
**Stocktake of diffuse pollution
attenuation tools for New Zealand
pastoral farming systems**

**NIWA Client Report: HAM2007-161
December 2007**

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Stocktake of diffuse pollution attenuation tools for New Zealand pastoral farming systems

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Executive Summary

New Zealand's surface waters (streams, lakes, rivers, wetlands) and groundwater systems are coming under increasing pressure from intensive farming. With increasing awareness of the environmental risks accompanying intensification, several farmer-focussed initiatives have been put in place to create a more environmentally sustainable farming industry (e.g., Dairying and Clean Streams Accord and the Dairy Industry Strategy for Sustainable Environmental Management targets for the next decade). In order to meet these targets both reduced generation and improved attenuation of nutrients and pathogens is required. This report reviews existing attenuation tools, assesses their cost-effectiveness and identifies gaps in knowledge and communication.

Attenuation is the permanent loss or temporary storage of nutrients, sediment or microbes during the transport process between where they are generated (i.e., in the paddock) and where they impact on water quality (i.e., a downstream water body, such as a lake). Generic attenuation processes include flow attenuation, deposition, microbial transformations, vegetation assimilation and other physical and biogeochemical processes. The driving force behind pollutant transfer from land to waterbodies is water, because it provides the energy and the carrier for pollutant movement.

Cost-effective utilisation of attenuation processes within a particular farm or catchment requires an understanding of the key hydrological pathways operating in the landscape and concomitant opportunities to intercept these. This information, along with regional or local water quality targets, can then assist with prioritisation of pollutants and choice of appropriate attenuation tools. These may include exploiting natural features of the landscape (e.g., seepage wetlands) that should be maintained or enhanced (e.g., by fencing, blocking drains, and/or planting), or addition of engineered attenuation tools such as riparian filter strips, constructed wetlands or reactive filters.

The attenuation toolbox contains a number of existing tools that can be used on farms if the conditions are suitable. For any particular paddock or farm there may be several attenuation options available to a farmer and this report provides a framework to compare their efficacy, applicability, landscape fit and cost-effectiveness. The tools reviewed include: (1) reducing hydrologic connectivity, (2) riparian management, (4) livestock exclusion (e.g., fencing wetlands and streams), (4) wetlands, (5) drainage manipulation, (6) plant and algae harvesting, (7) reactive filters (e.g., denitrification walls and wood chip filters) and (8) reactive materials (e.g., alum, P socks and subsurface drain materials). Some tools such as livestock exclusion are highly and universally applicable. However, there are constraints on the application of many of the tools. For example denitrification walls are primarily limited to loam soils where shallow subsurface flows can be easily intercepted and thus are not widely applicable. Gaps in communication, research and farm scale modelling tools are also identified for each group of attenuation tools.

Seven generic scenarios encompassing dairy, intensive sheep/beef and hill country sheep/beef farms have been used to evaluate the cost-effectiveness of the attenuation tools. The generic scenarios are loosely based on monitored research catchments for a variety of landscapes and farming types (e.g.,

Bog Burn, Toenepi, Whatawhata, etc.). Annual sediment, nitrogen and phosphorus loads (kg/ha/y) were estimated for each scenario at both the paddock and catchment scales and each attenuation tool was costed to derive an annualised cost value (\$/ha/y). The annualised cost-effectiveness for each applicable attenuation tool was estimated (\$/kg) for each model farm scenario.

Tools included in the scenarios investigated in this report are generally those that have been more widely tested and thus can be reasonably evaluated. Other tools for which there is insufficient information available require further investigation and consideration before their potential can be properly assessed.

Simple scoring systems were developed to summarise the results of the scenario analysis. A hydrology score between 5 and 14 was calculated for each flowpath × pollutant × tool combination and was based on (i) hydrological importance of that flowpath, (ii) opportunities for interception and (iii) proportion of the total paddock load of each pollutant carried. The hydrological scoring system revealed that drainflow, small springs and seeps, and surface runoff are important paddock flowpaths to tackle with attenuation tools. A pollutant removal score was designed to reveal the tools with the most potential for each scenario. Three indices were included based on (i) ease of use, (ii) proportion of the total paddock or catchment load attenuated and (iii) cost-effectiveness. The pollutant removal (attenuation) scores ranged between 4 and 15, with livestock exclusion and bottom of catchment wetlands scoring highly for every scenario. Seepage wetlands also scored well where these were applicable.

The simple scenario scoring systems have been used to prioritise research gaps and needs for tools that are widely applicable, effective and target the major flowpaths. This information was combined with the detailed knowledge gaps identified for each tool to prioritise research recommendations.

The science research priorities identified are:

Develop tools suitable for drainflow and subsurface flow that target multiple pollutants.

The major flowpaths requiring attenuation are drainflow and subsurface flows. Traditionally these “less visible”, low concentration but high volume flowpaths, have been considered to be insignificant transporters of pollutants (compared to high concentration, low volume surface runoff). However, recent research has highlighted their importance. Attenuation tools for these flowpaths are typically pollutant specific rather than multi-pollutant. Cost-effectiveness improves by targeting multiple pollutants. Specific opportunities include:

- Enhancing P attenuation in constructed wetlands e.g., P filters on the outlet structure, P retaining additions to wetland soils
- End of drain filters encompassing sediment, nitrogen and phosphorus attenuation tools

- Further research on the inclusion of reactive materials for removal of P (e.g., lapilli) and N (e.g., woodchips) along new tile drains.

Field test bottom of catchment wetlands.

Bottom of catchment wetlands have potential in both baseflow and stormflow dominated systems (depending on outflow structure design). They become a more cost-effective attenuation tool when marginal or community land is available. There are also options for cost-sharing with the community in recognition of their wider ecological and environmental benefits. An opportunity exists to augment Environment BOP monitoring of the Lake Okaro wetland near Rotorua to include sediment and pathogen monitoring.

Quantify nutrient and pathogen reductions as a result of livestock exclusion and other alternative strategies from hill-country perennial streams,

Investigate the benefits of livestock exclusion on intermittent streams, wetlands and seasonally saturated areas.

Little data exists on nutrient and pathogen reductions due to direct livestock deposition in ephemeral pathways. Current research projects in New Zealand cannot fill this gap because they involve the implementation of multiple BMP at whole of catchment scales. Livestock exclusion is a high profile issue for the dairy industry and is gaining profile in the sheep & beef industries. Livestock exclusion may be problematic on hill-country. Potential research issues include: (1) simple solutions for off-stream watering and (2) in landscapes where total exclusion is impractical alternatives, partial exclusion or modifying animal behaviour (e.g., troughs, supplements or shade). Exclusion could also be beneficial beyond permanent stream margins as seasonal channel network expansion may increase the probability of livestock access to surface water.

Field test seepage wetlands attenuation performance, particularly for SS and P, and evaluate their potential to be reinstated where drained

Much of the research effort on natural seepage wetlands has been on short term (hours) nitrate removal and denitrification rather than total N removal performance. Research is needed to measure the net sediment, N and P exports from a range of seepage wetlands under baseflow and event conditions.

Field-test TN, TP, SS and faecal microbe attenuation from surface drainage by facilitated and constructed wetlands.

Most of the research on treatment of diffuse run-off using constructed wetlands in New Zealand has focussed on subsurface tile-drain flows transporting mainly dissolved nutrients. Wetlands treating surface drainage flows with higher sediment loads are likely to perform well for all of the key pollutants, but further information is necessary to quantify the long-term performance of these systems under local conditions and develop appropriate design guidance.

Communication gaps needing attention include:

Develop simple tools, supported with training courses, to assist with the selection of suitable attenuation tools for different landscape and soil types, and farming systems

None of the existing guidelines provide tools to help farmers/land management officers/farm advisors identify flowpaths and attenuation tools suitable for their particular landscape and farming operations.

Integrate information on a wider range of pollutant attenuation options into farm-scale nutrient-budgeting tools such as Overseer®.

Develop practical guidelines for farmers to support appropriate protection, rehabilitation and management of natural attenuation features on farms (e.g., wetlands).

Develop practical guidelines for farmers to support proper design, implementation and on-going management of other widely applicable attenuation tools (e.g., sediment traps, constructed wetlands).

1. Introduction

New Zealand's surface waters (streams, lakes, rivers, wetlands) and groundwater systems are coming under increasing pressure from pollutants mobilised by intensive farming (PCE 2004). Pastoral farming has degraded New Zealand's freshwater quality by: mobilising sediment, nutrients and faecal microbes; altering stream bank and channel morphology; draining wetlands; and removing riparian shade resulting in nuisance algal growths, heating and oxygen stress (MfE 1997; Parkyn et al. 2002; Parkyn & Wilcock 2004; Smith et al. 1993; Wilcock 1986; Wilcock et al. 2007). Rural stream habitats are typically degraded, with large diel changes in pH, dissolved oxygen and temperature, as well as poor visual clarity (Davies-Colley & Nagels 2002; Wilcock et al. 1999).

Farming in New Zealand continues to intensify. Since 1994 the dairy industry in particular has expanded, and the drive to increase production per hectare and per cow continues to escalate (LIC 2007; PCE 2004). Between 1994/1995 and 2006/2007 the number of dairy cows increased by 38% and the 2006/07 average stocking rate of 2.81 cows/ha is the highest recorded (LIC 2007). Production per cow also continues to increase; between 1994/95 and 2006/07 the average increased 22% from 271 kg milksolids/cow to 330 kg milksolids/cow. Nitrogen fertiliser use has increased from almost none in 1995 to an average of 115 kg N/ha in 2005 (Clark et al. 2007). In addition to N fertiliser the use of supplementary feed sources, such as maize silage and palm kernel extract, has increased (Clark et al. 2007).



Figure 1: Dairy industry trends (adapted from PCE 2004).

The sheep and beef industry is also becoming more intensive. Despite a decline in stock numbers, production has increased as a result of higher lambing rates and heavier livestock weights. Fertiliser use has also increased, particularly on intensive farms (Figure 2).

With increasing awareness of the environmental risks accompanying intensification, several strategies or programmes have been put in place to promote an environmentally sustainable farming industry. For example, in May 2003 the Dairying

and Clean Streams Accord was signed by MfE, MAF, regional councils and Fonterra. The Accord is a statement of intent and framework for actions to achieve the goal of clean, healthy water in dairy catchments. The Accord includes targets for (1) excluding livestock from streams and regionally significant wetlands, (2) managing stream crossings, (3) nutrient budgeting and (4) effluent disposal. The dairy industry has also established a number of environmental goals, outlined in the “Dairy Industry Strategy for Sustainable Environmental Management” (Dairy Environment Review Group 2006). The following targets have been set for the next decade:

- Nitrogen loss – 50 per cent less than benchmark.
- Phosphate loss – 50 per cent less than benchmark in heavy soils, 80 per cent less than benchmark in free draining soils.
- Microbial – capable of delivering contact with standard water in all water-flow leaving the farm property. (Presumably this means achieving standards for contact recreation as specified by MfE/MoH).

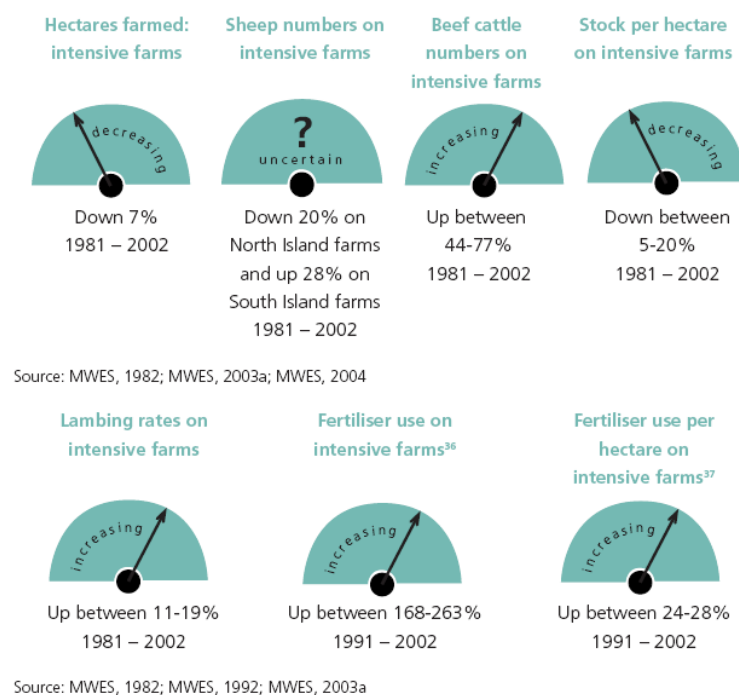


Figure 2: Sheep and beef industry trends (PCE, 2004).

In the paddock, nutrients and sediment are perceived as a resource promoting plant productivity, but downstream in receiving waters they can become pollutants. Relatively small agronomic loss rates (<5% of applied N and P) can translate to significant loads to aquatic systems, especially for lakes and enclosed coastal embayments. Concentrations in farm runoff are typically 0.3-3 gN/m³ and 0.02-0.3

gP/m³. In lakes concentrations above 0.2 gN/m³ and 0.01 gP/m³ can cause eutrophication (Vant 1987), while concentrations above 0.04-0.1 gDIN/m³ and 0.015-0.03 gFRP/m³ are considered to be non-limiting for periphyton growth in flowing freshwaters (MfE 1992). Loading of pastures by grazing livestock provides the potential for the transfer of pollutants to watercourses. The driving force behind pollutant transport from land to waterbodies is water, as it provides the energy and the carrier for pollutant movement (Figure 3). Loss is controlled by physical location, pollutant form, fate and environmental availability, coupled with hydrologic processes, the conveyors of pollutants (Gburek et al. 2000).

This report is a stock-take of pollutant attenuation options for pastoral farming, with specific reference to the dairy, sheep and beef industries. The report briefly examines the sources and forms of key pollutants, defines attenuation and key attenuation processes. A brief examination of attenuation in the context of farm water quality management and planning is made and the key steps for the successful use of attenuation tools explored. Potential attenuation sites and farm hydrology are reviewed. A toolbox of all currently available attenuation tools is developed and the tools are reviewed. New Zealand literature is used wherever possible, with overseas research filling knowledge gaps. The report demonstrates the farm planning steps needed to select attenuation tools from the attenuation toolbox by working through seven generic scenarios and evaluating the cost-effectiveness of relevant tools. Lastly, future research needs and recommendations are outlined.

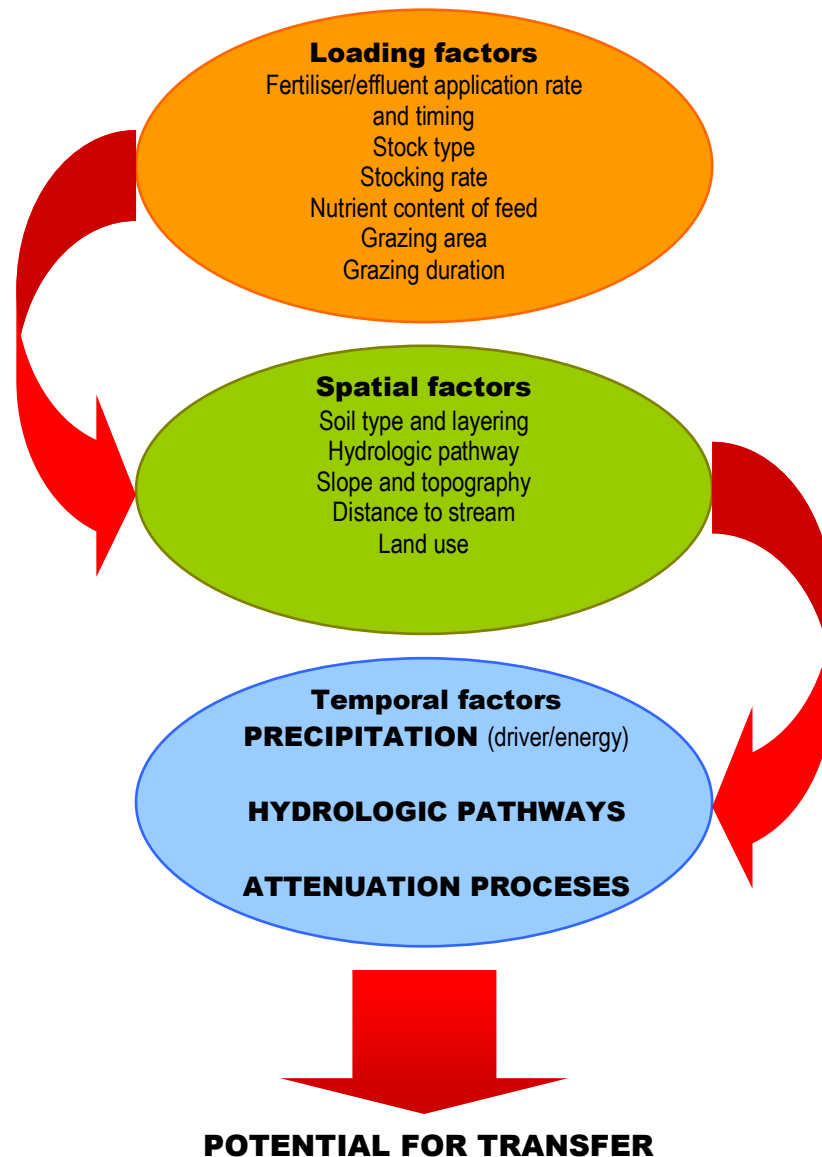


Figure 3: Controls governing pollutant transfers from pasture (adapted from Oliver et al. 2005).

1.1 Key pollutants

Given proper management of point sources of wastewater, the key water quality concerns stemming from pastoral farming relate to the three major diffuse (or non-point source) pollutants: (1) nutrients from livestock wastes, fertiliser application and eroded sediment, (2) microbial contamination from livestock faeces, and (3) sediment impacts (reduced water clarity and sedimentation).

Nitrogen can be transported by water in several different forms, including dissolved inorganic N (nitrate, ammonium and nitrite), dissolved organic N and particulate-associated N (e.g., particulate organic N and adsorbed ammonium) (Figure 4). Forms

of nitrogen can change during transport due to biogeochemical transformations (e.g., mineralisation of organic N, nitrification of ammonia). The main routes for nitrogen transfer from hillslopes to streams are generally (1) nitrate leaching, (2) direct inputs of animal excreta to streams, (3) transport of excreta by surface runoff, and (4) soil erosion. In New Zealand the major sources of nitrogen are from leaching of livestock urine patches and applied N fertiliser (Ledgard & Menneer 2005), as nitrate.

total nitrogen (excluding nitrogen gas)				
organic nitrogen		inorganic nitrogen		
dissolved	particulate	ammonium	nitrite	nitrate
	detritus plankton	sorbed dissolved		

Note: total Kjeldahl N = organic N + ammonium

total phosphorus			
filterable/dissolved/soluble		particulate	
reactive	organic	organic	inorganic
		detritus plankton	

Figure 4: Main forms of nitrogen and phosphorus in water (after McCutcheon et al. 1993).

Phosphorus may be transported in soluble and particulate forms, with particulate P including P sorbed (incorporated into or adhering to the surface) by soil particles and organic matter (Figure 4). In this report the term filterable P is used, rather than dissolved or soluble P, as the filtrate could be a mixture of dissolved forms and P attached to colloidal material that passes through the 0.45 µm filter. Phosphorus may come from many sources, including fertilisers, soils and any weathered rock, plants, microbial biomass and grazing animals. In headwater streams, direct deposition of P fertiliser into the stream channel can constitute a significant proportion (~8%) of total P exported (Cooke 1988).

Faecal material is a source of enteric viruses, bacteria, cysts and oocysts and parasitic protozoa. Faecal contamination is usually detected by testing for *indicator micro-organisms*, such as *E. coli*, that are consistently present in faecal wastes (Donnison & Ross 1999). It is assumed that if these organisms occur in stream water, then other more pathogenic micro-organisms are also likely to be present, such as *Campylobacter*, *Salmonella*, *Cryptosporidium* and *Giardia*. The key source of faecal contamination on farms is grazing livestock, although wild and feral animals can be an additional source. Faecal microbes may be introduced to freshwaters via “direct” (i.e., deposited directly into stream) or “indirect” pathways, such as the transmission of fresh or aged faecal matter in surface runoff, subsurface flows or drainage (Collins et al. 2007).

Sediment is made up of particles derived from mineral or biological material transported by water. Particle size exerts a major control on entrainment, transport and deposition of sediment. Suspended sediment is operationally defined as fine-grained particles that are retained by a 1.5 μm filter. It is mostly fine grained mineral particles <63 μm or low density organic particles up to ~ 1 mm (Davies-Colley & Smith 2001). Fine sediments, for example clay size particles, have large specific surface areas and therefore can be enriched with pollutants. Sediment can be sourced from paddocks or from stream banks and beds. Generation and transport of sediment from paddocks to streams is strongly influenced by particle size, soil hydrology, slope and land management.

2. Attenuation

Attenuation is the permanent loss or temporary storage of nutrients, sediment or microbes during the transport process between where they are generated (i.e., in the paddock) and where they impact water quality (i.e., a downstream water body, such as a lake). Generic attenuation processes include flow attenuation, deposition, microbial transformations, vegetation assimilation and other physical and biogeochemical processes (Table 1).

These processes can alter pollutant concentrations and loads by (i) decreasing the mean concentration or load, (ii) decreasing variability of concentration or load, (iii) decreasing the concentration or load maxima or minima, (iv) increasing the total removed and (v) reducing the frequency of high concentration or load (Viaud et al. 2004). In this report load (or export) changes are given precedence over concentration changes, as the cost-effectiveness calculations are completed on annual time steps.

2.1 Deposition (and other processes enhancing deposition)

Once a particle is entrained in water it begins to sink under gravitational forces. The distance it travels depends on the drag force of the water and the settling velocity of the particle. Deposition begins once the flow velocity falls below the settling velocity of a particle. Settling velocity is closely related to particle size and density, so that the coarse and dense particles are deposited first, with finer and lower density particles settling later, often only as the flow velocity falls.

Suspended sediment may also form flocs or composite particles that are deposited as they increase in mass and settle at a faster rate. Flocculation is the process whereby smaller particles (inorganic and organic) aggregate to form larger particles (flocs). Flocs are composite structures composed of biological material (e.g., bacteria, detritus), inorganic particles (e.g., clay) and water (Droppo 2001). Flocs form within a water column or on the surface of the bed by a variety of complicated physical (e.g., turbulence), chemical (e.g., ionic concentration) and biological mechanisms (e.g., bacterial population) (Droppo 2001). The composition and structure of a floc is in a continuous state of change and as a result a floc's physical (e.g., transport), chemical (e.g., adsorption) and biological behaviours may vary markedly.

Sediment deposition stores material and thereby reduces the concentrations and export (load) of suspended sediment from the system. Sediment storage may be short or long term. For example, sediment deposition in a grass filter strip will probably be short term (days-tens of years), while a blanket of sediment over a floodplain will be stored for longer (tens to hundreds of years; Fryirs et al. (2007)).

Sediments are a mixture of living organisms, inorganic and organic particles, faecal microbes, microorganisms and may be a substantial nutrient store. Suspended sediment can contain a suite of sorbed and structural chemicals notably phosphorus (Baldwin et al. 2002). Deposition of pollutants with suspended sediment will increase their residence time in the system and potentially expose them to other physical, biological and biogeochemical attenuation processes (e.g., deep burial and uptake by plants). Deposited sediment may also assist with creating suitable environmental conditions for other attenuation processes to occur. For example, anaerobic organic matter-rich conditions are conducive to denitrification. On the other hand physical, biological and biogeochemical processes may result in the release of soluble nutrients from deposited particulate forms. This process may reduce the long-term effectiveness of mitigation measures such as wetlands and grass filter strips – while they remove particulate nutrients during flow events these may subsequently breakdown and release soluble nutrient.

2.2 Biota uptake and stores

Plants and their residues can be important sinks and sources of nutrients. Growing plants and microorganisms immobilise inorganic P, N and other macro- and micro-nutrients from the soil into their biomass. A proportion of these nutrients are released when plant cells die and decay, and can be transported in particulate or dissolved organic forms, or (as a result of mineralisation) in inorganic forms. Microbes grow, die and decompose on a much faster time scale than do vascular plants, but they can be important in short-term pollutant immobilisation (Kadlec & Knight 1996).

Assimilation refers to a variety of biological processes that convert inorganic nutrients (e.g., nitrate or ammonium and phosphate) into organic compounds that serve as the building blocks for cells and tissues.

Plants can also be used for phytoremediation - the in-situ treatment of contaminated soils, sediment and water by plants. Organics, nutrient and metal pollutants can be accessed by the roots of plants and transpired, sequestered, degraded, immobilised or metabolised.

Table 1: Attenuation process definitions and constraints.

Attenuation process	Description/definition	Constraints	References
Deposition	settling of sediment, flocs, detritus, phytoplankton from the water column e.g., floodplains, soil deposits	low velocities promote settling; larger particles settle quickly, fines slowly.	(Knighton 1984)
Infiltration	entry of water and associated pollutants into the soil	ability of soil to transmit water away from soil surface	(Hillel 1971)
Filtering	sieving of coarse particles by plants or finer particles/microbes by soil matrix	porous barrier e.g., dense grass cover or soil	(Dosskey 2001; Oliver et al. 2005)
Plant/algae uptake	removal of dissolved inorganic or organic nutrients from water or soil water by plants	unless harvested and removed nutrients will be released during senescence and decomposition	
Denitrification	microbial production of nitric oxide (NO), nitrous oxide (N ₂ O) and N ₂ from nitrate	low oxygen (suboxic) conditions, carbon source, nitrate available	(Burgin & Hamilton 2007; Seitzinger et al. 2006)
Flow attenuation	Storage and attenuation of flood runoff	sufficient storage, location in landscape, flooding	(Mitsch 1992)
Adsorption	physical or chemical bonding of molecules to the surface of solids	P: soil chemistry (FeO _x , AlO _x and clays), solid type, pH, P concentration; P released under anoxic conditions faecal microbes: presence of salts and organic matter, pH	(Baldwin et al. 2002; Ferguson et al. 2003; Reddy et al. 1999)
Precipitation	removal of components from solution by their mutual combination forming a new solid-phase compound	P: soil chemistry (Fe III, Ca, Al)	(Baldwin et al. 2002; Reddy et al. 1999)
Immobilisation (microbial uptake)	accumulation of nutrients into microbial biomass	most significant when there is a large supply of carbonaceous material, e.g., plant litter, sawdust	
Microbial inactivation	inactivation of faecal microbes by unfavourable environmental conditions	many stressors including temperature extremes, pH extremes, low soil water potential, high ammonia concentrations and organic matter contents, UV exposure, oxic conditions, and predation.	(Ferguson et al. 2003; Oliver et al. 2005)

2.3 Biogeochemical transformations

Denitrification is the conversion of simple organic carbon and an electron acceptor such as nitrate, to energy, carbon dioxide and gaseous oxides (NO, N₂O) or nitrogen gas (N₂). Denitrification occurs when three conditions are satisfied: nitrate is available, oxygen concentrations are low (suboxic to anoxic, <0.2 mgO₂/L) and there is sufficient organic carbon (Seitzinger et al. 2006). Denitrification occurs in microsites within well drained soils in agricultural land, partially to fully saturated soils, groundwater aquifers, surface and riparian sediments and suboxic bottom waters of lakes and estuaries (Seitzinger et al. 2006).

Another microbially-mediated nitrate removal process is dissimilatory reduction of nitrate to ammonium (DNRA). DNRA occurs under anaerobic conditions in soils and sediments and converts nitrate to ammonium which is presumed to be more available than nitrate for plant uptake.

An additional N attenuation process is anammox (anaerobic ammonium oxidation), although little is known about this process in freshwaters (Burgin & Hamilton 2007). Anammox is the combination of ammonium and nitrite under anaerobic conditions producing N₂. The nitrite is derived from the reduction of nitrate, possibly by denitrifying bacteria. Anammox may be important in ecosystems with limited labile carbon or an excess of nitrogen relative to carbon inputs (Burgin & Hamilton 2007).

2.4 Faecal microbe inactivation

Faecal microbes can be inactivated by a large number of processes in soil and water. Unfavourable environmental conditions such as temperature extremes, pH extremes, low soil water potential, high ammonia concentrations and organic matter contents, UV exposure, oxic conditions, and the presence of other micro-organisms will all contribute to death of faecal microbes (Ferguson et al. 2003; Oliver et al. 2005). Microbe survival may be greater in the stream bed sediments in comparison with the overlying water column. Bed sediments can provide protection against predation, UV inactivation and act as a source of nutrients (Oliver et al. 2005). Artificial flood experiments in a Waikato stream generated total in-channel *E. coli* stores of approximately 10⁸ cfu/m² streambed (Muirhead et al. 2004), confirming the in-stream survival of *E. coli*.

2.5 Chemical processes

Adsorption is the physical or chemical bonding of molecules to the surface of solids (soil, sediment, organic particles). Sorption may be physically mediated and reversible, or chemically mediated and partly or wholly irreversible. Adsorption is particularly important for phosphorus, ammonia, faecal microbes, pesticides, metals

and organic pollutants (e.g., PCBs). Over time the adsorption sites on particles may fill and the adsorption capacity becomes saturated.

Precipitation is the removal of components from solution by their mutual combination forming a new solid-phase compound. If the solid has sufficient mass, it will fall out of suspension and sink to the bottom of the water column. For example, phosphorus is made unavailable to plants and microbes by the precipitation of insoluble phosphates with ferric iron, calcium and aluminium under aerobic conditions.

2.6 Flow attenuation

Natural and constructed attenuation systems may have water storage and attenuation functions. Flow attenuation increases the time that water and pollutants spend within a system and can also reduce the velocity of water (reducing entrainment and enabling deposition). Water storage may be short or long term depending on the size of the flow relative to the storage available. The degree of attenuation at a given peak inflow will be far greater for low volume floods and as the flood peak and volume increase, attenuation will decrease.

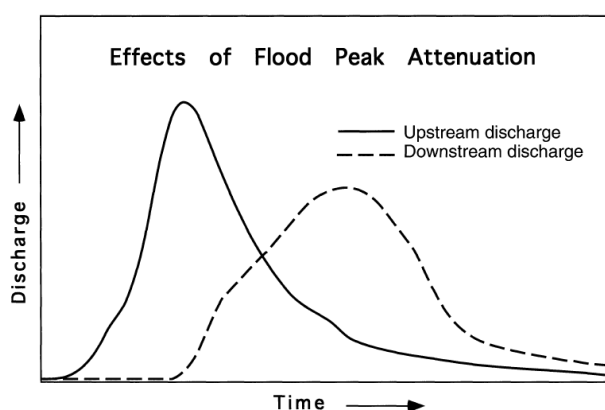


Figure 5: Generalised hydrographs illustrating the effects of flood peak attenuation. In the process of flowing from the upstream site to the downstream sites, peak discharge is reduced while the total flood volume is unchanged (Woltemade 1994).

