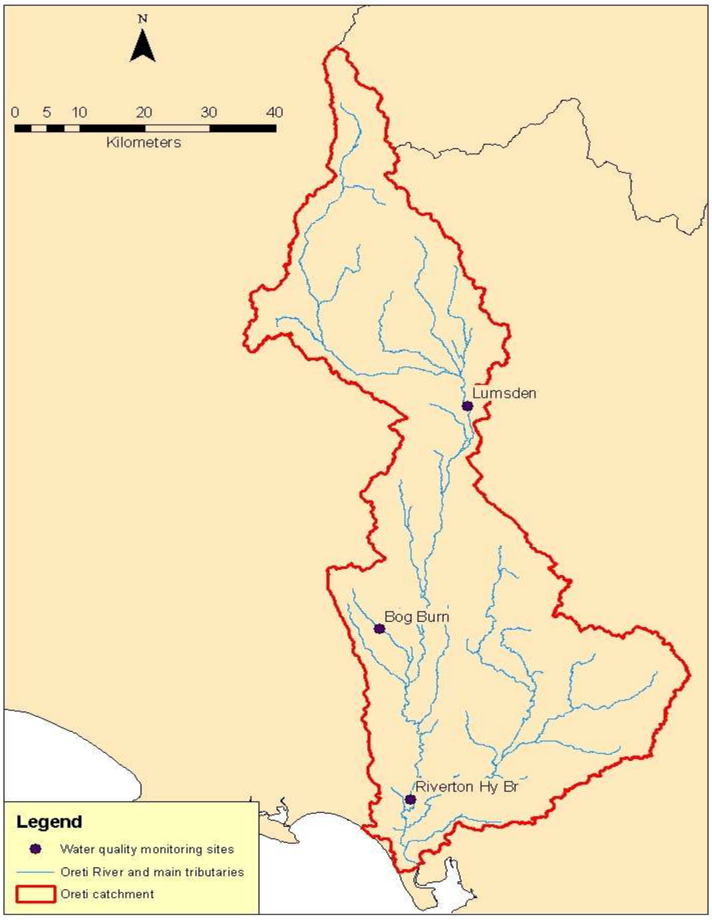


Figure 6.2 Location of water quality observation points in the Oreti River catchment.

The flow rates within CLUES match the measured flows well (Table 6.1) suggesting that there is no large-scale loss or gain of water into or out of the catchment. However, it is known that some stream reaches in the upper catchment lose water to the local unconfined groundwater systems. Riparian aquifers may provide seasonal storage throughout the catchment, and in the lower reaches, streams may be fed from confined aquifers which are recharged beyond the local surface water catchment (personal communication, Karen Wilson, Environment Southland groundwater scientist). Hence, CLUES is used to provide a 'big picture' of surface water quality, rather than a comprehensive and detailed view of the complex groundwater-surface water hydrological system.

The results for TN are given in Table 6.2 and for TP in Table 6.3.

The TN loads, concentrations and yields match those measured well, except for the Riverton HB site. This discrepancy is unlikely to reflect recent dairy intensification, because the measurements are based on a 20-year average. Also, considering that the model performs satisfactorily for the dairy-dominated Bog Burn site, the differences at Riverton HB are unlikely to be related to dairy land use. The under-prediction may reflect delayed responses to historical land-use changes. Some of the confined aquifers in the regional have groundwater ages in the order of 100 years (personal communication, Karen Wilson, Environment Southland). However the surficial unconfined aquifers have much shorter residence times, in the order of 10 years, and would not show such a slow response to historical land development or intensification. The under-prediction could also relate to underestimation of the intensity of sheep and beef land-uses in the lower catchment.

For TP, the predictions for load and yield are satisfactory for Bog Burn and Riverton HB, but the model over-predicts the load and yield at Lumsden, which has relatively undeveloped land-uses.

The model over-predicts TP concentrations at Lumsden and Riverton HB by a considerable degree. This is partly a reflection of load over-estimation (at Lumsden), but also reflects the way that median concentrations are calculated from the flow-weighted concentration, which in turn is calculated from the load and flow rate. For Riverton HB and Lumsden, the model predicts that the median concentration is approximately half of the flow-weighted concentration, whereas the measurements show that the median concentration is approximately a quarter of the flow-weighted concentration. This reflects a greater degree of flashiness or seasonal variability in concentrations, which may be due to in-stream uptake during baseflow and re-mobilisation during storm flow in these low-P and P-limited streams.

While the model reflects overall trends in loads and concentrations, the absolute values are sometimes inaccurate, especially for TP. Hence, it is best to focus on the relative changes predicted under various mitigation scenarios, rather than the absolute values.

Table 6.1. Water quality monitoring site information with observed and CLUES simulated mean discharge.

		Lumsden	Riverton Highway Bridge	Bog Burn
NZ River Reach		15033324	15058642	15046046
Up-stream catchment area (km^2)		1129	2143	21
Source		National Water Quality Network	National Water Quality Network	Wilcock et al., 2007
Mean discharge (m^3/s)	CLUES	28.6	42.5	0.3
	Measured	27.1	38.0	0.3

Table 6.2. Observed and CLUES-simulated total nitrogen (TN) loads and concentrations (default settings and current stock exclusion). Concentrations are medians.

	Lumsden	Riverton Highway Bridge	Bog Burn
TN Load ($t\ y^{-1}$)			
Predicted (default)	345	1036	22
Predicted (current stock exclusion)	342	980	19
Measured	470	1602	21
TN concentration ($mg\ m^{-3}$)			
Predicted (default)	333	624	1381
Predicted (current stock exclusion)	330	590	1216
Measured	480	885	1100
TN Yield ($kg\ ha^{-1}\ y^{-1}$)			
Predicted (default)	3.1	4.8	10.5
Predicted (current stock exclusion)	3.0	4.6	9.3
Measured	4.2	7.5	10.1

Table 6.3. Observed and CLUES-simulated total phosphorus (TP) loads and concentrations (default settings and current stock exclusion). Concentrations are medians.

	Lumsden	**Riverton Highway Bridge**	**Bog Burn**
TP Load (t y^{-1})			
Predicted (default)	42	80	0.7
Predicted (current stock exclusion)	42	75	0.5
Measured	21	63	1.0
TP concentration (mg m^{-3})			
Predicted (default)	18.6	30.8	52.6
Predicted (current stock exclusion)	18.4	28.7	40.8
Measured	5.3	13.9	50.0
TP Yield (kg ha^{-1} y^{-1})			
Predicted (default)	0.4	0.4	0.3
Predicted (current stock exclusion)	0.4	0.3	0.3
Measured	0.2	0.3	0.5

6.3 Mitigation scenarios

The effect of implementing mitigations in a particular sub-catchment or selection of sub-catchments was simulated using CLUES 3.0 by specifying the percentage decrease in nutrient loss and *E. coli* generated by dairy or sheep and beef farming for each of the sub-catchments targeted. The mitigation factors for the scenarios and their decrease factors are summarised in Table 6.4 and were provided by AgResearch. For each sub-catchment investigated, the mitigation factors were weighted according to the proportional area satisfying the mitigation criteria given in the table. This was done by splitting each sub-catchment into mitigation and no-mitigation zones using standard GIS tools and then calculating the percentage area of the total sub-catchment area for each zone in MS Excel using pivot tables. The mitigation factors were then attenuated accordingly. Current stock exclusion was assumed to coincide with simulation of nitrification inhibitors, herd shelters, wetlands and FDE management where these mitigations were applied to areas with an LUC of 1-3.

Table 6.4. Assumed mitigation factors in farm losses of N, P and *E. coli* under a range of mitigation scenarios (Oreti River catchment). Figures in italics/bold are best guesses.

Mitigation type	Mitigation Criteria			Mitigation (% decrease)		
	Soil drainage	**Land Use Capability**	**Landuse**	**N**	**P**	***E. coli***
CLUES default (no mitigation)	All	All	All	0%	0%	0%
Stock exclusion from streams – current situation[1]	Not specified	1-3	Dairy	15%	30%	15%
			Dry stock (all sheep and beef)	3.5%	10.5%	3.5%
Stock exclusion from streams – future mitigation[2]	Not specified	1-3	Dairy	20%	40%	20%
			Dry stock (all sheep and beef)	10%	30%	10%
Nitrification inhibitors[3]	Not specified	1-4	Dairy and dry-stock (sheep and beef intensive)	30%	0%	0%
Herd shelters[3]	Not specified	1-4	Dairy	30%	***10%***	***10%***
Wetlands[3]	Poorly drained	Not specified	Dairy and dry-stock (all sheep and beef)	25%	0%	***20%***
Improved FDE management[3]	Free draining	Not specified	Dairy	5%	5%	5%
	Poorly draining	Not specified	Dairy	10%	10%	40%

[1]Assumes current stock exclusion of 75% dairy cattle and 35% sheep and beef in LUC classes 1-3.
[2]Assumes total stock exclusion of all stock in LUC classes 1-3.
[3]Scenario simulated in combination with current stock exclusion.

