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Prepared for Mackenzie Water Research Limited

Cumulative Water Quality Effects of Nutrients from Agricultural Intensification in the Upper Waitaki Catchment

Mitigations Toolkit

August 2009

INFRASTRUCTURE | MINING & INDUSTRY | DEFENCE | PROPERTY & BUILDINGS | ENVIRONMENT

Contents

Table Index

Figure Index

Appendices

1. Glossary

Assimilative capacity – Theoretical amount of nutrients that a catchment can assimilate without significant environmental effect.

Denitrification – Natural process within nitrogen cycle that converts nitrate to gaseous nitrogen.

Dissolved Reactive Phosphorus (DRP) - Fraction of dissolved phosphorus that is readily available for plant uptake, and contributes to nutrient enrichment in water bodies.

Highly developed soils - Soils that no are no longer building organic matter and have reached their nitrogen storage capacity.

Hydrologically connected areas – Areas from which runoff can enter a watercourse, either directly e.g. a riparian margin, or indirectly e.g. via a ditch or track.

Leaching – The loss of dissolved nitrogen species from the soil profile.

N immobilisation – Conversion of inorganic nitrogen into its organic form where it can be stored in the soil.

Nitrate (NO_3^-) – Inorganic nitrogen species available to plants, highly soluble and vulnerable to leaching loss from the soil. Excess nitrate contributes to nutrient enrichment in water bodies and at high concentrations is toxic to humans.

Piston flow – Process of water being pushed along in a soil by more water entering at the surface.

Point source pollution – Pollution that can be traced to a single source.

Rapid bypass flow – Process by which water or effluent passes rapidly through the soil via cracks, macropores and drains and does not pass through the soil matrix.

Riparian margin – Strip of land on either side of a watercourse.

Total Ammonia (NH_4^+ and NH_3^+) -Inorganic nitrogen species available to plants, bound in the soil and not easily leached. Excess ammonia contributes to nutrient enrichment in water bodies and at high concentrations is toxic to aquatic life.

Total Kjeldahl Nitrogen (TKN) – Measure of both the ammonia and organic forms of nitrogen.

Total Nitrogen (TN) – Measure of all forms of nitrogen (organic and inorganic).

Total Phosphorus (TP) – Measure of all forms of phosphorus (dissolved and particulate).

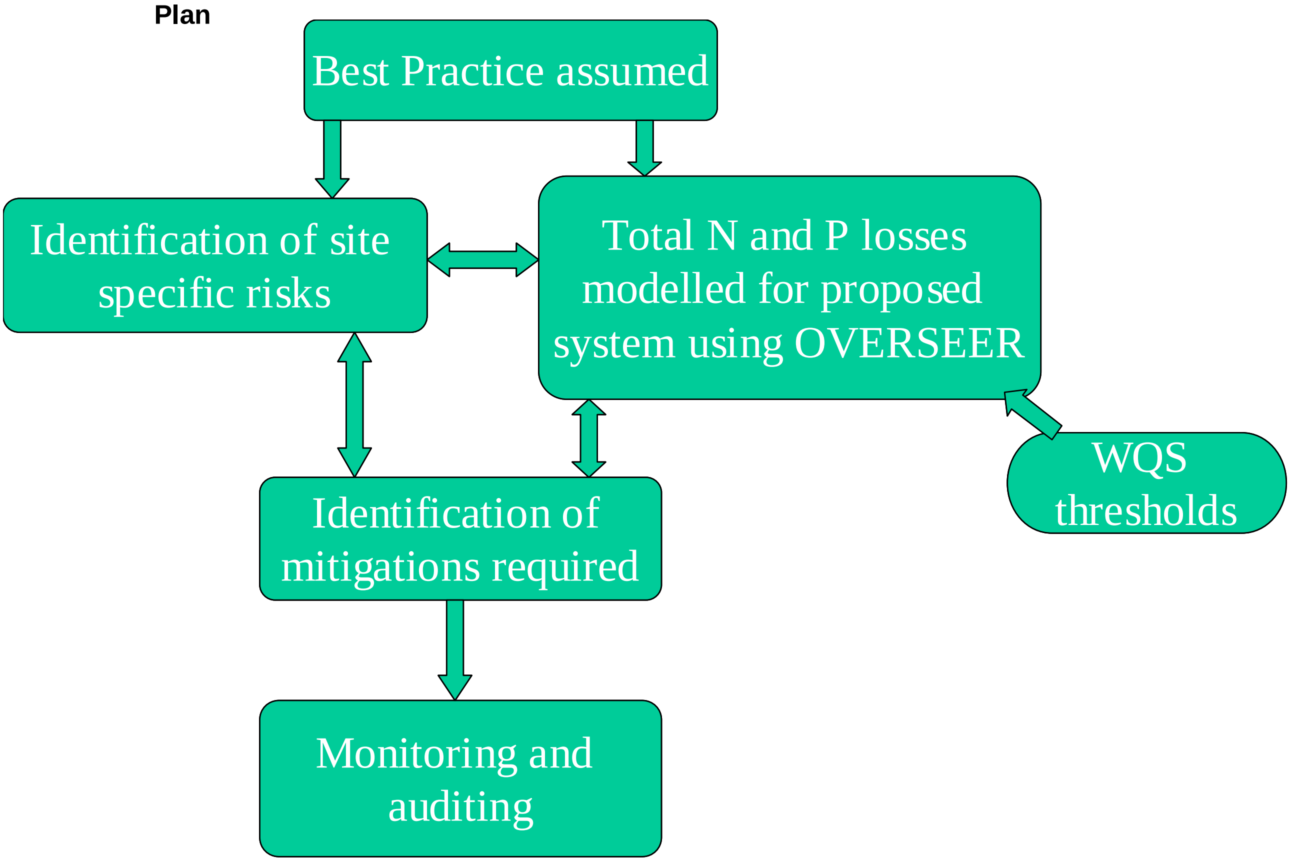
2. Introduction

2.1 Background

In the Water Quality Study (WQS) (GHD, 2009), that assessed cumulative effects of nutrients on water quality from agricultural intensification in the Upper Waitaki, it was found that the additional irrigation proposed in the catchment could take place without significant adverse effects on the environment providing that sufficient nutrient reduction was effected on farm.

The process advocated for effecting this on-farm nutrient reduction is through Farm Environmental Management Planing. A clear process for building a Farm Environmental Management Plan (FEMP) is laid out in this toolkit. An overview schematic of the process of building a FEMP is shown in Figure 1 below.

Figure 1 An overview schematic of the process of building a Farm Environmental Management Plan



2.2 What is a Farm Environmental Management Plan?

Farm management planning and the use of best management practices are methods commonly used to reduce diffuse pollution from farms, and have been trialled in New Zealand and internationally, both as voluntary initiatives and through regulation. Farm Environmental Management Plans assist farmers in reducing their environmental impact through providing a framework for assessing losses and identifying and mitigating site-specific environmental risks.

Diffuse pollution, as the name suggests, does not come from a single traceable source. In many cases the impacts are both temporally and spatially distanced from the source. This makes measurement from and traceability to an individual property difficult. For this reason, instead of measuring the losses, the emphasis is placed on the implementation of techniques that are known to reduce nutrient generation and transport.

2.3 Purpose of a Farm Environmental Management Plan

The FEMP serves two purposes, to ensure the proposed farm system can meet the nutrient mitigation requirements set out by the Water Quality Study, and to identify and mitigate other farm specific environmental risks that arise from the inherent characteristics of the farm or from the proposed farm system and its management. These specific farm risks include uncontrolled discharges that are not identified in the farm nutrient budget but that may still have an environmental effect.

2.4 Scope of report

This toolkit is divided into three principal sections:

» A description of nutrient loss pathways from farms in general and in the Upper Waitaki in particular;

» The process of building a Farm Environmental Management Plan; and

» A section that describes each of the mitigation measures in detail with reference to implementation, efficacy, limitations to use and impact on other pollutants. In addition, a brief suggestion of how to monitor the measure is given.

The building of a FEMP is also divided into four stages:

» Stage 1 describes some mandatory good agricultural practices that need to be implemented across the farm, and include the base assumptions of the OVERSEER model[1]. This helps to validate the use of the model on the property.

» Stage 2 is the construction of a representative farm model in OVERSEER and the confirmation of any required mitigation. Once the mitigation requirements have been established, the proposed farm practices can be altered and modelled to give the required mitigation for the sub-catchment and receiving environment. The changes to proposed farm practice will be noted to be included in the FEMP.

» Stage 3 is the identification of site–specific environmental risks through the completion of a Farm Environmental Risk Assessment (FERA), and the choosing of management options or techniques to mitigate these specific risks. The site-specific environmental risks on the farm include both the physical characteristics of the farm as well as the farming system and how the farm is managed. These site specific issues may not be reflected in a modelled nutrient budget on the irrigated areas, but will need to be addressed.

» Stage 4 describes the proposed monitoring and auditing strategy.

[1] In the future, should an alternative model be used, the assumptions for that model would need to be specified in this good agricultural practice section.

A central tenet of this process is proportionality: The requirements to effect a low nutrient reduction are less onerous than those for a high nutrient reduction. In this way 'overkill' for minor issues is avoided and in areas where more environmental protection is required, larger scale or more numerous strategies are required. In the OVERSEER nutrient budgeting model, a small mitigation requirement may be achieved with relatively small changes to farm management; however, large mitigation requirements may require farm system changes. However, even when the mitigation requirement is low or zero, the initial section detailing good agricultural practices are still mandatory, a representative farm model must be constructed and the FERA still needs to be completed and the identified issues addressed.

2.5 Implementation of a Farm Environmental Management Plan

The responsibility for meeting the nutrient reduction requirements set out by the WQS and the implementation and monitoring of the FEMP lies with the farmer. The efficacy of the all mitigations depends on their implementation as well as other site factors, therefore the magnitude of reductions cited are indicative only, and a robust monitoring programme is essential.

3. Nutrient Loss from Farms

3.1 Critical Factors

Nutrient loss from farms can be costly and has the potential to cause degradation of waterways, groundwater and to release greenhouse gases.

There are three critical factors in assessing the risk of causing environmental degradation from nutrient loss:

» A source of nutrients;

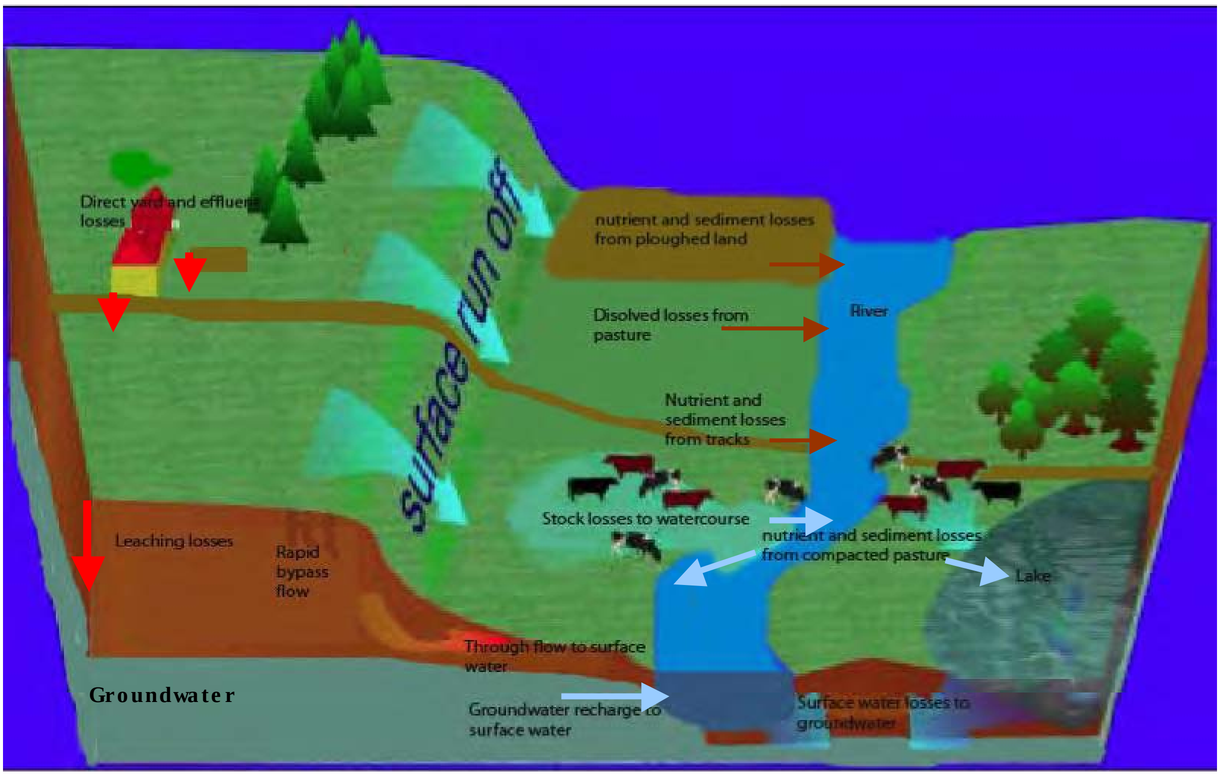
» Nutrient transport and pathways; and

» A receptor or receiving environment.

The source of nutrients represents the availability of nutrients in the system, usually in or on the soil. Transport is the vector required to remove the source nutrients and the pathway is the route of transport. The receptor is the receiving environment that the pathway leads to.

Using the concept of the receptor allows us to assess the nutrient losses in relation to the sensitivities of the receiving environment, setting the farm in the environmental context of the area. For example a 'high risk' field that would run off into another field may not be as great a concern as a 'medium risk' field that would run into a watercourse. Figure 2 shows the typical runoff pathways.

Figure 2 The typical pathways to receptors[2]



[2] Adapted from Defra, 2004

3.2 Mitigation Measures

Mitigation measures can be divided into three groups; source, pathway and receptor. The emphasis is on farm control of nutrients, therefore the toolkit focuses on source and pathway measures.

The key to successful application of appropriate mitigation measures is to match them to the methods of loss and transport. The loss and transport of phosphorus and nitrogen are described below.

3.2.1 How is Phosphorus lost and transported

Phosphorus from agriculture is mainly lost in one of three ways (McDowell *et al*., 2005):

- Point source losses commonly associated with yards, tracks and manure, or effluent storage facilities.
- Incidental losses through surface runoff, where a rainfall event coincides with a recent application of manure or fertiliser, or through surface run off or by subsurface flow via deep cracks and artificial drainage. The 'incidental P' lost will be in the same form as the applied material as there will not have been reaction with the soil. This incidental P loss may account for substantial losses and typically occurs when best management practices are not employed.
- Background losses, where the incorporated P has reacted with the soil, and then transported via water or wind. The majority of phosphorus when it enters soil solution will be either adsorbed onto soil particles or immobilised into organic matter, leaving only very small concentrations in the soil solution at any one time, therefore typically only small quantities of phosphorus are available to be leached (Brady, 1990). Generally P leaching in New Zealand soils is not regarded as a significant P loss risk (Ants Roberts. *Pers Comm*. 2009). However, P leaching losses may occur in coarse textured soils with little active clay content and low organic matter accumulation where the P status is high and drainage volumes are also high e.g., some soils on the West Coast (Ants Roberts. *Pers Comm*. 2009). Where soil P sorption is low and drainage volume high, the soluble P is more vulnerable to be lost with drainage, although where there are subsoils with higher clay content this P may be retained deeper in the profile (US EPA, undated).

In pastoral situations dissolved P losses predominate (McDowell, 2004). This predominance is because when water moves overland, the stabilising influence of the pasture reduces the loss of soil particles causing soil erosion, and soil wash occurs instead. As water runs overland it picks up fine particles, dissolved P and any manure and fertiliser not incorporated into the soil. This dissolved fraction is the most bio-available form of P and therefore represents the most risk to receiving waters (e.g. US EPA, undated). Although, as the P content and reactivity to P of the eroded material is usually greater than the source soil, the sediment can readily adsorb dissolved P in the runoff (US EPA, undated), so not all dissolved P picked up will necessarily be immediately available once in the watercourse.

Soil particles, through runoff or wind erosion, or transport down the soil profile into artificial drains can also represent a major P loss as particulate P in pastoral systems. An important consideration is the fact that the top few millimetres of soil contains the greatest amounts of accumulated P, and erosive forces such as water and wind generally interact with this top layer soil.

3.2.2 How is Nitrogen lost and transported

Nitrogen from agriculture is lost either through:

- Small point sources;

» Incidental losses;

» Denitrification; or

» Leaching.