2.7 Infiltration and filtering

Infiltration of surface runoff into the soil matrix provides attenuation opportunities. As water infiltrates large particles may be retained on the soil surface. Fine pollutants that infiltrate with the water will be exposed to adsorption sites within the soil. Once water enters the soil, soil type, condition and moisture content determine pollutant movement and removal.

Filtering of surface water by plants and/or the soil matrix is an important attenuation process during rainfall events. Mesh-like plant stems, such as dense grass, and

associated litter layers can sieve large soil aggregates and debris from surface runoff (Dosskey 2001). There is also evidence that bacteria and protozoa may physically block soil pores, effectively straining micro-organisms from suspended flow (Oliver et al. 2005).

2.8 Use of attenuation in farm water quality management

In order to successfully use attenuation tools in farm planning, the movement of pollutants from pasture to streams must be understood. We consider that there are three key steps to maximising farm water quality:

- setting water quality targets
- minimising risk of pollutant generation
- minimising risk of pollutant transport
 - identify and prioritise flowpaths
 - evaluate the relevance and cost-effectiveness of existing and additional attenuation tools.

Once the need for action has been recognised, priority pollutants and water quality targets need to be developed. Targets must be carefully defined, including the time frame they are to be achieved in, the indicators to be monitored and then benchmarks must be set against which future changes will be monitored set. Catchment targets need to be developed locally, with consideration of water quality outcomes desired by the local community, the sensitivity of downstream receiving waters and the natural of the pollutant sources.

An underlying principle behind water quality management is that if pollutant generation can be minimised, then transport to receiving water bodies will automatically be reduced. Source control strategies limit the generation and loss of pollutants from their source (e.g., reducing fertiliser application rates, nitrification inhibitors, wintering-off). Livestock exclusion is regarded by some as a source control technique. It has been included in this report as an attenuation tool as it is generally targeted along waterways and is the most basic form of riparian management.

In many instances source controls alone will not be sufficient and complementary attenuation tools will also be required to meet water quality targets. These may not be as effective as source control and are generally more difficult to design and maintain. Time lags in the movement of pollutants mean that in some catchments nutrient concentrations in streams will increase despite the introduction of source controls. For example, around Lake Rotorua many of the catchments are underlain by thick layers

of fractured ignimbrite and the streams have a high proportion of baseflow supplied by groundwater flow. The water in some of these streams has been dated using tritium, CFCs and SF₆ and the mean estimated residence time of the groundwater varies from 16 to 127 years (Morgenstern et al. 2004). The legacy of intensive pastoral farming is yet to be seen in streamflow and a range of attenuation tools could be applied to treat current and future stream nutrient loads before they reach the nutrient-sensitive lake.

Attenuation tools can provide a buffer between pastoral land use and receiving water bodies. For example, ponds and wetlands can buffer stream habitats against extreme concentration fluctuations and downstream of farm dairy effluent infrastructure they can reduce the severity (and associated penalties) of accidental spills (e.g., Sukias et al. 2008).

The following sections focus on using attenuation tools to minimise pollutant transport. Water flowpaths and their connectivity are explored and then the tools in the attenuation toolbox are reviewed and their relevance and cost-effectiveness examined.

3. Potential attenuation sites

The driving force behind nutrient transfer from land to water is water, as it provides the energy and the carrier for pollutant movement. The hydrology determines how water moves, its physical characteristics (e.g., flow rate and volume), and the area and time of contact with the soil (Nash et al. 2002). There are two basic types of hydrologic energy – slowly varying low flow between events, and rapidly varying high flows resulting from high energy rainfall events. Some catchments with porous soils and extensive groundwater will have fairly stable flow regimes with limited variation in flow rate through time, while others with shallow soils may have seasonally variable baseflows and frequent floods. Rainfall events can therefore affect water quality in streams as illustrated by large and sudden increases in pollutant concentrations generated by a flood event compared to the preceding dry period.

Hydrology can be considered at various scales: (1) soil or local scale (<10 m), (2) paddock/hillslope scale (10-1000 m) and (3) catchment scale (>1000 m). While the scale boundaries are arbitrary, the key point is that the important pathway may change with scale. For example, as one moves from the paddock to catchment scale groundwater may be an increasingly dominant flowpath.

3.1 Water pathways

At the paddock/hillslope scale (10-1000 m), hydrologic flowpaths can be broadly separated into surface and subsurface pathways (Figure 6). Subsurface pathways can be further classified into matrix, preferential or drainage, and groundwater. The important question at the paddock/farm scale is which pathway, surface or subsurface, dominates with respect to flows, pollutant concentrations or loads (concentration \times flow). The dominant flowpath will differ for sediment, nitrogen, phosphorus and faecal microbes. For example, most sediment will be transported by surface runoff.

Surface runoff or overland flow passes across the paddock/hillslope as a visible flow of water over the ground surface (Goudie et al. 1994). It can be a mix of infiltration-excess overland flow and saturation-excess overland flow. Infiltration-excess overland flow (IEOF) occurs when water enters a soil system faster than the soil can absorb or soak it up. It is common on impervious surfaces such as tracks and roads, low porosity soils or where heavy grazing results in pugging and compaction, or seals the soil surface. Saturation-excess overland flow (SEOF; Figure 7) is a combination of rain falling directly onto saturated areas and shallow subsurface flow that re-emerges at the soil surface (Chorley 1978). Soil saturation and return flow often occur when there is a reduction in slope angle (e.g., at the base of a hillslope), and where slowly permeable or impermeable layers are close to the soil surface. Surface runoff will have a short contact time with the hillslope, unless it infiltrates before entering the stream. The residence time is typically in the order of hours to days.

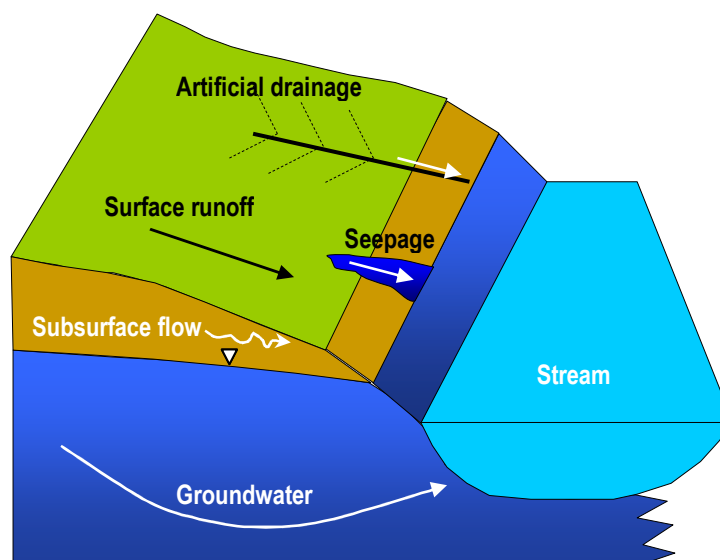


Figure 6: Water pathways from pasture to streams.

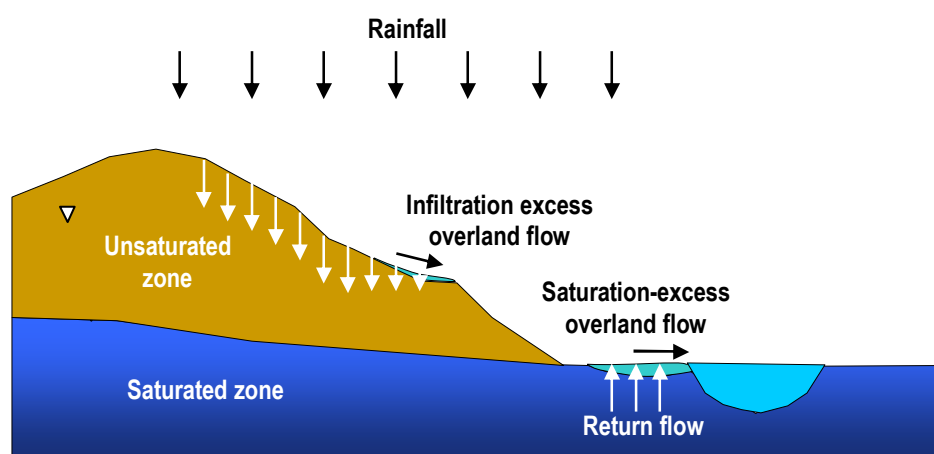


Figure 7: Surface runoff generated by infiltration-excess overland flow and saturation-excess overland flow.

Subsurface pathways include lateral flow through the soil and vertical drainage to groundwater. Subsurface flow is generated following infiltration of rain into the soil, and water may flow rapidly via macropores, (small channels through the soil – artificial or natural) or saturated soil horizons, or slowly through the soil matrix. Matrix flow passes slowly through the soil and has a long contact time (weeks to years). The retention time of water and soluble pollutants is therefore dependent upon many soil properties (e.g., texture, porosity, bedrock/impermeable layers, slope), position in the landscape and amount and frequency of rainfall. Artificial drainage, such as mole and tile drains, can rapidly divert water and pollutants to a stream, thus short-circuiting other pathways and reducing the opportunities for nutrient attenuation.

This transfer will take place in the order of minutes to hours. Water that infiltrates the surface and moves vertically through the soil profile may become groundwater flow. Groundwater flow is deeper saturated flow (Figure 6) and groundwater residence times and flow rates depend on the underlying soil and rock (aquifer) properties. Groundwater may re-merge at the ground surface as springs or flow directly into receiving water bodies.

3.2 Connectivity

The delivery of pollution to a stream from a paddock/hillslope depends on the spatial and temporal movement of water via flow pathways and their connectivity. Hydrological connectivity refers to the passage of water from one part of the landscape to another (Bracken & Croke 2007). Understanding the connectivity between parts of the landscape (e.g., paddocks and streams) is crucial in explaining the behaviour of pollutants.

Landscape setting and configuration shapes the operation of hydrological processes over a range of spatial and temporal scales. Some landscapes have well connected flowpaths and rapidly transmit water during rainfall events. Pastoral systems with artificial drainage, soil structural degradation or impermeable areas such as tracks and races have increased connectivity resulting in highly pulsed hydrology. For example, event hydrographs for mole and tile drains on silt loam near Palmerston North peaked in less than 5 minutes (Magesan et al. 1995). The delivery of large quantities of pollutants over such short time periods presents challenges for attenuation systems. Other landscapes may be less well connected resulting in significant lag times in pollutant movement. During events water may travel short distances before infiltrating into porous soils, particularly during small-moderate events. For example, on pasture at Kaharoa near Lake Rotorua, surface runoff was recorded on eight days in one year, but most re-infiltrated and on only three of these days was there significant surface runoff (McKergow, unpublished data). Disconnection in some landscapes may be the result of time lags introduced by slow moving groundwater.

Connectivity can also vary on seasonal scales. For example, Wigington et al. (2005) documented seasonal channel network expansion in an Oregon stream. During winter the channel network included drains and intermittent streams and the active stream drainage density was 8 km/km², while during summer flow was restricted to a single perennial channel (0.24 km/km²). Expansion of saturated areas (e.g., Cooke & Dons 1988) can also increase connectivity seasonally.

Landscapes can be broadly classified into units that have similar flow paths. While there is no unifying landscape classification system used in hydrology (McDonnell & Woods 2004), some generic classifications have been derived for northern hemisphere

near-stream zones (from Haycock & Muscutt 1995). Such classification systems could be useful at a regional or landscape unit level to help identify flowpaths and suitable attenuation tools.

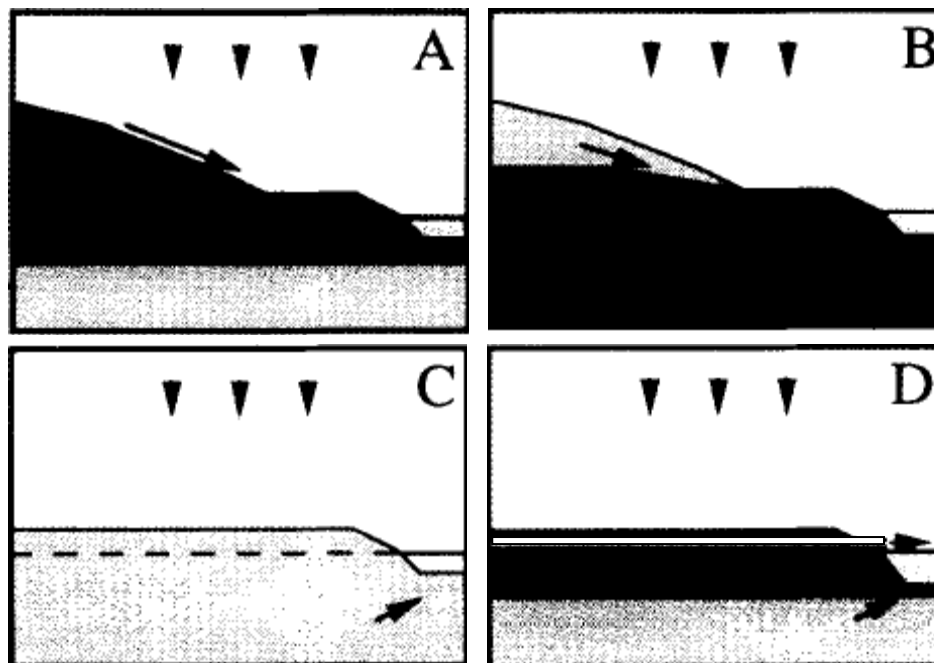


Figure 8: Simple classification of hillslope hydrogeology, with impermeable substrate in black and permeable substrate in grey. A. Impermeable hillslope and floodplain; B. Permeable hillslope; C. a flat permeable landscape; D. an artificially drained impermeable landscape (Haycock & Muscutt 1995).

Opportunities to treat agricultural pollution include: (i) close to the source, preventing mobilisation, (ii) along the transport pathways and (iii) before a stream or aquifer exits into a sensitive environment (e.g., bottom of catchment treatment). Pollutants may pass through multiple attenuation tools on their way to the bottom of the catchment (Figure 9). This tiered approach introduces an element of safety to the attenuation cascade.

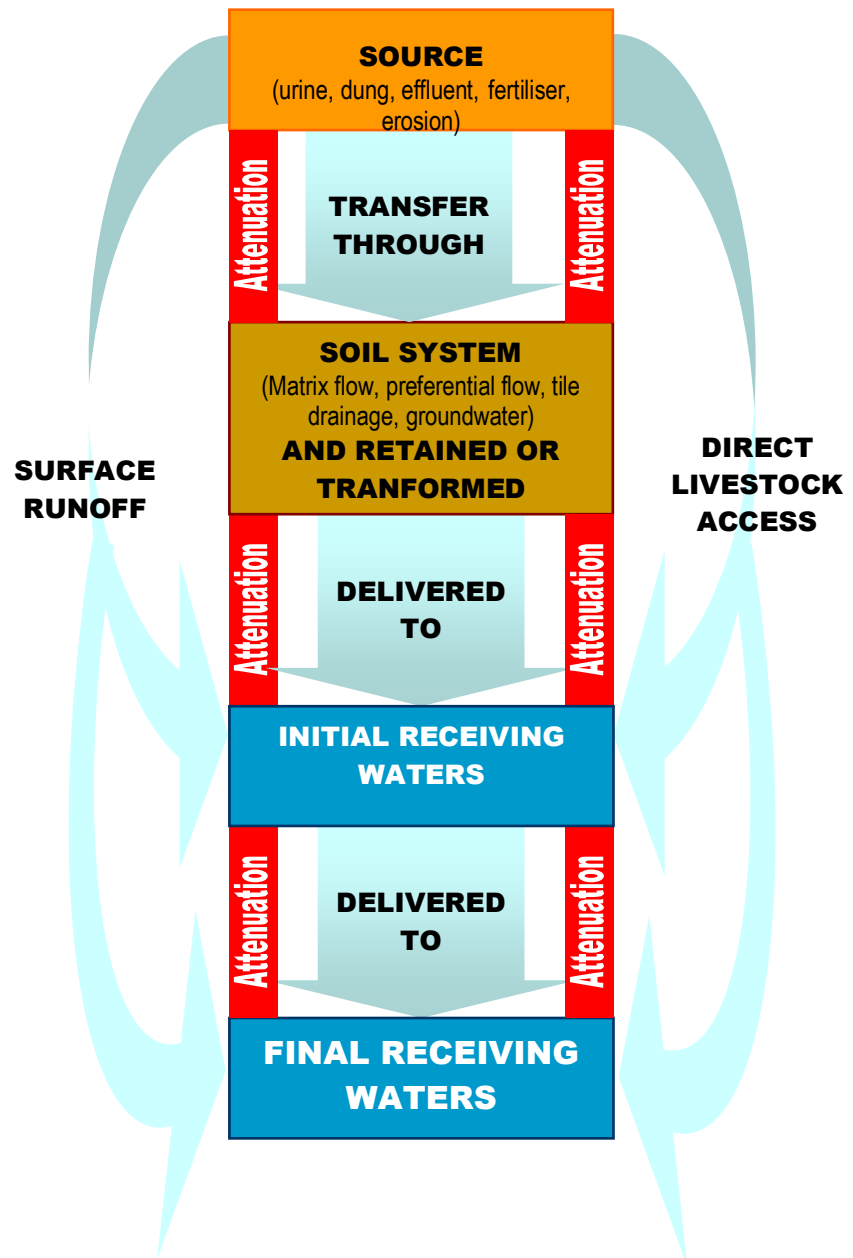


Figure 9: Conceptual model of pollutant transmission and attenuation from the pasture surface to receiving waters (adapted from Oliver et al. 2005).

Agricultural pollution may be distributed throughout the landscape (diffuse source; e.g., seepage of groundwater or sheet flow surface runoff) or focused at discrete discharge points (point source; e.g., effluent pond discharge). Diffuse sources are often individually minor, but collectively significant. Point sources are usually simpler to recognise and treat. Pollutant pathways may shift between the point and diffuse extremes. Neal & Jarvie (2005) expand the simple point-diffuse classification to include point-diffuse and diffuse-point sources. For example, mole and tile drains collect diffuse runoff which is then discharged to a stream as a point source (diffuse-point). In contrast, land disposal of dairy farm effluent from a storage or treatment pond converts a point source to a diffuse source (point-diffuse).

Many attenuation tools could be placed at several different locations in the landscape. For example, constructed wetlands can be placed on the end of each tile drain outfall, or one larger wetland could be constructed to treat the streamflow from a small catchment.

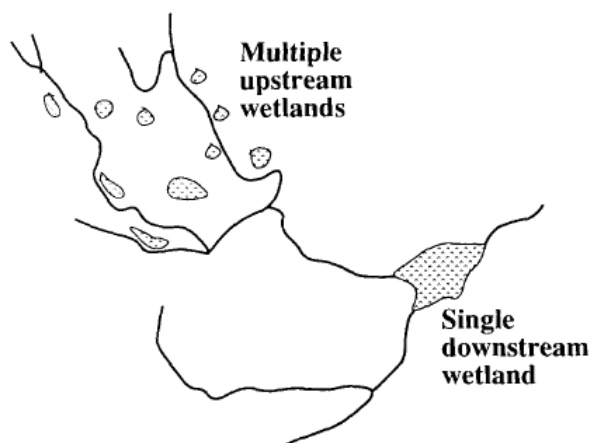


Figure 10: Alternatives for locating wetlands in a catchment – many smaller upstream wetlands intercepting small streams or tile drains versus one larger downstream wetland (from Mitsch 1992).

As well as being significantly affected by farm water management (e.g., irrigation and drainage), the water-yield and hydrological regimes (flooding, ephemeral and low-flows) of waterways and waterbodies in agricultural catchments may be locally and cumulatively affected by attenuation systems. For example, filter strips, ponds and wetlands can all attenuate flows, increase infiltration and evaporation losses, and change the physicochemistry of runoff. Such hydrological effects can influence the ecology, and human uses and values of downstream aquatic systems.

4. Attenuation toolbox

After dominant pollutant flowpaths have been identified on a farm and their ease of interception has been evaluated, attenuation tools can be compared and assessed. We use the analogy of a toolbox to convey the concept that each tool has been designed to do a particular task. The attenuation toolbox contains a number of natural and constructed tools that could be used on farms if conditions are suitable (Table 2). Preserving and protecting natural attenuation tools such as seeps and wetlands is a cost-effective attenuation option. For each tool a range of variations is listed, depending on the landscape characteristics, intercepted flowpath and chosen implementation scale. Some tools, for example constructed wetlands, can be applied at a range of scales from paddock (e.g., small wetlands at the end of tile drain) to larger wetlands at the outlet of a small catchment.

For any particular paddock or farm there may be several attenuation options available to a farmer. Table 2 contains a summary of the likely applicability, landscape fit, knowledge gaps, efficacy and cost, and additional benefits/disbenefits for each attenuation tool. Some tools such as livestock exclusion are highly and universally applicable (Table 2). However, there are constraints on many of the tools. For example, denitrification walls are primarily limited to loam soils where subsurface flowpaths can be easily intercepted and thus are not widely applicable (Table 2).

Natural, rehabilitated and constructed attenuation systems (e.g., riparian protection and wetlands) can in themselves have important habitat, biodiversity, and aesthetic functions and values in agricultural landscapes. Additionally, emissions of greenhouse gases (CO_2 , CH_4 , N_2O) are an indirect outcome of many contaminant transformation processes, whether they occur in an attenuation system or at a downstream site where they have impacts (e.g., lake or estuary). Therefore, in addition to direct reduction of contaminants, the broader influence of the attenuation options being applied needs to be understood and assessed. An accurate economic evaluation of many attenuation options will only be possible once research is conducted so that these ancillary benefits (ecosystem services) can be taken into account.

Six broad groupings of attenuation tools have been identified and are used to structure this section of the report – reducing hydrologic connectivity, riparian management, drainage manipulation, sediment traps/dams/ponds, wetlands and plant/algae harvesting, and reactive materials/filters. For each group the efficacy and applicability of individual tools is evaluated. Some of the attenuation tools are widely used already (e.g., riparian buffers), while others are the subject of full-scale trials in several landscapes (e.g., constructed wetland receiving subsurface drainage) or are emerging technologies at the ‘proof of concept’ phase (e.g., floating wetlands). The attenuation

tools are placed on the spectrum from ‘proof of concept’ to ‘action’ to illustrate their development status (Figure 11).

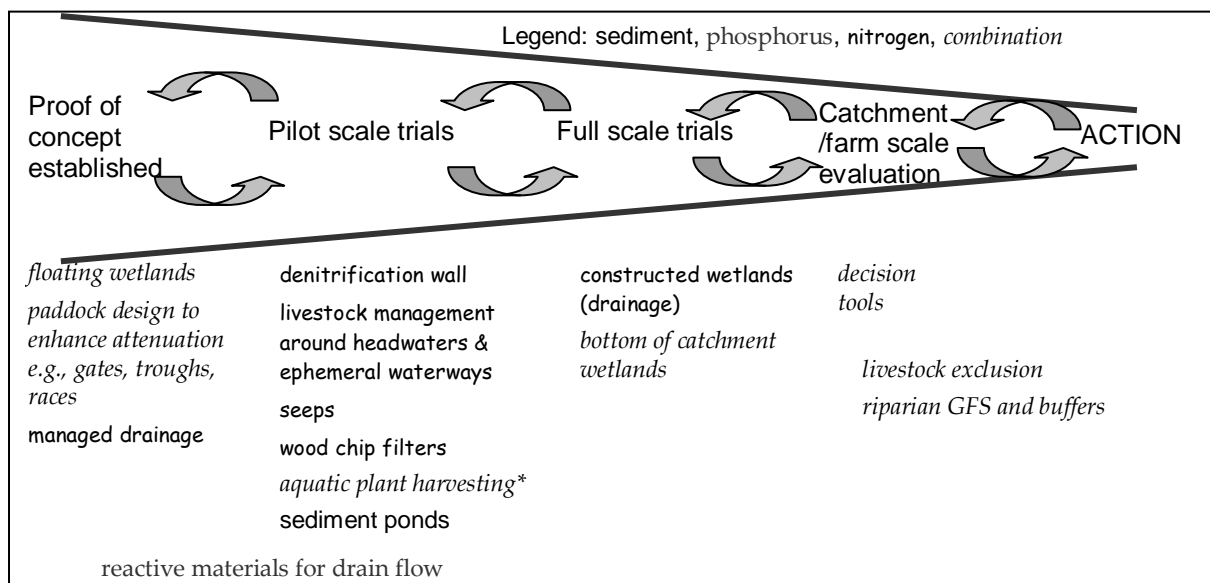


Figure 11: Spectrum of attenuation tool development status in New Zealand. Fonts indicate the pollutant targeted by each tool.

Where sufficient information is available the tools (minimum pilot scale trials; Figure 11) are costed and a short overview of the assumptions underlying the costing of the tool is provided. Gaps in communication, research and farm scale modelling tools are also listed for each group of attenuation tools. Some tools have been field tested in several landscapes (e.g., constructed wetland receiving subsurface drainage), while others are emerging technologies that may have been evaluated at one site (e.g., P sock).

All cost estimates are based on realistic commercial rates (exclusive of GST) as if done by an external contractor, including consultants fees, contractor establishment fee, digger and operator rates of \$120/hr, labour at \$50/hr, and transport of materials 50 km to site. No specific allowances have been made for any resource consent-related costs that may be involved in implementing these attenuation options. Fencing and weed spraying costs are applicable to several tools and standard costs have been adopted for these. Two wire electric fencing (No. 2 round posts at 8 m spacing) is used on dairy farms and costs \$1.90/m. Five wire (three electric, no battens) fencing is the chosen option for drystock farms and costs \$4.80/m. These costs are based on an article in QEII National Trust Open Space Magazine (No. 58, September 2003). Weed spraying costs are estimated at \$55/ha.

Table 2: Attenuation toolbox (rankings are High, Moderate, Low).