The following GIS layers were used to select areas where mitigation should be applied:

- Land Use Capability (LUC) taken from the LRI (Newsome, 1995). LUC classes for the Oreti catchment are given in Figure 6.3. A full description of LUC classes and their application nationwide can be found in Lynn et al. (2009).

- Soil data for the lower Oreti catchment were supplied by Environment Southland (ES topo-climate soil data). Data for the upper catchment were taken from the LRI Fundamental Soil Layer (Wilde et al., 2004). Both datasets have five drainage classes. The catchment was split into poor (LRI drainage classes 1 and 2) and free-draining (drainage classes 3 to 5) areas (Figure 6.4).

- Pastoral land used for dairy and sheep and beef farming were identified using the CLUES default land use layer. This layer records land use for 2001 and therefore does not include subsequent dairy conversions. Note that *other land uses* includes some grazed tussock.

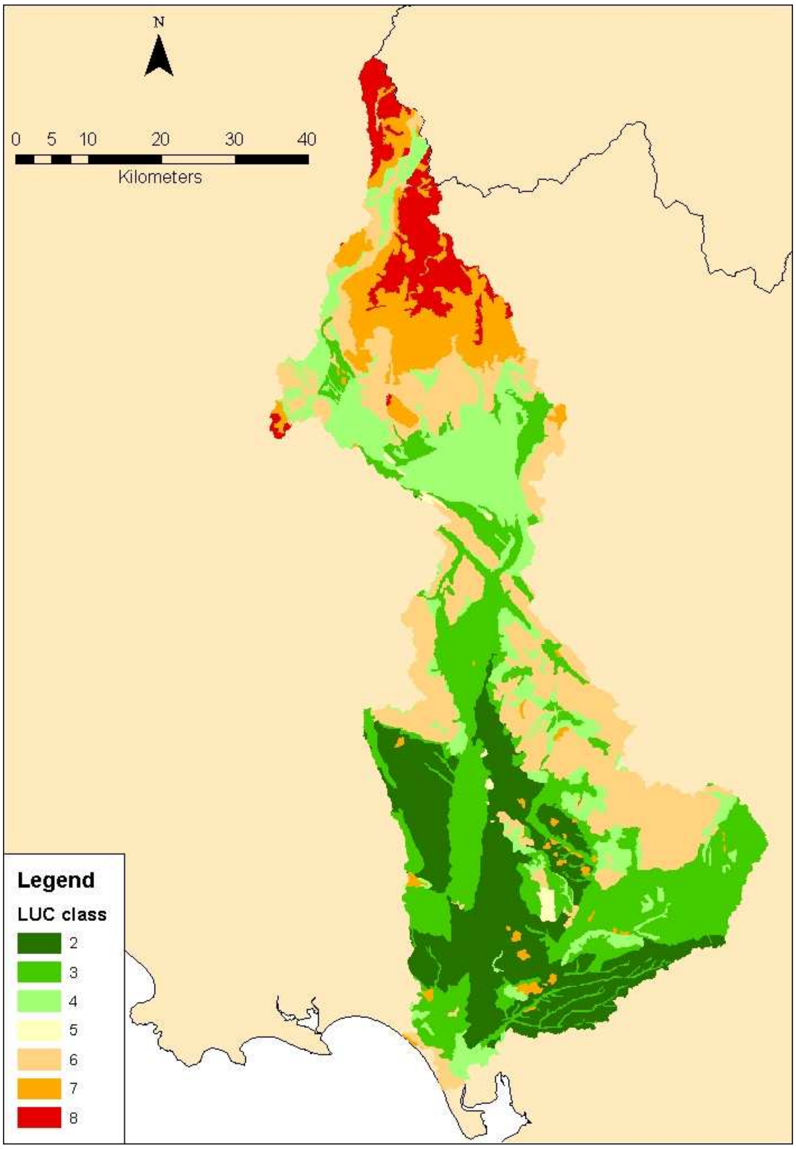


Figure 6.3. Land Use Capability classes for the Oreti catchment (source, LRI; Newsome, 1995). Pastoral LUC classes (1-4) used to develop the mitigation scenarios are mapped in green.

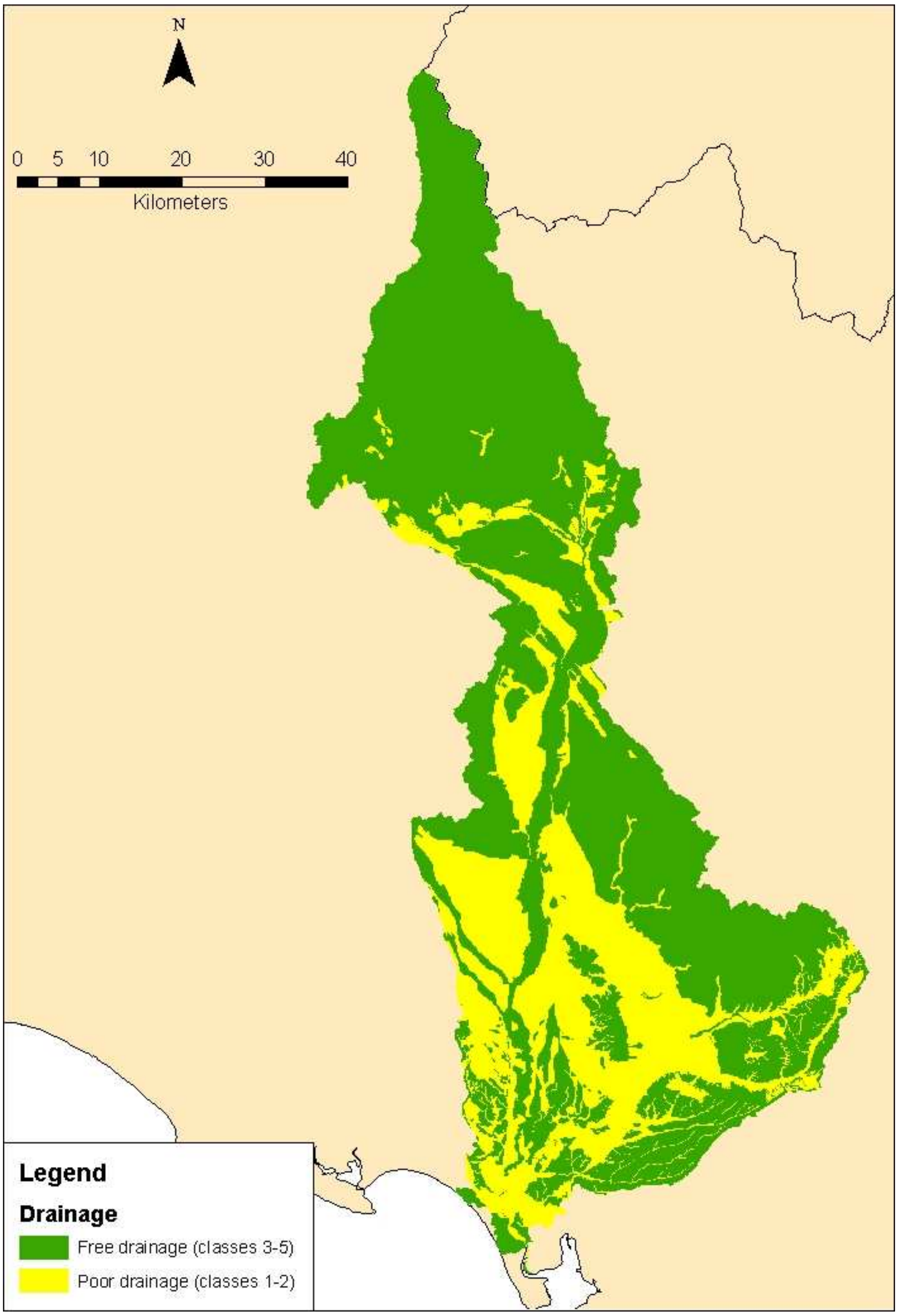


Figure 6.4 Areas of free- and poorly-draining soils in the Oreti catchment (derived from data supplied by Environment Southland and the LRI Fundamental Soil Layer (Wilde et al., 2004)).