Of principal concern here are the inorganic species of N that are associated with nutrient enrichment of water, nitrate-N and ammonium-N. Nitrate is negatively charged, therefore is not held on to the negatively charged colloids that dominate most soils. Nitrate is very soluble and when it is in the soil it is dissolved in the soil water. This makes it very accessible for plants and other organisms; however, it also means that once the crop has been removed or has stopped growing, any unused nitrate left in the soil water is vulnerable to leaching. During periods when rainfall exceeds evapo-transpiration, the predominant direction of water movement through the system is down. When water in the soil profile begins to drain by gravity or is saturated and pushed down by more rainfall, the unused nitrate is carried down the soil profile into ditches, drains, shallow and deep groundwater. Ammonium-N can also cause nutrient enrichment and has specific toxicity at elevated concentrations, however it is readily fixed in the soil's cation exchange capacity (CEC) and is therefore less vulnerable to leaching losses.

In stocked systems in New Zealand, the most important source of N loss will be leaching from urine patches (e.g. Di and Cameron, 2004, Monaghan *et al*., 2007). N is applied to soil far in excess of the plant requirement in a urine patch (Di and Cameron, 2004) therefore much of the N is not taken up and is consequently vulnerable to leaching. Where soil is very thin, any unused nitrate in the soil profile is especially vulnerable to being leached beyond the root zone.

The increase in intensification of (dairy) systems tends to an increasing 'leakiness' of future N leaching losses (Monaghan *et al*., 2004). Ledgard *et al*, (2000) showed that a 3-fold increase in total N inputs resulted in a 4-fold increase in N surplus and a 4-5-fold increase in leaching with a halving of N use efficiency.

3.2.3 Nutrient losses in Upper Waitaki

The principal concern in the Upper Waitaki Catchment is nutrient enrichment of groundwater, streams and lakes that can lead to degradation of environmental values. The two pollutants of concern are N and P. The mechanisms of loss for the key pollutants are important for effective mitigation strategies.

The risk of N loss will be primarily through leaching although practices such as stock encroachment in waterways and direct discharge of tailgate losses from border dyke irrigation will also contribute to N losses.

The high proportion of dissolved organic nitrogen in some surface and groundwater samples (GHD, 2009) suggests that in some areas of the catchment the greatest N losses are in the organic form. This is likely to reflect the background losses from very extensively farmed or natural environments with relatively minor anthropogenic or agricultural activity.

The soils in the Upper Waitaki Catchment are in general well drained, although many of the associations have waterlogged riparian margins and there are many wetlands, swamps, lagoons and tarns across the basin (after Webb, 1992). These waterlogged margins and wetland areas suggest a capacity for denitrification which has been modelled in the WQS.

The risk of P loss will be associated with overland flow from hydrologically connected areas, which in many areas will be restricted to riparian margins, through runoff from hill country, and direct deposition or discharge into watercourses. In addition, in areas with low P retention and well drained soils, elevated concentrations of P have been measured in groundwater, indicating that P losses with drainage water are also occurring (GHD, 2009). Wind erosion is an issue in some parts of the catchment and may have implications for some soil associated pollutants such as phosphorus, however the nutrient model used in the WQS does not specifically account for wind blown losses and therefore these have not been considered.

The average nutrient loss from farms in the Upper Waitaki Catchment is low by national standards. Extrapolated leaching losses ranged from less than 3 to 26 kg N /ha with higher losses coming from dairy units, farms with high stocking rates and farms with a high percentage dedicated to forage or arable crops (GHD, 2009). Extrapolated phosphorus losses ranged from 0.01 to 1.1 kg P /ha with higher losses associated with border dyke irrigation, hill country farming, dairy and deer units (GHD, 2009).

4. Developing a Farm Environmental Management Plan

This section describes the process of building a FEMP.

4.1 Data requirements

4.1.1 Farm nutrient generation

The estimates for each farm have been provided individually to farmer funders of this study. These are summary estimates for the whole farm and are provided for the current land use and that proposed under irrigation.

These estimates are a good guide as to the magnitude of nutrient generation on a property. However, for detailed farm nutrient management and development of management measures then each farm must construct a robust individual farm OVERSEER nutrient budget model[3]. These must be prepared by or validated by a suitably qualified person. The individual farm OVERSEER outputs will also clarify the nutrient reductions required. If the WQS has modelled higher loads than the farm specific OVERSEER nutrient budget, then any required nutrient reduction will be lessened and vice versa.

4.1.2 Mitigation requirements

Within streams, rivers and lakes there may potentially be a natural capacity to assimilate the effects of additional nutrient runoff without there being any adverse environmental impact. The magnitude of this capacity varies from catchment to catchment.

The WQS has suggested thresholds at nodes throughout the catchment and in Lake Benmore. Where the nutrient load at the node is predicted to exceed the threshold under proposed scenarios, nutrient mitigation will be required. Mitigation requirements for each property, translated into nutrient thresholds, have been provided to funders.

4.1.3 Nutrient allocation

For the purposes of the WQS, where the assimilative capacity in a sub-catchment is predicted to be exceeded, the nutrient 'overburden' has been divided equally between all the areas of land draining to the node. This method of allocation has been used for simplicity to indicate mitigation requirement and is not endorsed by the WQS as the preferred method.

4.2 Building a Farm Environmental Management Plan

4.2.1 Stage 1 – Mandatory good agricultural practices

The implementation of these practices established a commitment to a good base level of practice. It also, by including the base assumptions of the OVERSEER model that the inputs to the farming system are made using good practice, means that the consequent farm modelling can be viewed with more confidence.

[3] The OVERSEER model should be used according to the protocol in Appendix B

The mandatory good agricultural practices adopted in this study are shown in Table 1.

Table 1 Mandatory good agricultural practices

Mandatory good agricultural practices	
Fertilisers applied according to code of practice for fertiliser use (FERT Research, 2002)	This is a basic assumption in OVERSEER. The fertiliser users' code of practice aims to ensure that where fertilisers are used that they are used safely, responsibly and effectively and in away that avoids, remedies or mitigates any adverse environmental effects. The code of practice includes guidance on fertiliser use, application, storage, transport, handling and disposal.
Use of a recommendation system for fertiliser	Planning fertiliser applications to all crops, determining crop requirement and accounting for soil nutrients and organic nutrient supplies, all reduce the risks of applying excessive fertiliser above the crop requirement. This maximises the economic return from the use of fertilisers and reduces the risk of causing nutrient pollution of the environment (MAFF, 2000).
All sources of nutrients including applied effluents and soil reservoirs accounted for	This includes imported sources as well as soil reservoirs and entails regular soil testing. This is an important factor in all farming systems, especially in more intensive systems that produce effluent and/or manures or where manures or organic material is imported onto the farm. The re-application of organic manures to land is often thought of as a disposal of a waste product, and the available nutrients within the organic manures are not accounted for. The use of an integrated nutrient budgeting tool such as OVERSEER automatically accounts for nutrients supplied in organic manures.
Fertiliser application applied evenly	The even application of fertiliser is an assumption of the OVERSEER model as included in the fertiliser code of practice. Fertiliser spreaders should be tested and calibrated in-house at least annually and by an external tester every 5 years.
Irrigation and effluent applied evenly	The even application of water and or effluent is an assumption of the OVERSEER model. Irrigators should be tested and calibrated in-house at least annually and by an external tester every 5 years.
Crop, cultivation, nutrient inputs and yield records kept per farm management unit	Maintaining good crop input records is important for: » The calculation of cumulative annual organic fertiliser applications and also their contribution to long term nutrient supply; » The determination of realistic crop yields that are used to determine crop requirement; and

	» The provision of accurate inputs to the OVERSEER nutrient budgeting model that is used here as a proxy for measuring diffuse pollution.
Good design of irrigation systems	Design will match soil properties and low application amounts on shallower soil to prevent summer drainage.
Robust irrigation scheduling	Good irrigation scheduling to prevent summer drainage.
Supplement and feeding out management	Proper storage of supplements and responsible methods of feeding out that do not result in accumulations of excreta on small proportions of the farm. Where large amounts of supplements are fed out, a feed pad should used.
Winter grazing management	Winter management of stock to prevent pugging and high densities of stock in one area for long times.

4.2.2 Stage 2 – OVERSEER and meeting WQS mitigation requirements

The use of OVERSEER as a proxy measure for nutrient losses from farms for managing nutrients at a catchment scale has a precedent both in Environment Waikato's proposed rule changes for nutrient regulation around Lake Taupo, and in Horizons Regional Council's proposed 'One Plan'. Protocols for the use of OVERSEER were published by Manderson and Mackay (2008. A. Manderson. *Pers comm*. July 2009) for Horizons Regional Council, see Appendix B. These protocols should be observed.

Once the individual OVERSEER model has been constructed for current and proposed (without additional mitigation) farming systems, the exact nutrient reduction required can be confirmed. Required changes to farm management practices or farm systems can now be modelled and refinements made until the required mitigation is achieved. The inputs and outputs from OVERSEER will need to be verified and will form an auditable part of the FEMP.

4.2.3 Stage 3 – Identification and mitigation of site specific environmental risks

All farms are unique and will have environmental problems unique to them as well. Site-specific environmental issues may not be reflected in a modelled nutrient budget, but will still need to be addressed. Site-specific environmental issues can be identified through a Farm Environmental Risk Assessment (FERA). A FERA examines the physical characteristics of the farm, the potential receiving environments and risks associated with current and proposed farm practices, An example of a FERA can be found in Appendix A.

Once a FERA has been completed, the table of site specific mitigations (Table 2) is consulted and appropriate mitigation tool is selected[4]. Chapter 5 in the toolkit provides information on how the mitigation options work, their likely efficacy and a suggestion on how to monitor.

[4] The mitigation toolkit is not exhaustive; therefore if no appropriate mitigation is found, seek advice from a suitable qualified person. In the case of all mitigations, how the option is to be monitored and audited should be considered.

Table 2 Site-specific mitigation options

Specific mitigation area	Issue present on your farm? Y/N	Mitigation measure	Target pollutant
Soil	Y/N		
		Tillage and residue management	N and P
		Minimise soil erosion	N and P
		Reduce compaction	N and P
		Catch cropping	N and P
		Nil or restricted winter grazing	N and P
		No winter grazing of forage crops	N and P
Effluent / Infrastructure	Y/N		
		Minimise effluent produced through clean and dirty water separation	N and P
		Increase effluent storage and practice deferred effluent irrigation	N and P
		Manage effluent application rate	N and P
Stock nutrient losses	Y/N		
		Nitrification inhibitors	N
		Urease inhibitors	N
		Reduce dietary N and P intakes	N and P
		Regularly move winter feeding areas	N and P
		Livestock restriction to watercourse	N and P
		Nil or restricted winter grazing	N and P
		No winter grazing of forage crops	N and P
Fertiliser	Y/N		
		Reduction of P index	P
		Split N applications	N
		RPR as phosphate fertiliser	P
Water / irrigation	Y/N		

Specific mitigation area	Issue present on your farm? Y/N	Mitigation measure	Target pollutant
		Fertigation	N
		Irrigation scheduling	N
		Change irrigation method	N and P
		Pond and Spray border dyke systems	N and P
		Laser level existing border dykes	N and P
		Border dyke irrigation management	N and P
		Minimise effluent produced/ clean and dirty water separation	N and P
Cropping	Y/N		
		Minimum till and residue management	N and P
		Catch cropping	N and P
		No winter grazing of forage crops	N and P
		Remove compaction	
		In field buffer strips	N and P
Runoff	Y/N		
		Riparian margins	N and P
		In field buffer strips	N and P
		Minimise soil erosion/reduce compaction	N and P
		Restrict livestock access to watercourse	N and P
		Constructed wetlands	N and P
		Swales	N and P
		Sediment traps	N and P
		Track management	N and P
		Cattle tracks	N and P

4.3 Next steps

Once all three steps have been competed, the proposed farm system with mitigation should be described and a table of the mitigation and management options should be collated with proposed monitoring and auditing options, an example is given below in Table 3.

Table 3 Example table of mitigation options, monitoring and auditing

Mitigation	Monitoring	Auditing
Fertilisers applied according to code of practice for fertiliser use		Self certification
Accounting for all sources of nutrients including applied effluents and soil reservoirs	Soil and effluent testing and cumulative effluent inputs per management unit	Reconciliation of fertiliser, effluent and soil records with nutrient budget for example blocks. Submission of examples soil and effluent tests
Even fertiliser application	Calibrate and optimise fertiliser spreaders annually or with change in fertiliser prill size and every 5 years by an external auditor	Submission of testing and calibration
Even irrigation and effluent application	Calibrate and optimise irrigators annually in house and every 5 years by an external auditor	Submission of testing and calibration
Record crop, cultivation, nutrient inputs and yields per farm management unit	Upkeep of records	Submission of example block records
Fencing stock out of waterways through riparian fencing and planting	Surface water testing of race as it enters and exits the property	Annual auditing visit.
No winter application of fertiliser	Field records	Signed field records
N fertiliser applications split to under 50 kg N/application	Field records	Signed field records

4.4 Monitoring and Auditing

4.4.1 Monitoring

The monitoring and auditing of the FEMP is as important as the FEMP itself. A monitoring plan should be included in the FEMP, describing the location, frequency and parameters to be monitored, the 'triggers', and if applicable the contingency plan to be put in place immediately should the triggers be exceeded while the root cause analysis is carried out. Table 4 below shows and example of a monitoring plan.

Table 4 Example of Location, frequency and parameters for monitoring

		Location	Frequency	Measured parameters to include	Triggers	Contingency plan if triggers are exceeded
Soil	Soil nutrient testing	All blocks in rotation	1 in 3 years	Standard suite of soil nutrients, pH C, N and organic matter	Olsen P of 30	Reduce or stop addition of P fertiliser to area and monitor
Soil	Soil compaction survey	All blocks	Annually	Surface and subsoil compaction	Compaction, surface capping	Remove compaction with appropriate tool
Soil	Wet weather survey	All blocks	Annually	Runoff from tracks and centre pivot tracks	Runoff occurring	Immediately review current runoff mitigation options for tracks and pivot tracks. Introduce further runoff removal infrastructure
Water	Irrigation application		Annually in house and 1 in 5 years by an independent	Application uniformity	<80 %	Optimisation of the irrigator performance will be performed at the time of testing
Fertiliser	Fertiliser application		Annually in house and 1 in 5 years by an independent	Application uniformity	<80 %	Optimisation of the spreader performance will be performed at the time of testing
Pasture	Ground cover	All blocks	2 x per year	% Ground cover	< 80 %	Soil nutrient and compaction testing should be performed to identify possible causes

Where triggers are exceeded, the immediate contingency plans should be implemented while a ‘root cause’ analysis is carried out. Any further mitigation measures to be adopted as a result of the monitoring should be added to Tables 3, 4, 5 and 6.

1) Is the current mitigation option implemented correctly?

No – Implement and monitor

Yes – to 2)

2) Has anything changed in the farm system?

Yes – remodel and monitor

No – to 3)

3) Have there been abnormal conditions at the time of trigger breach?

Yes – continue monitoring to see if trigger breach continues

No – Seek advice if suitably qualified person to further investigate root cause and suggest appropriate mitigation.

If emergency conditions occur that risk a pollution event, such as a severe flooding event that puts the effluent storage system at risk or a catastrophic failure of the effluent system, seek immediate guidance from you regional council.

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4.4.2 Auditing

The auditing process allows both the farm operator to illustrate and other interested parties to have confidence that the management practices and mitigations planned for the farm are being implemented. In addition the audit shows that there is a mechanism for the adaptive management of the property should the chosen mitigation or management not perform to expectations.

An annual audit is proposed, and requires both external and in-house input, and should be submitted on a fixed date annually. The proposed audit plan should be included in the FEMP.

Table 5 below gives and example of a proposed annual audit sheet.

Table 5 Table showing proposed contents of an example annual audit report

Audit measures	Action in the case of non-compliance
Additional auditing that must be done externally	
Check riparian planting and fencing is present where it should be and that it is intact, plus photographs	Any failure in the integrity of the fencing should be repaired immediately or a barrier placed around gap to prevent stock access until repair is made
Check fertiliser storage and filling area.	There should be no possibility of loss of fertiliser to drains or direct discharge to ground. Any drains should be covered, or the filling area moved to where no discharges will occur.
Annual audit of OVERSEER nutrient budget and report based on previous 3 years. Submission of compliance with thresholds	Should the OVERSEER report show losses exceeding the threshold, further mitigations should be adopted to effect a reduction in nutrient loss to below thresholds.
Fertiliser spreader and irrigation testing and calibration 1 in 5 years by independent auditor	Spreaders and irrigators not performing should be recalibrated
Additional auditing that can be done either externally or internally	
Submission of example irrigation schedules and calculated water use efficiency	Where calculated water use efficiency is such that the trigger is exceeded, remedial action of how the system is to be optimised should be submitted, and followed up in the next audit
Annual quadrat testing for % ground cover, submission broad findings	Where poor groundcover is found and cause assessed, the remedials should be implemented and followed up in the next audit
Annual soil compaction survey, submission broad findings and remedials	Where poor soil structure is found and cause assessed, the remedials should be implemented and followed up in the next audit
Annual wet weather survey, submission broad findings and remedials	Where runoff is found and cause assessed, the remedials should be implemented and followed up in the next audit
Annual fertiliser spreader and irrigation testing and calibration	Spreaders and irrigators not performing should be recalibrated

Audit measures	Action in the case of non-compliance
Auditing that must be done internally	
Self certification for application of fertiliser according to code of practice	Any failures in observing the code of practice for applying fertiliser should be rectified and followed up in the next audit
Submission of proof of 'approved handler' status	Inappropriate handling of chemicals should cease until an approved handler is in place

5. Site-specific mitigation strategies

5.1 Soil Management

Minimise soil erosion through increasing soil organic matter and groundcover

Background

Soil erosion involves the upheaval and transport of soil from the system mainly as a result of runoff where water picks up soil particles and transports them to a watercourse, or wind, which picks up particles and deposits them elsewhere. Soil erosion is a serious problem because it removes the topsoil, which is the most productive area in the soil profile.