Attenuation tool	Description	Variant(s)	Intercepted flowpath(s)	Primary attenuation processes	Scale(s)	Likely applicability in NZ	Target pollutants	Landscape fit	Knowledge level in NZ	Efficacy	Cost	Other indirect benefits /disbenefits	Key NZ references
Reducing hydrologic connectivity													
Farm design to reduce connectivity	water sensitive placement of troughs, gates and tracks to reduce connectivity with streams		surface runoff	disconnection	paddock, farm	H	Sed, P, N, faecal microbes	H	L	L-M	L		
Riparian management													
Livestock exclusion	exclude livestock from margins of streams, drains, stock water races, lakes, wetlands and estuaries	bridges and culverts	stream flow, surface drains, stock water races, lake water, wetlands, estuaries	avoid direct faecal deposition + grazing disturbance	paddock, farm, catchment	H	Sed, P, N, faecal microbes	H	H	H	L	b: reduced stock losses; aesthetics d: potential weed management issues	(Davies-Colley et al. 2004; McKergow & Hudson 2007)
Grass filter strip	managed band of dense grass	riparian, hillslope or ephemeral channel	surface runoff (sheet flow)	deposition, infiltration, filtering	paddock	M	Sed, P, N, faecal microbes	L	M	M-L	L	d: potential weed management issues	(Collins et al. 2004; Smith 1989)
Riparian buffer	managed band of shrubs/trees along stream bank	native plants; forestry; sequential GFS + buffer	surface runoff (sheet flow) + subsurface flow	deposition, infiltration, filtering, nutrient uptake, denitrification	paddock	H	Sed, P, N	M	H	M-L	M	b: channel shading; improved aquatic habitat, wood and leaf supply to stream; recreation; cultural harvesting of flax and other plants; biodiversity value; landscape aesthetics d: requires some active vegetation management.	(Cooper et al. 1995; Williamson et al. 1996)
Drainage manipulation													
Vegetated or partially-vegetated drains	vegetated surface drains with marsh vegetation or water-tolerant grasses.		surface runoff + subsurface flows in surface drains	deposition, denitrification, nutrient uptake	paddock, farm	M	Sed, N, P	L	L	M	L-M	b: improves biodiversity and provides seasonal aquatic habitat d: may require redesign of drainage systems to prevent flooding; potential weed management issues	(Nguyen & Sukias 2002; Nguyen et al. 2002b)
Managed or controlled drainage	manipulation of water table to temporarily retain drainage waters to promote denitrification	with and without effluent irrigation	subsurface drainage	denitrification	paddock	L	N	L	L	M-L	L-M	b: soil water storage; flood attenuation d: requires active management	(Singleton et al. 2001)
Sediment traps, dams & ponds													
Sediment trap			surface runoff	deposition	paddock, farm	M	Sed, P	L	L-M	M	L	b: can reduce drain clearance costs d: may alter drain hydraulics	(Hudson 2002)
Dams and ponds	sedimentation ponds (outlet throttled)		surface runoff in ephemeral channels, stream flow diverted during floods	flow attenuation; deposition; solar disinfection	paddock, farm, catchment	M	Sed, P, faecal microbes	M	L-M	M-H	M	b: stock water supply; duck shooting; flood attenuation; and improve landscape aesthetics d: can have negative impacts on downstream flows, water temperatures and dissolved oxygen impacting aquatic life.	(Sinton et al. 2002)
Wetlands and algae and plant harvesting													
Aquatic plant/algae uptake and harvesting	harvested beds of watercress or other aquatic macrophytes or filamentous algae		springs, stream flow	nutrient uptake + deposition, filtering, denitrification	paddock, farm, catchment	L	N, P	L	L	M-L	H	b: forage crop for stock d: requires active management	(Cox 2004; Craggs 2002; Craggs et al. 1996a; Howard-Williams et al. 1982; Howard-Williams & Pickmere 1999; Howard-Williams & Pickmere 2005; Tanner 1996; Tanner 2001a; Vincent & Downes 1980)
Natural seepage wetlands	seeps flowing via wetlands at edge of streams		subsurface flow + some surface runoff	denitrification, nutrient uptake, deposition, mineralisation, adsorption	paddock	H-M	Sed, N	M	M	M-H	L	b: aquatic habitat and biodiversity; improve landscape aesthetics; recreational hunting, cultural harvesting of flax and other plants; flood attenuation; water storage d: potential weed management issues	(Burns & Nguyen 2002; Matheson et al. 2002; Rutherford & Nguyen 2004; Sukias & Collins submitted)

Attenuation tool	Description	Variant(s)	Intercepted flowpath(s)	Primary attenuation processes	Scale(s)	Likely applicability in NZ	Target pollutants	Landscape fit	Knowledge level in NZ	Efficacy	Cost	Other indirect benefits /disbenefits	Key NZ references
Floodplain wetlands	stream flood flows intercepted by riverine wetlands, meanders, oxbows, billabongs, lagoons, deltas etc.	natural, created or restored	floods	flow attenuation, deposition, nutrient uptake, denitrification	farm, catchment	M-L	Sed, P, N	M	L-M	M-L	L-H	as above	
Constructed wetlands	artificial wetland created on key flowpaths	treat drainage or effluent; constructed, facilitated or bottom of catchment	stream flow, tile drain flow, surface drains	denitrification, nutrient uptake, deposition, adsorption, mineralisation	paddock, farm, catchment	M	Sed, N	U	H	M	M-H	as above	(Sukias et al. 2005; Tanner et al. 2005a; Tanner et al. 2005b; Tanner et al. 2005c)
Floating wetlands	wetland plants growing in a floating mat on the surface of the water	deployable in wetlands, dams, lakes, ponds	standing surface water	denitrification, deposition, nutrient uptake	catchment	L	N, P	L	L	M	H	b: able to cope with fluctuating water levels; can be used to improve attenuation in ponds, dams and irrigation storage reservoirs	Lake Rotouehu (EBoP) and Hamilton Lake (HCC) trials. (Headley & Tanner 2006; Headley & Tanner 2007a)
Permeable filters and materials													
Denitrification walls	addition of sawdust to soil to produce suitable conditions for denitrification		subsurface flow, surface drains, tile drains	denitrification (adsorption + immobilisation in short-term)	paddock, farm	L	N	L	M	M-H	H	b: below-ground so little reduction of usable grazing land	(Schipper & Vojvodic-Vukovic 1998; Schipper et al. 2005; Schipper et al. 2004; Schipper & Vojvodic-Vukovic 2000; Schipper & Vojvodic-Vukovic 2001)
Permeable reactive filters	addition of organic carbon-rich sources to produce conditions suitable for denitrification	woodchip or straw-bale filters	subsurface drains	denitrification (adsorption + immobilisation in short-term)	paddock	M	N	L	M-H	H	H	d: discharges from C-rich filters may initially have elevated BOD and humic colour	(Sukias et al. 2005; Sukias et al. 2006a; Sukias et al. 2006b)
Reactive materials	addition of reactive materials to flowpaths, critical source areas and/or effluent irrigation, riparian and wetland soils	<ol style="list-style-type: none"> filter sock addition to tile drains addition to wetlands alum addition zeolite addition 	<ol style="list-style-type: none"> stream flow tile drains natural, facilitated and constructed wetlands surface runoff soils, or porous filters for tile or surface drain flows 	adsorption, precipitation	<ol style="list-style-type: none"> catchment paddock paddock, farm or catchment paddock paddock 	M-L	P, (zeolite also NH ₄)	M-L	L	M-L	H	<p>b: in soil or below-ground so little visual impact or reduction of usable grazing land; loaded materials may be able to be reused as slow-release fertilisers or as aggregates on farm raceways</p> <p>d: instream filters may affect water quality and aquatic habitat; need to be close to suitable source of reactive materials; relatively expensive to retrofit existing drainage systems; likely to require periodic replacement or rejuvenation of materials; fly ash results in caustic discharge- alkalinity must be reduced to be useful.</p>	<ol style="list-style-type: none"> (McDowell et al. 2007) (McDowell et al. in press) (Pratt et al. 2007; Shilton et al. 2006; Sukias et al. 2006b) McDowell pers. comm. (Bolan et al. 2003; Bolan et al. 2004; Nguyen 1997a; Nguyen et al. 1998)

4.1 Reducing hydrologic connectivity

The basis for these tools is the simple concept of decoupling the movement of surface runoff from pasture to a water body. By using water sensitive paddock and farm design natural hydrologic buffers can be introduced. For example, the re-siting of gates away from high runoff risk areas such as the bottom of a slope or near a stream. Increased activity occurs around gates, including trampling by livestock and compaction by farm machinery. Repositioning gates away from these sensitive areas reduces the risk of sediment, P and faecal losses being entrained by surface runoff and transported to streams or drains. Similar principles can also be applied to locating troughs and farm tracks and races. These tools are at the ‘proof of concept’ stage of development (Figure 11) and no pilot scale trials have yet been completed locally or overseas.

4.2 Riparian management

Riparian management can provide a buffer zone between a paddock and stream. Water quality can be improved by (1) filtering of pollutants from surface runoff, (2) removing nutrients from shallow subsurface flow by nutrient uptake, and biogeochemical transformations, (3) stabilising stream banks and reducing erosion, and the (4) reducing sediment and nutrient supply (e.g., livestock grazing) close to streams. Riparian management can have the additional benefits of providing terrestrial and aquatic habitat, moderating stream water temperatures, providing recreational areas and enhancing stream aesthetics.

Livestock exclusion is the most basic form of riparian management and can improve water quality via functions (3) and (4). The addition of a grass filter strip (GFS), a managed band of dense grass to filter surface runoff (function 1), will provide additional benefits if surface runoff is an important flowpath. Riparian buffers (fencing & planting; Table 2) can have the additional benefits of plant uptake and may also promote conditions suitable for processes such as infiltration and denitrification. All four riparian management functions are unlikely to operate simultaneously in any given environment, unless a sequential riparian buffer is used. In addition the conditions for some processes will be incompatible (e.g., denitrification and P adsorption in shallow subsurface flow; Table 1).

There has been no research on plant uptake rates in riparian zones in New Zealand, and nutrient biogeochemical research has focused on riparian wetlands (see section 4.7). Therefore cost-efficacy calculations can only be evaluated for grass filter strips and livestock exclusion.

4.2.1 Livestock exclusion

Livestock exclusion is highly applicable to New Zealand livestock farming and is suitable for stream, lake, water race, wetland and estuary margins. The target pollutants are sediment, nutrients and faecal microbes (Table 2).

Incursion by livestock into streams may cause immediate damage to the stream banks and bed, and degrade water quality. Livestock can increase stream bank erosion directly through damage arising from trampling (Stassar & Kemperman 1997; Trimble & Mendel 1995), or indirectly through damage to vegetation. The susceptibility of banks to damage by grazing livestock depends on many factors, including bank height, bank water content and channel stability (Trimble & Mendel 1995; Williamson et al. 1992). Faecal material and urine may be deposited directly into the water by livestock, immediately elevating concentrations of faecal microbes, nutrients and particulate matter (e.g., Davies-Colley et al. 2004). Future events, such as storm events or additional livestock incursions can remobilise bed deposits of faecal microbes, particulate material and nutrients. Incursion of livestock thereby causes both immediate and long-term damage.

Damage from livestock access depends on: (1) livestock management in riparian areas (particularly whether or not strip-grazing or mob stocking is practiced) and (2) numerous site characteristics, particularly soil texture and drainage, topography, and climatic factors (McKergow & Hudson 2007). There is evidence that livestock access to streams causes long-term geomorphic and aquatic habitat damage at farm average stocking rates (for cattle and sheep) above 4 SU/ha (McKergow & Hudson 2007). At the paddock scale, any incursion into the stream by cattle can elevate microbial water quality above microbiological contact recreation guidelines (McKergow & Hudson 2007).

Overseas catchment-scale studies have shown that riparian fencing can appreciably reduce sediment and *E. coli* yields, while nutrient reductions vary depending on nutrient sources, forms and flow paths (Line et al. 2000; McKergow et al. 2003; Meals & Hopkins 2002; Owens et al. 1996). These studies used a “before and after” experimental design and surmised that the changes in concentrations and loads were attributable to riparian fencing. Suspended sediment loads were reduced by between 30 and 90% in a range of environments (Line et al. 2000; McKergow et al. 2003; Owens et al. 1996) and the reductions were attributed to reduced streambank erosion. Riparian fencing can reduce stream *E. coli* concentrations by 30 to 65% after fencing (Line 2003; Line et al. 2000; Meals & Hopkins 2002). Significant reductions in phosphorus loads have been reported in some catchments following riparian fencing (e.g., Line 2002), while other research has identified a change in the dominant form of

phosphorus transport from particulate to dissolved forms following reductions in sediment export (McKergow et al. 2003).

The effect of livestock exclusion cannot be separated from the impacts of other attenuation tools in New Zealand catchment scale studies (Wilcock et al. 2007; Williamson et al. 1996). For example, livestock exclusion is just one of several mitigation tools being implemented in the Dairy Focus catchments (see Wilcock et al. 2007), and therefore cannot be isolated from other water quality improvements in these catchments.

A growing number of stream reach scale studies demonstrate that unimpeded livestock access degrades water quality in both dry and wet channels (Table 13 in Appendix 1). Most reach-scale studies use an upstream-downstream experimental design to assess the impacts of direct livestock access on water quality. Elevated suspended sediment, turbidity, nitrogen, phosphorus and *E. coli* concentrations and exports have been measured when dairy cattle access streams in large numbers (e.g., herds fording streams on the way to milking) and when cattle and deer access streams or ephemeral channels during grazing (Table 13 in Appendix 1). The impact of livestock exclusion on seepage wetland condition and water quality is being monitored in a Lake Taupo sub-catchment (McKergow et al. 2007b). Preliminary data suggests that organic N and total N exports are 5-10 times higher when stock are grazing the wetland compared to baseflow conditions without grazing.

For hill country sheep and beef farms alternatives to livestock exclusion by fencing could be worth investigating. The provision of off-stream shade, troughs and supplements have been evaluated in the United States and the results are often not consistent (see review by Agouridis et al. 2005). Installing off-stream water in Virginia reduced SS and TP loads by > 90% and halved the faecal coliform load (Sheffield et al. 1997), while an increase in TP and no change in SS loads was reported in response to off-stream water in North Carolina (Line et al. 2000). A study on steep hill country in the Waikato found that the presence of a trough did not influence cattle use of the stream or riparian zones (Bagshaw et al. 2008).

Sufficient data is available to estimate the efficacy of livestock exclusion for permanently flowing stream channels (Table 3). For suspended sediment and particulate nutrients, livestock-induced bank erosion may be a significant part of sediment export. For suspended sediment, the range of catchment scale load reductions is large, and so lower (30%) and upper (90%) bounds on export reductions are given.

Direct inputs from dung and urine alone were used to estimate the TN, TP and *E. coli* loadings that could be eliminated from streams with livestock exclusion. Direct

deposition reductions were calculated at the farm scale, assuming an average channel density for the farm, and that cattle have access to channels all of the time. Computationally this is identical to having all channels on $\frac{1}{2}$ of the farm ($2 \times$ average channel density) and cattle accessing channels $\frac{1}{2}$ of the time. Dung inputs were estimated assuming that 1% of pats are deposited in streams, each cow deposits 13 pats/d and the average pat size is 1.0 kg. Nutrient concentrations of 27 gTN/kg dry weight and 5.5 gTP/kg dry weight and 10.2% solids content were used to estimate the nutrient mass added by direct dung deposition (McDowell & Stewart 2005). The measured range of *E. coli* concentrations in cattle dung is large and so minimum and maximum values were taken from Wilcock (2006) and Davies-Colley et al. (2004), respectively. For the urine N input it was assumed that 1% of urinations occur in-stream, each urination is 2.2 L, there are 12 urinations per day (Williams & Haynes 1994) and the N content ranges between 1 and 13.5 mg/L (Keith Betteridge, AgResearch, pers. comm.).

Two permanent fencing options were used to cost livestock exclusion – 2 wire electric for dairy and 5 wire (3 electric) for sheep/beef. For hill-country sheep and beef a comparison is made between 5 wire (3 electric) and 8 wire post and batten fences (see Section 5.4). The total length of fencing required (both banks) was estimated by doubling the stream channel density. Stream network density can be estimated using either ground mapped or existing topographic data and is typically in the range of 17-35 m/ha. The calculations assume that (1) livestock have easy access to the channels, (2) no exclusion fencing is currently in place and (3) half of the 2 m width to be excluded on each streambank is unstable and un-productive land. The costing does not include the provision of off-stream water supplies.

Table 3: Riparian management tools performance and costing.

Intercepted flow-path	Application sites	Situations where likely to be of significant benefit.	Target contaminant	Variant	Length of fencing m/ha of catchment	Area requirement of attenuation system/ ha	Sediment reduction range	N reduction range	P reduction range	E. coli reduction range	Attenuation system set-up costs (\$/ha of catchment)	Attenuation system maintenance costs (\$/ha of catchment/y)	Notes on assumptions
					m/ha	m ² /ha	g/m/y or g/m ² /y or %	g/m/y or g/m ² /y or %	g/m/y or g/m ² /y or %	%	\$/m or \$/ha of system	\$/m/y or \$/ha/y of system	
Livestock exclusion from stream channels													
Primary function is to remove sources of livestock faecal matter, and treading and pugging damage from waterways, waterbodies and their margins.	Edges of streams, creeks, drains, wetlands, dams, ponds and lakes, within 1-2+ m of stable bank edge.	Areas with high densities of stream and surface drainage channels, and soils susceptible to treading damage and erosion.	sediments, P, N, pathogens	Livestock excluded from stream channels	17-30 m/ha per bank	2 x average channel density m/ha for both banks to be fenced**	30-90% at catchment scale	direct: g TN/cow/m channel/y dairy 0.05-0.43 intensive beef: 0.03-0.22 hill beef: 0.01-0.09	direct: g TP/cow/m channel/y dairy: 0.005-0.008 intensive beef: 0.004 hill beef: 0.014	20-35% direct: E. coli/cow/m channel/y 10 ⁵ -10 ⁸	1. fencing options: sheep/beef 5 wire (3 electric); dairy 2 wire electric 2. Land removed from production 2 m ² per m channel length for both banks fenced**	assume 1 weed spray per ha/y	1. assumes 2 m fenced margin. 2. assumes easy stock access to stream 3. direct deposition reductions calculated at farm scale. Assumes that channel density is an average over farm and cattle have access to a channel at all times. Computationally this is identical to having all channels on 1/2 of the farm and cattle accessing the channels 1/2 of the year. 4. dung inputs: 1% of pats deposited in stream, 13 pats/d/cow, 1.0 kg/pat, wet weight conc from McDowell and Stewart (2006); E. coli concentrations low range from Wilcock (2006) and high end from Davies-Colley et al. (2004). 5. urine inputs: 1% of urine in stream, 12 urinations/d/cow, 2.2L/urination (Haynes and Williams, 1994), urine N concentration range 0-13 mg/L (Keith Betteridge, pers. comm.) 6. does NOT include livestock induced bank erosion for TP. 7. SS does include all sources - i.e., at catchment scale 8. off-stream water supply costs are not included ** assumes that 1/2 of the bank being fenced is not productive - i.e., heavily grazed/bare soil
Riparian grass filter strip													
Surface run-off entering streams, surface drains, wetlands and lakes	Riparian zones around streams, creeks, drains, wetlands, dams, ponds and lakes	Low-moderate permeability soils, moderate to steep slopes, and climate zones with high intensity rainfall, where surface runoff (IEOF) is a significant pathway of contaminant transport.	sediments, P, N	Riparian Filter Strip permeable soils (low clay content) channelised flow through riparian zone.	17-30 m/ha	10 x average channel density (m ² /ha)*	20-30%** + livestock exclusion load***	10-20%** + livestock exclusion load***	15-30%** + livestock exclusion load***		1. fencing options: sheep/beef 5 wire (3 electric); dairy: 2 wire electric	assume 1 weed spray per ha/y	1. most applicable where surface runoff occurs in moderate-high amounts e.g., below sloping, surface sealed (e.g., compacted or high clay content soils) pasture 2. assumes dense grass filter strip, well managed, with upslope edge on contour 3. assume filter is grazed twice annually to maintain grass cover 4. assume that livestock exclusion benefits are also applicable to this scenario - i.e., 2 attenuation functions for a small increase in cost * average filter width is 10 m on both banks. The actual width may vary above 2 m depending on local topography. No planting - retired pasture. ** assumes 50% of surface runoff bypasses filter strip in concentrated channels *** these benefits are independent as they are different flow paths. They MUST be calculated independently and then summed.
				Riparian Filter Strip permeable soils (low clay content) planar slopes encouraging even flow through filter strip.	17-30 m/ha	10 x average channel density (m ² /ha)*	40-80% + livestock exclusion load***	20-40% + livestock exclusion load***	30-60% + livestock exclusion load***	fencing options: sheep/beef 5 wire (3 electric); dairy: 2 wire electric	assume 1 weed spray per ha/y	as above, except for unchanneled sheet-flow, limited bypassing of filter strip	
				Riparian Filter Strip low permeability soils (high clay content) planar slopes encouraging even flow through filter strip	17-30 m/ha	10 x average channel density (m ² /ha)*	40-50% + livestock exclusion load***	10-20% + livestock exclusion load***	20-40% + livestock exclusion load***	fencing options: sheep/beef 5 wire (3 electric); dairy: 2 wire electric	assume 1 weed spray per ha/y	as above except for low permeability soils greater clay fraction (harder to trap - slower to settle out of the water column)	

4.2.2 Grass filter strips

A grass filter strip (GFS) is a band of managed grass that provides a buffer between possible contamination sources and a water body. Grass filter strips are designed to intercept surface runoff during rainfall or irrigation events and the key pollutant removal processes include deposition, physical filtering, and infiltration (Table 1). Careful consideration of the landscape characteristics are required in order to effectively intercept contaminants travelling from paddocks into streams as surface runoff. For example, a riparian GFS will be little or no use for attenuation if water mainly moves through the riparian zones as shallow subsurface flow or bypasses the riparian zone via drains, ephemeral channels, or deep groundwater paths.

Grass filter strips can take several different forms, including riparian GFS, in-paddock GFS, grass hedges and grassed ephemeral waterways (Table 2). Riparian and in-paddock filter strips are strips of grass along topographic contours and are designed to filter pollutants from surface runoff moving as sheet flow. Grass hedges are narrow, stiff-stemmed hedges placed where concentrated flows occur. Grassed waterways or swales are strips of grass in flow convergence zones (e.g., gully bottoms) primarily installed to convey excess water and use dense grasses to stabilise the soil surface against erosion. Grass filter strips may also be placed upslope of plantings in sequential riparian buffers.

The majority of research on filter strips comes from cropped land where they can be an effective water quality tool, significantly reducing sediment loads and concentrations in surface runoff (Dosskey 2001). Filter strips have been tested on land draining pasture, either with or without manure/effluent additions. Typically between 40 and 80% of the suspended sediment load is retained, but the variability in performance is large (e.g., Magette et al. 1989; Schellinger & Clausen 1992; Smith 1989). Nutrients can also be removed from surface runoff, but the nutrient form will affect removal. Removal of particulate or sediment-associated nutrients from surface runoff is generally lower than that of sediment (e.g., Magette et al. 1989; Smith 1989). Significant amounts of particulate P can be removed from surface runoff if it is associated with large particles with short settling times, but if P is moving with clay particles or colloids then longer settling times are required. Dissolved pollutants transported by surface runoff (e.g., nitrate, FRP) are generally reduced the least. Filter strips may also be useful for reducing faecal microbe concentrations, although load and concentration reductions can vary between 0 and 99% and efficacy decreases with increasing flow (Collins et al. 2004; Tate et al. 2006; Tate et al. 2004).

Grass filter strips have been trialled on pasture in New Zealand. Smith (1989) established retired pasture filter strips on a Waikato drystock farm and monitored them for two years. Flow weighted mean concentrations were reduced by between 40 and

50% for most parameters. Smith (1989) does not include data on infiltration of surface runoff, and suggests that the high trapping of dissolved nutrients (e.g., $\text{NO}_3\text{-N}$) was probably due to a reduced supply of nutrients within the retired strips. Collins et al. (2004) conducted a series of experiments to evaluate the ability of filter strips to retain faecal microbes from effluent and concluded that trapping was a function of flow rate. Under high flow rates (13 L/min) trapping varied between 0-85%, while at low flow rates (4 L/min) trapping was much greater (>95%). The grass filter strips were a temporary store for some of the microbes trapped - they were mobilised and washed out of the filter strip in a subsequent event, 5 days later.

Current GFS research projects are examining alternative forms of grass filter strips on pasture. In-paddock grass filter strips at the downslope margins of intensively grazed blocks are being trialled at Lake Rerewhakaaitu (dairy) and Kaharoa (drystock). The key advantages of this approach are the close proximity of the GFS to pollutant sources and little or no flow convergence, providing conditions suitable for high pollutant removal. Surface runoff occurs mainly during the winter months, concentration reductions vary between 20 and 70% for SS, TN, TP and *E. coli* and load reductions are typically less than 15%.

In steeper hill country with convergent flow paths, several different GFS forms may be appropriate. Natural swales are suitable locations for grass filter strips as flow naturally concentrates in these areas. However, channels, either natural or stock generated, are likely in natural swales and so GFS must either be designed to spread flows or withstand concentrated runoff. Filter strips with flow spreaders were trialled on an ephemeral channel near Rotorua (Ledgard et al. 2007). Runoff was rare and volumes were low, but the trial appeared to reduce suspended sediment and particulate P concentrations. A grass hedge trial has commenced near Rotorua recently (McKergow et al. 2007c).

Many factors influence GFS performance and three variants are presented in Table 3. Performance depends on soil characteristics, sediment size and load, width, slope, slope length, vegetation type and density, duration of rainfall, flow rate, and the propensity of flow to channelise (Barfield et al. 1998; Dillaha et al. 1989; Magette et al. 1989; Wilson 1967). Maximum pollutant trapping by riparian GFS can be expected when surface runoff is uniformly dispersed across the strip and does not concentrate into channels (Dosskey et al. 2002). Once runoff concentrates into channels a large amount of the surface runoff is able to bypass the GFS. For example, Dosskey et al. (2002) modelled sediment removal by GFS on four farms and estimated removals in the range 41-90% from paddock runoff for uniformly distributed runoff. However, because of topographically driven non-uniform runoff, only 15 to 43% would actually be removed.

The efficacy ranges in Table 3 are for a dense grass filter strip that is well managed and has its upper edge on the contour. The average width is assumed to be 10 m (per bank), but the actual width may vary from a minimum of 2 m (livestock exclusion) where there is little risk of surface runoff to 20-30 m where there are landscape features such as ephemeral channels. In addition to filtering functions, riparian grass filter strips on livestock farms have the additional benefit of livestock exclusion (because they require fencing to maintain the grass cover). For these multiple benefits the efficacy is calculated independently for each benefit and then the two load reductions are summed, i.e., total reduction = livestock exclusion (30-90% reduction in catchment SS export) + riparian GFS (40-80% reduction in surface runoff SS load). Biomass removal from GFS is required and it is assumed that this is done twice a year, along with one weed spray per year. No planting is required as the GFS is retired pasture.

Grass filter strips must be managed to maintain their functioning. Current advice is to lightly graze (preferably with sheep) or if the terrain permits summer haymaking (Quinn & McKergow 2007). This is probably best timed in early spring (to stimulate spring growth) and in autumn (early enough to ensure re-establishment of a good cover before winter). Stock grazing must be managed to minimise treading damage and compaction of the soil, erosion of streambanks, and direct animal input of nutrients and pathogens to the stream and filter strip. Just how this is achieved will vary with the situation. This grazing also needs to be timed in a fine weather window to allow some regrowth before a rain event.

4.2.3 Riparian buffers

A riparian buffer is a band or zone of managed vegetation between the pasture and a water body. Riparian buffers may provide conditions suitable for all four water quality functions (see section 4.1). Riparian buffers can be designed to fulfil a range of additional functions including, modifying stream temperatures and light, inputting organic debris, enhancing fish, invertebrate and bird communities, providing recreational areas and enhancing flood defence. Typically riparian buffers are planted with trees, and native species are commonly planted in New Zealand.

Riparian buffer design will depend on the desired range of functions and the landscape. For example a buffer designed to include recreational facilities might be considerably wider than one designed to modify stream water temperature. Riparian buffers designed to improve water quality must consider whether the buffer can meet the specific conditions required for the desired attenuation processes (see section 2) rather than using a prescribed width. For example, removal of nitrogen from subsurface flows by denitrification is a function of soil type, subsurface hydrology,

and subsurface biogeochemistry (organic C, high nitrate) rather than buffer width per se.

If the main attenuation process for soluble particulate nutrients is by adsorption onto soil particles, then there is potential for the soil P adsorption sites to become saturated with time (e.g., Cooper et al. 1995; Dillaha et al. 1989). In order to reduce the likelihood of P saturation the sustainable net removal capacity of the buffer needs to match the nutrient influx from upslope (Cooper et al. 1995). Actively growing vegetation and active soil microbial systems will help to maintain active soil adsorption sites (by removing bound P) so there is a need to manage the system to maintain healthy soils and growing conditions. Here periodic short-term grazing or other harvesting is desirable.

4.2.4 Gaps in knowledge or communication

Separation of livestock from water can improve water quality. However, there are some key research gaps (Table 4):

- Little data exists on nutrient reductions due to direct livestock deposition and current research projects in New Zealand cannot fill this gap.
- Exclusion could be beneficial beyond permanent stream margins and on streams that are smaller than those included in the Clean Streams Accord. Seasonal increases in flow and channel network expansion may increase the probability of livestock access to surface water. Stream channel network expansion will occur in some New Zealand landscapes and in these situations fencing of channels beyond the permanent stream network in winter may be required to reduce the probability of livestock incursion into flowing water.
- For low intensity sheep/beef farms there is a need to find alternatives to fencing (e.g., shade, salt licks, off-stream watering) at stocking rates that have minor damages. A recent review for Environment Canterbury (McKergow & Hudson 2007) revealed that little data exists to guide livestock exclusion policy development to achieve water quality outcomes on drystock farms.

Priorities for GFS research are evaluating the risk of surface runoff and the degree of channelisation in order to target GFS where they will be most effective (Table 4).

Table 4: Communication and science gaps for riparian management.

Attenuation tool	Communication status and gaps	Science gaps	Availability in NZ farm-scale modelling tools (OVERSEER® and NPLAS)
Livestock exclusion	<ul style="list-style-type: none"> Available: Regional council clean streams booklets/ guidelines + DEC (2006) Gaps: – need to include quantitative examples of reductions expected 	<ul style="list-style-type: none"> seasonal expansion of channel networks (including drains) on farms – measured for range of landscapes stocking rate/practices with minor impacts for range of landscapes and stream types loads from grazed saturated areas and/or ephemeral channels grazing behaviour information and assessing ways to alter this behaviour stream order/size – efficacy relationships 	
Grass filter strip	<ul style="list-style-type: none"> Available: Regional council guides + DEC (2006) general guidelines Gaps: Need for detailed practical guidelines for identification of surface runoff risk and assessment of paddock scale suitability for GFS. 	<ul style="list-style-type: none"> importance of surface runoff as a flow and pollutant pathway in various landscapes pasture efficacy – including particle size analysis & grazing relationships importance of channels on pasture and impact of grazing on sheet flow infiltration capacity changes without grazing P saturation of soil in GFS over time maintenance regimes - grazing/cut and carry in practice performance of "real" filter strips vs experimental pathogen loads and removal (cf <i>E. coli</i>) 	<ul style="list-style-type: none"> included in OVERSEER® (under development) but without the effect of riparian livestock exclusion included in NPLAS
Riparian buffer	<ul style="list-style-type: none"> Available: Regional council guides + DEC (2006) general guidelines + Landcare Research posters on native plant growth 	<ul style="list-style-type: none"> relative importance of plant uptake by riparian trees/shrub species, especially natives management regime required (weed control, biomass harvesting) flowpath identification to better target riparian buffer development. GHG implications and trade-offs 	<ul style="list-style-type: none"> included in NPLAS

4.3 Drainage system manipulation

4.3.1 Controlled drainage

Manipulation of the water table by controlling subsurface drain discharge (e.g., Figure 12) can increase pollutant attenuation in the soil column. It can also be used to attenuate flow peaks entering downstream attenuation systems, such as reactive filters or constructed wetlands. Temporarily maintaining higher water tables below pastures using these practices tends to reduce drainage water outflows by promoting deep and lateral seepage through the soil, unless there is a highly impermeable layer underlying the soil. Extensive data from North Carolina have shown that controlled drainage can reduce drainage water flows by 30%, N losses by 45% and P loss by 35% (Gilliam et al. 1999). Decreasing the depth of drain placement, whilst increasing the drain density is another option to increase soil water storage capacity, but requires closer spacing of drains to maintain drainage function, thus increasing drain installation costs. Modelling studies by Skaggs and Chescheir (2003) and field trials by Burchell et al. (2005) in the USA suggest that shallow drains can reduce both water and nitrogen losses. Such practices to increase soil water storage capacity and promote slow seepage through the soil, can both reduce contaminant losses and enhance water availability for subsequent pasture growth (Gilliam et al. 1999).

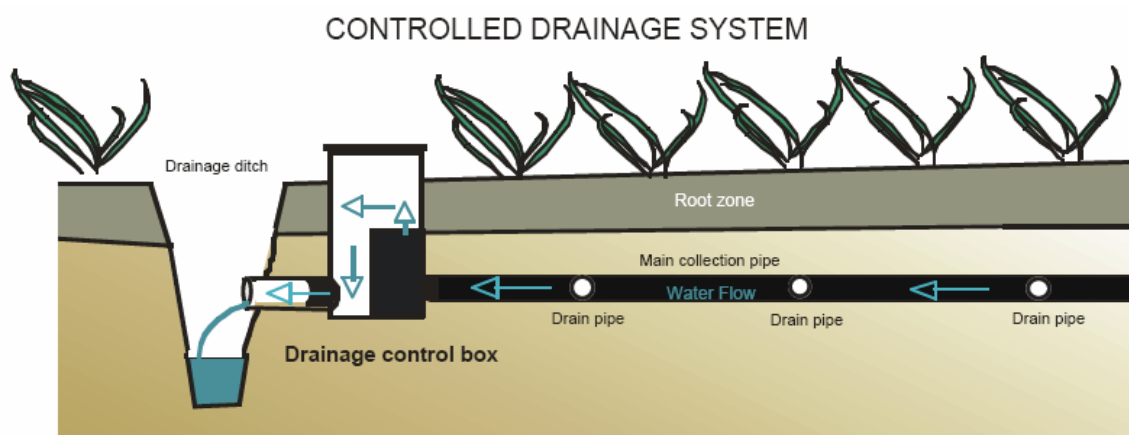


Figure 12: Cross section of a controlled drainage system enabling drainage depth to be seasonally adjusted via a control box with an adjustable weir (from Spillman 2002).