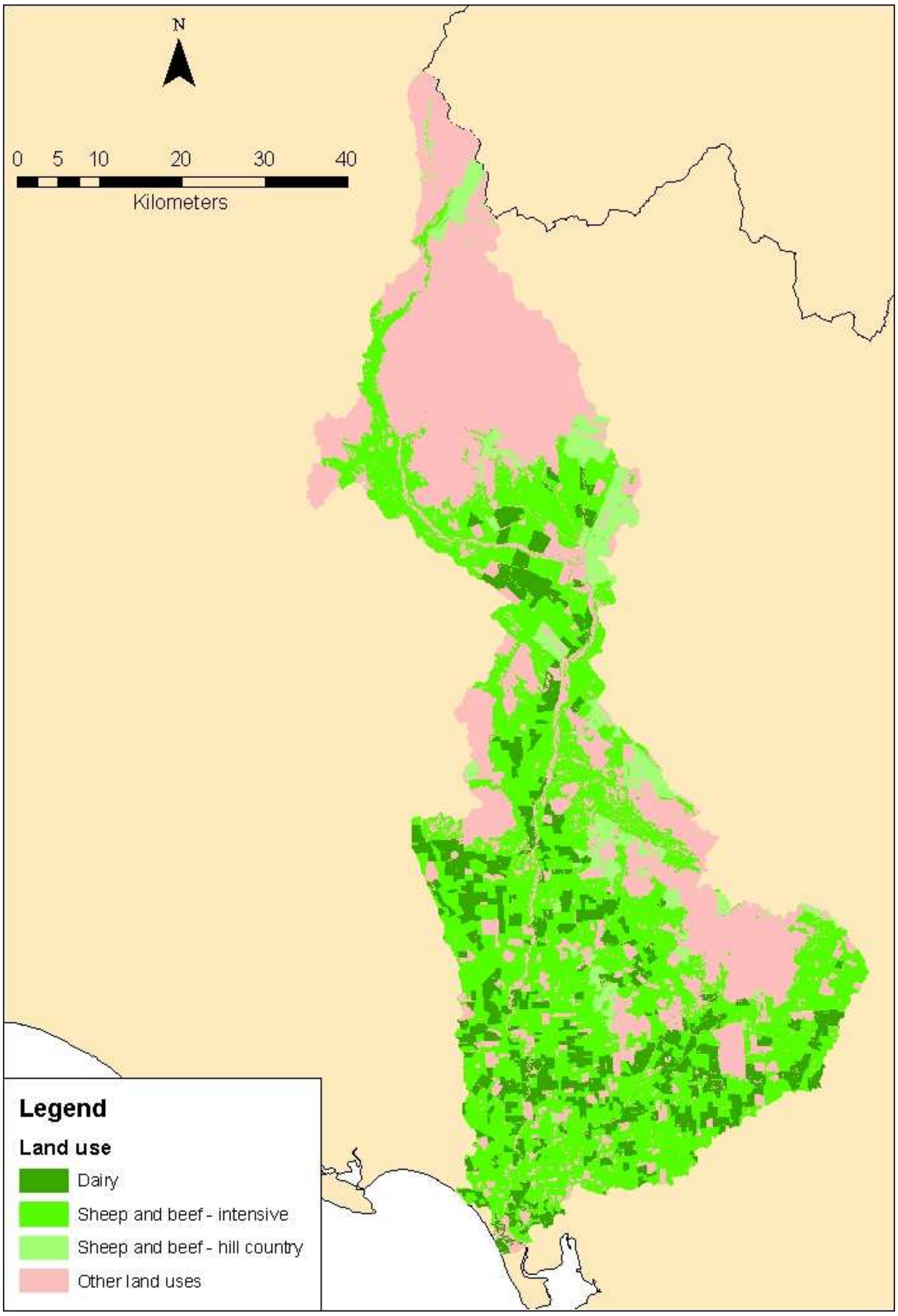


Figure 6.5 CLUES default dominant land use showing areas of dairy, and sheep and beef farming (derived from the LCDB2 and AgriBase).

6.3.1 Stock exclusion (current and future)

ArcMap selection tools were used to identify areas which satisfied the mitigation criteria for current and future stock exclusion given in Table 6.4. Approximately 40% of the total catchment area satisfies these criteria as can be seen in Figure 6.6.

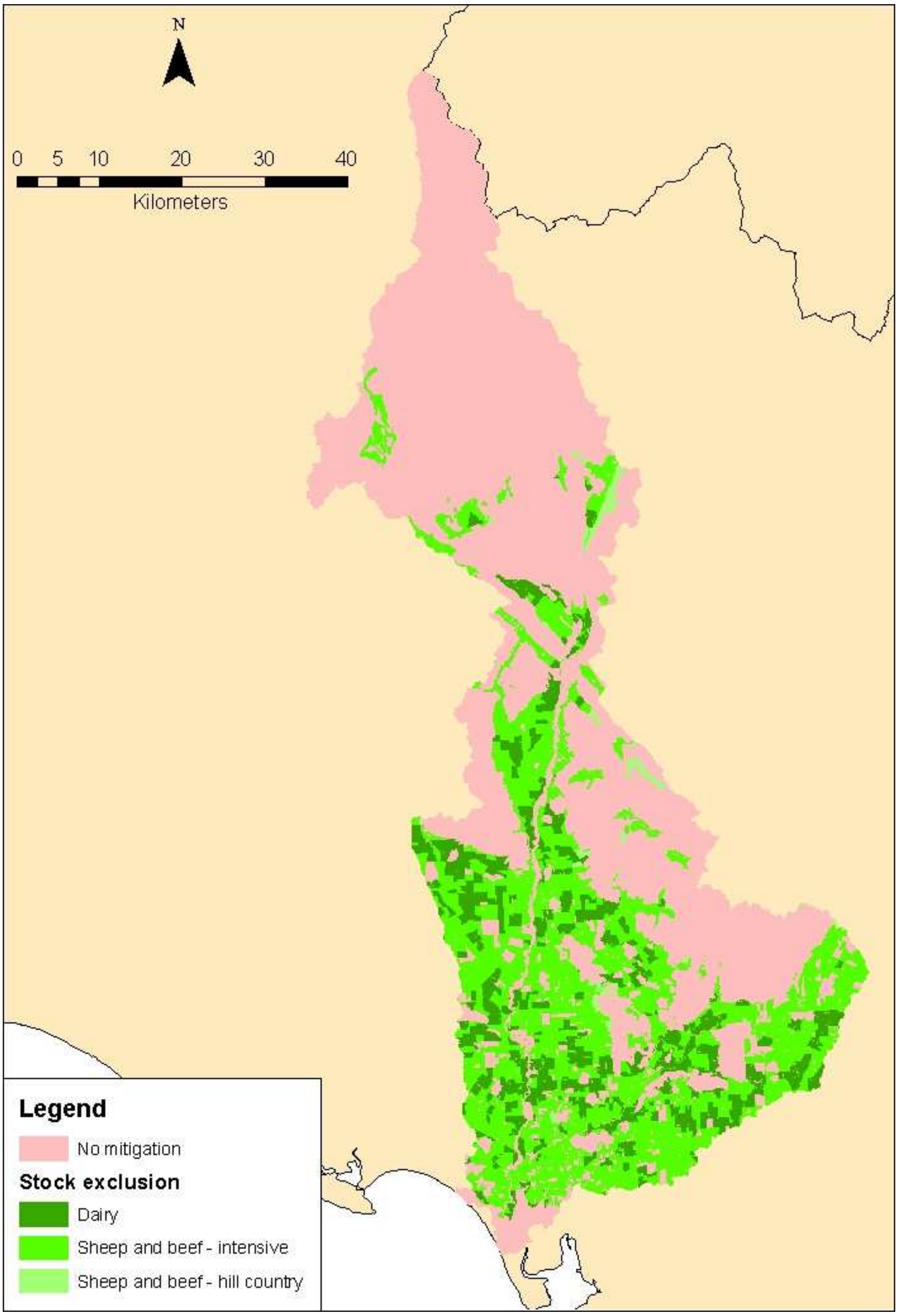


Figure 6.6 Areas which satisfy the criteria for stock exclusion in the Oreti catchment

6.3.2 Nitrification inhibitors

Nitrification inhibitors were applied to areas in LUC classes 1-4 that were either dairy or intensive sheep and beef farming. For areas with an LUC of 1-3, nitrification inhibitors were applied in addition to current stock exclusion. Approximately 50% of the total catchment area, marked in green on Figure 6.7, satisfies the criteria for nitrification inhibitors given in Table 6.4.