Soil erosion is the major pathway for sediment and P pollution of surface waters (Humphrey *et al*., undated). Inherent characteristics influencing risk and impact of soil erosion such as soil type, slope, and location cannot be altered, however management characteristics impacting on soil physical properties can be, and these will affect runoff risk.

Runoff and erosion from pastures increases markedly when ground cover falls below 70%. During intense rainfall or where runoff occurs in bare soil, nutrients attached to soil or organic matter can be lost via erosion. When ground cover is > 70% runoff is usually more isolated and can re infiltrate. When <70% the bare patches connect and the erosivity of runoff increases (Lang and McDonald, 2005). The importance of ground cover increases as rainfall intensity or amount increases (Murphy *et al.,* 2004, cited in Melland *et al*., 2007) or where wind erosion is an issue (Brown and Harris, 2005).

Measure

Measures that increase ground cover will help reduce the potential for erosion, as will techniques that enhance soil structure and aggregate stability will help reduce the potential for erosion. These include the regular use of soil improving crops such as forages, the building up of soil organic matter, and avoiding excessive tillage (use of minimum tillage). Excessive tillage leads to break down of soil aggregates and loss of organic matter and make it more prone to compaction and erosion.

Measures that decrease soil compaction at the surface and subsoil compaction and increase infiltration will reduce also soil erosion, however these are dealt with under a separate section on soil compaction.

The establishment of windbreaks can reduce wind erosion by reducing wind velocities over fields.

Efficacy of measure

Cuttle *et al*. (2006) reported estimated reductions in P loss of 1.11 kg/ha in the affected areas for the measure, maintenance or increase of soil organic matter levels. Increase of soil organic matter through manure or organic matter incorporation were designated as potentially effective measures to reduce diffuse pollution to rivers (Hilton *et al*., 2003). Building soil organic matter is a general farm management option in the UK DEFRA Soil Protection Review, designed to meet EU Common Agricultural Policy requirements.

Research in Poland has shown that soil loss intensity as a result of erosion is directly proportional to the number of days that the soil remained uncovered (Klima, 2002). Also, increasing the number of days without plant cover by one day gave rise to soil loss of 0.168 kg soil per hectare per year. The corresponding P loss will increase with higher Olsen P concentrations in the soil.

Where ground cover falls below 70 %m runoff and erosion by water and wind increase dramatically. Basher (1996) found average soil losses from the Upper Waitaki Basin were 25 mm over a 40 year period. Areas with full vegetation showed no signs of loss.

Using vegetation to protect soils is recognised as best farming practices for soil management in Best Farming Practice: Profit for change (Information sheets IS 2.4) (West Country Rivers Trust, undated).

Nil effect was reported by Cuttle *et al*., 2006, for decreasing soil compaction, however in severely compacted areas the improved soil physical condition is likely to improve plant growth and increase N uptake, thereby reducing losses. Building up organic matter will provide additional capacity in the soil for N storage, however with greater organic matter contents, there will be more mineralisation.

Lemunyon (undated) reports that conservation tillage and residue management can reduce soil erosion by between 30 and 90%, and a study by Hazel *et al*. (2008) using a strategy of reduced tillage (in comparison with conventional tillage) decreased total suspended solids leaving tobacco fields by 82%. As soil erosion is the major pathway for phosphorus pollution of surface waters (Humphrey *et al*., undated), this option will also reduce P losses. Cuttle *et al*. (2006) reported estimated decreases in TP losses by 1.12 kg/ha/year by adopting minimum tillage. The reductions are only applicable on the area where the measure is implemented. The measure is only suitable where there are no soil structural issues, or where soils are not prone to capping. Dampney *et al* (2002) reported positive results for adopting minimum tillage for N, P and sediment reductions in a Europe wide review of DWPA mitigations.

Other impacts of mitigation measure

The use of surface mulches and reduced soil disturbance will increase the moisture retention capacity of the soil by up to 30% (Lemunyon, undated).

Beare and Tregurtha (2004) found that soil aggregate stability, and therefore a soil’s ability to withstand erosive forces, decreased with increased tillage.

Examples of monitoring

Soil compaction should be monitored annually between autumn and spring, and can be assessed by using a soil penetrometer. Areas of subsurface compaction should be investigated further by digging a soil pit. Capping and surface compaction can also be identified visually. Areas of severe compaction, or where a receiving environment is at risk, should be addressed in the season following assessment. Where runoff and erosion has occurred the causes should be investigated and suitable mitigations implemented.

Ground cover as a % can be visually assessed on a field by field basis or using a quadrat, these should be supported by dated photographic evidence.

Windbreaks, where wind erosion has been a problem in the past should be identified on a map and along with the direction of the prevailing wind, locations of new or maintained windbreaks should be put on. These should be supported by dated photographic evidence.

Soil organic matter can be assessed at the same time as regular soil testing and can be monitored this way.

5.1.1 Reducing soil compaction

Background

Compaction from livestock and trafficking of machinery can lead to surface and deeper compaction of the soil. Soil aggregates are destroyed and as they settle or are compressed into the spaces between other aggregates they form a 'massive' surface structure with greatly reduced porosity. This compacted layer reduces the infiltration capacity of the soil and increases likelihood of runoff. This runoff may carry high concentrations of contaminants such as manure and fertiliser as these will not have been incorporated. Plant growth is also depressed in compacted conditions either physically or through poor soil aeration and this compromised crop growth reduces N uptake. In addition, compacted soils can depress plant root and shoot growth and can lead to the formation of anaerobic soil conditions.

Measure

Reducing compaction in soils to reduce surface runoff and increase infiltration by spiking, slitting, subsoiling, ploughing or lifting.

Efficacy of measure

Cuttle *et al*. (2006) reported estimated decreases in P losses by 1.11 kg TP/ha/year in the affected area through removing compaction. Dampney *et al*. (2002) reported positive results for P and very positive for sediment reductions in a Europe wide review of DWPA mitigations. Decreased overland losses are also likely to reduce ammonium-N delivery to watercourse (Haygarth *et al*., undated).

Other impacts of mitigation measure

Removing compaction is reported to have a net nil effect on nitrate leaching (Cuttle *et al*., 2006). Improved soil aeration will lead to increased root growth and deeper penetration, as they are not limited by anaerobic conditions. This will increase of uptake of nutrients including N, leaving less nitrate vulnerable to leaching, but increased infiltration allows more water to move down through the profile and will take available N with it. Under compacted conditions, N uptake will be lower but as there is less movement through the profile there will be less leaching, however, denitrification will be favoured under compacted conditions.

Examples of monitoring

Soil compaction should be monitored annually between autumn and spring, and can be assessed by using a soil penetrometer. Areas of subsurface compaction should be investigated further by digging a soil pit. Capping and surface compaction can also be identified visually. Areas of severe compaction, or where a receiving environment is at risk, should be addressed in the season following assessment.

Figure 3 Surface soil compaction

Figure 4 Soil erosion from compacted field

Figure 5 Runoff into watercourse from soil erosion in Figure 3

5.1.2 Catch cropping

Background

Catch cropping is the establishment of a green cover over the winter period to protect the soil surface and capture available soil nitrate. Cover or catch crops are planted or volunteered vegetation established prior to or following a harvested crop predominantly for seasonal protection and nutrient recovery (Lemunyon, undated; Hirschi *et al*., 1997).

The autumn and winter and early spring periods are the most vulnerable periods for N and P losses, especially if the ground is left bare. Autumn sown crops play an important role in decreasing the movement of nutrients to water by converting nutrient into biomass and minimizing soil erosion (FLRC, undated) through the uptake of nitrate that would otherwise be vulnerable to leaching and the stabilisation of the soil surface by the vegetative cover.

Autumn conditions allow mineralisation beyond the date many crops are harvested, resulting in an excess of nitrate in the soil. Higher rainfall than evapo-transpiration fills the soil profile and leads to the onset of drainage, and this drainage water leaches the excess nitrate below the root zone. McLeneghan *et al*. (1996) reported that more that 50% of the leached N is transported in the first 15-50 mm of drainage water. Therefore the earlier the over crops are established, the greater likelihood of uptake.

The exposed soil surface is vulnerable to damage by raindrop action and may already have some structural damage from harvesting. The raindrops break up soil aggregates and the resulting soil particles settle out in the pore spaces and create a structurally ‘massive’ cap, see Figure 3. The cap has greatly reduced infiltration capacity and therefore increases surface runoff and consequently increases the risk of P and ammonium-N pollution in watercourses.

Raindrop impact also detaches particle-associated P from the topsoil, making it vulnerable for runoff to transport it to a surface water body.

Cover crops are best adapted to warmer areas with abundant precipitation, however provided they are drilled early enough to form a cover, may be useful in the Upper Waitaki.

Measure

Establishment of cover or catch crops instead for leaving soil bare over winter period.

Efficacy of measure

Cover crops are suitable for any system where they can develop sufficient canopy and root system before the cold or dry weather restricts further growth. Good plant development is essential for nutrient uptake (Lemunyon, undated) and the longer the plant grows, the more nutrients it will secure.

Fowler *et al*. (2004) found that cover crops reduced N leaching in an organic system by 50%. The amounts of N taken up were 100, 162 and 126 kg N/ha for oats, lupin and oat lupin mix respectively. Mcleneghan *et al*. (1996) found that 2.5-13 kg N leached under cover crops compared with 33 kg/ha under fallow. Ryecorn and ryegrass clearly improved the quality of water leaving the root zone compared to fallow, reducing the amounts leached by 92 and 90% respectively. Their aggressive growth also reduced the volume of leachate. Cuttle *et al*. (2006) reported estimated reductions in N loss of 15-45 kg/ha in the year of establishment, 15 kg for low fertility situations and sandy soils and 45 kg for high fertility situations with frequent organic manure applications, and estimated reductions in P loss of 1.12 kg P/ha for cover crops applied to spring-grown roots and vegetables.

Dampney *et al*. (2002) reported very positive results for N, P and positive results for sediment reductions in a Europe wide review of DWPA mitigations.

Other impacts of mitigation measure

If the cover crop is established in poor conditions this can cause compaction and lead to an increase in P losses and reduced benefit for N as plant uptake of N will be minimal. A minimum cover of 50% canopy cover is required to produce desired protection and scavenging effects (Lemunyon, undated). The structural damage will need to be removed otherwise it will impact on the growth of the following crop as well.

Cover crops are best adapted to areas with good precipitation as water use by cover crops can adversely impact yields of subsequent crops in semiarid areas (Dabney *et al*, 2001), therefore destroying the cover crop early enough is critical to maintaining soil moisture levels for the succeeding crop (Hirschi *et al*., 1997).

Dry matter yields and N uptake by organic ryegrass was reported to be significantly greater (434-689 kg DM/ha and 17-22 kg/ha) for green manure amended soils compared to after fallow (297 kg DM/ha and 9.4 kg N/ha) (Fowler *et al*., 2004).

Examples of monitoring

This strategy should be documented in field records. Yields and fertiliser usage on paddocks with cover crops should be monitored to estimate efficacy of cover crop in retaining nutrients.

5.1.3 No winter grazing of forage crops

Background

In southern regions of New Zealand large numbers of cattle and sheep winter graze paddocks sown in forage crops that allow pastures in the remainder of the farm to be rested (Houlebrook and McDowell, 2008). Also where winter pasture growth is poor, this same system is used; grazing off forage crops such as turnip or ryecorn (Trolove, 2008), see Figure 6. In winter grazed forage systems, grazing of the forage crop only takes place once or twice a year, however the grazing traditionally takes place with very high stocking densities and at a time when the soil moisture is close to or at field capacity (Drewry and Paton, 2005 cited in Houlebrook and McDowell, 2008). Therefore grazing of winter forage crops is considered to be a practice with high potential for contaminant loss via drainage or runoff (McDowell *et al*., 2007, cited in Houlbrook and McDowell, 2008).

Figure 6 Sheep break feeding paddock next to watercourse

Measure

No grazing of forage crops in situ.

Efficacy of measure

In preliminary results for their experiment, Houlbrook and McDowell (2008) found that animals were the main driver for nitrate loss from winter grazed pastures and that the mitigation of animal urine effects is likely to provide the greatest mitigation for N loss. No winter grazing of forage crops was modelled using lysimeter experiments to reduce winter nitrate loss by over 90 % in a simulation of cattle grazed over winter and 60 % for sheep. The application of the Nitrification inhibitor DCD, while reducing the nitrate leaching, did not mitigate all of the urine N loss, reducing losses under simulated cattle by 35 % and under sheep by 10 %.

Previous work has show that grazing of crops over winter with its associated treading damage can increase P losses by 250% and sediment losses by 25% compared with ungrazed (McDowell *et al*., 2003).

Treading damage caused by stock in poor winter conditions can have longer term soil structural consequences. Beare and Tregurtha (2004) found poorer soil structural condition and fewer earthworms under winter grazed crops and a reduction in subsequent crop yields.

Examples of monitoring

This measure is not easily independently monitored. Self certification can be used to indicate the change in practice. The full impact of this measure can be estimated using a nutrient budgeting model such as OVERSEER.

Figure 7 Over grazed winter forage crop

5.2 Effluent Management

Effluents and manures contain significant concentrations of plant nutrients (particularly N, P) and organic matter. Although their application to land has, in many cases, been shown to result in significant increases in plant yields and improvements in soil physical conditions and chemical fertility (Cameron *et al.*, 1997), without proper management of the effluent application (i.e. the right times, places and rates of application), subsequent losses of nutrients via leaching or runoff to waterways will occur.

The composition of effluent depends on many factors such as retention time in storage, animal diet, and effluent system used. However, the organic forms of P are less readily absorbed by soil and are more vulnerable to runoff and loss than fertiliser forms (Melland *et al.*, 2007).

Effluents should never be discharged to waterways or directly to ground in the form of a soak-away. Raw dairy effluent has on average a dry matter of 0.9% and a mean concentration of 269 mg/l of N and 69 mg/l P (Longhurst *et al.*, 2000). Collected effluents even when stored and partially treated, are still high in pollutants. Traditional '2-pond system' effluents contain 98 mg/l BOD, 106 mg/l Ammoniacal-N, 27 mg/l P 198 mg/l suspended solids and 7000000 mpm/100ml E coli. Even advanced pond system effluents, although better, still typically contain 43 mg/l BOC, 39 g/l Ammoniacal-N, 19 mg/l P, 87 mg/l suspended solids and 918 mpn/100 ml E coli. (Craggs *et al.*, 2004).

Collection and treatment of effluent from feed pads and yard areas can help reduce and control pollutant losses to the environment and through re-application to land, the nutrients are recycled (US EPA, undated). The aim of land application of effluents is to utilise the soil/plant system to absorb, filter and breakdown the components of the effluent applied thus minimising the risk of nutrients leaching or draining to waterbodies (e.g. Cameron *et al.*, 1997).

Individual aspects of effluent management are discussed below.

5.2.1 Minimise effluent produced

Background

Reducing the volume of dirty water produced will make more efficient use of the existing storage volumes and reduce likelihood of exceeding storage capacity and having to irrigate in poor conditions that runs the risk of leaching or runoff. Technologies that reduce the volumes of effluent produced at the milking shed, such as covered yards that do not require daily wash down are likely to greatly reduce the frequency of unwanted discharges of effluent to water (Mongahan *et al.*, 2007).

Measures

Effluent can be minimised by:

» Reducing dirty yard areas by roofing loading and feeding areas: This will not reduce the amount of dung and urine deposited; however it will reduce the amount of water needed for washing down and therefore the volume of effluent to be collected, stored and applied to land;

» Avoiding excessive use of water in washing down: This is an easy and cheap mitigation option. In addition to simply using less water in the yard wash-down, there are many types of yard scrapers which can be attached to the automatic yard gates, towed behind a motorbike, or manually pushed. These reduce the wash down water consumption dramatically. Environment Southland recommends a maximum use of 50 l/cow/day (Environment Southland, 2007);

- Separating clean and dirty water: Ensuring that clean water is either collected and reused or is diverted and not allowed to drain to effluent stores. Clean water includes roof water and water that falls on unstocked or 'clean' yards. Allowing 'clean' water into the effluent system unnecessarily increases the volume of effluent that will be collected, stored and applied to land. Clean and dirty water separation will include the use of gutters, down pipes, water collection facilities, surface runoff diversions in the yard and strategic roofing;
- Cover effluent stores: This will reduce odours and NH3 volatilization. Keeping an effluent store covered may allow anaerobic conditions to prevail thus resulting in denitrification (the reduction of nitrate N to gaseous forms of N). However, where denitrification is not complete, N2O can be released. N2O is a potent greenhouse gas; and,
- The use of robotic milkers, thereby reducing the parlour washings generated.