Elevated levels of nutrient losses have been recorded in NZ for drained soils irrigated with wastewaters (Houlbrooke et al. 2004; Monaghan & Smith 2004; Singleton et al. 2001), and these are an obvious target for application of controlled or managed drainage systems. Controlled drainage is probably un-suited to mole-tile drained land as the moles may collapse under saturated conditions. In a study in Waikato, Singleton et al. (2001) investigated managing water tables on gley soils receiving effluent at the plot scale. The managed drainage treatment in which the water table was nearest the

soil surface resulted in less N being leached as nitrate (<2.5 kgN/ha/y) than the other drainage treatments (6-12 kgN/ha/y), but more organic and total N was leached (33 - 131 kgN/ha/y). In general, the managed drainage regimes trialled had limited effect on the amount and forms of N leached from the gleyed silt loam tested, due to rapid leaching through macropores. This suggests that this tool is unlikely to be useful in soils with preferential flow pathways, but further investigation may be warranted to evaluate potential benefits of controlled drainage in other situations.

4.3.2 Vegetated drains

The subsoil sediments exposed during excavation and maintenance of surface drains, and the sediments that accumulate within them, generally have considerable P adsorption capacity. Nguyen and Sukias (2002) found evidence that sediments in a range of NZ surface drains were acting as a semi-permanent sink for P from drainage water. About half of the P within surface drain sediments at 26 sites was loosely-bound (non-occluded Al/Fe-P or carbonate-bound P, 44-57%). A tracer test in a Waikato farm drain using a solution containing ammonium and phosphate showed 44% and 56% removal, respectively, over a distance of 150 m in a vegetated drain during a 5.5 h period (Nguyen et al. 2002b). Specific removal processes were not examined, but adsorption and absorption into sediment and nutrient uptake by plants and microbial biomass were considered likely mechanisms. Similarly, in studies of no-till cropped lands in Mississippi annual reductions of 44% of P loads and 57% of N loads were recorded during passage along drainage ditches with variable vegetative cover (Kroger et al. 2007; Kroger et al. 2008). Longer-term studies are needed to determine whether such attenuation is just temporary or longer-term, and how it may be enhanced under various flow conditions and drain management regimes (e.g., mechanical or herbicidal drain clearance).

Many surface drains only flow seasonally or in response to rainfall events, so experience a complex series of wetting and drying phases. They are also commonly subjected to periodic mechanical clearance and/or herbicide treatments, which together affect their capacity to take-up, transform or release nutrients (Barlow et al. 2006; Barlow et al. 2004; Barlow et al. 2003). Further work under New Zealand conditions is required to understand nutrient attenuation processes and quantify removal rates in surface drains, and how these may be modified so that they operate somewhat like long linear wetlands. Allowing establishment of vegetation in surface drains and/or increasing their residence time and retention of organic sediments using a series of low weirs has the potential to increase their capacity for pollutant attenuation (Bowmer et al. 1994). However, most existing drainage networks have been engineered solely with the aim of effectively draining excess soil water and their capacity to accommodate the increased hydraulic resistance of plants without causing

flooding is likely to be limited. Widening of vegetated sections of drainage channels near discharge points may be necessary to accommodate required flows.

4.4 Sediment traps, dams and ponds

Hudson (2002) has investigated the potential benefits of coarse sediment traps under NZ conditions and reviewed general design information. Coarse sediment traps are essentially excavations in the bed of a watercourse designed to settle and trap coarse particles (mainly sands and gravels). Such traps, based on rule of thumb design (1.5 times channel width, 10 trap widths long, and excavation to 1.5 m below existing channel base), are expected to remove 90% of fine sand. Sediment removal from the trap is likely to be required once or twice a year depending on sediment loads. Concentration of the retained sediments within the trap can substantially reduce drain management costs that would otherwise be incurred for clearance along the whole length of the drain.

Farm dams or retention ponds are a relatively low-cost option to capture runoff, sediments and associated contaminants. Stored run-off can later be used during dry periods as a source of irrigation and livestock drinking water, be allowed to infiltrate or released slowly to attenuate flood flows. Little research on farm dams or retention ponds as attenuation tools has been conducted in NZ, although anecdotal observations suggest sediment accumulation can be high. Dendy (1974) found sediment trapping efficiencies of 81-98% for 17 small flood-water retarding reservoirs in southern and western USA. Cooper and Knight (1990), studying a small reservoir on an ephemeral Mississippi creek over a five year period, found overall trapping of 77% of incoming suspended sediments, and 72% trapping of TP and 82% of nitrate-N during storm events. Net removal rates of 65-550 mgN/m²/d were recorded under base-flow conditions for 11 constructed ponds in Pennsylvania (Fairchild & Velinsky 2006). Nitrate-N was strongly removed, while particulate N (largely algae) and ammonium N (initially at low levels) increased moderately. The ponds sometimes increased downstream TP concentrations, mainly due to increases in particulate P (largely algae) and dissolved organic P.

Sediment trapping efficiency will depend on the size of the reservoir relative to its inflows, its shape and outflow configuration, and the size distribution and nature of incoming particles. In general, for high removal efficiencies (> 80%), systems receiving suspended silt-sized particles should have volume/average annual inflow ratios between 0.1 and 1, with event detention times approaching 1 day (Dendy 1974; Griffin 1979). Studies of N removal from nitrate-dominated agricultural streams in ponds and lakes in Sweden (Fleischer et al. 1994; Jansson et al. 1994) found residence times of 2-4 days were required for moderate N removal, with removal efficiency substantially higher in vegetated systems. Recently, significant information has been

accumulated on sediment and associated pollutant trapping in ponds receiving urban run-off (ARC 2003; Marselek et al. 2005; Schueler 1992), and, despite significant differences in contaminant sources, many of the principles employed and advances made also have relevance for treatment of agricultural run-off.

Collectively, large numbers of small ponds can have a significant effect on the hydrology and sediment yield from landscapes, affecting rates of water and material transport, elevating evaporative water loss, and altering rates, pathways and locations of biogeochemical processes (Smith et al. 2002). As well as creating potential barriers to fish passage, small impoundments can also significantly affect downstream habitat by modifying and fragmenting stream flow regimes, and modifying downstream physico-chemistry. In NZ studies of 6 on-line constructed ponds in the Auckland region, elevated temperatures and low dissolved oxygen concentrations were found to have significant negative impacts on stream macroinvertebrate communities hundreds of metres downstream (Maxted et al. 2005). Conversely, in many overseas studies such ponds are often considered as important sources of biodiversity in agricultural landscapes (Cereghino et al. 2008; Davies et al. 2008). Potential impacts on downstream aquatic life therefore need to be seriously considered when ponds or other impoundments such as wetlands are planned.

4.5 Wetlands

Wetlands occur in a wide range of landscapes and may support permanent or temporary standing water. They have soils, substrates and biota that have developed under conditions of waterlogging and restricted aeration. Wetlands utilised for nutrient attenuation may be natural or constructed. Natural wetlands on farms are often drained, either with tile drains or surface drains, and converted to pasture. Constructed wetlands attempt to replicate and optimise treatment processes that occur naturally in swamps, fens and marshes. Treatment efficiency is enhanced by optimising dispersion, flow paths, water depths, residence times, and vegetation characteristics. Construction and operating costs for simple constructed wetlands are relatively low providing suitable land is available, and enhancement of biodiversity and landscape aesthetics provides ancillary benefits (although potential effects on fish passage and downstream physico-chemistry also need to be taken into account, as noted above for ponds, section 4.10). Facilitated wetlands are a subset of constructed wetlands that use natural landscape features, such as depressions, to reduce construction costs. There are a wide range of possible different constructed wetland types. Here we focus on those most applicable to cost-effective treatment of diffuse agricultural flows; i.e., surface-flow wetlands, floating treatment wetlands, and harvested aquatic plant systems.

Wetlands are important nitrate and sediment attenuation tools. Wetlands can provide suitable conditions for deposition of sediment and particulate nutrients. The dominant

nutrient processes in wetlands are denitrification, plant uptake, deposition, adsorption and mineralisation (Table 1). Of these processes only denitrification represents permanent removal of nutrients from a wetland. The other nutrient processes are temporary stores, which can be re-released back to through-flowing waters. Plant uptake is a temporary nutrient store as unless aging plant material is removed from the system (e.g., by biomass harvesting, stock grazing etc.) a large proportion will eventually be converted back into soluble, bioavailable forms (60% of the $\text{NO}_3\text{-N}$ removed by watercress from the Whangamata streamwater is recycled when the plants die; Kit Rutherford, pers. comm.). Adsorption of phosphorus to inorganic sediments may be reversed under anaerobic conditions and fine particles that sorb onto vegetation and detritus may re-mobilise when they dry and/or during subsequent high flows. In practice, a small proportion (~5-10%) of the N and P taken up and cycled by plants is retained as long-lived humic compounds in accumulated sediments, or released in soluble humic forms with low bioavailability.

4.5.1 Natural seepage wetlands

Natural seepage wetlands occur at the heads and along the sides of streams. They may be also known as seeps, flushes, valley bottom or riparian wetlands. They are mainly fed by shallow subsurface flow that re-emerges via springs or seeps and their saturation status may range between temporary dryness and permanent saturation. These small wetlands are rarely identified in regional wetland inventories, although they may represent a large part of headwater catchments and can strongly affect the hydrology and water quality on a local scale (Merot et al. 2006).

Short and longer term studies around the North Island suggest that nitrate removal by seepage wetlands under baseflow conditions exceeds 75% (Cooper 1990; Downes et al. 1996; Rutherford & Nguyen 2004; Sukias & Collins submitted; Sukias & Nagels 2006). Lower removal rates are expected during events or when channels occur in the wetlands resulting in a significant fraction of water and nutrient bypassing the soil matrix (Burns & Nguyen 2002; Nguyen et al. 1999). Nguyen et al. (1999) reported 51% nitrate removal from a small seepage wetland (0.2% catchment area) over a six month period with numerous rainfall events. In a 15 month study of a natural seepage wetland receiving surface and subsurface flows from a Waikato dairy farm, Nguyen et al. (2002a) recorded reductions of 70-95% of nitrate-N, but found the wetland to sometimes be a source of ammonia-N, dissolved organic N and particulate N.

Areal removal rates for tracer input and output experiments in natural seepage wetlands are in the order of 5-15 $\text{mg/N/m}^2/\text{d}$ (Rutherford & Nguyen 2004; Sukias & Collins submitted). It would appear that there are localised “hot spots” within wetlands where denitrification rates are very high (e.g., where high nitrate groundwater first encounters organic, anoxic soils), while rates in other parts of a wetland may be significantly lower (e.g., Cooper 1990).

Total N, sediment and P removal by seepage wetlands have been less frequently investigated. Nguyen et al. (1999) reported 54% TN (inflow up to 20 g/d), 26% TP (inflow up to 2 g/d), 1% PP and 51% NO₃-N (inflow up to 30 g/d) retention over a six month period (which included storm events) by a small seepage wetland (0.2% catchment area) at Whatawhata. Higher flows are able to entrain fine sediment and when stock are grazing wetlands they may be sources of suspended sediment and nutrients (McKergow et al. 2007a; Nguyen et al. 1999).

Natural seepage wetland efficacy estimates are presented for two basic variants – 1% of catchment area and 5% of catchment area (Table 5). The perimeter of wetland requiring fencing is estimated using perimeter:area ratios for known wetlands (Collins 2004; McKergow et al. 2007b). Small, rectangular wetlands will have smaller perimeters than long, narrow wetlands.

Table 5: Wetland tool performance and costing.

Intercepted flow-path	Application sites	Situations where likely to be of significant benefit.	Target contaminant	Variant	Length of fencing m/ha of catchment	Area requirement of attenuation system/ ha	Sediment reduction range	N reduction range	P reduction range	E. coli reduction range	Attenuation system set-up costs (\$/ha of catchment)	Attenuation system maintenance costs (\$/ha of catchment/yr)	Notes on assumptions
					m/ha	m ² /ha	g/m/y or g/m ² /y or %	g/m/y or g/m ² /y or %	g/m/y or g/m ² /y or %	%	\$/m or \$/ha of system	\$/m/y or \$/ha/y of system	
Natural seepage wetlands													
Areas where shallow groundwater emerges or collects.	Natural boggy areas where groundwater seeps through saturated organic soils. Also receive some surface runoff from surrounding catchment, which includes particulate and dissolved contaminants.	Where groundwater emerges as seeps and springs in riparian zones and the toe-slope of hills.	sediments, N, P	Natural seepage wetlands. Low density -1% of catchment low perimeter/area ratio (0.35)	35 m/ha	100 m ² /ha	60% (of overland flow load entering wetland)**	50-75%*	No specific NZ or relevant overseas data but 10% PP expected from surface runoff		fencing options: sheep/beef 5 wire (3 electric); dairy: 2 wire electric	assume 1 weed spray per ha/y	* assumes most of N in form of nitrate and that denitrification is the key removal process. Assumes (1) incoming high nitrate water comes into contact with the soils; (2) residence time is sufficiently long (24-48 hours); (3) conditions are anoxic but not anaerobic; and (4) there is ample carbon 2. assumes wetland fenced and in good condition without channels of flowing water bypassing the soil matrix, no pugging or erosion 3. perimeter/area ratio = total perimeter/total area. Calculated for seepage wetlands at Whatawhata (Collins 2004) and Tutaeuaua (McKergow, unpublished data). Small square wetlands will have a low perimeter/area ratio (e.g., Collins 2004 wetland A 32 m x 7 m, perimeter = 78 m, area = 224 m, perimeter/area = 0.35), while long narrow wetlands will have a higher perimeter/area ratio. 4. assume that drained wetland is fully productive land which can be reinstated 5. add to costing reduced stock losses 0.01 sheep and 0.005 cattle/ha/y for undrained wetland 6. assumes all of surface runoff from paddock follows drainage lines and enters wetland ** estimated - no measured data for NZ
				Natural seepage wetlands. High density - 5% of catchment high perimeter/area ratio (0.75)	375 m/ha	500 m ² /ha	60% (of overland flow load entering wetland)**	50-75%*	No specific NZ or relevant overseas data but 10% PP expected from surface runoff		fencing options: sheep/beef 5 wire (3 electric); dairy: 2 wire electric	assume 1 weed spray per ha/y	
Constructed wetlands													
surface and shallow subsurface runoff	Where there are depressions, gullies and wet areas that intercept surface run-off and springflows.	Where landscape features such as depressions and gullies that intercept surface and subsurface run-off and drainage can be modified to form wetlands that retain and treat flows. Opportunity cost of land utilised expected to be low, with ancillary benefits in terms of reduced stock losses etc.	sediments, N, P	Facilitated wetlands -Low density - 1% of catchment	32 m/ha*	1% (100 m ² /ha) of contributing catchment	~60% plus of annual load in surface runoff	30% (likely annual range 10-40%) [#]	No specific NZ or relevant overseas data, but 50-60% reduction of particulate P in surface runoff expected	No specific NZ or relevant overseas data, but significant reduction of sediment associated E. coli particularly buffering of peak concentrations expected -likely 80% plus reduction in 90-percentile concentrations **	\$5.50/m ² wetland = \$550/ha of catchment***	\$15/ha/y****	*Assumes majority of N in form of nitrate (~80%); percentage removal performance likely to be similar across range of N loads, but to be better in warmer areas of the country, and in situations with lower run-off yields and/or flow variability. [#] Assumes that all surface runoff and ½ subsurface flow enters wetland (other ½ bypasses) *Assumes wetland sized to treat run-off from 5 ha subcatchment. Construction costs to dam and modify existing gullies or depressions estimated to be half that of excavation into flat land. Assume fencing on all sides assuming 9:1 length:width ratio, fence erected 1.5 m from wetland water edge. **E. coli performance estimates assume moderate to high incoming concentrations. There is potential for very low influent concentrations to be increased during passage through a wetland. *** Double handling of topsoil etc. expected to increase costs some-what for larger dimension wetlands, whilst fixed costs likely to be greater for smaller wetlands. ****Based on estimated cost for established wetland of one weed spray and one further inspection per wetland per year. Associated landscape and amenity plantings also likely to require maintenance and management.
				Facilitated wetlands - Moderate density - 2.5% of catchment	50 m/ha *	2.5% (250 m ² /ha) of contributing catchment	~80% plus of annual load in surface runoff	60% (likely annual range 40-80%) [#]	No specific NZ or relevant overseas data, but 60-80% reduction of particulate P in surface runoff expected	No specific NZ or relevant overseas data, but significant reduction of sediment associated E. coli , particularly buffering of peak concentrations expected -likely 90% plus reduction in 90-percentile concentrations **	\$6.50/m ² wetland = \$1625/ha of catchment***	\$25/ha/y****	

Intercepted flow-path	Application sites	Situations where likely to be of significant benefit.	Target contaminant	Variant	Length of fencing m/ha of catchment m/ha	Area requirement of attenuation system/ ha m ² /ha	Sediment reduction range g/m/y or g/m ² /y or %	N reduction range g/m/y or g/m ² /y or %	P reduction range g/m/y or g/m ² /y or %	E. coli reduction range %	Attenuation system set-up costs (\$/ha of catchment) \$/m or \$/ha of system	Attenuation system maintenance costs (\$/ha of catchment/yr) \$/m/y or \$/ha/y of system	Notes on assumptions
surface drainage	Surface drains carrying surface and shallow-subsurface run-off containing particulate and dissolved contaminants	Where surface drains transport a significant proportion of run-off.	sediments, N, P	Constructed wetlands Intercepting flows from surface drains Low density - 1% of catchment	18 m/ha *	1% (100 m ² /ha) of contributing catchment	~60% plus of annual load	30% (likely annual range 10-40%) [†]	No specific NZ or relevant overseas data, but 50-60% reduction of particulate P expected	No specific NZ or relevant overseas data, but significant reduction of sediment associated E. coli particularly buffering of peak concentrations expected -likely 80% plus reduction in 90-percentile concentrations **	\$11/m ² wetland = \$1100/ha of catchment***	\$10/ha/y****	[†] Assumes majority of N in form of nitrate (~80%); percentage removal performance likely to be similar across range of N loads, but to be better in warmer areas of the country, and in situations with lower run-off yields and/or flow variability. *Assumes wetland sized to treat run-off from 5 ha subcatchment. Assume fencing on 1 long and 2 short sides assuming 9:1 length:width ratio, fence erected 1.5 m from wetland water edge, and set along stream/open drain or existing fence. **E. coli performance estimates assume moderate to high incoming concentrations. There is potential for very low influent concentrations to be increased during passage through a wetland. *** Costs estimated in consultation with John Scandrett (Drainage Engineer, Southland) Double handling of topsoil etc. expected to increase costs some-what for larger dimension wetlands, whilst fixed costs likely to be greater for smaller wetlands. ****Based on estimated cost for established wetland of one weed spray and one further inspection per wetland per year.
				Constructed wetlands Intercepting flows from surface drains Moderate density - 2.5% of catchment	28 m/ha *	2.5% (250 m ² /ha) of contributing catchment	~80% plus of annual load	60% (likely annual range 40-80%) [†]	No specific NZ or relevant overseas data, but 60-80% reduction of particulate P expected	No specific NZ or relevant overseas data, but significant reduction of sediment associated E. coli, particularly buffering of peak concentrations expected -likely 90% plus reduction in 90-percentile concentrations **	\$13/m ² wetland = \$3,250/ha of catchment***	\$15/ha/y****	
subsurface/tile drainage	Subsurface mole and tile drains carrying subsurface run-off dominated by dissolved contaminants e.g., nitrate	Where subsurface tile/mole drains transport a significant proportion of run-off.	nitrate	Fully constructed wetlands Intercepting flows from subsurface tile drains Low density - 1% of catchment	18 m/ha *	1% (100 m ² /ha) of contributing catchment	30-50% assuming that the majority of sediment mobilised in mole/tile drains will be fine clays and silts	30% (likely annual range 10-40%) [†]	NZ data suggests minimal P removal without specific addition of P-sorbing materials	Buffering of peak concentrations -likely 80% plus reduction in 90-percentile concentrations **	\$11/m ² wetland = \$1,100/ha of catchment***	\$10/ha/y****	[†] Assumes majority of N in form of nitrate (~80%); percentage removal performance likely to be similar across range of N loads, but to be better in warmer areas of the country, and in situations with lower run-off yields and/or flow variability. *Assumes wetland sized to treat run-off from 5 ha tile-drained subcatchment. Assume fencing on 1 long and 2 short sides assuming 9:1 length:width ratio, fence erected 1.5 m from wetland water edge, and set along stream/open drain or existing fence. **E. coli performance estimates assume moderate to high incoming concentrations. There is potential for very low influent concentrations to be increased during passage through a wetland. *** Costs estimated in consultation with John Scandrett (Drainage Engineer, Southland) Double handling of topsoil etc. expected to increase costs some-what for larger dimension wetlands, whilst fixed costs likely to be greater for smaller wetlands. ****Based on estimated cost for established wetland of one weed spray and one further inspection per wetland per year.
				Fully constructed wetlands Intercepting flows from subsurface tile drains Moderate density - 2.5% of catchment	28 m/ha *	2.5% (250 m ² /ha) of contributing catchment	40-70% assuming that the majority of sediment mobilised in mole/tile drains will be fine clays and silts	60% (likely annual range 40-80%) [†]	NZ data suggests minimal P removal without specific addition of P-sorbing materials	Buffering of peak concentrations -likely 90% plus reduction in 90-percentile concentrations **	\$13/m ² wetland = \$3,250/ha of catchment***	\$15/ha/y****	

Intercepted flow-path	Application sites	Situations where likely to be of significant benefit.	Target contaminant	Variant	Length of fencing m/ha of catchment m/ha	Area requirement of attenuation system/ ha m ² /ha	Sediment reduction range g/m/y or g/m ² /y or %	N reduction range g/m/y or g/m ² /y or %	P reduction range g/m/y or g/m ² /y or %	E. coli reduction range %	Attenuation system set-up costs (\$/ha of catchment) \$/m or \$/ha of system	Attenuation system maintenance costs (\$/ha of catchment/yr) \$/m/y or \$/ha/y of system	Notes on assumptions
Stream-flow from agricultural catchment (100-500 ha.).	Off-stream wetland receiving drainage and streamflow comprising surface and subsurface run-off from grazed land. Weir constructed across stream/drain to divert normal flows through wetland and then returned back to stream or to adjacent receiving water. A proportion of flood-flows are likely to be bypassed over the weir and receive no additional treatment.	Where there is land suitable for wetland construction at the base of a catchment or adjacent to sensitive receiving waters (e.g., lakes or estuaries). Not likely to be suitable where large bed loads of sediment are transported downstream. Potential issues with fish passage - may require fish passes.	Sediments, N, PP Removal of dissolved contaminants such as nitrate likely to be best where baseflow dominates. Removal of particulate contaminants likely to be relatively efficient.	Farm or catchment-scale constructed wetlands intercepting stream and drain flows Low density - 1% of catchment	6 m/ha *	1% (100 m ² /ha) of contributing catchment	~60% plus of annual load, unless particularly fine and dispersible clays.	30% (likely annual range 10-40%) [†]	No specific NZ or relevant overseas data, but 50-60% reduction of particulate P expected	No specific NZ or relevant overseas data, but significant reduction of sediment associated E. coli, particularly buffering of peak concentrations expected -likely 80% plus reduction in 90-percentile concentrations **	1 ha of wetland /100 ha of catchment \$15-30/m ² wetland = \$1,500-3,000/ha of catchment ^{***}	\$10/ha/y ^{****}	<p>*Majority of N in form of nitrate (~80%); percentage removal performance likely to be similar across range of N loads, but to be better in warmer areas of the country, and in situations with lower run-off yields and/or flow variability.</p> <p>††Performance estimates for flow intercepted by wetland. Likely that a proportion of flood flows (50% in scenarios) and associated contaminants will be diverted from the wetland directly over the weir and hence will not receive any treatment in the wetland. Flood flows likely to be more important for mobilisation of sediment, P and faecal contaminant than for mobilisation of N, and to include materials eroded from stream banks and beds.</p> <p>*Wetland sized to treat run-off from 100-500 ha subcatchment.</p> <p>**E. coli performance estimates assume moderate to high incoming concentrations. There is potential for very low influent concentrations to be increased during passage through a wetland.</p> <p>*** Costs based on lake Okaro wetland (2.3 ha) built for EBOP (Tanner et al. 2007). Costs will vary significantly depending on extent of excavation and underlying soil materials -Estimates provided here assume clay subsoils at site of wetland without need for synthetic liner. Estimated costs include specialist engineering design and supervision, construction of timber weir structure on stream and pipeline to adjacent wetland, but no specific allowance for fish passage.</p> <p>****Based on estimated cost for established wetland of one weed spray and one further inspection per wetland per year. Associated landscape and amenity plantings also likely to require maintenance and management.</p>
				Farm or catchment-scale constructed wetlands intercepting stream and drain flows Moderate density - 2.5% of catchment	9.5 m/ha *	2.5% (250 m ² /ha) of contributing catchment	~80% plus of annual load, unless particularly fine and dispersible clays.	60% (likely annual range 40-80%) [†]	No specific NZ or relevant overseas data, but 60-80% reduction of particulate P expected	No specific NZ or relevant overseas data, but significant reduction of sediment associated E. coli, particularly buffering of peak concentrations expected -likely 90% plus reduction in 90-percentile concentrations **	2.5 ha of wetland /100 ha of catchment \$15-30/m ² wetland = \$3,000-\$7,500/ha of catchment ^{***}	\$15/ha/y ^{****}	

4.5.2 Constructed and facilitated wetlands

Shallow surface-flow wetlands or marshes are likely to be the most practical option for cost-effective treatment of agricultural run-off. Water is treated as it flows through shallow ponds or channels vegetated with emergent plants such as bulrushes, reeds or sedges. Facilitated and constructed wetlands attempt to replicate and optimise treatment processes that occur naturally in swamps, fens and marshes. Treatment efficiency is enhanced by optimising dispersion, flow paths, water depths, residence times, and vegetation characteristics. Construction and operating costs are relatively low providing suitable land is available, and enhancement of biodiversity and landscape aesthetics provides ancillary benefits.

Key features of facilitated or constructed wetlands that contribute to their nutrient removal functions include: (1) low flow velocities and tortuous pathways through aquatic vegetation, which favours sedimentation and accumulation of particulates that become incorporated into the wetland soils; (2) a mosaic of aerobic (oxygenated) and anaerobic (deoxygenated) micro-environments, which promotes bacterial transformations of nutrients (particularly nitrogen and sulphur) and other contaminants such as metals and pesticides; and (3) close contact between water, sediments, plants, detritus, and biofilms (bacterial slimes), which enhances uptake of nutrients, and promotes physical, chemical and biological interactions. Unless harvested and removed a significant proportion of the nutrients taken up by plants will be released again when they die and decompose.

There is now a substantial database on wetland treatment performance, particularly of wastewaters (IWA 2000; Kadlec 2005; Kadlec & Knight 1996; Tanner & Sukias 2003; Wallace & Knight 2006). Wetland performance has been measured under New Zealand conditions for 3 constructed systems treating nitrate-rich tile drainage from intensive dairy pastures in: Northland (3 years), Waikato (5 years) and Southland (3 years), and for an array of small experimental wetlands in Waikato (Sukias et al. 2005; Sukias et al. 2006a; Sukias et al. 2006b; Tanner et al. 2003; Tanner et al. 2005a; Tanner et al. 2005b; Tanner et al. 2005c). These studies show that newly constructed wetlands take a number of years to reach maturity, and that treatment levels vary with year-to-year differences in seasonal drainage patterns. In particular, N removal performance is better in warm than in cold seasons and when residence times are extended; i.e., when influent flows are spread out relatively evenly through time rather than arriving as a few large events. Nutrient budgets over 5 years for the longest-running wetland in the Waikato comprising ~1% of a 2.6 ha drainage area (without supplementary irrigation) showed TN removals varying from 40-406 g/m²/y (16-65% of influent loads; catchment yield 21-66 kg/ha/y). Typical TN removals of 100-120 g/m²/y (30-45% removal) were measured for the mature systems receiving loads of 250-350 g/m²/y. Removal of P was generally slightly negative in all the NZ wetland

systems studied, although this is likely to be able to be improved upon if wetland soils are supplemented with retentive materials (Sukias et al. 2006b). P retention is expected to be higher in situations where a significant proportion of the loading is in particulate rather than dissolved forms, but this may still be subject to desorption and re-release under anaerobic conditions. Identification of low-cost redox-insensitive P-sorbing supplements appropriate in different areas of the country, and the transformations and behaviour of P in response to seasonal wetting and drying of soils with and without these supplements is a subject for further investigation.

In the present review, constructed wetland efficacy is presented for a number of applications with varying flowpaths and percentage of contributing catchment based largely on information from NZ systems treating subsurface-drainage. Constructed wetlands can be used to intercept a range of flowpaths, including surface drains (see Figure 13), subsurface drains, overland flow, and stream flow, and can be constructed at a range of scales (Table 2). Runoff from each of these sources (and from different landscapes) has different characteristics (e.g., hydrological response and fraction of particulate nutrients), and there is still no good local information on the performance of constructed wetlands treating surface drainage or run-off from various landscapes and farm types. Recently, Environment BOP funded the design, construction and planting of a 2.4 ha wetland treating stream-flows entering Lake Okaro, near Rotorua (Tanner et al. 2007). Monitoring of this wetland, receiving stream-flows and seepage, from a grazed Rotorua catchment will provide valuable data on the performance of larger-scale systems.

To achieve 50-60% nitrate and TN removal, constructed wetlands generally need to cover 2-3% of the catchment from which they receive drainage water. Smaller wetlands (1% of catchment area) will generally removal ~30% of nitrogen, while larger wetlands (5% of catchment area) can achieve 70% nitrogen removal or greater. In practice, treatment performance will vary with year-to-year differences in the timing and magnitude of run-off events, which affect hydraulic retention times, inter-event duration and the seasonal timing of loadings in relation to microbial denitrification activity, and plant growth and senescence (Tanner et al. 2005a; Tanner et al. 2005b; Tanner et al. 2005c). Current NIWA research aims to develop models that can predict N removal performance under varying hydrological loading regimes.