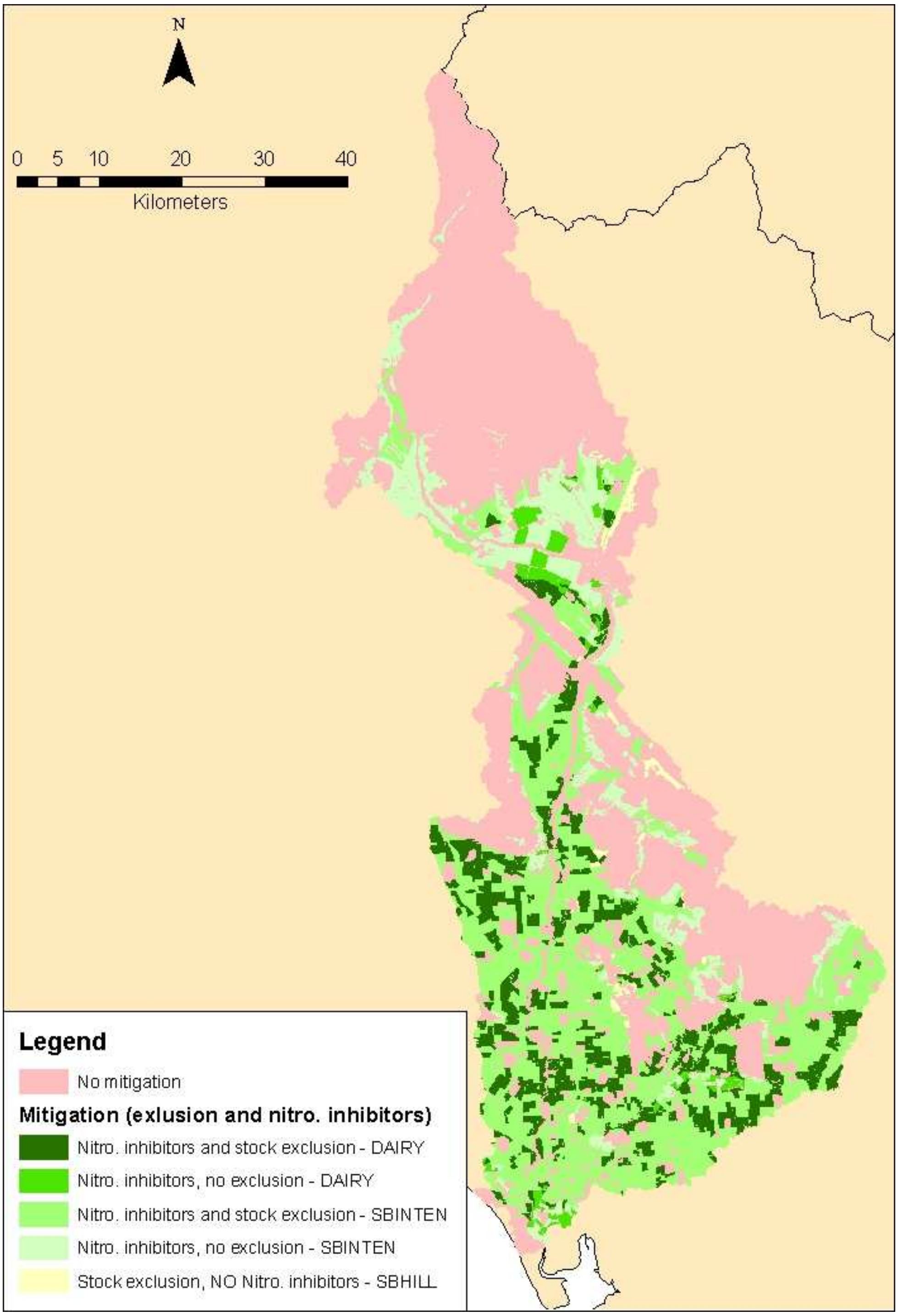


Figure 6.7 Areas which satisfy the criteria for nitrification inhibitors (with and without current stock exclusion).

6.3.3 Herd shelters

Herd shelters were applied to dairy farms within LUC classes 1-4. For areas with an LUC of 1-3, the mitigation was applied in addition to current stock exclusion. Approximately 12% of the total catchment area, marked in green on Figure 6.8, satisfies the criteria for dairy herd shelters given in Table 6.4

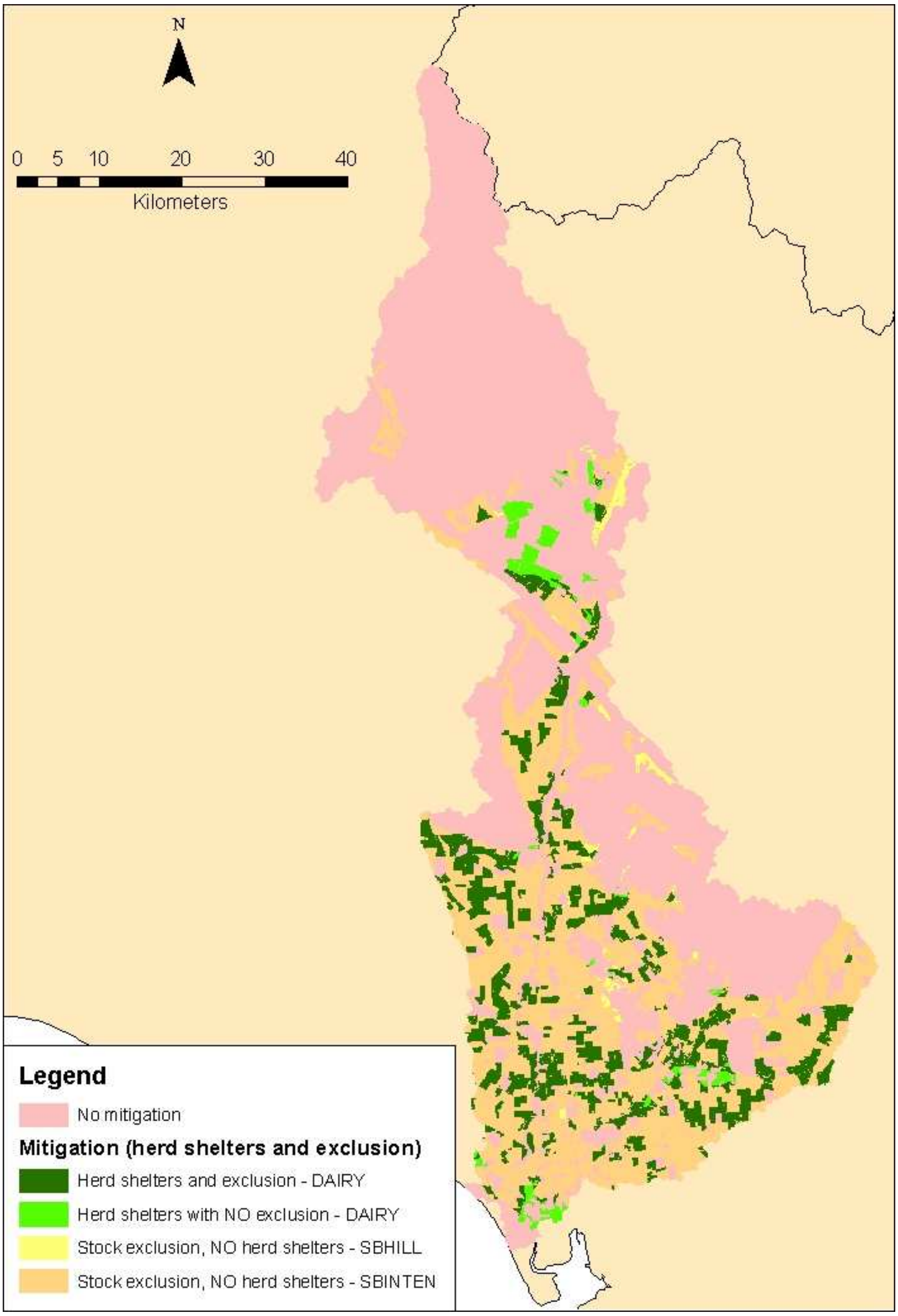


Figure 6.8 Areas which satisfy the criteria for dairy herd shelters (with and without current stock exclusion).

6.3.4 Wetlands

Constructed wetlands were applied to areas with poor drainage for all pastoral land uses (dairy, sheep and beef farming). For areas with an LUC of 1-3, wetlands were applied in addition to current stock exclusion. Approximately 26% of the total catchment area, marked in green on Figure 6.9, satisfies the criteria for wetlands given in Table 6.4.