Efficacy of measure

Manure reduction methods such as scheduling cleaning of alleys and holding areas, scraping yards before hosing and sloping floors in parlours to facilitate scraping all have high pollution reduction potentials according to Wright (1996, cited in Sharpley *et al*., 2004).

Cuttle *et al*. (2006) reported estimated reductions in N and P loss of 0-1 kg/ha and 0.03 kg/ha respectively averaged over the whole farm.

The used of low or no P detergents and acid rinses was reported as having a high P reduction potential at relatively low cost around $400 NZ per annum (Sharpley *et al*., 2004).

Other impacts of mitigation measure

Reducing effluent produced, through reducing the risk of runoff, will have a positive impact (<25% reduction) on ammonium-N losses (Haygarth *et al*., undated) and also Faecal Indicator Organisms (FIO) transfer to watercourses.

Reducing the volume of effluent produced will use less water in the wash-down process; however, it will make the effluent more concentrated in nutrients and pathogenic organisms.

Examples of monitoring

This measure can be measured by documenting reductions in volume of effluent pumped or number of times effluent store is emptied per season. Photographic evidence can also be produced of infrastructure changes that have taken place to minimise dirty water produced. Sub-metering of water usage in the dairy and yard can be measured and the impact of water saving mechanisms calculated.

5.2.2 Effluent storage

Background

Agricultural activities, including milking on dairy farming systems, cannot be put on hold due to bad weather, and consequently, effluent is still produced even when conditions should prevent it from being applied. In the late autumn, winter and early spring months, soil moisture deficits are generally low if this effluent is applied to the land at this time, the risk of leaching and nutrient runoff increases. In instances where a rainfall event immediately follows effluent application, large incidental losses of P and N will occur as runoff.

Effluent storage facilities permit farmers to spread to land in the most suitable conditions and at a time when the nutrients are in demand by growing plants thus limiting the risk of runoff. The provision of effluent storage was reported as one of the improved management systems for reducing farm P losses in best management dairy catchments (Monghan *et al*., 2004).

The collection and storage receptacle should be carefully sized to the area draining to it, the stock numbers and housed period and a suitable rainfall event magnitude. The storage area should be lined to prevent direct discharges of nutrients to ground.

Measure

Increasing effluent storage capacity.

Efficacy of measure

This measure will only be effective if the effluent application is applied according to good agricultural practice, including appropriate timing and application rates (Sharpley *et al*., 2004). For example, putting in a storage system for effluent, then irrigating this effluent onto the land in autumn when the soil moisture content is high would not only be detrimental to the environment, but a waste of an investment on the farmer's part. Giasson *et al*. (2003, cited in Sharpley *et al*., 2004) found that installing storage with at least 3 months capacity was the most cost effective control of diffuse P losses. Pennsylvania State University (1992, cited in Sharpley *et al*., 2004) found that animal waste systems, including collecting storing and recycling of effluent to land could remove 90% TP, 80 0% TN, 60% of sediment and 85% of faecal coliforms in losses.

Other impacts of mitigation measure

Applying effluent at low risk times will reduce risk of ammonium-N losses in runoff (Cuttle *et al*., 2006; Haygarth *et al*., undated) and FIO transfer. However, the increased storage is likely to increase ammonia and methane emissions (Cuttle *et al*., 2006). Also, if the effluent is stored in such a way that promotes anaerobic conditions, denitrification processes may occur.

Examples of monitoring

This measure can be measured by documenting the effluent infrastructure and volume of storage. By calculating the impermeable area draining to the effluent store and mean rainfall, your storage duration can be calculated. Designing an effluent storage to a suitable design' storm event and always maintaining a freeboard will help reduce point source losses from effluent storage systems. The design, storage capacity and storage duration should be documented and submitted as a part of an annual audit. The impact of this measure can be modelled using a nutrient budgeting tool such as OVERSEER.

5.2.3 Effluent application

Whilst land treatment of effluent represents a large improvement in the loss of nutrients in comparison to direct discharge, there is room for improvement as currently 2-20% of N and P applied as effluent is lost (Houlbrook *et al*., 2004). In particular it is necessary to prevent direct discharge of partially treated effluent by taking into account soil physical properties such as infiltration capacity, and soil moisture deficit. Scheduling irrigations of effluent based on soil moisture deficit can reduce direct drainage losses to zero. Correctly operated land application systems will minimise the risk of nutrients leaching or draining into fresh water (Cameron *et al*., 1997).

Incorporation in to the soil profile by subsurface placement (injection) decreases the potential for P loss in runoff and also reduces ammonia volatilisation thus improving the N:P ratio for crop growth (Sharpley *et al*., 2004). Table 6 shows volatilisation losses during and application of liquid manure. Meuller *et al*. (1984, cited in Sharpley *et al*., 2004) showed that incorporation with a chisel plough reduced total P losses in runoff from a maize field 20-fold, however, compacted soils are commonly associated with maize post harvest with high runoff risk.

Table 6 Nitrogen volatilisation losses during land application of liquid manure[5]

Application method	% N lost
Broadcast	15-30
Broadcast with immediate cultivation	1-5
Injection	0-2
Drag-hose injection	0-2
Sprinkler irrigation	15-35

Application rate

Background

Research has shown that the rate at which effluent is applied has a significant impact on the extent of leaching. Concentrations of N and P in drainage and runoff increases as rates and concentration of effluent increases (Melland *et al*., 2007). Rates of effluent application up to 200 kg N and 44 kg P applied in small doses over the dry season did not markedly increase nutrient losses compared to no application as the pasture used the available nutrients (Jacobs and Ward, 2006). The low application rates increases the contact time in the soil and hence the soil has a greater capacity to 'treat' the effluent (Houlbrook *et al*., 2004).

Most regional councils have established an upper N loading limit of between 150 and 200 kg of farm effluent N/ha/yr, and many use this as a guideline, however adhering to this does not necessarily protect surface waters especially of the effluent irrigation exceeds the soil's infiltration capacity (Houlbrook *et al*., 2004).

Using N rates can result in excessive K concentrations in soil that decreases Mg and Ca availability and can induce metabolic disorders in cows, such as hypocalcaemia and hypomagnasaemia (Roach *et al*., 2001, cited in Houlbrook *et al*., 2004). Typical K concentrations in effluent are 80% of the N concentration. Increases in salinity and sodicity are also likely after repeated effluent applications and therefore need to be monitored (Melland *et al*., 2007).

Measure

Reduce effluent application rates and applying less than the soil moisture deficit.

[5] % N lost within 4 days of application (Hirschi *et al*., 1997)

Efficacy of measure

Very low application rates, between 0.9 and 4 mm of effluent are very effective at reducing overland and bypass flow (Toor *et al*., 2004; Monaghan *et al*., 2007), and only applying less than the soil moisture deficit in the pasture rooting zone will reduce nitrate leaching (Monaghan *et al*., 2007). Environment Southland has set an effluent irrigation rate maximum at 10 mm/hr and no more that 15 mm per application (Environment Southland, 2007). McDowell and Sharpley (2003) found that P loss in runoff after surface applications of low rates of dairy manure (<50kg/ha/yr P, DM unknown) were not significantly different to losses from untreated soil. McLeod *et al.* (1998) found that at an irrigation rate of less than or equal to 10 mm per hour, the irrigated water stayed in the topsoil of both a well drained and poorly drained soil. At the irrigation higher rates examined, preferential flow and leaching below the root zone occurred.

The irrigation of effluent according to ‘deferred irrigation ‘criteria and applying effluent at a low rate were recommended as best management practices for reducing phosphorus losses from agricultural land in the Manawatu catchment, reducing farm P losses by up to 1 kg/ha (Parfitt *et al*., 2007) and have been included in Horizons Regional Council’s developing policy ‘framework for managing non-point source and point source nutrient contributions to water quality’ (Roygard and McArthur, 2008).

Longhurst *et al.* (1999) found that the efficiency of N use decreases with increasing effluent rates, from 85% use at 75 kg N/year to 4% at 375 kg N/yr.

Application Timing

Background

There is clear evidence that autumn and early winter application of manures and effluents will result in more diffuse pollution compared to spring application (e.g. Beckwith *et al*., 1989; Dampney *et al*., 2002). There will also be an increased risk of surface runoff as land. Autumn is often the most vulnerable time for nitrate pollution and winter for P losses.

Avoiding autumn and early winter application of manures/effluents and fertilisers will avoid any risk of their loss to the environment via leaching or surface runoff. If these applications are avoided there will still be background nutrient losses, however these losses won’t be as significant as they could have been.

The risk of loss of nutrients in runoff or drainage decreases with time after application, due to sorption of manure P by soil and dilution of applied P by rainfall insufficient to cause runoff (Sharpley *et al*., 2004). Effluents, like fertiliser, that are applied when rain is forecast can lead to considerable nutrient losses (Preedy *et al*., 2001).

When effluent is applied to soil close to field capacity, up to 30% of the applied effluent may be lost directly through the drainage network (Houlbrook *et al*., 2004).

Measure

No autumn of winter application of effluents and manures.

Efficacy of measure

Dampney *et al*. (2002) reported very positive results for N reductions for closed periods in a Europe wide review of DWPA mitigation.

Other impacts of mitigation measure

Reduced autumn applications of manure and effluent are likely to have associated benefits for both P and FIO losses to watercourses.

Examples of monitoring

Where deferred irrigation is practiced, evidence of soil moisture deficits and consequent effluent applications should be documented. Effluent application rate of equipment should be tested and calibrated annually in the field by using catch cans. The impact of this measure can be modelled using a nutrient budgeting tool such as OVERSEER.

5.3 Stock Nutrient Management

5.3.1 Nitrification Inhibitors

Background

N excreted by animals and in particular urine, is the most important determinant of N loss in stocked systems (Monaghan *et al*., 2007). The quantities of N that are deposited greatly exceed the immediate plant requirements in the area of the urine patch (equivalent to up to 1000 kg/ha Di and Cameron, 2004), leaving the excess N subject to leaching losses (de Klein and Ledgard, 2001).

Urea in urine patches is transformed by the enzyme urease, into ammonia. Under favourable soil conditions this ammonia (NH_3) is fairly quickly converted into plant available ammonium (NH_4^+). The ammonium is converted into nitrite (NO_2^-) by another enzyme produced by the bacteria Nitrosomonas. Dicyandiamide (DCD), a nitrification inhibitor, slows down this particular stage of the process by temporarily making the Nitrosomonas bacteria's enzyme ineffective. It is not a bactericide and doesn't affect the other organisms responsible for the soil's bioactivity. Slowing this conversion allows time for ammonium uptake by plants, or for binding onto soil components (Anon, 2005). Ammonium is positively charged and not so readily leached from the soil profile as nitrate (NO_3^-). Nitrification of ammonium N to nitrate following urine deposition occurs within 14-60 days depending on soil moisture and temperature (e.g. Field *et al*., 1985, Shand *et al*., 2000 cited in Smith *et al*., 2005).

The two nitrification inhibitor products on the market in New Zealand are Balance's DCn (granular) and Ravensdown's Eco-N™ (liquid).

The effectiveness of DCD depends on soil type, temperature and moisture (Smith *et al*., 2005). Kelliher *et al*. (2007) found that temperature accounted for over 90% of the variance in DCD performance, with slower degradation rates in cooler soils and therefore recommended that DCD application should be most effective if restricted to May – September when soil temperature < 12 °C (for example, when soil temperature was 4, 8 and 12 °C, Kelliher *et al*'s function predicted DCD half-life was 109, 73 and 49 days, respectively).

The application of DCD appears most effective in pasture situations. Beckwith *et al*. (1998) found that incorporation of nitrification inhibitor (DCD) with manures applied in mid autumn did not significantly reduce the manure N leaching from fallow or winter planted cover crops.

The rate of DCD application influences the effectiveness and longevity of action in the soil and inhibition of nitrification tended to be greater at 30 kg/ha applications compared to 15 kg/ha (Smith *et al*., 2005). By 70 days, this difference is significant. Kelliher *et al*. (2007) report that a suitable DCD application rate was 10 kg/ha and two applications each year, one following grazing in autumn and another following grazing in late winter.

Measure

Apply a nitrification inhibitor according to the manufacturers instructions.

Efficacy of measure

Smith *et al*. (2005) found that all formulations of DCD tested (DCD-Urea granules, DCD in Zeolite granules and liquid DCD) applied in late spring, limited nitrification of ammonium N to nitrite in soil for more than 100 days. Di and Cameron (2004) found that autumn application DCD as eco-n™ on to a grazed pasture reduced N leaching by 74-76% under spray irrigation, compared to non-treated pasture. Research by Ballance and AgResearch showed that over 3 years, DCD reduced N leaching by an average of 42% in Southland, with optimum performance when DCD was applied within a week of grazing (Anon, 2007).

The mitigation of animal urine effects will provide the greatest mitigation of N losses under grazed winter conditions (Houlbrooke and McDowell, 2008) who found N leaching reduced by 36% from cattle urine under grazed winter forage crop with DCD application.

In a double bottom line analysis of improved N management systems, Monaghan *et al*, (2004) found that the use of nitrification inhibitors was a cost effective measure for reducing N leaching from dairy farms in all catchments.

Ledgard *et al.* (2007) reported the first research study in New Zealand to examine the impact of DCD administering directly to animals to inhibit nitrification in the urine patch. 86-99% of the DCD administered to sheep was voided in the same form and therefore effective in reducing nitrification. About 2% was voided in the dung, however within several days after infusion had ceased, urine-DCD concentration reduced back to background level. Current research is investigating the use of a slow release mechanism to sustain the release of DCD (Ledgard *et al*., 2007). This could have significant impacts on nitrate available or leaching in stocked systems.

Other impacts of mitigation measure

DCD application was found to limit the nitrate accumulation in herbage to safe levels for ingestion (Smith *et al*., 2005). Reduced leaching of nitrate also reduces leaching of positively charged elements such as magnesium, calcium and potassium which get carried in association with the negatively charged nitrate. Di and Cameron (2004) found DCD applications reduced Ca and Mg leaching by 38-50% and 21-42% respectively.

Nitrification inhibitors also reduce nitrous oxide emissions. Kelliher *et al*. (2007) in a report for MAF on nitrification inhibitors recommended that DCD application be considered to effect a 67% reduction in direct N_2O emissions.

There has been one field trial, conducted at Lincoln, quantifying the effects of repeated applications of DCD to grazed pasture. The recently published, peer-reviewed paper about this trial (Moir *et al*., 2007) focussed on herbage production. Based on seasonal measurements over four years, on average, application of DCD corresponded with a 21% increase of dry matter production on whole paddock and annual bases (Kelliher *et al*., 2007).

Retaining more nitrogen in the soil in a plant available form gives more opportunity for plants to uptake, therefore growing more pasture for the same inputs, representing increased nitrogen utilisation efficiency.

Examples of monitoring

This measure can be monitored through field records of application dates and rates. The impact of these inhibitors can be estimated using OVERSEER.

5.3.2 Urease inhibitors

Background

Urea is the predominant N fertiliser used in New Zealand (Blennerhassett *et al*., 2007) at 80%, and its use has increased dramatically in New Zealand in the recent past. Six-fold increases to 250,000 t N in the last decade were reported by the Parliamentary Commissioner for the Environment (2004). However the traditionally expected responses of 10-15 kg DM/kg N from urea is based on only utilising 35-50% of the applied fertiliser (Blennerhassett *et al*., 2007). Urea is rapidly hydrolysed to ammonium and then can either be volatilised, nitrified to nitrate, bound by soil or taken up by plants. By slowing down the process of hydrolysis through the addition of urease inhibitors, direct losses of fertiliser are reduced and more fertiliser N is available to the plant (Watson, 2000 cited in Blennerhassett *et al*., 2007). Urea can also be taken up by plants directly, although various research cited in Quin *et al*., (2006) suggest that the rate of urea uptake is considerably lower than for nitrate and ammonium.

Measure

Apply a urease inhibitor according to the manufacturers instructions.

Efficacy of measure

Agrotain sold under the brand name SustaiN in New Zealand is a urease inhibitor that slows this hydrolysis process down. Blennerhassett *et al*. (2007) reported on a number of trials on SustaiN, showing that N response rates in pasture were on average 42% higher than urea alone. Edmeades (2004) in a review of nitrification and urease inhibitors cites the only trial of urease inhibitor Agrotain at the time of publication, this suggested significant benefits of urease inhibitors on both leaching (53% reduction) and ammonia volatilisation (69%) based on a single application of 150 kg/ha of urea. However, applications of this size are relatively uncommon in New Zealand and further research is required to quantify leaching reductions at lower rates.