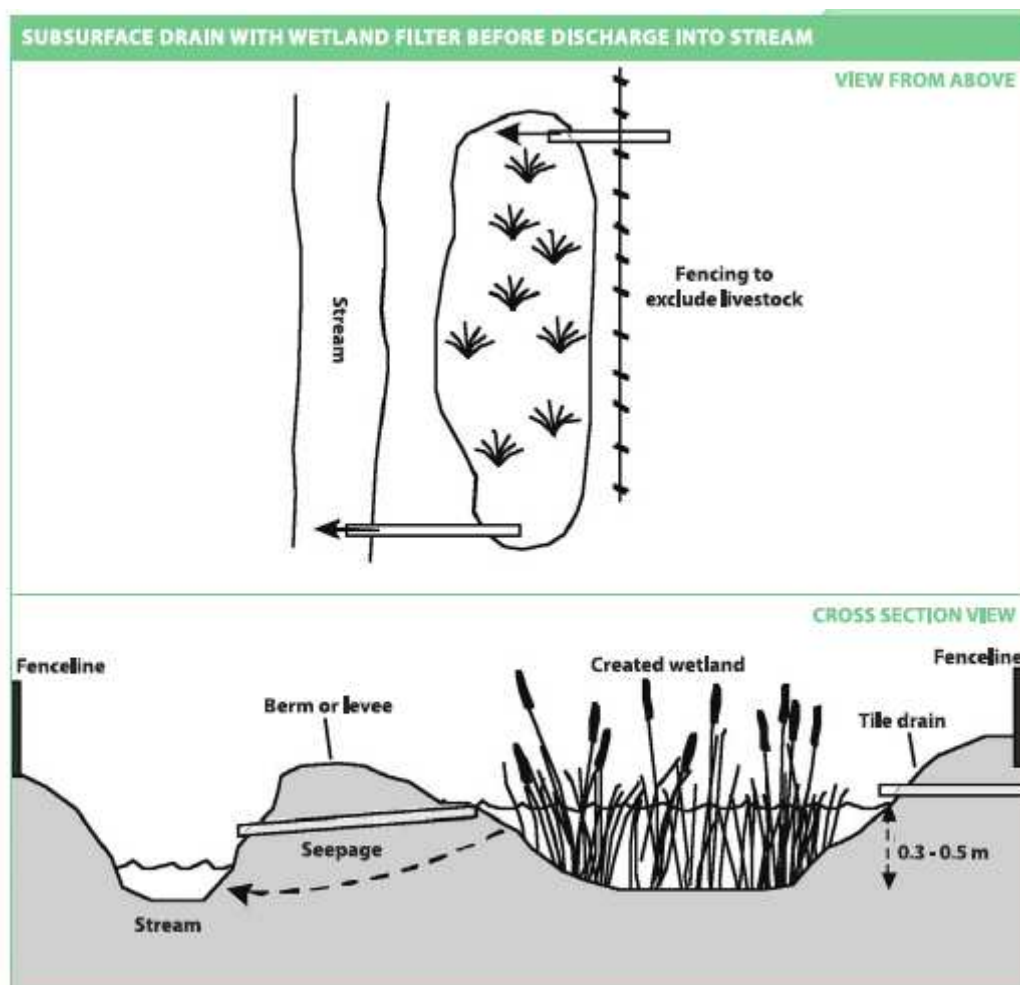


Figure 13: Example of how tile drain flows can be intercepted and treated in constructed wetlands set alongside streams (DEC 2006).

4.5.3 Floodplain wetlands

Natural processes of channel erosion and migration, and sedimentation in stream and river floodplains often form a complex shifting mosaic of old channels, oxbows, islands, levees, deltas, lagoons, ponds and wetlands. These areas become connected to the main channel during floods, acting to attenuate flood flows and promote settling of sediments and associated contaminant loadings. During non-flooded periods these riparian areas may remain connected to and interact with subsurface hyporheic flows, which percolate through sand, gravel and other permeable soils under and beside the stream channel. As well as promoting flow and contaminant attenuation, these spatially and temporally diverse areas are also incredibly important habitats for wildlife and biodiversity.

Retention of floodwater nutrients in a restored floodplain in Denmark was estimated to reduce river N export by 25% and P export by 30% (Kronvang et al. 1999). Walling and Owens (2003) estimated conveyance losses associated with the deposition of sediment-associated contaminants on the floodplains bordering two English rivers of 27 and 32% of suspended sediment and 14 and 9% of TP, respectively. Olde Venterink et al. (2003) studying two distributaries of the River Rhine found 20-45% P retention in the river where floodplains transported a substantial fraction of the flow, compared to negligible removal in the adjacent river with minimal floodplain.

Although most lowland floodplains in NZ used for agriculture are now highly modified and managed, so that they are largely disconnected from the streams and rivers that feed them, there is still considerable potential to restore and enhance contaminant attenuation functions. Remnant floodplains managed primarily for flood protection (e.g., lower Waikato River) undoubtedly still provide water quality benefits, but these could be improved by broadening their management objectives and utilising areas inside and alongside flood-banks as riparian wetlands (Petersen et al. 1992; Sheibley et al. 2006; Zedler 2003).

4.5.4 Floating wetlands

Floating wetlands are a novel ecological water treatment technology, in which emergent wetland plants grow hydroponically on floating mats or rafts (Headley & Tanner 2006). The roots of the plants form dense growths below the floating mats, taking their nutrients from the surrounding water. The root matrix provides a large surface-area for the development of microbial biofilms which promote sediment trapping, microbial transformations and ion adsorption. By shading the water column and reducing wind-driven circulation, the floating mats may also reduce algal growth and create anoxic zones that are conducive to microbial denitrification. The ability of floating treatment wetlands to cope with fluctuating water levels makes them particularly amenable to contaminant reduction in ponds, dams and irrigation storage reservoirs.

There is little quantitative data available on the nutrient removal performance of floating treatment wetlands. Mesocosm studies in New Zealand recorded removal of 58-67% of fine SS, 72-96% of $\text{NH}_4\text{-N}$ and 20-51% of DRP from artificial urban stormwater over ~7 days for mature planted floats, compared to negligible removal in equivalent unplanted floats (Headley & Tanner 2007a). Further trials are underway in association with Environment BOP to test their nutrient removal potential in simulated eutrophic lake waters (Headley & Tanner 2007b).

4.5.5 Plant and algae harvesting

Dissolved nutrients can be removed from water by uptake into aquatic plant tissues and subsequent harvesting and removal. By removing the plant biomass via harvesting, the associated nutrients that would otherwise be returned to the water and sediments when the plants degrade can be permanently removed. Aquatic plants can be highly efficient at scavenging nutrients down to remarkably low concentrations in the associated water. In particular, they absorb the nutrients necessary for plant growth; i.e., nitrogen, phosphorus, potassium, sulphur, magnesium and calcium and a range of micronutrients. Aquatic macrophyte tissues, however, have high water contents (90% or more), so large amounts of wet biomass must be removed in order to remove significant amounts of nutrients.

Nutrient uptake rates of 35-585 gN/m²/y and 9.2-113 gP/m²/y have been reported (Reddy & DeBusk 1987) for productive floating aquatic plants like water hyacinth and alligator weed in tropical climates, but most of these plants have restricted growth seasons in NZ and some are classified as invasive weeds. Floating duckweeds (*Lemna* and *Spirodella* sp.) are a possible exception being widely present around NZ. They also have high protein contents making them a potential livestock feed-source, however their nutrient uptake rates are at the lower end of the range for floating plants (35-120 gN/m²/y and 12-40 gP/m²/y) (Reddy & DeBusk 1987). Most studies investigating duckweed nutrient removal have been carried out in relatively nutrient-rich wastewaters, but data summarised by Korner et al. (2003) suggests uptake rates of ~0.4 gN/m²/d and 0.03 gP/m²/d are likely to be realistic at nutrient concentrations relevant to agricultural drainage waters in NZ, with roughly similar additional removal occurring collectively through other processes such as denitrification and algal uptake.

Productive emergent aquatic plants such as raupo, bulrushes and sedges have reported nutrient uptake rates of 13-263 gN/m²/y and 2-40 gP/m²/y (Reddy & DeBusk 1987). However, nutrients are mostly taken up from the sediments rather than from the water column, and a third to a half of the nutrient content is generally stored in below-ground tissues and thus not readily amenable to harvesting. Most studies of emergent wetland systems have found that potential nutrient removal by harvesting of emergent plants is small relative to removal via other removal mechanisms (particularly denitrification), and that harvesting is therefore not a cost-effective option (e.g., Tanner 2001a; Tanner 2001b), unless the biomass has specific economic value. See section on constructed wetlands above.

Watercress is likely to be a preferred species for use in a nutrient stripping situation because of its ability to take up nutrients directly from water, its relatively high nitrogen content (3-7 times higher than most aquatic plants) and uptake rates (Vincent & Downes 1980), and its ability to reduce in-stream nitrogen concentrations to very

low concentrations (Howard-Williams et al. 1982; Cox 2004). Vincent and Downes (1980) experimentally determined a nitrate uptake rate of up to 0.261 g N/m²/d in beds of watercress (*Nasturtium officinale*) in the Whangamata stream at Lake Taupo in mid-summer, but this reduced to negligible levels in mid-winter. Howard Williams et al. (1982) examined field uptake rates in the same stream and concluded that 1 g N/m²/d was accumulated by watercress during the growing season. Cox (2004), using microcosm studies of watercress, found supporting evidence with higher daytime nitrate uptake (expressed here relative to plant dry weight, 1.5 mg N/g dry weight/d) than during the night (1.35 mg N/g dry weight/d), although night rates were still significant. Cox suggested this may be associated with differences in study techniques (net community uptake compared with individual root incubations). The range of nitrogen removal rates in the literature varies by a factor of ten and does not explore the potential of managed harvested watercress beds to optimise nutrient removal. An experiment is currently underway at Wharenui Station, Rotorua to measure watercress nutrient uptake from stream water (McKergow et al. 2007c). Other palatable water-tolerant grasses such as *Glyceria* sp. may also have potential for nutrient removal via aquatic harvest, providing feed-quality and issues such as cyanide toxicity (Sharman 1967) can be managed.

Long term data for upstream and downstream sites on Whangamata Stream (Figure 14) clearly demonstrates that during periods of lower flows (e.g., 1986-1988) summer nitrate concentration decreased between the bottom and top sites. Under higher flow conditions or when the channel was moderately shaded nitrate concentration reductions between the sites were lower.

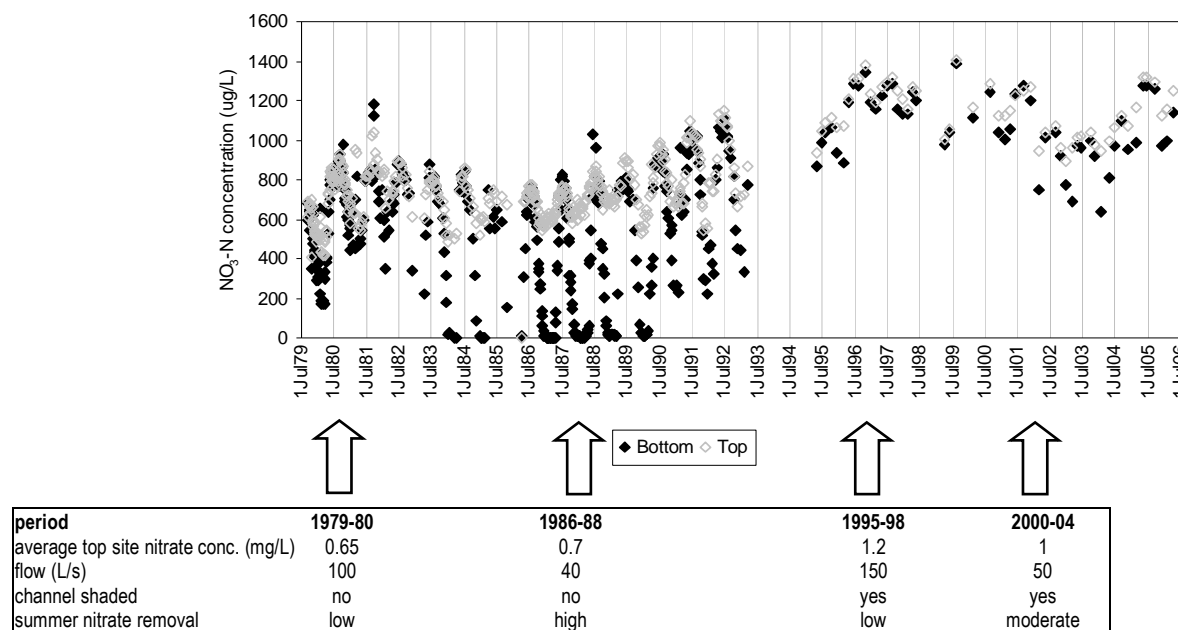


Figure 14: Whangamata Stream nitrate concentrations at the top and bottom sites with flow conditions, channel shade and summer nitrate removal for the four intensively sampled periods (Howard-Williams & Pickmere 1999 and unpublished data).

Algal turf scrubbers are a method of using filamentous algae (periphyton) to take up nutrients from water using shallow sloped flow-ways (Craggs 2002). The algal turf is periodically harvested to remove the accumulated biomass and nutrients. Symbiotic aerobic bacteria and fungi associated with the algae also break down organic contaminants. Relatively high mean nutrient removal rates of 0.95 g N m²/d (~ half accumulated in periphyton biomass) and 0.44 g P m²/d (partially by precipitation due to elevated pH) have been reported for wastewaters (Craggs 2002; Craggs et al. 1996a; Craggs et al. 1996b). Lower mean levels of 0.12 g P m²/d have been recorded for agricultural run-off with low nutrient content (Adey et al. 1993).

4.5.6 Gaps in knowledge or communication

Field scale research is required for all types of wetlands (Table 6). The research focus to date has typically been on nitrate removal and experimental in nature, rather than quantifying performance over the longer term and examining TN exports. More information is required about sediment and P processing and attenuation in wetlands as currently little information exists for New Zealand wetlands.

Much of the research effort on natural seepage wetlands has been on short term nitrate removal and denitrification rather than total N removal performance over the longer term. Research is needed to measure the net N and P exports from a range of seepage wetlands under baseflow and event conditions.

Field scale operational performance is required for several facilitated or constructed wetland variants (Table 6), particularly facilitated wetlands and bottom of catchment wetlands. Environment BOP has commenced monitoring on the recently established large-scale Lake Okaro wetland treatment system (2.4 ha wetland treating the main stream inflows to the lake), which will provide some valuable information on the efficacy of large bottom of catchment wetlands in one landscape context. Wetlands can be sources of greenhouse gases such as methane and nitrous oxide. Realistic emission rates for various systems need to be defined and their implications evaluated.

The SFF-funded trials in Rotorua investigating nutrient uptake by harvested watercress will provide a better basis for evaluation of the potential utility of aquatic plant uptake. If the trial results are positive, further work will be warranted to develop a practical system that could be used by farmers, including propagation, growth and harvesting methods, palatability and feed-value, and other market opportunities.

Table 6: Communication and science gaps for wetlands.

Attenuation tool	Communication status and gaps	Science gaps	Availability in NZ farm-scale modelling tools (OVERSEER® and NPLAS)
natural seepage wetlands	<p>Existing: scant mention in DEC (2006).</p> <p>Gaps: Need for detailed practical guidelines for seepage wetland identification and management.</p>	<ul style="list-style-type: none"> • importance of denitrification vs other N transformations and total N removal • role of plant uptake and influence of fencing on this • impact of livestock damage on performance e.g., channels with bypassing flow • importance at the landscape scale i.e., extent • SS and particulate nutrient trapping from incoming overland flow • performance of re-instated wetlands • productiveness of drained wetlands of adjacent pasture • <i>E. coli</i> and faecal microbe removal • P retention/release. 	<ul style="list-style-type: none"> • included in OVERSEER® (under development) • included in NPLAS
facilitated and constructed wetlands – all types and scales	<p>Existing: Basic concept and preliminary sizing guidelines given in DEC (2006)</p> <p>Gaps: Need for detailed practical guidelines.</p>	<ul style="list-style-type: none"> • field-scale operational performance of wetlands receiving flows from surface drains needs to be tested • field-scale operational performance and cost: effectiveness for larger wetlands (2.5% catchment area) receiving subsurface and surface-flows • field-scale operational performance and costs of facilitated wetlands • field-scale operational performance of bottom of farm/catchment scale wetlands intercepting drains and streamflow; Lake Okaro wetland being monitored, but monitoring in range of contrasting situations advisable. • For all constructed wetlands: • effect of variable flow rates on treatment performance • enhancement of N and P removal through use of subsoil as a rooting medium and addition of reactive materials • enhancement of denitrification by strategic seasonal harvesting of plant material and retention within the wetland • faecal microbe removal under varying loading regimes (being partially addressed for one system in CO1X0304). • enhancement of faecal microbe removal through incorporation of open-water zones • association between <i>E. coli</i> and pathogen risk after passage through wetlands • greenhouse gas implications and trade-offs 	<ul style="list-style-type: none"> • included in OVERSEER® (under development) • included in NPLAS
Aquatic plant harvesting	No guidelines available	<ul style="list-style-type: none"> • development, pilot and field-scale testing, and nutrient removal assessment of practical culture and harvesting systems, and watercress feed quality. 	

4.6 **Reactive filters and materials**

Nutrient attenuation can be enhanced by the addition of reactive materials to flowpaths. Reactive materials and filters are generally designed to target one attenuation process, typically denitrification or sorption (Table 7).

The addition of organic carbon has been used to enhance denitrification of nitrate in drainflow, shallow subsurface flow, and constructed wetlands. These filters or sediment additions to wetlands are most applicable where high nitrate concentrations occur, and, in wetlands, where availability of C limits denitrification rates.

Phosphorus sorbing materials such as melter slag, fly ash and alum have been trialled in subsurface drains, streams, wetlands and surface runoff source areas to attenuate dissolved P. The use of sorbing materials to capture P in flow is a function of cost and effectiveness. Although a number of natural products or industrial by-products have been shown to remove P from pipe or stream flows, the amounts required to achieve a long-lasting effect tend to be large due to the relatively low P sorption capacity of these materials. Consequently, their field use tends to be limited to areas where a local source is readily available and excessive transport costs do not preclude the use of the material. Conversely, despite having relatively high P-sorption capacities, artificial products such as alum tend to be relatively expensive and often short-lived.

Table 7: Reactive materials and filters performance and costing.

Intercepted flow-path	Application sites	Situations where likely to be of significant benefit.	Target contaminant	Variant	Length of fencing m/ha of catchment m/ha	Area/length requirement of attenuation system/ ha m ² /ha	Sediment reduction range g/m/y or g/m ² /y or %	N reduction range g/m/y or g/m ² /y or %	P reduction range g/m/y or g/m ² /y or %	E. coli reduction range %	Attenuation system set-up costs (\$/ha of catchment) \$/m or \$/ha of system	Attenuation system maintenance costs (\$/ha of catchment/y) \$/m/y or \$/ha/y of system	Notes on assumptions
Permeable reactive filters													
Shallow subsurface flow entering channel	Intercepting subsurface flow flowing within 3 m of the ground surface. Best positioned perpendicular to flow (e.g., parallel to stream channel). Dominated by dissolved contaminants e.g., nitrate	Where subsurface flow is confined by low permeability layers beneath (e.g., clay or bedrock)	nitrate	Denitrification wall: trenches back-filled with sawdust/soil mix	n/a	2 x channel density*	n/a	1 g/m ³ of wall/d	n/a	n/a	each m ³ costs \$50	nil	1. Assumes 80% of total N is nitrate, plume > 3 mg/L nitrate-N. 2. Assumes treatment of 1 g/m ² of wall/d = 365 g/m ³ wall/y 3. Estimated cost \$50 per m ³ , including trenching, spoil removal, labour, cartage, and material for sawdust, includes regrassing. Assumes road transport <50 km for off-site supply of sawdust. 4. Estimated 10 year lifetime before replacement of sawdust required. * Assumes subsurface flow is < 3 m below ground and trench captures all subsurface flow on both channel/drain sides, therefore trench is 2 x channel density in length.
subsurface/tile drainage	Subsurface mole and tile drains carrying subsurface run-off dominated by dissolved contaminants e.g., nitrate	Where subsurface tile/mole drains transport a significant proportion of run-off.	nitrate	Woodchip filters small	25 m/ha*	30 m ³ of woodchip filter/ha of tile-drained catchment	Sediment load likely to be minor. Likely to become blocked if tile-drains transport significant sediment load.	40-50%†	Long-term performance unknown; ~30% TP removal from tile drainage recorded in first 2 years operation, presumably due to filtration, sorption & immobilisation	Not known	\$2000-2,500/ha of catchment (depending on whether woodchip from on-farm chipping of woodlot or shelter belt trees, or from local supplier)	Estimate 10 year lifetime, before complete replacement required	†Assumes majority of N in form of nitrate (~80%); percentage removal performance likely to be similar across range of N loads, but to be better in warmer areas of the country, and in situations with lower run-off yields and/or flow variability. Size and performance based on field studies at Te Hoe, Waikato (Sukias et al. 2006) *Assumes wood-chip filters sized to treat run-off from 1 ha tile-drained subcatchments. Assume fencing on 1 long and 2 short sides assuming flow-path length of 5m, fence erected 1.5 m from filter edge, and set along stream/open drain or existing fence. *** Costing based on 1 m deep excavation below 0.9 m deep tile drain (total 1.9 m), filter lined and covered with LDPE and recovered with soil. Assumes road transport of 50 km for off-site supply of gravel and wood chips. ***** Cost estimates for wood chip supplies from Karl Schwitzer (pers comm); KS Developments Ltd, Cambridge.
				Woodchip filters large	30 m/ha*	50 m ³ of woodchip filter/ha of tile-drained catchment	Sediment load likely to be minor. Likely to become blocked if tile-drains transport significant sediment load.	70-80%†	Long-term performance unknown	Not known	\$4000-4,800/ha of catchment (depending on whether woodchip from on-farm chipping of woodlot or shelter belt trees, or from local supplier).	Estimate 10 year lifetime, before complete replacement required	†Assumes majority of N in form of nitrate (~80%); percentage removal performance likely to be similar across range of N loads, but to be better in warmer areas of the country, and in situations with lower run-off yields and/or flow variability. Size and performance based on field studies at Te Hoe, Waikato (Sukias et al. 2006) *Assumes wood-chip filters sized to treat run-off from 1 ha tile-drained subcatchments. Assume fencing on 1 long and 2 short sides assuming flow-path length of 6.5m, fence erected 1.5 m from filter edge, and set along stream/open drain or existing fence. *** Costing based on 1 m deep excavation below 0.9 m deep tile drain (total 1.9 m), filter lined and covered with LDPE and recovered with soil. Assumes road transport of 50 km for off-site supply of gravel and wood chips. ***** Cost estimates for woodchip supplies from Karl Schwitzer (pers comm); KS Developments Ltd, Cambridge.

Reactive materials

Intercepted flow-path	Application sites	Situations where likely to be of significant benefit.	Target contaminant	Variant	Length of fencing m/ha of catchment m/ha	Area/length requirement of attenuation system/ ha m ² /ha	Sediment reduction range g/m/y or g/m ² /y or %	N reduction range g/m/y or g/m ² /y or %	P reduction range g/m/y or g/m ² /y or %	E. coli reduction range %	Attenuation system set-up costs (\$/ha of catchment) \$/m or \$/ha of system	Attenuation system maintenance costs (\$/ha of catchment/y) \$/m/y or \$/ha/y of system	Notes on assumptions
subsurface/tile drainage	Subsurface mole and tile drains carrying subsurface run-off dominated by dissolved and fine particulate contaminants e.g., DRP and PP	Where subsurface tile/mole drains transport a significant proportion of run-off.	P, sediment	Reactive materials for tile-drain backfill: melter:basic slag (90:10 mix)	n/a	n/a	41%	0	56%	Not known	\$550/ha for cartage, handling & placement; assumes new trench and pipe required anyway	0	*cost-effectiveness directly proportional to cartage distance. **life expectancy of 25 yrs (cost incorporated into cost-effectiveness metric) ***estimated cost-effectiveness \$14 to \$27 per kg P removed
				Reactive materials for tile-drain backfill: fly ash	n/a	100 m trench	Year 1 58% Year 2 40% Year 3 27% Mean 42%	nil	Year 1 83% Year 2 55% Year 3 50% Mean 63%	90	138	0	*ash applied at 0.12m ³ per m of tiles, to a depth of approx 16 cm in bottom of trench **although highly effective for reducing P and sediment, alkaline drainage means that this mitigation option is inappropriate *** 25 year lifetime assumed
				Reactive materials for tile-drain backfill: standard pea gravel (control)	n/a	100 m trench	Year 1 31% Year 2 -22% Year 3 -23% Mean = nil effect	nil	Year 1 62% Year 2 7% Year 3 -19% Mean = nil effect assumed (certainly not significant in years 2 and 3)	60	40	0	Standard gravel backfill used for tile drains
Drain and stream flows	in-stream	small streams with flows < 15 L/s	P	Porous geotextile socks filled with iron-rich melter slag material	n/a	n/a	0	0	10% of TP 44% of DRP	Not known	\$7/sock, but each sock only removes 37g of P, so may need up to 10 socks/ha	\$2/sock to remove	*expensive technology (esp. labour component); estimated cost:effectiveness; \$256 per kg P removed **limited to small streams ***working lifetime <1 year
laneway runoff; flow channels	adjacent to lanes/tracks; flow cartridges	where flow from lanes discharges directly into streams	P	Altered steel melter slag filter	n/a	as and where required - hollows, depressions that drain to stream	98% - Significant potential for clogging by high sediment loads	0	93%	Not known	\$6-9 per m of lane treated	0	*Estimated cost: effectiveness; \$9 to \$13 per kg P removed - directly proportional to cartage distance for matrix source (matrices other than melter slag could also potentially be used). **life expectancy of 45 yrs assuming no clogging of material; geotextile cloth placed to minimise sediment clogging (this cost has been incorporated into cost-effectiveness metric along with other costs; P sorption capacity of 80% of Pmax assumed)
overland flow	critical source areas of overland flow	sprayed on fields where overland flow is an important source of P	dissolved P	Alum soil amendments in critical source areas	n/a	n/a	0	0	20%	Not known	0	60	*appropriate to cropped or recently grazed land mainly; works OK under the young pastoral soils at Inchbonnie, otherwise a fairly expensive option and limited effectiveness on pastures and weathered soils **working lifetime <2 months (1 application/y in scenarios) ***Estimated cost: effectiveness \$40 to \$300 per kg of P removed, depending on soil type

4.6.1 Denitrification walls

Aimed solely at nitrate removal, the trench is a permeable barrier of sawdust and soil mix, built perpendicular to groundwater flow (Schipper & Vojvodic-Vukovic 1998). They are best constructed where the full extent and flow direction of nitrate polluted groundwater can be determined and then intercepted with relative ease, for example, a consistent direction within several metres of the ground surface. They are best targeted to specific areas where plumes of nitrate are known to exist, for example, cattle feedlots, and old fertiliser dumps. More broadly, these systems may be suited to areas where there are shallow confining layers on which the groundwater perches and flow paths are well understood. Denitrification walls are not suitable for coarse-textured subsoils (Schipper et al. 2004) but have been observed to work well on loam soils. During construction it is imperative that the media is not returned to the trench such that it is less permeable than the surrounding soil as this will encourage bypass flow. The sawdust and soil media and physical construction set the conditions for microbes capable of carrying out denitrification to flourish. The average expected nitrate removal performance is 1g NO₃-N per cubic meter of wall per day (Schipper et al. 2005). Higher rates of nitrate (up to 15 gN/m³/d have been reported in Western Australia (Fahrner 2002) where soil temperatures and nitrate concentrations were higher. Hence the wall can be sized to accommodate the flow rate and concentration encountered.

Similar approaches are currently being explored where denitrification wall are constructed on either side of a tile drain in Iowa (Jaynes et al. in press). Jaynes et al. demonstrated that these denitrification walls decreased nitrate from 22 to about 9 mgN/L for more than 5 years with an average nitrate removal rate of 0.6gN/m³/d. These walls were narrower (about 0.6 m) and nearly 100% wood chip material. While this system has not been tested in New Zealand, it is possible they would be easier and cheaper to install than the walls built by Schipper. It is likely that these walls could be installed at the same time as tile drains but would be harder to retrofit when location of tiles are unknown.

4.6.2 Woodchip filters

A woodchip filter is a shallow lined excavation receiving runoff from subsurface drains. Woodchips are added to the excavation to promote conditions suitable for denitrification, and so are most suitable for drains where the majority (80%) of N is in the nitrate form. Woodchips are used, rather than sawdust, to provide a porous bed through which drainage waters can percolate. Sukias et al. (2005; 2006b) has evaluated the efficacy of three medium (1.2% of catchment area) and one small (0.6% of catchment area) pilot-scale woodchip filter receiving subsurface drainflow on a dairy farm in the Waikato. Annual mass load reductions of 55-79% for nitrate-N were recorded over 2 years operation representing average annual removals in the range of

~90-300 mgN/m³/d. Increases in levels of ammonium-N and, in the first year of operation, Org-N resulted in lower annual removal of total N (16-49%). FRP and TP removal was variable, with apparently reasonable removal of particulate forms entering during large storms, but later release in dissolved forms. There would be value in testing a range of P-sorbing media that could be incorporated within the woodchip, or used as a final P filter.

Similar woodchip systems are being trialled (Louis Schipper, pers. comm.) to treat a range of wastewaters including municipal and industrial (spent hydroponic solutions) and small streams. Removal efficiency of nitrogen removal from wastes for these denitrification beds currently range between 5 and 30 gN/m³/d depending on nitrate concentrations and loadings (Louis Schipper, unpublished data). Lower rates are observed in streams most likely due to low nitrate concentrations and high flows.

4.6.3 P socks

The P sock is a potential attenuation tool in which P is entrapped by a sorbing material in a casing that prevents it from washing away during a flood, but enables the material and P to be periodically removed. McDowell et al. (2007) evaluated the effectiveness of a stable, non-toxic (neutral pH) steel melter slag as the sorbing material for removing P in streamflow. One hundred and ninety socks 1 m long by 9 cm diameter were made using a woven geotextile (polyester) cloth with a 2 mm mesh. This was filled with a combination of melter slag (85%), electric arc furnace slag (10%) and basic slag (5%), sourced from a steel mill in South Auckland. The slag had a minimum mean particle size of 2 mm and maximum mean particle size of 5 mm. Socks were installed in a 200 m reach of streambed in a herringbone fashion, which allowed both fish passage and mixing with stream water. Overall, concentrations of DRP and TP decreased on average 35 and 21%, respectively after the P-socks were installed, while loads decreased 44 and 10%, respectively. While this was an effective P removal tool at low flows, relatively little P was retained at flow rates >20 litres per second. In addition, the technology is expensive at between \$200-300 per kg of P removed due to the high labour cost incurred during the placement and removal of socks. Hence, this approach to P retention is limited to small, slow flowing waterways and may be more useful in stopping P loss from sources such as runoff from lane ways where dung is regularly deposited. The material has been used successfully in wastewater filter beds and has been tested as backfill for tile drains, which showed good effectiveness in removing P from drainage (McDowell et al. 2005; Pratt et al. 2007; Shilton et al. 2006).