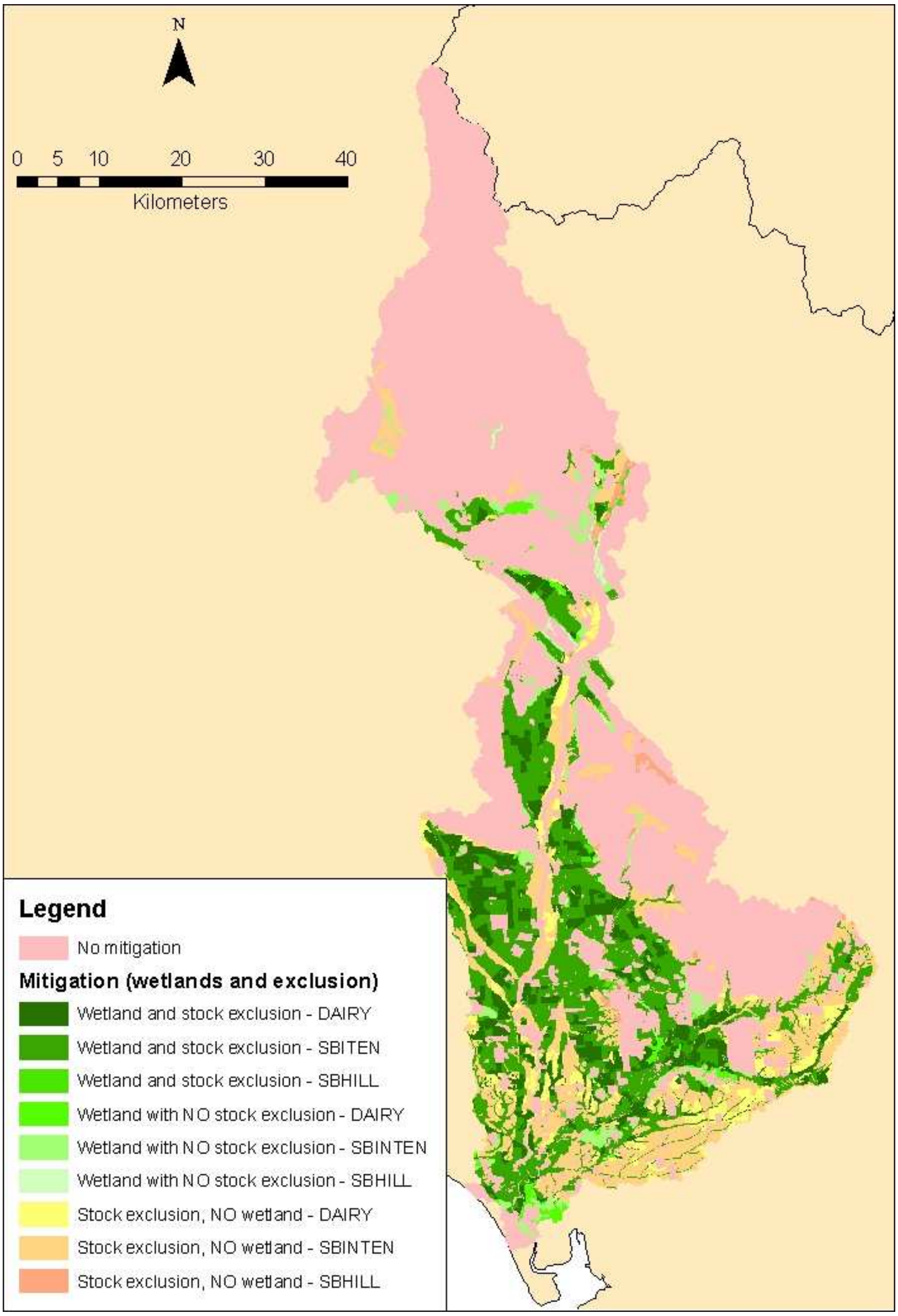


Figure 6.9 Areas which satisfy the criteria for wetlands (with and without current stock exclusion).

6.3.5 Farm Dairy Effluent (FDE) management

FDE management was applied to dairy farms, with different mitigation factors used according to soil drainage status (poor or free draining, see Table 6.4). For areas with an LUC of 1-3, FDE management was applied in conjunction with current stock exclusion. This mitigation covered approximately 13% of the total catchment area, marked in green on Figure 6.10.

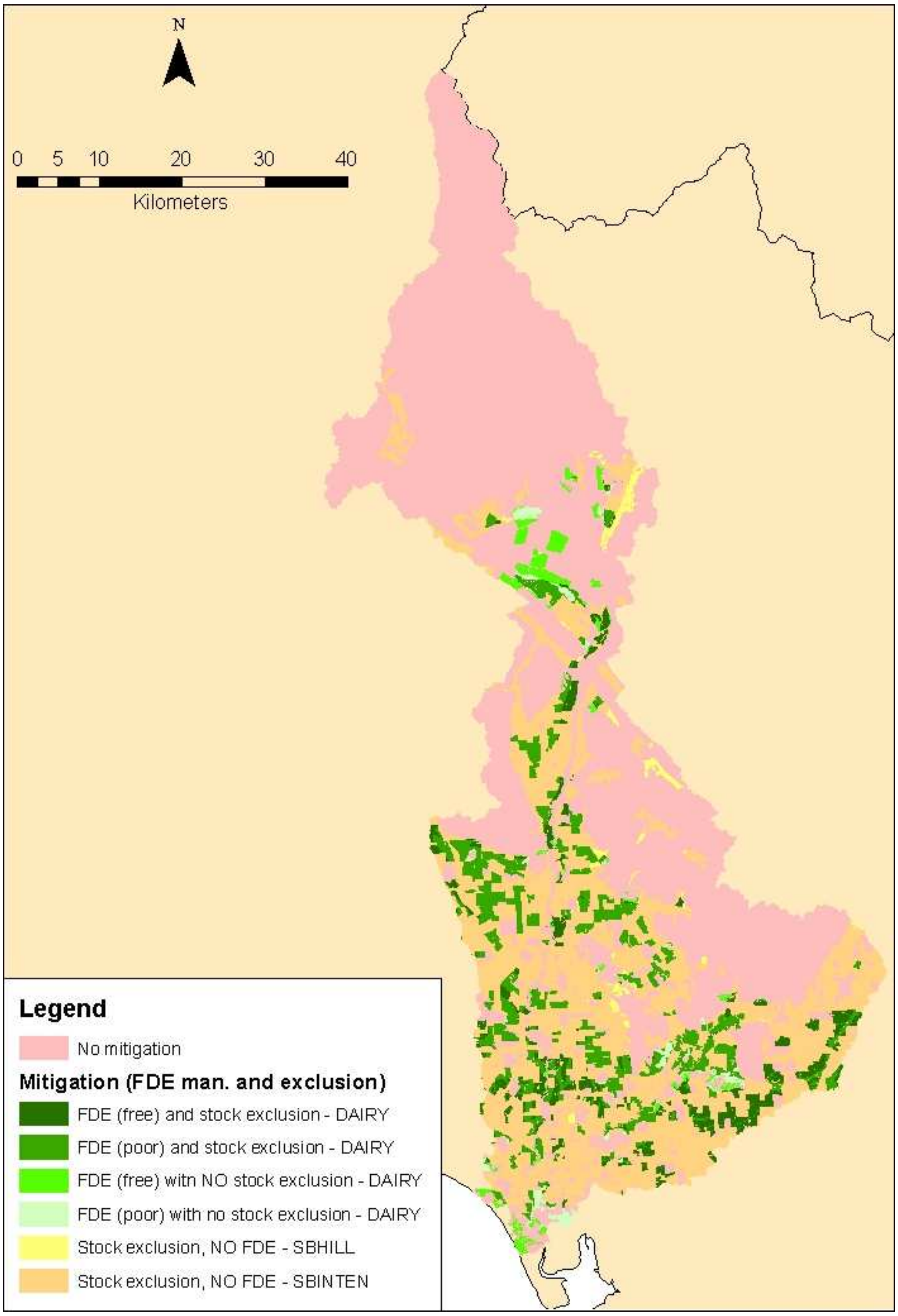


Figure 6.10 Areas which satisfy the criteria for improved farm dairy effluent management (with and without current stock exclusion). Areas in green were modelled assuming improved farm dairy effluent management was practised.

6.4 Model results and discussion

Results from the CLUES default and mitigation scenario runs are given in Table 6.5 and Figures 6.11 and Figure 6.12. Table 6.5 gives the water quality indicator values simulated for Oreti River terminal reach (NZ reach number 15060397) which flows into the Oreti tidal inlet. The loads and yields are in-stream values, which mean that they represent the cumulative totals, less losses and attenuation, of the reach and up-stream tributaries. The calculation of concentration is fairly new to CLUES (see Semadeni-Davies et al., 2009b), and is a median concentration which represents concentrations for typical flow conditions.

The greatest decrease in simulated TN is achieved with the nitrification inhibitor scenario; this is not surprising given that 50% of the catchment meets the criteria for inhibitor application and inhibitors have the highest TN decrease factor. However, nitrogen inhibitors do not decrease TP or *E. coli*. Herd shelters and wetlands offer decreases in TN of around 15%. The decreases in TN simulated with the future stock exclusion and improved FDE management scenarios are only slightly greater, with respect to the default load and concentration, than those that could be expected with current stock exclusion alone.

For TP, the future stock exclusion scenario gives the greatest decrease in TP, but the differences between the different scenarios are not great and there is generally little decrease over what can be expected with current stock exclusion alone.

The greatest decrease in simulated *E. coli* is achieved by wetlands, followed by improved FDE management. As with TP, it is assumed that the use of nitrification inhibitors does not decrease *E. coli* beyond the decrease achieved by current stock exclusion.

Figure 6.11 and Figure 6.12 give an indication of the nutrient sources and areas which can expect the greatest nutrient decreases under the different mitigation scenarios. The spatial pattern of yield decreases is related to the areal extent of the scenarios (see Section 6.3) and their mitigation factors (Table 6.4) respectively. The areas with the highest default-generated yields largely coincide with areas with pastoral land use (see Figure 6.5). It should be noted that for TN, the only sub-catchments with a generated yield greater than 60 kg ha^{-1} yr^{-1} contained point sources (i.e., two meat-works and a sewage outflow) discharging to the stream network. These sub-catchments similarly had the greatest TP generated yields (i.e., >4 kg ha^{-1} yr^{-1}). As the affected sub-catchments were small, the point sources have been marked to show their location. Nitrification inhibitors had the greatest spatial decrease in TN generated yields catchment wide, which reflects this scenario's widespread application. This mitigation scenario also had the greatest decrease in TN load at the Oreti River mouth (Table 6.5). There was very little difference in the mitigation scenario results for TP decreases with all the scenarios having fairly similar results and spatial patterns. This finding is in keeping with Table 6.5 which showed a close similarity in TP loads for the different scenarios.