Ramakrishnan *et al*. (2006) examined the use of urease inhibitor combined with nitrification inhibitors. They found that the urease inhibitor (Agrotain) was effective at slowing the hydrolysis of urea to ammonium and the combination of the treatments significantly reduced soil nitrate concentrations at the Ashburton site from 157 mg/l in the urine to 65 and 47 mg/l for the lower and higher rates of application respectively. Zaman and Blennerhassett (2008) found that the use of urease inhibitor combined with nitrification inhibitors reduced N leaching by up to 60% from urine patches compared to urine alone. A rate of 1:7 of urease:nitrification inhibitors appears to offer the best combination of reduced N losses and increased pasture production (Zaman and Blennerhassett, 2008).

Zaman *et al*. (2005) also studied the use combination of nitrification and urease inhibitors on N losses and reported around a 10-fold reduction in nitrate leaching over 70 days in comparison to urea alone.

Examples of monitoring

This measure can be monitored through field records of application dates and rates. The impact of these inhibitors can be estimated using OVERSEER.

5.3.3 Reduced dietary N and P intakes for grazing animals

The primary way to reduce the amount of nutrients excreted per animal is to decrease the amount that is consumed and increase the efficiency of utilisation of the dietary nutrients. The goal of efficient and productive feeding is to provide the essential nutrients with minimal excess amounts (US EPA, undated).

Nitrogen

When supplements are brought onto the farm, selecting and balancing the right type of protein sources are important to meeting the amino acid needs of the animal and minimising the excreted nutrients. Ruminants can utilise forages (lower digestibility) as well as concentrates (generally higher digestibility) however, when they consume excess proteins an increased amount of N is excreted, an indicator of this can be measured - milk urea nitrogen (MUN).

Grazing animals can return in excess of 75% of ingested herbage N through dung and urine (Di and Cameron, 2004), therefore the N in diets should be restricted to animal requirement, the N:Carbohydrate in rumen diets should be optimised and the proportion of rumen degradable protein should be reduced. In rumens, an increase in digestibility of the ration will reduce the N excreted (US EPA, undated; Cuttle *et al*., 2006). Feeding rations with ideal proportions of amino acids required for protein synthesis will reduce the surplus amino acids contributing to N excretion (Cuttle *et al*., 2006).

Phosphorus

Addressing farm-gate imbalances of P is fundamental to reduce diffuse P loss because the long-term accumulation of P on farms ultimately drives diffuse transfers from soil to water (Sharpley *et al*., 2004). P is routinely added to mineral mixes, however normal levels of P in most typical ration ingredients exceed the animal's P requirement. Overfeeding P for reproductive performance has no scientific basis and research has shown that accurate formulations will optimise performance. However it is a challenge to accurately know the inputs as forage is highly variable in P content.

Adding phytase enzyme to feed can increase uptake and reduce P in manures in non-ruminants, however any impact on ruminants is not known, although SEPA (undated) indicate that their use may be beneficial.

Other dietary amendments can reduce the N in cattle excreta for example it has been found that the ingestion of certain tannins (quebracho tannins) reduces the N concentration of livestock urine (Leibig *et al*., 2008). Livestock that had ingested the tannin had approximately 34% less N in their urine than those that didn’t ingest the tannin.

Measure

Match dietary N and P levels to stock requirement.

Efficacy of measure

This measure will be most effective in systems with high concentrate or supplementary feed use. The highest impact for N will be on shallow or sandy soils and incidental losses. For P the effects will take longer to realise due to the buffering of the soil. Cuttle *et al*. (2006) reported estimated reductions in N and P loss of 2 kg/ha and 0.3 kg/ha respectively for a reduction in crude protein in diet from 18% to 14%.

Selecting the optimal levels of the right protein to more accurately match the animal requirement can decrease N excretion by up to 25% in intensive beef (NRSC, 2003) and 15-25% in dairy (NRCS, 2003). Recent research shows that a 20-30% decrease in excreted P is achievable by not adding supplemental P to beef diets (NRCS, 2003) and up to 30% in dairy diets (NRCS, 2003).

Dietary additives such as salt to increase urination, decrease concentrations in urine patches and increase spread may also be effective.

Other impacts of mitigation measure

Reducing the amount of N excreted reduces the potential for that N to be lost via run off or as gaseous losses. However, there may be increases in methane emissions associated with the change in diet (Cuttle *et al*., 2006).

Examples of monitoring

This measure can be monitored through documenting the nutrient concentrations of imported supplements and concentrates and comparing these to requirements of the herd. The reduction in the surplus can be monitored in dairy cows through MUN concentrations or through regular effluent testing.

5.3.4 Nil grazing

Background

Losses from grassland systems are greater in the presence of animals than from fertiliser alone (Monaghan *et al*., 2003 cited in Melland *et al*., 2007). N excreted by animals, particularly urine, is the most important determinant of N loss (Monaghan *et al*., 2007) in stocked systems. Grazing animals can return in excess of 75% of ingested herbage N through dung and urine (Di and Cameron, 2004). The quantities of N that are deposited greatly exceed the immediate plant requirements in the area of the urine patch at applications equivalent to 300 – 600 kg N/ha and even up to 1000 kg N/ha (Haynes and Williams, 1993; Di and Cameron, 2004), and leaves the excess N subject to leaching losses (de Klein and Ledgard, 2001). Up to 70% of pasture gets no excretal N at all due to patchy distribution of dung and urine (de Klein, 2001), and the remainder of the paddock receives more nutrients than it can use. An estimated 10-15% of excretal returns are deposited on non-productive land, such as tracks, around feed and water troughs and in gateways, and unless collected and redistributed, this is both an inefficiency in the reuse of nutrient on farm but also is a loss from the system with associated environmental consequences.

Studies have shown that the urine-N in the soil at the start of the drainage season is directly related to the time of year that urine was deposited (Cuttle and Bourne, 1993, cited in de Klein and Ledgard, 2001). With 5-13% remaining from spring and early summer deposition and 30-50% from late summer deposition. The greatest losses are from overwintering stock in part, due to deposition of much excretal N onto grazed land when corresponding uptake is low (Monaghan *et al*., 2004).

Stocking when soil is wet or overstocking will increase the soil damage, decrease infiltration rates and increase volume and sediment load in runoff. Reduced pasture growth will decrease plant uptake of nutrients and potentially leave more N at risk of leaching. If overgrazing occurs and vegetated cover is depleted, the erosive force of rainfall can lead to increases in run off. Treading damage caused by high stocking rates can reduce pasture production by up to 38% with short-term reductions (up to five months) in dry matter production of up to 81% after intensive treading (Cluzeau *et al*., 1992 cited in de Klein 2001).

Monaghan *et al*, (2005) found that there were few lasting effects of increased stocking rate on soil properties and the periodic deterioration between early spring and early summer, probably from grazing in wet conditions in spring was ameliorated by natural processes over the summer. Amelioration will only occur if soil damage is not too severe and even with no permanent damage, overland losses in early spring and summer would deliver sediment and nutrients to the watercourse at a very sensitive time of the year.

Nil grazing systems reduce N leaching losses through zero 'patchy over application of N' in dung and urine patches, zero transfer losses where excretal N is deposited on non-productive areas of the farm, and increased pasture production and therefore N uptake, from lack of treading damage and probably the increase % utilisation of the nutrients in the effluent and manure through more even distribution.

Measure

Nil grazing systems where livestock are housed all year round. Pasture is grown on a cut and carry basis, conserved and fed back on the yard. Manure and effluent is collected, stored and spread back on to the land.

Restricted grazing systems where livestock spend part of the year grazed as normal and part withheld from grazing. Similar mitigation measures included are reduced stocking when wet and restricted grazing in winter (3 hours grazing per day).

Efficacy of measure – nil grazing

De Klein and Ledgard (2001) reported reductions of 55-65% of N leaching losses through nil grazing systems compared to conventional grazing systems. On the basis of the average annual NZ farm leaching losses of 30 kg/ha, nil grazing would have leaching losses of 10.5 – 13.5 kg/ha/yr. De Klein (2001) found a 20% pasture response under nil grazing compared to conventional grazing.

Efficacy of measure – restricted grazing

De Klein and Ledgard (2001) reported reductions of 35-50% of N leaching losses through restricted grazing systems compared to conventional grazing systems. On the basis of the average annual NZ farm leaching losses of 30 kg/ha, restricted grazing would have leaching losses of 15-21 kg/ha/yr. Cuttle *et al*. (2006) reported UK estimates of reduced N losses by 8-16 kg N/ha for reduced grazing day (daytime only in second half of season) or reduced grazing season (reduced by one month). Titchen *et al*. (1989, cited in Cuttle *et al*., 2006) found that more restrictive grazing '(nil grazing after mid summer) reduced soil mineral N contents in Autumn from 34 kg N/ha to 9 kg N/ha compared to conventional grazing with no restriction.

Restricted grazing in winter (3 hours/day) reduced total phosphorus (TP) losses by 75%, through reduction of particulate P loss from 0.4 to 0.1 kg TP/ha (McDowell *et al*., 2005).

A study by Betteridge *et al.* (2007) showed that in grazing systems, removing cattle between May and August reduced N-leaching from 13 to 5 kg nitrate-N/ha/yr and could be achieved by strategic selling and buying, grazing-off, or using feed-pad systems.

Reducing stocking rate when soils are wet is estimated to decrease losses of P by 0.2 kg/ha (Cuttle *et al*., 2006) and some impact (<25%) on reducing ammonium-N losses (Haygarth *et al*., undated).

Other impacts of mitigation measure

De Klein and Ledgard (2001) found that nil grazing lead to significantly increased gaseous N losses (5-fold) in comparison to conventional grazing. Between 9 and 11% of excretal N is lost through NH_3 volatilisation from housed stock and a further 2-15% (NZ average 6%), from manure storage, and around 6% through NH_4 losses from surface application of effluent (de Klein and Ledgard, 2001, Misselbrook *et al*., 2000 cited in de Klein and Ledgard, 2001).

Soil damage from treading and pugging will be eliminated due to no stock grazing, however, compaction from farm machinery may increase if soils are trafficked in wet conditions.

The even return of nutrients to the soil through manure and effluent spreading could reduce clover that thrives through out-competing grasses in low nutrient conditions (between urine and dung patches). Clover pressure can be maintained by lowering cutting height and increasing cutting frequency, thus restricting regrowth to 1200 kg DM/ha (de Klein, 2001).

5.3.5 Reduced stocking rate

Background

This mitigation option simply involves an overall reduction of stock numbers on a farm. N excreted by animals, particularly urine, is the most important determinant of N loss (Monaghan *et al*., 2007) in stocked systems in New Zealand. The availability of nutrients for movement from pasture in runoff, drainage and gaseous emissions generally increases with stocking rate (Melland *et al*., 2007), although different stock types have different propensities to cause diffuse pollution. To support higher stocking rates, usually higher nutrient inputs are required (Di and Cameron, 2002) and as grazing animals can return in excess of 75% of ingested herbage N through dung and urine (Di and Cameron, 2004), the quantities of N that are deposited greatly exceed the immediate plant requirements. In the area of a urine patch (equivalent of 300 – 600 kg N/ha even up to 1000 kg N/ha (Haynes and Williams, 1993; Monaghan *et al*., 1997; Di and Cameron, 2004), is deposited, leaving the excess N subject to leaching losses (de Klein and Ledgard, 2001).

Treading damage caused by high stocking rates can reduce pasture production by up to 38% with short-term reductions (up to five months) in dry matter production of up to 81% after intensive treading (Cluzeau *et al*., 1992 cited in de Klein 2001). Stocking when soil is wet will increase the soil damage, decrease infiltration rates and increase volume and sediment load in runoff. Reduced pasture growth will decrease plant uptake of nutrients and potentially leave more N at risk of leaching. If overgrazing occurs and vegetated cover is depleted, the erosive force of rainfall can lead to increases in run off (FLRC, undated).

Reducing overall stocking numbers will reduce N leaching primarily through a reduction in the excretal N returns through dung and urine patches, reducing the total amount of N available to be leached. Decreased stocking rates will also result in less treading damage to soils. Less stock will reduce incidence of overlapping urine patches.

Measure

Overall reduction of stock numbers on farm.

Efficacy of measure

This measure would reduce most forms of DWPA from livestock farms. Ledgard and Meneer (2005) reported that with a stocking rate of 2 cows/ha producing 600 kg Milk Solids/ha with no additional fertiliser use, the reduction of stock numbers (by 20%) would result in about a 50% reduction in N leaching (from 49 kg N/ha to 25 kg N/ha). Huging *et al*. (1995 cited in Vinten *et al*., 2005) found 27 kg/ha N lost from extensively grazed pastures in Germany compared with 87 kg N/ha from intensively grazed sites. Cuttle *et al*. (2006) reported estimated decreases in N leaching of 10 –25 kg N/ha (from 30-50 kg N/ha) for a 20% reduction in dairy numbers and 3-5 kg N/ha for the same reduction in beef numbers. *Although as the UK system is more heavily fertilized, but the stock are housed for 6 months, the losses may be greater in New Zealand.* Schepers and Francis (1982) measured 40% less P loss from pastures when cattle were not grazing compared to when they were stocked, and heavily grazed pastures were found to lose twice as much P as lightly stocked (Heathwaite *et al*., 1990). Dampney *et al*. (2002) reported very positive results for N, P and sediment reductions in a Europe wide review of DWPA mitigations.

Other impacts of mitigation measure

In terms of gaseous losses, de-intensification will affect the whole system, not just one particular component. Fewer cows, less feed, less N cycling through pastures leads to lower methane emissions, and lower energy requirements for growing and handling extra feed. (Monaghan *et al*., 2004).

Examples of monitoring

The full impact of this measure can be estimated using a nutrient budgeting model such as OVERSEER.

5.4 Fertiliser Management

5.4.1 Reduction of P index (soil Olsen P)

Background

Phosphorus is tightly cycled through the plant-soil continuum, but in agricultural systems, P is constantly removed through crop and animal production and is replaced to avoid problems associated with P deficiency. Where P has been applied to agricultural soils at levels that are surplus to requirements, this will lead to accumulation of P in the topsoil. In general, the higher the amount of accumulated P in the soil, the higher the risk of P transfer to a waterway. This P loss may be in the dissolved reactive form or in particulate form.

Dougherty *et al*. 2004 found a linear relationship between extractable P and dissolved reactive phosphorus (DRP) in runoff for a series of different pasture conditions in New Zealand. This relationship is important as DRP is directly available to aquatic plants and therefore represents a greater short term risk (US EPA, undated). The concentration of P in runoff from well-fertilised pastures (>30 soil Olsen P) may exceed the 0.03 mg/l P, which is thought to be sufficient to stimulate algal growth (NZFMRA, 2004), especially on low phosphate retention soils. Generalised figures indicate that responsible use of the agronomic optimum should avoid excessive P loss in drainage waters (NZFMRA, 2004).

If the P content of a high index soil is allowed to decline, the amount of P lost that is both adsorbed to soil particles and in solution will be reduced.

The reduction in soil P is dependant on the amount of readily desorbable P in the soil. Solubilisation of organic and inorganic P results from the desorption of PO_4^{3-} from solid colloidal surfaces. The removal of inorganic P from soil solution (by plant uptake) results in P ions moving into solution, while on the other hand, an increase in inorganic P in solution (for example following fertiliser application) results in P movement onto the solid phase. If the addition of P fertiliser is stopped (or significantly reduced), then as plants take up the P in soil solution, eventually, the accumulated soil P will be desorbed and subsequently taken up by plants, thereby over time decreasing the overall soil P status (Brady, 1990).

Other important factors affecting P desorption from soil include ionic strength, pH and temperature. Waterlogging of the soil can also release P through the reduction of Fe^{3+} and the mobilization of clay-humic complexes (and associated P). Biological mechanisms are also known to be critically important in determining soil solution P concentrations, however; this factor is under-investigated with respect to P transfer and more research is needed to determine definitive impacts.

Measure

A reduction in the soil P status.

Efficacy of measure

Of the many methods available to minimise P loss to waterways from New Zealand pastures, the best approach is to ensure that soil Olsen P is not above the agronomic optimum (Monaghan *et al*., 2007).

The running down of elevated P reserves is a gradual process and the full benefits will only be achieved in the long term (>10 years). (Cuttle *et al*., 2002). The greatest effect will be seen where the soil P is elevated. Cuttle *et al*. (2006) reported estimated reductions in P loss of 0.07 kg/ha for arable and 0.09 kg/ha for grassland.

In an investigation into hydrological approaches to delineate critical source areas for runoff to management of phosphorus, Srinivasan and McDowell (2007) found that P index and determining 50 m either side of a watercourse as hydrologically active were the two approaches highlighted to investigate further**.**

Examples of monitoring

This measure is easily monitored using regular soil testing in representative areas of the farm. Trends in Olsen P can then be displayed for all blocks on the farm. The impact of this measure can be modelled using a nutrient budgeting tool such as OVERSEER.