4.6.4 Tile drain additions

A number of materials capable of capturing P and sediment in tile drainage flows have been trialled locally. This has involved placing materials such as ash, river gravel and

volcanic lapilli around the drainage pipe. The effectiveness of placing lapilli directly into the mole channel has also been researched.

In the first year of field trials at Massey University the placement of volcanic lapilli in the mole channel or around the pipe drain removed 50% of total P loads in tile drainage. However, in the second year of field trials the lapilli mole fill and back fill systems failed to remove significant amounts of P from drainage waters. The poor performance of the mole fill system is believed to result partly from the drainage water by-passing the lapilli-filled mole channel by flowing in the cavity above the lapilli.

On-going field research in Southland indicates that the placement of a 20 cm layer of coal combustion ash (75:25 bottom ash: fly ash) around drainage pipe led to a 40% reduction in sediment loss and a 60% reduction in P loss. Although the ash mixture was highly effective for reducing P and sediment loads in drainage, the caustic nature of the resulting drainage means that this mitigation option is inappropriate for field use without a more aggressive pre-treatment of the ash material to neutralise its alkalinity.

An additional treatment at the same field site involved placing the pipe drain above a shallow (10 cm) layer of pea gravel to act as a sediment (and thus P) trap. Although Year trial 1 results looked promising, P and sediment loads in Years 2 and 3 actually increased above the control treatment, with an overall nil effect recorded for the 3 year trial.

4.6.5 Wetland additions

Although wetlands can effectively remove particulate P, retention of soluble forms, (either entering in inflows or released from retained sediments) is generally limited by predominantly anaerobic conditions common in wetland soils. Constructed wetlands treating tile drain flows have generally shown relatively poor or negative P removal in NZ trials (see section 4.4.2). There is, however, considerable potential to improve their performance by addition of P retentive materials to the soils of constructed wetlands or use of porous filters.

A range of materials with P sorption properties have been identified for use in constructed wetlands treating wastewaters, including natural sediments and industrial waste products such as smelting wastes and fly-ashes (e.g., Drizo et al. 1999; Mann & Bavor 1993; Sakedevan & Bavor 1998).

Melter slag aggregate filter beds have been used for final treatment of waste stabilisation pond effluents at a number of sites in New Zealand. Monitoring over 10 years at Waiuku showed that >70% TP removal over the first 5 years of operation, after-which retention declines markedly (Shilton et al. 2006). A maximum retention capacity of 1.23 kg TP per tonne of slag was found under these conditions. Subsequent

laboratory studies (Pratt et al. 2007) have shown optimal P retention for this material at neutral pH and high redox potential (i.e., aerobic conditions). Field trials of electric arc furnace slag filters added to constructed wetlands or operated as stand-alone recorded average P removal rates above 70% from farm dairy wastewaters (Weber et al. 2007).

Alum and calcium carbonate amendments to agricultural ditch sediments were found to reduce levels of readily exchangeable P and equilibrium P concentrations (Smith et al. 2005). Ann et al. (1999) found P immobilisation in a wetland soil was increased by a range of chemical amendments, with the effectiveness in the order: ferric chloride > alum > calcium carbonate. The calcium carbonate and alum-amended soils were less sensitive to redox changes and were considered to be more suitable for binding P in anaerobic wetland sediments (Ann et al. 1999).

Natural New Zealand zeolites can effectively retain ammonium and to a lesser extent phosphate from wastewaters and chemical solutions (Nguyen 1997a; Nguyen 1997b; Nguyen et al. 1998; Nguyen & Tanner 1998). Various types and grades of zeolite are amenable to use as wetland soil additives or porous filter media. Sukias et al. (2006b) undertook a preliminary laboratory trial comparing a range of P sorbent, precipitant and sediment sealing materials. A zeolite and a commercial P retention product (modified zeolite) being developed by Scion were found to have the best P retention characteristics. Further pilot trials are required to screen a wider range of materials and test performance under field conditions.

To maximise denitrification rates in constructed wetlands organic amendments such as sawdust or straw can be added to soil media. This was done in the Lake Okaro wetlands where sawdust wastes were readily available from a nearby mill (Tanner et al. 2007). Further research is warranted to determine the quantitative benefits on wetland denitrification rates and find optimal application rates.

4.6.6 Alum

Aluminium sulphate (alum) has been used to reduce P runoff in a small number of field studies. Alum additions to effluents or soil can reduce soluble P concentrations and thus losses of dissolved P. Smith et al. (2001) found that addition of alum to poultry manure that was applied to plots containing tall fescue (equivalent to 40 kg Al/ha) reduced dissolved P concentrations by 84%. On-going research on the West Coast (R McDowell, pers. comm.) indicates that 2 broadcast applications of alum (equivalent to 20 kg Al/ha/application, dissolved in 2 m³ of water) reduced P losses in overland flow from dairy pasture by approximately 20%.

4.6.7 Gaps in knowledge and communication

All of the reactive filter and material attenuation tools have been tested at a limited number of sites and so more field testing is required. In addition, a major opportunity exists to combine the properties of reactive material nutrient attenuation tools, and possibly also add suspended sediment attenuation (where applicable).

To facilitate use of wood-chip filters as a tool for farmers, development and refinement of a practical design and field-testing over a number of years is required. Current pilot-scale trials should ideally be continued to provide information on the operational life-time of wood-chip filters in terms of continued organic C supply to denitrifiers and maintenance of hydraulic conductivity. Information from these studies then needs to be incorporated into standard guidelines for farmers, which provide details for sizing, construction and management, and outline typical performance expectations.

To improve the P removal performance of constructed wetlands and woodchip filters a range of P-sorbing additives or filter media need to be identified and tested. These need to be able to sorb and retain P under the anoxic and anaerobic conditions common in denitrifying systems such as constructed wetlands and woodchip filters.

Further work is required to identify how the lapilli systems can be improved to capture most of the P in drainage waters. Options are:

- modification of the mole plough developed in the Massey University study to ensure that the channel left by the shank and the mole plug are completely filled with lapilli. Reducing the opportunity for drainage water to by-pass the lapilli material should further enhance the ability of lapilli to remove P. In this respect a “Hoskins plough” could be used to backfill a 20 cm high by 10 cm wide column above the mole channel;
- evaluating systems that are focused on end of pipe treatment, such as lapilli-based filter beds. Further research and development to improve these P sorbing systems will be undertaken by Massey University staff as other funding opportunities arise.

Further work is also required to establish appropriate pre-treatment protocols to ensure that combustion ash or steel slag products are safe for use in the field. It would also seem prudent to target these materials for use in filter beds placed at points where convergent flow enters streams. This is likely to significantly reduce installation and re-generation/replacement costs.

The longevity and effectiveness of alum treatment of a wider range of pastoral soils remains to be determined. Although preliminary evidence suggests the technology may be appropriate to cropped or recently grazed land, its effectiveness on more weathered soils that have higher background amounts of Fe- and Al-oxides is unclear.

Table 8: Communication and science gaps for reactive filters and materials.

Attenuation tool	Guideline status and gaps	Science gaps e.g., uncertainty in performance Top 5 ranked, others listed in no particular order	Availability in NZ farm-scale modelling tools (OVERSEER® and NPLAS)
Denitrification wall	no guidelines	<ul style="list-style-type: none"> • simple cost-effective construction techniques • utility around tile drains • long-term performance > 8 years • hydraulic conductivity effects for different soil types and flow rates • P and faecal microbe treatment performance • greenhouse gas emissions • potential negative water quality effects -BOD release, dissolved organic colour release 	
Woodchip filters	<p>Existing: Basic concept and preliminary sizing guidelines given in DEC (2006)</p> <p>Gaps: Need for detailed practical guidelines.</p>	<ul style="list-style-type: none"> • field-scale operational performance • operational lifetime under different flow conditions • long-term changes in hydraulic characteristics affecting hydraulic permeability • comparative performance characteristics (efficacy, lifetime, long-term porosity) of different types of woodchips (e.g., willow, poplar, pine) • P and faecal microbe treatment performance • potential negative water quality effects -BOD release, dissolved organic colour release 	
P sock	Application and performance still being refined	<ul style="list-style-type: none"> • Optimising the placement and configuration of P-sock variants to intercept convergent flow before stream entry 	
Tile drain backfill (pea gravel, melter slag, volcanic lapilli, flyash)	Application and performance still being refined	<ul style="list-style-type: none"> • field-scale operational performance • operational lifetime under different flow conditions • performance of other common gravel types; e.g., limestone, scoria, pumice 	
Wetland soil amendments and filter materials	Application and performance still being refined	<ul style="list-style-type: none"> • further screening and laboratory evaluation of redox-insensitive materials suitable for flooded wetland soils • field-scale operational performance • operational lifetime under different flow conditions 	
Alum soil amendments	Application and performance still being refined	<ul style="list-style-type: none"> • field-scale operational performance • operational lifetime under different flow conditions 	
Melter slag - laneway runoff	Application and performance still being refined	<ul style="list-style-type: none"> • affect of clogging with sediment on performance 	

5. Generic scenarios and cost-effectiveness

Seven generic scenarios encompassing dairy, intensive sheep/beef and hill country sheep/beef were formulated (Table 9). The generic scenarios are loosely based on monitored research catchments for a variety of landscapes and farming types (e.g., Bog Burn, Toenepi, Whatawhata, etc.). Generic scenarios were developed to illustrate the applicability and compare the cost-effectiveness of the attenuation tools. The use of generic scenarios reduces the assumptions and constraints placed on the analysis. For example, for livestock exclusion calculations we assume that no livestock exclusion is in place on each farm.

Note: Both the efficacy estimates and cost calculations in this section rely on the assumptions in Tables 3, 5 and 7 being acceptable.

5.1 Scenario characteristics

Paddock and catchment-scale attenuation tools were evaluated for each scenario. In order to assess the applicability of each tool, the general hydrologic characteristics were characterised. At the catchment scale this requires assessing the relative importance of the baseflows (flows between events) and storm events on the catchment exports. In some catchments (e.g., scenarios 1, 2, 3) storm exports are dominant, while in others storms play a minor role (e.g., scenarios 4 and 6; Table 9). At the paddock scale, the relative importance of each hydrological pathway (see section 3.1) was identified. In addition modifications to pathways, for example by artificial drainage, were included. This enabled the scenario model to explore the potential of re-instating the natural hydrology to take advantage of features such as natural seepage wetlands.

Sediment, nitrogen and phosphorus loads (kg/ha) were estimated for each scenario at both the paddock and catchment scales. These estimates are based on research and model data. The catchment loads include all sources, for example the suspended sediment load includes sediment derived from bank erosion. It was assumed that sediment could only be transported successfully by overland flow or drainflow. Nutrients can be transported by any flowpath. The sediment and nutrient loads were apportioned to each water pathway on a pro-rata basis. For example, if drainflow transported 40% of the flow, then it also transported 40% of the nutrient load. Some tools only attenuate particulate P and so it was assumed that 50% of TP exported from dairy farms was particulate, while for drystock farms 80% of TP was particulate (based on Table 3 in Monaghan et al. 2007).

These paddock average exports will not reflect the paddock to paddock variation that occurs on farms. For example, on dairy farms targeting critical source areas or

activities such as effluent irrigation blocks, sacrifice paddocks and forage cropping areas with appropriate attenuation tools is likely to increase cost-effectiveness.

5.2 Assessing cost-effectiveness

5.2.1 Approach

In recognition of the landscape constraints placed on many attenuation tools, an assessment was made of the appropriateness of each tool for each of the seven scenarios. For example, for scenario 1 only one paddock and two catchment-scale attenuation tools were applicable as losses to groundwater dominate the paddock scale losses.

The load reduction estimates for each tool were calculated using the information and assumptions summarised in Table 3, Table 5 and Table 7. The majority of load reductions are percentage based, with an expected range. The upper and lower load reductions were calculated for the scenario \times tool \times pollutant matrix. The cost-effectiveness (\$/kg) was then estimated by dividing the annualised cost (\$/ha) by the annual load reduction (kg/ha).

Each attenuation option was costed to derive an annualised cost value. This represented 3 cost components:

- The opportunity cost of any capital work required (8%), such as fencing, excavation and material costs. If a cost range was provided the mid-point was used. For example, bottom of catchment wetlands cost between \$15 and \$30/ m² wetland and so \$23/ m² wetland was used.
- Annual maintenance costs for operations, spraying, repairs or replacement materials. In the case of natural seepage wetland options, an annual credit of 0.005 cows/ha or 0.01 sheep/beef units was accrued due to the assumed decrease in stock losses that would result from fencing these areas.
- The cost of lost productivity due to any removal of productive land from agricultural use. For the dairy farms that were modelled, it was assumed that farm inputs would remain constant (i.e., per ha inputs increased slightly). UDDER and financial modelling of this strategy indicated that this intensification of the remaining farm area virtually off-set the lost productivity from the land retired for wetlands and riparian grass filter strips.

Table 9: Generic farm scenarios and nutrient export characteristics.

Scenario	Farm							Exports exiting paddock						Exports exiting catchment					
	Land Use	channel density (m/ha)	stocking density (SU/ha)	No cattle	No. sheep	Farm size (ha)	topography	soil	artificial drainage	TN	TP	SS	flow pathways ²	Baseflow	stormflow	Loads			
										kgN/ha/y	kgP/ha/y	kgSS/ha/y		%	%	kg N/ha/y	kg P/ha/y	kg SS/ha/y	
1	Intensive dairy	Dairy	25	22	620	0	200	flat	well drained	no	40	0.1	8	2% infiltration excess OF, 98% groundwater	5	95	25	1	100
2	Typical dairy	Dairy	30	19	432	0	160	flat/easy	poorly drained, heavy subsoil	yes	30	0.7	80	10% infiltration excess OF, 20% subsurface, 20% groundwater, 50% drains	15	85	20	1	100
3	Typical dairy	Dairy	30	21	330	0	110	flat/easy	moderately well drained	yes	40	0.7	10	5% infiltration excess OF; 45% subsurface; 40% drain (includes 30% flow through wetlands, seeps, springs drained ³ , 10% seasonally saturated areas drained); 10% groundwater	15	85	20	1	100
4	Intensive sheep/beef	Sheep/Beef	22	13	473	1300	300	rolling	well drained	no	10	0.7	10	5% infiltration excess OF, 10% shallow subsurface flow, 85% groundwater	95	5	15	1	300
5	Intensive sheep/beef	Sheep/Beef	25	13	473	1300	300	rolling	heavy subsoil	yes	7	0.7	10	20% subsurface, 50% drained (made up of 20% flow through drained seeps, springs, wetlands, 30% seasonally saturated areas drained), 10% infiltration excess OF, 10% saturation OF, 10% groundwater	15	85	15	1	300
6	Sheep/beef hill country	Sheep/Beef	17	9	500	6300	1000	rolling-steep	well drained topsoil	no	3	0.5	15	10% saturation excess OF, 40% shallow subsurface flow (all in seeps, springs that feed wetlands, 5% catchment area), 50% groundwater	95	5	15	1.5	1500
7	Sheep/beef hill country	Sheep/Beef	22	9	500	6300	1000	rolling-steep (with lots of small channels)	poorly drained	no	3	0.5	30	5% infiltration excess OF, 5% saturation OF, 50% groundwater, 40% wetland seepage	15	85	15	1.5	1500

5.2.2 Cost comparisons

The cost-effectiveness of applicable tools for each model farm scenario are summarised and compared in Appendix 2. For each scenario the basic characteristics plus paddock and catchment scale exports are summarised. Pie charts of paddock exports via each flowpath are presented to illustrate dominant flowpaths for each pollutant. Catchment scale exports are required to estimate load reductions resulting from livestock exclusion (sediment) and bottom of catchment wetlands (all pollutants).

For each applicable tool the annualised cost per kg of sediment, nitrogen or phosphorus removal (\$/kg) is plotted on a log scale bar chart. Tools not applicable are omitted. If only a small fraction of a pollutant is able to be removed by an attenuation tool (either because little travels by that route and is able to be intercepted, or the removal efficacy is low) the \$/kg is high.

Each flowpath (or combined flowpaths) is shaded to assist with comparisons of “like with like”. The range of \$/kg is presented for the minimum and maximum estimated load reductions. The cost estimates used are the same for both maximum and minimum load reductions and so the \$/kg is lower for the maximum than for the minimum load reduction.

For the three dairy farms modelled results are only presented for the status quo (Appendix 2). Intensification to offset any lost productivity through the use of productive land for an attenuation tool increases decreases \$/kg. The major gain was for the attenuation tool with the largest land requirement - riparian grass filter strips – and \$/kg is decreased by about an order of magnitude (e.g., a reduction from 10 \$/kg to 1 \$/kg). The decreases in \$/kg for the remaining attenuation tools are smaller, generally less than 30 \$/kg for sediment and nitrogen and up to \$1000/kg for phosphorus.

Sediment is the most cost-effective pollutant to target, followed by nitrogen and then phosphorus. Sediment removal mostly costs less than 50 \$/kg, but costs vary widely by scenario and flowpath. Livestock exclusion is the attenuation tool with the lowest \$/kg for sediment, ranging from 0.01 to 0.33 \$/kg.

Nitrogen removal costs are in the order of 10 to 600 \$/kg. Livestock exclusion is often the most cost-effective, but for scenarios 2-5 several attenuation options are equally cost-effective (Appendix 2).

Costs of phosphorus removal are generally an order of magnitude higher than for nitrogen, mostly ranging between 100 and 10,000 \$/kg. Livestock exclusion from waterways is one of the most cost-effective options, except in hill-country where

channel densities and fencing costs are higher. Flyash and slag additions to tile drains are comparable in \$/kg for the scenarios with artificial drainage.

For all pollutants, bottom of catchment wetlands generally show good cost-effectiveness, which may be able to be improved further through use of lower-value or non-productive land, and/or cost-sharing with the wider community in view of their potential multiple-benefits (e.g., wildlife habitat, biodiversity and landscape values).

Two fencing options were examined for livestock exclusion hill country sheep/beef model farm scenarios (6 & 7). The cost-effectiveness of 5-wire (3 electric) (\$4.80/m) and 8-wire post and batten fences (\$12/m) were compared. For sediment the costs for each fencing option were all <0.1 \$/kgSS. Costs increased slightly for nitrogen, ranging from 19-324 \$/kgN (5-wire) to 46-591 \$/kgN (8-wire). Costs increased again for phosphorus ranging from 1000 and 1300 \$/kgP (5-wire) to 2500-3000 \$/kgP (8-wire).

A brief overview of two contrasting scenarios results are examined in more detail here to illustrate the results contained in Appendix 2. Scenarios 4 and 5 are both intensive sheep/beef farms with rolling topography. Scenario 4 has well drained soils, while the subsoil on model scenario farm 5 is heavy and artificially drained. Catchment exports are the same for both scenarios, but paddock flowpaths and exports vary (Figure 15).

For scenario 5 all of the attenuation tools are relevant, including the option of re-instating the natural hydrology by removing artificial drainage and maximising the natural attenuation options. This would take land out of production and this has been factored into the cost estimates, but re-instating wetlands could be a cost-effective option compared to installing an end of drain attenuation tool. Fifty percent of the flow leaves the paddock via drain flow, of which 2/5 is flow from drained seeps, springs, wetlands, and 3/5 is from seasonally saturated areas drained (Table 9). These flows have been re-instated for the alum&DF (seasonally saturated areas) and seepage wetland (drained seeps, springs and wetlands) attenuation tools. For scenario 4, a more limited range of attenuation tools is applicable as there is no drain flow (Figure 15).

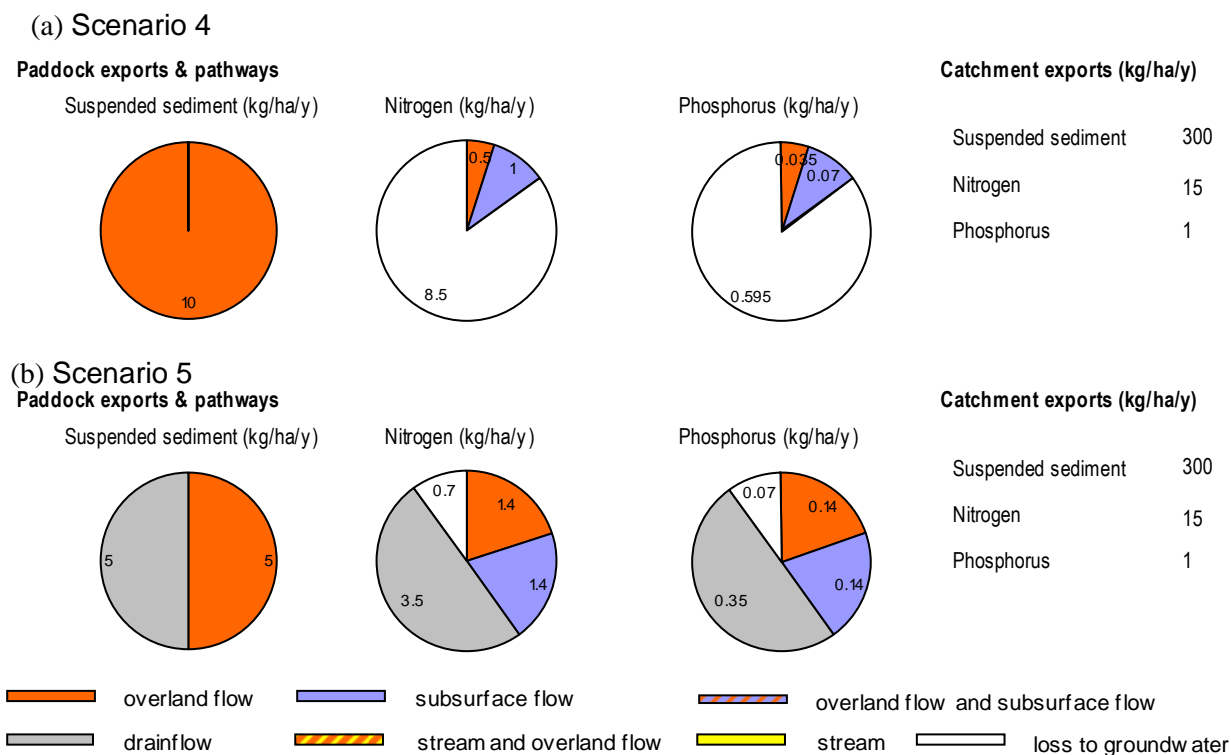


Figure 15: Paddock (via flowpaths) and catchment scale exports for model farm scenarios (a) 4 and (b) 5.

Figure 16 shows the range of cost-effectiveness for each nitrogen attenuation tool (see Appendix 2 for attenuation tool codes). Note that alum, flyash and slag are all phosphorus attenuation tools so are therefore not included in the nitrogen figures. For scenario 4 livestock exclusion alone and in combination with riparian grass filter strips, and bottom of catchment wetlands are the most cost-effective tools (Figure 16a). For scenario 5, livestock exclusion alone and in combination with riparian grass filter strips, and reinstatement of natural wetlands are the most cost-effective tools (Figure 16b).

The cost-effectiveness figures must be put in the context of the overall loads exported via each flowpath or the catchment total. For example, with scenario 4 most of the nitrogen leaves the paddock via losses to groundwater, so there may be more cost-effective source control tools available.

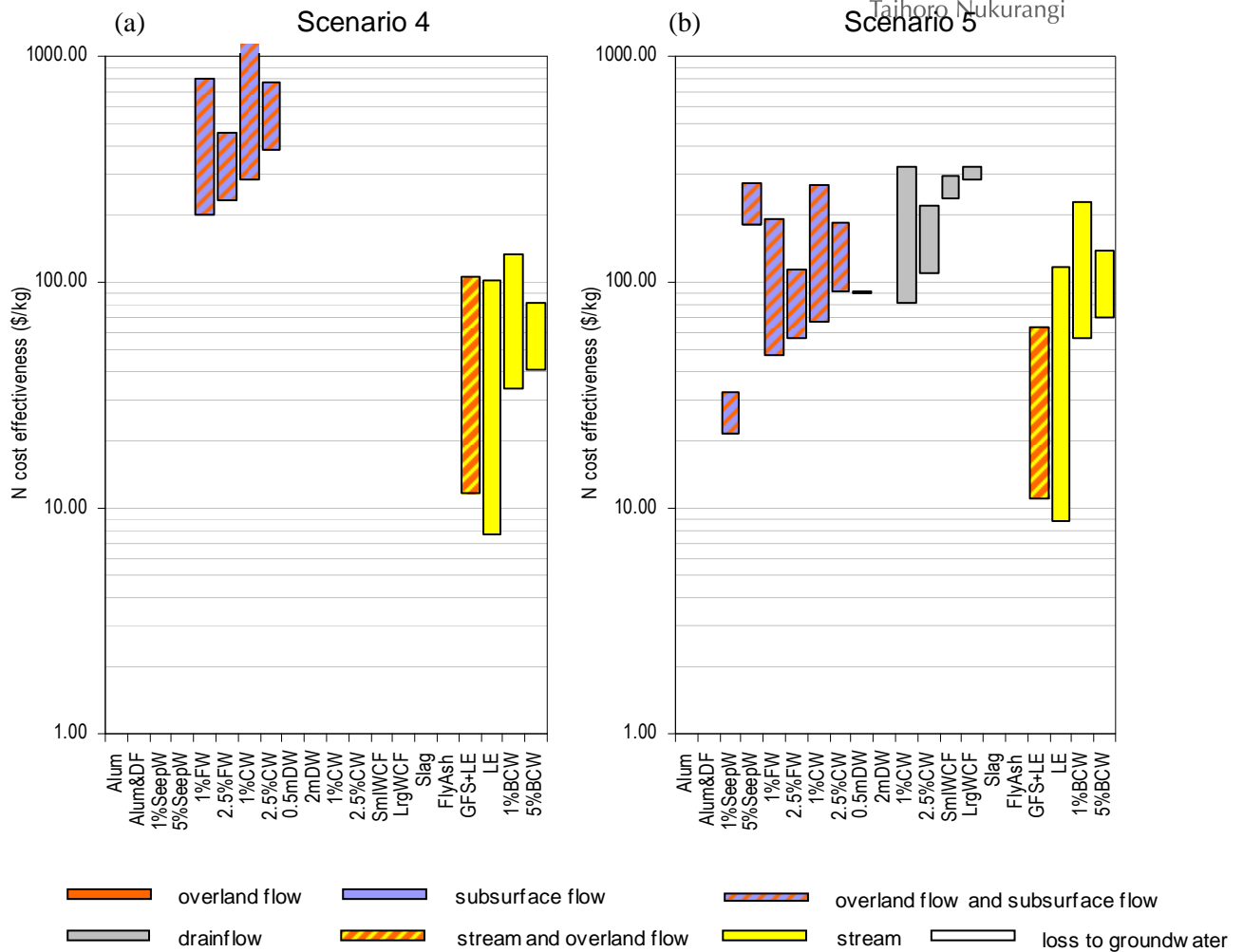


Figure 16: Nitrogen cost-effectiveness for model farm scenarios (a) 4 and (b) 5. (Note the cost-effectiveness range for 1% constructed wetlands in scenario 4 is 284 to 1136 \$/kg). See Appendix 2 for tool codes and \$/kg for other scenarios.

The range of scenarios illustrates the need to (i) identify and prioritise flowpaths and (ii) recognise natural attenuation assets in order to cost-effectively use attenuation tools to improve water quality.

5.3 Scoring systems

Simple scoring systems were devised to summarise the scenario results and assist with prioritising research needs. Two scoring systems were developed – one based on pollutant removal and the other on hydrology.

5.3.1 Hydrology scores

The hydrological scoring system was designed to identify important paddock scale flowpaths, reveal research gaps and help prioritise research needs. While attenuation

tools have been developed for all flowpaths, some flowpaths may have received more or less research attention than is hydrologically warranted on a national scale. Each flowpath is evaluated and scored for dominance, ease of interception and the proportion of the total paddock load of each pollutant carried. For example, a drain (the easiest flowpath to intercept) that carries >50% of the runoff and >50% of the pollutant load from a paddock would score the maximum of 15 (Table 10).

Table 10: Scoring system for paddock hydrology.

Index	Score	Description/classification
Paddock flowpath dominance	1	<5%
	2	5-10%
	3	10-20%
	4	20-50%
	5	>50%
Ease of interception	1	groundwater
	2	diffuse subsurface flow
	3	diffuse surface runoff
	4	natural wetland seepage
	5	drainflow (diffuse-point)
Proportion of total paddock load carried by flowpath	1	0
	2	1-10%
	3	10-25%
	4	25-50%
	5	>50%

The hydrological scoring system (Table 11) reveals that for suspended sediment both surface runoff and drainflow are important flowpaths to tackle with attenuation tools. For nutrients, drainflow, surface runoff and wetland subsurface flow (scenarios 6 and 7) score highly in several scenarios. For two scenarios (1 and 4) losses of nutrients to groundwater are significant. These losses can only be tackled using source controls and are not amenable to attenuation by any of the tools discussed in this report.

Table 11: Paddock hydrology scores (summed) for each model farm scenario and pollutant.