6.5 Catchment scale modelling summary

CLUES 3.0 has been used to compare the relative decreases in nutrient loads, yields and concentrations and *E. coli* loads that can be expected in the Oreti River catchment with mitigation. The catchment was simulated using default land use with no mitigations and with six mitigation scenarios: stock exclusion (current and future levels); nitrification inhibitors; herd shelters; improved farm dairy effluent (FDE) management; and constructed wetlands. For LUC lasses 1-3, it is estimated that 75% of stream reaches on dairy farms, and 35% of stream reaches on sheep and beef farms, already have stock exclusion in place. For this reason, the scenarios for nitrification inhibitors, herd shelters, FDE management and wetlands were run in conjunction with current stock exclusion.

To summarise, the main findings of the CLUES model simulations were:

- there was no one mitigation strategy that can substantially decrease all of the pollutant loads;
- nitrification inhibitors and herd shelters were most effective for decreasing catchment N losses;
- stock exclusion, herd shelters and improved FDE management were most effective for decreasing P losses;
- wetlands and improved FDE management were most effective for decreasing *E. coli* losses; and
- for greater decreases of all 3 pollutants, full stock exclusion will likely be required on higher LUC classes. The implementation of combinations of GEPs will clearly also deliver greater decreases in pollutant loads.

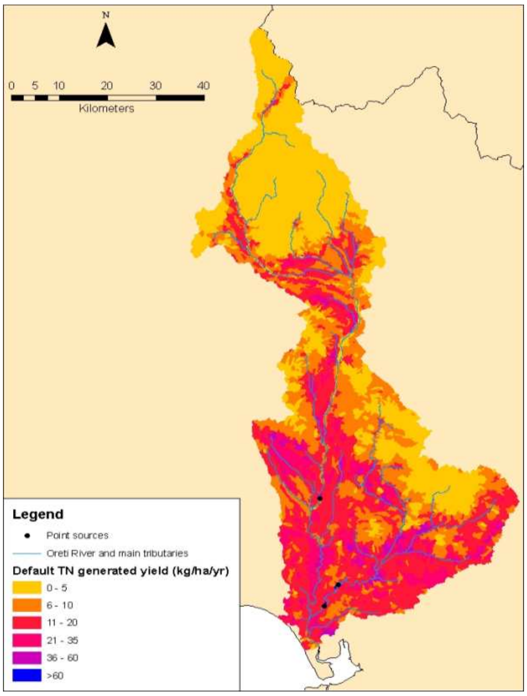
Table 6.5. Water quality indicators simulated for the terminal reach of the Oreti River. The most effective GEP option for each indicator is written in red. Percentage decreases from the default CLUES run are given in parentheses.

	Default no mitigation	Stock exclusion (current)	Stock exclusion (future)	Nitrification inhibitors[1,2,3]	Herd Shelters[1]	Wetlands[1,2]	Improved FDE management[1]
Total Nitrogen							
Load (tonnes/year)	2323	2207 (5%)	2138 (8%)	1821 (22%)	2015 (16%)	2001 (14%)	2151 (7%)
Median concentration (mg/m^3)	966	918 (5%)	889 (8%)	758 (22%)	839 (13%)	832 (14%)	895 (7%)
Total Phosphorus							
Load (tonnes/year)	192	180 (6%)	175 (9%)	-	177 (8%)	-	177 (8%)
Median concentration (mg/m^3)	70	66 (6%)	64 (9%)	-	65 (7%)	-	65 (7%)
E. coli							
Load (10^{15} organisms per year)	48.8	46.3 (5%)	44.0 (10%)	-	45.2 (7%)	40.2 (17%)	42.2 (14%)

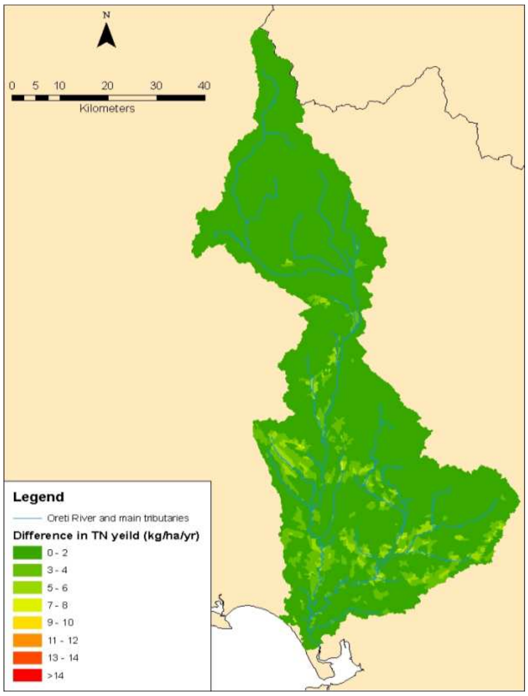
[1]Including current stock exclusion

[2]Nitrification inhibitors and wetlands were assumed to have no effect on TP and are therefore not included in this analysis

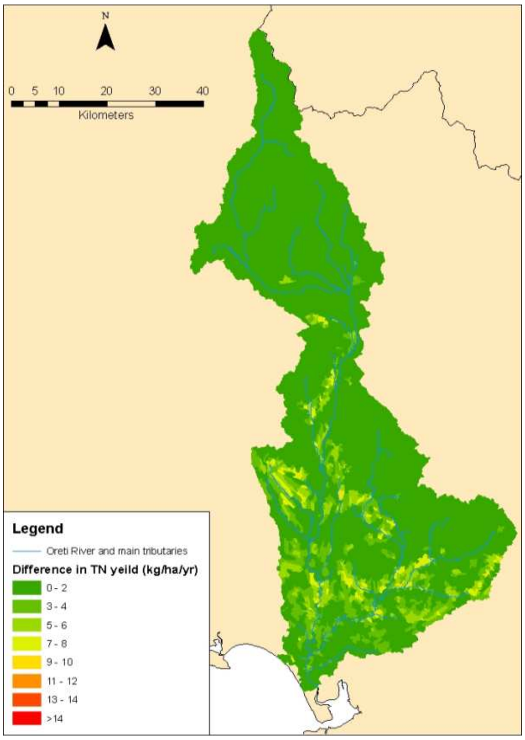
[3]Nitrification inhibitors were assumed to have no effect on *E. coli* and are therefore not included in this analysis