5.4.2 Split N applications on leaching prone soils

Background

Splitting up spring N applications rather than one bulk N application will reduce the potential for N leaching. Plants cannot uptake all the N in the soil solution; rather, they only take up what they need. If there is more N in the soil solution than what the plants can physically take up, then upon rainfall or irrigation, it is likely that this N will be leached below the root zone – particularly in leaching prone soils. Splitting up the N applications will result in a smaller amount of N in the soil solution following fertiliser application; therefore it is likely the plants will take up most, if not all of this N, leaving none or very little available for leaching. When the N in soil solution falls back, another small N application will again boost plant growth, yet still minimise the potential for leaching.

Measure

Split applications of N to below 50 kg N/application.

Efficacy of measure

Split applications of N are a recommended practice for controlling nitrate leaching to groundwater. Eckard (2004) found that applying greater than 50 kg N/ha in any single application will exponentially increase the N losses in drainage.

In a study by Malhi *et al*. (2002) the effects of single versus split applications of N on dry matter yield and protein yield of bromegrass was compared. The single application gave a higher dry matter yield over the short term, while split applications produced a higher yield. Malhi *et al*. (2004) also found that splitting annual application tended to redistribute forage production in a longer portion of the growing season.

According to Monaghan *et al*. (2007), many New Zealand farmers use split applications of < 50 kg N/ha and few use in excess of 250 kg N, which would indicate a low risk of direct N leaching under these management conditions.

Other impacts of mitigation measure

Split N applications were also found to be effective at reducing N_2O emissions from cropping systems, particularly in years where there is significant rainfall during the period between planting and tilling (Malhi *et al*., 2004).

Examples of monitoring

This measure can be monitored through field application records. The impact of this measure can be modelled using a nutrient budgeting tool such as OVERSEER.

5.4.3 No fertiliser at high-risk times or in high risk places, including avoiding autumn applications

Background

Applying fertiliser at a time, or in a place when it is unlikely to be taken up by the plant is a waste of resources and increases the environmental risk associated with farming in that area, and applying N at rates greater then can be taken up by the pasture at that time will increase the risk of N loss in drainage (Whitehead 1995, cited in Melland *et al*., 2007).

Timing

Fertiliser applications applied in the runoff/drainage season can directly increase nutrient concentrations in the water. According to Melland *et al*. (2007), the effect of P fertilisation in the runoff/drainage season will be greater than the soil P index on runoff concentrations. However, the risk of fertiliser nutrient loss in runoff and drainage decreases with time after the application. In the few days after application most fertilisers are highly available to plants and susceptible to loss. Concentration of P in runoff decreases exponentially with time, halving every 4 days after application and the direct effects of P application are negligible after a few months (Melland *et al*., 2007; Monaghan *et al*., 2007).

Placement

Examples of high risk places are hill slopes or depressions/channels in the ground that provide a ‘pathway’ from the land to a waterway, water courses, riparian margins, bores and springs non-productive areas e.g. tracks. Other high-risk places include areas of free-draining soils that are at/near field capacity, or clayey soils that are dry and prone to cracking.

The determination of what are known as “critical source areas” is important in this context. Critical source areas are “hot spots” or identifiable areas within a catchment that are most vulnerable to nutrient loss via leaching, surface and subsurface flow. Critical source areas result from the co-location of source areas and transport processes, which together can cause significant nutrient loss from a catchment. While source areas can often be locally controlled at the farm scale, transport areas are controlled by landscape scale hydrologic processes, but can to some extent be managed at the farm scale. This mitigation measure will prevent nutrients being applied at times when nutrient loss is most prominent (when runoff / leaching / rapid bypass flow is likely to occur), and also prevent fertiliser being applied on vulnerable areas (e.g. beside a watercourse or in a channel leading to a watercourse), thus reducing the risk of transport.

Measure

No fertiliser application at high risk times or in high risk places.

Efficacy of measure

Applying P fertiliser within 2 weeks of runoff occurring will significantly affect P concentration in runoff (Sharpley and Menzel, 1987). Applying superphosphate in winter resulted in losses of 2.3 –6.7 kg of P applied at a rate of 50 kg/ha (Sharpley and Syers, 1997, cited in Monaghan *et al*., 2007).

N applications made in late autumn/winter, losses may be up to 1/3 of applied N (Ledgard *et al*., 1988, cited in Monaghan *et al*., 2007). Also, as less than 50% of applied N is taken up in the short term, (Whitehead, 1995; Melland *et al*., 2007) with the remainder being microbially bound or vulnerable to loss, a moderate risk remains for 2-3 weeks following N applications.

Cuttle *et al*. (2006) reported estimated reductions in N loss of 0-1 kg/ha for non-placement of fertiliser in high-risk areas and 0.2 kg/ha for avoidance of fertiliser at high-risk times, both averaged over the whole farm. In affected areas, avoidance of fertiliser at high-risk times could reduce N losses by 15 kg/ha/yr with an average N fertiliser application rate of 150 kg N (applied in both high risk and low risk times throughout year). Estimated P losses for the same measures were 0.23 kg/ha and 0.13 kg/ha for avoidance of high-risk places and high-risk times respectively.

Other impacts of mitigation measure

Avoidance of fertiliser in high-risk places and at high-risk times was reported to be effective in reducing ammonia-N losses (Haygarth *et al*., undated).

Examples of monitoring

A map showing high risk and no application areas of the farm should be in all cabs. Fertiliser application dates and rates should be recorded on field application sheets and can be submitted for auditing. The impact of this measure can be modelled in part using a nutrient budgeting tool such as OVERSEER.

5.4.4 Use of a slowly soluble fertiliser instead of soluble phosphate

Background

McDowell *et al*. (2003) found that dissolved reactive P in runoff remained elevated for 60 days after the application of soluble P fertilisers, such as superphosphate. This highlights the higher risk of incidental P losses when using soluble P fertilisers. Whereas with a less soluble P fertiliser, based on rock phosphate, P is only slowly released and is therefore less vulnerable to losses. The rate of dissolution is influenced by RPR properties (i.e., particle size, chemical make-up of the RPR used) and farm properties (i.e., soil pH, rainfall, drainage and exchangeable magnesium) (NZFRMA, 2007).

Measure

The use of a slowly soluble P fertiliser instead of soluble phosphate fertiliser.

Efficacy of measure

According to Monaghan *et al*. (2007), using RPR reduced P losses to 0.01 kg P /ha/yr, and McDowell *et al*. (2003) reported reductions of 0.43 to 0.31 kg TP and 0.13 to 0.05 kg Dissolved Reactive P. In this same paper, McDowell found that after a fourth application of P treatments, 5.4 mg/l TP was lost from the 376 kg superphosphate treatment while only 0.11 mg/l was lost from the RPR treatment. Although the RPR losses were still greater than the Zero P treatment, losing around 0.02 mg/l TP.

Direct losses of P from border dyke system (Eyre soils, Canterbury) using RPR were predicted to be <0.1 kg P/ha/yr, significantly lower than the average loss from soluble P fertilisers (Monaghan *et al*., 2004).

Other impacts of mitigation measure

It is recommended soil pH is below 6.0 and that there is at least 800mm annual rainfall for RPR to be most effective.

To avoid 'acute fluorosis' it is recommended that stock are not grazed on pastures which have received phosphate fertilisers for 21 days or until 25mm rain has fallen.

Examples of monitoring

The impact of this measure can be modelled using a nutrient budgeting tool such as OVERSEER.

5.4.5 Fertigation

Background

Fertigation is the technique of applying dissolved fertiliser to crops or grassland through an irrigation system. When combined with an efficient irrigation system, both nutrients and water can be manipulated and managed to obtain the maximum possible yield. It is the effectiveness of the application and the plant availability of the nutrients that delivers the results (PacificAg, 2008).

Often, solid fertiliser side dressings are timed to suit management constraints rather than the horticultural requirements of the crop. If fertiliser is spread a day before a heavy rainfall event, much of this fertiliser will be lost by runoff and leaching below the root zone. Continuous, small applications of soluble nutrients overcome these problems. Fertigation also results in the fertiliser being placed uniformly around the plant roots, allowing for rapid uptake by the plant.

Fertigation controllers are also needed which control irrigation and a fertiliser injection system. Modern fertigation controllers have a range of trigger sources to start irrigation including time-of-day programmed irrigations, solar integration with the ability to modify the irrigation rate depending on humidity, temperature and wind speed, for example, the solar integrator will automatically increase the frequency of irrigations in sunny, hot dry weather and reduce it in cool, damp weather (Autogrow Systems Ltd, 2008).

Measure

Application of fertilisers through the irrigation system.

Efficacy of measure

Fertigation is an efficient mode of application as although the N fertiliser is readily available, it is applied exactly when the crop needs it (Smith and Cassel, 1991, cited in Lal and Stewart, 1994). Nitrate leaching can be reduced by applying just enough fertiliser to meet yield goals (Lal and Stewart, 1994).

In a study that compared the effect of drip fertigation on leaching losses and the transformation of N in the soil with that of surface fertilization combined with flood irrigation, the fertigation decreased leaching loss of N from the soil and increased the mineral N in the soil.

Fertigation and irrigation based on soil moisture content was reported to result in nitrate leaching to below 1% of the applied N from Bermudagrass turf (Snyder *et al*., 1984, cited in Lal and Stewart, 1994).

A study by Schepers *et al*. (1995) looked at N leaching and fertigation possibilities with different irrigation systems. A spoon-feeding strategy, based on chlorophyll meter readings to schedule fertigation, saved 168 kg N ha^{-1} in the first year and 105 kg N ha^{-1} in the second year, with no reduction in crop yields.

Other impacts of mitigation measure

Fertigation not only helps to reduce nutrient leaching and runoff as a direct result of untimely solid fertiliser spreading, but with an efficient and accurate irrigation system, it increases plant yields using less fertiliser. It is important to note however, that fertiliser distribution is only as accurate as the irrigation water distribution.

Many fertilisers are corrosive therefore it is important to know the materials of all pump, mixing and injector components that are in contact with concentrated fertiliser solutions. Cast iron, aluminium, stainless steel and some forms of plastic are generally less subject to corrosion and failure (New and Fipps, 2008).

Ammonia solutions are not recommended for fertigation because (1) it is very volatile and (2) ammonia solutions tend to precipitate lime and magnesium salts, which are common in irrigation water. These precipitates form on the insides of irrigation pipelines and can clog nozzles (New and Fipps, 2008).

Examples of monitoring

The type of fertigation system adopted will determine the most appropriate monitoring points. These should be discussed with the designer of the system.

5.5 Irrigation Water Management

A well-designed and managed irrigation system will reduce waste of irrigation water, improve water use efficiency and reduce the total pollutant discharge from an irrigation system. Any irrigation management system should aim to minimise water loss to deep percolation and runoff, minimise any soil erosion, and reduce water loss to evaporation.

Due to inefficiencies in the delivery of irrigation water, (for example runoff, wind drift, evaporation and deep percolation losses) the amount of water needed for irrigation is greater than the consumptive use. The aim of irrigation best management practices are to minimise the irrigation of water over and above that needed for consumptive use.

There are a range of applicable management strategies that will mitigate the effect irrigation has on diffuse pollution.

5.5.1 Irrigation Scheduling

Background

The purpose of irrigation scheduling is to determine the exact amount of water to apply to a field, and the exact timing for application. The amount of water to be applied is determined by a range of criteria and different areas of irrigated land will need site-specific strategies.

The application of water to meet the soil's moisture deficit and no more will result in less drainage and therefore less leaching. Nitrate, while it remains in the soil profile will be able to be taken up by plants.

The most important criteria include a soil's antecedent moisture content and its water holding capacity, climate, evapotranspiration (ET) rate, average rainfall, and also the tolerance of crops to water stress.

Yields can be significantly reduced in crops that are less tolerant to water stress. Effective irrigation is only possible through regularly monitoring soil and crop development conditions in the field and forecasting future crop water needs. Several scheduling tools are available to assist farm managers in irrigation scheduling such as soil probes, moisture sensors, in-field weather stations, computerised soil water balance accounting profiles and private consultants (Wright, 2002).

Using the water balance approach is a common method of irrigation scheduling and is often expressed as being analogous to a chequebook balance. In the field, daily evapo-transpiration amounts are withdrawn from storage in the soil profile and any rainfall is added. Should the water balance calculations predict the soil water to fall below a certain level, irrigation will be required (Broner, 2005). Weather forecasts enable the prediction of ET rates and rainfall, allowing projection of soil water balance and subsequently indicating whether/how much irrigation is needed in the near future (Broner, 2005).

Measure

Use irrigation scheduling to determine irrigation application amounts.

Efficacy of measure

When used with fertigation, irrigation based on soil moisture content was reported to result in nitrate leaching to below 1% of the applied N from Bermudagrass turf (Snyder *et al*., 1984, cited in Lal and Stewart, 1994).

Irrigation scheduling that maintains soil water at or near field capacity has been found to significantly enhance the yield and N uptake of crops (Home *et al*, 2001). For example, under sprinkler irrigation, schedules were compared at 15, 30, 45 and 60% of the maximum allowable depletion of soil water (MAD). Crop yields under the schedule of 15% MAD were almost double (7420 kg ha^{-1}) those under 60% MAD (4508 kg ha^{-1}) (Home *et al*. 2001).

The increased uptake of N by the crop leaves less in the soil profile and therefore vulnerable to leaching.

Dechmi *et al*. (2003) reported a 12% decrease in crop yield due to deficit irrigation or large intervals between irrigations, thus again highlighting the importance of effective irrigation scheduling.

Examples of monitoring

Accurate measurement of the volume of irrigation water applied is very important to derive the maximum benefits from irrigation and management, as is uniform water distribution across the field. The irrigation equipment will require testing to accurately identify the application rate and to ensure that it uniformly spreads the water. Testing in-house is recommended annually and by an external auditor every 5 years.

5.5.2 Change to a more efficient irrigation system

Over time, the efficiency and technology associated with irrigation systems has increased markedly. This includes various equipment, automation tools, field evaluation techniques and models for design and analysis (Pereira, 1999). Farms that use older types of irrigation systems (i.e. border dyke) are likely to have higher water and nutrient losses than farms with more recent irrigation technology (i.e. centre pivot systems). For simplicity reasons, this section will only compare border dyke to centre pivot systems, however it should be noted that many other irrigation systems exist with a wide range of efficiencies (i.e. travelling guns, booms, k-lines,) and may be more applicable to some sites than others. Upgrading or changing an irrigation system completely could therefore result in significant increases in the efficiency of water use. The most important factors considered in selecting the most appropriate irrigation system include land slope, water intake rate of the soil, water tolerance of the crops, and wind.

Measure

Change to a more efficient irrigation system.

Efficacy of measure

Border dyke systems are an example of surface irrigation and work by flooding agricultural land. If water needs are overestimated this can result in significant volumes of 'tailgate' losses (losses from end of border dyke) along with sediment and nutrients being discharge to watercourses, as well as increased risks of nutrient leaching due to large volumes of water being applied instantaneously. The conveyance of water in a border dyke system is often uneven resulting in a lower quality crop yields as some areas are under irrigated and other areas over irrigated. The potential reductions in nutrient losses associated with border dyke irrigation are outlined in sections 5.5.3 to 5.5.5.

Centre pivot systems on the other hand are an example of sprinkler irrigation and consist of evenly spaced sprinklers mounted on wheeled towers, which move around a central point in a circular motion. The water flow through each nozzle is even and is designed to simulate light rainfall, which *when used effectively* results in even and efficient irrigation. However, a poorly designed or managed centre pivot can generate significant losses.

Examples of monitoring

The impact of this measure can be modelled using a nutrient budgeting tool such as OVERSEER.

5.5.3 Pond and Spray Border Dyke Systems

Background

The South Island of New Zealand has some very large areas of border dyke irrigated land and runoff water as a result of poor management of these border dyke systems has been identified as a substantial contributor to poor water quality measured in local water bodies (Houlbrooke *et al.*, 2008). With border dyke irrigation systems there is a high risk of over-watering paddocks whereby excess water either ponds at the end of the run or drains into surface water bodies. This water typically contains large concentrations of contaminants, namely phosphorus and faecal bacteria. Studies in South Canterbury have shown that this water loss (termed wipe-off losses in this paper[6]) is usually about 50% of the inflow irrigation volume.

The 'pond and spray' mitigation option captures all the wipe-off and ponding water on a farm scale and re-uses it via spray irrigation to either a previously poor performing border dyke area or to an area not previously watered.

Where tailgate water is collected, this opens the opportunity of using natural or constructed wetlands to treat or 'polish' the water, reducing contaminant loads, before leaving the farm.

Measure

Collect and reuse losses from border dyke systems.

[6] Elsewhere wipe-off losses are referred to as tailgate losses.