Model farm scenario	Flowpath	Flowpath dominance	Ease of interception	Proportion of total load			Score		
				SS	TN	TP	SS	TN	TP
1 - Intensive dairy, well drained	Overland flow	1	3	5	1	1	9	5	5
	Subsurface flow								
	Drainflow								
	Groundwater flow	5	1		4	4		10	10
2 - Dairy, heavy subsoil	Overland flow	3	3	4	2	1	10	8	7
	Subsurface flow	4	2		2	2		8	8
	Drainflow	5	5	4	4	4	14	14	14
	Groundwater flow	4	1		2	2		7	7
3 - Dairy, mod. well drained	Overland flow	2	3	4	1	1	9	6	6
	Subsurface flow	4	2		3	3		9	9
	Drainflow	4	5	4	3	3	13	12	12
	Groundwater flow	3	1		2	1		6	5
4 - Intensive sheep/beef, well drained	Overland flow	2	3	5	1	1	10	6	6
	Subsurface flow	3	2		2	1		7	6
	Drainflow								
	Groundwater flow	5	1		4	4		10	10
5 - Intensive sheep/beef, heavy subsoil	Overland flow	4	3	4	4	2	11	11	9
	Subsurface flow	4	2		2	2		8	8
	Drainflow	5	5	4	4	4	14	14	14
	Groundwater flow	3	1		2	1		6	5
6 - Hill country sheep/beef, well drained	Overland flow	3	3	5	2	2	11	8	8
	Subsurface flow	4	4		3	3		11	11
	Drainflow								
	Groundwater flow	5	1		4	4		10	10
7 - Hill country sheep/beef, poorly drained	Overland flow	3	3	5	2	2	11	8	8
	Subsurface flow	4	4		3	3		11	11
	Drainflow								
	Groundwater flow	5	1		4	4		10	10

5.3.2 Pollutant removal scores

The pollutant removal scoring system was designed to reveal the tools with the highest attenuation potential for each scenario. Three indices were included based on (i) ease of use, (ii) proportion of the total paddock or catchment load attenuated and (iii) cost-

effectiveness. The 1-5 scoring system is outlined in Table 12. Cost-effectiveness scores were developed separately for each of the pollutants as the cost scales vary. For example, most phosphorus attenuation costs at least 100 \$/kg, while suspended sediment attenuation costs are in the order of 1 to 10 \$/kg.

The pollutant attenuation scores range between 4 and 15 (Figure 17). Livestock exclusion and bottom of catchment wetlands score highly for every scenario. Seepage wetlands also score well for the applicable scenarios; for scenarios 3 and 5 seepage wetlands are reinstated, while for scenarios 6 and 7 they currently exist. Overall these three tools have scores >30 as they can attenuate all three pollutants to some degree. The tools with overall scores <10 are able to attenuate only one pollutant.

Table 12: Scoring system for pollutant removal.

Index	Pollutant	Score	Description/classification
Ease of use	All	1	difficult
		2	high level of specialist expertise required, e.g., flowpath location
		3	moderate level of expertise required, e.g., wetland design with suitable guidelines
		4	
		5	low level of specialist expertise required e.g., natural wetland identification readily useable by most farmers without need for additional specialist expertise.
Removal capability	All	1	0-5% of total load (paddock or catchment)
		2	5-10% of total load (paddock or catchment)
		3	10-20% of total load (paddock or catchment)
		4	20-40% of total load (paddock or catchment)
		5	>40% of total load (paddock or catchment)
Cost - effectiveness (\$/kg)	Sediment	5	<1
		4	1-10
		3	10-50
		2	50-100
		1	>100
	Nitrogen	5	<10
		4	10-50
		3	50-100
		2	100-500
		1	>500
	Phosphorus	5	<100
		4	100-1,000
		3	1,000-10,000
		2	10,000-100,000
		1	>100,000

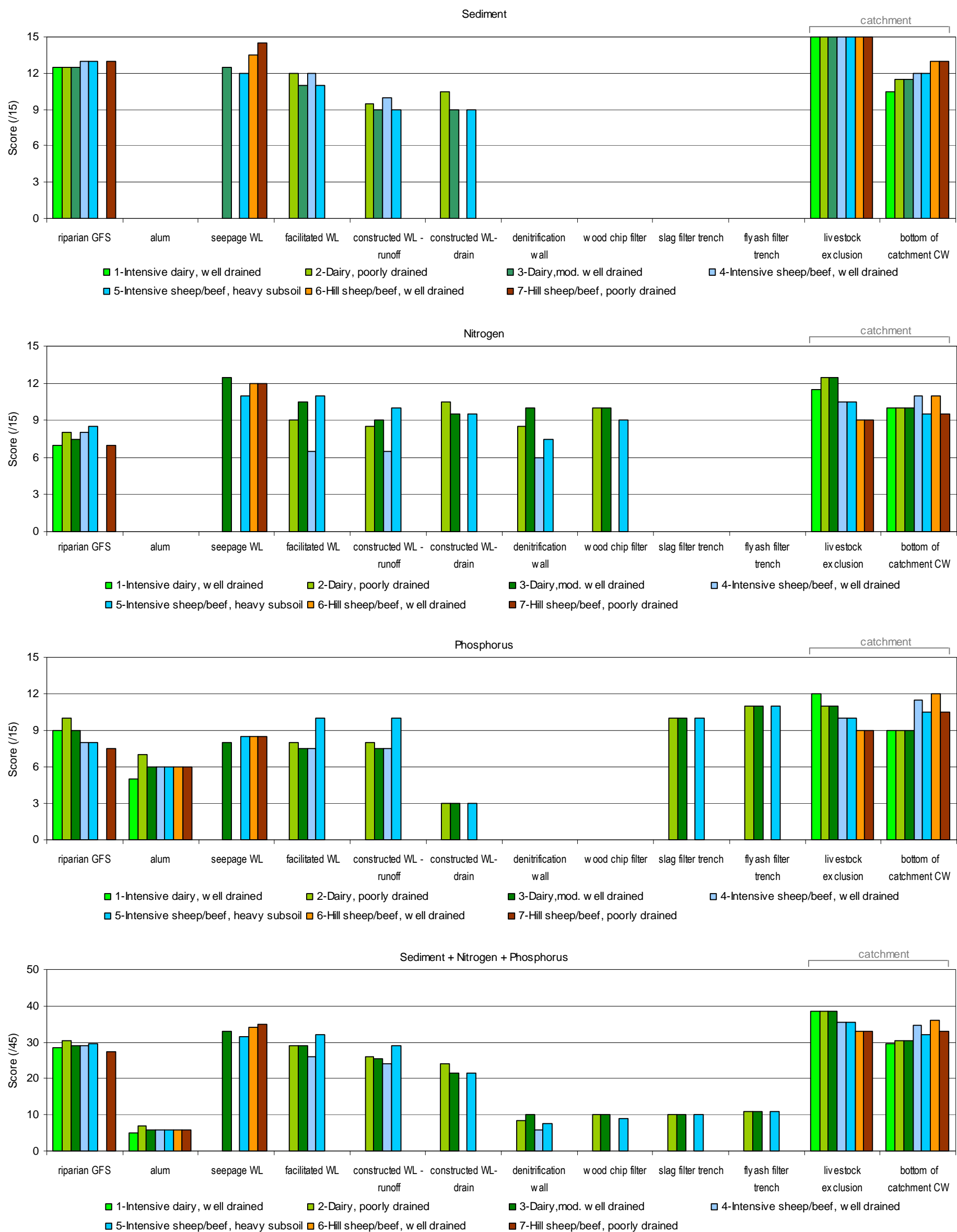


Figure 17: Pollutant removal scores for sediment, nitrogen, phosphorus and all three pollutants combined

6. Recommendations for research

Attenuation of sediment, nutrient and pathogens from flowing water is an integral component of sustainable farming. This review of existing attenuation tools has highlighted a number of research and communication gaps.

6.1 Research gaps

The simple scenario scoring systems have been used to prioritise research gaps and needs for tools that are widely applicable, effective and target the major flowpaths. This information is combined with the detailed knowledge gaps identified in Sections 4.6, 4.17 and 4.25 to develop research recommendations.

Tools included in the scenarios investigated in this report are generally those that have been more widely tested and thus can be reasonably evaluated. Table 2 also includes tools that have gained less traction, have had limited testing in New Zealand or are the subject of exploratory new trials. These tools or variations on them (including: controlled subsurface drainage, vegetated-sections of surface drains, aquatic plant harvesting) require further investigation and consideration before their potential can be properly assessed. A watching brief is also needed to identify novel attenuation options and approaches that could be applied in New Zealand pastoral farming systems.

- **Develop attenuation tools suitable for drainflow and subsurface flow that target multiple pollutants.**

The major flowpaths requiring attenuation are drain flow, and seepage and diffuse subsurface flows. Traditionally these “less visible”, low concentration but high volume flowpaths, have been considered to be insignificant transporters of pollutants (compared to high concentration, low volume surface runoff). However, recent research has highlighted their importance.

Attenuation tools for these flowpaths are typically pollutant specific (e.g., wetlands receiving drain flow or wood chip filters for N or reactive filters for P) rather than multi-pollutant. For example, a drain flow attenuation tool that combines SS, TN and TP attenuation rather than attenuating only one pollutant could have widespread application. Research has often been focused on one landscape where the key pollutant has been identified, but in other landscapes there may be multiple pollutants that need addressing. Cost-effectiveness improves by targeting multiple pollutants. Source controls (such as nitrification inhibitors, improved effluent management and less easily leached fertiliser types etc.) will assist with reducing the concentrations of pollutants in drain flow and subsurface flow, particularly where there is high connectivity between the pollutant source and flowpath. However, residence times for

subsurface flows in some catchments may introduce time lags and attenuation tools will be required.

Specific opportunities include:

- Enhancing P attenuation in constructed wetlands e.g., outlet P filters.
- End of drain filters encompassing sediment, nitrogen and phosphorus attenuation tools (for existing drains).
- Further research on the inclusion of organic carbon and reactive materials (e.g., lapilli) in new mole/tile drains.

In addition to these opportunities, there is a need to evaluate basic performance attributes, practicality and cost of promising less-researched and novel attenuation options which target these priority flowpaths. There may be new or emerging tools that have yet to be evaluated for New Zealand conditions.

- **Field test bottom of catchment wetlands, including ancillary community and environmental benefits.**

Bottom of catchment wetlands have potential in both baseflow and storm flow dominated systems (depending on outflow structure design). They become a cost-effective attenuation tool when marginal or community land is available, and where wider community and environmental benefits are taken into account.

Environment BOP has commenced monitoring on the recently established large-scale Lake Okaro wetland treatment system (2.4 ha wetland treating the main stream inflows to the lake), which will provide one year of data on the efficacy of a large bottom of catchment wetland designed for nutrient attenuation. Gaps exist in the current monitoring programme with respect to sediment and pathogen attenuation, and a multi-year (≥ 3) monitoring is required to gauge year to year variability in performance. Wider testing of this attenuation tool in different landscapes, particularly those with more intensive land use would be valuable.

- **Quantify nutrient and pathogen reductions as a result of livestock exclusion and other alternative strategies from hill-country perennial streams.**

Little data exists on nutrient and pathogen reductions due to direct livestock deposition and current research projects in New Zealand cannot fill this gap due to the concurrent implementation of multiple BMPs in research catchments. Livestock exclusion is become a high profile issue for the sheep & beef industries and may be problematic on hill-country. One issue is the provision of off-stream water and research on simple alternatives to troughs is needed. In addition, in some landscapes total exclusion may

be impractical and research on alternatives, such as partial exclusion or changing animal behaviour (e.g., troughs, supplements or shade) is needed.

- **Investigate the benefits of livestock exclusion on intermittent streams, wetlands and seasonally saturated areas.**

Targeted livestock exclusion could be beneficial beyond permanent stream margins and on streams that are smaller than those included in the Clean Streams Accord (e.g., seasonally saturated source areas and ephemeral stream headwaters). Seasonal increases in flow and channel network expansion may increase the probability of livestock access to surface water and hydraulic connectivity to pollutant reservoirs.

- **Field test seepage wetlands attenuation performance, particularly for SS and P, and evaluate their potential to be reinstated where drained.**

Much of the research effort on natural seepage wetlands has been on short term nitrate removal and denitrification rather than total N removal performance over the longer term. Research is needed to measure the net sediment, N and P exports from a range of seepage wetlands under baseflow and event conditions.

- **Field-test TN, TP, SS and faecal microbe attenuation from surface drainage by facilitated and constructed wetlands.**

There have been no New Zealand studies of wetland pollutant removal from surface drains. Performance estimates have been derived mainly by reference to overseas studies, where farming systems are frequently quite different (e.g., seasonally housed livestock and cropping). Wetlands treating surface drainage flows are likely to very effectively remove suspended sediments and associated particulate nutrients during flow events, but there is considerable uncertainty about re-release of retained nutrients. This information is necessary to quantify the long-term performance of these systems and develop appropriate designs.

6.2 Information needs and guidelines

We have also identified communication gaps, overall and for specific attenuation tools. The main priorities are:

- **Develop simple tools, supported with training courses, to assist with the selection of suitable attenuation tools for different landscape and soil types, and farming systems**

None of the existing guidelines provide tools to help farmers/Land management officers/farm advisors identify priority pollutants, key hydrological flowpaths and attenuation tools suitable for their particular combination of receiving waters,

landscape and farming operation. Realistic local water quality targets should be developed for rural catchments to guide mitigation actions. For example, a decision tree approach could be formulated to guide attenuation tool selection.

- **Integrate information on a wider range of pollutant attenuation options into farm-scale nutrient-budgeting tools such as Overseer®.**

The recent development of a hydrology model for OVERSEER® provides opportunities for attenuation tools to be included in the model. The first attenuation tools to be included are riparian grass filter strips (without livestock exclusion effects), natural seepage wetlands and constructed wetlands. Further attenuation options should be added, where possible, to increase the range of options able to be considered by farmers.

- **Develop practical guidelines to support appropriate protection, rehabilitation and management of natural attenuation features on farms (e.g., wetlands).**

Natural landscape features that perform important attenuation roles, such as existing wetlands, have not been adequately recognised and valued. A possible cause for this could be a lack of appreciation of how widespread wetlands actually are in the landscape. Many farmers cannot identify seepage wetlands (Ben Banks, EBoP pers. comm.; Simon Stokes, HBRC pers. comm.) and are unaware of their potential to attenuate nutrients, particularly nitrogen. Wetlands (seeps) were included in the DEC (2006) guidelines in passing, but practical management guidelines are required.

- **Develop practical guidelines to support proper design, implementation and on-going management of other widely applicable attenuation tools (e.g., sediment traps, constructed wetlands).**

The cost-effectiveness of these tools such as constructed wetlands depends on proper design, construction, planting and maintenance. Although guidelines exist for some widely-applicable attenuation tools, monitoring, review and further development is required to ensure that these tools are used as effectively as possible. Given the financial risks to farmers and the environment from improper implementation, thought needs to go into what is an appropriate level of testing and development for establishment and industry endorsement of new BMPs, and what are the guideline and training requirements to support their wise use.

7. Appendix 1

Table 13: New Zealand studies on water quality impacts of cattle and deer on streams, wetlands and ephemeral channels.

Reference	Waterbody	Livestock/Location	study design	Stock units	Results [†]	Relevance to Canterbury water bodies
(Environment Southland 2000)	stream	deer, Southland	one-off upstream + downstream		downstream concentrations increased 19-35 x SS, NH ₄ -N and faecal coliforms	all
(Southland RC V New Zealand Deer Farms Ltd 2004)	stream	deer, Moss Burn, Southland	one-off upstream + downstream & tributary	paddock: 13-36 SU/ha (100-200 deer, 10-15 ha)	25 x increase in SS and turbidity from upstream of 6.8 mg/L & 6.9 NTU (tributary 2000 mg/L) 5 x increase in <i>E. coli</i> from 1100 MPN/100 mL (tributary 6 x 10 ⁴ MPN/100mL)	all
(Stassar & Kemperman 1997)	stream	cattle, Whatawhata, Waikato	upstream + downstream	farm average: 10 SU/ha paddock (15 mixed age cattle, 1.06 ha) = 70 SU/ha	Rise in turbidity from background of 10 NTU to 70-80 NTU (approximately 65 mg/L suspended solids), but up to 250 NTU.	all
(Davies-Colley & Nagels 2002)	stream	deer, Piakonui, Waikato	upstream + downstream, 13 samples		2-3 x increases in <i>E. coli</i> from upstream site to downstream site.	all
(Davies-Colley et al. 2004)	stream	246 dairy herd, Tasman	one-off upstream + downstream monitoring of herd crossing	246 cows.	plumes of turbid water. Sharp spike of <i>E. coli</i> 5 x10 ⁴ MPN/100 mL. Two crossings yield 35.2 kg SS, 4.5 billion <i>E. coli</i> , 1.4 kg TN.	all
(Smith et al. cited in Davies-Colley et al. 2004)	stream	145 cow herd, Puremahia Ck, Golden Bay	one-off upstream + downstream monitoring of herd crossing		<i>E. coli</i> peaked at 8 x10 ⁴ MPN/100 mL, yield estimated as >11 billion <i>E. coli</i> & 10 kg SS	all

Reference	Waterbody	Livestock/Location	study design	Stock units	Results [†]	Relevance to Canterbury water bodies
(Nagels unpublished data) (see Collins et al. 2007)	stream	dairy, Waikato	upstream and downstream monitoring of paddocks when stock grazing	paddock: 150-200 cows in paddocks 1.2-3.1 ha = 450-900 SU/ha	<i>E. coli</i> concentrations increased except where cows could not easily access the channel <i>E. coli</i> concentrations highest at Hopkins (median 4600-5200 MPN/100 mL, about 30 X background) – small stream + easy access	all
(McDowell 2006)	stream	dairy, south Otago	monitoring upstream and downstream for 2 years, monthly sampling	10 heifers (2002) 20 cows (2003)	TP loads increased. Upstream TP load increased after ephemeral channel access (2002). Downstream TP load increased after forced stream grazing (2003).	all
(McDowell 2007)	stream	deer, Telford, Otago, 14-65 day rotation	monitoring outlet small catchments (6.1-32.1 ha), monthly + flow sampling	farm average: 1000 hd/155 ha ≈ 13 SU/ha	Loads of <i>E. coli</i> , SS, FRP, PP, TP, NH ₄ -N and NO ₃ -N higher when deer in wallows	all
	stream	deer, Invermay, Otago, 21-56 day rotation	monitoring outlet small catchments (4.1 ha), monthly + flow sampling	farm average: 1200 hd/160 ha ≈ 15 SU/ha	Loads of <i>E. coli</i> , SS, FRP, PP, TP, NH ₄ -N and NO ₃ -N higher when deer in wallows	all
(Buck et al. 2004)	streams	Otago, rolling- hill country Lee: sheep/beef Tuakitoto: sheep/beef/dairy Barbours: ungrazed tussock	multiple sites within 3 catchments, one-off summer baseflow sampling at 60 sites	Lee: farm average 7 (4.7-10.6 SU) Tuakitoto: farm average 11.3 (6.8-24.5) Barbours: 0	Lee: catchment stocking rates positively correlated to conductivity, turbidity, NH ₄ -N, TP and TN. Tuakitoto: catchment stocking rates correlated to TP	all
(Collins 2004)	wetland	beef, hill country, Waikato	two wetlands within 2 small catchments, storm and baseflow sampling	farm average: 12 SU/ha, wetland A: 95 SU/ha	Concentrations of <i>E. coli</i> highest during storm events shortly after livestock have been in wetland	all

[†]SS = suspended sediment, TN = total nitrogen, NH₄-N = ammonium, NO₃-N = nitrate; TP = total phosphorus, FRP = filterable (or dissolved) reactive phosphorus, PP = particulate.

8. Appendix 2: Scenario results

8.1 Scenario codes

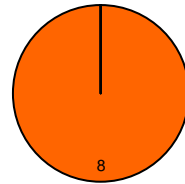
Code	Definition
Alum	Alum
Alum&DF	Alum with subsurface drains removed - increasing SEOF
1%SeepW	seepage wetland 1% of catchment area estimated for existing and re-instated wetlands
5%SeepW	seepage wetland 5% of catchment area estimated for existing and re-instated wetlands
1%FW	facilitated wetland 1% of catchment area
2.5%FW	facilitated wetland 2.5% of catchment area
1%CW	constructed wetland 1% of catchment area, receiving surface runoff and subsurface flow
2.5%CW	constructed wetland 2.5% of catchment area, receiving surface runoff and subsurface flow
0.5mDW	0.5 m deep denitrification wall
2mDW	2 m deep denitrification wall
1%CW	constructed wetland 1% of catchment area, receiving drain flow
2.5%CW	constructed wetland 2.5% of catchment area, receiving drain flow
SmIWCF	small woodchip filter
LrgWCF	large woodchip filter
Slag	trench backfill: slag
FlyAsh	trench backfill: fly ash
GFS+LE	riparian grass filter strip & livestock exclusion
LE	livestock exclusion
1%BCW	bottom of catchment wetland, 1% catchment area
5%BCW	bottom of catchment wetland, 5% catchment area

8.2 Scenario 1: Intensive dairy, flat topography, well drained soil.

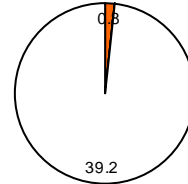
Stocking density (SU/ha)	22
No. cattle	620
Farm size (ha)	200
Topography	flat
Soil	well drained
Channel density (m/ha)	25
Channel length (m)	5000
Artificial drainage	no

Paddock exports & pathways

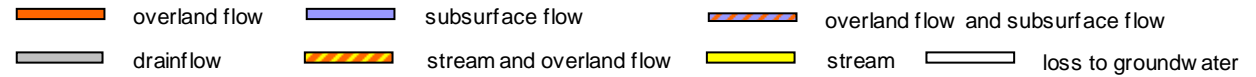
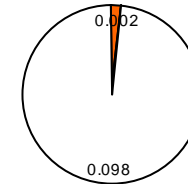
Suspended sediment (kg/ha/y)



Nitrogen (kg/ha/y)

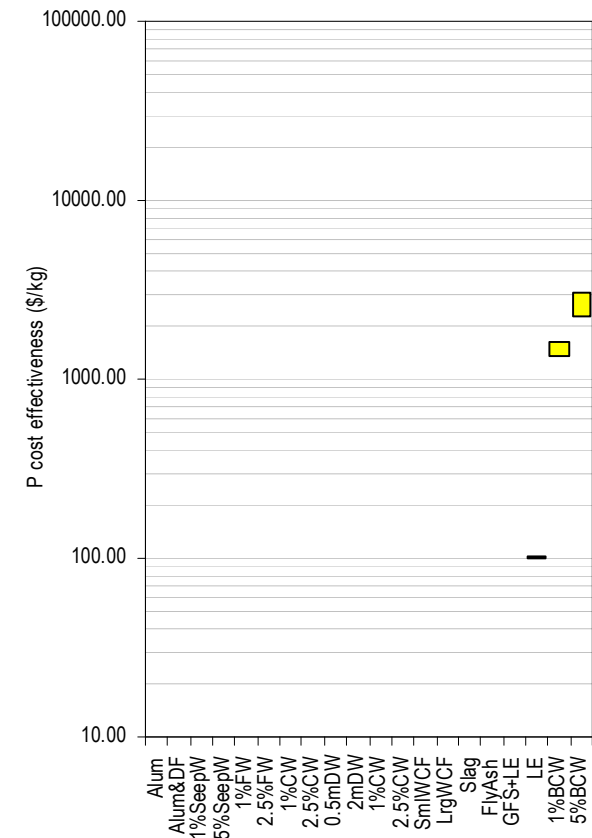
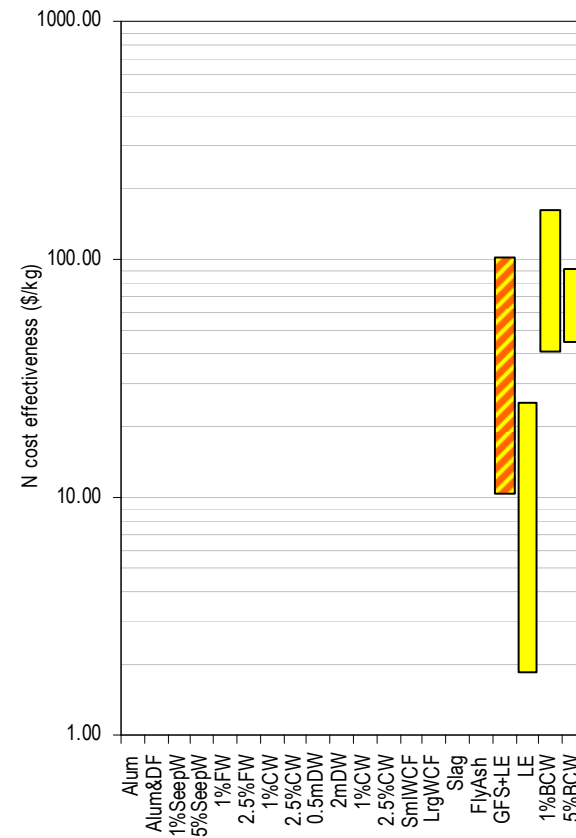
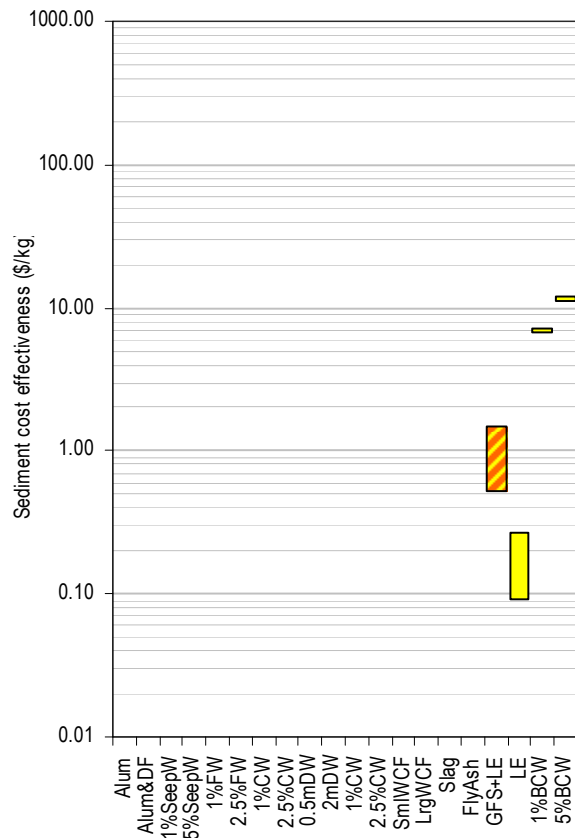


Phosphorus (kg/ha/y)



Catchment exports (kg/ha/y)

Suspended sediment	100
Nitrogen	25
Phosphorus	1

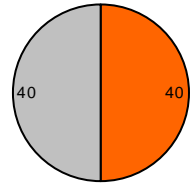


8.3 Scenario 2: Dairy, flat/easy topography, poorly drained, heavy subsoil

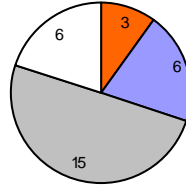
Stocking density (SU/ha) 19
 No. cattle 432
 Farm size (ha) 160
 Topography flat/easy
 Soil poorly drained, heavy subsoil
 Channel density (m/ha) 30
 Channel length (m) 4800
 Artificial drainage yes

Paddock exports & pathways

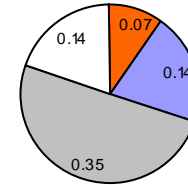
Suspended sediment (kg/ha/y)



Nitrogen (kg/ha/y)



Phosphorus (kg/ha/y)



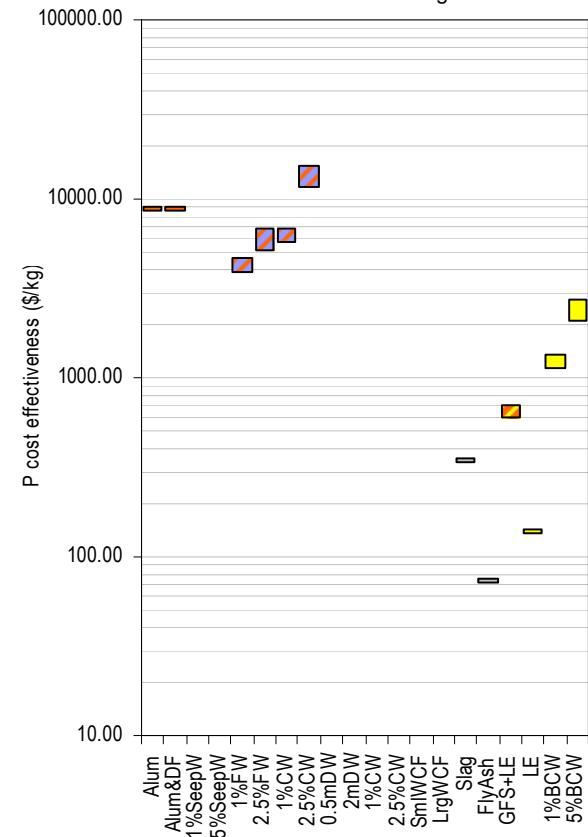
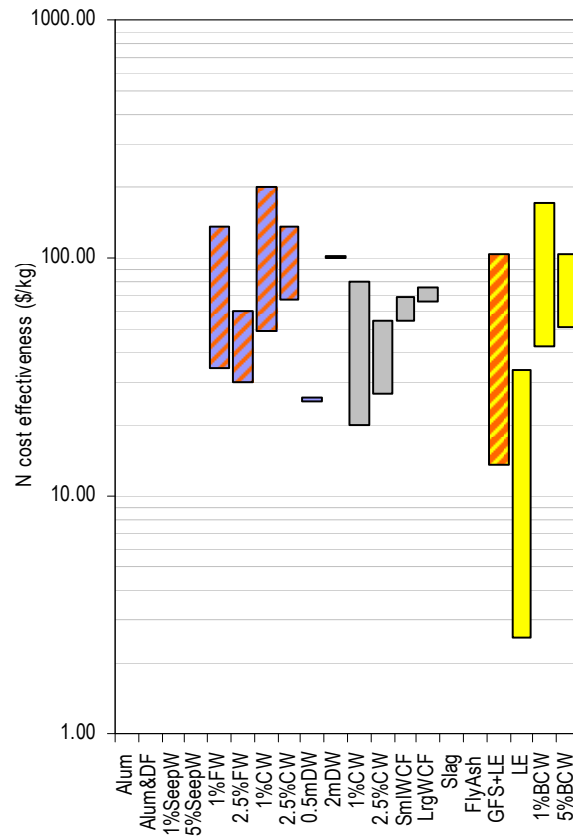
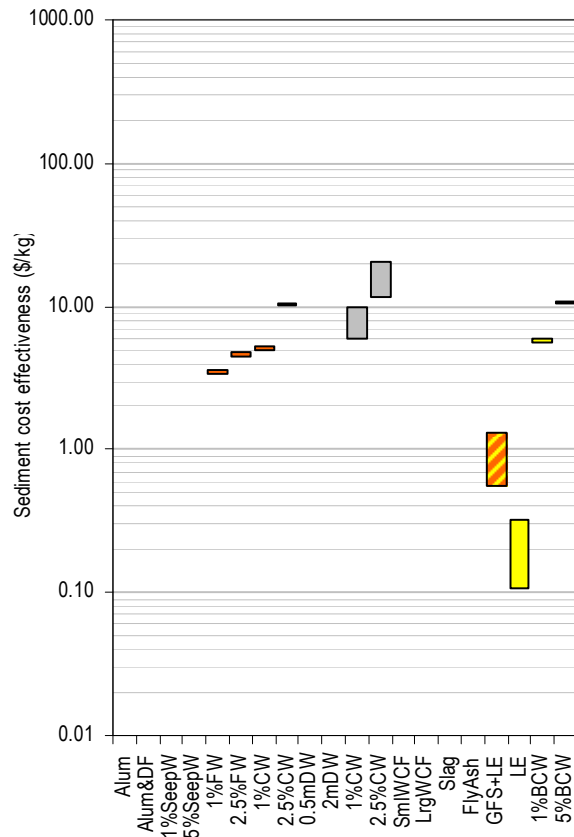
Catchment exports (kg/ha/y)