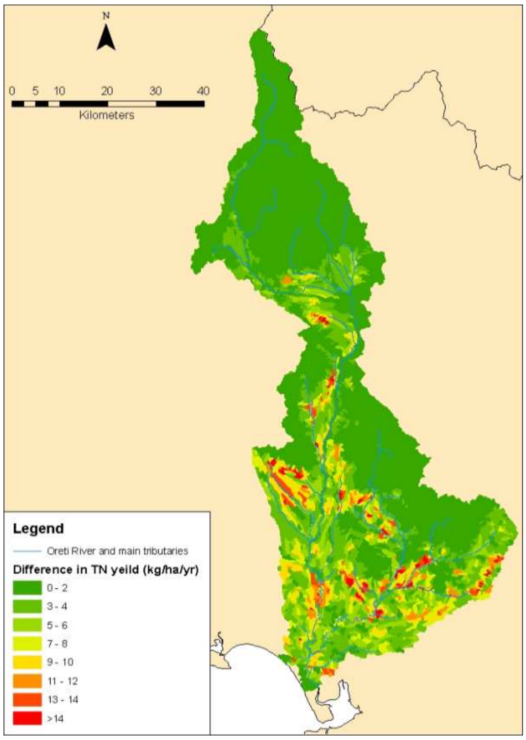
Default



Current stock exclusion

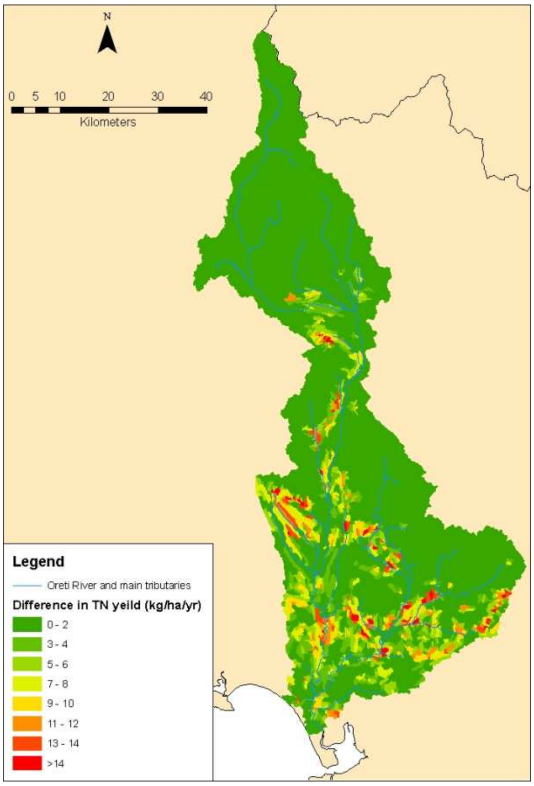


Future Stock exclusion

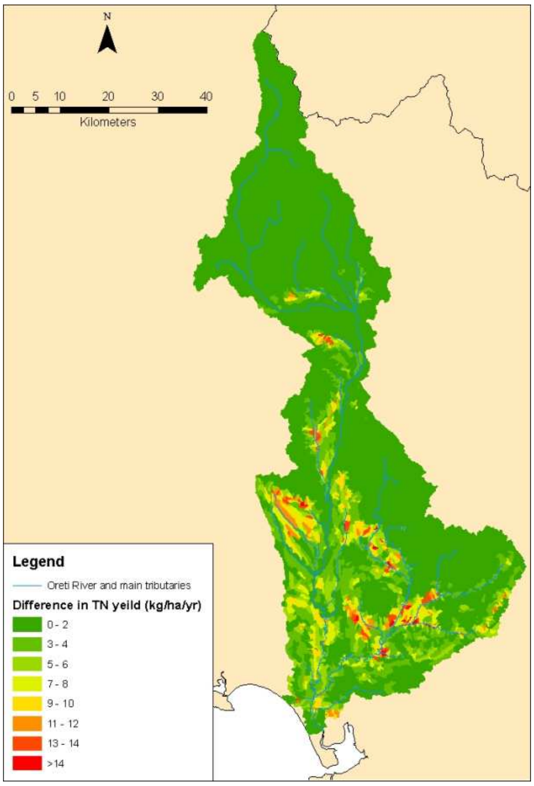


Nitrification inhibitors

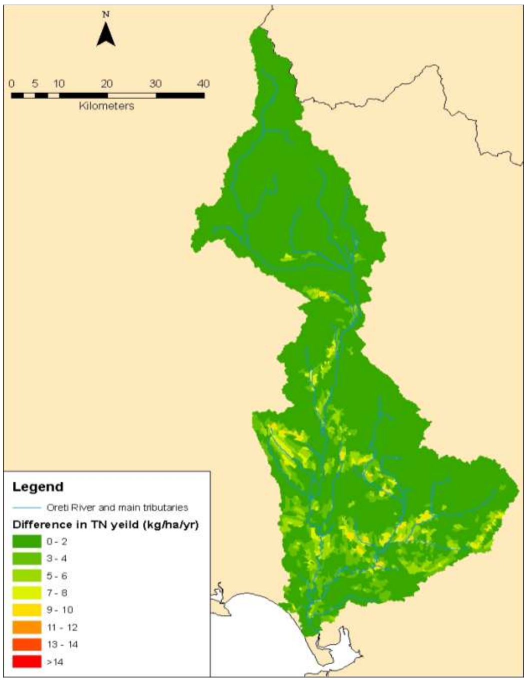
Figure 6.11 TN generated yield simulated for the Oreti catchment showing the default generated yield and differences from the default for the mitigation scenarios. Continued over page.



Herd shelters

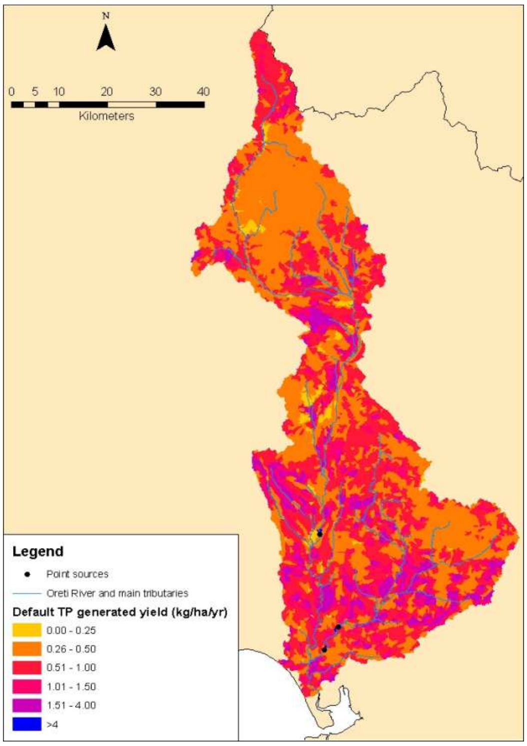


Wetlands

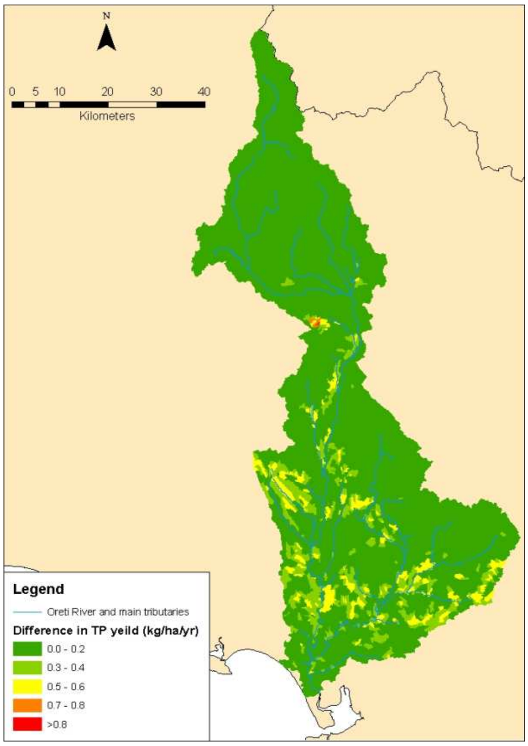


Improved FDE management

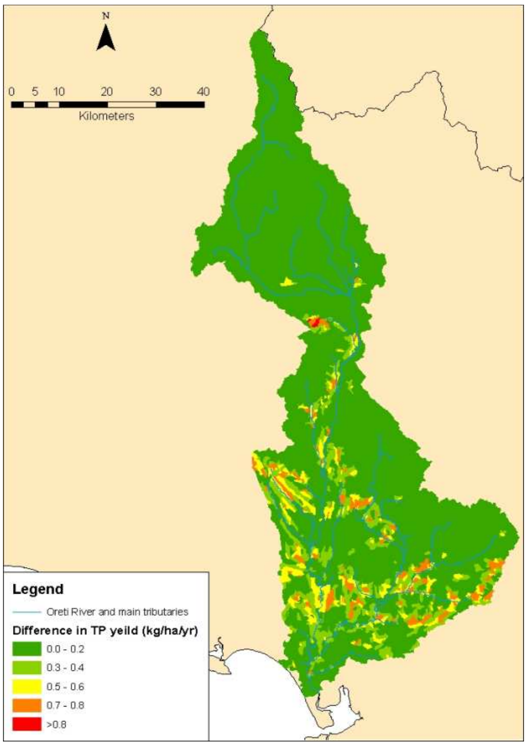
Figure 6.11 continued.



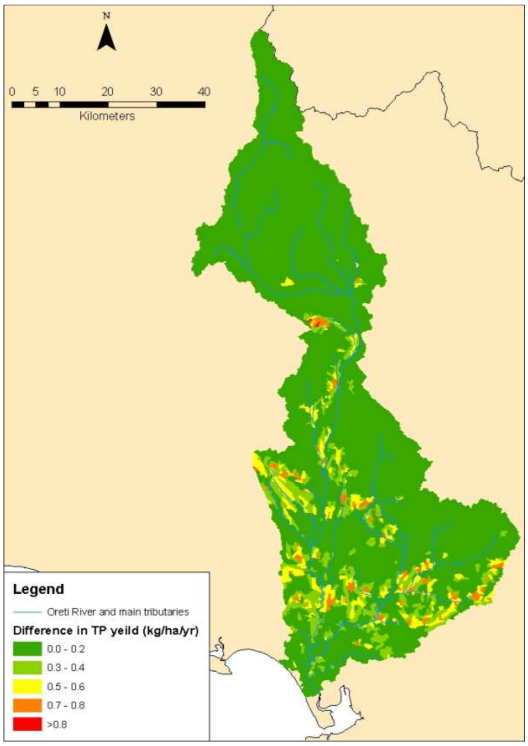
Default



Current stock exclusion

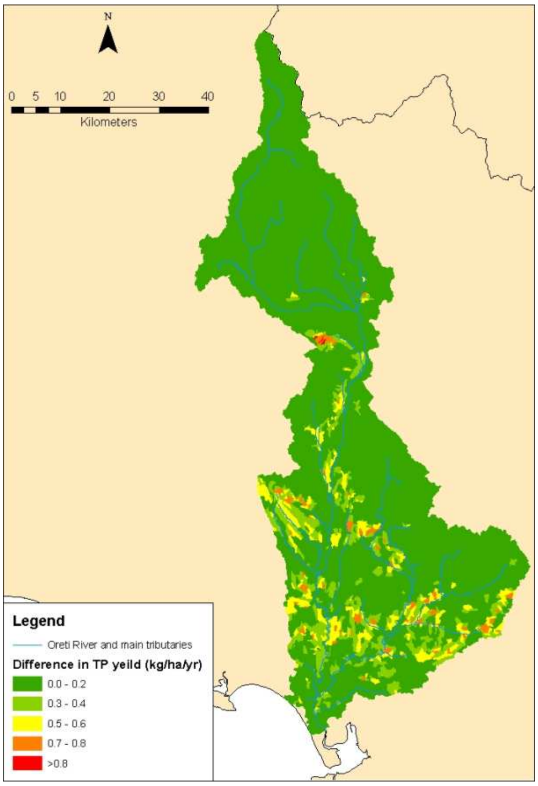


Future Stock exclusion



Herd shelters

Figure 6.12 TP generated yield simulated for the Oreti catchment showing the default generated yield and differences from the default for the mitigation scenarios. Nitrification inhibitors and wetlands do not decrease TP and are not included. Continued over page.