Efficacy of measure

The study by Houlbrooke *et al*. (2008) collected wipe-off water from 157 ha of border-dyke irrigated land and stored it in a large collection pond. A total volume of 285 thousand cubic metres over 5 irrigation rotations was collected, representing 36% of the irrigation water. Concentrations of total phosphorus and total nitrogen were 25 and 5 fold (respectively) greater than the water quality guidelines to prevent the stimulation of aquatic weed growth. By extrapolating mean nutrient losses to a seasonal basis, it is estimated that approximately 4 kg P and nearly 16 kg N were being lost on a per hectare basis from the border-dyke irrigation area.

An Australian study suggests that large water ponds, in the right place can be successful at capturing 98% of wipe-off water.

Other impacts of mitigation measure

In order for this management strategy to effectively reduce wipe-off losses to surface water bodies, the collection pond needs to be situated in a place that catches a maximum amount of the wipe-off water. This is generally the lowest point possible on the property, and would preferably make use of any existing drainage networks.

Examples of monitoring

This strategy should be documented using maps where border dyke and pond systems are marked and accompanied by dated photographic evidence. The dimensions of the pond and the subsequent spray irrigation should also be documented. The impact of this measure can be modelled in part using a nutrient budgeting tool such as OVERSEER.

5.5.4 Install wide laser levelled borders on already established border dyke systems

Background

This irrigation management option involves re-engineering the narrow borders of already existing border dyke systems to improve water use efficiency. Narrow borders in a border dyke system encourage water to travel at a faster velocity down the runs. Widening these runs will encourage the water velocity to decrease, and subsequently reduce the likelihood of water loss to drain or surface watercourse at the end of the run. In addition, if the fields under border dyke irrigation are not level, even by the slightest amount, the water conveyance over that field will be irregular, which will subsequently result in uneven crop growth. Laser levelling equipment is used to scrape a field flat before the planting of a crop. This ensures the even distribution of water along the border dyke runs.

Measure

Create wide, laser levelled borders on already established border dyke systems.

Efficacy of measure

The study by Houlbrooke *et al*. (2008) assessed the performance of these wide, laser-levelled borders with that of the typical narrower border dyke systems. After the first irrigation event of the study, the volume of wipe-off water measured from the widened, laser-levelled border was 200 m^3, which equated to less than 10% of the total irrigation water. This was much less than the wipe off water measured by Carey *et al*. (2004) from the narrower, conventional border dyke systems on the same property. A loss of 0.1 kg P ha^{-1} was calculated from the same irrigation event in the widened levelled borders, which was significantly less than the 0.5 kg P ha^{-1} per event measured by Carey *et al*. (2004).

Other impacts of mitigation measure

This management strategy would achieve the best results if coupled with other strategies such as restricting watering times and amounts on border dyke systems.

Examples of monitoring

This strategy should be documented using maps where border dyke systems are marked and accompanied by dated photographic evidence of levelling. The collection of tailgate water will indicate the efficacy of this measure by reductions in losses. Wetting profiles can be dug along the border dyke can be used to assess the uniformity of application.

5.5.5 Restrict Watering Times in Border-Dyke Systems

Background

This mitigation option involves the careful timing and pasture cover calibration to adjust watering times for each border dyke strip across the farm. The ultimate aim of this is to reduce the amount of irrigation water that leaves the farm and enters a watercourse.

The management of irrigation timing is site specific and based on the length of each run, slope characteristics, pasture cover, antecedent soil moisture.

Currently, most border dyke operators will have timers set to close the floodgates once the water is about 75% of the way down the run, therefore ensuring there is enough flow to bring the water to the end of the strip (Houlbrooke *et al*, 2008). However, even a small overestimation of the water requirement will result in the increase of tailgate losses.

Measure

Calculating site specific watering times for border dykes.

Efficacy of measure

In the study by Houlbrook *et al*. (2008), a site was established to minimise 'wipe-off' water by only managing irrigation times. Scheduled times were based on the length of each run, slope characteristics and pasture cover and wipe-off volumes were measured over five irrigation rotations. The cumulative loss of water to wipe-off was approximately 45 thousand cubic metres, which represents a loss of 10% of the border dyke irrigation at the farm scale. Compared to typical wipe-off volumes of between 25% and 32% of the total applied irrigation water, this is a significant decrease. The loss of nutrients equated to 0.9 kg P ha^{-1} and 1.4 kg N ha^{-1} of irrigated land when extrapolated to a whole season.

Examples of monitoring

The collection of tailgate water will indicate the efficacy of this measure by reductions in losses. Wetting profiles can be dug along the border dyke can be used to assess the uniformity of application.

5.6 Runoff Management

The driving force behind pollutant transport from land to water is water, as it provides the energy and the carrier for pollutant movement (McKergow *et al*., 2008). Attenuating this water can provide a permanent loss or temporary storage of nutrients, sediment or microbes during the transport process between where they are generated and where they impact on water quality. Attentuation tools can also provide an important buffering function between pastoral land use and receiving water bodies (McKergow *et al*., 2008).

5.6.1 Riparian buffer strips

Background

Buffer strips are highly relevant to all types of agricultural and horticultural systems (drystock, irrigated and cropping) and are most effective when used in combination with good in-field soil management.

Riparian buffer strips are areas of land that are immediately alongside water bodies and in close interaction with them. Buffer strips act as an interceptor of surface flow and shallow subsurface interflow. Riparian buffer strips also act as a gap between farming activities such as fertiliser spreading or spraying and the watercourse, helping to minimize direct contamination. The proximity of nearby watercourses is commonly used as an indicator of risk of nutrient delivery (Bundy and Ward Good, 2006), as the longer the flow path and greater time travelled, the more opportunity for reinfiltration or redistribution (Melland *et al*., 2007).

Measure

To establish an unfertilized vegetated strip either with grass or woodland alongside a watercourse or within a field along the contour.

N removal

Buffer strips intended for N reductions rely predominantly on denitrification to remove nitrate (Howard-Williams *et al*., 1986; Cooper 1990). They can be effective provided that there is sufficient residence time (FLRC, undated) and are consistently reported to reduce nitrate in subsurface flow (Muscott *et al*., 1993). The mechanism and effectiveness for N reduction by denitrification will be examined under creation of artificial wetlands. However, the rapid growth of plants in a buffer strip will remove N from the soil (Prosser *et al*., undated) and the retention and infiltration of surface flows will capture and attenuate ammonium in the runoff allowing it to be absorbed into the soil matrix. Deep-rooted buffer plants can also take up nitrate that has already leached below the crop-rooting zone (US EPA, undated).

P removal

The pollutant removal mechanism for P and sediment, is through infiltration, deposition, filtration, absorption and adsorption. Buffer strips slow down the rate of water movement and can trap nutrients (Chambers *et al*., cited in Melland *et al*., 2007). However, they will only do so if the overland flow is diffuse and less than approximately 1 cm in depth (Prosser *et al*., undated). Naturally occurring macropores in the soil under a buffer strip will increase infiltration and decrease runoff (Melland *et al*., 2007).

Buffer need to be maintained in different ways for N and P reductions. A combined N and P reduction system is possible with the implementation of double strips, with one section managed for nitrate removal, where waterlogged conditions occur (Hilton *et al*., undated), and another section managed to optimise attenuation and infiltration.

To optimise the long term value of buffer strips as nutrient filters, the strips should be used in conjunction with improved land management practices to reduce runoff further up the catchment or slope, and there should be periodic harvested of plant material to remove nutrients.

Where the land is steeper, the buffer strips should be proportionally wider to allow the velocity to slow down (Barling and Moore, 1994), and if more than 5% gradient a dense continuous ground cover of vegetation will need to be used in conjunction with trees along the riparian margin (Prosser *et al*., undated). Runoff should not be channelled onto buffer strips, but where possible, spread out along the entire length. Where the required width is not available, flow barriers, sediment traps, silt fences and other vegetative plantings such as swales should be used. The efficacy of the strip will be dictated by the volumes and speed of water. Buffer strips will only be effective where the flow is slow and sediment has the chance to settle out, and therefore in-field or up-catchment works to reduce overland flow should be implemented. If volumes are too great, no reasonable width of buffer strip will be effective.

Woodland areas (not strips) however, can intercept and slow down larger volumes of water and used in conjunction with buffer strips, encourage the deposition of sediments (and associated P) before the water reaches a surface water body.

Efficacy of measure

Cooper *et al*. (1992) using New Zealand data from Smith (1989) reported that a 10 m buffer strip of a dense grass sward was adequate for attenuation using CREAMS - a US physical diffuse pollution model. Haycock and Burt (1991) found that 5-8 m of riparian margin was sufficient for nitrate removal to occur. Mander *et al*. (1999) found that a wider strip (30-50 m) was necessary for 67-94 % N and 81-97% P removal. Smith (1989) reported that strips of 10-13 m were capable of removing 80% of sediments and 67% and 55% of dissolved N and P respectively. Withers *et al*. (1997) found strips could be effective in removing over 90 % of sediment bound P and Gharabaghi *et al*. (2002) found that the first 5 m were critical for sediment removal with nearly all particles >40 microns being captured. However, the only mechanism to remove the smaller sized sediment was infiltration. Buffer strips of 20 m with moderate to low flow rates were found to remove 90% of the sediments and in this way a significant proportion of the particulate P is also removed.

Cumulatively buffer strips can have a significant impact. In catchments draining to Lakes Rotorua and Rotoiti, 80% of the stream margins draining pastoral land were taken out of production and planted with trees and shrubs. This lead to a significant reduction in loads of suspended solids and P, but not N (Cooper *et al*., 1992). Riparian plantings have also been used as a strategy to manage elevated N and P levels in Lake Kapoai, Northland (Fertresearch, 2003).

Cuttle *et al*. (2006) reported estimated decreases in P losses at 1.63 kg P/ha of riparian buffer strip and 2.17 kg P/ha of infield buffer strip. Cuttle *et al*. (2006) estimated little direct reduction of N leaching, but some reductions, between 1-5 kg/ha for each ha of buffer strip, were gained by default of land being taken out of fertilized production and the possibility of denitrification at sites with appropriate soil conditions. Dampney *et al*. (2002) reported positive results for N, P and sediment reductions in a Europe wide review of DWPA mitigations.

It is critical that an open soil structure is maintained below buffer strips intended for P reduction. If these strips are used as 'green tracks' for farm machinery, this will drastically reduce their efficacy.

Buffers may become saturated with P therefore sediment and organic matter should be removed from the system to extend their effective life (Barling and Moore, 1994).

Ettema *et al* (1999) and Schoonover and Williard (2003) found that a buffer strip, if anaerobicity was maintained could remove almost all of the nitrate passing through, and a forested buffer of 10 m achieved 70% removal. The trees provide nutrient uptake to a greater depths and reduce nitrate in lateral subsurface flows.

Other impacts of mitigation measure

Buffer strips can reduce FIO by arresting the low of surface flow carrying contaminants. Under low flow conditions, Collins *et al*. (1994) found that buffer strips trapped >95% of faecal microbes.

There is a risk of remobilising deposited sediments and nutrients before they are adsorbed or taken up if high flow situations occur in quick succession.

Examples of monitoring

This strategy should be documented using maps where riparian margins and in-field strips are marked and accompanied by dated photographic evidence. Where previous water quality monitoring has taken place, the impact of the riparian margins should be measurable. The impact of this measure can also be modelled using a nutrient budgeting tool such as OVERSEER.

5.6.2 Restrict livestock access to watercourse through fencing and crossings

Background

Stock with direct access to watercourses can cause severe environmental damage. Trampling by stock can damage watercourse banks causing them to destabilise and erode. Access to watercourse for drinking destroys areas of the bank and soil becomes badly pugged acting as a source for sediment, excreted N and P and FIOs. Stock can also damage streambeds through treading, stream vegetations through grazing and will defecate and urinate directly into the watercourse.

As a formal step to improve the environmental performance of New Zealand's dairy farms, the dairy industry, in partnership with local regulatory and central government authorities developed the Clean Streams Accord. This initiative was a commitment of the industry's farmers to implement some key environmental goals. One of the five targets under this commitment is to ensure that cattle are excluded from 50% of streams, rivers and lakes by 2007 and from 90% of these waters by 2012 (Ministry for the Environment, 2003, cited in Beswell *et al.*, 2007). However, small streams (< 1m wide) and ephemeral streams are excluded from these targets and thus the environmental impact of implementation will be less significant.

The study by Beswell *et al*, (2007) investigated the factors influencing farmers' decisions to adopt stream fencing. The most common reason farmers' fence off stream areas is for stock management and safety.

Measure

Stop stock access to watercourses.

Efficacy of measure

Dampney *et al*. (2002) reported positive results for N and P reductions in a Europe wide review of DWPA mitigations. Cuttle *et al*. (2006) reported estimated reductions in N loss of 0-1 kg/ha over whole farm, but obviously all of that benefit will be in the riparian zone. Modelling for an analogous parameter reported reductions in P loss of 0.3 kg/ha, but these did not account for the cattle gathering at watering points, and so the reduction in loss could be many times this reported figure (Cuttle *et al*., 2006).

In a 3-year innovation project to protect Lake Kapoai, Northland, riparian fencing was one of the principal measures used (FertReseach, 2003).

Other impacts of mitigation measure

Riparian fencing and providing alternative drinking water sources will reduce water pollution risks from ammonium-N by 25-75% (Haygarth *et al*., undated), sediments and FIO contamination by reducing defecation and urination in or nearby the river onto compacted soils prone to runoff.

Examples of monitoring

This strategy should be documented using maps where lengths of restricted are marked and accompanied by dated photographic evidence. Where previous water quality monitoring has taken place, the impact of the restricted access should be measurable. The impact of this measure can be modelled using a nutrient budgeting tool such as OVERSEER.

Figure 8 Stock watering point

5.6.3 Farm wetlands

Background

Wetlands essentially provide a natural water treatment process by capturing and retaining water, slowing its velocity and thus allowing sediment to come out of suspension. Biological processes within wetlands such as nutrient uptake by plants will remove nutrients from the water. Treatment wetlands mimic water treatment processes of natural wetlands such as high primary productivity and low flow conditions. Research has shown that treatment wetlands can be reliable and practical to build and operate and more sustainable than conventional treatment technology (Hawkins, undated).

Constructed farm wetlands are an ecologically engineered system, often compromising of a series of one or more constructed ponds, which are intended to receive and treat 'lightly contaminated' water. 'Lightly contaminated' can be defined as a mixture consisting predominantly of rainwater and drainage from roofs or from yards used by livestock and/or machinery on an occasional basis. It may also be contaminated with soil, silt, excreta or other organic material. The term 'constructed farm wetlands' cover a number of different systems that are designed to contain shallow water planted with emergent wetland plants (SEPA, undated).

P removal

P removal by wetlands depends on sorption of dissolved P by wetland substrates, sedimentation of particulate P and assimilation by microbial and plant biomass (Nguyen, 2000). P removal by wetlands gradually declines after years or decades (McDowell, 2004a) as the soil becomes saturated with P and annual growth of plants slows as they mature. Therefore plant biomass should be harvested to remove P from the system and ensure plants are in a growth stage that requires rapid nutrient uptake, and thus prolonging the active life of the wetland.

N removal

Wetland areas and seeps that intercept drainage and runoff before the flow enters a watercourse have been identified as areas with conditions that are favourable for denitrification (after Woods *et al*., 2004).

Notes on wetland design

The key to designing a suitable wetland system is to know the influent water characteristics, timings of flow, catchment area, plants, permeability and soil types (Hawkins, undated). Low permeability soils are essential for an effective wetland, so where these soils are not found on site, alternative measures should be taken such as importing clay or using an artificial plastic liner (Carty *et al*., 2008).

The wetland function improves when the flow enters one end of an elongated area with a length:width ration of 4:1 to 10:1. Multiple cells may be needed. The depth of water will impact on treatment with 30 – 60 cm giving a 36 hour retention period compared to 0-10 cm allowing enough time for plants to take up nutrients. Wetlands must be designed to have sufficient detention volumes to store 'design storm runoff', as the first flush in an event will carry a large proportion of the pollutants. The design event may be a 1 in 5 year storm. The use of sediment traps before wetlands to remove large sediment particles and organic debris is recommended to optimise the function of the wetlands, and also sediment traps are easier to clean out.

If the influent water is high is organic C, N and solids, some pre-treatment will be necessary to prevent overloading the system with high BOD pollutants (Hawkins undated).

Sukias *et al*. (2006) suggests that wetlands need to be between 2 and 5% of the contributing catchment area to be effective, and that improved nitrate removal performance (and potentially a smaller size of treatment has been achieved by using wood chip filters or by additions of supplemental carbon (e.g. straw bales to constructed wetlands).

In the short term, the creation of wetlands can be effective in controlling sediment and P delivery to a watercourse. However, the wetland will eventually fill up and will become a source for sediment and P, therefore wetlands need to be managed to ensure their continued efficacy.

Measure

Use of farm wetlands to treat of polish lightly contaminated water.