Suspended sediment 100
 Nitrogen 20
 Phosphorus 1

overland flow
 drainflow

subsurface flow
 stream and overland flow

overland flow and subsurface flow
 stream
 loss to groundwater

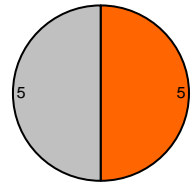


8.4 Scenario 3: Dairy, flat/easy topography, moderately well drained soil

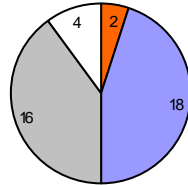
Stocking density (SU/ha)	21
No. cattle	330
Farm size (ha)	110
Topography	flat/easy
Soil	moderately well drained
Channel density (m/ha)	30
Channel length (m)	3300
Artificial drainage	yes

Paddock exports & pathways

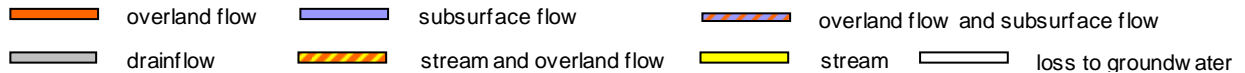
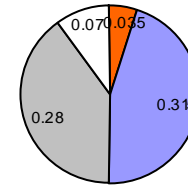
Suspended sediment (kg/ha/y)



Nitrogen (kg/ha/y)

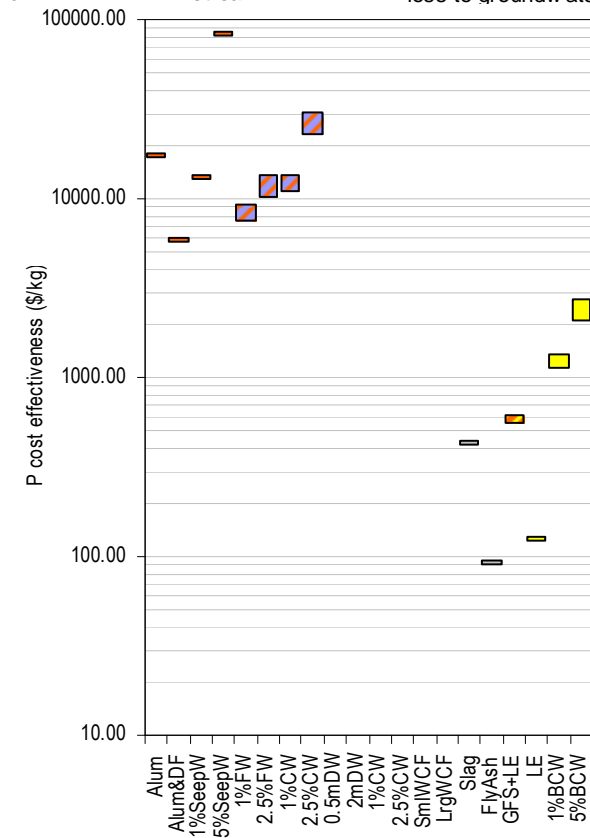
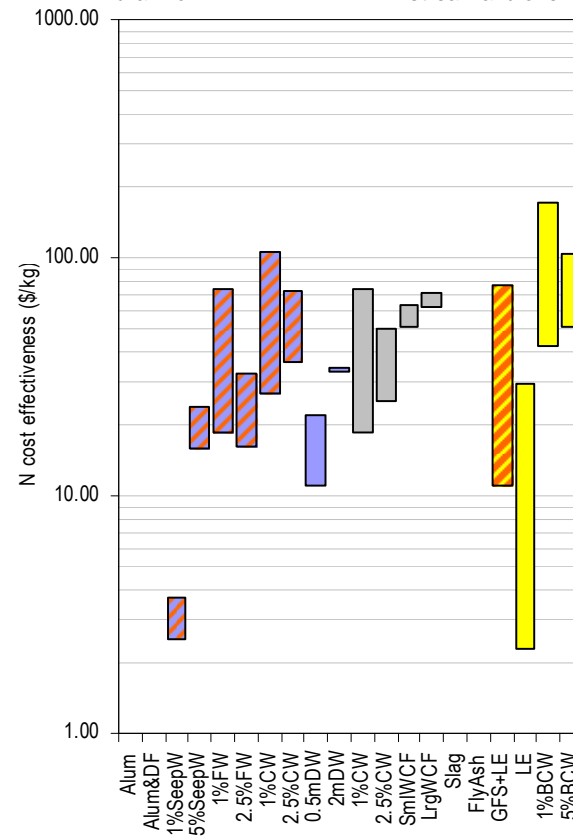
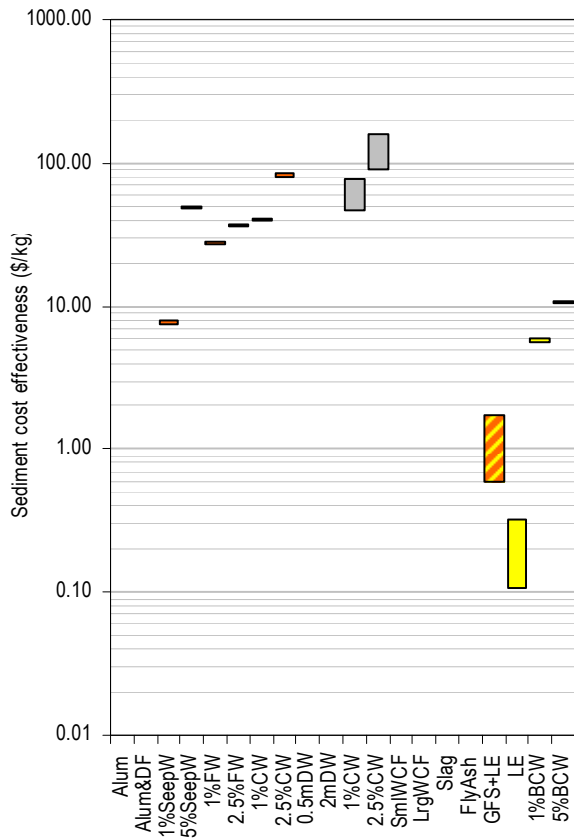


Phosphorus (kg/ha/y)



Catchment exports (kg/ha/y)

Suspended sediment	100
Nitrogen	20
Phosphorus	1

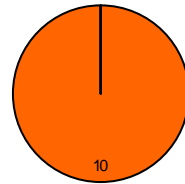


8.5 Scenario 4: Intensive sheep/beef, rolling topography, well drained soils

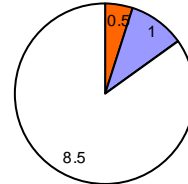
Stocking density (SU/ha)	13
No. cattle	473
Farm size (ha)	300
Topography	rolling
Soil	well drained
Channel density (m/ha)	22
Channel length (m)	6600
Artificial drainage	no

Paddock exports & pathways

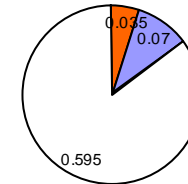
Suspended sediment (kg/ha/y)



Nitrogen (kg/ha/y)

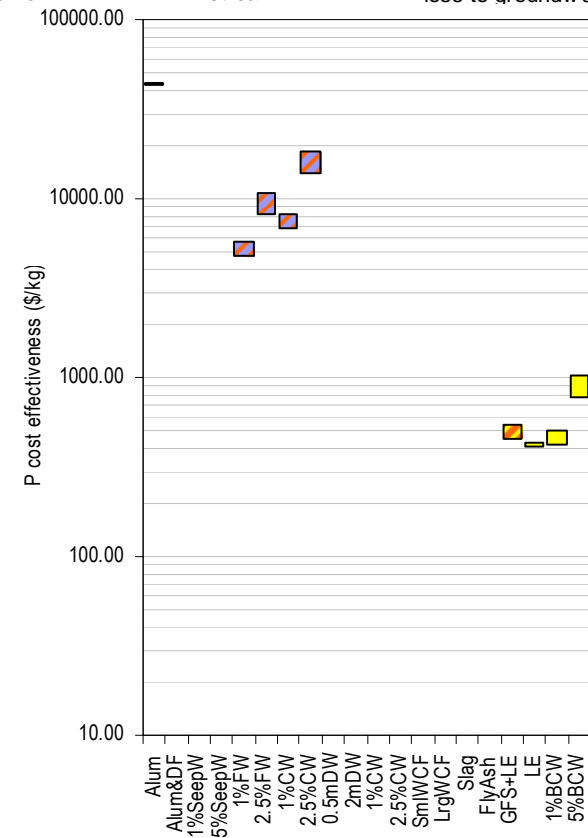
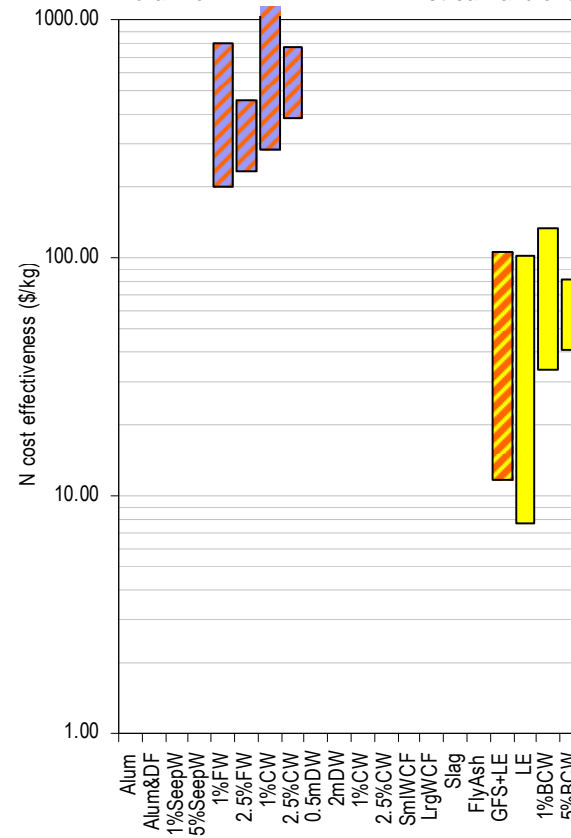
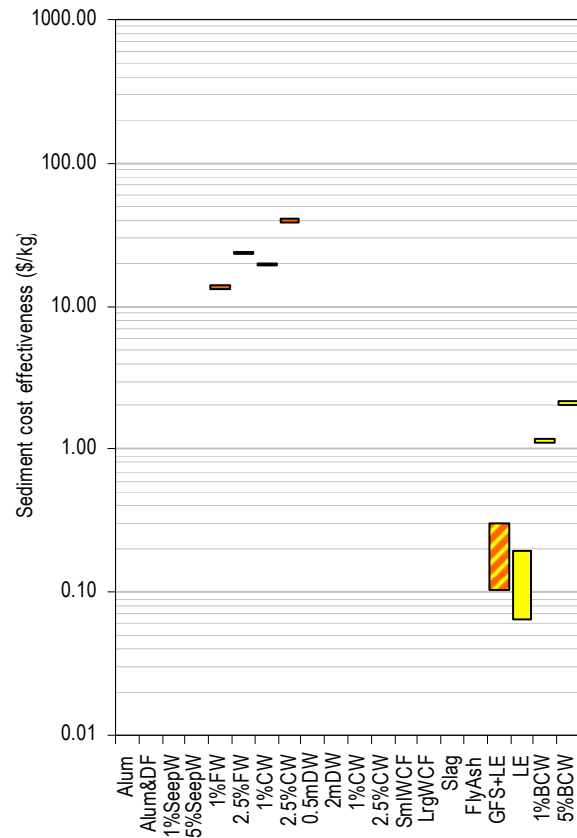
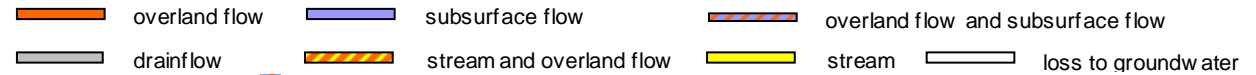


Phosphorus (kg/ha/y)



Catchment exports (kg/ha/y)

Suspended sediment	300
Nitrogen	15
Phosphorus	1

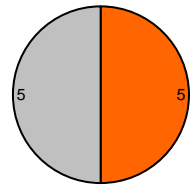


8.6 Scenario 5: Intensive sheep/beef, rolling topography, heavy subsoil

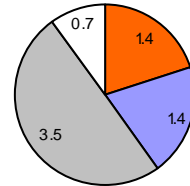
Stocking density (SU/ha)	13
No. cattle	473
Farm size (ha)	300
Topography	rolling
Soil	heavy subsoil
Channel density (m/ha)	25
Channel length (m)	7500
Artificial drainage	yes

Paddock exports & pathways

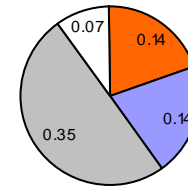
Suspended sediment (kg/ha/y)



Nitrogen (kg/ha/y)

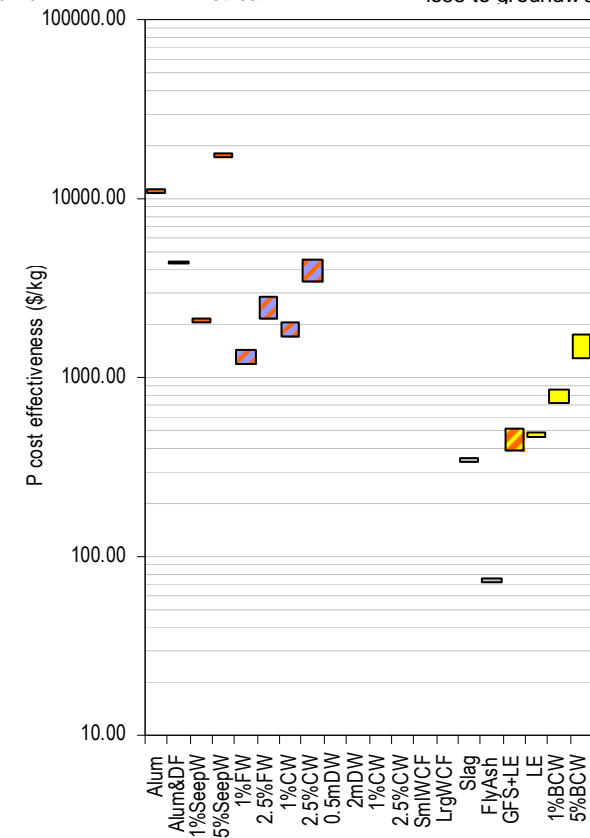
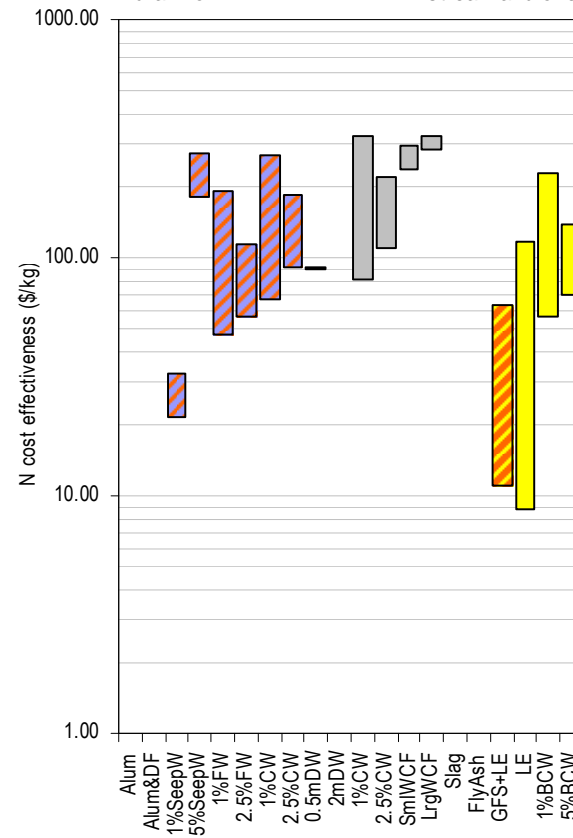
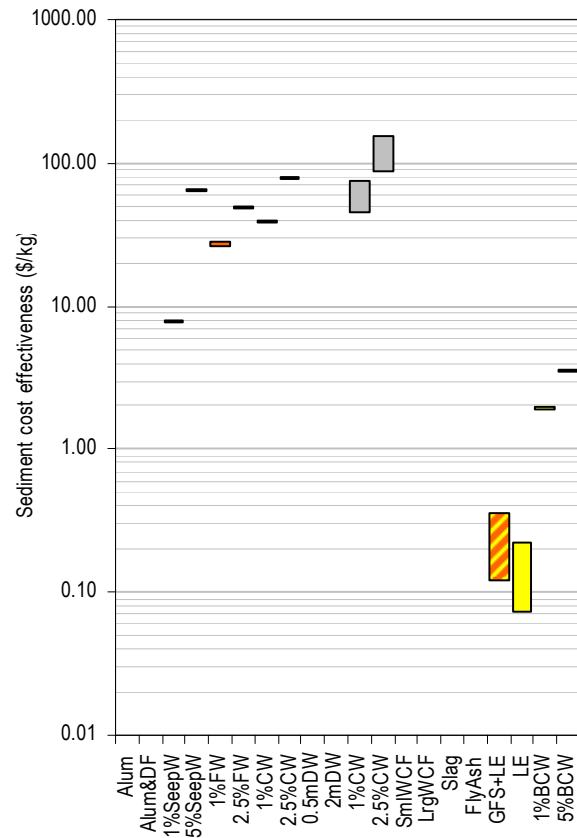
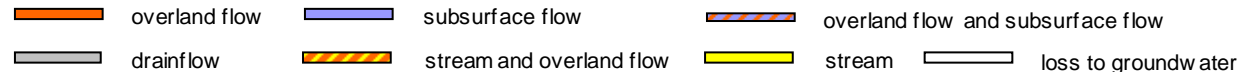


Phosphorus (kg/ha/y)



Catchment exports (kg/ha/y)

Suspended sediment	300
Nitrogen	15
Phosphorus	1

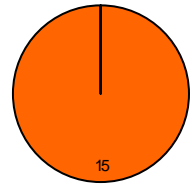


8.7 Scenario 6: Hill country sheep/beef, rolling-steep topography, well drained topsoil

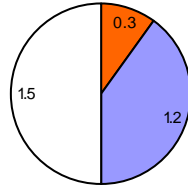
Stocking density (SU/ha)	9
No. cattle	500
Farm size (ha)	1000
Topography	rolling-steep
Soil	well drained topsoil
Channel density (m/ha)	17
Channel length (m)	17000
Artificial drainage	no

Paddock exports & pathways

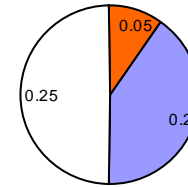
Suspended sediment (kg/ha/y)



Nitrogen (kg/ha/y)

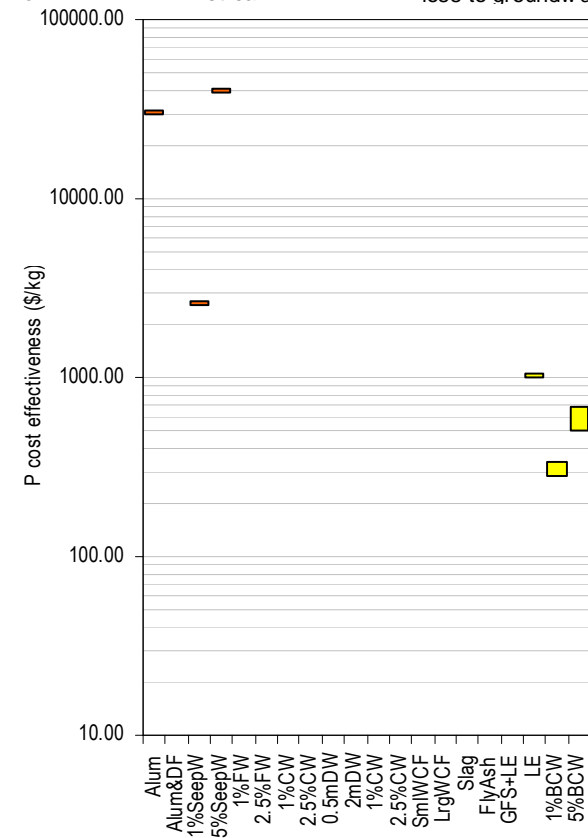
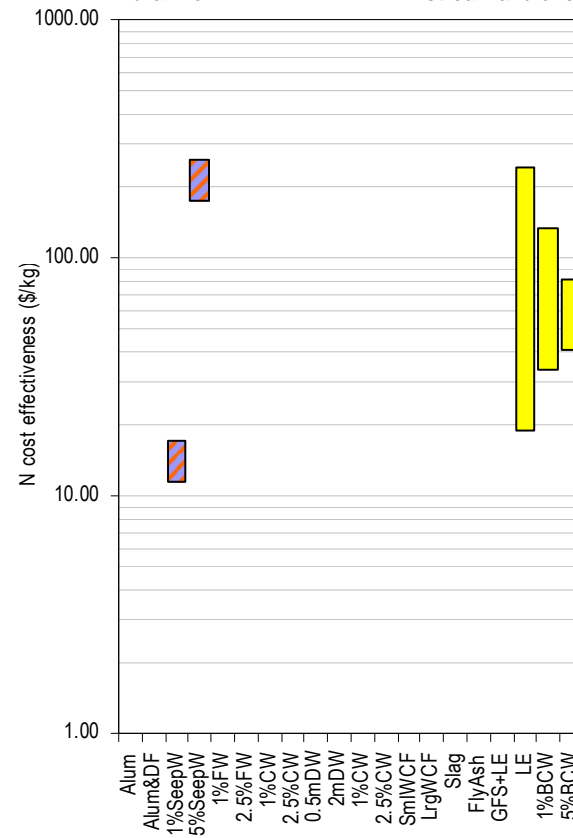
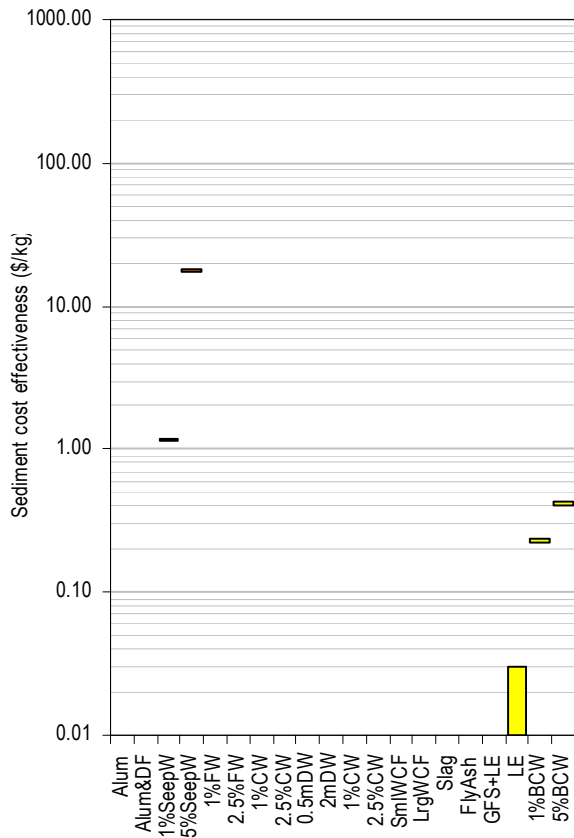
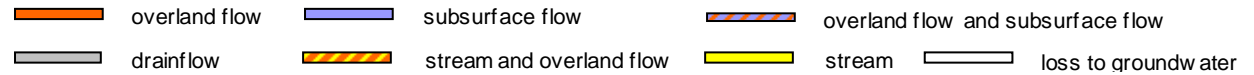


Phosphorus (kg/ha/y)



Catchment exports (kg/ha/y)

Suspended sediment	1500
Nitrogen	15
Phosphorus	1.5

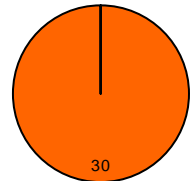


8.8 Scenario 7: Hill country sheep/beef, rolling-steep topography, poorly drained soil

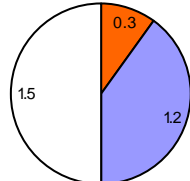
Stocking density (SU/ha)	9
No. cattle	500
Farm size (ha)	1000
Topography	rolling-steep (with lots of small channels)
Soil	poorly drained
Channel density (m/ha)	22
Channel length (m)	22000
Artificial drainage	no

Paddock exports & pathways

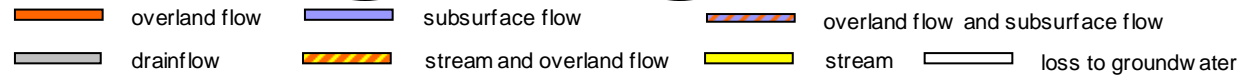
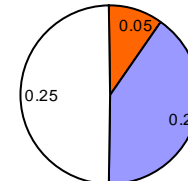
Suspended sediment (kg/ha/y)



Nitrogen (kg/ha/y)

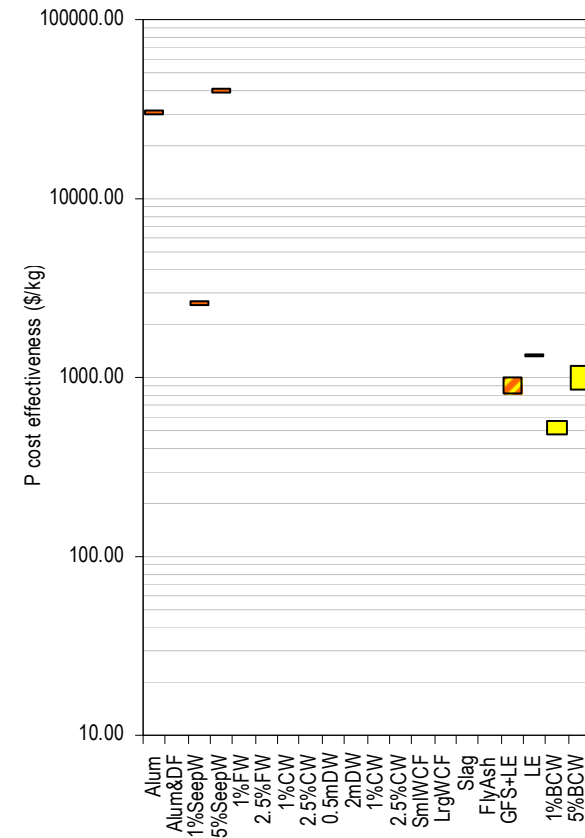
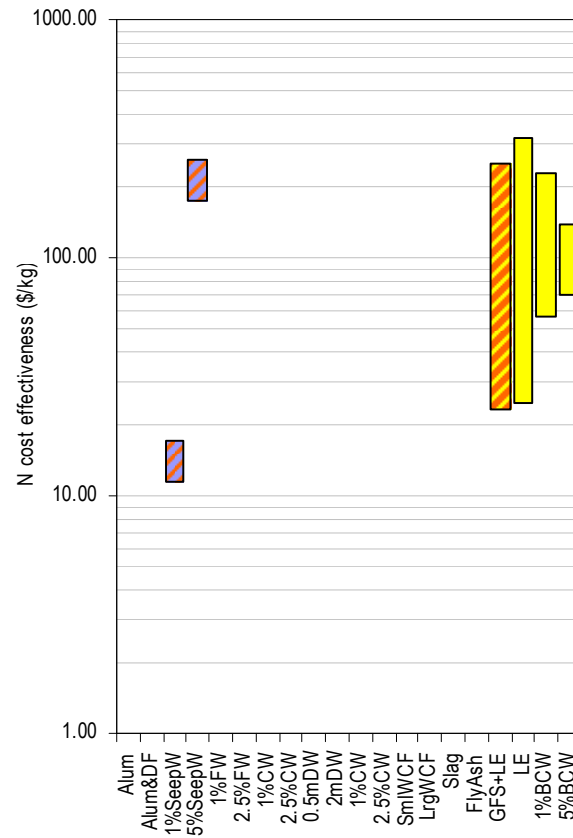
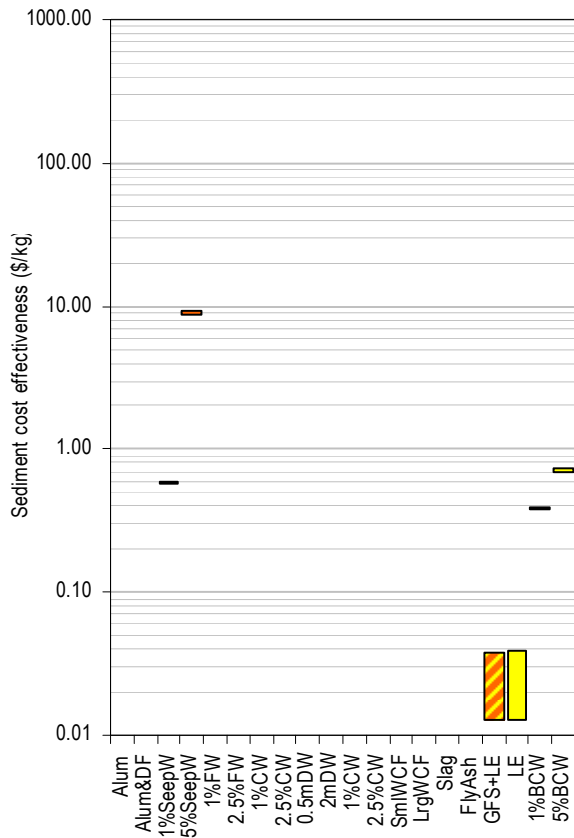


Phosphorus (kg/ha/y)



Catchment exports (kg/ha/y)

Suspended sediment	1500
Nitrogen	15
Phosphorus	1.5



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