Improved FDE management

Figure 6.12 continued**.**

7. Knowledge gaps

It is apparent from our review that there are many knowledge gaps remaining in our understanding of the effects of land use and management on water quality. Some priority areas for further research are suggested below.

<u>Defining the impacts of animal wintering systems:</u>

- Measurement of nitrate leaching losses from grazed winter forage crops under a much wider range of soil types and climate and management scenarios is required. Particular focus should be placed on the stony shallow and/or very free draining soils that are commonly used. Experimentation within the Pastoral21 (Environment) programme of work has begun to address this, although is mainly limited to the Pukemutu, Lyntley, Lismore and Paparua soils (the latter 3 can be described as stony shallow soil types). A major research trial supported by the Pastoral 21 programme and Environment Southland is currently underway at Five

Rivers to provide data relevant to shallow stony soils in Northern Southland. At completion, it will provide 3 full years of leaching data under treatments of with and without a nitrification inhibitor (DCD) applied.•

- A better understanding is required of flow pathways and the factors that contribute to the variability evident where loads of P and sediment have been documented under winter forage grazing.
- There is only one limited, small scale runoff plot study reporting the role of grazed winter forage crops as sources of *E. coli*. As noted for P, a better understanding of flow pathways and yields of *E. coli* in runoff from these crops is required.

<u>Improving our understanding of contaminant losses to water from dry stock farms:</u>

- Field experimental data documenting contaminant losses to water from dry stock farms is sparse, and the little information we do have mostly comes from trials conducted near Palmerston North. There is a need for more datasets, under a broader range of climate and landscape features, and documenting losses of multiple contaminants. Assessments of the relative effects of dry stock classes against dairy land use would be especially helpful, particularly for P, sediment and *E. coli* losses.

<u>Improving our understanding of contaminant losses to water from horticultural farms:</u>

- To the authors knowledge, no data is available that documents contaminant losses to water from intensive horticultural operations such as bulb production. Given its growing prevalence in Southland, this is an important knowledge gap that needs to be addressed.

<u>Better description of spatial and temporal patterns of contaminant losses:</u>

- Our current modelling frameworks do not adequately account for spatial and temporal variability in contaminant losses from farms. Improved understanding of this variability is important if we are to better define ***when*** mitigation practices will have their greatest effect (e.g. the case for stock exclusion, as described in section 4.2) and ***where*** mitigations should be targeted (the CSA management concept). FRST-funded research programmes are currently being proposed to address these issues. These improved understandings are required if we are to ensure we get the greatest value from mitigation expenditure.

<u>More detailed CLUES modelling of current and future land uses</u>

The CLUES modelling analysis has highlighted two issues which could be addressed with further CLUES simulation:

- The CLUES default land use is based on geo-spatial data from 2001. Over the intervening years, the Oreti catchment has undergone substantial dairy expansion through conversion of dry stock farms. This means that the results presented may not reflect current land use – as was discussed in the comparison of CLUES results with observations. One of the main features of the CLUES package is its ability to create comprehensive land use change scenarios. New land use scenarios could be created for the catchment which could be used to simulate water quality with both current land use and future trends to give an indication of water quality changes over time. An example of this sort of CLUES application can be found in Semadeni-Davies et al. (2009a).
- While nitrification inhibitors, herd shelters, FDE management and wetlands were run in conjunction with current stock exclusion, no other combinations of mitigation strategies were tested. Nor were changes in stocking rates investigated. CLUES could be run with combinations of mitigation strategies and stocking rates to determine whether further improvements to water quality can be attained. For example, can limiting stocking rates or combining herd shelters with nitrogen inhibitors in sub-catchments with high generated nutrient yields improve catchment-wide water quality?

Additional modelling

The effectiveness of mitigation measures on groundwater quality will also need to be assessed at some stage. Appropriate modelling tools (e.g. AquiferSim) will need to be utilised for this type of exercise.

8. Acknowledgements

The authors would like to thank Drs Tony van der Weerden and Richard McDowell for valuable input to and feedback on this report. The assistance of Mr Jim Risk in the description and mapping of Southland soils is also gratefully acknowledged. Feedback from staff at Environment Southland on draft versions of the report is also gratefully acknowledged. This report was funded by FRST through the Envirolink fund.

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10. Appendix I. Glossary and description of GEPs.

Nutrient budgeting: Nutrient budgets account for nutrient flows into and away from farm blocks in fertilizer, feed, animal transfer, animal product and via loss pathways such as leaching and volatilisation. The planning objective is to ensure that nutrient inputs and outputs are balanced to avoid situations of deficit or surplus. The OVERSEER® nutrient budgeting program is a tool that has been developed to assist with such planning decisions.

EffluentEffluent storage: The provision of pond storage is an important management practice for dairy farms on soils with artificial or impeded drainage. This allows farmers to store effluent during wet periods (typically spring) when the soil is too wet to store the liquid applied in the FDE. Sometimes referred to as a "deferred effluent irrigation", it also has the benefit of avoiding the need to irrigate FDE during the busy spring calving period.

LowLow rate effluent application: Low rate effluent application systems typically use sprinkler-type delivery nozzles to deliver instantaneous rates of effluent application of 10 mm per hour or less. This is much lower than delivered by a rotating twin gun travelling irrigator, and allows effluent more time to infiltrate the soil, helping to ensure the liquid and nutrients contained in the effluent remain in the root zone, available for plant uptake. Runoff or drainage that may occur will at least have had some degree of filtering by the soil if a low rate application system has been used.

Nitrification inhibitors: these are chemicals that inhibit the transformation of ammonium to nitrate in the soil. Due to its positive charge, and in contrast to nitrate, ammonium is more readily retained in the soil. Commercial products such as eco-n and DCn contain these chemical inhibitors and thus help to decrease losses of nitrate in drainage water.

Off-paddockOff-paddock wintering: on-going research in Southland indicates that grazed winter forage crops are a significant source of the nitrate lost in drainage from the dairy farm system. Strategies that minimise the deposition of urine to these grazed crops can help to decrease these leaching losses. Stand-off pads (preferably covered) or wintering barns are some of the infrastructure options that could be considered to allow for an off-paddock wintering system.

Restricted autumn grazing: research in Southland has also shown that restricting autumn grazing rounds to 3-4 hours per break, then excluding the animals (removing them to a pad or barn) can significantly decrease urine deposition to land prior to the on-set of winter drainage. This management system has been shown to decrease nitrate losses in drainage from the milking platform by about 40%.

Facilitated wetlands: these types of wetlands utilise naturally poorly drained parts of the landscape where seepage flows can more easily be intercepted. Fencing and planting of these areas helps to create a wetland environment where contaminants that may be entrained in flow can be captured, especially N and sediment.

Constructed wetlands: these types of wetlands are designed to capture sediment and nutrients discharging from obvious discharge points such as tile drains. Some excavation is usually required to create a wetland bed that can be planted with wetland plants such as raupo to help disperse and decrease the velocity of water flowing through it. This also helps to promote the settling of particulate material.

Incorporating low N feeds into diets: ruminant animals consuming a pasture-based diet typically ingest far more N that they require. Consequently, more than 70% of ingested N is excreted via urine and dung. Because urine is the major source of N lost from grazed pastures in NZ, any strategy that can decrease the amount of urinary N deposited to pasture will help to decrease N leaching losses. The incorporation of low N feeds such as maize or cereals into diets has been shown to decrease urinary excretion and N leaching losses. The effectiveness of these types of feeding strategies can be assessed using the OVERSEER® nutrient budgeting tool.

Elimination of stock stream crossings: animals walking or standing in stream beds or on banks cause a number of undesirable effects. Firstly, they tend to deposit significant amounts of nutrient and faecal bacteria to the stream when they urinate or defecate. Secondly, they also destroy stream habitat through the erosive action of their hooves, exposing areas of bare ground that can subsequently act as a source of sediment. Fencing stock out of streams is a simple and cost-effective measure that avoids both of these harmful effects.

Grass buffer strips: A grass buffer or filter strip is a fenced-off area containing dense grasses that runoff water passes through before reaching a water body. These areas act as infiltration or deposition zones that are particularly effective at intercepting particulate material. Their recommended size varies depending on soil and landscape features. It is particularly important that they are located in areas where surface runoff is known to occur or converge. Recent upgrades to the OVERSEER® model also allow users to evaluate the effectiveness of this mitigation practice.