Efficacy of measure

N removal

Chavan and Dennett (2008) reported that in a series of pilot scale treatment wetland systems, during increased influent nitrogen loading, they functioned as sinks for total N with removal efficiency increasing from about 45% to about 87%. Nguyen *et al*. (1999) found 54% removal of N in a 6-month period in a wetland at Whatawhata, and also noted the importance of contact time in the removal of nitrate – the longer the contact time, the greater the removal. Sukias *et al*. (2006) reported that constructed wetlands could deliver 40-50% reduction in nitrates and up to 70% using alternative filters such as wood-chip. Table 7 shows some data for wetland N removal from the US.

Table 7 Nitrogen loading rates and mass removal efficiencies for constructed wetlands[7]

Nitrogen	System	% Mass removal[1]
3 kg/ha/day	Rush/bulrush	94
	Cattails/bur-reed	94
8 kg/ha/day	Rush/bulrush	88
	Cattails/bur-reed	86
15 kg/ha/day	Rush/bulrush	85
	Cattails/bur-reed	81
25 kg/ha/day	Rush/bulrush	90
	Cattails/bur-reed	84

[1] % Mass removal = % mass removal of N (NH_3-N + NO_3-N) in the effluent with respect to the nutrient mass inflow.

Tanner *et al*., (2005) reported that treatment wetlands removed 55-61% nitrates in drainage water in an irrigated subsurface drained catchment in Northland.

Dampney *et al*. (2002) reported very positive results for N reductions for wetlands, although with a negative impact of farm profits in a Europe wide review of DWPA mitigation.

[7] Duplin Co., NC (Rice *et al*., 199-, cited in Hirschi, 1997)

P removal

Nguyen *et al*. (1999) found 27% removal of P in a 6-month period in a wetland at Whatawhata. Hawkins (undated) reported that average reductions in P from wetland systems from dairy, beef, pig and poultry sites was 42%, however increasing pre-treatment or wetland area increased this to over 57%. Cuttle *et al*. (2006) reported estimated reductions in P loss of 1.53 kg/ha in the areas draining to the wetland.

Sukias *et al.* (2006) reported that newly constructed wetlands can be a source of P and that P absorbent materials such as alum residuals or smelter slag are being investigated to counteract this release.

Other impacts of mitigation measure

The denitrification process, when incomplete, releases nitrous oxide (N_2O), which is a greenhouse gas so wetlands have the potential to contribute large amounts of N_2O to the atmosphere.

As well as wetland plants taking up nutrients, they may take up other environmental toxins such as heavy metals, which will prevent them entering the food chain to be eaten by cattle/sheep, and also from being transported to watercourses.

Farm constructed wetlands are not only efficient at water treatment, but they also provide landscape and biodiversity enhancements to the area (Carty *et al*., 2008; Hawkins, undated).

Examples of monitoring

This strategy should be documented using maps where natural and artificial wetland areas are marked and accompanied by dated photographic evidence. Where previous water quality monitoring has taken place, the impact of the wetland areas should be measurable. Where water is channelled through wetland systems, the efficacy of wetlands can also be measured by water quality monitoring at up and downstream sites. The impact of this measure can also be modelled using a nutrient budgeting tool such as OVERSEER.

5.6.4 Swales to reduce peak flow, remove pollutants and promote runoff infiltration

Background

Swales are grassed ditches or constructed waterways that are generally used for providing stormwater, road/track runoff or simply excess water, conveyance and filtration prior to downstream drainage waters or receiving waters. They are constructed on very shallow slopes and can meander with land contours. Often they are constructed with 'check dams' that hold small quantities of runoff and allow it to infiltrate into the soil. This pooled water will then infiltrate into the ground allowing for the natural breakdown of compounds in the soil (i.e. pesticide compounds) and also the attenuation and plant uptake of nutrients, namely nitrogen and phosphorus.

Although swales do not prevent soil erosion, they can prevent the loss of eroded soil from the catchment and they slow down the speed and volume of water reaching the watercourse during or directly after rainfall. To be most effective, they should be dry most of the year and only carry water during or after heavy rainfall (SEPA, undated).

Measure

Install swales to carry runoff water.

Efficacy of measure

Average pollutant removal efficiencies reported for the test swales vary from 14 to 99% for total suspended solids, chemical oxygen demand, total nitrogen, and total phosphorus (Yu, *et al*., 2001). The wide range of performance results indicates the importance of such design parameters as length, longitudinal slope, and the presence of check dams. Minimum design guidelines for use of swales as a BMP are suggested.

Experimental programmes by Deletic and Fletcher, (2005) reported reductions in sediment inflow concentrations by 61-86% in Scotland, and removals of 69, 46 and 56% of sediment, total phosphorus and total nitrogen respectively (Deletic and Fletcher, 2005). SEPA (undated) report 30 -80% removal of suspended solids and 15-45% removal of N and P using swales.

Figure 9 Swale with check dams to manage runoff

Other impacts of mitigation measure

During autumn and winter months, the use of swales could cause areas of localised N leaching. Therefore frequent harvesting of the plant matter and vigorous plant species will reduce the amount of nutrients available for loss in the soil.

Depending on the size of the swale and the amount of productive land able to be sacrificed, trees and shrubs can be planted which will take up more water (and nutrients) than grass alone, reducing the potential for nutrient leaching.

On farms with a high level of soil erosion, over relatively short periods of time swales may intercept large amounts of sediment. This sediment will build up and eventually reduce the efficacy of the swales. This may reach the point where excavation and re-vegetation is needed. In a situation like this the swales are best used in conjunction with sediment traps.

The performance of swales is greatly decreased with increased flow rates (Deletic and Fletcher, 2005).

As with many pathway mitigation measures, they should be used with good soil management to allow them to work effectively.

Examples of monitoring

This strategy should be documented using maps where swales are marked and accompanied by dated photographic evidence. Where previous water quality monitoring has taken place, the impact of the swales should be measurable.

5.6.5 Sediment traps and infiltration basins

Background

Sediment traps are designed to temporarily trap water in its transit from the landscape to a watercourse and slow its velocity enough so that any suspended sediment in the water is deposited before the water reaches the watercourse. An infiltration basin is a shallow surface impoundment where storm water run-off is stored until it gradually infiltrates through the soil.

Figure 10 Field sediment trap

Figure 11 Concrete sediment trap designed to be easily dug out with bucket digger for track runoff

The design of the sediment trap must slow down water movement sufficiently to allow suspended sediment to drop out of the water column. While this measure does not reduce soil loss from the productive areas of land, it will reduce sediment loss to the stream below.

Measure

Install sediment traps in a runoff zones.

Efficacy of measure

The efficacy of a sediment traps and infiltration basins will be determined by the amount of sediment it is trapping and how often the accumulated sediment needs to be removed. SEPA (undated) report 80 – 99% removal of suspended solids and a consequent high efficacy for removing particulate P – up to 80% removal.

The efficacy of these measures will be increased where used in conjunction with source management strategies.

Other impacts of mitigation measure

Sediment traps require regular maintenance (i.e. the removal of sediment) in order to function properly, and therefore the construction needs to aid ease of sediment removal.

Reducing sediment loss with sediment traps requires relatively detailed knowledge of soil loss fluxes, and the origins and transfer mechanisms associated with a particular area.

Both these measures have low ability to deal with dissolved nutrients.

Examples of monitoring

This strategy should be documented using maps where sediment traps are marked and accompanied by dated photographic evidence. Where previous water quality monitoring has taken place, the impact of the riparian margins should be measurable. In addition, the direct reductions in sediment load delivery can be estimated when cleaning out and maintaining the sediment traps.

5.6.6 Track and cattle track management

Background

Because of their compacted surface, farm roads and tracks can respond rapidly to storms, producing large volumes of erosive run-off, which is channelled along the road to a low discharge point, and in some cases close to a watercourse. Poor track maintenance coupled with accumulation of various farm waste materials (e.g. manure, urine, oil, soil material carried by tyres, fertiliser, etc.) can all result in contamination of the run-off.

Access tracks can act as both a source of eroded soil and also act as conduits for moving run-off from an eroded area to a water body. As the velocity of the runoff increases, erosion of the soil particles increases (Melland *et al*., 2007). In sloping areas, tracks are a major conduit for water and pollutants even in arable areas.

In both intensive systems such as dairy, but also for beef and sheep, there will be a disproportionate deposition of urine and dung on tracks and laneways (Melland *et al*., 2007). The risk of nutrient loss is increased as landscape features such as stock tracks quickly remove water from paddocks and increase water speed and energy compared with the tortuous flow routed through undrained landscape (Cox and Pitman, 2001). Soils become particularly prone to erosion and surface run-off when constant movement by livestock removes the vegetation cover, exposing bare earth. Run-off from these tracks could be a potential source of faecal bacteria, soil particles and nutrients. There is some evidence to suggest that livestock foot problems can be reduced through use of dedicated livestock tracks (possibly most beneficial for dairy cattle using regular tracks) (SEPA, undated).

When tracks are constructed, they can be constructed with a camber and drains to channel water to one side where it can be redirected. Where a track is already existing, cross drains or 'sleeping policemen' can be used to direct water in combination with grips at the side of the track to remove the water. Once the water is removed, it can either be allowed to drain away naturally in a field for example, or if needed can be used in combination with swales or sediment ponds. Livestock tracks are designed to reduce the amount of poaching and run-off associated with the movement regular movements of livestock around the farm. These tracks can be bark/wood chipping tracks or covered with hardcore or concrete for heavier/frequent use, or where machinery also uses the track.

Tracks are important conduits for sediment and by association P transport, especially on sloping land. Once water has built up on a track it is difficult to attenuate at the bottom of the hill or before the receiving environment – the flows and volumes are usually too great, therefore track management should start at the top of the slope and continue down.

Measure

Manage tracks and cattle tracks to ensure regular removal of water from track surface, discharges to a safe site.

Efficacy of measure

Where track are managed well for water management, they can prevent almost all of water and the associated sediment and P getting into the watercourse.

Examples of monitoring

This strategy should be documented using maps where tracks and cattle tracks are marked and accompanied by dated photographic evidence. Where previous water quality monitoring has taken place, the impact of the track management in hydrologically connected areas should be measurable.

Figure 12 Grips on side of track for water removal

Figure 13 Cross drains on track for water removal

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Appendix A

Identification of farm environment risks

Identification of farm environment risks

The exercise of identifying specific environmental risks on the farm can either be undertaken by an experienced independent consultant, or the farmer can complete and have it audited by an experienced independent consultant.

The purpose of the exercise is to identify areas of environmental risk on the farm and in the farm practice. Some risks pertain to inherent risks on the farm, i.e. the natural environment or conditions, and some risks arise from the way the farm is managed. Additionally, some risks may be answered immediately, such as 'Are soils left bare over winter?' whereas others such as ' are the compacted consolidated or capped soils?' a cursory investigation will be needed to answer.

The questions listed here are an illustration of the type of information that is needed to assess the environmental risks of the farm, and is not comprehensive. For all risks, the relevant locations should be marked on a farm map.

Soil

Below are some examples of questions to ascertain whether there are inherent or managed soil risks on your farm.

Some example questions on soil health	Notes/description
Previous incidence of soil erosion or wash?	
Are there compacted, consolidated or capped soils?	
Is the soil trafficked when wet?	
Are there slaking or highly erodible soils present on farm?	
If arable or fodder crops are grown, are measures taken to conserve or build soil organic matter on arable land?	
If these crops are grazed over winter, are remedial measures in place after winter grazing?	
Is stock over wintered outside?	
Other soil issues or incidences? Please describe.	
If you have answered yes to any red questions or no to any blue questions, then there are inherent or managed soils risks on the farm	

Cropping

Below are some examples of questions to ascertain whether there are inherent or managed cropping risks on your farm.

Some example questions on cropping	Notes/description
Is inversion tillage used?	
Are soils left bare over winter?	
If arable or fodder crops are grown, are measures taken to conserve or build soil organic matter on arable land?	
Are remedial measures in place after winter grazing?	
Is there a possibility of run off from winter grazed crops reaching a watercourse?	
Other cropping issues or incidences? Please describe.	
If you have answered yes to any red questions or no to any blue questions, then there are inherent or managed cropping risks on the farm.	

Effluent/Infrastructure

Below are some examples of questions to ascertain whether there are inherent or managed effluent risks on your farm.

Some example questions on effluent	Notes/description
Do you produce effluent?	
Do you have less than 4 weeks storage of effluent?	
Is your effluent storage facility fully sealed?	
Do you separate clean and dirty water in the yard?	
Do any direct discharges occur off the yard?	
Do you spread effluent by a travelling irrigator? If not, how?	
What rate do you apply effluent at?	
What depth of effluent do you typically apply?	
Do you use soil moisture deficits to decide on application depth?	

Some example questions on effluent	Notes/description
How do you determine application depth?	
Do you apply more than 150 kg N/ha/yr of effluent N?	
If silage is made on farm, is effluent collected and spread to land?	
Are there any direct discharges from silage pit?	
Other effluent issues or incidences? Please describe.	
If you have answered yes to any red questions or no to any blue questions, then there are inherent or managed effluent risks on the farm.	

Fertiliser

Below are some examples of questions to ascertain whether there are inherent or managed fertiliser risks on your farm.

Some guideline questions for fertiliser	Notes/description
Do you apply more than 50 kg N per application?	
Do you apply N fertiliser during late autumn and winter?	
Do you apply P fertiliser within 3 weeks of irrigation?	
Do you regularly soil test?	
Do you have Olsen P levels over 30?	
Are fertilisers ever applied within 20 m of a watercourse or borehole?	
Are fertiliser spreaders calibrated regularly?	
Are spreaders calibrated to each fertiliser type?	
Are there 'no-fertiliser' areas on farm?	
Other fertiliser issues or incidences? Please describe.	
If you have answered yes to any red questions or no to any blue questions, then there are inherent or managed fertiliser risks on the farm.	

Water

Below are some examples of questions to ascertain whether there are inherent or managed water risks on your farm.

Some guideline questions for water	Notes/description
Is your water metered?	
Do you compare annual water usages?	
Do you collect rainwater?	
Do your stock drink from watercourses?	
Do you sample other than metered water to check for suitability for use?	
Do you use irrigation scheduling?	
Do you calculate soil moisture deficit?	
Do you use surface irrigation (border dyke, wild flood).	
Do you collect tailgate losses?	
Are these tailgate losses discharged to a watercourse.	
Are your borders laser levelled?	
If you have spray irrigation, do you practice fertigation?	
Is clean water yards collected separately and discharged or used?	
Are back siphoning prevention measures in place?	
Other water issues or incidences? Please describe.	
If you have answered yes to any red questions or no to any blue questions, then there are inherent or managed water risks on the farm.	

Biodiversity/natural features

Below are some examples of questions to ascertain whether there are inherent or managed biodiversity risks on your farm.

Some guideline questions for biodiversity	Notes/description
Are there any special areas or species of interest or conservation on the farm?	
Are there any water or wetland features on the farm?	
Are these features actively protected?	
Are surface water features protected from stock access?	
Is there evidence of bankside erosion.	
Other biodiversity issues? Please describe.	
If you have answered yes to any red questions or no to any blue questions, then there are biodiversity risks on the farm.	

Stock nutrient losses

Below are some examples of questions to ascertain whether there are inherent or managed stock nutrient loss risks on your farm.

Some guideline questions for stock nutrient loss	Notes/description
If stock over wintered outside on the farm, are strategies in place to reduce winter nutrient losses?	
Are measures taken to control dietary intakes of N and P? (Intensive beef and dairy).	
Are stock restricted from entering watercourses?	
Are troughs rotated/moved during winter in hydrologically connected fields?	
Other stock nutrient issues or incidences? Please describe.	
If you have answered yes to any red questions or no to any blue questions, then there are stock nutrient risks on the farm.	

Track management and runoff

Below are some examples of questions to ascertain whether there are track management and runoff risks on your farm.

Some guideline questions for track management and runoff	Notes/description
Are there tracks in hydrologically connected areas?	
Do any tracks run through streams?	
Do any tracks directly runoff to a water course.	
Are devices in place for removing and/or treating contaminated water from tracks?	
Are tracks for stock specifically maintained?	
Do stock regularly pass through water courses?	
Are there any sloping fields adjacent or hydrologically connected to a water course?	
Any previous runoff or soil wash?	
If arable or fodder crops are grown, are measures taken to conserve or build soil organic matter on arable land?	
Are remedial measures in place after winter grazing?	
Is there a possibility of run off from winter grazed crops reaching a water course?	
If you have answered yes to any red questions or no to any blue questions, then there are runoff risks on the farm.	

Appendix B

Assurance protocols for using OVERSEER nutrient budgeting model

Assurance protocols for using OVERSEER nutrient budgeting model for generating whole farm nutrient losses (A Manderson, 2009. Pers Comm. After Manderson and Mackay, 2008)

- Rainfall figures must be from a consistent database;
- Only qualified OVERSEER operators;
- No change to default settings without full justification;
- Submission of model input parameters;
- Submission of key model outputs;
- Monitoring and recording of organic and inorganic fertiliser applications; and
- Random auditing.

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